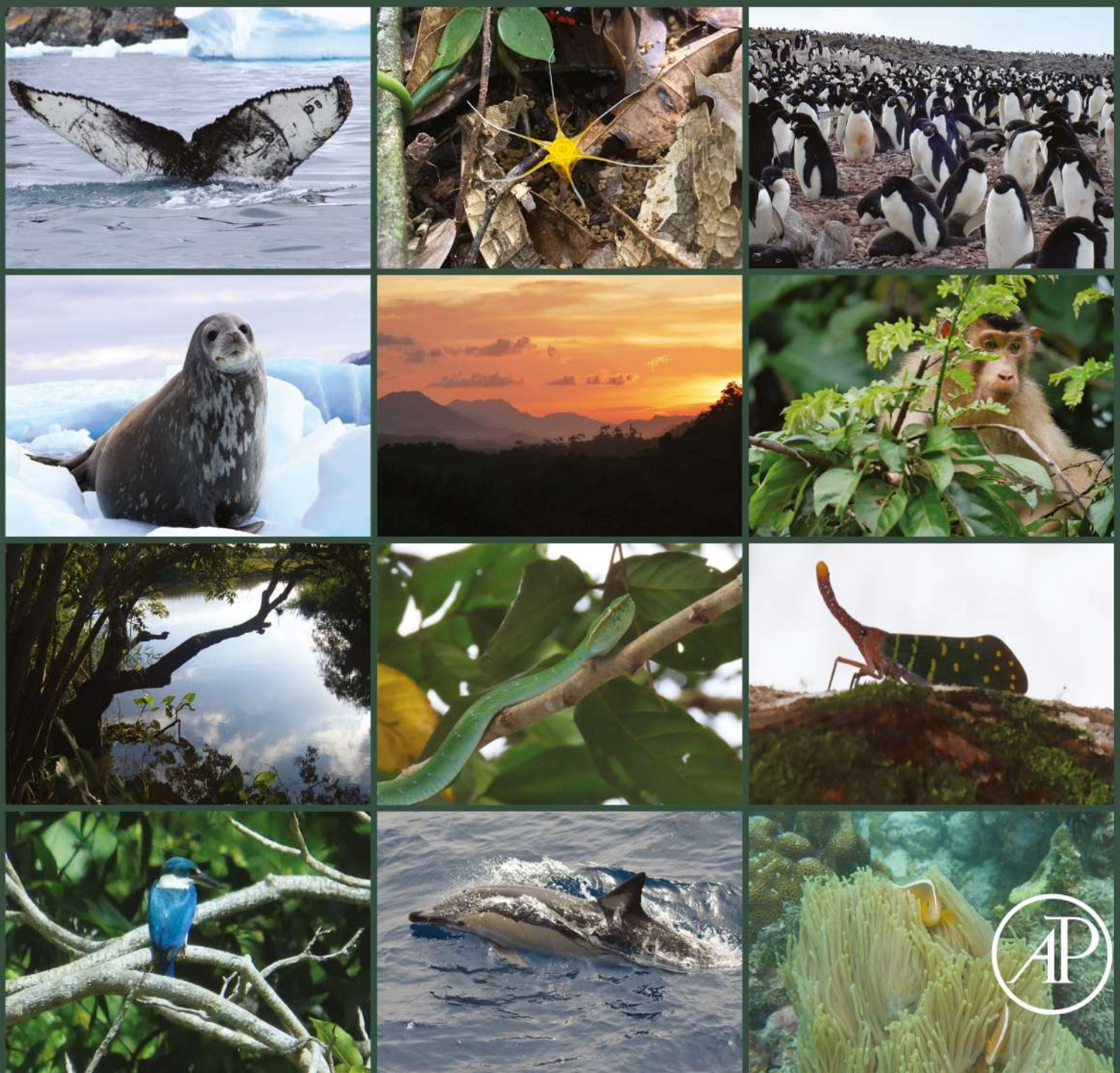
Volume 2

Edited by

Germano Leão Demolin-Leite

TERRESTRIAL BIOMES:

Global Biome Conservation and Global Warming Impacts on Ecology and Biodiversity



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Germano Leão Demolin-Leite

Universidade Federal de Minas Gerais, Montes Claros, Minas Gerais, Brazil

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Dedication

**I dedicate this book to my mother, Leuma Leão Demolin-Leite,
for her dedication to our family
and the Brazilian people who pay my salary.**

Germano Leão Demolin-Leite
Universidade Federal de Minas Gerais,
Montes Claros, Minas Gerais, Brazil

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Introduction

Human activities exert significant pressure on Earth's most diverse biomes, which has escalated notably since the Industrial Revolution, exemplified by increased greenhouse gas emissions. This trajectory could propel the planet into an unprecedented era of ecological transformation.

Islands confront heightened vulnerability due to climate-induced changes, threatening unique endemism and jeopardizing the persistence of various species. The planet's rising temperature also impacts other terrestrial biomes, including deserts, leading to increased fires in Mediterranean biomes, shrublands, savannas, and wetlands (e.g., Pantanal). Additionally, boreal forests, grasslands, and steppes, crucial for global carbon dynamics, experience rising temperatures and altered precipitation patterns, which influence plant species composition and biome functioning. These changes elevate the risk of invasive species, affecting biodiversity in temperate forests, among other biomes. Tropical forests, exemplified by the Amazon, renowned for their biodiversity, face threats from deforestation, changing precipitation patterns (e.g., extreme droughts), and rising temperatures, endangering numerous species. Global warming triggers permafrost thaw, alters migration patterns, and shifts tundra vegetation dynamics, resulting in profound implications for Arctic biodiversity. Subterranean biomes, often overlooked, harbor unique species adapted to specific conditions, facing threats from changing hydrological regimes and land-use practices.

Therefore, this book *Terrestrial Biomes: Global Biome Conservation and Global Warming Impacts on Ecology and Biodiversity* adopts an approach that explores the interaction between global climate change and Earth's diverse biomes, which form the foundation of biodiversity. This second volume delves into terrestrial biomes, investigating innovative conservation and remote sensing techniques while considering the imminent trajectory of global warming over the next half-century in these biomes.

Moreover, Volume 2—*Terrestrial Biomes* encompasses themes such as deserts (e.g., Sahara Desert), insular biomes, the Mediterranean basin, prairies, savannas, shrublands, steppes, subterranean biomes, taiga, temperate, and tropical forests (e.g., Amazon Forest), and tundra. The chapters in this volume, in general, present an introduction, geographical description, biodiversity, conservation areas and their ecological indicators, human activities and their impacts, operational and potential monitoring tools for conserving or recover the biome, conclusions, and perspectives.

Reflecting on the interaction between global warming and biome conservation underscores the imperative of a holistic approach to preserving biodiversity and biome resilience. Conservation efforts must transcend geopolitical borders, emphasizing sustainability, resilience, and adaptive management as imperative principles for biome conservation and the survival of human civilization. Integrating conservation with climate mitigation and adaptation measures is not only scientifically sound but also crucial for preserving life on Earth, representing a collective responsibility. The scientific community must continually enhance its understanding, by employing cutting-edge research methodologies, monitoring technologies, and modeling approaches. Furthermore, the knowledge presented in the *Terrestrial Biomes: Global Biome Conservation and Global Warming Impacts on Ecology and Biodiversity* serves to raise awareness and engage society, including policymakers, in fostering a global commitment to sustainable practices, biodiversity preservation, and climate mitigation.

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Section A

Deserts

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Chapter 1

Antarctica desert

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1 Introduction

The Antarctic area comprises about 10% of the planet's surface and the continent has approximately 14 million km^2. It is considered the coldest, driest, and windiest continent with the highest average altitude and a vast ice sheet, exceeding 4600 m in thickness at certain locations. Technically classified as the largest desert globally, Antarctica holds approximately 70% of the Earth's freshwater, primarily in the form of ice. Furthermore, it is the least polluted continent on Earth (Goldsworthy and Hemmings, 2009; Bell and Seroussi, 2020).

About 34–22 million years ago, a strong Antarctic Circumpolar Current (ACC) was established (Fig. 1.1), leading to the isolation of the Antarctic region from other continents. The oceanographic frontal system of ACC is termed the Antarctic Polar Front, where Antarctic cold surface water descends beneath the warmer, less saline water of the middle latitudes. This created a physical barrier for Antarctic colonization by organisms insufficiently adapted to polar temperatures. Under the selective pressure exerted by successive glaciations, endemic and cold-adapted organisms emerged (Clarke and Johnston, 1996; Barker and Thomas, 2004). Terrestrial endemism varies greatly across the biogeographic regions of Continental Antarctica, Maritime Antarctica, and Subantarctic islands (Fig. 1.2) (Cucini et al., 2022). Most continental species are endemic in contrast to Maritime Antarctica, where endemism is not as pronounced. The Subantarctic islands also show marked endemism, but there are some differences in the distribution of species between the islands located to the west (South Georgia) and the east (Macquarie Island) (Rogers, 2007).

In contrast to the relatively stable thermal conditions of the Antarctic marine environment, characterized by temperatures of approximately −1.96°C at high latitudes and around +3°C near Antarctic islands, the terrestrial environment exhibits much more expressive variations. The surface air temperature in the continent reached as low as −89.2°C at the Vostok Antarctic station, but even colder temperatures have been detected by satellites in East Antarctica. Conversely, the Antarctic Peninsula region has undergone accelerated warming, with the highest temperature recorded thus far in the region occurring in February 2020 at the Esperanza Argentine Antarctic Station (+18.3°C) (Barker and Thomas, 2004; Barnes et al., 2006; Turner et al., 2021).

Temperature has been identified as Antarctica's main environmental driver of faunal biodiversity. Except for birds and mammals, other Antarctic organisms are poikilothermic, in which seasonal thermal variations profoundly impact their growth, locomotion, and respiration rates. Adaptation to low temperatures involved, in part, the selection of more flexible protein structures that exhibit distinct physicochemical properties, enabling activity at subzero temperatures. However, more flexible structures are thermally less stable and more susceptible to denaturation under heating conditions. Thus, some Antarctic organisms are stenothermal and highly susceptible to increases in ambient temperature (Hochachka and Somero, 2002; Clarke, 2003; Somero, 2004).

2 Geographic location

Antarctica occupies an asymmetric position around the Earth's South Pole and is encircled by the Southern Ocean (Fig. 1.1). This ocean is formed by the oceanic provinces south of the Atlantic, Indian, and Pacific Oceans.

Topographically, Antarctica is divided by the Transantarctic mountain range into East and West Antarctica (Fig. 1.3).

East Antarctica originated from the fragmentation of the Gondwana supercontinent and two rocky mountain ranges stand out there: the Gamburtsev, reaching heights of 3000 m at its apex in the center of the continent (Dome A) at 80°S; and the Transantarctic Range, boasting elevations up to 4500 m, which serves as the boundary between East Antarctica and the West Antarctic rift system. Situated between the East Antarctic Ice Sheet and the Ross Sea lies the McMurdo Dry Valleys, a primarily ice-free

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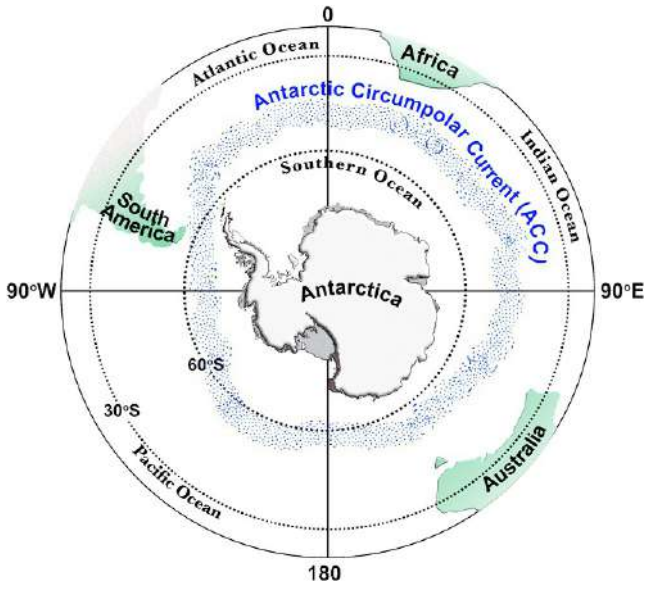

FIG. 1.1 Geographic location of Antarctica. The Antarctic Circumpolar Current (ACC) is a surface ocean current that physically isolates the Southern Ocean from other oceans. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credits: The authors redrawn from Barker, P.F., Thomas, E., 2004. Origin, signature and palaeoclimatic influence of the Antarctic Circumpolar Current. Earth Sci. Rev. 66, 143–162. https://doi.org/10.1016/j.earscirev.2003.10.003.)*

region starkly contrasting with the vast expanse of the East Antarctic Ice Sheet. Encompassing an area of 4800 km^2 (77–78°S, 160–164°E), this arid polar desert originated from the obstruction of ice flow from East Antarctica by the Transantarctic Range. The limited snow precipitation in the McMurdo Dry Valleys does not accumulate as it sublimates before melting. Consequently, water infiltration into the soil is restricted, impacting biodiversity (Obryk et al., 2020).

The geological origin of West Antarctica is more recent and complex. This region, 4700 km long and up to 1600 km wide, resulted from tectonic activity between the Paleo-Pacific plate and the East Antarctic continental margin. Physiographically, West Antarctica can be divided into three provinces: (1) the Weddell Sea sector, encompassing a network of rifts situated in the shallow waters of the Weddell Sea, the elevated terrain of the Nunataks Haag, and the Ellsworth-Whitmore mountain range; (2) the Ross Sea sector, including the Marie Byrd Land Dome, the West Antarctic rift system, and the relatively low-lying Ross Embayment; and (3) the Antarctic Peninsula sector, comprising the Antarctic Peninsula and Thurston Island regions (Fig. 1.3). The South Shetlands Islands, situated at the northern tip of the Antarctic Peninsula and hosting numerous scientific facilities, are of particular significance. King George Island, the largest island within the archipelago, accommodates 10 Antarctic Stations and attracts thousands of tourists annually. Additionally, an Antarctic Specially Managed Area (ASMA) and six Antarctic Specially Protected Areas (ASPA) are located at King George Island (Braun et al., 2001; Hall, 2009; Council of Managers of National Antarctic Programs (COMNAP), 2017; Jordan et al., 2020; Secretariat of the Antarctic Treaty (SAT), 2024).

A group of isolated and scattered oceanic islands surrounding the Antarctic Continent, situated at latitudes ranging from 46 to 54°S, is categorized as Subantarctic islands. The principal ones include South Georgia, South Orkney, Marion, Heard and MacDonald, Crozet, and Macquarie. The ecosystems of these islands exhibit greater diversity and complexity compared to the coastal environments of Maritime Antarctica and Continental Antarctica (Degma and Gábrišová, 2023).

Beneath the Antarctic ice sheet and glaciers lie 675 subglacial lakes. The thickness of the ice sheet covering these lakes varies considerably, with Mercer Lakes in West Antarctica and Vostok Lake in East Antarctica having ice sheets of 1100 and 4000 m thicknesses, respectively. Subglacial lakes are isolated environments that may serve as habitats for microbial communities with a unique evolutionary trajectory. Insights into this biota can contribute to a better understanding of Antarctic ecosystems. The basal hydrology and biogeochemical fluxes of these lakes are also critical, as they influence the dynamics of Antarctic ice flow (Livingstone et al., 2022).

The negative mass balance of Antarctic ice is currently not deemed significant. According to global climate models, augmented snowfall in central East Antarctica is counterbalancing the hastened melting of glaciers in West Antarctica (Fig. 1.4), which are experiencing accelerated warming. This phenomenon would minimize the impacts of Antarctic ice melting on global sea levels (Bell and Seroussi, 2020).

Antarctic ice cover can be categorized into three main types: (1) slow-moving ice anchored above the ground; (2) fast-moving glacier ice; and (3) the floating ice of ice shelves. Slow-moving ice moves due to its internal deformations and displaces at a rate of approximately 1 m per year. Most of this ice is found in East Antarctica and contains enough water to elevate sea levels by 58 m. Fast-moving glacier ice advances toward the ocean at speeds of up to 4 km per year, with some glaciers discharging ice streams as wide as 100 km. Ice shelves are flat, floating bodies of ice that are interconnected with outflow glaciers. The area of these platforms exceeds 1.5 million km^2 around Antarctica (Bell and Seroussi, 2020).

3 Biodiversity

3.1 Terrestrial biota

The main terrestrial ecosystems are primarily situated along the peripheries of the Antarctic Continent, the Antarctic Peninsula, and Subantarctic islands. These areas are free of ice or snow during a certain period of the year and

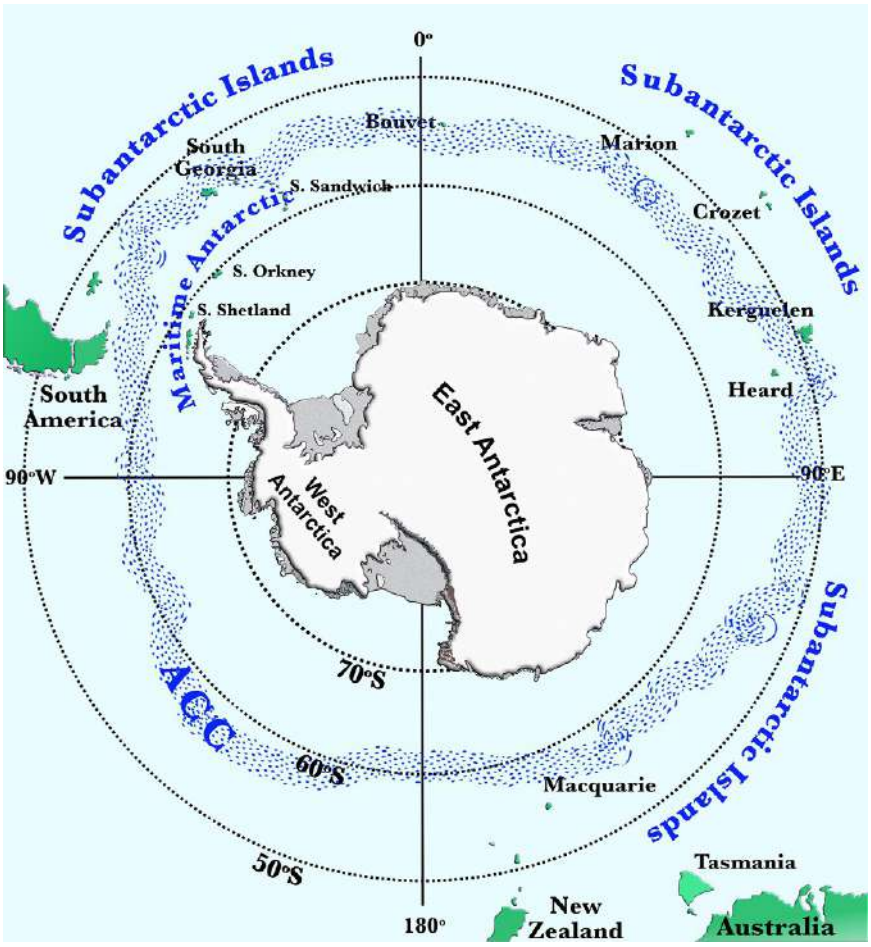

FIG. 1.2 The terrestrial biogeographic zones: Continental Antarctica (East and West Antarctica), Maritime Antarctica, and Subantarctic, within the Antarctic region. ACC: Antarctic Circumpolar Current. The map delineates study areas and does not necessarily depict accepted national boundaries. *(Credits: The authors redrawn from Convey, P., 2017. Antarctic ecosystems. In: Reference Module in Life Sciences. Elsevier, pp. 179–187. https://doi.org/10.1016/B978-0-12-809633-8.02182-8.)*

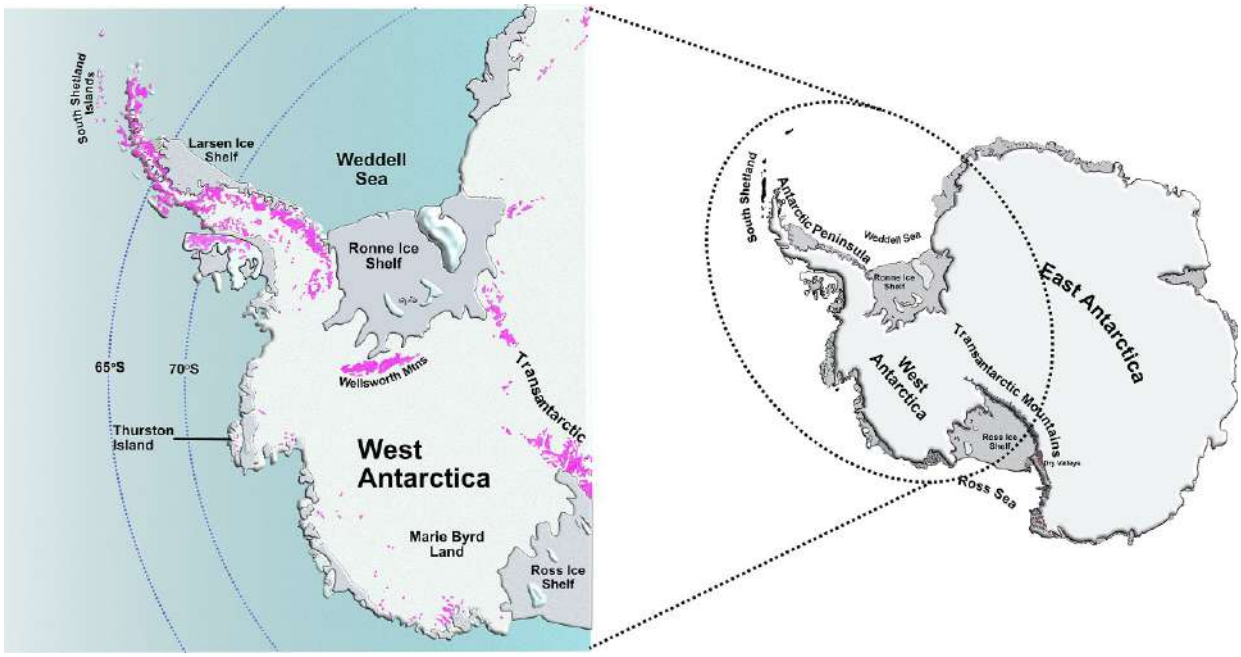

FIG. 1.3 Geographic location of East and West Antarctica. The predominant geological formations found in the three regions of West Antarctica are depicted in magenta coloration. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credits: The authors redrawn from Paulsen, T., Deering, C., Sliwinski, J., Valencia, V., Bachmann, O., Guillong, M., 2017. Detrital zircon ages and trace element compositions of Permian-Triassic foreland basin strata of the Gondwanide orogen, Antarctica. Geosphere 13(6), 2085–2093. https://doi.org/10.1130/GES01482.1.)*

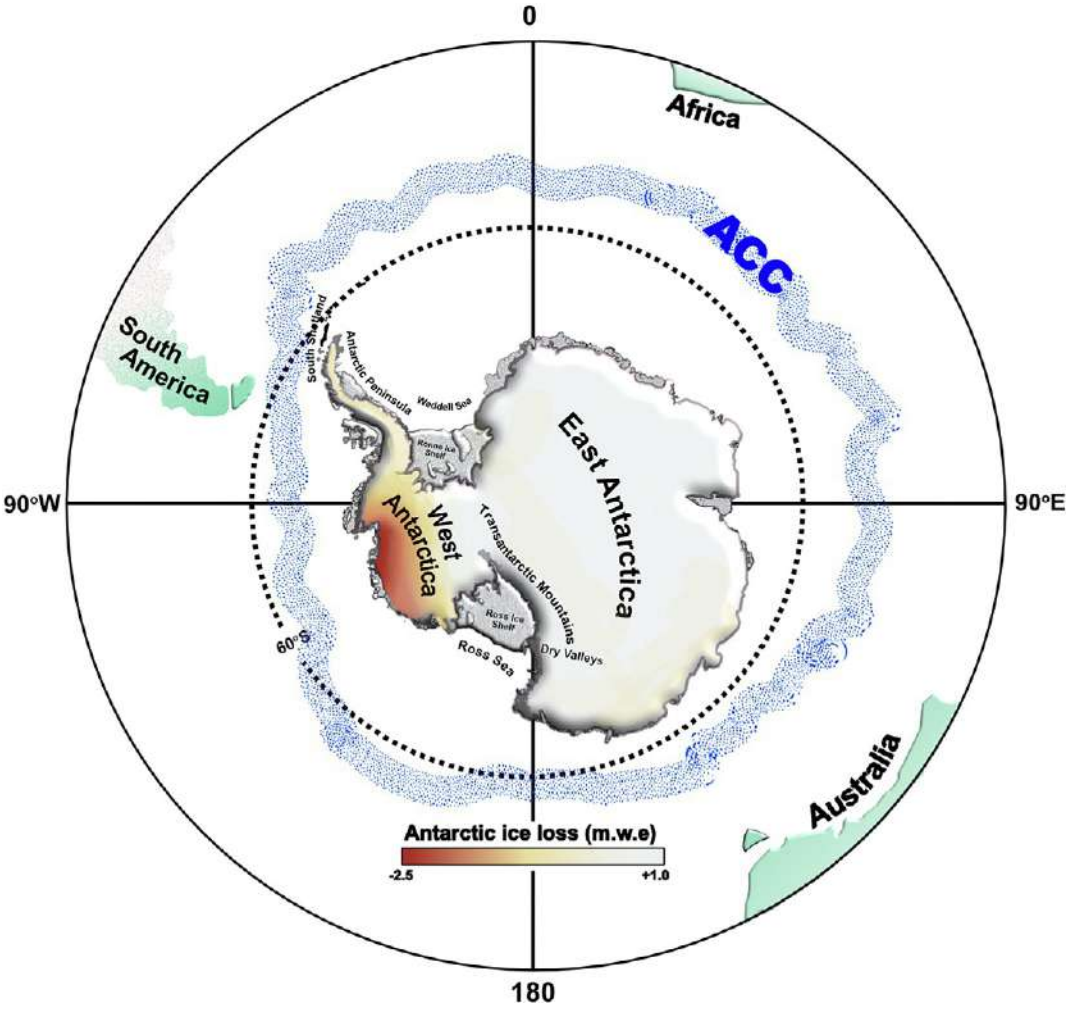

FIG. 1.4 Antarctic ice mass balance. Spatial ice mass loss (m.w.e.) in the period from 2002 to 2017. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credits: The authors redrawn from Bell, R.E., Seroussi, H., 2020. History, mass loss, structure, and dynamic behavior of the Antarctic Ice Sheet. Science 367(6484), 1321–1325. https://doi.org/10.1126/science.aaz5489.)*

account for less than 0.5% of Antarctica. Conversely, the ecosystems within the interior of the continent, found on nunataks, rocky mountains, and frozen desserts, exhibit lower diversity and endure even more extreme conditions of temperature and low humidity. Except for microorganisms and viruses, terrestrial species diversity is limited and heavily influenced by liquid water availability (Convey and Peck, 2019). Warmer summers tend to expand ice-free areas but can affect the distribution and abundance of freshwater. The rapid melting of snowbanks during spring and summer provides an abundance of freshwater for plants; however, as these banks deplete, organisms encounter periods of water scarcity (Lee et al., 2022b).

The food chain of Antarctic terrestrial ecosystems is relatively simple and dominated by a few species of plants (grasses and mosses) and invertebrates (springtails, mites, flies, Tardigrada, nematodes, rotifers, and enchytraeids). The harsh condition of extreme cold and food shortages prevents birds, mammals, amphibians, and reptiles from surviving permanently in the icy interior of the continent. However, birds and seals seasonally frequent the coastal regions of Antarctica and the Subantarctic Islands, either to form colonies or in search of resting spots. These visits promote the transfer of nutrients from the marine to the terrestrial environment and fertilize the soils, especially around the bird colonies (Convey and Peck, 2019; Bokhorst et al., 2022). Because Antarctic vertebrates depend on the ocean for their food, they are classified as marine. Unlike most of the Arctic, Antarctica's severe conditions do not support the development of plant life sufficient to sustain vertebrate populations, resulting in the absence of herbivorous vertebrates in Antarctica (Eisert, 2015). In addition, there are no native freshwater fish from Antarctica, although several salmonid fish have been introduced into Subantarctic lakes (Dartnall, 2017).

The extreme seasonality of the photoperiod promotes a veritable explosion of life during the austral summer and imposes a long period of famine during the winter. The high luminosity of summer and the milder temperatures accelerate primary production and provide food abundance in the ice-free areas around the continent. In winter, low light and ice/snow cover on land and sea environments limit primary production and impose a long period of famine. Birds and marine mammals typically migrate during the winter in search of food. Survival strategies for nonmigratory organisms, primarily ectothermic species, involve metabolic dormancy and the reduction of energy expenditures by dynamic actions for feeding. This metabolic seasonality directly impacts the growth rates of these organisms, which are significantly higher in the summer than in the winter (Clarke et al., 2004; Rodrigues et al., 2007).

In the extreme polar deserts of Antarctica, such as the dry McMurdo valleys, diversity is low consisting of no more than five main taxonomic groups. The absence of higher plants and the sparse presence of mosses, algae, and lichens contribute to the very low carbon concentrations and low primary productivity in this environment. The most biodiverse communities are found in moist soil habitats near ephemeral streams and glacier meltwater sources, environments characterized by higher levels of net primary productivity (Bamforth et al., 2005).

3.1.1 Vertebrates

Birds

Native terrestrial vertebrates found in the Subantarctic area, on South Georgia island and Kerguelen archipelago, are limited to a single endemic insectivorous passeriform (*Anthus antarcticus*), freshwater ducks (*Anas georgica georgica* and *Anas flavirostris*), and two scavenging sheathbills (Holdgate, 1977; Martin, 2013; Convey, 2017). However, 29 species of birds breed on South Georgia Island, and this is the main breeding site for six species: Macaroni Penguin (*Eudyptes chrysolophus*), gray-headed Albatross (*Thalassarche chrysostoma*), Northern Giant Petrel (*Macronectes halli*), Antarctic Prion (*Pachyptila desolata*), White-chinned Petrel (*Procellaria aequinoctialis*), and Common Diving Petrel (*Pelecanoides urinatrix*).

South Georgia was designated by BirdLife International as an Important Bird Area (IBA). However, it faces several threats, including predation by rats, which prey on bird eggs, as well as the impacts of regional climate change and accidental mortality resulting from human fishing activities in the region (Clarke et al., 2012). A new potential threat is the ingestion of microplastics by King Penguins (*Aptenodytes patagonicus*) foraging from South Georgia, the consequences of which are still unknown (Le Guen et al., 2020).

Among the avian fauna inhabiting Antarctica, four species are categorized as near threatened (NT) according to the International Union for Conservation of Nature (IUCN) Red List. These species comprise the Emperor penguin (*Aptenodytes forsteri*), Gentoo penguin (*Pygoscelis papua*), Adélie penguin (*Pygoscelis adeliae*), and Light-mantled Albatross (*Phoebetria palpebrata*). Additionally, the Macaroni penguin (*E. chrysolophus*) is classified as vulnerable (VU) (Harris et al., 2015).

Climate change has exerted a distinct impact on Antarctic organisms. The warming trend observed in the surface temperatures of the Southern Ocean has led to a substantial reduction in circumpolar sea ice coverage. Over the past seven summers, this reduction has surpassed the previous record three times. Furthermore, sea ice extent during autumn and winter has exhibited exceptionally low levels. Such changes have notably affected sea-ice-dependent organisms, particularly the Antarctic krill, which, in its larval phase, relies on the biota associated with sea ice for sustenance. This crustacean is a pivotal component of the marine food web since all Antarctic vertebrates feed on krill directly or indirectly. The diminishment of krill populations, whether due to increased harvesting or decreased recruitment, exerts a profound impact on the dynamics of coastal ecosystems in Antarctica (Flores et al., 2012; Hughes et al., 2021; Purich and Doddridge, 2023).

Penguins rank among the species most susceptible to the warming trends observed in Antarctica. Populations of Adélie and Chinstrap penguins inhabiting colonies situated in the Palmer Archipelago and the South Shetland Islands, located on the Antarctic Peninsula, have experienced notable declines, with some colonies ceasing to exist altogether in recent decades. These two species of penguins are planktivores and feed on krill of different sizes. Sea ice serves as a resting habitat for these birds during their foraging activities. Conversely, demographic analyses spanning two 4-year periods (1978–81 and 2014–18) reveal a marked increase of 216% in the population of Gentoo penguins within the ASPA#128 colony, situated on King George Island. Unlike their Adélie and Chinstrap counterparts, Gentoo penguins are piscivorous and exhibit distinct ecological traits. Consequently, the reduction in sea ice and krill abundance has not adversely impacted the reproductive rates of Gentoo penguins (Trivelpiece et al., 1987; Adams et al., 1989).

Most emperor penguin colonies are situated on fast ice (sea ice anchored to soil and grounded icebergs). These penguins (the tallest and heaviest Antarctic penguin) primarily rely on fish (e.g., *Pleurogramma antarcticum*) as their main dietary component, besides small crustaceans and squids. Their successful feeding requires the proximity of colonies to the junction of fast ice and offshore pack ice. Projections under a scenario of 1.5°C warming by 2100

indicate a substantial decrease in the extent of offshore pack ice regions. Consequently, this will escalate the foraging demands of emperor penguins, rendering them the most susceptible Antarctic taxon to warming trends (Lee et al., 2022a).

Seals

There are six species of seals inhabiting Antarctica, with five belonging to the Phocidae family: the Weddell seal (*Leptonychotes weddellii*), crabeater seal (*Lobodon carcinophagus*), leopard seal (*Hydrurga leptoynx*), Ross seal (*Ommatophoca rossii*), and elephant seal (*Mirounga leonina*). The eared seal, also recognized as the Antarctic fur seal (*Arctocephalus gazella*), falls under the Otariidae family and is colloquially termed a seal despite not being taxonomically classified as one. Climate change exhibits nonuniform impacts on these seals, eliciting varied responses within Antarctic ecosystems. Crabeater seals demonstrate highly specialized predation behaviors targeting Antarctic krill, primarily breeding on seasonal ice packs. In contrast, Weddell seals, acting as generalist predators, breed on stable fast ice. Consequently, crabeater seals endure a more pronounced impact from climate change compared to Weddell seals (Wege et al., 2021).

3.1.2 Protists and invertebrates

The terrestrial Antarctic fauna basically comprises Protozoa and invertebrates, encompassing various taxonomic groups including Protozoa, Rotifera, Tardigrade, Nematoda, Platyhelminthes, Gastrotricha, Annelida (Oligochaeta), Mollusca, Crustacea, Insecta, Arachnida (Araneida, Acarina), and Myriapoda (Convey, 2017).

Protozoa

Regarding phagotrophic protists (formerly Protozoa), Antarctica is estimated to harbor 539 distinct taxa (Thompson et al., 2019). The North Antarctic Peninsula, East Antarctica, and South Victoria Land show greater diversity, with 155, 52, and 50 species, respectively, compared to the rest of the regions. Ciliophora is the main group in Antarctica, followed by Amoebozoa, Cercozoa, Discoba, and then by members of Opisthokonta, Apicomplexa, Stramenopiles, and nonciliated alveolates. Overall, 81% of taxa are endemic to one region, with no single region encompassing the entirety of phagotrophic protist diversity on the continent. Only 19 of the 236 identified species were found in three or more regions. Among these 19, nine were testate amoebae (four Amoebozoa and five Cercozoa), nine were Ciliophora, and one was a flagellated cercozoan; no naked amoebae were identified. The most common species is *Corythion dubium* (a testate amoebae cercozoa) found in 8 of the 11 regions surveyed. Cercozoans have been recorded in all sampled regions, but the group is predominantly composed of testate amoebae (27 out of a total of 39 Cercozoans), with flagellated cercozoans accounting for only 23%. Mosses host the highest diversity of phagotrophic protists totaling 116 species, with *C. dubium*, *Trinema lineare*, *Centropyxis aerophila*, *Assulina muscorum*, *Difflugia lucida*, and *Euglypha rotunda* being the most frequent and abundant. Fewer species ($n = 35$) were found exclusively in soil, whereas 50 species were present in both mosses and soil. *C. dubium* is the most common species in mosses and soil, whereas cercozoan flagellates designated as *Heteromita globosa* are the most widely distributed species, found exclusively in soil (Thompson, 2021). However, *H. globosa* was reclassified by Howe et al. (2009), resulting in the creation of five new genera and 29 species, rendering the name invalid (Thompson et al., 2019). Further studies are warranted to accurately identify Antarctic species previously designated by this name.

Rotifera

Rotifers are abundant in almost all habitats of Antarctic regions, including permanent freshwater, brackish, and marine water bodies, as well as in proximity to mosses, lichens, soils, and peculiar habitats such as glaciers, meltwater pools, algal mats, and cryoconite holes. Monogonant rotifers are mainly present in aquatic environments, whereas bdelloids, although aquatic, live predominantly in limnoterrestrial environments such as mosses, lichens, and soils (Fontaneto et al., 2015). Antarctic bdelloids need to survive in freezing and desiccated environments and exhibit cryptobiosis, during which they enter a dormant state with temporary cessation of any metabolic activity (Lukashanets and Maisak, 2023). The number of rotifer species is higher in the Subantarctic ($n = 103$) compared to Continental ($n = 59$) and Maritime ($n = 55$) Antarctica (Fontaneto et al., 2015). However, new Antarctic species have been reported (Iakovenko et al., 2015). *Adineta grandis*, *Philodina alata*, *Philodina antarctica*, *Philodina gregaria*, *Philodina jeanelli*, *Habrotrocha angularis*, and *Mniobia ostensa* are endemic species of Antarctica (Iakovenko et al., 2015). Interestingly, neither *A. grandis* nor *P. gregaria* have been found in the South Shetland Islands, an archipelago that seemingly falls within their range (Dartnall, 2017).

Tardigrada

Tardigrades live in marine or freshwater environments and among mosses, liverworts, and lichens. The phylum comprises two main classes—Heterotardigrade and Eutardigrade (Dartnall, 2017). Limnoterrestrial environments are most commonly colonized by female populations of eutardigrade and heterotardigrade reproducing by parthenogenesis. Tardigrades are exclusively oviparous and survive

in harsh environments through two forms of dormancy: cryptobiosis and diapause. Four types of cryptobiosis are known: anhydrobiosis, cryobiosis, anoxybiosis, and osmobiosis. The first two forms have been of particular importance for limnoterrestrial tardigrades of Subantarctic regions. Diapause occurs through encysting and resting eggs. Cyst formation is observed in tardigrades inhabiting moss, soil, and freshwater, with cysts capable of surviving for months without depleting their energy reserves (Degma and Gábrišová, 2023). The eutardigrade *Acutuncus antarcticus* and its eggs frozen at −20°C for 30.5 years were successfully revived, and the individuals reproduced, as well as their frozen eggs developed into fertile individuals (Tsujimoto et al., 2019).

About 75 species of Antarctic tardigrades are known. Endemic species include *Barbaria pseudowendti*, *Dastychius improvisus*, and *Mesobiotus blocki*, all exclusively recorded in Enderby Land, East Antarctica. *A. antarcticus* is a pan-Antarctic species, however, as it reproduces by parthenogenesis, isolated populations may be undergoing speciation processes (Kaczmarek et al., 2020). On the Subantarctic islands, 49 species have already been found, however, yet 29 of them are of doubtful identification, requiring further confirmatory studies (Degma and Gábrišová, 2023).

Nematoda

The total number of terrestrial, freshwater, and saline nematode species in the Antarctic region stands at approximately 68. Among these, roughly 34 species have been documented in mainland Antarctica, while 37 are found in Maritime Antarctica and only three species are shared between both regions (Dartnall, 2017).

The McMurdo Dry Valleys in Antarctica are one of the coldest and driest deserts on the planet. Despite these extreme conditions, nematodes are the most abundant and widely distributed metazoans in this region where *Plectus murrayi*, *Eudorylaimus antarcticus*, and *Scottnema lindsayae* are the most common species. *S. lindsayae* is a generalist that feeds on bacteria and fungi and is found in large numbers in the dry, saline soils of the McMurdo Dry Valleys. *P. murrayi* is a bactivore, while *E. antarcticus* is an omnivore that feeds on algae, cyanobacteria, and probably other metazoans. *P. murrayi* and *E. antarcticus* prefer habitats with higher soil moisture levels than *S. lindsayae* (Caruso et al., 2019; Robinson et al., 2023). Since 2001, the McMurdo Dry Valleys have experienced relatively moderate warming compared to other parts of Antarctica. However, this warming trend has decreased the population of *S. lindsayae* while increasing that of *P. murrayi*, with possible consequences for the composition of the soil community in this area (Robinson et al., 2023). In the Darwin Glacier region of Victoria Land, East Antarctica, *S. lindsayae* is the only nematode found in the soil, and its abundance increases with altitude, which in turn correlates with soil moisture (Zawierucha et al., 2019).

Platyhelminthes

In the Southern Ocean, about 30 species of flatworms (Turbellaria) are known (Volonterio and Ponce de León, 2018). However, few species have been found in Antarctic freshwater, namely: *Prorhynchus insularis*, *Neoophora* sp., *Kalyptorhynchia* sp., *Curtisia secluse,* and *Duplominona amnica* (Dartnall, 2017).

No freshwater parasitic worms (Trematoda, Monogenea, and Cestoda) have been reported in the Antarctic region (Dartnall, 2017). Conversely, approximately 60 species of parasitic trematodes of Antarctic fishes are known, belonging to 28 genera and 10 families (Faltýnková et al., 2022). Additionally, Cestoda species have also been found in penguins (Diaz et al., 2016).

Gastrotricha

The occurrence of Gastrotricha has been reported in marine sediments of the Antarctic region (Kieneke, 2010; Fonseca et al., 2017) and few species have been found in freshwater environments within this region, including old records of specimens from the genus *Chaetonotus* and *Lepidodermella*. The South Orkney Islands exhibit the highest diversity, hosting three Gastrotricha species (Dartnall, 2017).

Annelida

Polychaeta and Clitellata (comprising subclasses Oligochaeta and Hirudinea) are classes within the phylum Annelida (Schmelz et al., 2021). Still, only oligochaetes have been found in the terrestrial or freshwater environment of Antarctica. These oligochaetes belong to six families: Acanthrodrilidae, Lumbricidae, Naididae, Phreodrilidae, Tubificidae, and Enchytraeidae. In Antarctica, most species belong to Enchytraeidae (Dartnall, 2017). An estimated 23 species of oligochaetes occur on Subantarctic islands (Convey, 2017), with the greatest diversity on Macquarie Island, where nine species have been found (Dartnall, 2017). Within the Warren subglacial cave on Mount Erebus, an active volcano in Victoria Land, Antarctica, evidence of the occurrence of two species of oligochaetes from the genera *Achaeta* and *Bryodrilus* was found (Fraser et al., 2018).

Mollusca

No freshwater snails or bivalves have been recorded on the continental Antarctic. However, terrestrial snails and slugs have been discovered on Subantarctic islands, including species such as *Notodiscus hookeri*, found on Prince Edward, Crozet, Kerguelen, and Heard islands (Pugh and Scott, 2002).

Crustacea

Among the freshwater invertebrates of the Antarctic region, crustaceans are one of the most diverse groups, where eight main orders of crustaceans are represented (Anostraca, Cladocera, Podocopida, Calanoida, Cyclopoida, Harpacticoida, Amphipoda, and Isopoda). In mainland Antarctica, in the region called Maritime Antarctica (Antarctic Peninsula, South Orkney, and South Shetland Islands) and in the Subantarctic islands, seven, nine, and 46 species were found, respectively. Many taxa are members of zooplankton communities, occupying a broad array of ecological niches and adapting to environmental changes, such as rising temperatures (Díaz et al., 2019, 2021). Furthermore, four species of terrestrial crustaceans have been documented on the Subantarctic islands (Convey, 2017).

Invasive species of marine crustaceans have been found in the Antarctic regions where warming has occurred (Aronson et al., 2015), but little is known about the impact of this phenomenon on terrestrial and freshwater crustaceans within this region.

Insecta

Indigenous insect diversity is very low in the Antarctic Peninsula, where only *Belgica antarctica* was recorded. In the South Shetland Islands, *Parochlus steinenii* and *B. antarctica* were the only identified species. However, other islands demonstrate a higher species richness, with the combined McDonald and Heard islands presenting 12 taxa, South Georgia Island with 17 taxa, Marion and Prince Edward islands combined hosting 19 taxa, Macquarie Island with 24 taxa, and Kerguelen Island displaying the highest diversity with 28 taxa (Chown and Convey, 2016, and their Supplemental Table 1).

One of the most known features of Antarctic insect fauna is the prevalent occurrence of flightlessness and disharmony. The term disharmony was initially introduced to characterize biotas featuring a considerable abundance of species capable of easy dispersal; however, subsequent evolutionary processes often led to rapid adaptations resulting in the loss of flight ability. The inability to fly has been attributed to the concept that significant allocation of resources toward the development and maintenance of the flight apparatus has been foregone in favor of maximizing growth and reproductive output, particularly in stable, low-energy environments; however, the presence of strong winds in the region may have also favored the prevalence of this trait (Leihy and Chown, 2020).

Invasive alien species, climate change, and their interactions are the main threats to Antarctic insects. Substantial impacts of introduced predators, such as rodents, on Indigenous insect populations have been extensively documented across several islands, resulting in population declines and alterations in ecosystem processes, notably mineralization, and peat formation. It is anticipated that under climate warming conditions, rodents will experience heightened survival rates and reproductive success, exacerbating their impact further. While it could be hypothesized that elevated temperatures might be advantageous for insects, empirical evidence suggests otherwise. Many species studied exhibit significant mortality after prolonged exposure to 15°C, despite possessing higher short-term and lethal thresholds. Additionally, the introduction of alien insect species has been observed to induce alterations in indigenous populations on the islands (Chown and Convey, 2016).

Arachnida

A total of over 260 species of arachnids are estimated to inhabit the Antarctic region, with the highest biodiversity found on the Subantarctic islands, followed by mainland Antarctica and maritime Antarctic islands (Convey, 2017). Pseudoscorpions, harvestmen, and spiders are present on Subantarctic islands, while certain groups of mites are documented on the mainland (Beron, 2018). Antarctic freshwater arachnids encompass taxa from the orders Acaridida, Oribatida, Gamasida, and Prostigmata, with aquatic spiders remaining unreported on mainland Antarctica (Dartnall, 2017).

Heat stress diminishes the fertility of male Antarctic mites, potentially impeding their population growth. Even brief periods of nonlethal heat stress can induce significant reproductive effects on local populations (Benoit et al., 2023).

Myriapoda

The Myriapoda are notably absent on the Antarctic continent (Schierwater and DeSalle, 2021) as well as on various islands including the South Shetland, South Orkney, South Georgia, South Sandwich, Bouvetøya, Prince Edward, Kerguelen, Heard, Macquarie, Antipodes, and Bounty islands (Pugh, 2013). However, three Myriapoda species have been reported on the Crozet Islands, located in the Subantarctic region, belonging to the genera *Symphylella*, *Pauropus*, and *Stylopauropoides* (Scheller, 1974a, b; Convey, 2017).

3.1.3 Plants

Cryptogams (bryophytes and lichens) constitute the predominant vegetation of Antarctica. Bryophytes are subdivided into mosses and liverworts and are abundant in humid environments. Lichens preferentially colonize dry sites and rock surfaces throughout the Antarctic region (Sancho et al., 2019).

Concerning bryophytes, 133 moss taxa have been reported in the Antarctic region, with 24 found on the continent itself (Colesie et al., 2023). Additionally, 27 species

of liverworts are known, with *Cephaloziella varians* being the most common and widely distributed, being the sole liverwort species occurring in continental Antarctica (Newsham, 2010). Despite the extremely cold conditions of the Antarctic winter, certain moss species exhibit photosynthetic apparatuses capable of functioning at relatively warm temperatures, with maximal CO_2 uptake occurring between 20°C and 30°C and maximal mitochondrial respiratory rates at 35°C (Perera-Castro et al., 2020). However, due to the physiological diversity among Antarctic bryophytes, predicting their survival under climate change is a complex task. Warmer temperatures are thought to potentially stimulate the growth and reproduction of these bryophytes, but they may also experience harmful effects from more frequent desiccation cycles in some regions (Colesie et al., 2023).

According to Mundim et al. (2021), 500 species of lichens are found in the Antarctic region. Antarctic lichens can colonize all types of solid substrates and exhibit minimal requirements for mineral nutrients. Among them, *Usnea antarctica* stands out due to its circumpolar distribution, being found in the Antarctic Peninsula, Subantarctic islands, and continental Antarctica (Laguna-Defior et al., 2016). Optimal conditions for lichen photosynthetic activity occur within the temperature range of 10–17°C (Sancho et al., 2019; Colesie et al., 2023). In the maritime Antarctic islands, there is a tendency for lichens to thrive due to increased temperatures and humidity. However, on the mainland, they may experience reduced productivity attributed to higher rates of evapotranspiration and drier environmental conditions (Colesie et al., 2023).

Only two species of vascular plants are considered endemic to the Antarctic region: *Colobanthus quitensis* (Caryophyllaceae) and *Deschampsia antarctica* (Poaceae). They are found on the Antarctic Peninsula and the islands of South Shetland and South Orkney. In the case of *D. antarctica*, it also occurs in the South Sandwich Islands but does not extend its distribution to mainland Antarctica (Convey et al., 2011; Cavieres et al., 2016).

These plants exhibit resilience to the extreme Antarctic climate owing to their xeromorphic anatomy. In the case of *D. antarctica*, xeromorphism is characterized by the presence of small epidermal cells, small and thick leaves, a thick cuticle, a high density of stomata, and a high number of cells per area (Cavieres et al., 2016). Both species are also tolerant to freezing and water stress and can cope with excessive photosynthetic active radiation (PAR). *D. antarctica* has a superior photosynthetic capacity in comparison to *C. quitensis*, with an optimal temperature for CO_2 uptake recorded at 13°C for the former and 19°C for the latter species. Therefore, it is hypothesized that climate warming may enhance CO_2 assimilation in these plants (Sáez et al., 2018; Bertini et al., 2021); evidence of this proposition has been documented on Signy Island in the South Orkney Islands, where both species have expanded their range, but mainly *C. quitensis*. This phenomenon has been associated with an increase in air temperature and a decrease in the impact caused by fur seals, which trample or crush vegetation, in addition to altering the nitrogen content of the soil with their urine and feces (Cannone et al., 2016, 2022).

Two nonindigenous and invasive species have established populations in the Antarctic region. *Poa annua* has been reported in the Antarctic Peninsula, South Shetland Islands, Signy Islands, Crozet Islands, and the Kerguelen archipelago (Frenot et al., 1999; Malfasi et al., 2020). *Poa pratensis* was found at Cierva Point on the Antarctic Peninsula (Pertierra et al., 2013). Both species are targeted for an eradication program (Fuentes-Lillo et al., 2017).

The subspecies *Coprosma perpusilla* ssp. *subantarctica* (formerly *Coprosma pumila*), a Rubiaceae, is found on Macquarie Island and nearby islands and is native to these islands but not endemic to Antarctica (Lloyd and Horning, 1979; Orchard, 1986).

3.2 Intertidal zone biota

Located at the intersection of terrestrial and marine environments, the littoral region is a transitional zone that sustains a diverse array of flora and fauna generally tolerant of significant environmental changes driven by the tidal cycle. However, in polar environments, the winter ice foot and growlers from summer melting introduce extra stressors to the intertidal organisms. These organisms already contend with challenges from temperature variations, UV radiation, salinity changes, and terrestrial runoff (Peck et al., 2006).

In Antarctica, the littoral zone, generally conducive to the establishment of biological communities, is primarily concentrated in the Antarctic Peninsula, as well as Antarctic and Subantarctic islands and some ice-free zones around the continent. During the summer thaw, the intertidal zones of these areas become exposed, promoting an intense biological activity. Most of these areas are generally composed of hard substrate, formed by boulders, cobbles, and pebbles, and some regions also have sandy strips under the hard substrate. Owing to the persistent impact of ice, the intertidal zone appears seemingly devoid of life, leading to its biodiversity being long underestimated. However, with the observed rise in global temperatures, already recorded mainly in the Antarctic Peninsula, there has been an increased imperative to survey the biodiversity of the Antarctic coastline. Furthermore, the potential impacts of heightened melting in these regions on the structure of biological communities have garnered attention, resulting in a surge in studies focusing on Antarctic biodiversity over the past two decades (see Griffiths and Waller, 2016).

Despite the apparent scarcity of life in the intertidal region, the biodiversity in these areas can be remarkably high compared to other cold regions on Earth. In a comprehensive study of the biota in the Antarctic and Subantarctic coastal zones (Griffiths and Waller, 2016), a total of 356 taxa were found from 30 analyzed studies. The Antarctic Peninsula exhibited greater diversity with 185 taxa compared to Southern South America and Antarctic (e.g., King George Island) or Subantarctic (e.g., Kerguelen Island) islands. Conversely, certain Antarctic islands (e.g., King George Island) showed lower richness values, potentially influenced by both natural spatial variations and inadequate sampling efforts that hinder precise estimates of local biodiversity.

Generally, the fauna inhabiting Antarctic intertidal zones is predominantly composed of small, vagile, or sessile invertebrates such as mollusks (gastropods and bivalves), arthropods (typically amphipod crustaceans), annelids (polychaetes and oligochaetes), and flatworms (turbellarians). Additionally, sessile species from taxa such as bryozoans, hydroids, and spirorbid polychaetes are also prevalent in these environments (Bick and Arlt, 2013; Waller, 2013; Liu et al., 2015; Griffiths and Waller, 2016; Chelchowski et al., 2022). To a lesser extent and more infrequently, nemerteans, pycnogonids, and even echinoderms (sea urchins and starfish) may be encountered, with the latter typically inhabiting the lower regions of the intertidal zone. Meiofauna (organisms smaller than 0.5 mm) is also present, with Nematoda and Copepoda being the most common and abundant groups (Bick and Arlt, 2013). Additionally, the presence of terrestrial mites (e.g., *Alaskozetes antarcticus*) has been recorded in the upper intertidal zone of certain sites such as King George Island (Siciński et al., 2011), along with other terrestrial arthropods such as the mite *Gamasellus racovitzai* and the springtail *Cryptopygus antarcticus* on Adelaide Island (Waller et al., 2006).

The fauna within Antarctic intertidal zones is generally cryptic, with many organisms being found beneath rocks or within crevices of the substrate (Waller, 2008), likely in response to the impacts of ice scouring and encasement. Consequently, clear zonation patterns are often absent, and richness, diversity, and abundance tend to increase within subsurface layers of the substrate (Waller, 2013), extending toward the infralittoral zone (Bick and Arlt, 2013). Moreover, the composition and dominance of invertebrate species may vary depending on substrate type (e.g., sand, gravel, cobbles, or boulders). For instance, oligochaetes and bivalves are more abundant on sandy substrates (e.g., *Lumbricillus* and *Kidderia subquadrata*), whereas gastropods such as *Nacella concinna* are predominantly found on rocky substrates (Liu et al., 2015).

Regarding primary producers, the microphytobenthos primarily comprises benthic diatoms that extensively cover substrates across much of the Antarctic coast. Macroalgae, predominant during the summer season, are represented by a smaller number of taxa in the intertidal zone compared to the infralittoral zone, with over 120 species recorded for the Antarctic marine environment (Oliveira et al., 2020). Green algae such as *Monostroma hariotti*, brown algae like *Adenocystis utricularis*, and red algae such as *Iridaea cordata* are commonly found, spanning from the upper to lower intertidal zones (Kim, 2001; Marcías et al., 2017), and can cover up to 50%–55% of available substrates (Ko et al., 2023). Although certain algae, such as the rhodophyte *Pyropia endiviifolia*, can thrive in the upper intertidal zone due to their high tolerance to desiccation and freezing (Marcías et al., 2017), macroalgae are more prevalent in the intermediate and lower zones, often occurring in tidal pools as well. *A. utricularis* can completely (100%) cover substrates in intertidal regions of Fildes Bay, King George Island, while red algae like *I. cordata*, *Phaeurus antarcticus*, or encrusting algae such as *Lithothamnium* are more commonly found in tide pools or the lower coastal zone (Aghmich et al., 2016). The notable ability of the green alga *Prasiola crispa* to colonize the supralittoral region and ice-free areas, under the effect of sea spray, and to occur near penguin colonies stands out (Dalto et al., 2010). This alga serves as a habitat for microinvertebrates such as tardigrades, nematodes, mites, and rotifers in these environments. Indeed, the presence of macroalgae in the intertidal zone directly influences the diversity and abundance of macrofauna by providing additional microhabitats, thereby supporting a rich and abundant associated fauna that varies according to the species of macroalgae present (Martín et al., 2016).

In terms of zoning, small mollusks such as *Laevillitorina caliginosa*, *Laevilacunaria antarctica*, *Altenaeum charcoti* (formerly known as *Mysella charcoti*), *K. subquadrata*, and/or amphipods (e.g., *Paramoera edouardi* and *Gondogeneia antarctica*) tend to colonize the upper and intermediate zones of the intertidal region. Meanwhile, polychaetes are more frequently found in the intermediate and lower regions, as well as echinoderms (Bick and Arlt, 2013).

The abundance and occurrence frequency of coastal Antarctic organisms exhibit temporal variability. Seasonal fluctuations have been documented, with the peak abundance of intertidal zone fauna or even in the fringe of the infralittoral occurring in late autumn, between April and June (Jazdzewski et al., 2002; Chelchowski et al., 2022). During winter, part of the benthic fauna migrates to the infralittoral, exemplified by *N. concinna*, to evade intertidal freezing conditions, persisting in the infralittoral during the severe winter months. However, *N. concinna* and sessile invertebrates can endure winter by seeking refuge in rock crevices beneath the ice (Waller et al., 2006).

Because the intertidal zone is the transition region between the terrestrial and marine environments, it represents an important link between these two environments in terms of energy and matter flow. The marine gastropod *N. concinna* serves as an important prey item for certain migratory bird species during the Antarctic summer. Specifically, the kelp gull (*Larus dominicanus*) preys upon *N. concinna* in Antarctic islands, such as the South Shetlands, as documented by Silva et al. (1999). This relationship underscores a substantial degree of trophic connectivity between the food webs of these terrestrial and marine environments. Consequently, both natural and anthropogenic environmental changes within the Antarctic terrestrial ecosystem can disrupt the community structure of the intertidal zone and vice versa.

In contrast to the thermally stable environment of the Antarctic subtidal zone, the temperature of the intertidal zone varies widely. For instance, during the 2010/2011 Antarctic summer, the temperature of the intertidal zone at King George Island, located at the northern extremity of the Antarctic Peninsula, fluctuated between −2.26°C and +21.18°C. Exposure to terrestrial climate conditions renders intertidal zone organisms particularly susceptible to the effects of climate change (Kuklinski and Balazy, 2014). Nonetheless, the precise impact of climate change on intertidal organisms remains a subject of ongoing research. Studies on the gastropod *N. concinna*, which can serve as an effective biomonitor for these changes, have enhanced our understanding of the potential warming risks to intertidal biota (refer to Suda et al., 2015). The thermal niche of *N. concinna* from South Georgia ranges from −12.9°C to +15.6°C. However, temperatures exceeding 10°C have been correlated with significant physiological changes in this gastropod, including elevated heartbeat and feeding rates (although radula rasping ceases above +12.3°C), and diminished locomotion speed. The righting ability and tenacity (adhesive force to remain on the substrate) decrease above +2°C. Additionally, metabolic shifts have been documented, such as a transition from aerobic to anaerobic metabolism, oxidative stress, upregulation of heat stress proteins, and a reduction in phospho-L-arginine levels (Suda et al., 2015).

4 Conservation areas and their ecological indicators

Conservation areas in Antarctica are delineated within the framework of the Antarctic Treaty System (ATS), a series of international agreements established to coordinate, manage, and regulate activities and international relations in the Antarctic region. This region encompasses all continental territories, islands, and marine waters situated south of the 60°S latitude. The Antarctic Treaty System (ATS) includes the 1959 Antarctic Treaty, which became effective in 1961. It also encompasses the Convention on the Conservation of Antarctic Seals (1978), the Convention on the Conservation of Antarctic Marine Living Resources (1982), and the Protocol on Environmental Protection to the Antarctic Treaty (1998). The Convention on the Regulation of Antarctic Mineral Resource Activities, signed in 1988 but not yet in force, is also part of the ATS framework (Figueiredo dos Santos and Korber Gonçalves, 2022).

Article 2 of the Protocol on Environmental Protection to the Antarctic Treaty designates Antarctica as a "natural reserve, devoted to peace and science." While this designation implies comprehensive protection for Antarctica, the protocol does not explicitly delineate the activities falling within the scope of peace and science, except for the prohibition of mineral resource-related activities as stipulated in Article 7. However, the Antarctic Treaty ensures that the continent remains free from military bases and fortifications, prohibiting military maneuvers and tests of any type of weapon, including nuclear ones, and radioactive waste disposal through Articles I and V (Coetzee et al., 2017).

Despite these protective measures, the International Union for Conservation of Nature (IUCN) does not classify Antarctica as a protected area, and there lacks a global strategy for managing the continent as a protected area. Consequently, Antarctica cannot be regarded as highly protected overall (Chown et al., 2017). Globally, Antarctica ranks in the lowest quartile in terms of the total percentage of protection compared to other countries (Shaw et al., 2014).

Protected areas in Antarctica are currently designated as Antarctic Specially Protected Areas (ASPAs) and are defined under Annex V of the Protocol on Environmental Protection to the Antarctic Treaty. These areas are intended to remain inviolate and encompass representative examples of the main ecosystems, important species assemblages, and areas of scientific research interest (SAT, 2021).

There are 79 registered ASPAs, but three of them were de-designated in 2014 (SAT, 2023). The majority of ASPAs are situated in terrestrial environments (Fig. 1.5) and are mainly located in the Antarctic Peninsula and the Ross Sea region. Terrestrial biodiversity in Antarctica is largely confined to permanently ice-free areas, constituting 0.2%–0.5% of the Antarctic continent. However, only 1.5% of this ice-free area is protected (Shaw et al., 2014; Wauchope et al., 2019).

The ASPAs together cover 44% of most Antarctic terrestrial species (including birds, plants, lichens, and invertebrates), despite the small percentage of the ice-free area reserved for this purpose, which demonstrates the relevance of ASPAs for the conservation of Antarctic biodiversity. However, this level of protection varies from group to group of organisms. While all the birds (flying birds and penguins) in Antarctica can be found in at least

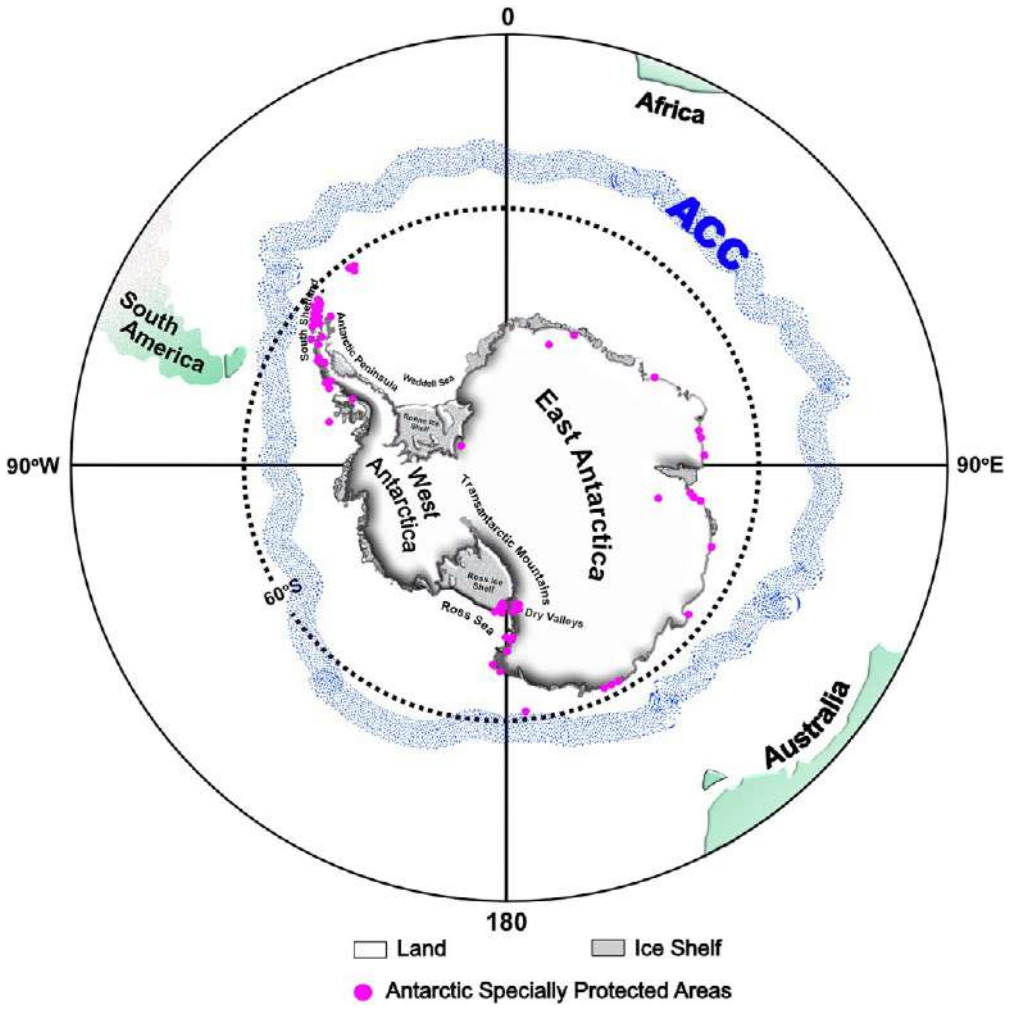

FIG. 1.5 Localization of Antarctic specially protected areas. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credits: The authors redrawn from Burrows, J.L., Lee, J.R., Wilson, K.A., 2023. Evaluating the conservation impact of Antarctica's protected areas. Conserv. Biol. 37(3), e14059. https://doi.org/10.1111/cobi.14059.)*

one ASPA, only half of the invertebrates can be found in a certain protected area. In the case of plants and lichens, the level of protection is even lower, with only a third of the species occurring in ASPAs (Wauchope et al., 2019). These differences may be related to insufficient sampling efforts regarding those species that are not easily detectable or charismatic as vertebrates (e.g., arthropods). In this context several authors argue that the current ASPA system inadequately represents Antarctic biodiversity and advocate for improvements in this area, questioning the effectiveness of ASPA management plans. Issues raised include: (a) the implementation of strict measures to reduce the transfer of propagules of invasive alien species to an ASPA may be ineffective if similar restrictions are not applied beyond its boundaries and (b) human infrastructure activities, such as the construction of new research stations or hard rock runways, do not appear to have been significantly modified, despite requirements to assess their potential impacts (Chown et al., 2017 (see S1 text); Coetzee et al., 2017; Burrows et al., 2023).

In 2015, BirdLife International and the Scientific Committee on Antarctic Research (SCAR) compiled a report listing 204 Important Bird and Biodiversity Areas (IBAs) in Antarctica. Mainland Antarctica and island groups such as South Shetland, South Orkney, and Balleny represented a significant gap in the global coverage of IBAs in terrestrial environments. This study gathered available data on Antarctic avifauna, according to IBA standard criteria, to identify and describe sites of particular importance for species conservation. About 27 IBAs were located within ASPAs designated for avifauna protection, three IBAs were within ASPAs designated for other purposes, and nine IBAs were situated within three ASMAs (Harris et al., 2015).

Moreover, there is limited representation of inland aquatic ecosystems, specifically melt streams, ponds, and

lakes, within ASPAs. These ecosystems are widespread in Antarctica and possess characteristics unique to polar regions. They are also susceptible to human impacts from Antarctic research stations (Howard-Williams et al., 2021). The Committee for Environmental Protection (CEP), an expert advisory body established by the Madrid Protocol, has approved revising the ASPA system (Burrows et al., 2023).

Other categories of protected areas in Antarctica include Antarctic Specially Managed Areas (ASMAs) and Historic Sites and Monuments (HSMs). The establishment of ASMAs aims to improve the planning and coordination of activities within these areas, facilitate coordination among the parties, avoid potential conflicts, and minimize environmental impacts. ASMAs are also required to have a management plan; however, entry permits are not mandatory, and their regulations are exhortative rather than obligatory (Hughes et al., 2021). One or more ASPAs may be contained within an ASMA, as exemplified by ASPA#128, which is situated within Admiralty Bay ASMA#1 (Fig. 1.6).

ASMA#1 is situated at Admiralty Bay of King George Island in the South Shetland Islands, at the northern tip of the Antarctic Peninsula, an area experiencing rapid warming. The island hosts scientific facilities from five countries: Brazil, Poland, Peru, Ecuador, and the United States of America. Among these, two scientific stations operate year-round: Comandante Ferraz Brazilian Antarctic Station and Arctowski Polish Antarctic Station. During the spring and summer, the region attracts a significant number of tourists interested in visiting scientific stations and observing the local fauna and flora, including penguin and seal populations that breed in the region (Fig. 1.6). Consequently, King George Island has been identified as a proxy for studying climate change and environmental impacts. Specifically, the decline in Adélie and Chinstrap penguin populations in the South Shetland Islands over 30 years, as well as the reduction in the cover of the lichen *Urticinopsis antarctica* on Signy Island in the South Orkney Islands, have been linked to warming trends (Siciński et al., 2011; Trivelpiece et al., 2011; Bokhorst et al., 2016, 2022).

5 Human activities and their impacts

Currently, Antarctica hosts 112 scientific research stations or national facilities that operate either during the summer,

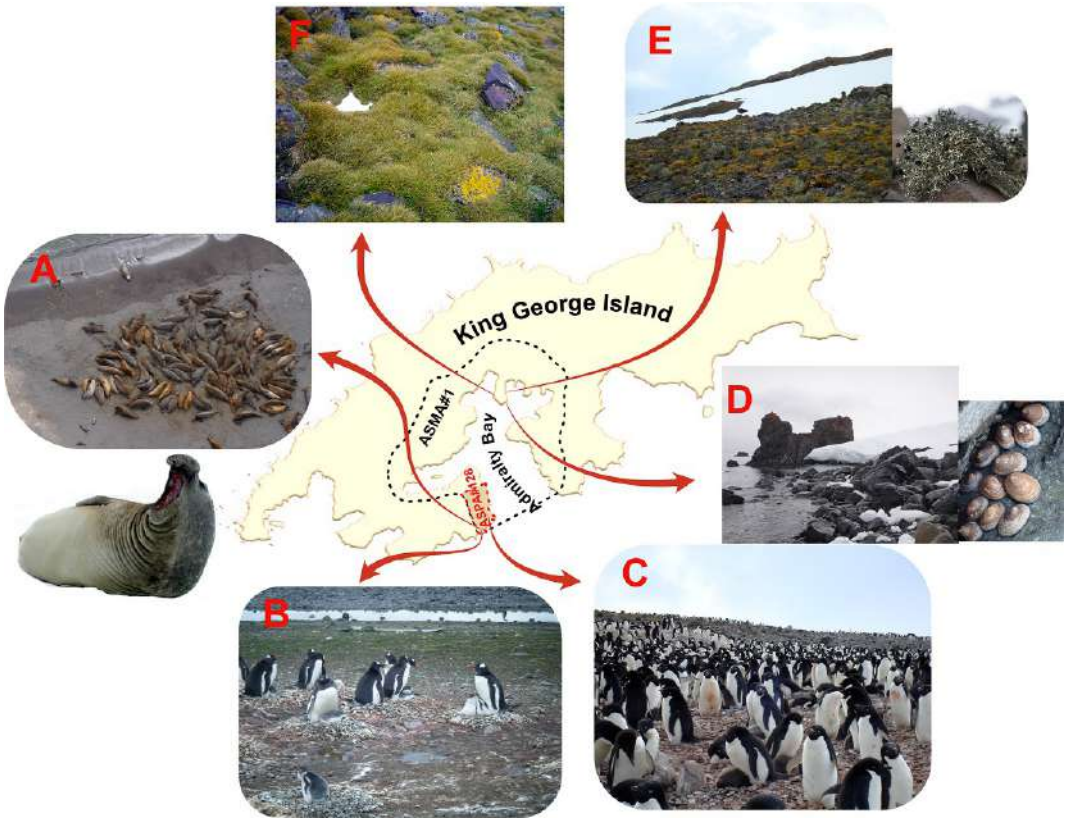

FIG. 1.6 Geographic location of ASMA#1 and ASPA#128 of King George Island. Groups of Elephant seals (*Mirounga leonina*, A), Gentoo (*Pygoscelis papua*; B), and Adélie (*Pygoscelis adeliae*, C) penguins at ASPA#128. On the rocky shores and intertidal zone of ASMA#1 of Admiralty Bay, there are several representatives of the Antarctic fauna and flora, such as the gastropod *Nacella concinna* (D), the lichen *Usnea antarctica* (E) and the grass *Deschampsia antarctica* (F). *(Credits: The authors.)*

winter, or both. Most of these facilities are situated in ice-free coastal regions. During the austral summer, approximately 5500 persons, including researchers and logistics teams, inhabit these scientific facilities. The disposal destinations for solid and liquid waste are not always well documented, which impedes a comprehensive assessment of local risks and anthropogenic impacts (Stark et al., 2023).

The Environmental Protection Protocol to the Antarctic Treaty, also known as the Madrid Protocol, recommends solid waste removal from Antarctica. Liquid waste, including sewage effluents and wastewater, may be discharged at sea. However, this disposal method requires consideration of the assimilative capacity of the local marine environment, ensuring adequate conditions for the dilution and rapid dispersion of pollutants. Since the Madrid Protocol does not clearly define the terms assimilative capacity, rapid dilution, and dispersion, the consultative parties to the Antarctic Treaty which comprise treaty member countries, rely on their respective domestic legislation to implement environmental protection measures within their operational areas in Antarctica. Studies indicate that human activities in Antarctica affect an estimated area exceeding 5.2 km^2, encompassing more than 50% of ice-free regions (Stark et al., 2016, 2023; Cordero et al., 2022).

The dispersal of sewage effluents from Antarctic research stations into the marine environment has not proven effective in preventing the exposure of organisms living near sewage discharge points to pollutants. Elevated concentrations of bacteria, heavy metals, persistent organic pollutants (POPs), fecal steroids (particularly hormones), hydrocarbons, and pharmaceuticals have been detected in marine organisms and sediments adjacent to sewage outfalls. These pollutants have been spatially distributed more than 2 km from the sewage discharge points. The extent of the ecological impact remains incompletely understood. However, both in situ and laboratory-controlled experiments with Antarctic fish and invertebrates have demonstrated that pollution can induce oxidative stress, genotoxicity, metabolic disruptions, and tissue damage (Feijó de Oliveira et al., 2015; Rodrigues Jr. et al., 2015; Stark et al., 2016, 2023).

Tourism in Antarctica has increased exponentially in recent decades. During the 2022–23 season, the region welcomed over 104,000 tourists, marking a surge of more than 40% compared to the 2019–20 season (prepandemic of COVID-19). The International Association of Antarctic Tour Operators (IAATO) projects that the 2023–24 season will see an even higher number, with estimates surpassing 118,000 tourists (Bastmeijer et al., 2023). This trend has been on the rise since the 1960s, marking the onset of modern Antarctic tourism. Several types of impacts have been recorded throughout this period, such as soil compaction, vegetation degradation, introduction of nonnative species, changes in microbial activities in the soil, disturbances in macrofauna (reduction of reproductive success, recruitment, and population abundance), sewage pollution, fuel spills from vessels, and interactions with marine mammals leading to stress and disruptions in the reproductive and feeding behaviors of birds and marine mammals, among others. Tourist visits predominantly occur during the spring and summer seasons, focusing primarily on the ice-free coastal regions of the Antarctic Peninsula where the majority of scientific stations are located (Tejedo et al., 2022).

The deposition of dust and soot, commonly referred to as black carbon, onto snow surfaces diminishes albedo, amplifies light absorption, and elevates surface temperatures, thereby accelerating snowmelt. Black carbon sources include forest fires, combustion of diesel oil, coal, and residential wood burning. These particles can be transported through the atmosphere to distant regions or even across continents. A minimal proportion of Antarctic black carbon is attributed to intercontinental aerosol transport from South America and Australia. The local burning of fossil fuels, particularly fuel oils, is the primary source of black carbon in Antarctica and has been attributed to human activity in the region, encompassing both research and tourism. Measurements taken from snow around the Amundsen-Scott, McMurdo, and Palmer Antarctic Stations have revealed that black carbon levels exceed background levels. The expansion of scientific and tourism activities in Antarctica has led to increased fossil fuel combustion in vessels, land vehicles, helicopters, aircraft, and diesel power generators. The accelerated snowmelt in areas impacted by black carbon ranges from 5 to 23 kg/m^2 during the summer, corresponding to the rapid melting of a snow layer of up to 23 mm (Cordero et al., 2022).

6 Conclusions and perspectives

Antarctic ecosystems are presently experiencing a notable thermal shift, characterized by warming trends. Since the 1960s, the Antarctic Peninsula has encountered rapid and sustained warming, surpassing any observed warming trends in terrestrial environments within the Southern Hemisphere. Should global warming reach 1.5°C by 2100, a substantial increase in the frequency of days with temperatures exceeding 0°C is expected on the Antarctic Peninsula. Projections suggest that the northern tip of the Antarctic Peninsula could experience up to 130 days per year with temperatures above 0°C. This anticipated change is projected to extend rainy seasons and diminish snowfall in lower latitudes, impacting the region's fauna and flora. Conversely, certain regions of East Antarctica have exhibited cooling trends and an increased number of colder days annually. Thus, should greenhouse gas concentrations in the atmosphere continue to rise, a multidegree increase in surface air temperature across Antarctica is anticipated by

the end of this century (Siegert et al., 2019; Carrasco et al., 2021; Turner et al., 2021).

In addition to the overall increase in Antarctic air temperatures observed in recent decades, there has been a notable uptick in the intensity of heat waves compared to the period between 1950 and 2020. In February 2020, an unprecedented heat wave resulted in temperatures soaring by +4.5°C above the regional average for the Antarctic Peninsula. The Esperanza Antarctic Station, situated at the northernmost tip of the Antarctic Peninsula, recorded a record temperature of +18°C. Furthermore, six other Antarctic stations in the same region registered temperatures exceeding +10°C. East Antarctica has also experienced the effects of increasingly severe heat waves. In January 2020, Casey Station, located in East Antarctica, similarly recorded record-breaking temperatures. The likelihood of heatwaves with temperatures exceeding +2°C above the regional average has increased tenfold compared to the period between 1950 and 1984 (González-Herrero et al., 2022).

Compared to the preindustrial period, the warming of polar regions was approximately twice the global average. During the second half of the 20th century, the average air temperature on the Antarctic Peninsula increased by 4°C, establishing it as the Antarctic region with the most pronounced warming. The loss of Antarctic ice mass has been notably more significant in West Antarctica, where over 500 glaciers transport ice from the central plateau to the ocean. The disintegration of Larsen Ice Shelves A and B in 1995 and 2020, respectively, has diminished their role as major barriers to glacier discharge in the eastern sector of the Antarctic Peninsula. There is evidence suggesting that warming may have destabilized these platforms, facilitating their breakup (Bell and Seroussi, 2020).

Approximately 87% of the marine glacier fronts on the Antarctic Peninsula have already retreated, leading to alterations in the local landscape. The expansion of these ice-free areas has been accompanied by an expansion of vegetation cover, predominantly mosses, which have exhibited a growth rate increase of 4–5 times relative to 1950 levels. The combination of warming, exacerbated by the high influx of tourists to the region, elevates the risk of establishing nonnative plant species, particularly vascular plants, which could have potential impacts on terrestrial ecosystems (Griffiths and Waller, 2016; Amesbury et al., 2017; Bokhorst et al., 2021).

The warming of the Southern Ocean surface has led to a significant reduction in the coverage of circumpolar sea ice. Over the past seven summers, the decline in sea ice extent has surpassed the previous record three times. During autumn and winter, sea ice coverage has also been exceptionally diminished. This has impacted sea-ice-dependent species, particularly Antarctic krill, which, in its larval stage, rely on the biota associated with sea ice for food. This crustacean serves as a pivotal link in the marine food web, as virtually all Antarctic vertebrates either directly or indirectly depend on krill for sustenance. A decline in krill populations, whether due to increased harvesting or reduced recruitment, will profoundly influence the dynamics of Antarctica's coastal ecosystems, particularly affecting birds and seals (Flores et al., 2012; Hughes et al., 2021; Purich and Doddridge, 2023).

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Chapter 2

Northern Great Rift Valley: Deserts and other biomes

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1 Introduction

Biologists have always been fascinated by the very rich diversity in the areas known as the Great Rift Valley (GRV), which is the largest such structure in the world stretching around 6800km from Mozambique to Syria. To understand the geological and climatic history of the GRV, we must examine its faunal and floral distributions. The southern part of the GRV has separated into a wide area, creating mountains and savannas in East Africa with rich faunal diversity, attractive today for tourist safaris. Here, we focus on the narrower Northern Rift Valley (NRV) because we find, within a very small area, fauna and flora involving several phytogeographical zones. For example, going from Jerusalem to Jericho, a distance of only 30km, one encounters Mediterranean, Irano-Truranian, Saharo-Arabian, and Sudanese-Aethiopian fauna and flora. Understanding the causes of this biodiversity requires studying the geographic distribution of fauna and flora and relating them to the geological history of this area.

Studies of the biodiversity in the areas of the NRV commenced in the 19th century with visiting scientists (Tristram, 1884; Hart, 1891). Many more studies were conducted in the 20th century on species of plants and animals in the area, including interesting attempts at reconstructing biogeographical history (e.g., Tchernov, 1968; Zohary, 1973; Shmida and Aronson, 1986; Holt et al., 2013). These studies were limited to attempts to relate species distribution to climatic and geological history. Recent advances in molecular biology allowed for a better understanding of biological diversification in relation to geography based on DNA sequence divergence (e.g., Pook et al., 2009; Carranza et al., 2002; Metallinou et al., 2012).

Here, we focus on the distribution of plants and reptiles and attempt to expand on earlier studies by using new studies (like molecular biology) to comeup with explanations for distributions and the history of expansion, especially of Sudanese-Aethipian and Saharo-Arabian elements in the NRV. Plants were chosen because they usually respond to climatic, elevational, and soil conditions and provide good phytogeographic markers. From animals, we chose reptiles because of their ubiquitous presence and that many species show very strict distribution along phyogeographic areas.

2 Geographic and geologic history

For millions of years, what is now the eastern Mediterranean area was covered by the Tethys Sea, which regressed during the late Eocene and exposed most of the land. The land of what is now the Eastern Mediterranean Region (including Sinai) was uplifted from the Tethys sea drained into the Mediterranean from 35 to 5 MYA (Bandel and Shinaq, 2003; Segev et al., 2017). This continuous landmass was critical for the evolution of animals and plants making the NRV more contiguous with what is now Persia/Iran and Africa. The formation of the GRV began in the south during the Oligocene and gradually spread north during the Miocene, with the bulk of what we now know as the GRV opening up around 20 MYA. It started with the splitting of the African plate into two plates when uplift due to tectonic plate movement resulted in the gradual uplift of mountains and opening up of the NRV (Marcus and Slager, 1985). That geological history is why the valley system, which stretches nearly 7000km from Syria to Mozambique, includes in its Southern part secondary rifts that created lake systems and wide savannas that are of critical importance to animal and human evolution (Fig. 2.1).

The GRF is narrower and peculiar in its shape, stretching from the Gulf of Aqaba through Wadi Arabah, the Dead Sea, and the Jordan Valley. The Red Sea area (including the Gulf of Aqaba) opened up as a result of a "paar," a depression caused by the movement and migration of Arabian and African tectonic plates. This depression was not connected to the Gulf of Eden or flooded until the Pliocene (Swartz and Arden, 1960). Furthermore, the last ice

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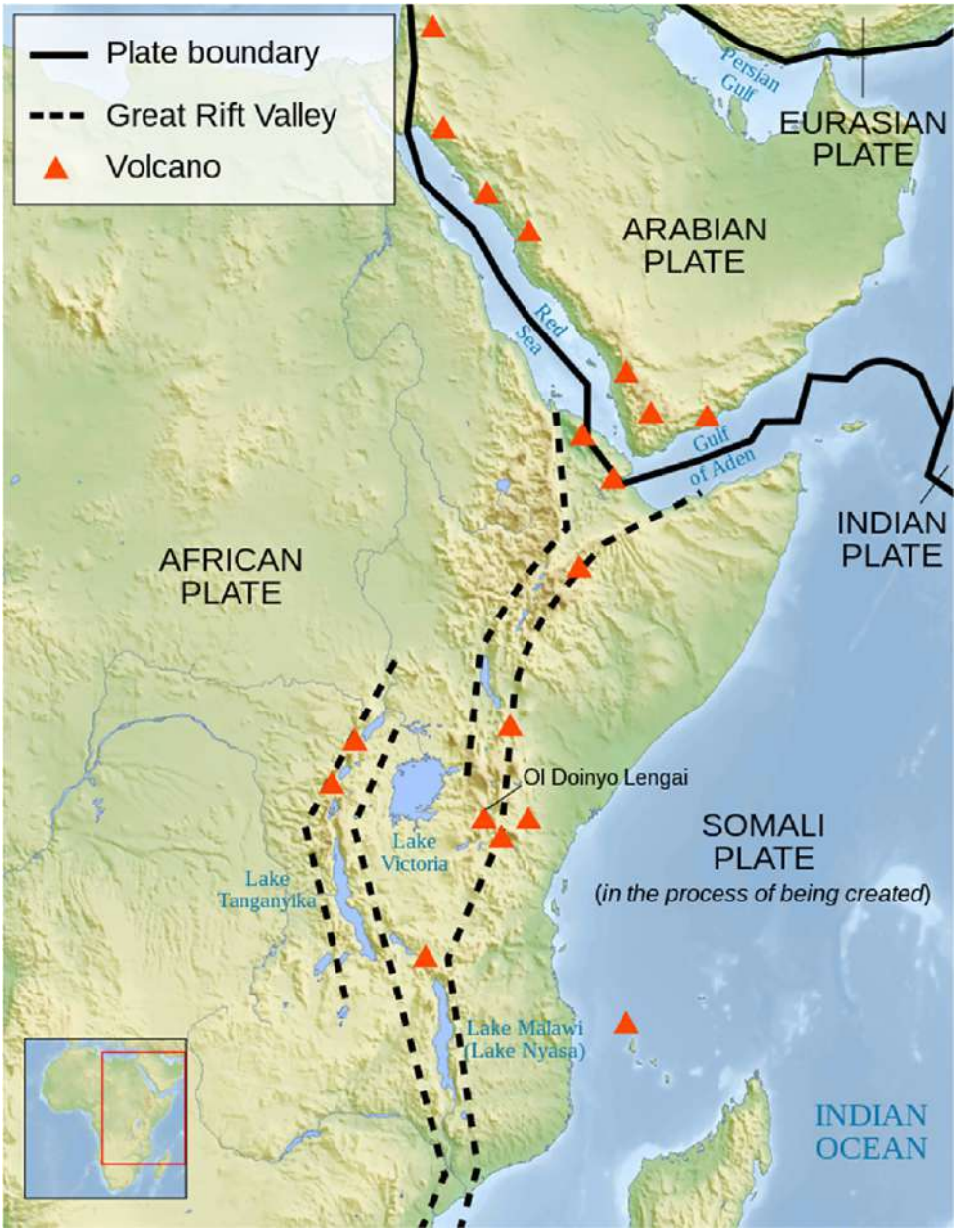

FIG. 2.1 The Great Rift Valley showing Eurasian, Arabian, African, and Somali tectonic plates. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Wikemedia Commons.)*

age ensured that few temperate animals and plants were found in the northern parts of Palestine and Jordan. Qumsiyeh (1985) conducted an analysis of the distribution of mammals (including bats) in the Arab world and showed associations not related to geographic proximity, especially around Sinai, Palestine, and Jordan, which can only be attributed to geologic history.

In very recent times, desertification has been discussed (Edgell, 2006; Groucutt and Petraglia, 2012). Yet, it is important to understand what the concept of desert is and its relation to biodiversity and the unique adaptation of certain Sudanese-Aethiopian fauna and flora to "desert conditions." Most authors understand desert conditions in relation to aridity. For example, Grove (1977) classified deserts into hyper-arid (rainfall <25 mm), arid (25–200), and semiarid (200–500). Others have considered latitude and evaporative potential in forming arid conditions (Köppen, 1931; Belda et al., 2014). But clearly arid regions are expanding (desertification), especially in the era of climate change and this has ramifications for many areas including humans and biodiversity in our region (Soultan et al., 2019; Bayram and Öztürk, 2021).

There is significant evidence that the Arabian Desert, what now stretches to the NRV, had a moist subtropical climate during the Pleistocene, including an assemblage of what is now present in tropical and subtropical regions of Africa and Africa (Groucutt and Petraglia, 2012). The initial desertification due to climatic changes is estimated to have started no earlier than 8000 years ago (Bate, 1937; Avnimelech, 1937). Typical arid or semiarid land snails, for example, are noted from this period in Palestine and Jordan in the hills around the NRV but the desertification removed most semitropical elements of the large vertebrates leaving only small ones like *Procavia capensis* (see Picard, 1937).

3 Notes on Fauna and Flora

Vegetation zones along the NRV include mountains, steppe, desert, and wetlands/reeds. But each of those can have flora and fauna elements that are related to Sudanian, Irano-Turanian, or Saharo-Arabian elements. For example, if we take the Sudanian flora, we find species widespread in Wadi Araba, around the Dead Sea, and in the Jordan valley. Sometimes the penetration of Sudanian elements is surprising. For example, *Moringa peregrina* tree (Fig. 2.2) is found not only in Wadi Araba, Dead Sea, and Jordan valley but has also been recorded reaching the Yarmouk River valley system (Al-Himma Al-Ordoniyya area). Shmida and Aronson (1986) summarized two hypotheses presented by earlier authors about the origin of these elements of our flora: (1) relicts from the Miocene era (Tchernov, 1968; Zohary, 1973) and (2) recent Pleistocene migration (Tristram, 1884; Hart, 1891; Bodenheimer, 1935). Due to the lack of a high degree of endemism, adaptation to aridity, and vicariance, they supported the second hypothesis. However, this issue remained unresolved, and new molecular data suggest that the earlier spread hypothesis (Miocene) may be valid. The Red Sea rifting occurred geologically around 20–22 MYA (Zachos et al., 2001) and significant climatic changes began to happen over 8 MYA. These two dates explain the Saharo-Arabian distribution of some, but not all, Sudanese animals and plants.

FIG. 2.2 *Moringa peregrina. (Credit: PIBS.)*

The split of the area that is now the Red Sea that began around 24 million years ago had a profound impact on speciation. Molecular data/clocks show this split into African and Arabian populations around 20 million years ago, for example, in the genera *Echis* (Pook et al., 2009) and *Stenodactylus* (Metallinou et al., 2012). In some reptile genera like *Uromastyx*, the split seemed to have happened later around 11–15 MYA (Amer and Kumazawa, 2005). The data on molecular relationships of reptile species are clearly concordant with geological shifts creating vicariant populations, allowing speciation to proceed rapidly afterward (Metallinou et al., 2012). But one could also cite early Miocene divergence in other species like scorpions (Cain et al., 2021a, b). Soultan et al. (2020) based on vertebrate distributions believed that those animals previously described as Saharo-Arabian (Holt et al., 2013) should not be grouped together and that there is indeed an Arabian and Saharan distinct fauna. The molecular data cited above confirm that the Miocene vicariant split by the formation of the Red Sear does indeed lend credence to this designation. Indeed, there is evidence of the absence of post-Miocene land bridges over the Red Sea area, with the North Sinai land bridge being available only after the glacial periods of 40,000 YA (Fernandes et al., 2006).

In terms of historical species presence, since the Pleistocene in the upper Jordan Valley, there has been a remarkable persistence of species of Sudanese origin (Biton et al., 2019). Examples of reptiles include the ornate mastigure, *Uromastyx ornata*, the Palestine Mole Viper, *Atractaspis engaddensis*, Mueller's two-headed snake, *Micrelaps muelleri*, and species of the genus *Tropiocolotes* (Fig. 2.3). Mammals of African origin that spread through the Middle East are exemplified by the Rock Hyrax, *P. capensis*, the extinct Cheetah, *Acinonyx jubatus*, the Egyptian Fruit Bat, *Rousettus aegyptiacus*, and the Hyrax *P. capensis* (Fig. 2.4). The distribution of *Paraechinus aethiopicus* (Ethiopian hedgehog) along the NRV area is another interesting example (Qumsiyeh, 1996).

Another driver of evolution in our region is paleoclimatic shifts. Based on molecular phylogenetic data, Tamar et al. (2016) concluded that diversification of the genus *Pseudotrapelus* likely started in Western Arabia during the mid to late Miocene around 8.1 MYA and that species spread into other areas of what is now the Arab world, including the northern part of the GRV. This was driven by the climatic changes (expansion of arid areas) that affected the region starting mid-Miocene (Flower and Kennett, 1994; Griffin, 2002; Edgell, 2006;

FIG. 2.3 Interesting reptiles of the NRV: (A) the Palestine Mole Viper, *Atractaspis engaddensis*. (B) Mueller's two-headed snake, *Micrelaps muelleri*. (C). *Tropiocolotes nattereri*. (D) Ornate mastigure, *Uromastyx ornata*. *(Credit: Prof. Zuhair Amr.)*

FIG. 2.4 Interesting Ethiopian-Sudanese mammals of the NRV: (A) Hyrax *Procavia capensis*. (B) Fruit Bat *Rousettus aegyptiacus*. *(Credit: Authors.)*

Steinthorsdottir et al., 2021). This was another critical period for creating the shape of the fauna and flora of the NRV.

The NRV hosts a number of endemic species related to the recent expansion of arid regions including land snails (i.e., *Buliminus diminutus*, *Buliminus therinus*, and *Euchondrus pseudovularis*) (Amr et al., 2018) and scorpions (*Buthacus arava*, *Buthacus amitaii*, and *Buthacus levyi*) that were recently described from Al Naqab desert (Cain et al., 2021a, b). Finally, we note that studies in Fainan (South Jordan) using bird bones found in Neolithic community deposits showed that the area was far more biologically diverse due to the presence of water bodies and woodlands that subsequently disappeared (Mithen et al., 2022). There is evidence of increased habitat changes affecting the distribution of flora in our region, concomitant

with human development since the beginning of the Netufian agricultural period (Soto-Berelov et al., 2015). Only a few remnant freshwater wetlands remain in the NRV like Ein Gedi, Ain Al-Fashkha, Wadi Al-Mujib, and the marshlands in Mallaha. These changes were dramatic and are linked to human activities (Schaffer and Levin, 2014) and have accelerated in the past 200 years.

4 Human activities and their impacts

The rich biodiversity related to geology and paleoclimatic events now contends with major global threats (i.e., climate change, habitat destruction, pollution, invasive species, and overexploitation) as well as local threats like colonization, wars, and sociopolitical instability (Qumsiyeh and Abusarhan, 2021). Climate change, for example, has been modeled to have negative but variable regional effects on endemic species (Soultan et al., 2019). Desert-adapted species are also vulnerable to climate change (Vale and Brito, 2015; Kechnebbou et al., 2021). There need to be tailored responses (action plans, activities, targets) to the regional, national, and local community situations which also link directly to global/international strategies, agendas, and policies adopted by international conventions. Here, we need to address some specific threats to the NRV not discussed enough in the literature and then propose how we may address the challenges.

5 Dead Sea and Jordan Valley area

Before 1967, Israel diverted and used most of the water resources of the Jordan River basin for irrigation farming through the so-called "Israel national water carrier/canal" (Elmusa, 1998). From 1250 million cubic meters (mcm) per year in the 1940s, the river's flow declined to 20 mcm (Chen and Weisbrod, 2016). Israel uses 68.5% of the Jordan River basin and the rest is divided among Jordan, Syria, and Lebanon and none is used by Palestinians in the West Bank (Isaac and Hilal, 2011). Palestinians used 140 pumping units along the Jordan River before 1967 and all were destroyed or confiscated by the occupying authorities. The US-sponsored Johnston Plan 1950s was adopted to address shared water resources, but Israel refused to accept this arrangement (see Alatout, 2011; https://history.state.gov/historicaldocuments/frus1961-63v17/d8). According to the plan, Palestinians were to share 250 million cubic meters annually. According to Johnson's plan, the mountain aquifer in the West Bank provides approximately 850 million cubic meters annually, while the coastal reservoir in the Gaza Strip provides 150 million cubic meters, which indicates that there was an abundance of water that was more than the population's needs. But since then more resources also become available. Significant depletion of the Jordan River basin water was occurred because of megaprojects like the draining of the Hula Wetlands and the diversion of significant amounts of water from upstream (Lake Tiberias) (Messerschmid and Selby, 2015; Phillips et al., 2005). The Dead Sea is a unique natural phenomenon being the lowest point on the Earth. The economic, historical, sociocultural, and touristic importance of the basin region of this inland salt lake cannot be overstated. The area has essentially a tropical climate (but with little rain) making it a winter destination for the people of our region for thousands of years (e.g., the Roman Edomite vassal King Herod had a winter palace in Jericho area near the basin and a summer palace in Tequa area—Herodion). The muddy shores with high concentrations of salts and rare elements have therapeutic uses for both locals and internationals. It is also fed by the Jordan River and springs like Ain Al-Fashkha (a protected area). The part of the Dead Sea that is supposed to be part of the State of Palestine is under Israeli occupation (the whole state is under occupation). Contrary to the Fourth Geneva Convention, its resources are being utilized by the State of Israel (Al-Haq, 2012).

This diversion of the water via the Israeli National Water Carrier from Lake Tiberias, and the draw that Israel makes further downstream, has resulted in a significant decline in the level of water in the Dead Sea (Salameh and El-Naser, 2008; Figs. 2.5 and 2.6). The current rate of decline in the level of the Dead Sea is estimated at about 80–100 cm annually. As a result, the water level has declined from −397 (in 1968) to about −434 in 2018 over the past 50 years. This has also resulted in a decline in the surface area from 950 to 637 km^2. In the event that the situation continues as it is, the level of the Dead Sea surface is expected to drop by about 46 m by 2070, and is accompanied by a decline in the surface area of the Dead Sea.

The first military order given by the state of Israel at the conclusion of the so-called Six-Day War was military order number 92 on June 7, 1967, regarding the issue of water in the occupied territories, giving exclusive authority over all water sources and even controlled rainwater harvesting. This was followed by order 158 on November 19, 1967, ensuring control of "installations of water infrastructure" and on December 19, 1968, Israeli authorities declared all previous water dispute settlements invalid and all water resources in WB to be state-owned (military order 291). On March 23, 1971, military order 418 centralized decision-making under a High Planning Council and in the 1980s, the council declared that the upper limit of Palestinian consumption of West Bank water resources shall be limited to 125 mcm of the Mountain Aquifer (DIHLC, 2013). Water was stolen and Palestinians were made to buy more than 50% their own water from Israel (Rouyer, 2000). In parallel, Israeli authorities and settlers devastated the system of springs that existed in the WB before 1967 (Braverman, 2019). Israel destroyed all the

FIG. 2.5 The current "shrunk" dead sea. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Google Earth.)*

pumping stations on the Jordan River. It prevented the Palestinians in the West Bank from using the Jordan River while simultaneously drawing 600 million cubic meters annually from Lake Tiberias at the head of the Jordan River. The Mekorot Water Company Ltd. (Mekorot) was founded in 1937 as a joint Zionist venture by the Jewish Agency for Israel, the Jewish National Fund, and others. In 1949, Mekorot became a government company. According to Al-Haq (2022) "in 1982 the West Bank water infrastructure controlled by the Israeli Occupying Forces (IOF) was sold to Mekorot for the symbolic amount of 1 NIS. Since then, Mekorot has been the largest single water supplier for Palestinians and has taken over the entire water supply in the West Bank, in violation of the Palestinian inalienable rights of self-determination and permanent sovereignty over their natural resources."

The Oslo Agreement was supposed to end the occupation, but instead it simply entrenched it, further strengthening Israeli control over the natural resources including water. International treaties and laws pertaining to water have been ignored in favor of "might makes right" approach. Regardless of political outcomes, there is simply a very small geographic territory (historic Palestine) with one hydrological system (Elmusa, 1998; Daibes and Daibes-Murad, 2003). A democratic state ensuring distribution of water to its citizens based on international guidelines is actually the most logical.

6 There is also pillage of resources of the Dead Sea

The Israeli settlements in the northern part of the Dead Sea explicitly and actively utilize the natural resources of the area (Al-Haq, 2012). For example, within two decades of its launch in 1988, the annual global sales of Ahava, an Israeli cosmetics company that uses minerals from the Dead Sea, reached almost $150 million. The company has been the target of boycott campaigns. Meanwhile, Palestinians

FIG. 2.6 Overlook area over the Dead Sea, showing shrunk areas that now have sinkholes, whose impact on biodiversity is still not well understood. *(Credit: PIBS.)*

are prevented from using the Dead Sea area for tourism (Abu-Baker and Farah, 2020; Al-Haq, 2012). According to ARIJ (2016) "Access to the Dead Sea is completely sealed off for Palestinians as far as economic activities are concerned." For the Palestinian economy, this represents a loss proportional to the potential economic value from the exploitation of these resources. Accordingly, if Palestinians had free access to their share of the Dead Sea and were allowed to invest in and develop their mining industries, their production value could range between $917.70 million and $2366.40 million, or equivalent to 7.2%–18.6% of 2014 Palestinian GDP. The average potential production would equal $1642.05 million, or 12.9% of GDP. Access to Dead Sea salts and minerals will also allow Palestinians to invest in the cosmetic industry with the potential to generate even more revenue.

From the Knesset Dead Sea Concession law 1962 "[the Israeli government gave to Dead Sea Works Company] the exclusive right to extract by way of evaporation (solar or artificial), cooling, mining, quarrying, or in any other way, mineral salts and minerals and chemicals, whether in solution or in solid form, which are in and beneath the Dead Sea… the exclusive right to make, expand, modify, maintain and demolish, in and beneath the Dead Sea…including—but without derogating from the generality of the aforesaid—embankments, evaporation ponds, culverts, water barriers, pumping stations, canals, pipes, electricity lines and electricity cables, roads, anti-flood installations, wells and bores, and other installations." (Hasson, 2022).

The Dead Sea Works basically transformed the southern half of the original salt lake into an evaporation pond. However, since it is now completely dry, to extract the minerals water is pumped from the northern half (which is legally Jordanian and Palestinian territory) to the southern half by the Dead Sea Works (the Israeli company). As a result, the northern half is shrinking very fast (due to the diversion of Jordan River and pumping by Israel). In 2021, a huge new pumping station called P-9 started operating, pumping 450 mcm/year from north to south (Hasson, 2022).

Jericho is a town that has been continuously inhabited since the dawn of the agricultural era, about 11,500 years ago. As one of the oldest (if not the oldest) town/village, it has amazing features including the archeological site of Tel Al-Sultan. There were two refugee camps in Jericho, stemming from the Nakba (Ain As Sultan and Aqbat Jaber) but most residents fled to Jordan in 1967. Wadi Al-Qelt brought water from three springs (Fara, Fawwar, Qelt) for thousands of years (Fig. 2.7). Together with local springs like Tel-AlSutan, Ain Al-Auja and others, Jericho had ample agriculture including in the old days sugar cane. There is a notable sugarcane press and other archeological sites (Tela Abu Al-Alayiq, Herod's Winter Palace, the Sycamore tree mentioned in the Bible in association with Jesus, the Baptismal site, etc.). There is also Deir Hijleh Monastery (one of the oldest historically 455 AD by St Gerasimus) and Deir Qarantel (Monastery on the so-called Mount of Temptation). Qumran caves are also the caves where the Essenes kept bible scrolls that were initially housed in the Palestine Museum (aka Rockefeller Museum) in Jerusalem but were illegally taken to the Israel Museum in West Jerusalem.

According to B'Tselem, 2011 "East of Route 90, along the Jordan River, are 64 sections of land—16,973 dunams, constituting 1.1 percent of the Jordan Valley and northern Dead Sea—in which Israel buried hundreds of thousands of mines, most of them antipersonnel. Even following the Israeli-Jordanian Peace Treaty, in 1994, Israel did not remove the minefields. On the East Bank, the Jordanians cleared some 100,000 mines, enabling the land to be used once again as farmland…In 1995, Israel signed the

FIG. 2.7 Ein Al-Qelt. *(Credit: January 2023 by PIBS.)*

Convention on Prohibitions or Restrictions on Certain Conventional Weapons, and in 2000, joined Protocol II of the Convention, which restricts the use of mines that are liable to be excessively injurious. In November 2006, Israel reported to the Convention's committee that the army had removed 40 minefields against tanks in the northern Jordan Valley, at a cost of five million shekels.60 Israel has not signed the Mine Ban Treaty, which took effect in 1999. The treaty completely bans the use of mines and requires their removal." However, we also note that Israeli occupation forces set up both "nature reserves" and, most of the time, overlapping military zones in the Jordan Valley (Fig. 2.8). All of this impacts both the environment and the human well-being in the area.

The Red Sea-Dead Sea Canal project is a prime environmental problem and should not be implemented as it could damage the environment in the whole region (see Salem, 2009; Rosenberg, 2011; Georges et al., 2021; Qumsiyeh and Amr, data available/unpublished). Its impact in SP will be most acutely felt in the unnatural "replenishment" of the Dead Sea, leaving the Jordan Valley essentially dry and with continued environmental deterioration. Some work has been done on this, but much more research needs to be done and a summary of these works is beyond the scope of this report.

7 Conservation

The area is notable for its rare/endangered birds (both resident and migratory) such as the bearded vulture *Gypaetus barbatus*, Dead Sea sparrow *Passer moabiticus* (LC), and the lesser kestrel *Falco naumanni* (LC). Several species have a conservation status, including *Aquila heliaca* (VU), *Circus macrourus* (NT), *Neophron percnopterus* (EN), *Streptopelia turtur* (VU), *Falco concolor* (VU), and *Lanius senator* (NT). In addition, 24 key (notable) species of mammals belonging to 11 families are known to occur in this PA. This includes eight species of bats, three felines, two canids, and three bovids. Several of these species have conservation status, including *Gazella dorcas* (VU), *Gazella gazella* (NT), and *Hyaena hyaena* (NT). The local status of the Arabian leopard, *Panthera pardus*, requires confirmation. Furthermore, the area is known for its rare and endangered plants (Al-Sheikh and Qumsiyeh, 2022).

In collaboration with IUCN ROWA (International Union for Conservation of Nature and Natural Resources Regional Office for West Asia), Palestine (OIBS & EQA with support from the Ministry of Agriculture and Ministry of Local Economy) worked on a new Protected Area Network (PAN) for the West Bank. The work on the new PAN was completed in December 2022. However, neither the PAN nor the new NBSAP has been published yet. The PAN designates protected areas according to IUCN criteria. The protected landmass increased from 9% to 11% and the

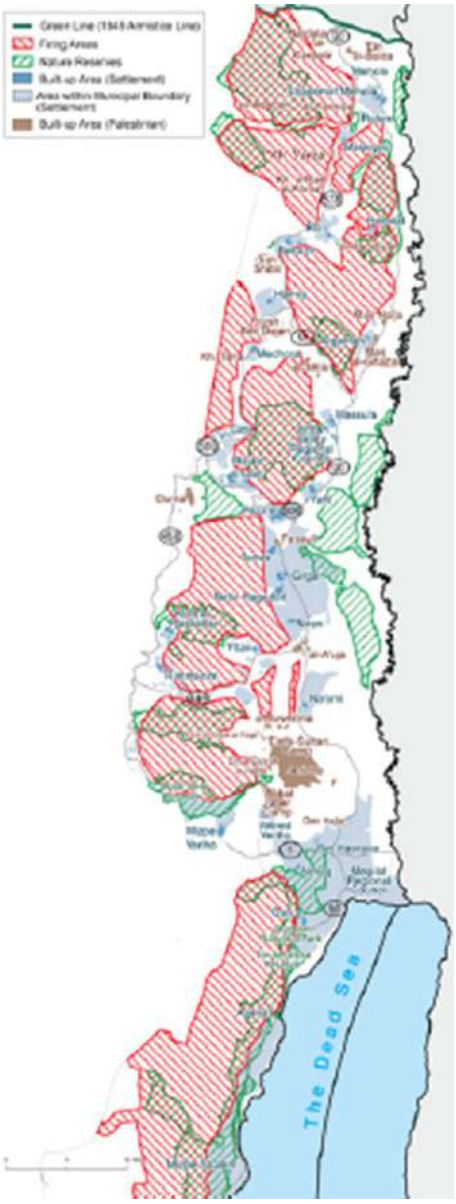

FIG. 2.8 Firing range and Nature reserves. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: B'Tselem, 2011. Dispossession and Exploitation Israel's Policy in the Jordan Valley and Northern Dead Sea. https://www.btselem.org/sites/default/files/sites/default/files2/201105_dispossession_and_exploitation_eng.pdf.)*

27 protected areas (PAs) represent all Palestinian vegetation types and phytogeographical zones. The PAN protects key biodiversity hotspots and ecoregions, but so far, only five of the PAs have management plans, and intensive surveys of the remaining areas are necessary to ensure effective protection (Qumsiyeh and Al-Sheikh, 2023). The Dead Sea Protected Area (for outlines of the PA see Fig. 2.9) is the largest PA in the West Bank and designated based on IUCN criteria as category Ia Strict Nature

FIG. 2.9 Redrawn protected area network portions around the Dead Sea. Old areas in black, new in Red. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: PIBS.)*

Reserve. That means, it is a "strictly protected area set aside to protect biodiversity and also possibly geological/geomorphological features, where human visitation, use and impacts are strictly controlled and limited to ensure protection of the conservation values" (Dudley, 2008).

Plant biomes and altitudinal effects are also conservation priorities in the NRV (Aynekulu et al., 2012). That this area includes the lowest point on Earth and that it has a mixture of fauna and flora related to African, Asian, and Saharan elements make it of particular conservation status. We recommend the following action for management:

(1) Significant new data to aid conservation efforts need to be collected on the state of the Palestinian environment (UNEP, 2020).

(2) Establish a Geopark and combine it with the protected areas on both sides of the Rift Valley to create the "Northern Rift Valley Global Protected area." The Dead Sea PA encompasses the lowest point on Earth and has interesting geological and geomorphological features. According to Gordon (2019), "the integration of geo-conservation across the full range of IUCN protected area management categories (…) should help to enhance their management and to deliver more effective nature conservation, with consequent benefits for geo-heritage, biodiversity, and people." The lowest point on Earth can be classified as a cultural ecosystem service, as it provides a *sense of place* (for a definition see Ryfield et al., 2019). Therefore, an analysis of the feasibility to incorporate a geopark in the Dead Sea PA will be an integral part of the developing recommendations for an ecosystem services-based management plan.

(3) This area should be also listed on IUCN's list of Red Ecosystems (Keith et al., 2015) because of its unique fauna and flora and the many endemic forms occurring around the lowest point on land globally.

(4) Transboundary and regional issues should be taken into consideration in strategic actions (Mason et al., 2020; Perrings and Halkos, 2012). In fact, there could be transboundary protected areas that increase cooperation (Sandwith et al., 2001). There is also increased desertification and a risk of fires (Turco et al., 2017). The main threat to the effective management of the new PAN is the lack of sovereignty, followed by the need for capacity building and a better Palestinian management structure.

(5) Tackling the threats facing the valley including Israeli projects that damage the environment. The best remedy for the shrinking Dead Sea and its sink holes (Arkin and Gilat, 2000; Ezersky et al., 2013; Salem, 2020) is to restore the flow of the Jordan River. Furthermore, we must address the power imbalance (Qumsiyeh and Albaradeiya, 2022(. While the state of Palestine needs to develop better plans for managing the resources, a big impediment to implementing many existing plans (e.g., for water resources) or complying with signed International treaties is that the State essentially has no control over most of its natural resources (due to the Israeli occupation).

(6) The most important aspect of success in conserving the area is to ensure local participation and empowerment (Rowe, 2006; Orenstein and Groner, 2015; Al-Assaf et al., 2022).

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Chapter 3

Ecosystem services in the Atacama region, Chile

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1 Introduction

1.1 Ecosystem services

Ecosystem services (ESs) are aspects of ecosystems that are used, either actively or passively, to maintain human well-being (Fisher et al., 2009). ESs are usually divided into three categories: provision, regulating, and cultural services. Provisioning services are products used directly by people, such as firewood, food, water for human consumption, resources for developing livestock, medicinal plants, and fodder. Regulating services make indirect contributions to human well-being through on their ecological functions, such as climate regulation, pollination, and erosion control. Cultural services provide nonmaterial benefits to people from ecosystems, such as tourism and recreation, research possibilities, spiritual development, cultural identity, social cohesion, and religious ties.

The study of ESs has become an important area of research during the last decade and also motivated political interest in the conservation and management of natural and social systems (Posner et al., 2016; Bidegain et al., 2019). There have been several initiatives to understand, operationalize, and classify ESs, such as the Common International Classification of Ecosystem Services (CICES), the Economics of Ecosystems and Biodiversity (TEEB), and the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES). This latest initiative aims to strengthen the collaboration between science and politics in the field of ESs to support the management of territories, the well-being of people, and the sustainable conservation of nature (Pascual et al., 2022; Díaz et al., 2018; Cerda and Tironi, 2017).

The relevance of studying the ESs provided by ecosystems of arid and semiarid zones is currently recognized as extremely relevant (Smith-Ramírez et al., 2023). Although research on ESs and their evaluation has considerably increased worldwide, the analysis of such services generated in semiarid and arid zones has not received the same attention as other ecosystems, such as temperate, humid, and coastal zones (Costanza et al., 2017). Approximately 20% of the world population inhabit arid and semiarid areas, which represent about 47% of the world's land surface (Forage, 2007; Gaur and Squires, 2018; Wickens, 1998), many of them rely on the flow of ESs and contributions in these territories for their livelihoods, cultural traditions, and collective relationships. Besides, water resources and biodiversity have significantly deteriorated in these areas worldwide, consequently affecting human well-being (García-Llorente et al., 2015). In these zones, ESs such as water, food supply, medicine, and raw materials are relevant for the well-being of both local inhabitants and surrounding communities (Bidak et al., 2015). In particular, water availability plays a key role in sustaining vegetation, which, for example, in many deserts, represents the basis of ancestral forms of life (O'Farrell and Anderson, 2010; Reyers et al., 2009; Safriel et al., 2005; Tapia, 2023), and its structure and dynamics control the production of ESs (Reyers et al., 2009). The valuation of ESs in arid and semiarid zones contributes to understanding historical society-ecosystem connections and discovering means, contextualizing research and thus allowing for better policy-making processes. In fact, the assessment of the diverse values associated with ESs is strongly emphasized (Pascual et al., 2022).

Semiarid and arid regions are also prone to significant impacts of climate change, which decreases, for example, water availability, and disturbs the entire natural and social systems. This is particularly important for the semiarid and arid regions near the Atacama Desert (Alvarez-Garreton et al., 2023), which, under the more optimistic scenario of climate change, predicts a decrease of about 10% for this region (Alvarez-Garreton et al., 2023), enhancing the

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probability of conflicts between the different stakeholders that make use of the limited amount of water. Therefore, it is urgently necessary to understand the ESs that sustain ecosystems and the benefits that sustain livelihoods. Nevertheless, in Latin American countries like Chile, these areas have not received necessary attention in scientific studies or research on ecosystem services (De la Barrera et al., 2015).

1.2 The region of Atacama in Chile and ecosystem services

The Atacama region in Chile is located between 26° and 29° 20′ south latitude. It comprises three provinces and nine communes, headed by the city of Copiapó (regional capital). It has an area of 75,176.20 km^2, equivalent to 9.94% of the Chilean territory, and borders the Antofagasta region to the north and the Coquimbo region to the south. It constitutes a complex arid socioecological system in Chile, where the relationships between indigenous and other communities with the natural systems have created sociocultural and ecological conditions to deliver a diverse flow of ESs. The high level of conflicting uses of ESs such as water availability for mining, agriculture, and conservation of biodiversity, among others, has consolidated a complex cultural region (Vargas, 2014; Cerda et al., 2018) including the construction of sometimes conflictive values regarding ESs (Madaleno and Gurovich, 2007). The governance of water resources characterized by a water private market and legislation that prioritizes private interests, causes inequity in water accessibility, affecting the construction of values toward ESs (Prieto, 2017; Babidge, 2016). Although analysis and proposal for new water governance mechanisms have been conducted (e.g., Sánchez-Galvis et al., 2018), and biophysical regional models have been developed, which expect substantial decreases in precipitation and extreme temperature events in the near future (Fuentes et al., 2021), from the perspective of the ESs approach, the ecological conditions that determine human well-being and the social importance that people attribute to the services have not been sufficiently explored. Aiming to confront this gap, we present a case study in the Huasco River Valley in the Atacama Region, whose goal was visualizing ESs that are relevant to different local actors that live in the valley, and the areas that provide such services from the perspective of the participants. In addition, we also provide some examples of participants' narratives regarding ESs, which allows to identify value domains associated with ESs.

2 Methodology

2.1 Study area: The Huasco river valley

The Huasco River Valley is a socioecological system located in the Province of Huasco in the Atacama region, where diverse human actors, including indigenous communities among them, concur. These populations interact with and inhabit these ecosystems defining their long-standing experiences of the world, with a great intergenerational-accumulated knowledge over generations. The watershed covers an area of 9.850 km^2 (MOP, 2009) and includes the regions of Atacama and Coquimbo. The relationships between people and the natural system, together with the effects of climate change, have created the sociocultural and ecological conditions to deliver a flow of ES in the area. The watershed harbors different components such as water bodies, 980 species of vegetation, many of them are part of the fragile xerophytic vegetation formations, more than 65 species of fauna, and other ecological attributes that provide ESs to the local population, e.g., water for human consumption, food derived from agriculture (e.g., olives, citrus fruits, and table grapes), water for mining, possibilities for tourism, food for cattle, extraction of medicinal plants, extraction of shrubs for coal and wood, among others (Letelier-Gálvez et al., 2008; MOP, 2009). In the basin, the phenomenon of the flowery desert also manifests and motivates national and international scientific interests in conservation (Cerda et al., 2018). The watershed includes the communes of Huasco (10.262 inhabitants) and Alto del Carmen (6.305 inhabitants), as well as parts of Freirina (6.915 total inhabitants at communal level) and Vallenar (53.087 inhabitants at the communal level) (INE, 2017). Except for Vallenar, the watershed is characterized by significant rural population. These places are certainly important scenarios for agroindustry, mining, and power generation within the country (Vargas, 2014; MOP, 2009). The local population of the watershed engages in different activities such as industrial and subsistence agriculture, mining, commerce, services, fishing, and tourism. The relation of local communities with ESs has changed over time. In the case of indigenous communities, their economic activities reflect indigenous base readapted with Hispanic elements (breeding activities, use of medicinal plants, hunting the *guanaco*,[a] pottery and textiles), with activities of European origin that became traditional in the region (production of dried fruits, cultivation of grapes, production of wine and syrup, and goat herding) (Otárola and Muñoz, 2017), which gives the territory a productive and, at the same time, a cultural conservation character. For example, in the highest part of the basin, the indigenous population still maintains ancestral practices that align with natural systems (Otárola and Muñoz, 2017). In the *diaguita* cultural context, some communities have defended their territory from development projects.[b]

Actors operate at different territorial scales and are interested in various benefits generated by ES (Vargas, 2014). Some actors directly depend on ESs, such as

a. *Lama guanicoe*.

b. https://www.cidh.oas.org/annualrep/2009sp/chile415-07.sp.htm#_ftn1.

(a) small farmers (organized in local agricultural communities and small farmers' associations), (b) medium and large farmers, (c) indigenous communities, and (d) territorial and civil organizations that are also present in the area with particular interest in different ecosystem services. More powerful actors, whose management significantly affects or may affect the flow of ESs, are also present. Among these actors, we find: (a) sectoral development agencies, (b) regulatory bodies, (c) regional, provincial, and communal governments, and (d) enterprises. The zone is also ecologically fragile (e.g., desertification processes), which motivates the interest of scientists working on issues related to biodiversity conservation, whose interests often contrast with those of local stakeholders (Bidegain et al., 2019; Cerda and Bidegain, 2019; Romero et al., 2012; Valladares, 2012). Fig. 3.1 shows the location of the study area.

2.2 Definition of local actors

To identify ESs that flow in the valley, we identified key local actors to be interviewed. We defined key local actors as those who have a great interest in different ecosystem services that flow in the territory comprised of the Huasco River Basin, due to their dependence on these services (Bidegain et al., 2019). For our study, key local actors include farmers, leaders of institutions, members of indigenous communities, members of local government, scientists, and companies, among others.

2.3 Design and application of the interview

We conducted semistructured interviews with 58 local actors (Table 3.1), who were consulted individually to identify relevant ESs and areas relevant for the manifestation of these services (Newing et al., 2011; Biggs et al., 2022). The individual format of the interview allowed people to freely identify different services of interest. To ask about the areas relevant for the manifestation of services, we used maps of the river basin, which were designed and used in such a way as to provide local actors with the opportunity to recognize areas perceived as relevant for individual and collective use of ecosystem services that they mentioned. Participants engaged with printed maps on paper. Each participant received the same map, which was representative of the entire basin. This allowed

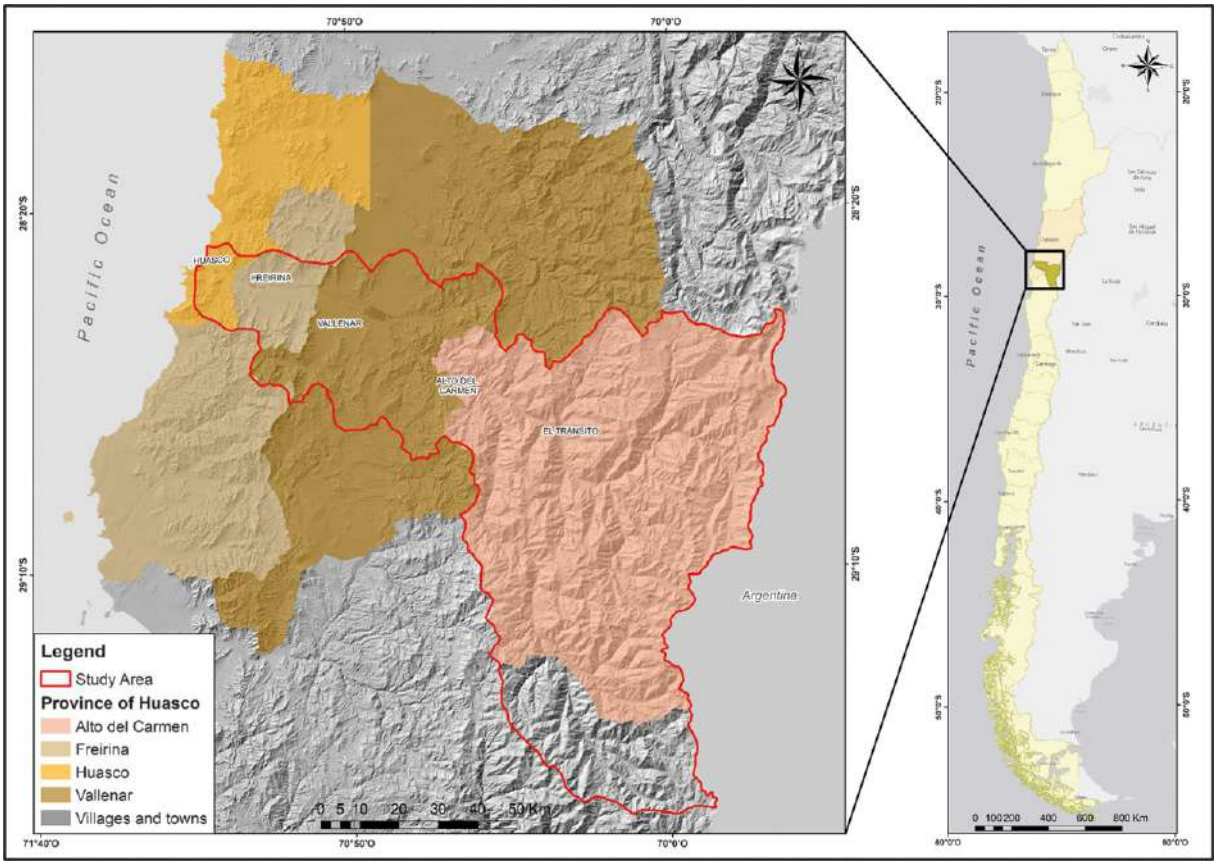

FIG. 3.1 Location of the Huasco river valley in the Atacama region, Chile. Own elaboration. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Alyzon Valdés-Rojas.)*

TABLE 3.1 Local actors interviewed and the frequency.

Local actor group	Frequency
Farmer (beekeeper, olive, grower, winegrower)	18
Artisan	6
Rancher/breeder	2
Small scale mining worker	1
Professional (government entity an others)	22
Tourism workers	9

participants to reflect on their links with the entire territory under analysis, and consequently, obtain a territorial approach to the analysis of the results. The maps showed the current uses and coverage of land, bodies of water, roads, and towns, among others that were considered relevant. They were prepared at an appropriate scale that facilitated the exercise of the participants (1:250,000). Square cells, where each cell represents a real surface, were used in the map to facilitate the work of the participants.

This phase was not focused on any ecosystem service, so that they emerged spontaneously from respondents. The interview, in addition to being designed to identify relevant ESs and relevant areas that provide them, contained questions to investigate reasons why these areas and services matter, and thus we obtained an approximation of the diversity of values related to ESs.

The sampling was not aimed at obtaining many interviews, but rather at obtaining sociocultural and location diversity of the actors in the basin (Bidegain et al., 2019). A total of 58 interviews with key actors were conducted between January and April 2023. These actors are distributed throughout the territory included in the basin. Participants were contacted by a sociologist and anthropologist living in the area, who were previously trained for this.

2.4 Data analysis

The information obtained through maps was digitized and processed in a geographic analysis platform, which allows working in raster and vector layers. The interviews/mapping were digitized independently, assigning to each grid a particular identifier (ID) in the attribute table, thus avoiding duplication. For each grid, relevant areas providing the ESs identified by the interviewees were obtained. Through a binary system of presence/absence, a value was assigned to each grid, 0 for the absence of ES and 1 for the presence of ES. Once these values were assigned in the table of attributes of the vector entity, the different types of ecosystem services identified by the interviewees were incorporated and differentiated, strengthening the database. The identification of relevant areas was done in a general way and did not specify the type of key actors.

Once the binary system of the grids was created, the ESs present in each grid defined in the Huasco basin were summed, thus allowing the establishment of ranges of service provision and the elaboration of a traffic light map, identifying the areas of the basin that provide more or less ecosystem services, according to the perception of the interviewees, with the following ranges and classification (HIGH = Red; MEDIUM = Yellow; LOW = Green). Four maps were obtained (Fig. 3.2), one showing the total ESs present in the basin and the others differentiating between provision ESs, regulation ESs, and cultural ESs. In addition, there is coverage in vector format with all the information.

Concerning the valuation of ecosystem services, we codify 28 interviews with ATLAS.ti (2023) (24.1 version) to understand prioritizing ESs that respondents have about the surrounding environment. To do it, we first used the framework developed by Arias-Arévalo et al. (2017, 2018) for the plural valuation of ESs. In this sense, we use a triple perspective of intrinsic, instrumental, and relational values to see the different valuations used by respondents. By intrinsic value, we refer to the value of ecosystems as ends in themselves and are represented as moral duties (Arias-Arévalo et al., 2018). By instrumental value, we refer to the value of ecosystems as merely means to an end and are often measured in monetary terms (Arias-Arévalo et al., 2018). Finally, relational values are defined as those concerned with the relationships and responsibilities between people or between nature and people (Arias-Arévalo et al., 2018; Chan et al., 2016). Second, we also codify those ecosystem services that suffered transformations over time as perceived by respondents, and if the transformations were positive or negative. Finally, a codification was done according to the ESs that produced some type of conflict between different stakeholders. The sector that transforms the ESs or generates conflicts was of special interest, always according to the perception of the respondent. In this sense, both the transformation of

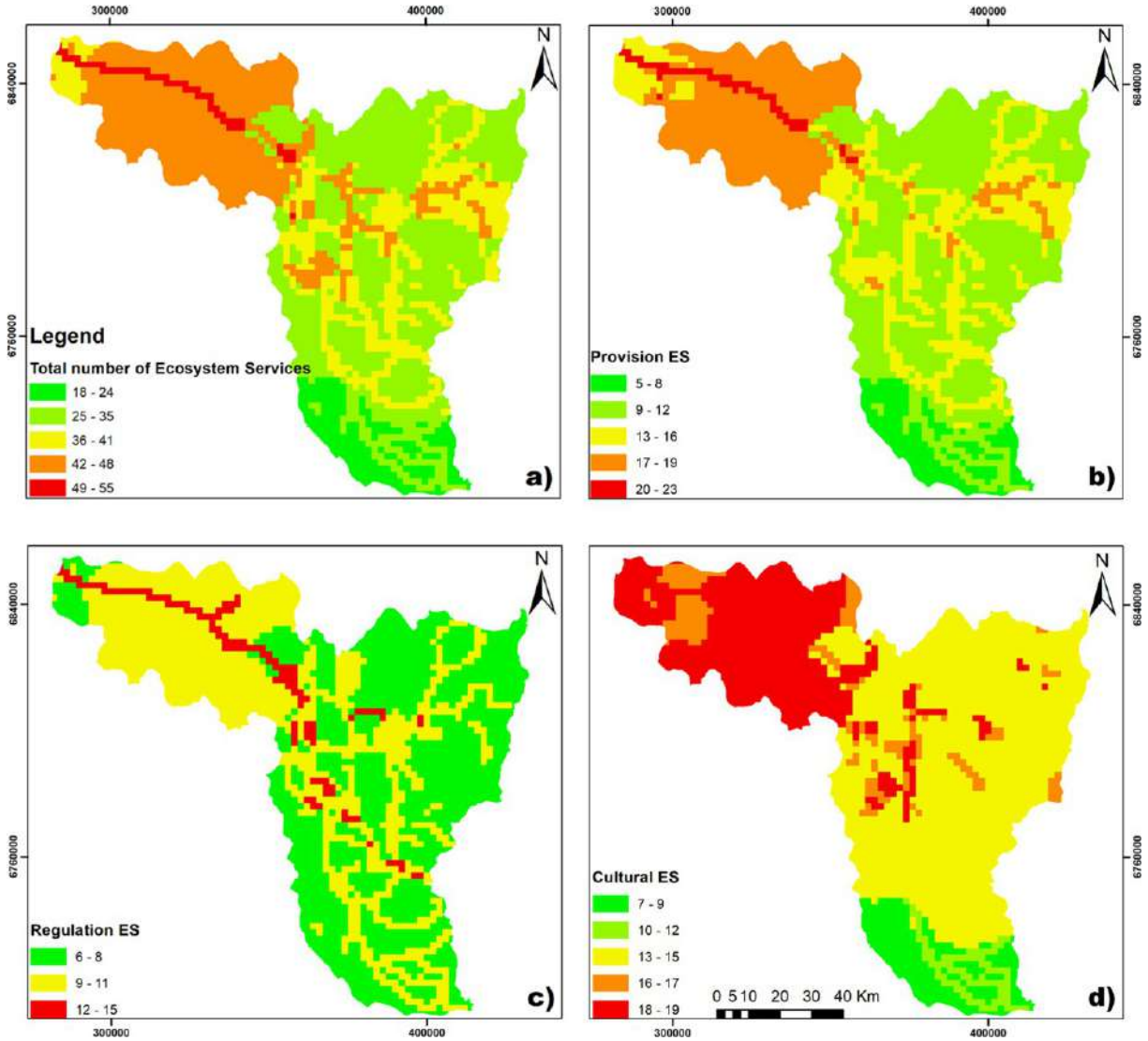

FIG. 3.2 Ecosystem service identified by local stakeholders. (A) Total amount of ecosystem services present in the watershed, (B) provision ecosystem services, (C) regulation ecosystem services, and (D) cultural ecosystem services. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Alyzon Valdés-Rojas.)*

ESs and the ESs in conflict were subcodified by two properties: resource (water, ecological attribute, territory, agriculture production, climate, livestock activity) and stakeholder (mining sector, industrial agriculture, real estate sector, energy production, fishing, poultry, aggregate extraction, and tourism).

3 Results

Table 3.2 shows the ecosystem services identified by local actors. A wide variety of ecosystem services is highlighted. Provision ecosystem services are mentioned most frequently by participants.

Based on the interviews conducted with various key actors (Table 3.1), the ESs present in the territory are identified and spatially represented in cartographies categorized by SEs (Fig. 3.3). Among those who were interviewed, the majority identified themselves as inhabitants of the commune of Alto del Carmen, which is the municipality with the largest surface area in the basin and also has localities that are difficult to access, due to being located in the heart of the Andes mountain range, which leads to the development of activities such as livestock farming (transhumance), agriculture (cultivation of grapevines, citrus fruits and subsistence or family agriculture), and small mining, thereby expanding the spectrum of interviewees.

As mentioned, we also provide some examples of participants' narratives regarding ESs that are important to them, which allows identifying value domains from the plural valuation perspective associated with ESs.

TABLE 3.2 Ecosystem services identified by local actors.

Provision ecosystem services	Regulation ecosystem services	Cultural ecosystem services
Foods derived from industrial livestock farming	Erosion protection	Ethnic identity
River water as a source of electric energy	Protection against wind	Research development possibilities
	Bioremediation by plants	Nature sounds for relaxation
Wood fuel	Regulation and water retention by fog	Symbolic flora
Use of plants for cosmetics	Maintenance of invertebrate populations and habitats (including protection of the gene pool)	Conservation of threatened biodiversity
Water for mining	Regulation of air quality	Sports tourism in nature
Horse fodder	Maintenance of fish and amphibian populations and habitats (including protection of the gene pool)	Nature-based religious festivities
Materials derived from animals (skins, leather, wool, horns, other)	Maintenance of flora populations and habitats (including protection of the gene pool)	Diaguita ethnic identity
Water for industrial agriculture	Filtration/sequestration/storage/ accumulation of nutrients by microorganisms, algae, plants, and animals	Wetland and estuary conservation
Goat fodder	Microclimate regulation	Conservation of plant species and plant endemisms
Alcoholic beverages derived from industrial agriculture	Water quality (minerals, taste, smell)	Nature tourism
Use of medicinal plants	Microclimate regulation for agriculture	Spiritual and religious value
Extraction of clay for the manufacture of ceramics, adobe, and construction	Maintenance of fauna (non-bird) populations and habitats (Including protection of the gene pool)	Biodiversity conservation
Foods derived from artisan fishing	Maintenance of bird populations and habitats (Including protection of the gene pool)	Scenic beauty that allows tourism
Minerals extraction	Glacier water cycle regulation	Scenic beauty
Foods derived from industrial agriculture	Regulation and retention of water (lagoons, waterholes)	Archaeological sites
Food derived from small-scale fish farming	River channel maintenance	Social cohesion linked to a territory
Water for livestock feeding		Leisure, relaxation, and enjoyment of nature
Availability of underground water		Sense of belonging and cultural identity
Alcoholic beverages derived from traditional agriculture		
Small-scale livestock farming (extensive grazing)		
Water for agriculture		
Preparation of processed foods from crops		
Water for domestic agriculture		
Foods derived from small-scale/traditional livestock farming		
Drinking water		
Foods derived from traditional agriculture		

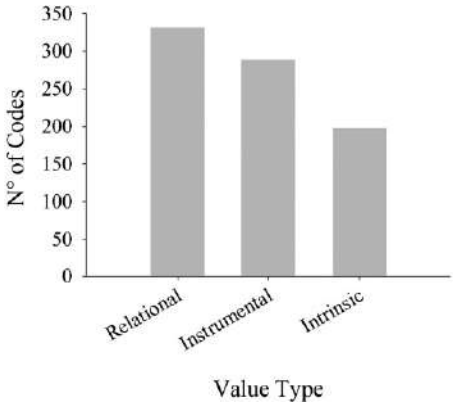

FIG. 3.3 Number of codes per valuation type concerning respondent's perceptions about ESs of the Huasco watershed. *(Credit: Matías Guerrero-Gatica.)*

Table 3.3 shows examples of narratives that were analyzed and classified into typologies of values associated with ESs. One important result is the number of codifications concerning the plural valuation of ESs (Fig. 3.3). Here, we find an important number of relational values (331 codes) that refer to the relational values, followed by instrumental values (289 codes) and intrinsic values (198 codes). In this sense, the valuation of water appeared as the most relevant one, as it was the most frequently mentioned resource. Concerning stakeholders, the mining industry was the most mentioned activity by respondents.

4 Discussion

The application of the ecosystem services approach has proven useful in a diverse set of decision contexts, in a wide range of biophysical, social, and governance systems (Posner et al., 2016; Ruckelshaus et al., 2015). However, in semiarid and arid regions, this approach has been less explored and applied, which can aggravate the degradation of ecosystems and their links with human well-being, impeding better decision-making processes in a clear and comprehensive way. In this chapter, we visualize the Atacama region as a provider of relevant ecosystem services to local communities through an example of the Huasco river valley that constitutes a complex socioecological system with multiple ecological and social conditions to deliver a flow of ecosystem services.

Our methodological approach allowed us to identify which ecosystem services are relevant to local actors, which can be useful for designing management scenarios that consider local realities (Costanza et al., 2017). The results show a wide variety of relevant ecosystem services that flow at different scales. For example, industrial agricultural benefits stand out, as well as local agriculture and other local activities, such as the benefits from local cattle, are significant. A variety of cultural ecosystem services was also found. In this group, people mentioned cultural benefits associated with biodiversity conservation, tourism, and benefits related to identity and social cohesion that are favored by the existence of the river basin and its biological attributes. This reflects the multiple values that emerge to express the importance that local actors attribute to the socioecological system in the Atacama region, highlighting intrinsic, instrumental, and relational values of ecosystem services. Integrating and visualizing these values make it possible to understand the worldviews and dynamics between ecosystems and human communities (Valle, 2020; Pascual et al., 2017; Tapia, 2023).

Concerning the valuation of ESs, we highlight the relevance that water provision has for the three types of values. It is not surprising that, in the context of a semiarid and arid environment, people value more the relevance of water availability. What is more interesting is the awareness of ecosystem processes interlinked in the entire watershed, not only in instrumental terms but also in relational terms. Water provision, as a culturally relevant ES, must be considered when planning ES management due to the relevance that water has for the local community. More alarming is the prevalence of the industrial mining companies that jeopardize this ES, which is now developing prospects for a big new project at the head of the watershed. The structuring valuation that the ESs of water have evidences a series of local knowledge concerning the changes over time regarding this resource. Some people recognize changes in water quality and quantity because of the impact of a big mining project, Pascua Lama, which has been actually closed by the state for violating environmental legislation. While we do not differentiate between water and glaciers in the analysis, several times, the valuation of water has been established through concerns about glacier melting.

The relevance of our analysis is highlighted given that it makes visible the relevance of ecosystems present in the Atacama region of northern Chile as providers of multiple ecosystem services to local communities. In Chile, the management of ecosystem services is characterized by a hierarchical system, concentrating the important decisions by a few elites (local or national) and disabling the capacity of local communities to participate, even though they still deeply depend on the ESs for their livelihoods. While some researchers argue that the management of ESs is better achieved when local groups are involved in the decision-taking processes, the political dynamic goes in the contrary (Chhatre and Agrawal, 2008; Bennett and Dearden, 2014; Fernández-Llamazares et al., 2016; Waylen et al., 2010). Thus, there is a need to include local actors in the management of their territories through interdisciplinary perspectives that embrace social values around biological attributes. This will allow strengthen the development of public policies that harmonize with the contexts of local and indigenous communities and allow real participation in the management of their territories.

TABLE 3.3 Examples of valuations made by respondents concerning the relevance of ESs that are present in the Huasco watershed.

Intrinsic values	From mountain to sea, you have different microclimate types. That is something really important. That also helps to generate what is the vegetation, wetlands, everything that makes different types of microclimates. So, that is the reason we have different types of productions, inside the different towns [+Instrumental].
	Even though we are in the most arid desert in the world, we have a valley, we have a life system and we have the opportunity to preserve life through our watershed, through those of us who are living here.
	[one of the most important things of the valley is] the vegetation itself. The native vegetation that born from what nature is delivering.
	From the mountain, there, you can see how life is born, how life flows, how life endures.
	[native animals are important] because they are part of our ecosystems, they are with us, we protect them, we teach people that come here to not kill them, that they have to protect them, to take care of them, they are also part of our lives [+Relational].
	High Andean lagoons are something that the man has never done. They only did the gates and they have been practically since the beginning of times and they are something that favors us and, as I said before, we are, in one way or another, blessed for what we have.
	In my vision, an important animal is the bee. Guanaco is also relevant, both for its preservation and well, it was also relevant for feeding. The puma, the fox, and the condor, are relevant animals too. Pollinators of all kinds too. The shrimps are also important [+Instrumental].
Instrumental values	The particular things of the products we have here. The organoleptic characteristics are much better than in other places. What do I mean? The flavors are better, the smells are more intense, they are less transformed than in other places, despite I have the center region to stock up on products, but is not the same.
	In productive terms, the goats are important for the high Andean places for domestic animals, the transhumance.
	I use the water mainly for food. Hygiene is another important thing. Playing. Relaxation. Those are my most important uses for water.
	The hills of color from Pinte are oriented to La Totora and it gives us a benefit to the quality of Waters from the Huasco River, because they are mineralized in the mountain by the natural conditions. However, they arrive at this side and they are mixed with carbonates, and is like they are entirely filtered and very exquisite water is generated.
	The mountain is the sustenance of all the things that our province do
	The springs are relevant because natural ecosystems fed on them, wildlife fed on them, domestic animals fed on them, and people fed on them [+Intrinsic].
	[In the high mountains] I go to buy products, I go to see friends, I go providing advice, I go to relax, to eat exquisite.
	The glaciers are the source of drinking water we have, the reservoir of drinking water.
Relational values	There I, again, come back to what is the mountain, I come back to what is life, in this concrete case, in the sense of making use of it. In this case, the animals, that is another relevant thing. Because I insist, this is a chain and if the chain is broken for giving relevance to one of them, it no longer has the relevance it had.
	Of course, there is also a you can say biological, cultural or even an archaeological richness. But I feel that, from the indigenous point of view, the connection is there, with the river, the water, and livelihoods.
	Everything related to the river and water is also related with life. If there is no water there is no life, and I feel that the local community has united to defend it.
	[Riversides are important] Because they maintain nature. They give life to fauna and the livestock.
	Just seeing the water, the people come there, they swim there and there are more birds, of all kind of animals, and the valley has more life. Without water we are nothing.
	I believe that, when we talk about the watershed and the relevance of the river or water that runs there, the relevance is from mountain to sea, of course. I mean, what happens here up in the mountains influences totally downstream and not only downstream.
	[The mountain is relevant] Because it is the place where one goes with your animal or family, you go with tourists. The important thing is to have keeping in a good state the places, keeping clean. That is why I worry about.

In brackets [] are additional values in conjunction with the main value expressed.

It is necessary to continue advancing knowledge regarding the perception and assessment of ecosystem services in arid and semiarid regions. Studies of the benefits of ecosystem services in arid and semiarid regions in Chile are notably lacking. The management of the ecosystems of Atacama presents a significant challenge that requires further study and synthesis of social knowledge regarding the value of such ecosystems.

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Chapter 4

Loss of biodiversity and effect of climate change on the vascular flora of the Chihuahuan Desert of Mexico

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1 Introduction

Natural capital is the set of ecosystems, the organisms that live in them, and the natural processes that are the basis of social well-being on which anthropocentric activities are developed (Sarukhán et al., 2017). The sustainable use of a country's biodiversity offers great potential for its development and the generation of benefits for the entire population, so there must be a balance between the exploitation of natural resources and the maintenance of biodiversity. Overexploitation of natural resources and changes in land use cause the loss of biodiversity, which implies not only the loss of species but also genetic diversity. This negatively affects the viability, fertility, and capacity of organisms to respond to environmental changes, particularly in species with isolated and reduced populations, as is the case for many endemic species, and those with restricted distributions (Bertola et al., 2022).

Conservation biology, applied sciences, and political sciences are closely related because natural biological processes and biological diversity provide ecosystem services that are essential for the survival of living beings, including humans. In this context, the Convention on Biological Diversity proposes conserving the three elements of biodiversity, namely, genetic, species, and ecosystem diversity, in addition to guaranteeing sustainable development and the distribution of benefits obtained from nature. Therefore, it suggests guidelines and policies that allow the objectives of the agreement to be achieved among member countries (Hoban et al., 2020).

2 Geographic location

The size of the Chihuahuan Desert is between 450,000 and 630,000 km^2 depending on the study (Shreve, 1942; Johnston, 1977; Medellin-Leal, 1982; Henrickson and Johnston, 2007). It includes parts of the states of Arizona, New Mexico, and Texas in the United States, while in Mexico, it is delimited by the Sierra Madre Oriental and the Sierra Madre Occidental, which includes the states of Aguascalientes, Chihuahua, Coahuila, Durango, Guanajuato, Hidalgo, Nuevo León, Querétaro, San Luis Potosí, Tamaulipas, and Zacatecas. The relief of the Chihuahuan Desert is highly variable and comprises plains with a minimum elevation of 300 m asl, and a considerable part of the Chihuahuan Desert extends between 1000 and 1500 m. In mountainous areas, elevations can reach between 1820 and 3130 m asl. Precipitation is between 200 and 300 mm annually, with abundant rainfall occurring in mountainous areas with higher elevations, which generate water bodies that are highly important for the region. The predominant climatic zones are semiarid-semiwarm (BS1hw), semiarid-temperate (BS1kw), arid-semiwarm (BS0hw), arid temperate (BS0kw), and arid-semiwarm (BWhw), with an average annual temperature of 21°C. Due to the diversity of its orography, the temperature in the Chihuahuan Desert can be extreme because on it drops to below 0°C on cold days and reaches 50°C on the warmest days (Descroix, 2004; Ruhlman et al., 2012).

According to the US Environmental Protection Agency (EPA) (https://www.epa.gov/eco-research/ecoregions-north-america) (accessed October 13, 2023), the Chihuahuan Desert is the largest desert area in North America; it is an ecoregion with an approximate area of 509,429 km^2 that is located within the ecological region of hot deserts. In this work, we follow the criteria of the EPA and delimit it to the part corresponding to the territory of Mexico, hereinafter called the Chihuahuan Desert of Mexico (CDM), with an area of 347,282 km^2 (Fig. 4.1).

Terrestrial Biomes. https://doi.org/10.1016/B978-0-443-36569-0.00004-5
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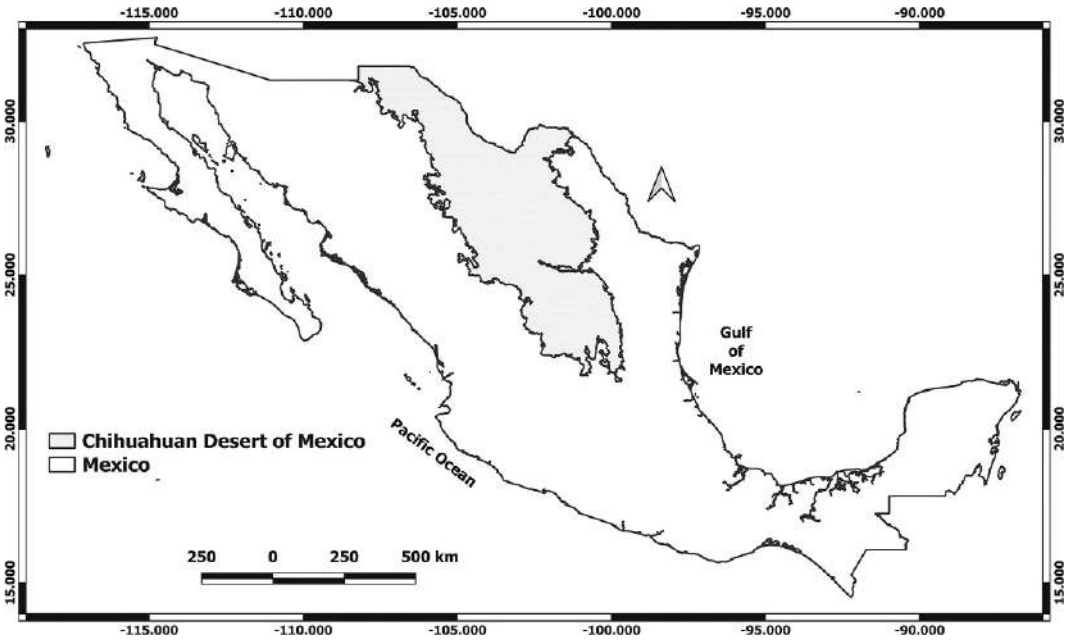

FIG. 4.1 Location of the Chihuahuan Desert of Mexico (CDM). Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: Alberto González-Zamora and Rebeca Pérez-Morales.)*

3 Biodiversity of the Chihuahuan Desert of Mexico

Due to its topography, nested microclimates are generated in the CDM, which has allowed this ecoregion to be recognized as an area of great importance for terrestrial and aquatic species (McDaniel, 2022). The vegetation is characterized by the predominance of various types of shrublands and grasslands, a small area of forests and hydrophilic vegetation in mountainous areas, as well as gypsophilous and halophytic xerophilous vegetation (Zavala-Hurtado and Jiménez, 2020).

The CDM harbors a great diversity of species from chordate groups, including 123 species of mammals such as the pronghorn *Antilocapra americana* Ord, the jaguar *Panthera onca* (Linnaeus, 1758), the collared peccary *Dicotyles tajacu* (Linnaeus, 1758), rodents such as *Dipodomys* spp., *Perognathus* spp., *Neotoma* spp., and *Peromyscus* spp.; 333 species of birds, the most common of which are the roadrunner *Geococcyx californianus* (Lesson, 1829), the thrush *Toxostoma curvirostre* (Swainson, 1827), the quail *Callipepla squamata* (Vigors, 1830), and the thrush *Icterus parisorum* (Bonaparte, 1838), in addition to numerous birds of prey such as owls and hawks including *Bubo virginianus* (Gmelin, 1788), *Buteo albonotatus* (Kaup, 1847), *Buteo swainsoni* (Bonaparte, 1838), and *Falco femoralis* (Temminck, 1822). The reptile diversity of the CDM is among the highest of all desert ecoregions and several lizards are endemic, including the geckos *Coleonyx brevis* (Stejneger, 1893) and *Coleonyx reticulatus* (Davis & Dixon, 1958), the lizards *Aspidoscelis marmoratus* (Baird & Girard, 1852), *Cophosaurus texanus* (Troschel, 1852), and *Sceloporus* spp.; the snakes *Bogertophis subocularis* (Brown, 1901), *Thamnophis eques* (Reuss, 1834), *Masticophis schotti* (Baird & Girard, 1853), and *Masticophis flagellum* (Shaw, 1802); in addition to the endemic turtles *Gopherus flavomarginatus* (Legler, 1959) and *Terrapene coahuila* (Schmidt & Owens, 1944). There are also 23 species of fish (Dinerstein et al., 2000).

In addition to the groups mentioned above, the functioning of the ecosystem depends on the high diversity of invertebrates, such as subterranean termites of the order Isoptera, which are the main consumers of dead plant material and animal dung; other invertebrates, such as shrimp, mites, and moths; and a wide diversity of bee species (https://www.inecol.mx/inecol/index.php/es/ct-menu-item-25/ct-menu-item-27/17-ciencia-hoy/1113-diversidad-faunistica-del-desierto-chihuahuense) (accessed November 21, 2023).

Scrub is the vegetation type with the greatest cover. It includes some of the most common species in the CDM such as the bushes *Larrea tridentata* (DC) Coville, *Flourensia cernua* DC, the trees *Neltuma glandulosa* (Torr.) Britton & Rose, and various species of the genus *Acacia* Mill., as well as cacti such as those of the genera *Coryphantha* (Engelm.) Lem., *Mammillaria* Haw., and *Opuntia* Mill., as well as various species of the genus *Agave* L. In contrast, grasses and shrubs are observed in grasslands, including species of the genera *Aristida* L., *Bouteloua* Lag., *Hilaria* Kunth, and *Sporobolus* R.Br. Additionally, the CDM is widely recognized for its

diversity and high levels of endemism of Cactaceae because up to 25% of the 1500 known cactus species are found there (Dinerstein et al., 2000; Hernández and Gómez-Hinostrosa, 2005).

For this work, we compiled the floristic diversity of the CDM from three main sources: by consulting the Texas Oklahoma Regional Consortium of Herbaria (TORCH) database, which contains approximately 4 million plant specimens, many of which were digitalized from more than 50 herbaria in North America (https://portal.torcherbaria.org/portal/index.php) (accessed May 14, 2023); a review of the literature for the region; and data obtained from our field trips. The search for information in TORCH included records of specimens that were within the previously defined limits of the CDM. For each record, the coordinates assigned to each location were verified to detect possible georeferencing errors. In cases where any inconsistency was detected, the coordinates were corrected by consulting the cartography of the National Institute of Informatics and Geography of Mexico (INEGI).

The literature reviewed included original descriptions; taxonomic revisions of various groups; and inventories of the flora of the CDM and the states it covers (Balleza and Villaseñor, 2002; Estrada-Castillón and Villarreal-Quintanilla, 2010; García-Arévalo, 2002; González et al., 1991, 2009; González-Elizondo et al., 2017; Henrickson and Johnston, 2007; Hernández and Gómez-Hinostrosa, 2011, 2015; Herrera et al., 2010; Valdés, 2015; Villarreal-Quintanilla, 2001; Villarreal-Quintanilla and Encina-Domínguez, 2005; Villarreal-Quintanilla et al., 2017).

The information obtained from the different sources was analyzed, taxonomically reviewed, synthesized, and combined in a database. To do this, only the unique records of each species per locality were taken, and records of specimens without data from a specific locality were eliminated from the analysis to avoid georeferencing errors. Taxa reported as weeds in the Mexican weed portal of the National Commission for the Knowledge and Use of Biodiversity (CONABIO) were removed from the database (http://www.conabio.gob.mx/malezasdemexico/2inicio/home-malezas-mexico.htm) (accessed on August 26, 2023), as well as species grown in agricultural fields, introduced into gardens in urban areas and used to feed livestock. Only taxa at the species level were considered.

The World Flora Online database (https://www.worldfloraonline.org) was used to define the nomenclature and taxonomic locations of the groups. To identify the endemic species in the CDM, the list of species of the flora of Mexico by Villaseñor (2016) and the Plants of the World Online database of the Royal Botanic Gardens of Kew (https://powo.science.kew.org) were reviewed. The classification proposals of Christenhusz et al. (2011a,b) were taken for ferns and related plants and gymnosperms, and the APG IV (The Angiosperm Phylogeny Group) (2016) was used for angiosperms to assign higher taxonomic categories.

The compiled database consists of 32,981 historical records from 1830 to 2022 (Fig. 4.2). Given the environmental conditions recorded in the CDM, the specimen collections were not homogeneously distributed, so many of the records are located close to the most important population centers and on the main roads that cross the CDM. This heterogeneity in collection activity reflects a ratio of less than 1 record/km^2 if the total number of collections is divided by the surface area of the CDM. Likewise, collecting activity has not been uniform in the almost 200 years for which records exist, with 1971–90 being the period where the greatest collecting activity was carried out within this area, after which it decreased considerably in the past 30 years (Fig. 4.3).

The total diversity of the vascular flora represented in TORCH for the CDM corresponded to 3334 species from 918 genera in 152 families, the majority of which were angiosperms, with 3188 species. Asterids constituted the most diverse clade of vascular plants in the study area, representing 40.6% of the species richness, followed by Rosids and, to a lesser extent, Gymnosperms (Table 4.1). Of these, 1087 species (34%) are endemic to Mexico, and 254 are restricted to the limits of the CDM proposed in this work.

Of the 10 families best represented in the CDM, Asteraceae contained approximately 20% of the species with 620, followed by Fabaceae and Poaceae with 265 and 255 species, respectively (Table 4.2); this family also had the largest number of genera. The case of the Cactaceae family is unique since most of the endemic species of this family have an official conservation status in the IUCN Red List (https://www.iucnredlist.org) (accessed January 11, 2024).

The representation of each species on average corresponds to 10 collection records; in the case of endemic species, the majority does not exceed 5 collection records, so they can be considered rare. Many of these groups possibly correspond to species that emerged recently in geological terms according to the age of the CDM (González-Zamora et al., 2023).

For some species in the CDM, studies have been carried out that address genetic diversity mainly for conservation purposes. In this regard, some of the works that emphasize this topic include those presented by Mandujano et al. (2020) and that of González-Zamora et al. (2023); however, greater effort is needed to achieve greater understanding, especially of species that have small populations and/or are at risk. An example of a research effort carried out to understand other aspects of biodiversity is the study by Sosa et al. (2020) on phylogenetic diversity and the effects of changes in climate.

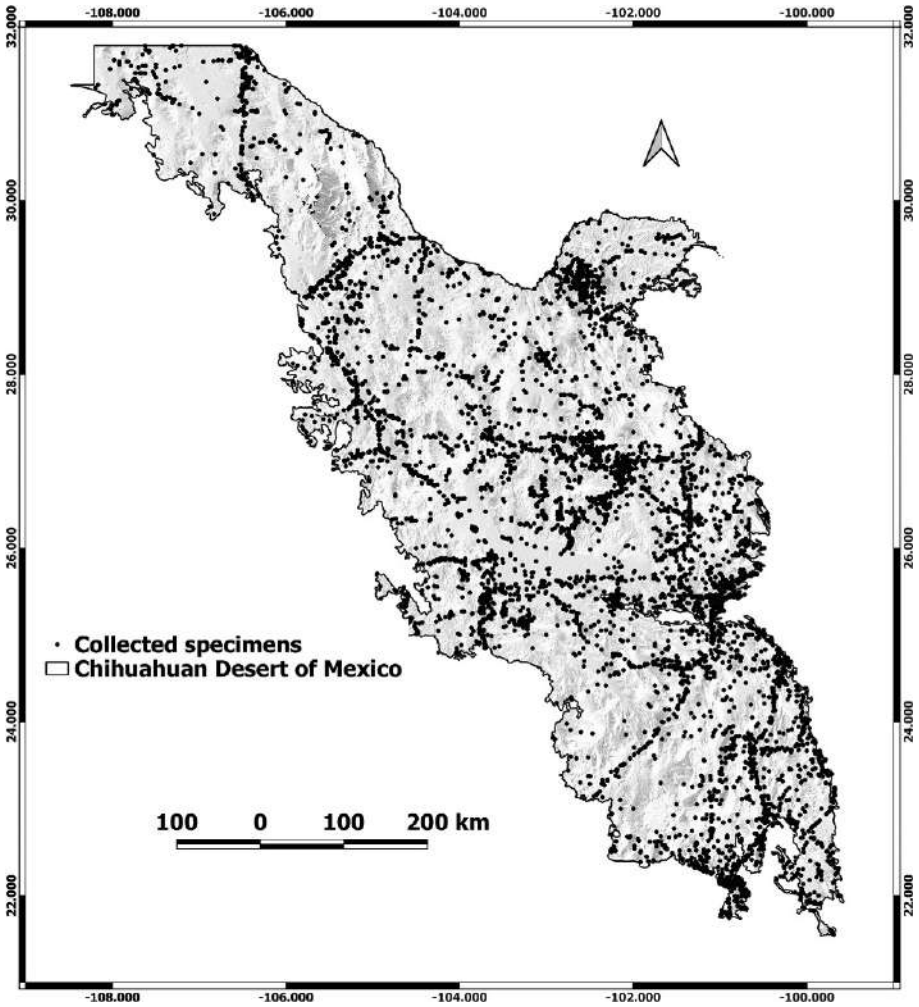

FIG. 4.2 Collection locations for the vascular flora of the CDM. The records include all records from 1830 to the year 2022 in TORCH. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: Alberto González-Zamora and Rebeca Pérez-Morales.)*

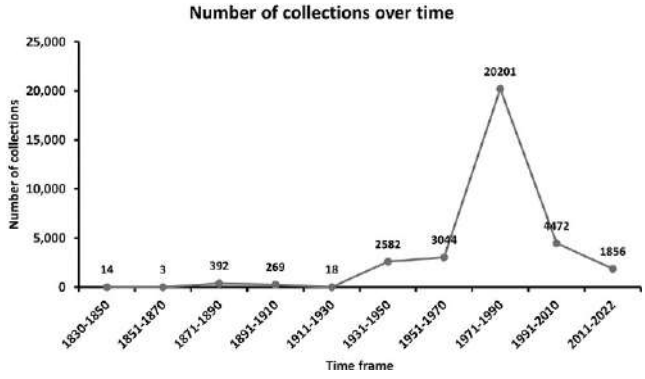

FIG. 4.3 Collections of botanical specimens over time from 1830 to 2022 in TORCH. *(Photo credits: Alberto González-Zamora and Rebeca Pérez-Morales.)*

4 Conservation areas and their ecological indicators

In the CDM, 70 biodiversity conservation areas were formally established: at the federal level, called Federal Protected Natural Areas of Mexico (FPNAM), 18 were decreed; 42 areas were decreed State, Municipal, Ejidal, Community, and Private Protected Natural Areas of Mexico (SMECPPNAM); and 10 sites were named Areas Voluntarily Designated for Conservation (AVDC) (Table 4.3 and Fig. 4.4). The state of Coahuila is where the greatest effort was made to conserve biodiversity as it encompasses half of the decreed areas, followed by Nuevo León, where 20 areas for conservation are located (Table 4.4).

TABLE 4.1 Quantitative participation of vascular plant groups in the CDM flora.

Taxonomic group	Families	Genera	Species	Species in IUCN	Species endemic to Mexico	Species endemic to CDM
Ferns	14	29	109	1	12	1
Gymnosperms	3	7	37	27	11	0
Angiosperms	135	882	3188	349	1064	253
Asterids	35	375	1355	58	486	116
Eudicots	5	15	39	3	14	5
Magnoliids	5	6	8	0	4	1
Monocots	21	132	500	36	115	22
Rosids	48	260	886	131	262	52
Superasterids	17	84	355	121	148	52
Superrosids	3	9	43	0	35	5
Nymphaeales	1	1	1	0	0	0

TABLE 4.2 The families of vascular plants in CDM, with the highest numbers of genera and species.

Family	Genera	Species	Species in IUCN	Species endemic to Mexico	Species endemic to CDM
Asteraceae	173	620	12	268	56
Fabaceae	67	261	35	89	12
Poaceae	62	255	5	36	8
Cactaceae	30	154	120	96	28
Euphorbiaceae	10	116	0	25	5
Boraginaceae	19	92	1	29	14
Lamiaceae	12	91	0	48	15
Asparagaceae	14	72	28	40	7
Brassicaceae	26	66	0	28	13
Malvaceae	21	61	1	12	6

In the total areas designated for the conservation of biodiversity in the CDM, 1974 species of vascular plants are distributed, 463 of which are endemic to the limits established in this study. The SMECPPNAM practically conserve the same number of species as the FPNAM, which shows the importance of the participation of citizens and local authorities in the conservation of biodiversity (Table 4.5). The FPNAM area with the largest number of vascular plant species is the APRN CADNR 004 Don Martín, with 1172 species, of which 233 are endemic. Among the SMECPPNAM, REc Sierra y Cañón de Jimulco has the most species, with 328, 88 of which are endemic. In the case of the AVDC, Tierra Silvestre Cañón del Diablo, with a total of 218 species and 13 endemic species, is the one that harbors the greatest biodiversity. The three above-mentioned areas are located in the state of Coahuila.

5 Human activities and their impacts

Historical accounts show that in the mid-19th century in the CDM, the native grasslands were lush and relatively free of shrubs, the riparian areas were surrounded by gallery forests, and unchanneled streams extended to form wetland systems called ciénegas. In addition to wild vegetation,

TABLE 4.3 Areas intended for the protection of biodiversity in the CDM.

Name	State(s)	Type of area	Year of decree
Federal Protected Natural Areas of Mexico			
Cañón de Santa Elena	Chihuahua	APFF	1994
Janos	Chihuahua	RB	2009
Medanos de Samalayuca	Chihuahua	APFF	2009
Mapimí	Chihuahua, Coahuila, and Durango	RB	1979
Ocampo	Chihuahua and Coahuila	APFF	2009
Río Bravo del Norte	Chihuahua and Coahuila	MN	2009
CADNR 004 Don Martín	Coahuila	APRN	1949
Cuatrociénegas	Coahuila	APFF	1994
Maderas del Carmen	Coahuila	APFF	1994
CADNR 026 Bajo Río San Juan	Coahuila and Nuevo León	APRN	1949
Cumbres de Monterrey	Coahuila and Nuevo León	PN	2000
Ríos y Montañas de la Comarca Lagunera	Durango	APRN	2024
Sierra Gorda	Querétaro and San Luis Potosí	RB	1997
Gogorrón	San Luis Potosí	APFF	1936
Sierra de Álvarez	San Luis Potosí	APFF	1981
Sierra de San Miguelito	San Luis Potosí	APFF	2021
Sierra La Mojonera	San Luis Potosí and Zacatecas	APFF	1981
Semidesierto Zacatecano	Zacatecas	APFF	2024
State, Municipal, Ejidal, Community, and Private Protected Natural Areas of Mexico			
Bosque Urbano Ejército Mexicano Polígono 1	Coahuila	PE	2013
Bosque Urbano Ejército Mexicano Polígono 2	Coahuila	PE	2013
Bosque Urbano Ejército Mexicano Polígono 3	Coahuila	PE	2013
Bosque Urbano Ejército Mexicano Polígono 4	Coahuila	PE	2013
Bosque Urbano Ejército Mexicano Polígono 5	Coahuila	PE	2013
Bosque Urbano Ejército Mexicano Polígono 6	Coahuila	PE	2013
Cuatro Gorriones	Coahuila	RNV	2017
El Cuatralbo	Coahuila	RNV	2017
El Palmar	Coahuila	RNV	2017
El Rescalco	Coahuila	RNV	2016
El Tulillo Lote 2	Coahuila	ADPEBIV	2013
El Tulillo Lote 4	Coahuila	ADPEBIV	2013
El Tulillo Lote 15a	Coahuila	ADPEBIV	2013
La India	Coahuila	RNV	2017
La Misión	Coahuila	RNV	2017

TABLE 4.3 Areas intended for the protection of biodiversity in the CDM—cont'd

Name	State(s)	Type of area	Year of decree
La Muralla	Coahuila	RN	2017
La Viga	Coahuila	ADPE	2016
Sierra y Cañón de Jimulco	Coahuila	REc	2003
Tomás Garrido	Coahuila	ADPEBIV	2014
Villa de Bilbao	Coahuila	ADPEBIV	2014
Zapalinamé	Coahuila	ZSCE	2008
Zapalinamé	Coahuila	ZR	2014
Cañón de Fernández	Durango	PE	2017
Acuña	Nuevo León	ZSCE	2000
Cañón Pino del Campo	Nuevo León	ZSCE	2000
Cerro La Mota	Nuevo León	ZSCE	2000
El Refugio de Apanaco	Nuevo León	ZSCE	2000
La Hediondilla	Nuevo León	ZSCE	2002
La Trinidad	Nuevo León (Aramberri)	ZSCE	2000
La Trinidad	Nuevo León (Galeana)	ZSCE	2002
Llano la Soledad	Nuevo León	ZSCE	2002
San Elías	Nuevo León	ZSCE	2000
San Juan y Puentes	Nuevo León	ZSCE	2000
Sandía El Grande	Nuevo León	ZSCE	2000
Sierra Corral de Los Bandidos	Nuevo León	ZSCE	2000
Sierra El Fraile y San Miguel	Nuevo León	ZSCE	2000
Trinidad y Llano Salas	Nuevo León	ZSCE	2000
Huiricuta, los Lugares Sagrados y la Ruta Histórico Cultural del Pueblo Huichol	San Luis Potosí	REPC	2000
Manantial de la Media Luna	San Luis Potosí	PE	2003
Palma Larga	San Luis Potosí	PE	1998
Real de Guadalcázar	San Luis Potosí	REs	2006
Parque Estatal Ruta Huichola	Zacatecas	PE	2009
Areas Voluntarily Designated for Conservation			
Dulce amparo del águila	Chihuahua		2018
Área de Germoplasma Relevante Rancho La Puerta	Coahuila		2015
Loma del Gorrión	Coahuila		2019
Rancho Media Luna	Coahuila		2005
Sierra San Vicente	Coahuila		2007
Tierra Silvestre Cañón del Diablo	Coahuila		2009
Zona de restauración Ecológica del Lobo Mexicano La Providencia	Nuevo León		2008

Continued

TABLE 4.3 Areas intended for the protection of biodiversity in the CDM—cont'd

Name	State(s)	Type of area	Year of decree
Zona de restauración Ecológica del Lobo Mexicano Potrero de Zamora	Nuevo León		2007
Zona de restauración Ecológica del Lobo Mexicano San Francisco	Nuevo León		2012
Zona de restauración Ecológica del Lobo Mexicano San Joaquín de Soto	Nuevo León		2007

ADPE, area destined for the preservation of ecosystems; *ADPEBIV*, area destined for the preservation of ecosystems and their biodiversity of voluntary initiative; *APFF*, flora and fauna protection area; *APRN*, natural resources protection area; *CADNR*, National Irrigation District Feeder Basin; *MN*, natural monument; *PE*, State Park; *PN*, National Park; *RB*, Biosphere Reserve; *REc*, Ecological Reserve; *REPC*, State Cultural Landscape Reserve; *Res*, State Reserve; *RN*, Nature Reserve; *RNV*, Voluntary Natural Reserve; *ZR*, Restoration Zone; *ZSCE*, Zone Subject to Ecological Conservation.

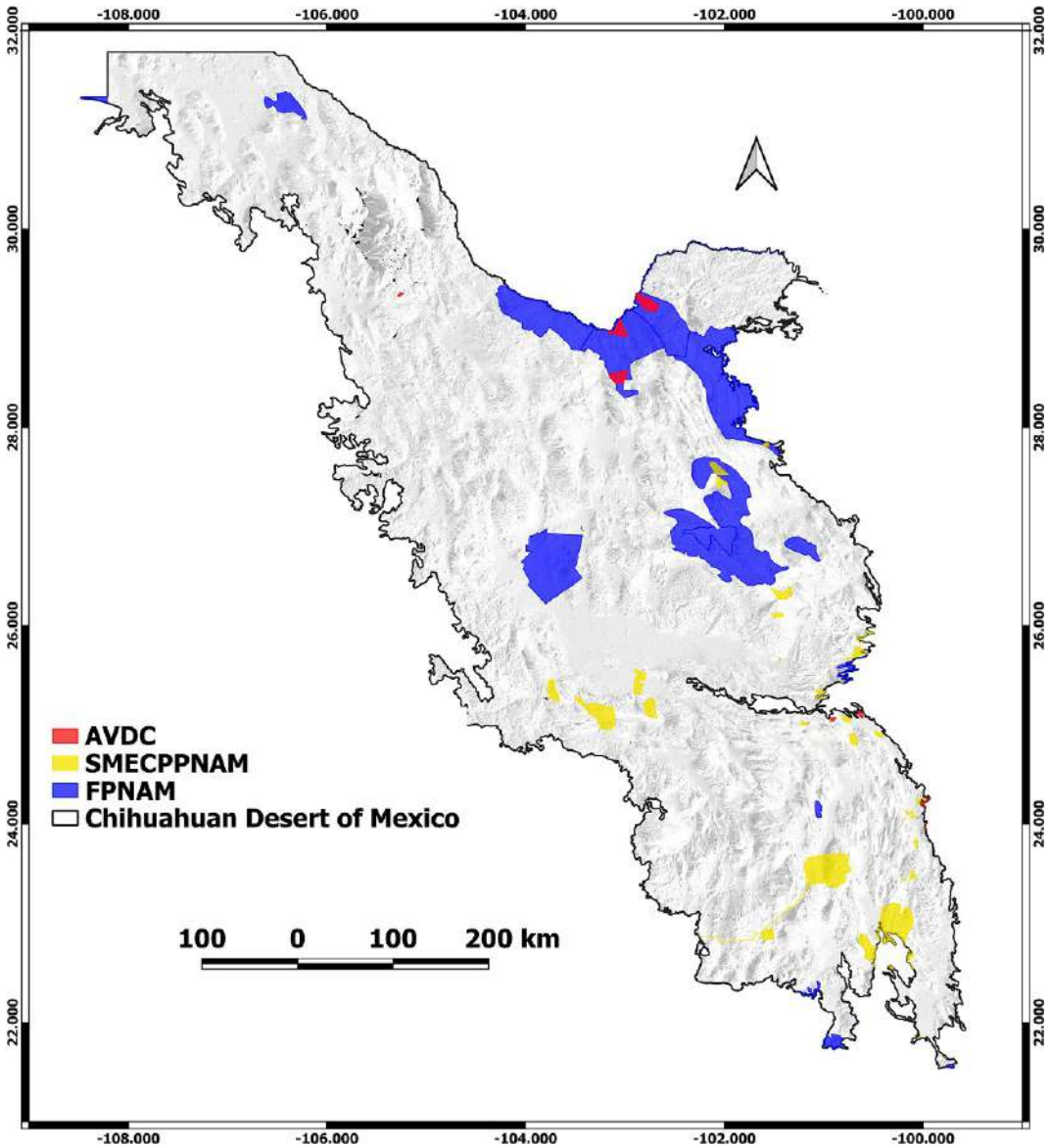

FIG. 4.4 Biodiversity conservation areas in the CDM. *AVDC*, Areas Voluntarily Designated for Conservation; FPNAM, Federal Protected Natural Areas of Mexico; SMECPPNAM, State, Municipal, Ejidal, Community, and Private Protected Natural Areas of Mexico. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: Alberto González-Zamora and Rebeca Pérez-Morales.)*

TABLE 4.4 Number of areas for the protection of biodiversity in the CDM.

Number of areas	Chi	Coa	Dgo	NL	Qro	SLP	Zac	Total
FPNAM	**6** (3)	**8** (5)	**2** (1)	**2** (2)	**1** (1)	**5** (2)	**2** (1)	18
SMECPPNAM	0	22	1	14		4	1	42
AVDC	1	5	0	4		0	0	10
Total	**7** (3)	**35** (5)	**3** (1)	**20** (2)	**1** (1)	**9** (2)	**3** (1)	70

AVDC, Areas Voluntarily Designated for Conservation; *Chi*, Chihuahua; *Coa*, Coahuila; *Dgo*, Durango; *FPNAM*, Federal Protected Natural Areas of Mexico; *NL*, Nuevo León; *Qro*, Querétaro; *SLP*, San Luis Potosí; *SMECPPNAM*, State, Municipal, Ejidal, Community and Private Protected Natural Areas of Mexico; *Zac*, Zacatecas. The number in bold indicates the total areas within the state while the number in parentheses indicates the number of areas shared with other states.

TABLE 4.5 Number of species within the areas designated for conservation in the CDM.

Area type	Number of species	Number of endemic species of Mexico	Number of endemic species of CDM	Number of species in IUCN
FPNAM	1694	312	87	187
SMECPPNAM	1144	291	41	143
AVDC	226	13	0	24
Total	**1974**	**463**	**114**	**238**

pronghorns, prairie dogs, and Mexican wolves were abundant. Currently, the ecosystem is highly disturbed due to changes in land use, including excessive grazing, land erosion, and the irrational use of water for agriculture; this excessive practice has caused most riparian forests and swamps to disappear. At present, some streams and springs are seasonally wet, and in many cases, the rivers and streams are completely dry. The disturbance of the ecosystem allowed the establishment of invasive species, such as grasses and the shrub *Tamarix ramosissima* Ledeb, which is considered aggressive since it establishes easily, displacing native vegetation (Ruhlman et al., 2012).

Human populations have grown substantially over the last few decades and have a greater demand for goods and services. This has caused extensive deforestation, intensive agricultural practices that degrade ecosystems, and increased industrialization that generates pollutants and greenhouse gases that lead to global warming, in addition to mass extinction of animal and plant species due to ecosystem degradation (Baena and Halffter, 2008).

In the CDM, some regions have undergone great economic and commercial development since the mid-20th century, among which agriculture, livestock, maquiladoras, and mining stand out. Most of the region's territory are plains, which make these areas more prone to the development of agricultural activities and the settlement of various urban localities, where anthropocentric activities increase the demand for surface water. This requires water to be extracted from greater depths, and combined with soil, water and atmospheric pollution, land use change, and desertification, the resulting environmental impact puts the species in the CDM at risk (González-Zamora et al., 2020).

Pollution problems in different areas of the CDM are due to agricultural and mining activities, mainly the accumulation of nitrates, arsenic, lead, cadmium, and even uranium (Méndez-García et al., 2016). The accumulation of these contaminants, which sometimes exceed the limits allowed by national and international standards, is an important factor in the illnesses experienced by human populations. Salas-Luévano et al. (2017) reported that different species of plants typical of the CDM, which grow in mines in the state of Zacatecas, were able to accumulate various contaminants such as arsenic, lead, and cadmium. That study proposed using these species for pollution programs as an alternative to contamination remediation.

Global climate change has been associated with human activities since the Industrial Revolution. Since that time, industrial processes have been carried out by burning fossil fuels such as oil, gas, and their derivatives. Gases such as carbon dioxide, methane, and nitrous oxide, which are generated by human activities at urban and rural sites, are released into the atmosphere, altering its composition by generating a greenhouse effect that promotes an increase

in temperature. Therefore, international agreements have been established to regulate and reduce these emissions (Scarpelli et al., 2020).

Another factor related to global warming is desertification, which is defined as the irreversible degradation of soil. This process has increased due to the overexploitation of natural resources, which can cause soil erosion through the loss of nutrients and microorganisms. This phenomenon can be reversed; however, it takes many years for soil recovery, so the land loses its productivity. Intense soil erosion has been documented in the CDM since the 1990s and is mainly due to changes in land use (Guevara et al., 2020).

Climate is dependent on a large number of factors that interact in complex ways, in addition to climatic variables such as precipitation, temperature, the composition of the atmosphere, and its interactions with the hydrosphere, which determines the state of the climate in different regions of the planet. There are several reasons for climate change, such as the instability of the atmosphere and hydrosphere and variations in the levels of solar radiation that the Earth receives; all of these changes have occurred in the planet, giving rise to cycles of glaciation and global warming; however, accelerated warming caused by land-use change and high greenhouse gas emissions derived from anthropocentric activities has been documented (Murray-Tortarolo, 2021).

In the last 100 years, accelerated vegetation loss has been observed in very large regions of Mexico. The United Nations Environment Program estimates that the country lost approximately 50% of its forests due to the deforestation of 6.3 million hectares of its forests, which caused changes in vegetation cover. The loss of vegetation cover, coupled with changes in land use, desertification, and high greenhouse gas emissions, has contributed to climate change and coincides with the loss of biodiversity in large parts of the Mexican territory.

Recently, the importance of climate variability as a catalyst for the loss of biodiversity in ecosystems has been highlighted because climate is one of the factors that delimit the distribution of plants, and each species requires certain light, temperature, and humidity conditions to establish its reproductive cycle. Although many species may tolerate climate change, when the degree of tolerance is exceeded, they cannot complete some phase of their life cycle, leading to a decrease in population density and eventually extinction. Climate change also represents a selective pressure on ecosystems that do not always lead to their extinction. For other species, climate change modifies their distribution ranges, which can favor their abundance and distribution. In a study carried out in Mexico, the HadCM2 models predicted a decrease in the number of species in coniferous and oak forests, xeric scrub, mesophilic forests and grasslands, as well as an increase in species in evergreen tropical forest, deciduous tropical forest, subdeciduous forest, and thorny forest (Arriaga and Gómez, 2004).

Therefore, the effect of climate change on ecosystems is expected to be reflected in the modification of the distribution and abundance of species, which can lead to the extinction of some species but can also lead other species to shift their distribution toward higher latitudes and altitudes different from those where they are currently found.

We carried out an analysis of species richness in the CDM in seven different time periods; for this purpose, we used the vegetation and land use maps of the INEGI Series I to Series VII (INE-INEGI, 1997; INEGI, 2001, 2005, 2009, 2013, 2016, 2021). In each period, the area allocated to agricultural activities and the number of species and endemic species that were lost at the end of each period due to changes in land use were quantified. We carried out an analysis of the possible loss of biodiversity in the CDM due to agricultural activities. Since this activity involves the modification of the habitat, we hypothesize that the historical records are no longer represented in the altered sites. Thus, we carried out an analysis of the area destined for agriculture according to INEGI data in seven time periods, namely, 1997, 2001, 2005, 2009, 2013, 2016, and 2021 (Fig. 4.5). The data show that the size of the altered surface area has increased relatively slowly but steadily, so that it currently corresponds to more than 42,000 km^2, which represents just over 12% of the total surface area of the CDM (Table 4.6 and Fig. 4.6).

The analysis of species richness showed that the number of species decreased as the surface area destined for agricultural activities increased in such a way that due to this increase, at the moment, approximately 347 vascular plant species have possibly been lost in the CDM, of which 157 are endemic species (Table 4.7). The lack of records of current collections in the study area prevents us from corroborating whether representatives of these species survive at other sites.

Data were taken from the CDM endemic species recorded to be present after 2021, that is, those that are not distributed in sites with agricultural activity. Of the total number of species, those that presented at least 10 distribution records were selected (Pearson et al., 2007; Ballesteros-Barrera et al., 2022). To model the potential distribution of the species under current conditions, 19 environmental variables from the WorldClim database (http://www.worldclim.org) (Fick and Hijmans, 2017) were used at a resolution of ~1 km^2. All layers were clipped according to the map representing the CDM. A correlation analysis (Pearson's correlation coefficient) was carried out to rule out the collinearity between the environmental layers and to identify redundant environmental variables at the sites of occurrence of each species (Wiens et al., 2009). A cutoff point of $r \leq 0.85$ was used (Elith et al., 2010).

For the distribution models and the projection due to climate change, future climate variables were obtained

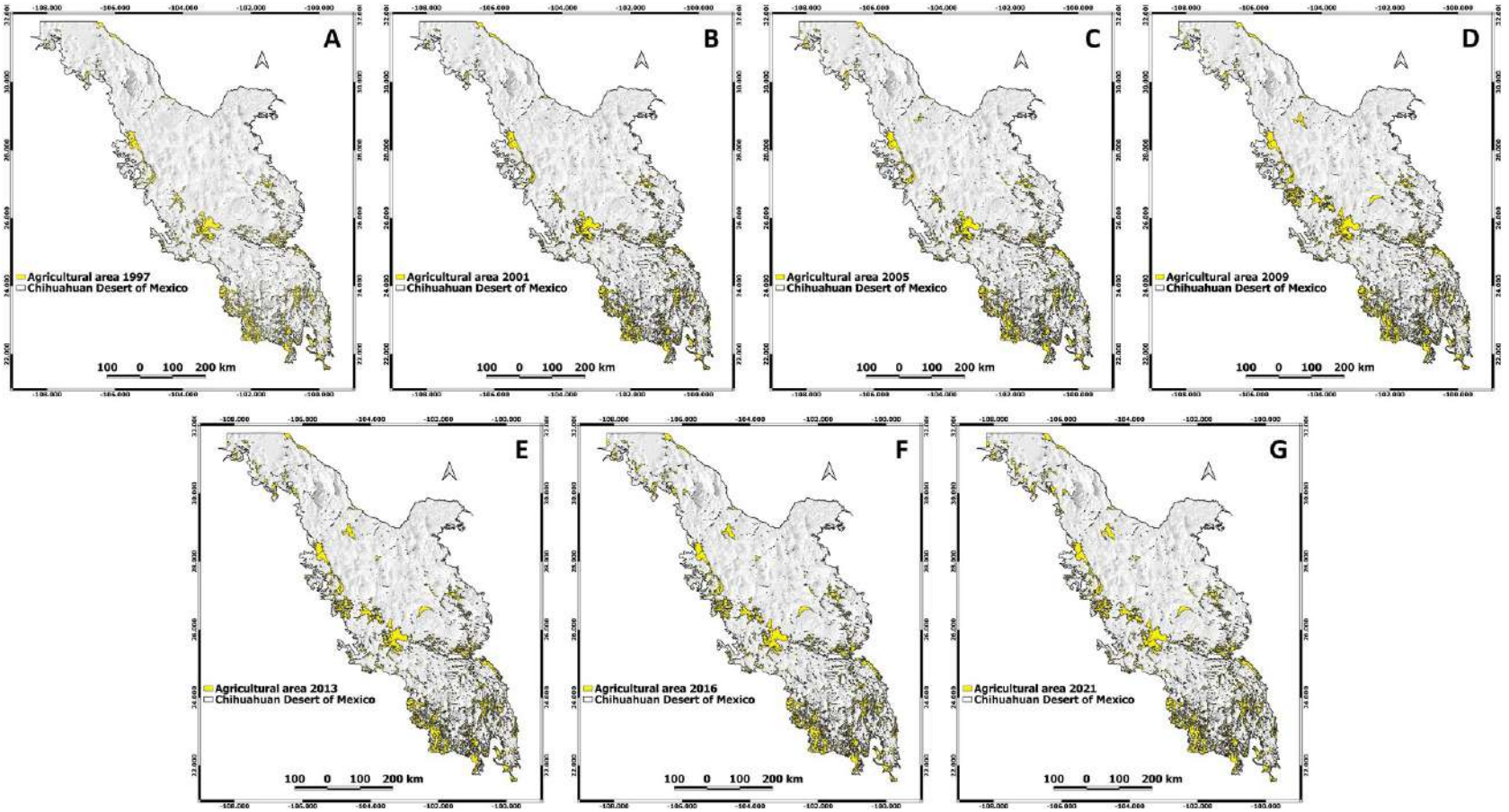

FIG. 4.5 Change in land use toward agricultural activities in the CDM according to INEGI. (A) 1997; (B) 2001; (C) 2005; (D) 2009; (E) 2013; (F) 2016; (G) 2021. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: Alberto González-Zamora and Rebeca Pérez-Morales.)*

TABLE 4.6 Area allocated to agricultural activities in the CDM.

Year	Agricultural area (km^2)	Surface increase (km^2)	Surface increase (%)
1997	21,329.5		
2001	29,134.4	7804.9	36.6
2005	31,382.5	2248.1	7.7
2009	36,920.1	5537.7	17.6
2013	39,089.2	2169.0	5.9
2016	40,539.1	1449.9	3.7
2021	42,038.7	1499.6	3.7

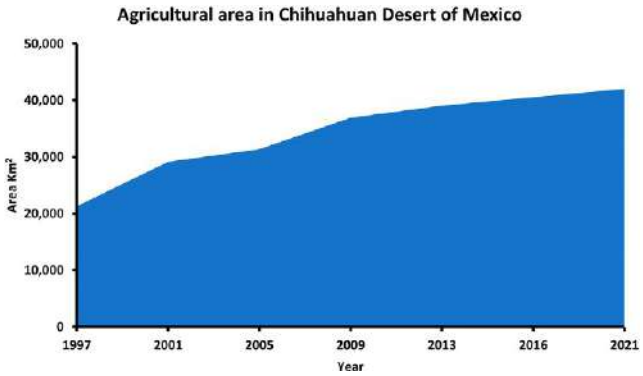

FIG. 4.6 Increase in the area destined for agricultural activities in the CDM. *(Photo credits: Alberto González-Zamora and Rebeca Pérez-Morales.)*

from the general circulation model (GCM) ACCESS-CM2 ssp585 (O'Neill et al., 2014), which predicts the combination of future radiative forcing; the GFDL-ESM4 ssp370 model (Krasting et al., 2018), which predicts a substantial change in land use (decrease in global forest cover) and high emissions of near-term climate forcers (NTCFs). Both models predict climate changes associated with alternative paths of socioeconomic development by the year 2060 (2041–60).

To develop the potential distribution models, the maximum entropy algorithm (MaxEnt v3.4.4) was used (Phillips et al., 2006) since this algorithm has been used to model species with small sample sizes (Elith et al., 2006; Ballesteros-Barrera et al., 2022). The estimate of the percentage contribution of each variable was obtained by a jackknife analysis (Meza-Joya et al., 2020). Modeling was performed by dividing the data set and using 80% for training and 20% for testing. The configuration in MaxEnt consisted of a maximum number of iterations = 500 and a convergence threshold = 0.00001 (Phillips et al., 2006). Models were validated through cross-validation of the area under the curve (AUC) of the receiver operating characteristic (ROC) curve. To maximize sensitivity and minimize specificity, the 10th percentile was used as the cutoff point for the preparation of the prediction maps. These maps were subsequently converted into binary maps to obtain the surface area potentially occupied by each species. This process was carried out both for the current environmental variable data and for the two climate change scenarios used in the present analysis. The changes in the surface of the potential distribution were obtained by calculating the difference between the surface of the current potential distribution and the surface of the distribution under the

TABLE 4.7 Changes in the diversity of vascular plants due to changes in land use in the CDM.

	Collection records			Species			Endemic species		
Year	Beginning of the period	End of period	Loss	Beginning of the period	End of period	Loss	Beginning of the period	End of period	Loss
1997	28,518	26,140	2378	3140	2012	1128	1009	662	347
2001	26,971	25,657	1314	2991	2246	745	933	720	213
2005	26,710	26,398	312	2940	2709	231	900	870	30
2009	26,946	26,623	323	2946	2734	212	901	854	47
2013	27,569	27,443	126	2968	2865	103	918	890	28
2016	28,007	27,725	282	2983	2765	218	926	875	51
2021	28,047	27,868	179	2978	2837	141	921	887	34
2023	28,069			2984			927		

climate change scenarios; the final result was expressed as a percentage.

Of the total species endemic to the CDM, only 38 have at least 10 unique collection records. Once the correlation analysis and modeling analyses were carried out, only 23 species belonging to 12 botanical families met the requirements for modeling the potential distribution, namely, Amaranthaceae: *Atriplex prosopidum* I.M. Johnst. and *Tidestromia tenella* I.M. Johnst.; Asteraceae: *Brickellia glutinosa* A. Gray, *Calanticaria brevifolia* (Greenm.) E.E. Schill. & Panero, *Sartwellia gypsophila* A.M. Powell & B.L. Turner, *Strotheria gypsophila* B.L. Turner, and *Varilla mexicana* A. Gray; Boraginaceae: *Nama constancei* J.D. Bacon and *Nama rotundifolia* (A. Gray) J.F. Macbr.; Brassicaceae: *Nerisyrenia castillonii* Rollins, *Nerisyrenia incana* Rollins, and *Nerisyrenia johnstonii* J.D. Bacon; Cactaceae: *Grusonia bradtiana* (J.M. Coult.) Britton & Rose, *Grusonia bulbispina* (Engelm.) H. Rob., and *Grusonia moelleri* (A. Berger) E.F. Anderson; Caryophyllaceae: *Drymaria elata* I.M. Johnst.; Gentianaceae: *Sabatia tuberculata* J.E. Williams; Malvaceae: *Sphaeralcea endlichii* Ulbr.; Nyctaginaceae: *Acleisanthes purpusiana* (Heimerl) R.A. Levin and *Acleisanthes undulata* (B.A. Fowler & B.L. Turner) R.A. Levin; Poaceae: *Sporobolus spiciformis* Swallen; Rubiaceae: *Houstonia teretifolia* Terrell; and Zygophyllaceae: *Zygophyllum scoparium* (Brandegee) Christenh. & Byng. None of the selected species appears on any list of endangered species.

Validation tests for the 23 species indicated that the models have good predictive capacity. The analysis of the ROC curves of 15 of the species showed high AUC values (greater than 0.9), while for eight species, values between 0.72 and 0.89 were obtained, which shows that the models are sufficiently robust to predict the presence of species in the study area. Of the climatic variables, BIO15 [precipitation seasonality (coefficient of variation)], BIO6 (minimum temperature of the coldest month), and BIO9 (mean temperature of a driest quarter) were the variables that presented the highest contributions under current conditions.

Under current climatic conditions, the suitability of the habitat for the modeled species indicates that, on average, for the 23 species, the potential surface area is 108,374 km^2. The smallest surface area is presented by *S. gypsophila* with 3344 km^2 while *C. brevifolia* presents the largest surface area, with 268,970 km^2.

Under the ACCESS-CM2 ssp585 scenario, for the year 2060, 15 of the species are predicted to have a decrease in their distribution area, the most severe being for *S. spiciformis*, with an approximately 36% decrease in surface area. Eight species will experience a decrease in the distribution area of less than 10%. In contrast, *A. purpusiana* will experience a positive effect on its distribution since it could increase its surface area by more than 170%, while six species will expand their area by less than 10%. In general, for species whose distribution decreases, the average contraction is 12%, while for those whose distribution expands, the average is 36%.

The results of the modeling under the GFDL-ESM4 ssp370 scenario showed some differences from those of the previous scenario; 15 species will experience a decrease in distribution area, 11 of which by more than 10%. *S. tuberculata* was the most affected species, with an approximately 50% loss in potential area. In the opposite case, *A. prosopidum* will experience an expansion of more than 180%, and the distribution area of the other five species will expand by more than 10%. For species whose distribution area decreases, the average contraction will be 17%; for those whose distribution area will increase, the average expansion will be 57%. The distribution areas of 10 species under the two scenarios will decrease, while the distribution areas of three species in both scenarios will increase. The remaining species will exhibit different behaviors in response to the effects of climate change according to the modeled scenarios (Fig. 4.7).

The results obtained in the present study are consistent with those of Cortés et al. (2014), Mosco (2019), and Iracheta-Lara et al. (2021), who also reported a not large decrease in the potential distribution areas of Cactaceae species under one scenario and an increase in the same under another scenario. The adaptation to the arid conditions of the CDM will apparently shield the distributions of plant species from being affected as strongly as those in other ecosystems, where the plants adapted to conditions of greater humidity and fewer temperature oscillations. In the CDM, the greatest risk to biodiversity will continue to be land use change, the abandonment of sites due to soil nutrient depletion, and the accumulation of contaminants.

6 Operational and potential tools to conserve or recover the biome

Efforts to conserve biodiversity in the CDM date back to 1936, with the establishment of the Area for the Protection of Flora and Fauna Gogorrón in the state of San Luis Potosí. After that, in 1949, the Areas for the Protection of Natural Resources Feeder Basin of the National Irrigation District (CADNR) 004 Don Martín and CADNR 026 Bajo Río San Juan were established in the states of Coahuila and Nuevo León. However, the establishment of areas for biodiversity protection and conservation in the CDM was very slow after, and from 1950 to 1987, no new area was decreed. Starting in the 1990s, there was a push to establish conservation areas at lower administrative levels, and this is when the SMECPPNAM the first AVDC was decreed through an extraordinary effort given the economic conditions of the country at the time (Fig. 4.8).

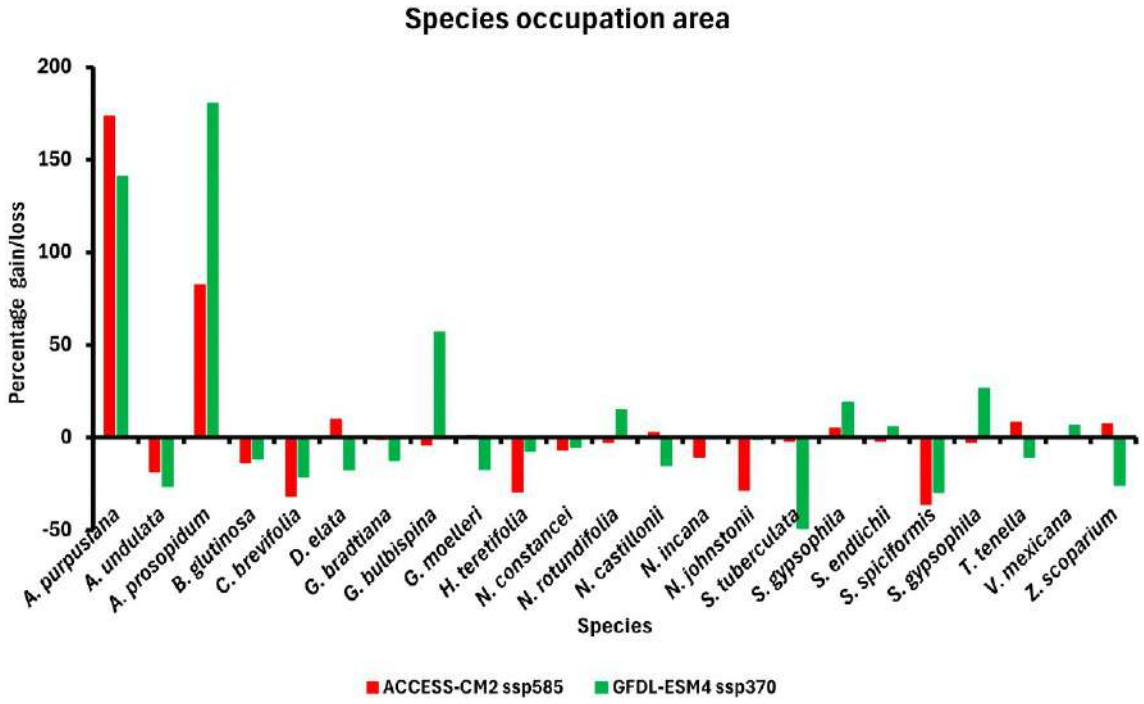

FIG. 4.7 Rate of change of potential range under two climate change models with respect to current environmental conditions. *(Photo credits: Alberto González-Zamora and Rebeca Pérez-Morales.)*

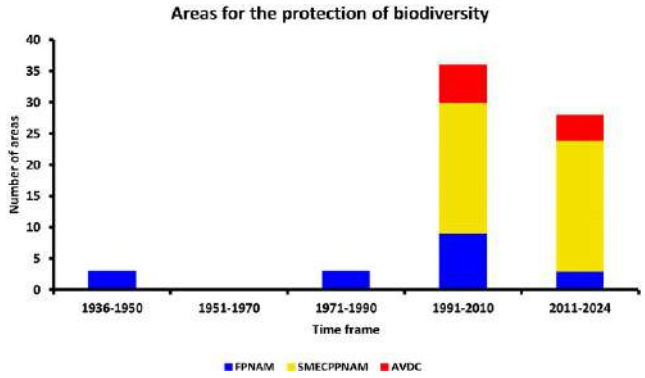

FIG. 4.8 Number of areas for biodiversity conservation in the CDM over time. *(Photo credits: Alberto González-Zamora and Rebeca Pérez-Morales.)*

Mexico participates in the Convention on Biological Diversity. This agreement proposed the National Strategy on Biodiversity, where the strategic axes of the Action Plan 2016–30 are knowledge of national biodiversity, conservation and restoration, sustainable use and management, attention to pressure factors, education, communication and environmental culture, and integration and governance (CONABIO, 2016). Addressing these axes requires exhaustive and continuous research to understand the biodiversity of specific areas, such as the CDM, as well as the causes of ecosystem deterioration and biodiversity loss. This is an essential requirement for identifying conservation zones, proposing the restoration of ecosystems and sustainably managing natural resources. This chapter presents data that provide relevant information for this important task, which requires reflection on the anthropocentric events that significantly affect climate change and contribute to the deterioration of the CDM.

The establishment of areas to protect biodiversity through voluntary initiatives of communities has increased; as noted by Arato (2016), at different sites of the CDM, the participation of communities in which women play an extremely important role has increased. In the conservation of resources, it is essential to maximize the benefits of socioeconomic development; this development is enhanced when private initiatives are involved (Gándara and Pérez, 2022).

Since 2011, the number of conservation decrees has decreased. It should be noted that in January 2024, the establishment of the area for the protection of flora and fauna in the Zacatecano Semidesert was decreed in Zacatecas (Diario Oficial de la Federación (DOF), 2024a), and the area for the protection of the natural resources of the rivers and mountains of the Lagunera Region was established in Durango, both at the federal level (Diario Oficial de la Federación (DOF), 2024b). The latter will undoubtedly help mitigate the deterioration of the region that has been documented in various works, which, together with the Agua Saludable para la Laguna (Healthy Water for the Laguna) project (Diario Oficial de la Federación (DOF), 2020), aim to reduce the impact on human health caused by

water pollution in the region (Gandarilla-Esparza et al., 2021; García-Torres et al., 2022).

7 Conclusions and perspectives

The biodiversity of a region has historically been related to the richness of species at a given time and environment; however, more recently, biodiversity has been recognized to encompass different aspects and levels of organization, from the genetic diversity of organisms to population variation and their interactions, which allow communities to be established and ecosystems to form. Therefore, biodiversity has been widely analyzed in conservation biology, where the main objective was to identify species loss, generally due to anthropocentric causes, and propose conservation zones to maintain species diversity and richness.

To mitigate the effects of changes in climate, in addition to the efforts made by the different levels of government, the involvement of citizens in the conservation and sustainable use of natural resources is necessary.

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Chapter 5

Simpson Desert

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1 Introduction

Lying between the Indian Ocean in the west and the Pacific Ocean in the east, and occupying latitudes between 10° 40′ S and 39° 08′ S, the Australian continental land area covers 7,595,342 km^2, with another 97,488 km^2 comprising islands of varying size. As might be expected for a continental landmass, Australia is climatically and geographically diverse and supports a very wide range of land systems and land forms. These varied environmental conditions and the continent's long and complex geological and tectonic history—notably its isolation from other land masses after its separation from East Antarctica in the mid-Paleogene—have helped to shape a rich biota that is largely endemic to the Australian region (Dickman, 2018). Although we focus in this chapter on the biome of the Simpson Desert in central Australia, we first briefly consider the history and physical geography of the continent to better appreciate the origins of the desert biota and the processes that influence and currently threaten it.

1.1 The central Australian environment through time

The soils of the Simpson Desert, like those of much of continental Australia, are mostly ancient and nutrient-poor, restricting opportunities for plant growth. These edaphic conditions result in part from the antiquity of source materials as well as from limited soil-building activities arising from orogenic and glaciation events. The world's oldest known terrestrial materials—crystals of zircon—have been identified from deposits in the Jack Hills in Western Australia. Dating to 4.374 billion years, these crystals formed just 125 million years after the formation of the Earth-moon system and represent remnants of the Earth's proto-crust (Valley et al., 2014). Other materials dating to 3.5 billion years occur in the Pilbara and Yilgarn cratons, or proto-continents, in Western Australia (Griffin et al., 2004; Hickman and Van Kranendonk, 2008), with sedimentary rocks dated between 2.5 and 3.5 billion years in the Pilbara region providing evidence of rivers and large bodies of water in which eroded sediments were deposited (Johnson, 2004).

Around 1.2–1.3 billion years ago, tectonic shifts and collisions between the North, West, and South Australian cratons resulted in the upthrusting of mountain ranges in parts of what is now central Australia. These ranges have since been heavily eroded, but remnants occur as the Musgrave Ranges in northern South Australia, Northern Territory, and eastern parts of Western Australia (Johnson, 2004). Mountain-building activity in central Australia has since been limited, with the Petermann Orogeny 630–520 million years ago and the Alice Springs Orogeny 450–300 million years ago being the last significant tectonic events (Quentin de Gromard et al., 2019). Major glaciation events also have been relatively unimportant for most of Australia's central regions. Although ice cover was extensive across much of the continent during the worldwide Sturtian and Marinoan glaciations (780–700 and 610–575 million years ago, respectively), ice cover since these intensely cold phases has been localized and of short duration (Johnson, 2004). In the Pliocene and Pleistocene, for example, Australia oscillated between arid glacial and humid interglacial conditions, but ice accumulated only at higher elevations in the Snowy Mountains of south-eastern Australia and in Tasmania (Colhoun and Barrows, 2011).

Except for the uplifting of the Great Dividing Range in eastern Australia that began in the mid-Cretaceous, the antiquity and geological stability of the Australian continental plate have resulted in vast areas being exposed to prolonged weathering. As a result, landscapes in many regional areas are characteristically gentle and undulating, with few topographically dramatic features. The average land height above sea level is 330 m (Johnson, 2004), with the highest peak—Mount Kosciuszko—reaching a modest 2228 m and the lowest point—Lake Eyre—being 16 m below sea level. With limited orogenic and glacial activity to generate "new" soil, the prolonged weathering of central

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Australian landscapes has also resulted in soils that are leached, often skeletal, and nutrient-depleted. The impoverished soils place certain constraints on plant growth, and have been important in shaping the evolution of Australia's biota (Augee and Fox, 2000; Dickman, 2007; Orians and Milewski, 2007; Bradshaw, 2018).

Central Australia is now predominantly arid and dominated by perennial grasslands and low shrublands, with tall trees restricted to riparian corridors or other local sites that provide reliable access to water. However, the prevailing aridity is punctuated irregularly by heavy rainfall events that drive pulses of high productivity and give the country a "green sheen" that may last several months (Wardle et al., 2013). Rainfall stimulates the germination of annual herbs and grasses and provides conditions that allow subsequent flowering and fruiting of many perennial plant species. Rains replenish water holes, restore flows in creeks and rivers and, because of the gently undulating landscapes, often lead to widespread flooding that creates vast but ephemeral lakes and wetlands (Morton, 2022). Visitors to central Australia often time their trips to coincide with the "boom" conditions that follow heavy rainfall, with sites such as Kati Thanda-Lake Eyre—covering up to 10,000 km^2 when full—being popular destinations (Wright, 2017).

Despite the arid conditions that prevail today, the climate of central Australia has oscillated between aridity and humidity at least since the early Miocene, beginning 23 million years ago. At this early date, the continent experienced tropical greenhouse conditions; the land was cloaked with extensive forests and high rainfall resulted in the formation of rivers and large areas with freshwater lakes (Fujioka and Chappell, 2010). Icehouse conditions began to take hold about 8 million years later with the formation of the Antarctic ice sheet, heralding the replacement of forests with grasslands, the drying of riverine systems, and the replacement of lacustrine systems with salt lakes (Fujioka and Chappell, 2010). Arid conditions intensified throughout the Pliocene, with the first characteristically arid landforms such as stony desert pavements forming 4–1 million years ago and sand dunes spreading largely over the last 1 million years (Byrne et al., 2008). Cold, arid conditions fluctuated with brief periods of high humidity, but culminated in the last very cold and dry period in the late Pleistocene 20,000–18,000 years ago (Hesse et al., 2005). The shift from mesic and even tropical conditions to widespread aridity in central Australia has had profound effects on the biota of the Simpson Desert and neighboring regions, as we discuss further below. The Deserts and Xeric Shrublands biome, which contains the Simpson Desert and Australia's four other major deserts—the Great Sandy Desert, Gibson Desert, Great Victoria Desert, and Sturt's Stony Desert—now covers 3,468,165 km^2 and is characterized by low but variable rainfall (<375 mm per year) and air temperatures that fall below 0°C in winter and can exceed 50°C in summer (Williams and Calaby, 1985; Olson et al., 2001).

1.2 Historical biogeography

Physical geography and climate provide the template for species' persistence, shaping species' niche sizes and shapes and contributing to the assembly and composition of their communities (Rosenzweig, 1995). However, these factors say little about the origins of biota, why some lineages occur in a region and why others do not, and why some taxa are locally or regionally endemic. These questions are of particular relevance for a now-isolated landmass like Australia. Of course, Australia has not always been an island continent, but was at the beginning of the Triassic a part of the supercontinent Pangaea. At this time, around 250 million years ago, mobile organisms were potentially able to occupy any parts of this landmass. Pangaea began to separate during the Triassic, with the northern landmass Laurasia (or Laurussia) sundering initially from the supercontinent and the southern landmass Gondwana separating and breaking apart throughout the Mesozoic Era (Johnson, 2004). Australia's last Gondwanan land connection, with Antarctica, began to break down approximately 85 million years ago. By the Eocene, 45–38 million years ago, the last land links between Australia and eastern Antarctica were severed, and the Australian continent became independent for the first time in more than 1.6 billion years (Johnson, 2004). The process that drove the separation of these land masses—continental drift—continues to move the Australian plate northward at around 6–7 mm per year, increasing the separation distance between Australia and Antarctica (White, 2006). In drifting northward, the Australian plate collided with the Pacific plate, resulting in buckling and uplift of land along the plates' active edges. Beginning around 30 million years ago, these tectonic events drove the creation of much of the Central Cordillera of New Guinea (White, 2006). Although isolated islands were in place prior to this time in the Moluccas, the formation of land to the immediate north of Australia reduced the continent's isolation and increased opportunities for dispersal of biota from continental Asia (White, 2006). Falls in sea level during the late Pleistocene created temporary land connections between northern Australia and New Guinea, further increasing opportunities for overland movement of biota.

The titanic events affecting Australia's location and land connections since the Triassic and earlier times have had a profound influence on the composition of the continental biota. Before the break-up of Pangaea, fossil evidence indicates that bryophytes, ferns, and gymnosperms were widespread and diverse; modern taxa in these groups

that now occur in different parts of the world share deep affinities that hearken to their Pangaean origin. Indeed, Augee and Fox (2000) noted that 86 families of vascular plants are found worldwide, and that representatives of more than half (189) of the world's 355 vascular plant families occur in Australia. Pangaean affinities can be traced also in many of the world's modern groups of invertebrates. Other elements of Australia's biota were either widespread in the Gondwanan section of Pangaea before separation from Laurasia, or arose in Gondwana post separation. Examples include angiosperms such as Cyperaceae, Myrtaceae, and Proteaceae, freshwater galaxiid fish, monotremes, and flightless ratite birds (White, 2006; Lamont et al., 2024). Modern marsupials also are descended from Gondwanan stock, moving into Australia from South America around 60–65 million years ago via the Antarctic land bridge (Beck, 2023). More recent immigrants to Australia appear to have made oceanic voyages from other southern continents (e.g., some spiders, the ancestors of the continent's nasute termites; Arab et al., 2017), island-hopped from the north via Melanesia (e.g., the ancestors of old endemic rodents; Aplin and Ford, 2014), or have arrived recently as a result of human activity (e.g., the dingo, *Canis dingo*; Dickman et al., 2021). Fossil evidence of the origins of different elements of the Australian biota is often fragmentary, but phylogenetic analyses are augmenting many of the gaps and confirm the diverse origins of many of the continent's major (and minor) plant, animal, and fungal groups (Heads, 2014; Oliver and Hugall, 2017; Archer et al., 2023; Lamont et al., 2024).

Whatever their origins, organisms in Australia's central desert regions are now subject to the prevailing environmental conditions. Phylogenetic studies indicate that several lineages of arid-adapted plants and animals diverged from mesic-adapted ancestors as conditions dried out from the mid-Miocene, with some taxon groups such as the large kangaroos appearing with the later expansion of grasslands (Dawson and Dawson, 2006). Further divergence took place in situ for many groups as marked climatic fluctuations initially isolated populations and then reassembled them, with this expansion—contraction process being particularly notable during the climatic oscillations of the late Pleistocene (Byrne et al., 2008, 2018). During the Pleistocene, the most iconic and now-widespread central Australian biome appeared, the arid grasslands characterized by spinifex or "hummock" grasses, *Triodia* spp. (Jacobs, 1982; Kershaw et al., 2000). Grasslands dominated by *Triodia* now cover 2,823,578 km^2 of arid Australia (Smyth et al., 2012), and also comprise the dominant vegetation of the Simpson Desert. The ancestors of all, or almost all, modern lineages of arid-adapted plants and animals were present during the Pleistocene, with a great many of these associated with the expansive spinifex grasslands (Byrne et al., 2008, 2018).

2 Geographic location

2.1 Landforms

The Simpson Desert lies in the center of continental Australia, with the largest area occurring in the Northern Territory and smaller areas in Queensland and South Australia (Fig. 5.1). About three quarters of the 176,500 km^2 expanse of the desert comprises sand dunes, with smaller areas of gibber plain, dissected residuals or "breakaways," playas and salt pans, especially in the south (Purdie, 1984; Fig. 5.2). The sand dunes are linear and run parallel for up to 200 km in a north-north-westerly to a south-south-easterly direction, and range in height from less than 5 m in the west to over 30 m in the east (Shephard, 1999). After rain, rivers flow intermittently and south-south-easterly through eight main channels along dune swales and produce ephemeral but spectacular pulses of productivity (Free et al., 2013). The desert is bounded in the east by Eyre Creek, the Mulligan and Diamantina Rivers, in the west by the Finke and Macumba Rivers, and in the south by the Warburton River and Kallakoopah Creek (Shephard, 1999). The floodplains of the latter two water courses, in particular, cover large areas during floods and support ephemeral saline wetlands that are unique in the broader desert region (Shephard, 1999). Freshwater springs and soaks occur principally in the eastern and southern parts of the desert (Fig. 5.2), supported by the vast (1.7 million km^2) Great Artesian Basin (Kingsford, 2017).

2.2 Climate

The Simpson Desert traverses a steep north-south rainfall gradient, with northerly areas receiving up to 300 mm a year, mostly in summer and frequently from monsoon troughs that are centered further north; southerly areas receive on average less than 150 mm a year. Classified as a hot desert, maximum daily temperatures in summer are 46–50°C, and minima in winter can fall to −6°C (Bureau of Meteorology, 2024). Median and extreme annual maximum and minimum temperatures have increased across the desert over the last century, but changes in rainfall over this period have been inconsistent (Greenville et al., 2012). Wildfires are common in the dominant hummock grasslands (see Section 3.6.1) and, while expansive (e.g., covering >2000 km^2), do not usually penetrate far into shrubby or woodland communities. Extreme rainfall events (>93rd percentile) predictably lead to wildfires 1–2 years later; rain triggers pulses of primary productivity that cover much of the soil surface which, when dry, forms a continuous layer of ground fuel (Greenville et al., 2009; Verhoeven et al., 2020).

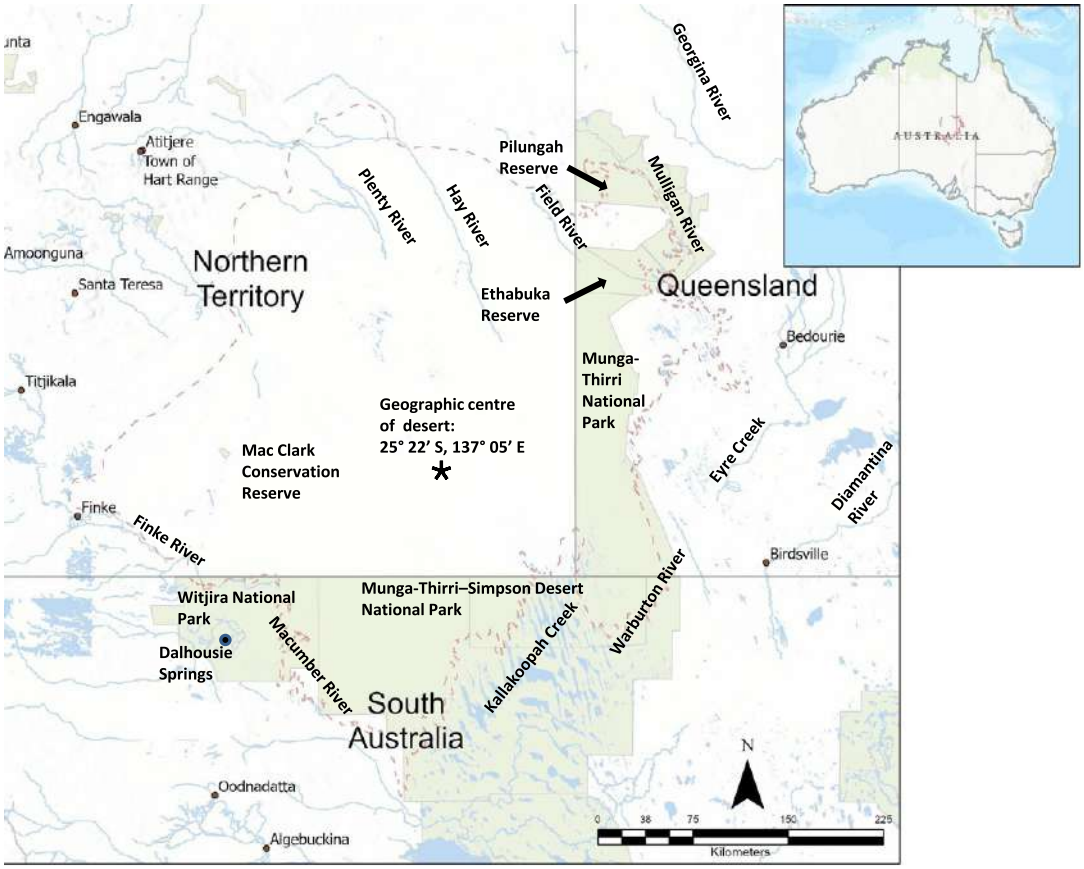

FIG. 5.1 The Simpson Desert, shown bounded by the *dashed orange line*, lies across the Northern Territory, Queensland, and South Australia and is located in the heart of the Australian continent. Light shading indicates land protected for conservation; no shading, mostly in the Northern Territory, indicates land used for cattle grazing.

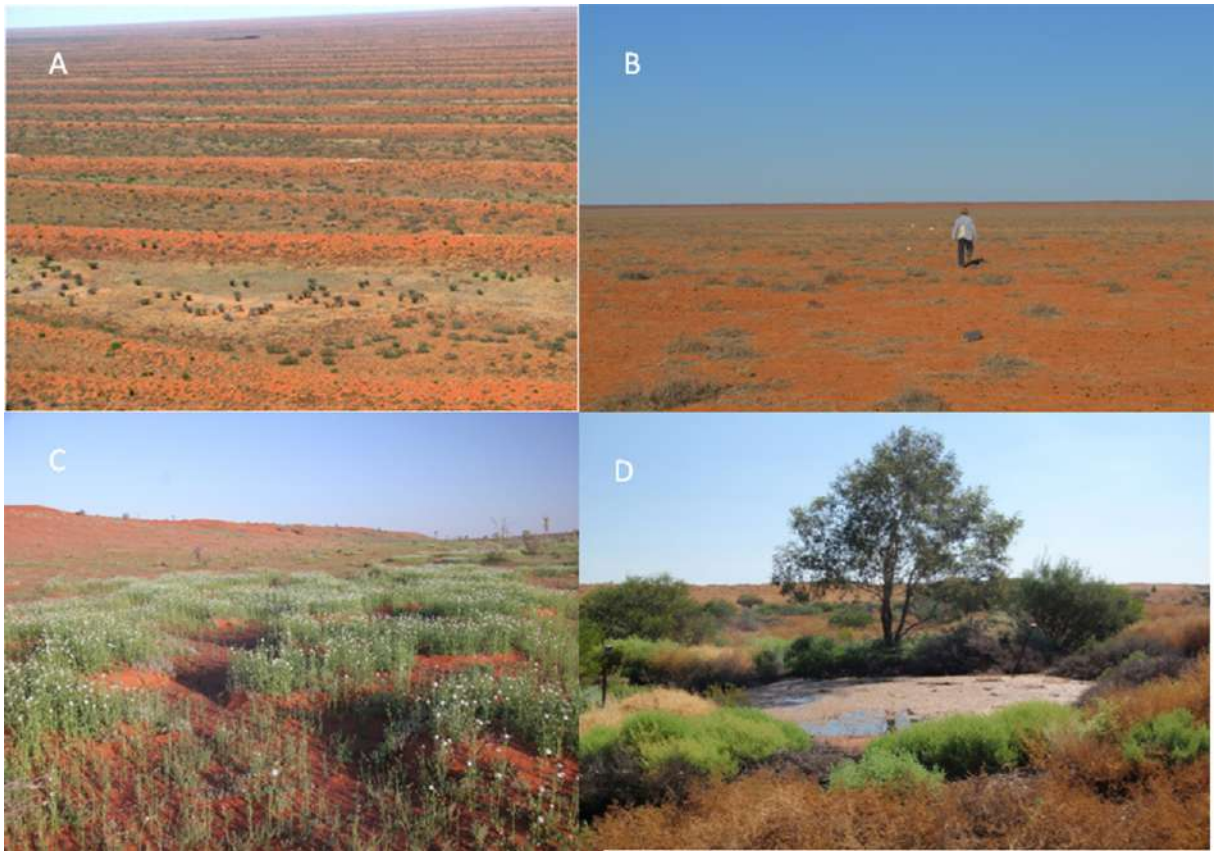

FIG. 5.2 (A) Long, linear sand dunes in the eastern Simpson Desert; (B) open gibber plain in the western Simpson Desert; (C) wildflowers after heavy rainfall; (D) a freshwater spring fed by underground aquifers of the Great Artesian Basin. *(Photo credits: Chris Dickman.)*

2.3 Land use and tenure

Mounting evidence indicates that Aboriginal people moved into parts of central Australia up to 49,000 years ago (Hamm et al., 2016; Tobler et al., 2017), weathering the severe climatic oscillations that characterized late Pleistocene environments. It is unclear how long the Simpson Desert has been occupied, but people used the desert when seasonal conditions allowed, often moving between *mikiri* or groundwater wells and Artesian springs (Roth, 1897). They also established permanent or semipermanent sites around riparian and lacustrine systems (Hercus and Clarke, 1986; Westaway et al., 2021). The desert provided resources for food, shelter, and ceremonial use, but materials for stone tools usually had to be transported from quarries beyond the desert boundaries. The Simpson Desert was used by several different groups of people, notably Wangkamadla in northern areas, Wangkangurru and Yarluyandi in eastern and southern areas, and Aranda, or Arrernte, in the west (Shephard, 1999).

European exploration of the Simpson Desert began in the mid-19th century, when a party led by Captain Charles Sturt penetrated the desert's south-eastern margin in 1845. Despite Sturt's assessment of the desert, made candidly in a letter to his wife Charlotte, that "A kind of dread came over me…. It looked like the entrance into hell." (Rudolph, 2013, p. 222), this was not enough to deter other explorers or slow the relentless expansion of the pastoral industry. With the encouragement of the Queensland and South Australian governments, pastoral settlement and exploration continued into the early 20th century, reaching the fringes and in some places entering the Simpson Desert and thus invading the traditional lands of the Indigenous custodians. The severe Federation Drought between 1895 and 1902 prompted some Aboriginal people to leave the desert in search of better conditions in settled areas but, for many more, traditional lands were usurped by the pastoral pioneers (Hercus, 1985). Many Aboriginal people were forcibly moved into missions or distant towns, others died from diseases they had not previously encountered, and others still were killed by pastoralists or armed forces such as the Native Mounted Police in Queensland (Watson, 1998; Griffiths, 2010). Known as the Arunta Desert for many years, the Simpson Desert was named by Cecil Madigan after Alfred Allen Simpson in 1929. Simpson, then President of the South Australian branch of the Royal Geographical Society of Australasia, provided material support for Madigan to make the first crossing of the desert for scientific research in 1939 (Madigan, 1946).

Despite its aridity and intermittent production of grasses and herbage for fodder, cattle grazing occurs in large parts of the Simpson Desert in the Northern Territory, Queensland, and South Australia. Grazing is permitted (and may even be required) under term leases in the latter two States, but in the Northern Territory the desert is grazed under freehold or perpetual lease conditions. However, large areas also have been set aside for conservation purposes, including Munga-Thirri National Park (>10,000 km^2) in Queensland, Witjira National Park (7726 km^2) and Munga-Thirri-Simpson Desert National Park (36,073 km^2) in South Australia, and several other protected areas of varying size in both States as well as in the Northern Territory (https://www.dcceew.gov.au/environment/land/nrs/science/capad, accessed March 23, 2024). Especially notable are Ethabuka and Pilungah Reserves (combined area 4463 km^2) on the eastern edge of the Simpson Desert that are owned by the nongovernment organization Bush Heritage Australia (BHA), and Mac Clark Conservation Reserve (30.4 km^2) in the Northern Territory, set aside for the protection of the rare and geographically restricted waddywood tree *Acacia peuce* (Fig. 5.3). The latter reserve is jointly managed by the Northern Territory Parks and Wildlife Service and Lower Southern Arrernte, Eastern Arrernte, and Central Arrernte traditional owners.

In recognition and partial recompense for past privations suffered by Aboriginal people on their traditional lands, much of the protected area estate in the eastern and southern Simpson Desert is now recognized as Aboriginal land under the *Native Title Act* 1993 and is comanaged by Aboriginal people with relevant State authorities. Comanagement combines Indigenous and Western knowledge of caring for the land and natural resources, facilitates continuing traditional use and protection of culturally significant sites, and also allows for appropriate recreational use by members of the public. Owing to the remoteness of the Simpson Desert from major population centers and difficulty of access via unsealed roads, there is virtually no permanent human settlement throughout the desert's vast expanse.

Tourism is low key and mostly restricted to four-wheel drive enthusiasts. However, remote camping, organized car rallies, and treks with camels occur from time to time during the cooler months of the year. Tourism is virtually nonexistent over the summer months, and the national parks may be closed during periods of extreme heat. Mining exploration is the only other main land use in the Simpson Desert, but this activity has to date been limited largely to road construction and test drilling in spot locations. Nonetheless, networks of seismic exploration tracks, or "shotlines," have been created in southerly and central parts of the desert, providing ingress routes for intrepid travelers. Early mining exploration, beginning in the 1960s, was for oil and gas, but more recently prospecting has occurred for gold and varied minerals for use in renewable industries (Shephard, 1999).

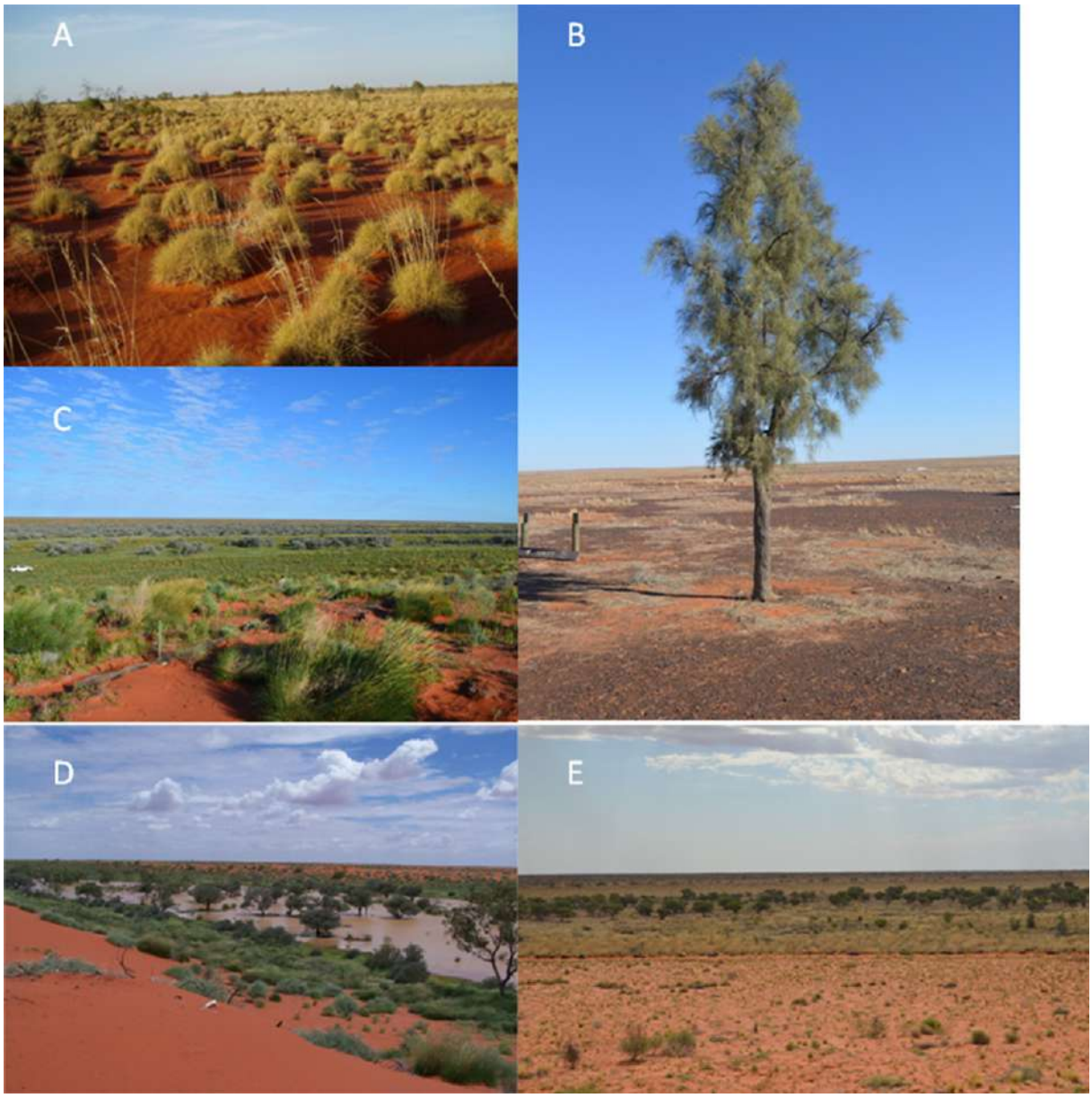

FIG. 5.3 (A) Spinifex grassland showing hummocks of the dominant *Triodia basedowii*; (B) waddywood tree *Acacia peuce*, Mac Clark Conservation Reserve, Northern Territory; (C) hummock grassland in the foreground on dune crest, with Georgina gidgee *Acacia georginae* woodland in the background; (D) flooded claypan after extreme rainfall; and (E) fire scar with sparse above-ground vegetation postwildfire in foreground and unburnt vegetation in the background. *(Photo credits: Chris Dickman (A, B, C, E); Glenda Wardle (D).)*

3 Biodiversity

3.1 Indigenous biocultural knowledge and use

Archaeological evidence suggests that humans occupied inland Australia by 50,000–40,000 years ago (see also Section 2.3). However, inland colonization by Aboriginal people is generally considered to have occurred during periods of higher rainfall, with groups adapting to gradual desertification (post-35,000 years ago) by modifying foraging and social strategies used in wetter environments. This is known as the desert transformation model (Hiscock and Wallis, 2005). The Last Glacial Maximum (LGM), occurring 25,000–18,000 years ago, featured intense desiccation and expansion of the arid interior. During this time, there was local or regional abandonment of much of the arid interior by people, apart from refuge sites. Indigenous knowledge and use of flora and fauna in the Simpson Desert today is viewed

as being part of the strategy of late Holocene desert adaptation that evolved after the LGM (Hiscock and Wallis, 2005).

The Simpson Desert was permanently occupied by Aboriginal people prior to European settlement. It is home to Indigenous people who come from multiple language groups, but most speak dialects that are within the Arandic or Arabanic languages (Smith, 2005). The Wangkangurru occupied the southern Simpson Desert. They obtained water from claypans after rain and, during these periods of surface water availability, spread out across the desert (Hercus and Clarke, 1986). When rainwater was not available, the occupation of the Simpson Desert by the Wangkangurru was centered on locations with *mikiri*. The *mikiri* were wells that had been dug as much as 7 m into the ground and usually tapped into groundwater in surface aquifers. People obtained water by climbing down into the wells through a system of narrow tunnels (Hercus and Clarke, 1986). As there is no permanent surface water in the Simpson Desert, the *mikiri* were essential for permanent occupation of this region.

Aboriginal people have an intimate relationship with the fauna and flora of the Simpson Desert that includes a spiritual connection and use as well as a deep knowledge of the ecology of many species of plants and animals. Simpson Desert people were nomadic hunter-gatherers who used, and continue to use, a wide range of plant and animal species for food and other resource requirements. They are similar to traditional Aboriginal people throughout Australia in their acceptance of a cosmology based on what is known as the Dreaming (Sutton and Walshe, 2021). To Aboriginal people, the Dreaming is the ultimate spiritual foundation of life and a system in which economics agrees with ancestral authority. Under this system, people manage important resource species through spiritual propagation, which involves performing maintenance or increase rituals at species-specific sites, singing the songs of resource species during ceremonies, and maintaining systems of totems for species. Totemism is characterized by reference to mythological ancestral beings that emerged from the Earth at the beginning of time. These ancestors moved across the Earth and brought shape to the continent and animated the landscape (Spencer and Gillen, 1875–1912). The mythology of important resource sites is well illustrated in the account by Hercus and Clarke (1986) of nine desert wells or *mikiri* in the southern Simpson Desert. Each one of the *mikiri* has myths associated with it and they are often central sites for myths of ancestral beings.

Aboriginal people in the Simpson Desert used, and continue to use, a very wide range of plants and animals for food and other resource requirements. Bread and small patties/cakes were a staple component of the diet and were made from the seeds of grasses and other plants including nardoo, *Marsilea drummondii* (Clarke, 2008). The preparation of cakes from nardoo was an involved process that included pounding and sluicing to make flour and then baking. Grinding stones were used to remove spore cases from nardoo and also the husks from grass seeds (Clarke, 2008). Nectar from the flowers of a range of shrubs, including the desert myrtle *Aluta maisonneuvei* and grevilleas, was consumed directly or mixed with water and drunk. Fruits of many shrubs and trees were consumed when available. Large animal species including the red kangaroo *Osphranter rufus*, emu *Dromaius novaehollandiae*, and Australian bustard *Ardeotis australis* were sought after as food. However, a very wide range of animals was consumed, including invertebrates, reptiles, small birds, and mammals. Famous examples include edible beetle and moth larvae that live in the roots and lower trunks of shrubs.

3.2 Vegetation

Hummock grassland dominated by hard spinifex *Triodia basedowii* occurs on the sides of sand dunes and in the swales between the dunes throughout the Simpson Desert (Fig. 5.3). Overall ground cover can be as high as 40%–50%, but this declines to 20%–25% during prolonged drought, and to 1%–2% coverage after wildfire (Nguyen et al., 2015). Mast seeding can occur after heavy summer rainfall; the large amount of seed produced fuels irruptions of rodents, granivorous birds such as budgerigars *Melopsittacus undulatus*, and zebra finches *Taeniopygia castanotis*, and grass regeneration postfire. Shrubs dominated by *Acacia* spp., *Eucalyptus* spp., *Eremophila* spp., *Grevillea* spp., and many others occur singly or in small stands among the hummock grassland, and a high diversity of ephemeral grasses, herbs, and forbs appears after rain (Purdie, 1984). Larger stands of Georgina gidgee *Acacia georginae* trees occur on loamy and heavier clay soils in dune swales (Fig. 5.3), and tall coolibahs *Eucalyptus coolabah* and bloodwoods *Corymbia terminalis* line the banks of ephemeral creeks and rivers. Areas dominated by tussock grasses such as *Aristida* spp. and *Astrebla* spp. and chenopod shrubs including *Atriplex* spp., *Maireana* spp., and *Sclerolaena* spp. occur patchily throughout the desert. Samphire communities with *Halosarcia* spp. and *Tecticornia* spp. have restricted occurrences around clay pans and saline flats (Purdie, 1984).

Building on extensive knowledge of the Simpson Desert flora gained by Aboriginal people, collections and descriptions of desert plants began to be made by European explorers in the second half of the 19th century. Charles Sturt in 1845, Charles Winnecke in 1880 and 1883, and naturalists on the Horn Scientific Expedition in 1894 returned significant numbers of plants, with many new species and genera being described. Eardley (1946) listed 71 flowering plant species in 22 families, as well as 6 species of fungi that were collected during the crossing of the desert by Madigan

in 1939. Symon (1969) listed 180 flowering plant species from the Simpson Desert, and Shephard (1999) expanded the list further to 789 species in 75 families. Surveys by the authors and other researchers over the last 25 years have added substantially to these numbers. In total, 240 species of algae and 2 species of moss have been recorded, all from mound springs and ephemeral or semipermanent water bodies (Table 5.1). Small numbers of fungi and lichens occur. Vascular plants occur throughout the desert in all habitats, the vast majority of species being angiosperms (Table 5.1), with about a third belonging to the families Amaranthaceae, Asteraceae, Fabaceae, and Poaceae.

Except for the vascular plants, it is likely that the numbers of taxa presented in Table 5.1 are underestimates of true species-level diversity because of the limited sampling that has been carried out in most parts of the Simpson Desert. Apart from waddywood, which is listed as vulnerable under national legislation, no desert plant species are listed as at risk of extinction, and none is known to be endemic. However, there is local variation in some species. For example, leaves of the widespread shrub, pituri *Duboisia hopwoodii*, contain nicotine, but particularly high levels of this narcotic in pituri are found only in the north-eastern Simpson Desert, leading to a substantial trade in leaves from this area by Aboriginal people across inland Australia (Silcock et al., 2012). Compared with other Australian regions, the Simpson Desert has relatively few introduced plant species and—with the exception of buffel grass *Cenchrus ciliaris*, which is spreading in many areas—no introduced plants are widespread. Shrubs such as mesquite *Prosopis pallida* and mimosa bush *Acacia farnesiana* are currently present in peripheral localities but can spread and establish rapidly after heavy rains (Milson, 1995).

TABLE 5.1 Plant and fungus species of the Simpson Desert.

Native/introduced species	Higher order classification	No. of species
Fungi		
Native	Fungi	13
Lichen		
Native	Lichen	5
Algae		
Native	Chlorophyta	54
	Cryptophyta	1
	Cyanophyta	19
	Dinophyta	10
	Euglenophyta	25
	Heterokontophyta	28
	Streptophyta	103
Bryophyta		
Native	Moss	2
Pteridophyta		
Native	Ferns	2
Gymnospermae		
Native	Cupressaceae	1
Angiospermae		
Native	83 families	908
Introduced	10 families	17

Results collated from Eardley (1946), Symon (1969, 1984), Gibson and Cole (1988), Latz (1995, 2007), Milson (1995), Shephard (1999), Kutsche and Lay (2003), Blake (2009), Thomas and Turpin (2009), Williams and Bailey (2009), ALA (2024), and the authors.

3.3 Vertebrates

Surveys carried out at different times and in different parts of the Simpson Desert by the authors and by other researchers have documented at least 286 extant species of native vertebrates (Table 5.2). In addition to these native species, a further 9 species—all mammals—have been introduced over the last 150 years, while at least 14 native mammals have become regionally or wholly extinct over the same period (Letnic and Dickman, 2006). That so many species persist in a harsh and variable desert environment is ostensibly surprising, especially as taxa from five classes within the subphylum Vertebrata are represented, including water-dependent ray-finned fish (or actinopterygians) and amphibians. Apart from four species of fish that are endemic to the Dalhousie Springs complex on the south-western edge of the desert, no vertebrates are restricted to the Simpson Desert; all have distributions that extend beyond the desert's boundaries. Some species are rarely encountered despite targeted surveys in the Simpson Desert, and others are listed as at risk by the International Union for Conservation of Nature (IUCN) (Table 5.2); these taxa may be particularly susceptible to the impacts of climate change.

Like vertebrates in other arid environments, those exploiting the Simpson Desert use several different survival strategies. Some species are restricted to habitats that provide relatively sheltered conditions at all times, and include riparian corridors, permanent water holes, and rock outcrops. Other species retreat to such "oases" when conditions in the broader desert environment deteriorate, while some use dormancy to avoid periods of low resources and emerge only when conditions have improved (Dickman et al., 2007a). However, most vertebrates use one of two broad strategies to persist: they move in from

TABLE 5.2 Vertebrate species of the Simpson Desert.

Native/ introduced species	Higher order classification	No. of species	Extant	Extinct	IUCN status of extant species		
					Least concern	Near threatened	Threatened[a]
Mammalia							
Native	Marsupialia	25	15	10	12	2	1
	Monotremata	1	1		1		
	Rodentia	11	8	3	5	1	2
	Chiroptera	13	12	1	11	1	
	Carnivora[b]	1	1		1		
Introduced	Rodentia	1	1		1		
	Carnivora	2	2		2		
	Lagomorpha	1	1		1		
	Artiodactyla	3	3		3		
	Perissodactyla	2	2		2		
Aves: Birds							
	Casuariiformes	1	1		1		
	Galliformes	1	1		1		
	Anseriformes	8	8		8		
	Caprimulgiformes	2	2		2		
	Apodiformes	1	1		1		
	Otidiformes	1	1		1		
	Cuculiformes	3	3		3		
	Columbiformes	6	6		6		
	Gruiformes	3	3		3		
	Podicipediformes	2	2		2		
	Charadriiformes	14	14		12		2
	Suliformes	2	2		2		
	Pelecaniformes	9	9		9		
	Accipitriformes	11	11		11		
	Strigiformes	2	2		2		
	Coraciiformes	2	2		2		
	Falconiformes	6	6		5		1
	Psittaciformes	8	8		8		
	Passeriformes	55	55		53	1	1
Reptilia							
Native	Serpentes	17	17		17		
	Sauria	68	68		68		
Amphibia							
Native	Anura	9	9		9		
Actinopterygia							
Native		18	18		11	2	5

[a]*For simplicity, numbers of species listed under IUCN categories Vulnerable, Endangered, Critically Endangered and Data Deficient have been combined in this column.*
[b]*There is debate about whether the single species in this category—the dingo* Canis dingo*—should be considered native or introduced (Dickman et al., 2021).*
Results collated from Gibson and Cole (1988), Shephard (1999), Bailey and Cook (2009), Emmott and Wilson (2009), Gynther et al. (2009), Wells (2009), Kerezsy (2017), Wilson and Swan (2021), ALA (2024), and the authors.

more mesic areas beyond the desert fringe and stay temporarily for as long as conditions permit, or they use physiological, behavioral, or ecological means to remain active in situ. Although these strategies have not been clarified for all vertebrates of the Simpson Desert, we note them where appropriate in the accounts below.

3.3.1 Mammals

Most of the extant mammals of the Simpson Desert are small (<250 g) and remain in situ over long periods (Fig. 5.4). The 12 species of dasyurid marsupials are wholly or largely nocturnal; all include invertebrates in the diet, but larger species such as the kowari *Dasyuroides byrnei* and mulgaras *Dasycercus* spp. also hunt small vertebrates. All species appear to have either large home ranges or ranges that shift and allow individuals to track food resources over the desert landscape. Except for a single species of false antechinus, *Pseudantechinus* cf. *mimulus*, which is confined to rock outcrops in the north of the desert, all species shelter in burrows to escape daily temperature extremes, and most enter daily torpor to conserve energy and water (Dickman and Pavey, 2023). The eight species of extant native rodents also use burrows for shelter by day; although none is known to use torpor, most species consume green plant material, roots, seeds, invertebrates, and even fungi if available, and conserve water by producing concentrated urine (Murray et al., 1999). Indeed, the spinifex hopping-mouse *Notomys alexis* produces more concentrated urine than any other mammal that has been studied (Baverstock and Watts, 1974). Most of the desert's native rodents are

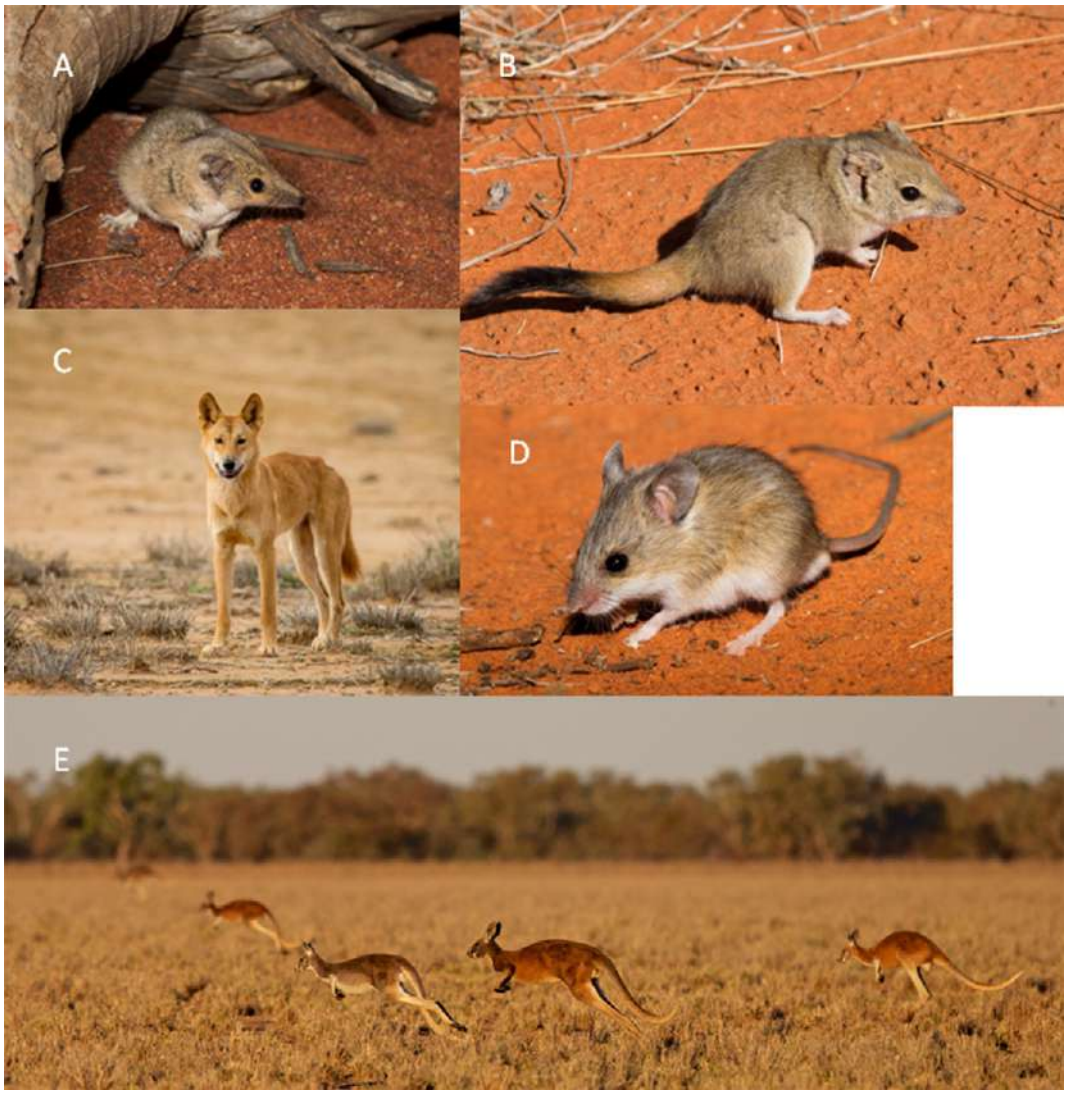

FIG. 5.4 Some mammals of the Simpson Desert: (A) Narrow-nosed planigale *Planigale tenuirostris*; (B) brush-tailed mulgara *Dasycercus blythi*; (C) dingo *Canis dingo*; (D) sandy inland mouse *Pseudomys hermannsburgensis*; and (E) red kangaroos *Osphranter rufus*. *(Photo credits: David Nelson (A, B); Angus Emmott (C, E); Bobby Tamayo (D).)*

capable of long-distance (>5 km) travel, and some species such as the sandy inland mouse retreat to small wooded refuge areas during drought (Pavey et al., 2017). Recent research has demonstrated how genetic diversity declines during long periods of confinement in refuge patches but reestablishes after rain when animals move out into the broader sand dune environment (Stringer et al., 2024).

Very little research has been carried out on the 11 extant species of microbats in the Simpson Desert, but some evidence indicates that most species shelter in clefts and small hollows in trees, and emerge after dusk to forage in patches of woodland, especially along riparian corridors (Williams and Dickman, 2004). Unlike its smaller and relatively sedentary microbat relatives, the little red flying-fox *Pteropus scapulatus* is an occasional visitor to the desert, traveling along riparian corridors to feed on eucalyptus blossoms when these are available, but retreating to mesic areas outside the desert when flowering ceases.

Larger native mammals include the echidna *Tachyglossus aculeatus* (5 kg), dingo *C. dingo* (15 kg), and three species of macropodid marsupials. The echidna occurs throughout the desert, escaping extreme temperatures on the soil surface by burrowing to depths of 1 m or more. It is active at any time in the diel cycle and preys upon two reliable food sources, ants and termites. By contrast, the three macropodid species rest by day, the euro *Osphranter robustus* and spectacled hare-wallaby *Lagorchestes conspicillatus* in rock outcrops, and the red kangaroo under the shade of shrubs or trees in more open areas. All three species are grazers, although the spectacled hare-wallaby also includes the leaves of forbs and shrubs in its diet. The red kangaroo is the largest extant marsupial (males weigh up to 90 kg) and is the only marsupial capable of moving in and out of the desert. Both sexes take only a day or so to travel distances of 25–50 km to access areas that have received local rainfall, but movements of 250 km have also been reported (Dawson, 2012). The dingo, a placental carnivore that was introduced to Australia ~4000 years ago, is the largest terrestrial predator in the Simpson Desert and a significant predator of the macropodids and other vertebrates.

The 14 species of native mammal that no longer occur in the Simpson Desert are either wholly extinct or have disappeared from the desert but remain extant in other areas (Table 5.3). It is likely that further species have

TABLE 5.3 Native mammal species that have disappeared from the Simpson Desert since European settlement.

Common name	Scientific name	Weight (g)	Last known desert record	Extant elsewhere?
Western quoll	*Dasyurus geoffroii*	1100	Pre-1900	Yes—Western Australia
Southern marsupial mole	*Notoryctes typhlops*	55	1900s	Yes—western sand deserts
Golden bandicoot	*Isoodon auratus*	1500	Pre-1900	Yes—north-western Australia
Yirratji	*Chaeropus yirratji*	~500	Pre-1900	No
Greater bilby	*Macrotis lagotis*[a]	1500	1980s	Yes—eastern and western desert areas
Lesser bilby	*Macrotis leucura*	400	1960s	No
Common brush-tailed possum	*Trichosurus vulpecula*	2500	1960s	Yes—peripheral continental areas
Burrowing bettong	*Bettongia lesueur*	700	1900s	Yes—Western Australia
Desert rat-kangaroo	*Caloprymnus campestris*	~850	1974/75?	No
Crescent nail-tailed wallaby	*Onychogalea lunata*	3500	1956	No
Short-tailed hopping mouse	*Notomys amplus*	~100	Pre-1900	No
Gould's mouse	*Pseudomys* cf. *gouldii*	50	Pre-1900	Yes—Western Australia
Lesser stick-nest rat	*Leporillus apicalis*	150	Pre-1900	No
Ghost bat	*Macroderma gigas*	150	1950s	Yes—northern Australia

[a]*This species possibly persists on the extreme eastern and western margins of the desert.*
Results collated from Finlayson (1961), Gibson and Cole (1988), Letnic and Dickman (2006), ALA (2024), and the authors.

disappeared; for example, based on early records or recent subfossil material from sites on the fringe of the Simpson Desert (Finlayson, 1961; Watts and Aslin, 1981), rodents such as the long-tailed hopping mouse *Notomys longicaudatus* and pale field-rat *Rattus tunneyi* may have occurred there before any surveys were undertaken. All species are within a "critical weight range" (35–5500 g) identified for terrestrial mammals by Burbidge and McKenzie (1989) as being at particular risk of decline and extinction, largely because of their vulnerability to two species of introduced predators, the European red fox *Vulpes vulpes* and feral cat *Felis catus*. We defer discussion of these species to Section 4.1. Reasons for the loss of the ghost bat *Macroderma gigas* from the Simpson Desert are unclear; Finlayson (1961) reported records from the Field River in the northern reaches of the desert, but surveys along this ephemeral water course in recent years have failed to detect it (Gibson and Cole, 1988; Free et al., 2015).

3.3.2 Birds

The number of bird species recorded in the Simpson Desert is relatively high, with almost 140 species detected so far from two long-term monitoring sites on the desert's eastern and western margins (Fig. 5.5). This compares favorably with, for example, a total of 106 species recorded across 197 sites during an intensive 9-month survey of the adjacent Finke IBRA region that took place during a period of high resource availability (Pavey and Nano, 2009). Species that are members of 19 orders have been recorded in the Simpson Desert (Table 5.2).

Species richness can vary greatly across time depending on whether productivity in the environment is high or low

FIG. 5.5 Some birds, reptiles, and frogs of the Simpson Desert: (A) Red-backed kingfisher *Todiramphus pyrrhopygius*; (B) wedge-tailed eagle *Aquila audax*; (C) thorny devil *Moloch horridus*; (D) three-lined knob-tailed gecko *Nephrurus levis*; (E) woma or sandhill python *Aspidites ramsayi*; and (F) desert spadefoot frog *Notaden nichollsi*. *(Photo credits: Aaron Greenville (A, B); Pavel German (C, D); Chris Dickman (E, F).)*

(boom vs bust) (Tischler et al., 2013). In addition, primary productivity varies across environments with sand dune vegetation being more stable and supporting generally higher resource availability over time than adjacent stoney clay plains (Nano and Pavey, 2013).

Given that there is virtually no permanent surface water in the Simpson Desert, water-dependent species move out of the Simpson Desert during dry periods or remain on its fringes in areas with available water often associated with human activities such as cattle ranching. They are, therefore, absent for long periods between successive high rainfall events. An influx of these species occurs when temporary wetlands hold water following rain, resulting in dramatic increases in the richness and abundance of birds over local or regional areas. Birds well known for such dramatic responses to rainfall include budgerigars, diamond doves *Geopelia cuneata*, orange chats *Epthianura aurifrons*, and wood swallows *Artamus* spp. These species require daily access to water and include many terrestrial species. Also included are "waterbirds," a term that refers to birds that live on or around freshwater. Waterbirds that occur in the Simpson Desert include species from a range of orders: Anseriformes (ducks, black swan), Gruiformes (dusky moorhen, Eurasian coot), Charadriiformes (shorebirds including migratory species), Suliformes (cormorants), and Pelecaniformes (Australian pelican, ibis, egrets, herons).

Smaller numbers of species are resident in the Simpson Desert irrespective of weather conditions and are able to survive despite a lack of water. Foremost among this group are those species of birds of prey that feed on invertebrates, reptiles, mammals, and carrion. These include the nankeen kestrel *Falco cenchroides*, brown falcon *Falco berigora*, and wedge-tailed eagle *Aquila audax*. A range of passerine birds is also resident in the Simpson Desert, although their numbers fluctuate over time. This list includes the Eyrean grasswren *Amytornis goyderi*, a small bird that is near-endemic to the Simpson Desert and occupies sandhill canegrass *Zygochloa paradoxa* on the crests and slopes of sand dunes.

Several birds of the Simpson Desert are listed as threatened nationally within Australia or are listed migratory species under the *Environment Protection and Biodiversity Conservation Act 1999*. The grey falcon *Falco hypoleucos* is endemic to the Australian arid zone and specializes in capturing birds, especially parrots, pigeons, and doves. This species ranges widely but is a breeding resident within the Simpson Desert. It is classified as Vulnerable. The painted honeyeater *Grantiella picta* is also Vulnerable and has been recorded within the eastern margin of the Simpson Desert as a vagrant. Four species of migratory shorebird that breed in the Northern Hemisphere spend the nonbreeding period in Australia during the Austral summer and sometimes occur in the Simpson Desert. One of these, the oriental plover *Anarhynchus veredus*, is an annual visitor as it is not reliant on the availability of surface water. It breeds mostly in Mongolia and occupies open plains and claypans in the Simpson Desert each year. The remaining three species occur in the desert only when surface water is present. The common greenshank *Tringa nebularia* is Endangered, the sharp-tailed sandpiper *Calidris acuminata* is Vulnerable, whereas the marsh sandpiper *Tringa stagnatilis* is a listed migratory species.

Climate change is projected to influence a range of weather variables in arid Australia over the coming years (see Table 3 in Dickman and Pavey, 2023). Although the direction of change in many variables is unclear, there is very high confidence in predictions of both an increase in average temperature across all seasons and an increase in the number and maximum temperature of hot days. The mean temperature in arid Australia increased over the period 1910–2013 by 0.9°C in the northern and 1.0°C in the southern arid zone. A recent analysis (Conradie et al., 2020) suggests that birds in arid Australia will suffer increased mortality risk as temperatures increase. Smaller species in the size range of 10–42 g are at particular risk from lethal dehydration and hyperthermia. Some parts of the geographic range of species such as the common and widespread zebra finch will no longer be suitable to meet their physiological needs and range shifts are predicted to occur. Mass mortality of birds during heatwaves has been reported from arid Australia, including the Simpson Desert, as far back as 1932 (Anon., 1932).

3.3.3 Reptiles

The reptilian diversity of the Simpson Desert is one of the greatest of any arid region in the world, with 17 species of snake and at least 68 species of lizard (Table 5.2; Fig. 5.5). With the possible exception of the perentie *Varanus giganteus*, a large carnivorous lizard that can attain body weights exceeding 15 kg and has been recorded to make movements over 10 km within a few days, all reptiles probably pursue a sedentary strategy and occupy fixed areas. They use an extraordinarily broad range of behavioral, ecological, and physiological adaptations to exploit food and shelter resources and cope with climatic extremes (Pianka, 1986; Bradshaw, 1997, 2018). Almost all the small lizards are insectivorous, as are three species of blind snake in the family Typhlopidae. With increasing size, some lizards such as the centralian blue-tongue lizard *Tiliqua multifasciata* include seeds, berries, and flowers in the diet, but most species increase their consumption of smaller vertebrates. This applies to most of the 8 species of varanid lizards of the Simpson Desert, 3 species of python, and 11 front-fanged snakes in the Elapidae family. The latter family contains brown snakes in the genera *Pseudechis* and *Pseudonaja*, as well as the inland taipan or fierce snake *Oxyuranus microlepidotus*, and the snakes are renowned as

being among the world's most venomous snakes (Emmott and Wilson, 2009).

Reptiles occur in all areas of the Simpson Desert, but often show strong associations with particular habitats or even habitat components. The jeweled gecko *Strophurus elderi*, for example, is restricted to hummocks of *Triodia* where its diet comprises almost solely termites; the regal ctenotus *Ctenotus regius* (one of 16 ctenotus species that cooccur in the desert) occurs only on clay soils in dune valleys and is a generalist insectivore, while the southern sandslider *Lerista labialis* is confined to dune crests where it "swims" through sand in search of termites (Pianka, 1986; Greenville and Dickman, 2009). Other species show marked preferences for dense ground-level vegetation or areas of open sand with no above-ground vegetation in the wake of wildfires (Letnic et al., 2004). There is also some regional differentiation within the Simpson Desert. The spiny-tailed monitor *Varanus acanthurus* and red-sided ctenotus *Ctenotus pulchellus* are restricted to rock outcrops in the northern part of the desert, whereas Aphrodite's ctenotus *Ctenotus aphrodite* and the fourteen-lined ctenotus *Ctenotus quattuordecimlineatus* occur only in the eastern and western portions of the desert, respectively (Gibson and Cole, 1988; Gynther et al., 2009). Differences in the ecology, resource requirements, responses to disturbances such as wildfire and heavy rainfall, as well as in local and regional occurrence, likely contribute to the extraordinary richness of reptile species within the Simpson Desert (Pianka, 1986; Downey and Dickman, 1993; Dickman et al., 2020; Pepper and Keogh, 2021). Unlike the mammals, no reptiles of the Simpson Desert are listed as threatened or near-threatened (Table 5.2).

3.3.4 Amphibians

The nine frog species recorded from the Simpson Desert (Table 5.2) are representatives of the families Limnodynastidae and Pelodryadidae (Hylidae). The most widespread and commonly encountered species are the desert spadefoot frog *Notaden nichollsi* (Fig. 5.5) and Sudell's frog *Neobatrachus sudelli* (Limnodynastidae), both of which escape dry periods by burrowing in open sand and reducing their metabolic rates to conserve energy. Both species regularly shed their skin and use this as a barrier to prevent water loss; Sudell's frog uses the shed skin to form a cocoon around the body, reducing water loss by up to 99% (Lee and Mercer, 1967). This species burrows to depths of 30–50 cm and can tolerate dry soil, whereas spadefoot frogs may descend to 1 m to maintain hydration (Dickman and Letnic, 2010). Although rainfall events of just 10 mm may prompt frogs to break their aestivation and come to the surface, they use the wet conditions opportunistically to feed and rehydrate; much larger rainfall events that fill claypans are usually required for reproduction (Predavec and Dickman, 1993; Dickman and Letnic, 2010). Another limnodynastid, Spencer's burrowing frog *Platyplectrum spenceri*, also burrows to escape hot, dry conditions on the soil surface, but prefers the sandy beds of ephemeral creeks in which it breeds after rain or water flows. A fourth limnodynastid, the spotted grass frog *Limnodynastes tasmaniensis*, may dig temporary burrows but often shelters under dense cover on the banks of temporary or permanent bodies of water.

Pelodryadid frogs are subject to ongoing taxonomic revision (Duellman et al., 2016; Brennan et al., 2024), but fall into two ecologically different groups. Three stout-bodied species dig burrows to a meter or more, usually in creek beds or on the edge of claypans, and may remain underground for several years until heavy rains prompt them to emerge from aestivation (Predavec and Dickman, 1993; Roberts and Edwards, 2018). These frogs maintain hydration during aestivation by storing, and using, free water in the bladder. In the water-holding frog *Dryopsophus* (=*Cyclorana*) *platycephala* the weight of water in the bladder may account for 130% of the animal's body weight (Hillman et al., 2009). By contrast, two species of hylid, the green tree frog *Dryopsophus caeruleus* (=*Litoria caerulea*) and desert tree frog *Litoria rubella*, do not burrow but use hollows in the trunks and limbs of trees to shelter. Suitable trees—usually eucalypts—grow on the banks of creeks and standing water bodies, with these frogs breeding after heavy summer rains.

Although hot deserts may seem to provide hostile environments for amphibians, frogs can be the most numerous vertebrates during and immediately post rain in some arid regions (Morton et al., 1993; Roberts and Edwards, 2018). The wet biomass of frogs in the Simpson Desert—over 2300 g/ha—may exceed that of reptiles and mammals post rain by up to two orders of magnitude (Predavec and Dickman, 1993), confirming that frogs can be an important but often overlooked component of desert faunas. No frogs in the Simpson Desert are considered to be at risk of extinction (Table 5.2).

3.3.5 Fish

Even more than amphibians, fishes require permanent free water and thus are patchily distributed in the Simpson Desert. Four endemic fish species occur in permanent water bodies at Dalhousie Springs on the south-western fringe of the desert, but all others are principally riverine species that occur largely on the desert's eastern and western margins (Glover, 1979, 1982; Australian Desert Fishes, 2024). On the eastern margin, Eyre Creek and inflows from the Mulligan and Georgina Rivers collectively support 11 fish species in 8 families, whereas in the west the Finke River system supports 9 species, 6 of

them shared with the drainages further east. Some fish species, such as spangled perch *Leiopotherapon unicolor*, occur also in smaller drainages such as the Hay River that runs through the center of the Simpson Desert, but diversity here is low.

Because the desert river systems frequently do not flow, fish must persist for periods of months to years in permanent water holes. Their widespread occurrence in these temporary rivers arises from their ability to migrate during times of flood, with several species such as bony bream *Nematalosa erebi*, silver tandan *Porochilus argenteus*, and glassfish *Ambassis* sp. recorded moving at least 300 km (Kerezsy, 2017). However, not all species are equally mobile. Species such as the Finke goby *Chlamydogobius japalpa* (IUCN: Vulnerable) and Finke mogurnda *Mogurnda larapintae* (IUCN: Near Threatened), for example, appear to be largely restricted to the headwaters of the Finke River in the MacDonnell Ranges, outside the Simpson Desert. Although they are included in the tally of species shown in Table 5.2, it is not clear how often or even whether such usually sedentary species recruit into the desert drainages (Australian Desert Fishes, 2024). Many desert fishes have distributions that extend beyond the desert margins, and affinities with congeners to the north or south of the Simpson provide clues to the origins of these species. The four species at Dalhousie Springs—all Critically Endangered under the IUCN criteria—are distinctive and are likely to have been restricted to this warm spring complex for many millennia (Kerezsy, 2020).

3.4 Invertebrates

The invertebrates of arid Australia are generally much less well studied than the vertebrates. Several detailed studies have been completed on particular invertebrate groups (e.g., termites (Watson, 1982); mygalomorph spiders (Main, 1982); freshwater crabs (Greenaway, 1984); land snails (Slatyer et al., 2007); subterranean invertebrates (Halse, 2018)) and suggest that the diversity of desert-dwelling invertebrates is manifold greater than for vertebrates in the same areas. However, targeted work on invertebrates in the Simpson Desert is limited (Fig. 5.6). Apart from a broad-scale survey of invertebrates running over 6 years in sand dune habitat (Kwok et al., 2016) and short-term (1-month) surveys in different habitats on Pilungah Reserve in the north-eastern portion of the desert (Feeney and Freemantle, 2009), focal studies have been carried out only on ants and pollinators, notably native bees. These studies have resulted in descriptions of many new species (e.g., Henschel et al., 1995; Feeney and Freemantle, 2009; Batley and Popic, 2013, 2016). Some of these species, such as the Ethabuka slit spider *Fissarena*

FIG. 5.6 Some invertebrates of the Simpson Desert. (A) Predatory carabid ground beetle; (B) herbivorous curculionid beetle; (C) desert scorpion *Lychas* sp.; and (D) Bird-eating spider *Selenocosmia* sp. *(Photo credits: Chris Dickman.)*

ethabuka, are likely to be endemic to the Simpson Desert (Henschel et al., 1995), but the distributions of most of the region's invertebrate taxa are poorly known. Invertebrates probably use a range of strategies similar to vertebrates to exploit arid conditions (e.g., dormancy, restriction to oases, mobility or remaining in situ), although these have been documented for a small fraction of taxa in the Simpson Desert (Duncan and Dickman, 2001, 2009).

The results of short-term and localized surveys in the eastern Simpson Desert hint at the extraordinary diversity of invertebrates that the desert supports. Sampling in and around water bodies, Powling and Russell (2009) described at least 68 species of zooplankton and other microfauna; Lemann and Weir (2009a) recorded 8 species of aquatic hemipteran bugs, while a further 26 species of other aquatic macroinvertebrates were detected by Powling and Bailey (2009). Further surveys for insects have reported >12 species of termite (Foster, 2005 and unpublished); 23 species of Collembola, with 7 of these undescribed (Greenslade, 2009); 17 species of cicada, with 14 undescribed (Ewart, 2009); 392 moth and butterfly species (Edwards and Glover, 2009); 124 beetle species (Lemann and Weir, 2009b); 50 native bees species (Popic et al., 2013) with 6 being newly described (Batley and Popic, 2013, 2016); and over 100 species of ant (Reid, 1995; Gibb et al., 2019). Among other arthropods, Estrada (2008) and Potter et al. (2018) documented at least 29 species of arachnid (27 araneids, 2 scorpionids) from sand dune habitats, while Richardson (2009) recorded a further 15 spider species (13 undescribed) near water sources in the single family Salticidae. Many other arthropods and other invertebrates occur in the Simpson Desert but remain poorly documented (Shephard, 1999).

Longer term studies of invertebrates in the Simpson Desert indicate that rainfall, temperature, dune position, vegetation structure, and productivity have strong but often disparate effects on the abundance of different invertebrate groups. Over a 6-year period, Kwok et al. (2016) found that numbers of mites, ants, and beetles increased with rainfall up to 18 months prior to sampling, whereas numbers of spiders declined; all taxa studied showed a positive relationship with the cover of non-*Triodia* vegetation. Over a period of 22 years, Gibb et al. (2019) showed further that while dominant ant species respond to long-term rainfall, subordinate species respond to short-term weather events; increases in local species richness were associated with increases in overall abundance. In later studies, Gibb et al. (2022) described the differential effects of productivity over time and dune position for different trophic groups of ants, concluding that the scale of rain-driven productivity pulses is a key determinant of ant assemblage dynamics.

3.5 Genetic diversity

Although biodiversity is sometimes viewed simplistically as the suite of species that occurs in a particular place, it also includes genetic diversity within species, diversity between species, and the habitats, communities, and ecosystems that arise when species interact with each other and the environment (Dickman et al., 2007b). In view of the great range of species in the Simpson Desert, from cyanobacteria, microfungi, and lichens to higher vascular plants and vertebrates, as overviewed above, the genetic diversity in the desert can be considered relatively high. Few taxa are known to be endemic to the Simpson Desert, and many occur outside the desert's boundaries. Nonetheless, the composition of species that might be found on any hectare of sand dune, gibber, or woodland habitat in the Simpson Desert would likely differ from that in comparable habitats in other arid regions (e.g., Gibson, 1986; Shephard, 1995; Robin et al., 2010), suggesting that the region is genetically unique. If measures of phylogenetic diversity are considered, incorporating species numbers, different geographical origins, evolutionary histories, and current rarity (Laity et al., 2015), the Simpson Desert can likely be viewed as an important repository of genetic diversity at the continental scale.

Within-species genetic diversity has been little studied in the Simpson Desert. However, an analysis of single nucleotide polymorphism (SNP) markers collected from two species of small mammals over 13 years in the Simpson Desert has shed light on how these species maintain genetic diversity through periods of rapid environmental change (Stringer et al., 2024). One species, the lesser hairy-footed dunnart *Sminthopsis youngsoni*, conserved genetic diversity by maintaining a relatively stable population size over time. The other species, the sandy inland mouse *Pseudomys hermannsburgensis*, lost genetic diversity during dry periods when subpopulations were confined to small patches of woodland, but regained genetic diversity after rain when animals moved out and intermixed in the surrounding sand dune environment (Stringer et al., 2024). Further research using SNPs and whole genome analyses is likely to uncover more aspects of species' population trajectories and evolutionary responses to desert conditions (Pepper and Keogh, 2021).

3.6 Ecological diversity

Desert environments provide templates for bewilderingly complex arrays of interactions, both among species and between species and the abiotic environment, and the Simpson Desert is no exception (Fig. 5.7). For brevity, we provide selected examples of interactions here; a more detailed treatment is provided by Dickman et al. (2014).

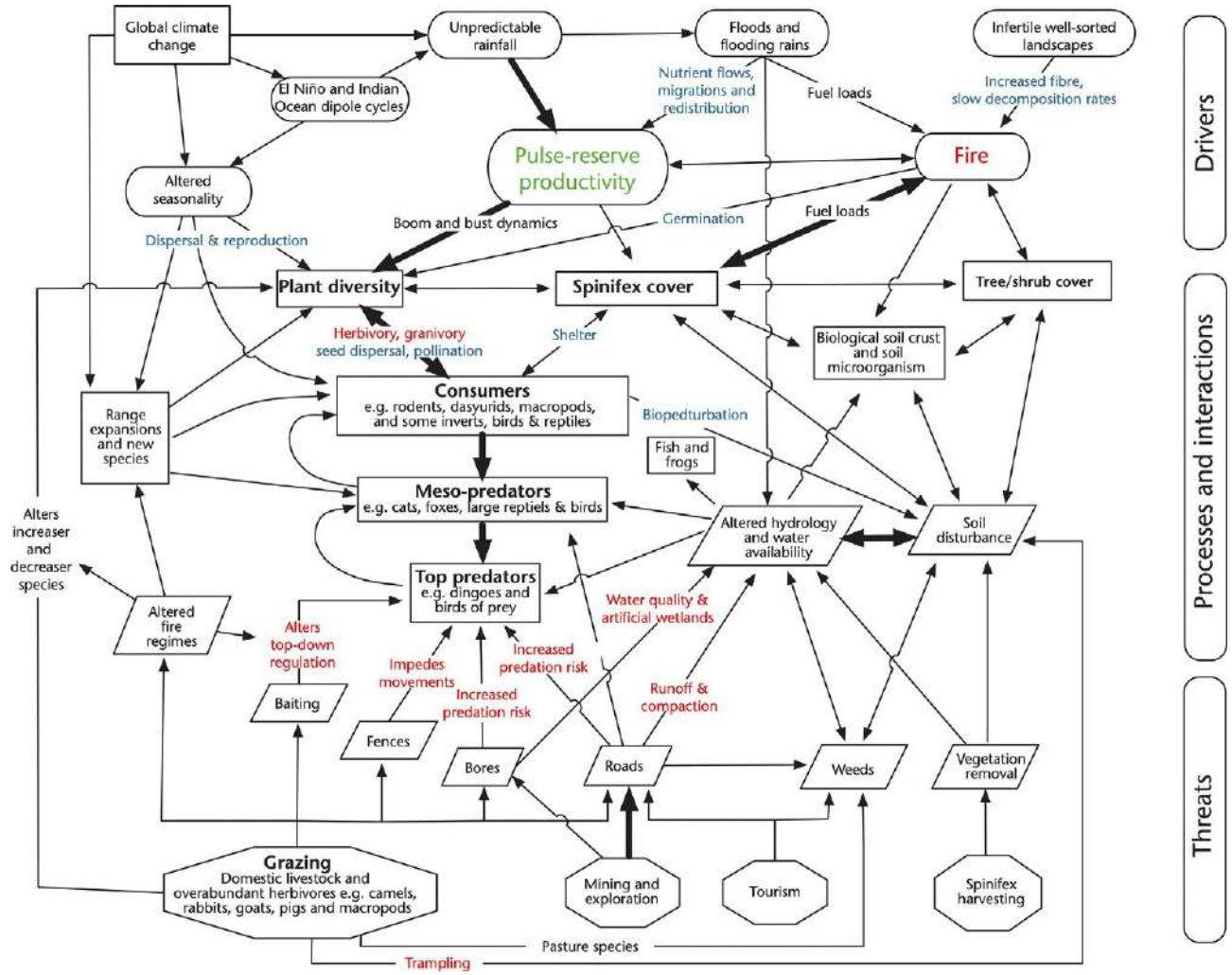

FIG. 5.7 Conceptual diagram depicting the drivers, threats, processes, and interactions in arid Australia. Abiotic drivers are shown in the top section of the figure, threats *(octagonal shapes)* and the flow-on effects of human activities *(parallelogram shapes)* are depicted in the lower section. The key drivers and threats vary in extent and magnitude throughout the Simpson Desert and in the dominant hummock grasslands and interspersed woodlands of the biome, and also vary depending on the phase of the boom-and-bust, or pulse-reserve, cycle driven by unpredictable rainfall and fire. Processes that have positive effects on biodiversity are shown in *blue type*; negative effects are shown in *red type face*. In some instances, two-way interactions and indirect effects are also possible but are not explicitly included. Some of the more dominant processes are shown in *bold arrows*. Climate change is a driver that, in association with its effects on other processes, looms as a major threat in future. *(Source: Dickman, C.R., Wardle, G.M., Foulkes, J., de Preu, N., 2014. Desert complex environments. In: Lindenmayer, D., Burns, E., Thurgate, N., Lowe, A. (Eds.), Biodiversity and Environmental Change: Monitoring, Challenges and Direction. CSIRO Publishing, Melbourne, pp. 379–438.)*

3.6.1 Rainfall and wildfire

Localized rainfalls and the transient pulses of productivity they produce can be detected by several species of mammals, birds, and probably some flying insects at distances of 10 km or more, prompting some animals to move in to exploit them. However, large rainfall events (e.g., >90th percentile), usually restricted to the summer months, can drive regional scale productivity pulses—germination of seeds and bulbs, vegetative growth, and reproduction—that last for many months and provide resources for herbivores and higher level consumers. Population booms occur synchronously across vast areas of the desert landscape for many, but not all, resident and migratory consumers (Pavey et al., 2008a; Pastro et al., 2013; Greenville et al., 2016a,b; Kwok et al., 2016; Gibb et al., 2019). Water flows provide dispersal opportunities for fish and some amphibians, while standing water that collects in claypans can break the dormancy of invertebrates such as the shield shrimp *Triops australiensis* and trigger rapid increases in their numbers. Animals that drown in flood waters may attract large numbers of mobile scavengers as water recedes, with corvids responding particularly quickly (Krige et al., 2024).

Wildfires may follow within 1–2 years of major rainfall events and remove above-ground cover of vegetation,

primarily hummock grass, over large areas (Fig. 5.3). The heat and/or smoke from the fires stimulates the germination of a wide range of herbs, grasses, and shrubs such as *Acacia* spp. and desert poplar trees *Codonocarpus cotinifolius*, especially if rainfall occurs postfire. Fires and subsequent rainfall are likely also to affect soil crusts, but research on this topic is scant in the Simpson Desert. The conditions postfire provide an open template for species that hunt on open sand such as the Eyre Basin beaked gecko *Rhynchoedura eyrensis*, central netted dragon *Ctenophorus nuchalis*, and hairy-footed dunnart *Sminthopsis hirtipes*. By contrast, species that prefer more shelter, such as military dragons *Ctenophorus isolepis*, little button-quails *Turnix velox*, and wongai ningauis *Ningaui ridei*, decline until vegetation cover increases (Letnic et al., 2004). Some research indicates that different vertebrate species prefer patches of vegetation, especially hummock grass, that are at different stages of recovery postfire, but such preferences appear labile and vary across different spatial scales of observation (Haythornthwaite and Dickman, 2006; Pastro et al., 2011, 2013).

Rainfall and wildfire are key drivers of interactions among native species in the Simpson Desert, but also provide opportunities for nonnative species to invade. We discuss these processes in more detail in Section 4.

3.6.2 Biotic interactions

Much theory and empirical evidence suggests that predation is a pervasive process shaping population and assemblage structure in hot desert environments (e.g., Polis, 1991a,b; Ayal, 2007). In the Simpson Desert, the outcomes of actual predation on the predator-prey interaction have been extensively studied, as have the effects of the risk of predation on prey. For example, some specialist predators such as the thorny devil lizard *Moloch horridus*, which is restricted to eating ants $\leq$1.5 mg (Withers and Dickman, 1995), occur in those parts of the environment where their requisite prey can be found. Other specialist predators such as the letter-winged kite *Elanus scriptus* and barn owl *Tyto alba*, which primarily consume rodents, expand their occupancy of the desert and reproduce primarily postrain when rodent populations have irrupted (Pavey et al., 2008a). The structure of lizard assemblages is related similarly to the availability of invertebrate prey (Pastro et al., 2013); different trophic groups of birds also invade the desert when their respective nectar, invertebrate or seed resources peak in availability (Tischler et al., 2013).

In terms of the effects of predators on prey, direct predation by native rodents on buried seeds selectively removes palatable seed species from the seed bank, creating different seed assemblages that are available to germinate at sites where rodents do, or do not, occur (Yip, 2023). In terms of predation risk, several species of small mammals have been shown to avoid odor-based cues of larger mammalian and reptilian predators, and avoid leaving their own odor cues by frequently shifting their burrows to reduce odor accumulation (Dickman, 2006; Dickman et al., 2022). Predation models incorporating both native and introduced predators highlight the importance of top-down forcing on prey assemblages, with differences in interaction strength occurring at different stages of boom-and-bust cycles (Greenville et al., 2017; Llewelyn et al., 2023).

Despite the apparent primacy of predation, other biotic interactions are also prevalent. Intense interference competition between the lesser hairy-footed dunnart and lycosid spiders often culminates in intraguild predation on the spiders (Potter et al., 2018), and diffuse competition from insectivorous dasyurid marsupials constrains the diets of nocturnal generalist lizards but not those of diurnal or dietary specialist lizards (Dickman et al., 2020). Competition for rodent prey has also been implicated between native letter-winged kites and introduced mammalian predators, particularly when population irruptions of rodents are beginning to subside and this shared prey resource is becoming more limited (Pavey et al., 2008b).

Mutualistic interactions are also important. Lichens, representing usually obligate associations between fungi and algae or cyanobacteria, occur mostly as thin crusts on hard substrates in the Simpson Desert (Williams and Bailey, 2009). By contrast, facultative associations occur in all desert habitats between flowering plants and animals that serve as vectors of their pollen and seeds. Working in *Triodia*-dominated grassland, Popic et al. (2013) described a flower-visitor network that comprised 50 native bee species and 52 plant species. These species interacted to form 168 different links, with 38% of these links resulting in the transport of pollen. Further work suggested possible pollination by birds of the regal birdflower *Crotalaria cunninghamii*, and raised the intriguing possibility that nectar-robbing by small rodents indirectly facilitated reproduction in this shrub by stimulating increased nectar flow or visits by pollinators (Popic et al., 2016). Dispersal of seeds of *Acacia* spp. has been documented by ants in the Simpson Desert (Davidson and Morton, 1984). Ingestion and passage of seeds through the gut of birds achieves dispersal and the breaking of seed dormancy, enabling germination soon after dispersal (Letnic et al., 2000). Facilitation interactions also occur frequently in the desert, with a wide range of animals using burrows created by digging vertebrates and invertebrates (Dickman, 2003). Prior to the loss of several mid-sized digging marsupials from the Simpson Desert (e.g., bandicoots, bilbies, bettongs; Table 5.3), facilitation interactions—and ecological interactions more broadly—were likely to have been still more pervasive than they are now.

4 Human activities and their impacts

Human activities in the Simpson Desert began with the utilization of natural resources by Aboriginal people. It is possible that activities such as patch burning of vegetation for clearing country or flushing game influenced the biota, but any effects are likely to have been rare or localized in this part of the arid zone (Kimber and Friedel, 2015). Other activities, such as fishing, hunting, and collecting plant material for food, medicine, or narcotics would also have had limited or transient effects on the biota. With the arrival of Europeans and their livestock in the late 19th century, however, fringing areas of the desert became more heavily impacted. Scarce local timber resources were depleted to construct homesteads, fences, and stockyards, and sites around natural water sources were degraded by herds of hard-hoofed cattle (Watson, 1998). Loss of these sites perhaps contributed to the extinction of mammals (Table 5.3) that would have relied upon them as refuges during periods of drought (Morton, 1990). In the 20th and early 21st centuries, the direct and indirect impacts of human activity have expanded to encompass all parts of the Simpson Desert, and the gathering threat of climate change looms as a major existential impact for the future.

4.1 Current threats

As noted in Section 2.3, cattle are grazed over large areas of the Simpson Desert. Although stocking rates are typically low, especially in sand dune habitats (1 beast per 250 ha), grazing is usually more intense in local areas near shelter and water, creating degraded "sacrifice zones" (Frank et al., 2012). Cattle disperse more widely after rain when palatable plants become available, and this can expand the area that is subject to trampling, soil disturbance, and depletion of sapling shrubs and trees (Fig. 5.7). Disturbance to the thin soil crust compromises the organisms that form the crustal communities and disrupts their ability to trap water, cycle nutrients, and reduce soil erosion, while reduced regeneration of perennial vegetation affects the structure of plant communities and habitat for the animals that depend on them (Morton et al., 2011). Cattle grazing has been able to expand throughout much of the Simpson Desert due to the sinking of bores to access water in the Great Artesian Basin. This infrastructure, and the track network that services it, provide opportunities for introduced species of animals and plants to move in and establish in the desert (James et al., 1999). Studies exploring the short-term effects of cattle removal in the Simpson Desert have demonstrated positive effects of removal on palatable plant species and on the abundance of flowers, small mammals, and red kangaroos, but have also identified interactions between habitats, timing of surveys, and differences in historical grazing pressure (Frank et al., 2014, 2016). Long-term monitoring is needed to tease out the separate and interactive effects of these factors.

Other current threats to the ecology and biodiversity of the Simpson Desert are associated, paradoxically, with key natural drivers of desert life: rainfall and fire. Extreme rainfall events drive pulses of primary productivity, recharge water sources, and provide opportunities for dispersal and reproduction for native biota (Wardle et al., 2013). However, these events also provide opportunities for introduced species, many of which have deleterious effects on biodiversity. For example, introduced plant species such as buffel grass, mesquite, and mimosa bush spread and establish readily after heavy rains, displacing native plant species and altering conditions for fauna (Dickman et al., 2014). These species are encroaching increasingly in moist areas, rainfall run-on sites, and around water bodies.

Introduced vertebrate species such as the feral pig *Sus scrofa* also move into the desert postrain; although localized, the rooting activities of pigs for food can disturb large areas around water sources. Other introduced vertebrates are always present in the desert and increase quickly following extreme rains. The house mouse *Mus musculus* and European rabbit *Oryctolagus cuniculus* rapidly exploit rain-driven pulses of primary productivity, while the European red fox and feral cat in turn exploit the smaller primary consumers, their populations peaking several months after those of their prey (Dickman et al., 2014). Predation by the fox and cat depletes the prey base that is shared with native predators (Pavey et al., 2008b) and drives populations of preferred native prey species to critically low levels (Spencer et al., 2017). Larger introduced vertebrates such as feral cattle, feral horses *Equus caballus* and one-humped camels *Camelus dromedarius* probably also benefit from the pulses of productivity resulting from extreme rainfall events. All three species pollute and degrade water sources during dry times (e.g., Brim-Box et al., 2010), but disperse more widely after rain. Camels are most widespread and abundant, and their selective browsing may kill or impede the recruitment of shrubs and trees such as long-leaf emubush *Eremophila longifolia* and whitewood *Atalaya hemiglauca* (authors, unpublished observations).

No introduced aquatic animals or plants are yet known in the Simpson Desert, but several species occur just outside in the broader Lake Eyre Basin (Kingsford, 2017) and could "wash in" following flooding rains. One of these species, the cane toad *Rhinella marina*, is now common in drainages to the east and north of the desert; as in other regions where this anuran has established, it would probably devastate populations of native snakes and varanids if it were to gain access to the water bodies and drainages of the Simpson Desert (Dickman et al., 2014).

If dry conditions prevail for 1–2 years after extreme rainfall, lightning strikes can ignite the dry plant biomass and hummock grass and lead to broadscale wildfires (Greenville et al., 2009). These events are key drivers of change in hummock grasslands and have been little affected by human activity (Wright et al., 2021). However, wildfires now interact with other factors such that their occurrence can now pose threats to desert biota. For example, fire facilitates the spread of introduced fire-tolerant plants such as buffel grass at the expense of native species; buffel in turn forms dense monocultures that can lead to more intense and more frequent fires (Schlesinger et al., 2013). The recovery of native vegetation postfire can also be slowed by the presence of introduced herbivores, and populations of small vertebrates can be impacted because recently burnt areas provide open hunting grounds for feral cats, European red foxes, and dingoes (Letnic et al., 2004; Pastro, 2011). Feral cats and foxes are particularly potent threats for native rodents (Greenville et al., 2017).

4.2 Future threats

The remoteness of the Simpson Desert and its unsuitability for agriculture and permanent human habitation mean that it is unlikely to face major new developments in future. It is possible that mining interest will intensify, as is predicted for the surrounding regions of the Lake Eyre Basin (Mudd, 2017), with concomitant increases in road and infrastructure construction and demand for water. However, such changes (if they occur) are likely to be confined to local areas that are relatively accessible on the desert's fringes. A similar appraisal can be made for potential increases in tourism. By contrast, climate change is likely to have pervasive effects across the entire desert landscape, especially with respect to increases in the frequency and severity of extreme events (Wardle et al., 2013).

Weather records from sites immediately surrounding the Simpson Desert show that median (50th quantile) and extreme (5th, 10th, 90th, and 95th quantile) annual minimum and maximum temperatures have increased at local and regional scales from the late 19th to early 21st centuries (Greenville et al., 2012; Fig. 5.8). Rainfall changes were less consistent over the same period, with individual sites showing increases in annual rainfall, increased frequency of large rainfall events or deeper droughts (Greenville et al., 2012). These past changes align broadly with predictions for future climate change, with increases in minimum and maximum temperatures projected up to 2090, together with increases in the intensity of extreme rainfall events and also of extreme droughts (Watterson et al., 2015). Such changes imply shifts in associated hydrological features such as soil moisture and runoff, and also have implications for changes in humidity and the size and intensity of wildfires. Although the magnitude of anticipated changes varies between modeled scenarios, the effects of all projected changes on biota will likely be profound.

Taking account of projected climate changes, Smyth et al. (2012) used bioclimatic modeling to estimate the current geographical distributions of major vegetation groups and then compared these with projected distributions for these groups in 2070 assuming either medium or high CO_2 emissions. While hummock grassland was expected to remain substantially intact (−0.6% to 3.5% change in area), several other major vegetation communities were projected to decline in area, with some, such as mallee woodlands and shrublands, declining by as much as 31%–78%. In further analyses of the Georgina gidgee ecosystem, which is embedded as a network of woodland patches within hummock grassland primarily in the northern parts of the desert, Wardle et al. (2015) concluded that the ecosystem should be considered vulnerable and faced a >30% decline in suitable habitat in eight modeled scenarios up to 2050. Notably, this latter study considered threats from overgrazing, weed encroachment and changed fire regimes in making its assessment. The Georgina gidgee woodland is associated with a high diversity of grasses, herbs, forbs, and shrubs and provides critical foraging or shelter habitat for many animal species (Wardle et al., 2015). Like other major vegetation associations, the decline or even collapse of the Georgina gidgee woodland ecosystem would seriously impoverish the biotic and ecological diversity of the Simpson Desert.

Animal responses to climate change are likely to vary substantially. Provided that sheltered refuge areas such as water holes and riparian corridors remain in future, species that depend on these features should persist. Species that use dormancy may fare relatively well, and mobile species could be expected to continue to visit the desert when conditions are suitable. However, animals that remain active in situ may fare less well. For example, Conradie et al. (2020) predicted a marked increase in extreme heat risks for small (10–42 g) and relatively sedentary birds by the year 2100, with species such as the zebra finch likely to experience 20–100 days per year of acute lethal hyperthermia and dehydration risk over different parts of its arid range. Biber et al. (2023) also predicted substantial decreases in Australian reptile species richness due to future temperature extremes, especially in the continent's arid regions. Although some reptiles such as geckoes are nocturnal and may escape daytime heat extremes if they can access sufficient shelter, they will still be exposed to much higher nighttime temperatures and may face shortened foraging opportunities as a consequence (Roberts, 1998). Small mammals and many nocturnal invertebrates are likely to encounter similar challenges.

Climate modeling predicts reductions in species richness for many taxon groups in the Simpson Desert

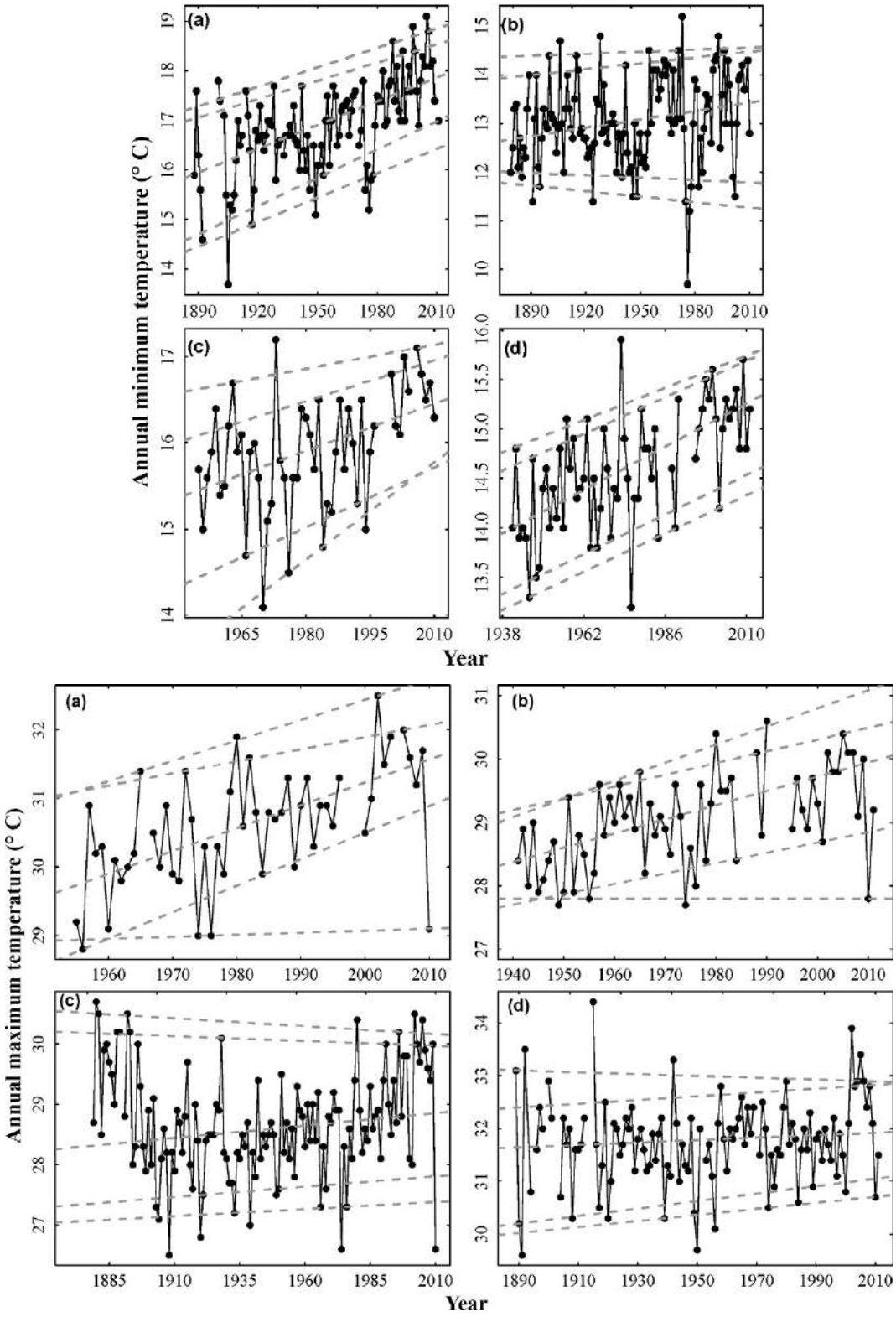

FIG. 5.8 Top: Mean minimum annual temperatures (°C) from weather stations at (A) Boulia, (B) Alice Springs, (C) Birdsville, and (D) Oodnadatta. Bottom: Mean maximum annual temperatures (°C) from weather stations at (A) Birdsville, (B) Oodnadatta, (C), Alice Springs, and (D) Boulia. All weather stations are located on the edge of the Simpson Desert, central Australia. *Broken gray lines* show 95th, 90th, 50th, 10th, and 5th quantiles. *(Source: Greenville, A.C., Wardle, G.M., Dickman, C.R., 2012. Extreme climatic events drive mammal irruptions: regression analysis of 100-year trends in desert rainfall and temperature. Ecol. Evol. 2, 2645–2658. https://doi.org/10.1002/ece3.377.)*

and other arid regions of Australia (Smyth et al., 2012), but modeling often ignores the effects of removing (or adding) species and how climate change will affect the interactions of species with each other and with the environment (Fig. 5.9A). Extreme rainfall events may facilitate ingress of more weed and feral animal species, and wildfires may become more extensive and severe, increasing the impacts of native and introduced predators. Some effects may emerge unexpectedly. For example, native varanid predators—notably the sand goanna *Varanus gouldii*—are usually inactive over winter, and many small mammals then occupy shallow burrows that are energetically inexpensive to dig. These species, as well as burrow-dwelling geckoes and other reptiles, are likely to face greatly increased risks

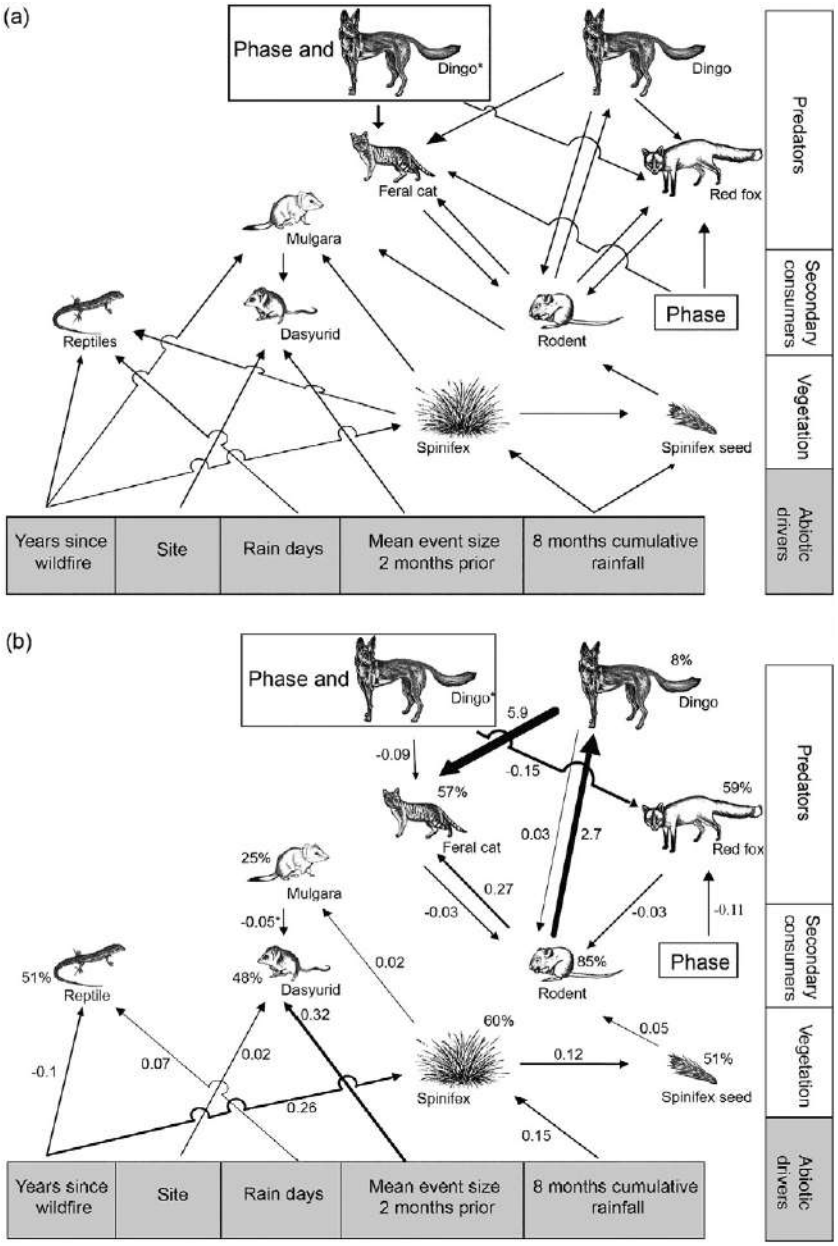

FIG. 5.9 (A) Conceptual model showing putative interactions between abiotic drivers, selected species, species groups, and trophic levels derived from long-term research in the Simpson Desert, central Australia. (B) Results from structural equation modeling quantifying the interactions. *Values* above drawings are percentage deviances explained. Standardized path coefficients are shown for each *arrow*, and *arrow thickness* is proportional to effect size. Nonsignificant interactions are not shown. *Interaction term between dingo numbers and boom or bust population phase of rodent prey. Artwork by A. Foster. *(Source: Greenville, A.C., Wardle, G.M., Dickman, C.R., 2017. Desert mammal populations are limited by introduced predators rather than future climate change. R. Soc. Open Sci. 4, 170384. https://doi.org/10.1098/rsos.170384.)*

of predation from goannas if these predators remain active as minimum winter temperatures increase (authors, unpublished). Using long-term (17–22 year) datasets on vegetation, vertebrates, rainfall, and wildfire from the Simpson Desert, Greenville et al. (2017) used structural equation models to quantify the strength of linkages between these ecosystem components (Fig. 5.9B). The models were then used to predict how climate-driven changes in rainfall and wildfire might influence the cover and productivity of hummock grass over a 100-year timeframe, and how this and the impacts of introduced predators would affect populations of native rodents. The results showed, intriguingly, that the dominant effect on the rodents was top-down suppression from feral cats and European red foxes (Fig. 5.10). More data-based models of this kind will be essential to forecast when and where management should be focused to retain desert biodiversity.

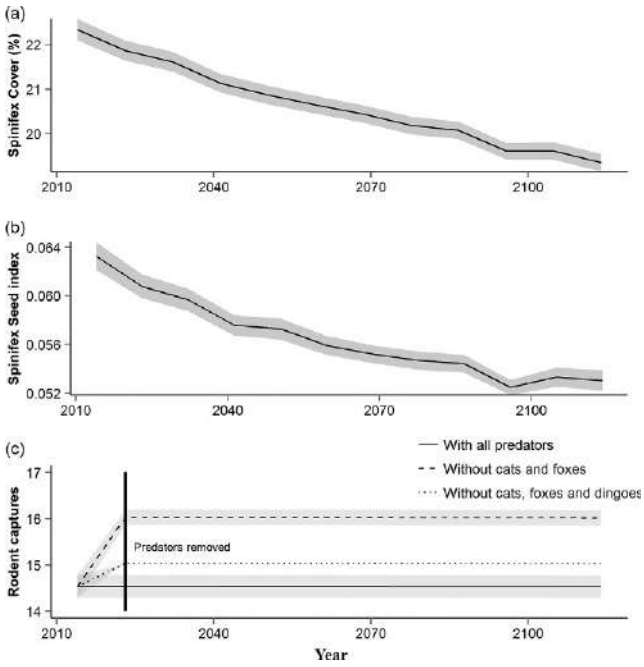

FIG. 5.10 Responses of spinifex (hummock grass) and rodents to simulated changes in rainfall and wildfire as a result of climate change over the next 100 years. Projected results (±s.e.) are shown for changes in (A) percentage spinifex cover, (B) spinifex seed index (a measure of reproductive output), and (C) captures of native rodents (standardized per 100 trap-nights per 1 ha sampling grid) with all mammalian predators present, introduced predators (feral cats and European red foxes) removed and all mammalian predators removed, under a changing rainfall and wildfire scenario. Despite projected declines in spinifex, rodent numbers were not affected by the decrease in this key food and cover resource, but rather by the presence of introduced predators. *(Source: Greenville, A.C., Wardle, G.M., Dickman, C.R., 2017. Desert mammal populations are limited by introduced predators rather than future climate change. R. Soc. Open Sci. 4, 170384. https://doi.org/10.1098/rsos.170384.)*

5 Operational and potential monitoring tools to conserve or recover the biome

In view of the extraordinary biodiversity of the Simpson Desert and the range of threats that it faces now and in the future, it is important to ensure that appropriate monitoring is established and interventions implemented so that conservation outcomes can be achieved. We have previously proposed fit-for-purpose monitoring regimes for the desert (Dickman and Wardle, 2012; Dickman et al., 2014, 2018), and summarize these briefly here.

In the first instance, ground-level monitoring is essential to track species in time and space. On the one hand, surveillance monitoring is needed to detect incursions of invasive plant and animal species so that control measures can be implemented in a timely fashion. For species such as buffel grass, cane toads, or feral pigs, incursions are most likely to follow heavy rainfall events and are likely to be focused along water courses or run-on areas in the landscape. Regular on-ground inspections should be conducted at likely incursion sites post rain. Alternatively, or additionally, remote detection methods such as camera traps, phenocams, or eDNA sampling can be used to check for incursions over more extended periods. Surveillance methods need to be targeted to ensure that potential invaders have a high likelihood of being detected. For example, fire ants *Solenopsis invicta* are spreading in Australia currently; they are most likely to be detected by the presence of their distinctive nests (National Fire Ant Eradication Program, 2024).

On the other hand, long-term monitoring should be carried out in the major habitats of the desert to detect trends in the biota. Locations of sampling sites should be stratified by habitat, be sufficient to detect real trends, and use methods that are appropriate for different taxon groups. Examples of such methods include, inter alia, quadrat-based surveys for vegetation, litter, and pitfall or sweep sampling for invertebrates, nets for fish and aquatic insects, and live-trapping, remote cameras, and audio devices for tracking vertebrates (Dickman et al., 2014). Long-term monitoring sites should also measure key environmental drivers of desert life such as rainfall, temperature, and the incidence of wildfires to help interpret any changes in the biota that are observed. Satellite-borne sensors can be used to track changes in vegetation attributes such as primary productivity at high resolution (1–5 m) over regional spatial scales, and can also be used effectively to map fires and floods. Although such remote sensing cannot track changes in individual species or communities, it provides the only means of monitoring "big picture" events as they develop across the desert landscape.

If surveillance monitoring reveals an incursion by an undesirable species or long-term monitoring uncovers worrying trends in target species numbers (e.g., populations fall

below predetermined thresholds), management intervention is likely to be needed. For invasive species, this may involve eradicating or containing the newly arrived organisms. On properties in the north-eastern Simpson Desert that are managed by the nongovernment organization BHA, for example, surveillance has detected early incursions of buffel grass that are now being controlled (Radford et al., 2012). For declining native species, some caution—based on knowledge of the species' life histories and biology—is advisable before management actions are imposed. This is because populations of many desert taxa decline precipitously and naturally after peaking during boom periods; for example, desert frogs and many plant species are detectable only in the soil during bust periods and can seem, without awareness of this quiescent life history stage, to be absent (Silcock et al., 2011). If "true" declines are detected that are outside expected norms, further investigation and, if necessary, intervention may be needed. Following identification of introduced mammalian predators as a key threat to small native prey species, for example, BHA has instituted a program of regular culling to mitigate this threat. A further critical function of monitoring is to determine whether conservation management interventions have the effect intended; management actions may not always have the effects that are desired, and monitoring is essential so that success or failure can be identified and management actions adapted, if needed (Dickman et al., 2018).

Successful monitoring that allows conservation objectives to be realized requires that data be maintained safely and securely, cleaned and made available for use soon after it has been collected in the field. Resources must be sufficient to deploy well-trained personnel at all stages so that reliable information is available to inform management decisions. Adequate resourcing will be particularly important as the climate changes further and arid systems experience more frequent and more severe extreme events. Unfortunately, monitoring of biota and environmental drivers in the Simpson Desert is currently very limited and woefully underfunded. The desert is remote, lacking in infrastructure and is virtually unpopulated, and this—with the exception of the BHA reserves of Ethabuka and Pilungah—provides significant impediments to biodiversity monitoring for conservation and management agencies that are headquartered in distant towns and cities. Improvements in monitoring capacity will occur only with a vastly increased allocation of resources.

6 Conclusions

The Simpson Desert boasts an extraordinary level of biodiversity, with numbers of species in some groups such as reptiles and insectivorous mammals far exceeding those found in other world deserts. Although Australia's arid regions are relatively recent, the biota have varied evolutionary origins and different biogeographical histories, with several plant and mammal species that have been introduced recently by people. Deep knowledge of the desert environment forged by Aboriginal people has been augmented in recent decades by insights derived from Western science. Collectively, the Simpson Desert is understood to be a boom-and-bust environment for most taxa that is driven by episodic heavy rainfall events, subsequent periods of rainfall deficit, and wildfire. The biotic and ecological diversity of the desert, while still remarkably high, has diminished over the last century or more following the arrival of European settlers, their livestock, and other introduced species. These changes saw the demise of Traditional culture, the local or regional extinction of at least 14 species of native mammals, and have left an enduring legacy of threats that place much of the present biota at risk. The most serious threat to desert life in future decades will come from climate change, notably an increase in the severity of rainfall, drought, and wildfire events and interactions of these drivers with other processes. The remoteness of the Simpson Desert from population centers poses logistic and practical difficulties for monitoring and managing the biota and mitigating the threats they face. However, such monitoring could occur with increased allocation of resources, and will be essential if the desert's living riches are to be conserved as anthropogenic climate changes become more severe.

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Chapter 6

The Indian Thar Desert

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1 Introduction

The Thar Desert, also known as the Great Indian Desert, is a vast arid region located in the northwestern part of India. The name Thar refers to the sand hills accumulated by the prevailing wilds. The term Desert means "region of death" in Hindi, while the term *marusthal* derives from the Sanskrit words "meri" (to die) and "sthala" (place that is desert or dry country—Sharma and Mehra, 2009). Its major portion is found in the state of Rajasthan in northwestern India, and partially in Punjab and Sindh province in eastern Pakistan. The Aravalli range of mountains divide Rajasthan unequally into two parts with a major portion covered by The Thar Desert in the northwest (Goyal et al., 2014). The Thar Desert's climate is classified as a hot desert environment, characterized by unusually high summer temperature and large temperature differences between day and night (Dhir and Singhvi, 2012). Summer temperatures frequently exceed ~45°C (113°F) throughout the months of April and June. The area receives little rain, with an average annual precipitation ranging from 100 to 500 mm (4 to 20 in.), with the majority raining during the monsoon season, which continues from July to September. However, rainfall in the region is highly unpredictable and unreliable, resulting in frequent drought situations (Singhvi and Krishnan, 2014). Winters in the Thar Desert are mild, with temperatures falling to ~10°C (50°F) during the day and near-freezing at night. During this season, the region perceives cold winds, which add to the area's overall dryness. Hence, despite its harsh environment, this region is home to a distinct ecology as well as rare flora and wildlife. The desert vegetation consists mostly of resistant species that have adapted to thrive in arid conditions, such as thorny shrubs, cacti, and grasses (Akhtar and McMichael, 1996; Sheikh et al., 2015).

The Thar Desert in the state of Rajasthan is classified into four primary zones based on rainfall and edaphic characteristics:

- Agriculture-rich zone: It includes the districts of Sri Ganganagar and Hanumangarh, which share borders with the states of Punjab and Haryana.
- The northern drainage zone: It covers the districts of Sikar, Jhunjhunu, and Northern Nagaur.
- The Luni basin: It includes the districts of Jalore, Pali, the southeastern part of Barmer, the eastern part of Jodhpur, the western part of Ajmer, Sirohi, and the southern part of Nagaur district.
- The true desert: It encompasses the entire district of Jaisalmer, northern Barmer, the western part of Jodhpur, Churu, and Bikaner districts.

Furthermore, prior researchers (Saxena, 1972; Bhandari, 1990; Prakash, 1994) also identified four major types of ecosystems based on vegetation and topography in the overall Thar Desert that are as follows:

- Sandy habitats: It occupies a large portion of the Thar Desert, and grasses are the dominant flora in this area. Sand dunes are typically separated from one another by sandy plains and low barren hills. These dunes change shape and size as they move in the wind direction (Fig. 6.1).
- Hills and rocky outcrops: This area is located near the boundary of the Aravalli Mountain range, and contains hills and rocky outcrops spread throughout the region. In this region, low barren hills known as "Bhakars." On the flat surfaces of these hills, a scrub community coexists with xerophytic plant groups.
- Wetlands: Ephemeral rivers, lakes, canals, tanks, and reservoirs are part of this ecosystem. The construction

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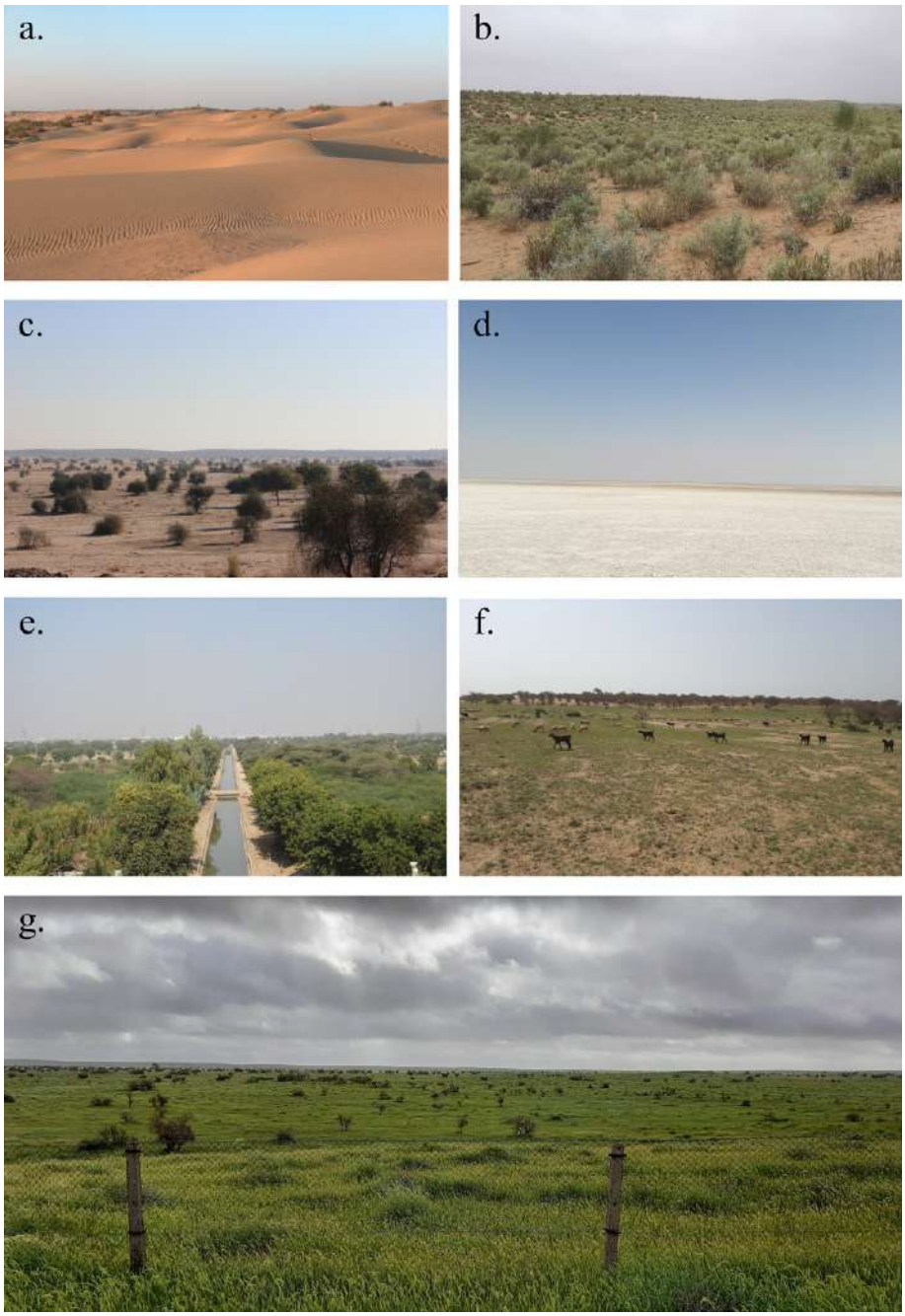

FIG. 6.1 Different habitats and landscapes of the Thar Desert: (A) Sand dunes; (B) scrub land; (C) dry land with scattered trees; (D) salt lakes; (E) Indira Gandhi Canal; (F) grazing activity at grassland; and (G) grassland during monsoon. *(Credit: Images A–D—Rishikesh Tripathi; image E—Pratap Singh.)*

of the Indira Gandhi Canal in the Thar Desert created a plethora of new habitats in the form of tiny ponds and swampy areas (Fig. 6.1).

- Ruderal habitats: The term "ruderal" is commonly used to describe the ecological relationship of plants around human habitations (Blatter and Hallberg, 1918–1921). This habitat can be found in a range of landscapes throughout the Desert, including rocky outcrops, sand dunes, sandy plains, river banks, and others, as well as scattered hamlet complexes.

Some of the best grasslands can be found in the districts of Bikaner, Barmer, Jaisalmer, and along the Indo-Pakistan border, where there is little human population and grazing pressure. However, following the implementation of the Indira Gandhi Canal Project (IGCP), water supply

increased in many of these locations due to the building of water channels (Islam and Rahmani, 2011). The majority of grasslands are dry open or mixed with trees and prickly plants (Islam and Rahmani, 2008, 2011). Based on species presence and their dominance, these grasslands are usually called *Dichanthium-Lasiurus-Cenchrus* type (Dabadghao and Shankarnarayan, 1973).

Despite its hard climate and arid environment, the Thar Desert has a rich and vibrant culture and known as one of the most populated deserts with very high human population and cattle heads. The desert is home to a number of indigenous people, including the Rajputs, Bhils, Raika, Gadouliya Lohar, Gypsy, Rabaris, Garasians, and Bishnois, who have established distinct lives and traditions centered on the desert environment (Gaur and Gaur, 2004). The Thar Desert's population relies on animal husbandry, agriculture (where water is accessible), and trade to make a living. The Thar Desert's economy is based on irrigation. Between June and October, Kharif crops are mostly grown in agricultural areas. Bajra *Pennisetum glaucum*, jowar *Sorghum bicolor*, maize *Zea mays*, sesame *Sesamum indicum* and pulses such as guar *Cyamopsis tetragonoloba*, and groundnut *Arachis hypogaea* are important crops (Dhir et al., 2018).

In recent years, attempts have been made to capitalize on the Thar Desert's renewable energy potential. Because of its ample sunlight, the region is seen to be perfect for solar energy generation, and various solar power projects have been developed in the area, adding to India's renewable energy goals (Singh et al., 2005; Bhandari, 2021). One of the largest Solar Park in the world as of 2023, is Bhadla Solar Park situated in The Thar Desert covering 56 km^2 area with an installed capacity of 2245 MW (The Hindu, 2021). Wind energy is also one of the major forms of renewable energy source, which in The Thar Desert, is used to generate electricity. Besides the Jaisalmer Wind Park situated at Jaisalmer district, Rajasthan is also one of the world's largest onshore wind parks in terms of electricity generation. Recently, the world's largest 600 MW Wind-Solar Hybrid Power Plant had also been commissioned at Jaisalmer, India (India Infrahub, 2022).

2 Geographic location

The Indian Desert formed as a result of geotectonic and climatic changes during the Himalayan upheaval and subsequent glacial epochs (Gupta and Prakash, 1975). It (area 285,000 km^2) is located in the arid western region of India's states of Punjab, Haryana, Rajasthan, and Gujarat (Fig. 6.2) and accounts for ~9% of the country's total geographical area. Geographically, it is defined by the Aravalli hill ranges in the east, the irrigated Indus plain in the west, the plains of Haryana and Punjab in the northeast, and the Rann of Kutch in the south. Its majority (70%) part is in the state of Rajasthan, with 23% in Gujarat and 7% in Punjab and Haryana. It has a varied scenery that includes sand dunes, rocky terrain, salt flats, and limited flora. The region has a dearth of surface water, with the lone Luni River flowing through the desert and joining the Arabian Sea through Gujarat's Rann of Kutch. The majority of the desert's surface is made up of sand, silts, and loess-like

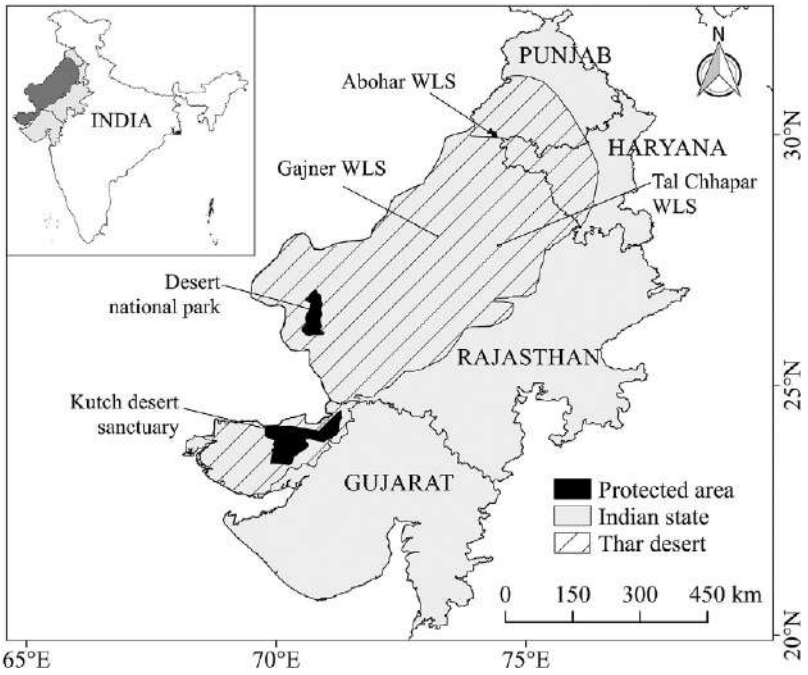

FIG. 6.2 Location and extension of the Thar Desert in four northwestern Indian states. Major protected areas, i.e., national park and wildlife sanctuary (WLS), in the Thar Desert are depicted in *black color*. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Dr. Vijay Kumar Koli.)*

material. Wind velocity is found to be related to temperature fluctuations, with greatest velocity recorded (mean 203 km/h) during summer and minimum velocity recorded (mean 5–6 km/h) during winter (Climate and Weather in Thar Desert, 2022).

The entire zone has been sculpted into a variety of morphological forms over a long time of complex interaction between parental rock formations and numerous exogenic and endogenic processes. In this zone, four primary physical provinces have been identified: (1) the Thar Desert, (2) the Great Rann of Kachchh and Little Rann, (3) mainland Kachchh, and (4) the Saurashtra uplands (Kar, 1995). The latter two zones are in contact with the Arabian Sea and the Gulf of Kachchh.

3 Biodiversity

The term "biodiversity" was initially coined to describe species richness and diversity. It is also used to define the variation among living organisms in all environments where they exist, which includes diversity within species, diversity between species, and biodiversity among ecosystems. The Thar Desert is home to a wide range of floral and faunal species. The region is estimated to have more than 2000 different types of fauna and 775 species of flora that have adapted to the harsh climate. Sections 3.1 and 3.2 provide a comprehensive overview of the diversity of flora and fauna in this region.

3.1 Floral biodiversity

The natural flora of Thar Desert consists of ~775 species, which are spread out among 384 genera and 90 families. A significant number of these species are crucial for maintaining climate stability in the region and are endemic to the area (Bhandari and Holm-Nielsen, 1979). Specifically, 23 species and 14 subspecies are endemic, accounting for 6.4% of the total recorded species. It is higher than the degree of endemism in the renowned Sahara Desert (Bhandari, 1991; Roy and Roy, 2019). However, the introduction of alien species has serious ecological and economic impacts, resulting in the extinction of species from different ecosystems (Islam and Rahmani, 2011). Recently, Charan and Sharma (2016) conducted a floristic survey of the Thar Desert and reported the highest prevalence of herbaceous vegetation (60.10%), followed by shrubs (16.26%), trees (14.29%), and climbers (9.36%). An updated list of the prominent plant species with their distribution, conservation status, and utilization, is provided in Table 6.1. Some important Thar Desert plants are also shown in Fig. 6.3.

Sand dunes and interdunal flats occupy the maximum number of recorded taxa in the Indian Desert, ranging from annual climbers and grasses to perennial shrubs and trees. More than 45 plant species from 23 families dominate the sand dunes and interdunal flats. Fabaceae and Poaceae were the most prevalent at the generic level, with five and seven genera, respectively. Ephemeral vegetation is observed during the rainy season. These habitats support a considerable number of species with local and commercial importance, including *Balanites aegyptiaca* (desert date), *Cenchrus ciliaris* (dhaman grass), *Lasiurus scindicus* (Sewan grass), *Prosopis cineraria* (Khejari), *Salvadora persica* (toothbrush tree), and some species of genus *Acacia* and *Ziziphus*.

A vast area of the Thar Desert is covered by saline depressions called "Playas or Salt flats," where the salinity level is exceptionally high. These habitats are characterized by the presence of halophyte species, which have evolved distinct morphological and physiological features to survive in highly saline environments (Joshi et al., 2020a). A total of 25 halophytic species belonging to 17 genera and 11 families are known to grow dominantly in the Thar Desert. At the generic level, Poaceae was the most prevalent with genus like *Aeluropus*, *Dactyloctenium*, and *Sporobolus*. Amaranthaceae was the second-most dominant family with four species of the genus *Suaeda*, including *Suaeda fruticosa*, *Suaeda maritima*, *Suaeda monoica*, and *Suaeda nudiflora*. These salt flats are inhabited by several tree species, including *Tamarix dioica*, *Tamarix indica*, and *Salvadora oleoides*. The vegetation is commonly used by local communities for animal feed, human food, and nutrition purpose. Some species, such as *Juncus maritimus*, *Salsola baryosma*, *Sporobolus marginatus*, and *S. fruticosa*, can be used to produce sustainable biofuels and pharmaceutical products. Additionally, *Sesuvium portulacastrum*, *Cressa cretica*, and *S. monoica* are promising halophyte candidates for the phytoremediation of saline and metal-polluted soil and water (Joshi et al., 2020a,b).

The eastern border of the Thar Desert abuts the Aravalli Mountain Range. The Desert also features a few discontinuous hills and rocky outcrops called "Magra" situated near Jodhpur, Sewana (in Jalore District), Tembda (in Jaisalmer District), Chohtan, and Kiradu (in Barmer District). A total of 25 species belonging to 23 genera and 17 families grow dominantly in the hilly and rocky outcrops of the Thar Desert. The highest prevalence was reported for Fabaceae (four species), followed by Euphorbiaceae, Poaceae (three species each), and Acanthaceae (two species). This region harbors some of the very important medicinal plants such as *Commiphora*, *Sida*, *Tinospora*, *Bacopa*, *Withania*, and *Phyllanthus*. These genera provide raw ingredients for industrial products, such as gums, tannins, oils, fiber, and other pharmaceutically active compounds.

The Thar Desert has limited water resources. The IGCP and Narmada Canal Project (NCP) serve as the primary water supply systems for most of the region. Some lakes,

TABLE 6.1 Dominant flora of Thar Desert, their conservation status and utilization.

Type of edaphology	Name of species	Family	Habitat	IUCN status	Common uses	Reference
Sand dunes and interdunal flats	*Acacia jacquemontii*	Fabaceae	Shrub	NE	Animal fodder, firewood	Ramawat and Arora (2021), Chlachula (2021), Charan and Sharma (2016), Khan and Frost (2001)
	Acacia senegal	Fabaceae	Tree	NE	Forage, gam arabic, wood	
	Acacia nilotica	Fabaceae	Tree	NE	Fodder, firewood, gam arabic	
	Acacia tortilis	Fabaceae	Tree	LC	Animal fodder, furniture	
	Aerva lantana	Amaranthaceae	Herb	NE	Food, forage, folk medicine	
	Aerva javanica	Amaranthaceae	Herb	NE	Soil binder, fodder, fuel	
	Aerva pseudotomentosa	Amaranthaceae	Herb	NE	Forage, medicine	
	Arnebia hispidissima	Boraginaceae	Herb	NE	Food, immune tonic, fever	
	Asparagus racemosus	Asparagaceae	Climber	NE	Food, pharmaceutical	
	Azadirachta indica	Meliaceae	Tree	NE	Animal fodder, pharmaceutical	
	Balanites aegyptiaca	Zygophyllaceae	Tree	NE	Food, folk medicine, firewood	
	Boerhavia diffusa	Nyctaginaceae	Herb	NE	Food, fodder, folk medicine	
	Calligonum polygonoides	Polygonaceae	Shrub	NE	Food, fodder, charcoal	
	Capparis decidua	Capparaceae	Shrub	NE	Food, folk medicine, gardening	
	Cenchrus ciliaris	Poaceae	Grass	LC	Animal forage	
	Cenchrus setigerus	Poaceae	Grass	NE	Animal fodder	
	Citrullus colocynthis	Cucurbitaceae	Climber	NE	Food, fodder, biofuel	
	Citrullus fistulosus	Cucurbitaceae	Climber	NE	Food, fodder	
	Convolvulus deserti	Convolvulaceae	Shrub	NE	Folk medicine, pharmaceutical	

Continued

TABLE 6.1 Dominant flora of Thar Desert, their conservation status and utilization—cont'd

Type of edaphology	Name of species	Family	Habitat	IUCN status	Common uses	Reference
	Crotalaria burhia	Fabaceae	Herb	NE	Soil binder, sheds and shelters	
	Cucumis callosus	Cucurbitaceae	Climber	NE	Food, fodder, folk medicine	
	Cucumis prophetarum	Cucurbitaceae	Climber	NE	Food, folk medicine	
	Cyperus rotundus	Cyperaceae	Herb	LC	Food, folk medicine	
	Dactyloctenium sindicum	Poaceae	Grass	NE	Fuel, fodder, soil stabilization	
	Eragrostis ciliaris	Poaceae	Grass	NE	Food, fodder	
	Eragrostis minor	Poaceae	Grass	NE	Food, fodder	
	Evolvulus alsinoides	Convolvulaceae	Herb	NE	Folk medicine, gardening	
	Fagonia cretica	Zygophyllaceae	Herb	NE	Food, fodder	
	Farsetia hamiltonii	Brassicaceae	Herb	NE	Folk medicine	
	Gisekia pharnaceoides	Gisekiaceae	Herb	NE	Food, folk medicine	
	Indigofera cordifolia	Fabaceae	Herb	LC	Food, folk medicine	
	Justicia simplex	Acanthaceae	Herb	NE	Folk medicine, pharmaceutical	
	Lasiurus scindicus	Poaceae	Grass	NE	Food, animal forage	
	Leptadenia pyrotechnica	Asclepiadaceae	Shrub	NE	Food, fodder, folk medicine	
	Mollugo cerviana	Molluginaceae	Herb	NE	Fodder, folk medicine	
	Mukia maderaspatana	Cucurbitaceae	Climber	NE	Animal fodder	
	Octhocloa compressa	Poaceae	Herb	NE	Animal fodder	
	Oligochaeta ramosa	Asteraceae	Herb	NE	Fodder, folk medicine	
	Panicum antidotale	Poaceae	Grass	NE	Forage, folk medicine	
	Pedalium murex	Pedaliaceae	Herb	NE	Forage, folk medicine	

	Pergularia daemia	Apocynaceae	Climber	NE	Fodder, folk medicine	
	Prosopis cineraria	Fabaceae	Tree	NE	Food, fodder, folk medicine	
	Pulicaria crispa	Asteraceae	Herb	NE	Folk medicine	
	Saccharum spontaneum	Poaceae	Grass	LC	Fodder, shelters, folk medicine	
	Salvadora persica	Salvadoraceae	Tree	NE	Food, folk medicine	
	Tecomella undulata	Bignoniaceae	Tree	NE	Fodder, wood, folk medicine	
	Tephrosia purpurea	Fabaceae	Herb	NE	Fodder, folk medicine	
	Tribulus terrestris	Zygophyllaceae	Herb	NE	Fodder, folk medicine	
	Ziziphus mauritiana	Rhamnaceae	Shrub	NE	Food, fodder, folk medicine	
	Ziziphus nummularia	Rhamnaceae	Shrub	NE	Food, fodder, folk medicine	
Saline flats or playa	*Aeluropus lagopoides*	Poaceae	Grass	NE	Fodder, folk medicine	Joshi et al. (2020b), Sharma et al. (2017), Arora and Ramawat (2013)
	Chloris virgata	Poaceae	Grass	NE	Food, fodder	
	Cressa cretica	Convolvulaceae	Herb	LC	Fodder, folk medicine	
	Dactyloctenium aegyptium	Poaceae	Grass	NE	Fodder, folk medicine	
	Desmostachya bipinnata	Poaceae	Grass	NE	Fodder, folk medicine	
	Haloxylon recurvum	Amaranthaceae	Shrub	NE	Fodder, folk medicine	
	Heliotropium curassavicum	Boraginaceae	Herb	NE	Fodder, folk medicine	
	Juncus maritimus	Juncaceae	Herb	NE	Fodder, folk medicine, biofuel	
	Portulaca oleracea	Portulacaceae	Herb	NE	Food, folk medicine, biofuel	

Continued

TABLE 6.1 Dominant flora of Thar Desert, their conservation status and utilization—cont'd

Type of edaphology	Name of species	Family	Habitat	IUCN status	Common uses	Reference
	Salsola baryosma	Amaranthaceae	Shrub	NE	Fodder, folk medicine, biofuel	
	Salvadora oleoides	Salvadoraceae	Tree	NE	Food, folk medicine, biofuel	
	Sesuvium portulacastrum	Aizoaceae	Herb	NE	Biofuel, phytoremediation	
	Sporobolus coromandelianus	Poaceae	Grass	NE	Fodder, folk medicine, biofuel	
	Sporobolus marginatus	Poaceae	Grass	NE	Fodder, folk medicine, biofuel	
	Suaeda fruticosa	Amaranthaceae	Under shrub	NE	Food, fodder, biofuel, folk medicine, antioxidant, phytoremediation	Joshi et al. (2019, 2023)
	Suaeda maritima	Amaranthaceae	Shrub	NE		
	Suaeda monoica	Amaranthaceae	Shrub	NE		
	Suaeda nudiflora	Amaranthaceae	Under shrub	NE		
	Tamarix aphylla	Tamaricaceae	Small tree	NE	Animal fodder	Joshi et al. (2018, 2020c)
	Tamarix dioica	Tamaricaceae	Small tree	NE	Animal fodder, folk medicine	
	Tamarix indica	Tamaricaceae	Small tree	NE	Fodder, phytoremediation	
	Trianthema portulacastrum	Aizoaceae	Herb	NE	Food, animal fodder	
	Trianthema triquetra	Aizoaceae	Herb	NE	Food, fodder, folk medicine	
	Zygophyllum simplex	Zygophyllales	Shrub	NE	Animal fodder and biofuel	
Aquatic physiques	*Ceratophyllum demersum*	Ceratophyllaceae	Herb	NE	Folk medicine, pharmaceutical	Sen (2023), Roy and Roy (2019)
	Cynodon dactylon	Poaceae	Grass	NE	Fooder	
	Hydrilla verticillata	Hydrocharitaceae	Herb	LC	Folk medicine	
	Ipomea aquatic	Convolvulaceae	Climber	LC	Food, fodder, folk medicine	

	Ipomoea pes-tigridis	Convolvulaceae	Climber	NE	Food, fodder, folk medicine	
	Lemna minor	Araceae	Herb	NE	Fodder, phytoremediators	
	Nelumbo nucifera	Nymphaeaceae	Herb	NE	Food, phytoremediators	
	Nymphaea nouchali	Nymphaeaceae	Herb	NE	Fodder, phytoremediators	
	Pistia stratiotes	Araceae	Herb	LC	Food, phytoremediators	
	Potamogeton crispus	Potamogetonaceae	Herb	NE	Folk medicine, biogas	
	Spirodela polyrhiza	Araceae	Herb	NE	Bioremediations	
Hills and rocky outcrops	*Aristida funiculata*	Poaceae	Herb	NE	Fodder, shelters	Dash et al. (2022), Arora and Ramawat (2014), Khan et al. (2003)
	Bacopa monnieri	Plantaginaceae	Herb	LC	Folk medicine, pharmaceutical	
	Barleria prionitis	Acanthaceae	Herb	NE	Folk medicine, hedge	
	Brachiaria ramose	Poaceae	Grass	LC	Food, fodder	
	Butea monosperma	Fabaceae	Tree	NE	Fodder, timber, resin, dye	
	Calligonum polygonoides	Polygonaceae	Shrub	NE	Food, fodder, charcoal	
	Cassia fistula	Fabaceae	Tree	NE	Fodder, folk medicine	
	Cassia tora	Caesalpiniaceae	Tree	NE	Fodder, folk medicine	
	Cayratia trifolia	Vitaceae	Climber	NE	Folk medicine, pharmaceutical	Arora et al. (2020)
	Cuscuta reflexa	Cuscutaceae	Climber	NE	Fodder, bio-stimulants	
	Commiphora wightii	Burseraceae	Tree	CR	Folk medicine, pharmaceutical	Roy and Roy (2019), Arora et al. (2010)
	Crotalaria medicaginea	Fabaceae	Herb	NE	Folk medicine	
	Euphorbia caducifolia	Euphorbiaceae	Shrub	NE	Folk medicine, biofuel	
	Euphorbia hirta	Euphorbiaceae	Herb	NE	Folk medicine	
	Ephedra foliata	Ephedraceae	Shrub	NE	Folk medicine	
	Lepidagathis trinervis	Acanthaceae	Herb	NE	Folk medicine	

Continued

TABLE 6.1 Dominant flora of Thar Desert, their conservation status and utilization—cont'd

Type of edaphology	Name of species	Family	Habitat	IUCN status	Common uses	Reference
	Moringa oleifera	Moringaceae	Tree	NE	Food, fodder, folk medicine	
	Phyllanthus amarus	Euphorbiaceae	Herb	NE	Folk medicine	
	Pupalia lappacea	Amaranthaceae	Herb	NE	Fodder, folk medicine	
	Pueraria tuberosa	Fabaceae	Climber	VN	Food, folk medicine, pharmaceutical	Kanthaliya et al. (2019, 2023)
	Tinospora cordifolia	Menispermaceae	Climber	NE	Folk medicine, pharmaceutical	Arora et al. (2010)
	Tragus biflorus	Poaceae	Grass	NE	Animal forage	
	Tephrosia purpurea	Fabaceae	Herb	NE	Fodder, folk medicine	
	Sida cordifolia	Malvaceae	Herb	NE	Folk medicine, pharmaceutical	
	Withania somnifera	Solanaceae	Herb	NE	Folk medicine, pharmaceutical	
Invaded alien species	*Argemone mexicana*	Papaveraceae	Herb	NE	Folk medicine	Charan and Singh (2018), Khan and Frost (2001)
	Calotropis procera	Asclepiadaceae	Shrub	NE	Animal forage, folk medicine	
	Cassia italica	Fabaceae	Herb	NE	Animal forage, folk medicine	
	Cassia occidentalis	Fabaceae	Herb	NE	Food, folk medicine	
	Cleome viscosa	Cleomaceae	Herb	NE	Folk medicine	
	Eichornia crassipes	Pontederiaceae	Herb	NE	Landscape gardening	
	Ipomoea eriocarpa	Convolvulaceae	Climber	NE	Folk medicine	
	Lantana camara	Verbenaceae	Herb	NE	Folk medicine	
	Parthenium hysterophorus	Asteraceae	Herb	NE	Phytoremediators, biogas	
	Prosopis chilensis	Fabaceae	Shrub	NE	Animal forage, timber	
	Pulicaria arabica	Asteraceae	Herb	NE	Animal forage	
	Typha angustata	Typhaceae	Herb	NE	Food, fodder	

LC, least concern; *NE*, not evaluated.

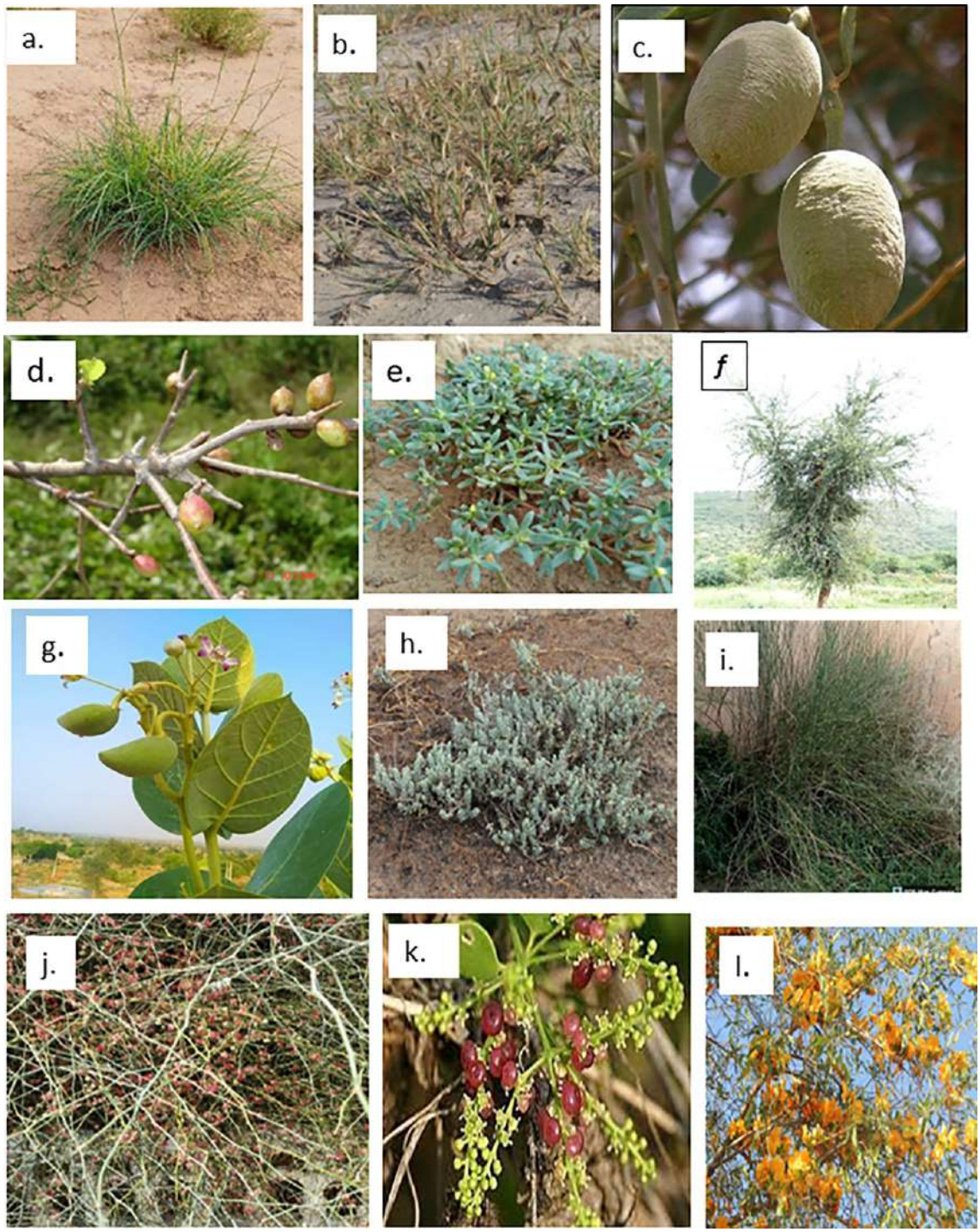

FIG. 6.3 Representative flora of the Thar Desert (A) *Lasiurus scindicus*, (B) *Sporobolus* sp. (C) *Balanites aegyptiaca*—fruits, (D) *Commiphora wightii*, (E) *Trianthema triquetra*, (F) *Prosopis juliflora*, (G) *Calotropis procera*, (H) *Suaeda fruticosa*, (I) *Leptadenia pyrotechnica*, (J) *Capparis decidua*, (K) *Salvadora persica*, and (L) *Tecomella undulata*. *(Credit: Images A, C, F, I—Ms Supriya Meena; images B, E, G, H—Dr. Abhishek Joshi; and images D, J, K, L—Retd. Prof. K.G. Ramawat.)*

such as Kaylana in Jodhpur, Kolayat and Gajner in Bikaner, and Gadisar in Jaisalmer, retain water throughout the year, resulting in the growth of weeds and aquatic plants. A total of 11 species belonging to 10 genera and 7 families grow dominantly in the major aquatic bodies of the Desert region. The highest prevalence was reported for the Araceae family, with three species, followed by Convolvulaceae and Nymphaeaceae with two species each. The remaining four families were represented by one species each. It was observed that *Nelumbo nucifera* was the predominant

species in the Kolayat lake of Bikaner, while *Cynodon dactylon* was observed to be dominant near the banks of almost all aquatic bodies. These aquatic species are potential candidates for animal forage, folk medicine, and bioremediation.

The native flora of the Thar Desert face severe threat from alien vegetation, which is frequently invaded by irrigation facilities and canal irrigation. The invasive species persist and propagate over multiple generations without any direct intervention from humans. The desert region is home to 12 invasive species belonging to 11 genera and 9 families. The region experiences a rapid spread of *Prosopis chilensis*, *Calotropis procera*, *Lantana camara*, *Typha angustata*, *Parthenium hysterophorus*, and *Eichornia crassipes*. These species have adapted to survive in almost in any desert condition due to their broad range of ecological adaptability.

3.2 Faunal biodiversity

More than 2000 faunal species have been identified from the Thar Desert, ranging from single-celled Protozoa to Mammalia, with 619 being vertebrates and the rest invertebrates (Sharma, 2013). This fauna constitutes ~2.12% of the total fauna of the country.

3.2.1 Invertebrates

Despite the harsh dry conditions, the Thar Desert has been identified as a species-rich zone for a wide range of faunal species that are normally adapted to desert life as a result of the juxtaposition of Oriental, Palaearctic, and Saharan features (Pocock, 1939, 1941; Prakash, 1963). Among the invertebrates identified from this region are 82 species of protozoa belonging to the classes Mastigophora, Sarcodina, and Ciliata (Mahajan, 1969, 1971, 1977). Mishra (1960) identified opalinid infection in monitor lizard feces, whereas Mahajan and Mukherjee (1977) discovered seven species of rectal ciliates in frogs. The Thar Desert is home to seven freshwater sponge species (Soota and Saxena, 1983). A total of 87 species of Platyhelminthes have been investigated in this area. Furthermore, there are 38 Trematode species and 50 Cestode species described from amphibians, reptiles, aves, and mammals (Gupta, 1976; Wason and Johnson, 1977; Nama, 1972, 1974, 1984, 1990a,b; Nama and Khichi, 1974).

Sewak (2009) identified 85 species of dung beetles belonging to 12 genera (subfamily: Coprinae) in human and animal dungs, whereas Tak (2009) discovered 35 species of ants belonging to 16 genera of the Formicidae family in Rajasthan and Gujarat states. More than 70 spider species have been identified; the number varies by district (Singh and Singh, 2022). Interestingly, 37 species were found only from sand dunes, indicating a typical tolerance to dry and sandy conditions (Kashmeera and Sudhikumar, 2019). On the other hand, more than 40 butterfly species with diverse host plant preferences have been reported (Kazmi et al., 2003; Meena et al., 2021).

3.2.2 Vertebrates

Despite the arid conditions, 142 fish species have been reported from the entire Thar Desert. Due to some perennial waterbodies, the districts of Bikaner, Jaisalmer, Pali, and Sirohi in the desert have the highest number of species and abundance (Banyal and Kumar, 2017; Pathak et al., 2020). This desert is also famous for its herpetofaunal richness, which includes lizards, snakes, and turtles. The Brahminy river turtle (*Hardella thurjii*), Indian flapshell turtle (*Lissemys punctata*), and Indian softshell turtle (*Trionyx gangeticus*) are the most common Chlonians. The Indian star tortoise (*Geochelone elegans*) is the only land tortoise found in this region, and has been observed in Pali district (Sharma and Gaur, 2005). A total of 19 lizard species has been identified and the most common are Brooke's house gecko (*Hemidactylus brookii*), Leschenault's leaf-toed gecko (*Hemidactylus leschenaultii*), Northern house gecko (*Hemidactylus flaviviridis*), Keeled Indian mabuya (*Eutropis carinata*), bronze grass skink (*Eutropis macularia*), Indian fringe-toed lizard (*Acanthodactylus cantoris*), Jerdon's snake-eye (*Ophisops jerdonii*), and Common Indian monitor lizard (*Varanus bengalensis*—Gaur et al., 2013). The Indian chameleon (*Chamaeleo zeylanicus*), Brilliant ground agama (*Trapelus agilis*), and Oriental garden lizard (*Calotes versicolor*) are commonly seen during the wet season. Eastern sand swimmer (*Ophiomorus raithmai*) may be easily seen in sand dunes. The Sind gecko (*Crossobamon orientalis*), Rajasthan toad-headed lizard (*Phrynocephalus laungwalaensis*), common dotted garden skink (*Riopa punctata*), Indian spiny-tailed lizard (*Uromastyx hardwickii*), Eastern sand swimmer and small-scaled lacerta (*Ophisops microlepis*), on the other hand, are uncommon. The Sind gecko (*C. orientalis*) has also been documented from the westernmost reaches of the Thar Desert in Jaisalmer district, which borders Pakistan (Solanki et al., 2015). This area is also home to >22 snake species, including venomous snakes such as the Indian cobra (*Naja naja*), Indian saw-scaled viper (*Echis carinatus*), Sindh saw-scaled viper (*Echis carinatus sochureki*), Russell's viper (*Daboia russelii*), and Sindh Krait (*Bungarus sindanus*—Kumawat and Purohit, 2021). Skittering frog (*Euphlyctis cyanophlyctis*), Indian bullfrog (*Hoplobatrachus tigerinus*), Boie's wart frog (*Limnonectes limnocharis*), green pond frog (*Euphlyctis hexadactylus*), Indian burrowing frog (*Tomopterna breviceps*), Ornate narrow-mouthed frog (*Microhyla ornata*), Asian common toad (*Bufo melanostictus*), and Indian marbled toad (*Bufo stomaticus*) are among the general amphibians (Sharma and Gaur, 2005).

The Thar Desert is home to a diverse range of resident and migratory bird species. So far, around 150 bird species have been reported (Rahmani, 1997a,b). The Great Indian bustard

(*Ardeotis nigriceps*), Demoiselle crane (*Anthropoides virgo*), imperial sand grouse (*Pterocles orientalis*), Houbara or Macqueen's bustard (*Chlamydotis macqueenii*), Indian Eagle-Owl (*Bubo bengalensis*), black Francolin (*Francolinus francolinus*), Indian roller (*Coracias benghalensis*), Indian peafowl (*Pavo cristatus*), and various species of vultures, falcons, and eagles are among the notable avian fauna identified. The Great Indian bustard is a critically endangered species, with ~90% of its population found in the Thar Desert, particularly in Desert National Park. Stoliczka's bushchat (*Saxicola macrorhyncha*) is another notable species in the area (Islam and Rahmani, 2011). Seven species of vulture observed in Thar Desert namely Cinereous vulture (*Aegypius monachus*), Egyptian vulture (*Neophron percnopterus*), King vulture (*Sarcogyps calvus*), Long-billed vulture (*Gyps indicus*), Eurasian griffon (*Gyps fulvus*), Himalayan griffon (*Gyps himalayensis*) and White-backed vulture (*Gyps bengalensis*) are found. Winter migratory species include the Eurasian griffon, Himalayan griffon, and Cinereous vulture, whereas other species are residents with breeding records (Chhangani, 2009). However, their numbers have been steadily declining over the past few decades. Another fragile and endemic Thar Desert species is the White-naped tit (*Machlolophus nuchalis*), which is located in semiarid areas next to the Aravalli range (Sharma and Koli, 2014). A huge flock of winter migrating Demoiselle cranes (*A. virgo*) may be seen in Khichan hamlet, where they are fed by residents (Ilyashenko et al., 2022).

The Thar Desert is home to 68 dry-adapted mammal species (18% of India's mammalian biodiversity). The majority of mammals move underground during the day to avoid excessive heat or to adapt to nocturnal foraging habits. Some notable mammals are the Indian gazelle or chinkara (*Gazella bennettii*), Indian wild ass (*Equus hemionus khur*), desert fox (*Vulpes vulpes pusilla*), Indian wolf (*Canis lupus pallipes*), Indian desert cat (*Felis silvestris*), Indian hare (*Lepus nigricollis*), Caracal (*Caracal caracal*), black buck (*Antilope cervicapra*), desert gerbil (*Meriones hurrianae*), wild boar (*Sus scrofa*), and blue bull (*Boselaphus tragocamelus*). Leopards (*Panthera pardus*) have been sighted in the desert area in recent decades, most likely as a result of increased water supply due to the Indira Gandhi Canal and changing irrigation practices (Sharma et al., 2021b). Except for chinkara and black buck in a few regions, the status of other larger mammals is not satisfactory. Caracal is a threatened species in India, and its presence in the Thar Desert has only been reported in Udaipur and Pali districts since 2000 (Khandal et al., 2020). Gerbils prefer open plain and interdunal regions among the 18 rodent species (11 genera and 3 families). Cutch rat (*Rattus cutchicus*) and flat-haired mouse (*Mus platythrix sadhu*) are only found in rocky settings, with *Euphorbia caducifolia* clumps in some places. The only nonhuman primate found in the desert area is the Hanuman langur (*Semnopithecus entellus*—Agoramoorthy, 2009). As per zoogeographical distribution patterns, the majority of rodent species in this desert represent a mixture of Oriental and Saharan origins (Prakash, 1963). Bats are a critical component of the desert environment, with three megachiropteran species (i.e., Indian flying fox *Pteropus giganteus giganteus*, greater short-nosed fruit bat *Cynopterus sphinx* and fulvous fruit bat *Rousettus leschenaultii*) and 12 microchiropteran species (i.e., trident leaf-nosed bat *Asellia tridens*, large rat-tailed bat *Rhinopoma microphyllum kinneari*, lesser mouse-tailed bat *Rhinopoma hardwickii*, Egyptian tomb bat *Taphozous perforatus*, Naked-rumped tomb bat *Taphozous kachhensis* (*nudiventris*), Blyth's horseshoe bat *Rhinolophus lepidus*, greater Asiatic yellow house bat *Scotophilus heathii*, Indian pygmy bat *Pipistrellus mimus* (*tenuis*), Indian false vampire bat *Megaderma lyra lyra*, Fulvus roundleaf bat *Hipposideros fulvus*, Egyptian free-tailed bat *Tadarida aegyptiaca*, and Dormer's bat *Scotozous dormer* (Purohit and Vyas, 2009; Senacha and Dookia, 2013). Some of the major faunas have been shown in Fig. 6.4.

4 Conservation areas and their ecological indicators

4.1 Conservation of flora

Conservation of biodiversity is essential to sustain the delicate ecosystem of the Thar Desert. Several informal protected areas, such as bani (small protected forests), beehads (larger protected forests), orans (sacred groves), and seenves (margins between cultivated land and open fields) have been discovered across the desert regions. These protected areas are similar to mini-biosphere reserves and play a vital role in maintaining the ecological stability of the Thar Desert (Pandey, 1999).

Sacred groves, which are commonly referred to as "Orans," are small patches of forests or stands of trees that local communities conserve due to their religious beliefs passed down through generations. They are also classified as sacred corridors (deo ghats), sacred gardens (baughs), and temple forests (Dagla et al., 2007; Choudhary and Gupta, 2014). Several tree species, such as *P. cineraria*, *Acacia catechu*, *Tecomella undulata*, and *Salvadora* spp., and shrub species like *C. procera* and *Anogeissus pendula* are commonly found to grow in these sacred groves, along with grasses like *Lasiurus scindicus*, and *C. ciliaris*. However, the overall composition of the vegetation in the orans is primarily determined by the local vegetation, soil conditions, and environmental factors (Gadi et al., 2016).

Trees and branches must not be cut from these regions, with only grasses and palatable herbs permitted to be harvested for animal forage. Sacred groves are also found to harbor a number of animals and birds that are rare and endemic

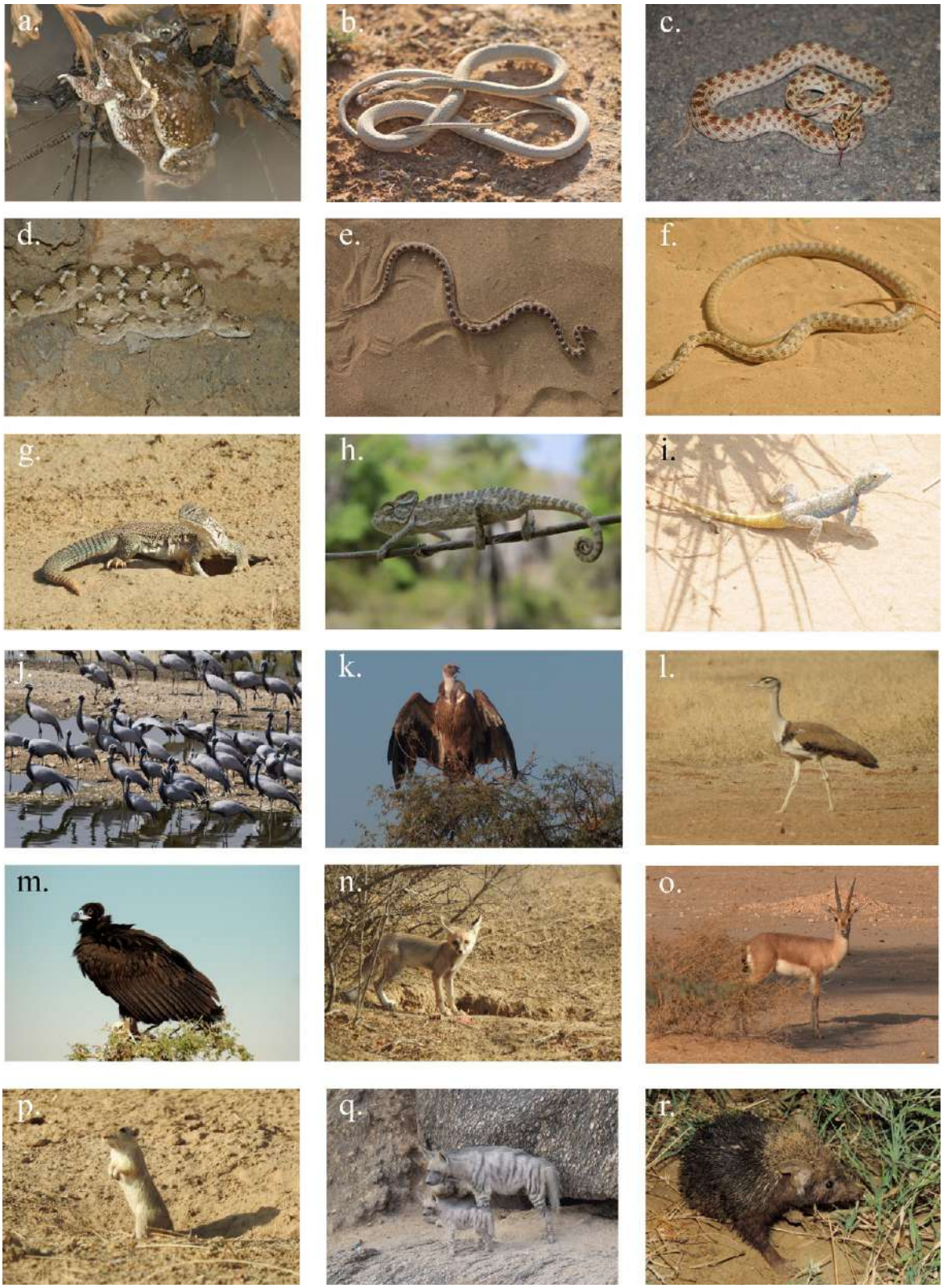

FIG. 6.4 Representative vertebrate fauna of the Thar Desert. (A) Indian marbled toad (*Duttaphrynus melanostictus*); (B) Afro-Asian sandsnake (*Psammophis schokari*); (C) red-spotted royal snake (*Spalerosophis arenarius*); (D) Sindh saw-scaled viper (*Echis carinatus*); (E) Sindh awl-headed snake (*Lytorhynchus paradoxus*); (F) glossy-billed racer (*Platyceps ventromaculatus*); (G) Hardwicke's spiny-tailed lizard (*Saara harwickii*); (H) Asian chameleon (*Chamaeleo zeylanicus*); (I) brilliant ground agama (*Trapelus agilis*); (J) a flock of Demoiselle crane (*Grus virgo*); (K) Himalayan griffon (*Gyps himalayensis*); (L) great Indian bustard (*Ardeotis nigriceps*); (M) cinereous vulture (*Aegypius monachus*); (N) desert fox (*Vulpes vulpes*) with its den; (O) chinkara (*Gazella bennettii*); (P) Indian gerbil (*Tatera indica*); (Q) hyena (*Hyena hyena*) at its den; and (R) collared hedgehog (*Hemiechinus collaris*). *(Credit: Images A, I, K, M, R—Dr. Pratap Singh; images B–F—Rakesh Kumawat; images G, N, P—Vijay K. Koli; and images J, Q—Utkarsh Prajapati.)*

such as the great Indian bustard, lesser florican, Houbara bustard, black buck deer, gray wolf, chinkara, and spiny-tailed lizard (Hall and Hamilton, 2014). In addition, these regions serve as shelter and breeding sites for many migratory and residential birds, providing sources of perennial water (ponds and small wells), maintaining the local microenvironment, and supporting biogeochemical cycles (Agarwal, 2016; Gaur et al., 2018). Therefore, sacred groves serve as a gene pool and offers shelter to a variety of endemic, endangered, and threatened species (Kumar and Sen, 2022). Some widely recognized sacred groves of Thar Desert are listed in Table 6.2.

4.2 Conservation of fauna

The Thar Desert has one national park, one wildlife sanctuary, and five conservation reserves (Table 6.3) that support several threatened and ecologically significant faunal species. Birdlife International and the Bombay Natural History Society (BNHS) have also identified 10 globally important bird and biodiversity areas (IBAs) in this region as high-priority conservation areas, including the Desert National Park (Jaisalmer and Barmer), Tal Chhapar Wildlife Sanctuary (Churu), Diyatra area (Bikaner), Khichan (Jodhpur), Sambhar lake (Nagaur, Jaipur and Ajmer), Badopal (Hanumangarh), Jawai dam area (Pali), Jor Beer (Bikaner), Kharda dam (Pali), and Sardar Samand lake (Pali—Rahmani et al., 2016).

5 Human activities and their impacts

The Thar Desert is the most densely populated desert in the world (84 people/km^2) due to increased development opportunities, putting a severe biotic strain on its ecology and natural resources. The majority of people live in scattered settlements of villages (known as dhanis); however, city areas are expanding as well. Besides this, the high livestock population and frequent droughts reduce ecosystem productivity and regeneration. Agriculture in this region is mostly reliant on groundwater irrigation. Excessive use of groundwater for agriculture, on the other hand, can lead to water scarcity and aquifer depletion, harming both human livelihoods and the desert's fragile environment. The area also has the world's highest cattle density (Venkateswarlu et al., 1992), ranging from 42 per km^2 (in Jaisalmer district) to 226 per km^2 (in Sikar district). Grazing livestock is a prevalent practice in the Thar Desert. The cattle population in many regions of Rajasthan state is nearly double that of the human population. Hence, the growing livestock population put enormous pressure on grasslands and native plants. Overgrazing can cause vegetation degradation, decreased species diversity and density, soil erosion, and biodiversity loss. It may create barren patches where vegetation is unable to regrow, thus contributing to desertification (Tewari and Arya, 2004).

However, the Indira Gandhi Canal has converted >11% of desert area into productive land, but it has also caused significant environmental issues, notably the extinction of local wildlife. The excessive water availability has altered soil texture, moisture content, and floral composition. Excessive irrigation practices, frequently fueled by subsidies and unrestricted water availability, lead to overexploitation of water resources. Excessive irrigation reduced groundwater and increased salinity in agricultural areas and ground waterlogging, resulting in decreased productivity (Chlachula, 2021). Owing to the presence of a hard layer of limestone, 1.5–6 m below the surface, irrigation water seeps down and collects on this layer. This water became nearly saturated and swampy as it rose. This can lead to an increase in aquifers as well as increased salt in the soil, which contributed to land degradation and lower agricultural output in the long run. In addition, Rahmani and Soni (1997) observed that the entrance of this canal modifies the avifauna in the area and introduces several new and invasive plant species.

The Thar Desert is also noted for its mineral richness such as limestone, gypsum, and marble. Mining and quarrying activities cause habitat destruction, soil erosion, and the discharge of pollutants into the environment; all of which have an impact on the area's biodiversity. Similarly, newly created companies in the Kutch Rann are spewing effluents that harm both land and coastal fauna. Mangrove depletion in this zone has resulted in a decrease in fish capture. For mangrove forests, the yearly degradation rate is ~11% (Srivastava and Mehta, 2021).

The presence of power lines and windmills in Thar Desert wildlife habitats has highlighted significant threats to numerous bird species, including the globally threatened great Indian bustard and vultures (Dookia, 2010; Uddin et al., 2021). Uddin et al. (2021) identified 22 species impacted by powerline wire networks. The number of wires was found to be positively associated to avian mortality. Wind farms also disrupt these bird species' flight paths, resulting in many deaths (Dookia, 2010). Many vulnerable species are also affected by many local factors such as increased road traffic in Little Rann, as well as excessive weed growth in these habitats that have been recognized as a contributing factor to the decline of the wild ass population. The Maldharis, nomadic cattle grazers, have also been cited as a significant threat to the wild ass population due to their encroachment on its habitat within the Kutch Rann.

Nowadays, the Thar Desert attracts a significant number of tourists for activities such as camel safaris and desert camping. If not managed properly, unregulated tourism can cause habitat disturbance, littering, and the destruction of natural ecosystems. To reduce the environmental impact, proper trash management and acceptable tourism practices are required (Cadène, 2017). Besides indiscriminate cutting of certain important medicinal plants such as *Commiphora*

TABLE 6.2 Some widely recognized sacred groves of Thar Desert and their ecological indictors.

Name of the sacred groves	Location	Area (ha)	Dominant floral species
Oran of Karni Mata	Deshnokh, Bikaner	370	*Cenchrus* spp., *Capparis decidua, Ziziphus nummularia*
	Mokha Charan, Bikaner	300	*Salvadora oleoides, Lasiurus scindicus*
	Khari Khurd, Jodhpur	23	*Prosopis juliflora, P. cineraria, Capparis decidua*
Oran of Ashapura	Devikot, Jaisalmer	1300	*Prosopis cineraria, Capparis decidua, Ziziphus nummularia*
	Nadole, Pali	14	*Acacia nilotica, Salvadora oleoides*
Oran of Tanot	Tanot, Jaisalmer	35	*Caralluma edulis, Commiphora wightii, Glossonema varians, Lasiurus scindicus, Leptadenia pyrotechnica, Tribulus terrestris*
Oran of Bhadariya	Bhadaria, Jaisalmer	15,000	*Prosopis cineraria, Ziziphus nummularia, Haloxylon* spp.
Oran of Ramdev	Ramdeora, Jaisalmer	12	*Acacia tortilis, Crotalaria burhia, Ziziphus nummularia*
Oran of Rohida	Akolai, Jalore	35	*Tecomella undulata, Prosopis cineraria*
Oran of Pabooji	Kolu, Jodhpur	3335	*Capparis decidua, Ziziphus nummularia*
	Bhopalgarh, Jodhpur	7	*Prosopis juliflora, Prosopis cineraria*
Oran of Viratra Mata	Dhok, Barmer	1859	*Prosopis juliflora, Ziziphus nummularia*
Oran of Bankal Mata	Ranigaow, Barmer	4600	*Tecomella undulata*

wightii has led to its inclusion in the Red List of International Union for Conservation of Nature (IUCN) under critically endangered species (Ved et al., 2015).

6 Operational and potential monitoring tools to conserve or to recover the biome

Conserving and recovering Thar Desert biodiversity is critical for maintaining the region's natural balance and sustaining the livelihoods of its residents. Management practices that are long-term are essential. This attempt can be aided by a number of operational and potential monitoring tools. Long-term priorities should include land protection or recovery from the risks of water scarcity and land erosion, regional ecology stability, socioeconomic awareness of the sustainable use of natural resources, protection of threatened and endemic flora and fauna, and promotion of ecotourism and geoheritage to improve economic development in local communities (Chlachula, 2021). Many endemic and threatened species, like the great Indian bustard, Houbara bustard, sand grouses, white-browed bushchat, falcons, and raptors, require special attention and a species-specific program. Protecting associated native scrub and rocky habitats requires equal effort.

Remote sensing technologies, such as satellite imaging and airborne surveys, along with Geographic Information Systems (GIS), can give useful data on land-use changes, vegetation cover, soil moisture, and encroachment in the Thar Desert. This information can aid in identifying regions that require immediate attention and developing appropriate conservation plans. Temporal monitoring and mapping of land degradation and desertification can aid in understanding the scope of the problem and developing appropriate solutions (Sharma et al., 2021a). Alternative land-management practices can aid in the pursuit of the natural sustainability of Thar Desert occupancy habitats. Under the UNCCD framework, spatial-temporal assessment of Environmentally Sensitive Areas (ESA) can aid in the fight against desertification (Sharma et al., 2021a). Effective management can conserve the Luni River, which may help to improve groundwater in that area.

TABLE 6.3 Protected areas present in the Thar Desert, India and their important fauna.

S. no.	State	District	Name of protected area	Area (ha)	Important fauna
1	Rajasthan	Jaisalmer and Barmer	Desert national park	316,200	Great Indian bustard, Blackbuck, Chinkara, wolf (*Canis lupus pallipes*), Caracal (*Caracal caracal*)
2		Churu	Tal Chhapar wildlife sanctuary	630,000	Blackbuck, Desert fox, Raptors
3		Jalore	Sunda mata conservation reserve[a]	11,749	Sloth bear (*Melursus ursinus*), Leopard, Striped hyena (*Hyaena hyaena*), Indian porcupine (*Hystrix indica*), Blue-bull, Jungle cat (*Felis chaus*), Desert cat
4		Bikaner	Gajner wildlife sanctuary	1920	Chinkara, Black buck, Desert fox, Wild boar (*Sus scrofa*)
5			Jod Beed Gaadwala[a]	5646	Black Buck, Wild Cat, Wild boar
6		Jodhpur	Gudha Vishnoiyan[a]	231	Chinkara, Black Buck, Wild boar
7		Nagaur	Gogelao[a]	358	Chinkara, Hare, Black Buck
8			Rotu[a]	73	Chinkara, Hare, Black Buck
9	Gujarat	Kutch	Kutch Desert Sanctuary	750,622	Wild ass, Spiny-tailed lizards, Chinkara, Desert fox, Hyena, Blue bull, Indian porcupine, Indian pale hedgehog (*Paraechinus micropus*), Indian hare and Flamingos
10	Punjab	Fazilka	Abohar wildlife sanctuary	18,650	Black buck, Bluebull, Wild boar, Porcupine, Indian jackal (*Canis aureus*)

[a]*Conservation reserve.*

The Thar Desert has some good grasslands, particularly of Sewan (*Lasiurus scindicus*); however, they are severely grazed. These places can be classified as multiple-use protected areas. Livestock overstocking, which leads to overgrazing, should be regulated. Controlled grazing has the potential to restore productivity and biodiversity, but it is dependent on an understanding of plant succession and resilience (Tewari and Arya, 2004). The Indira Gandhi Canal should not be introduced in these areas in order to preserve their natural renewing process. Agriculture and human settlement should be discouraged in these areas as well (Islam and Rahmani, 2011). Remote sensing can be used to track the productivity of these lands on a regular basis. Normalized Difference Vegetation Index (NDVI) can provide some insight into changing ecological conditions and help to improve management actions.

Monitoring soil health is critical for sustainable land management and agriculture. Soil erosion, salinity, and fertility assessments can help and guide appropriate conservation and improvement actions. Quantitative measurements of dry fluvial dynamics can aid in understanding erosion and desertification in the Thar Desert (Chlachula, 2021). Exotic plant species should be replaced by native or endemic plant species. Drought-adapted tree plantation, arid-land farming with newly developed xerophytic cultigens, and regional hydrological network regulation will be important in stabilizing sand dune motions in new areas (Chlachula, 2021).

As the impact of global climate change on the Thar Desert is apparent (Goyal and Surampalli, 2018), building weather monitoring stations around the Thar Desert will be critical in examining the region's climate trends, including rainfall, temperature, and humidity. This information will be useful for anticipating droughts, analyzing water availability, and controlling agricultural practices. Climate change can be tracked. The Thar Desert is primarily reliant on groundwater resources. Regular monitoring of groundwater levels and quality can assist prevent overexploitation and will assure its long-term use. Adoption and promotion of traditional knowledge and practices, together with current scientific intervention,

may help to alleviate the negative effects of climate change in the future. Continuous and long-term monitoring is essential to assess faunal changes, particularly in light of recent climate change.

The Indira Gandhi Canal that runs for 649 km alters the environment of the area it passes. The increased vegetation along with the canal changed numerous floral and faunal communities. Rahmani and Soni (1997) have already reported on changes in avifaunal diversity, but information on other groups is scarce. Hence, more research on the impact of this canal on the distribution and status of fauna, with a focus on insects, amphibians, reptiles, and mammals, is required.

Regular biodiversity surveys and monitoring in the Thar Desert can offer information on the state and distribution of many plant and animal species. This understanding is critical for implementing conservation measures and conserving its unique desert ecology. If used correctly, in vitro technologies can be important aids in biocultural conservation (Kulak et al., 2022). Currently, only one national park exists in the westernmost region of the Thar Desert; furthermore, protected areas should be established to safeguard the biodiversity of this dry landscape.

Engaging communities in monitoring initiatives aids to promote a sense of ownership and responsibility for Thar Desert conservation. Their traditional knowledge can be useful in identifying environmental changes and implementing sustainable practices. The invasion of Western lifestyles in people, particularly among the younger generation, and the waning interest in their rich cultural history must be reversed. Traditional knowledge must be passed down from generation to generation (Gaur and Gaur, 2004). Technology appears to be a key barrier to communication, which can be bridged through social media campaigns and programs. Local nongovernmental organizations (NGOs) can take the lead in promoting information exchange within communities, particularly among persons of diverse ages. Social and religious activities can also serve as communication venues. Raising awareness and education about the importance of Thar Desert conservation among the local population and the general public can help to gain support and encourage responsible behavior. Eco-tourism should be encouraged due to the expanding human population in this vulnerable ecosystem and the need for their employment; yet, such locations have problems with solid waste management, including polythene bags and other items that harm the natural ecology. As a result, strict rules, regulations, and guidelines must be developed and enforced.

7 Conclusions

Overall, the Thar Desert's geographic location and climate create a challenging climate characterized by severe temperatures, limited water resources, and a distinct ecosystem. It is still an important geographical feature of India in terms of both natural beauty and historical significance.

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Chapter 7

Sahara and other African deserts

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1 Introduction

1.1 Overview of African deserts

Drylands cover about 40% of the global land surface and are home to more than two billion people (Middleton, 2017). Deserts have always been an integral part of the story of humanity and a source of fascination for us (Mares, 2017; Li et al., 2024). They constitute unique ecosystems characterized by extreme aridity and limited precipitation, which significantly influence the distribution of flora and fauna (Noy-Meir, 1973; Laity, 2009). The definition of a desert varies, but it is generally described as an area that receives very little rainfall, typically less than 250 mm per year (UNEP (United Nations Environment Programme), 2013). African deserts, including the Sahara Desert, Namib Desert, Kalahari Desert, and other small deserts, exemplify these arid regions and exhibit distinct characteristics (Laity, 2009). The common feature among deserts is the high aridity. Fig. 7.1 shows African drylands classified based on the aridity index expressed as the ratio of annual precipitation over annual potential evapotranspiration (PET).

One defining characteristic of deserts is their high evaporation rates due to intense solar radiation and low humidity. This creates an imbalance between water input and output, leading to minimal surface water and limited vegetation cover (Dirmeyer and Shukla, 1996; Williams, 2014). As a result, African deserts are often characterized by vast expanses of sandy or rocky landscapes, interspersed with sporadic oasis ecosystems, with typical vegetation and fauna of each landscape type (Azizi et al., 2021; Bradai et al., 2015a; Bouallala et al., 2022). These deserts also feature unique landforms such as sand dunes, salt pans, wadis (dry riverbeds), and inselbergs (isolated mountains) that have been shaped by wind and water erosion over millions of years (Laity, 2009; Parsons and Abrahams, 2009).

Another notable characteristic of African deserts is their extreme temperature fluctuations. During the day, desert temperatures can soar to scorching highs, often exceeding 40°C, while at night, they can plummet to near-freezing temperatures (Laity, 2009). These extreme temperature ranges pose significant challenges to both plants and animals, necessitating adaptations to survive in such harsh conditions. Consequently, African desert ecosystems are inhabited by specialized flora and fauna that have evolved unique physiological and behavioral adaptations to conserve water and tolerate extreme temperatures (Whitford and Duval, 2019; Mitchell et al., 2020; Berdugo et al., 2022; Huntley, 2023; Mouane et al., 2024).

Furthermore, African deserts are often characterized by low and erratic rainfall patterns, leading to sporadic vegetation growth and scarce water resources (Le Houérou, 1989). Vegetation in these deserts typically consists of drought-tolerant and water-storing plants, such as cacti, succulents, and hardy shrubs (Quézel, 1965; Williams and Faure, 1980; Ozenda, 1983; Zahran and Willis, 1992; cf. Neffar and Chenchouni, 2025). These plants have adapted various mechanisms to minimize water loss, such as reduced leaf surface area, waxy coatings, and deep root systems that tap into underground water sources. These adaptations enable desert plants to survive in arid environments with limited access to water (Bezzalla et al., 2018; Ekwealor et al., 2020; Berdugo et al., 2022).

1.2 Background and significance of studying African deserts

The African deserts, particularly the Sahara Desert, are among the most iconic and intriguing ecosystems on our planet. Spanning over 9 million square kilometers, the Sahara Desert is the largest hot desert globally and

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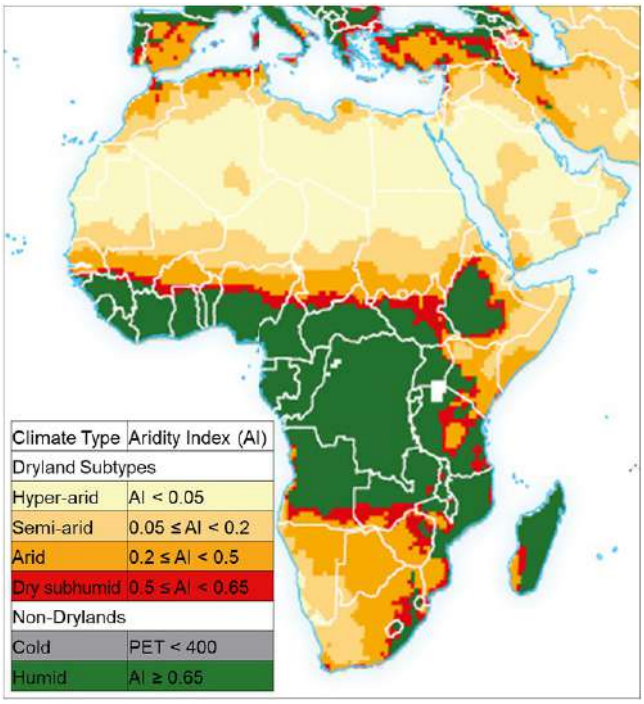

FIG. 7.1 Map of Africa with the climate classification and dryland subtypes based on the aridity index. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: This map is adapted from the World Atlas of Desertification (European Commission, Joint Research Centre, Hill, J., Von Maltitz, G., Sommer, S. et al., 2018. World Atlas of Desertification—Rethinking Land Degradation and Sustainable Land Management. Publications Office of the European Union, Luxembourg. https://doi.org/10.2760/06292.).)*

significantly influences the climate patterns of the African continent (Laity, 2009; UNEP (United Nations Environment Programme), 2013; Masih et al., 2014). Studying African deserts is of great importance due to their unique ecological characteristics and the critical role they play in maintaining global biodiversity. The extreme aridity, high temperatures, and scarce water resources in these deserts have shaped the adaptation strategies of plants and animals, resulting in specialized and resilient species (Whitford and Duval, 2019; Mitchell et al., 2020). Therefore, the understanding of the complex ecological dynamics of African deserts contributes to our knowledge of ecosystem functioning, adaptation mechanisms, and conservation strategies (Chenchouni and Neffar, 2025).

Investigating the African deserts also holds relevance in the context of global climate change. These deserts are experiencing the impacts of global warming at an alarming rate. Studies have indicated that the Sahara Desert is expanding both northward and southward due to changes in temperature and precipitation patterns, leading to desertification and land degradation (Laity, 2009; UNEP (United Nations Environment Programme), 2013; Terink et al., 2013; Ziadat et al., 2022). Such changes have far-reaching consequences for both local communities and global ecosystems (Allen et al., 2010; Vale and Brito, 2015; Al-Blooshi et al., 2020). Therefore, studying African deserts provides valuable insights into the effects of climate change on arid regions and aids in formulating effective mitigation and adaptation strategies.

Moreover, African deserts harbor a remarkable diversity of plant and animal species that have evolved to survive in extreme desert conditions. These unique ecosystems contain endemic species found nowhere else on Earth, making them important hotspots of biodiversity (Noy-Meir, 1973; Quézel, 1965; Ozenda, 1983; Huntley, 2023). According to Whitford and Duval (2019), by studying the flora and fauna of African deserts deepens the understanding of the evolutionary processes and ecological interactions that shape desert ecosystems. Furthermore, the identification and documentation of desert species contribute to global conservation efforts and support the development of sustainable management practices.

In addition to their ecological significance, African deserts have been home to indigenous communities for thousands of years. These communities have developed traditional knowledge and survival strategies that are closely intertwined with the desert environment (Keenan, 2013; Bradai et al., 2015b). According to Keenan (2013), African deserts have valuable insights into the cultural heritage, sustainable land-use practices, and traditional adaptation methods of these indigenous groups. Indeed, recognizing the value of indigenous knowledge and involving local communities in conservation efforts is crucial for the long-term preservation of African desert ecosystems (Chenchouni and Neffar, 2025).

Finally, understanding the dynamics of African deserts has broader implications for global biome conservation and sustainable development. Desert ecosystems provide valuable ecosystem services such as carbon sequestration, water regulation, and soil stabilization (Boudjabi and Chenchouni, 2022; Chenchouni and Neffar, 2022; Dhawi, 2023; Farinaccio et al., 2024). Preserving and restoring these services in the face of climate change and human activities is essential for maintaining planetary health (Knight and Holmes, 2018). We can uncover strategies by studying African deserts to mitigate the negative impacts of desertification, promote sustainable land management practices, and support the livelihoods of local communities while conserving biodiversity (Ziadat et al., 2022; Izah and Ogwu, 2023).

1.3 Objectives and scope of the chapter

The objectives of this chapter are to provide a comprehensive understanding of the Sahara Desert and other African deserts in the context of global biome conservation

and global warming. By exploring the unique ecological characteristics, biodiversity, and ecosystem dynamics of these deserts, we aim to shed light on the impacts of climate change on desert ecosystems and the associated implications for biodiversity conservation. The scope of this chapter encompasses an overview of the Sahara Desert, as well as other African deserts such as the Namib Desert and the Kalahari Desert. We delve into the geological history and landforms of these deserts, elucidating the processes that have shaped their current state. Furthermore, we examine the diverse flora and fauna that have adapted to survive in the extreme arid conditions of African deserts and explore the ecological functions and interactions within these ecosystems.

A significant focus of this chapter is dedicated to the impacts of global warming on African deserts. We discuss the historical climate trends and projections for future climate change in these regions, drawing upon studies that analyze temperature changes, precipitation patterns, and desertification processes. The consequences of these changes on desert ecosystems, including shifts in species distribution (e.g., Brito et al., 2016; Karssene et al., 2017; Arar et al., 2019), biodiversity loss (e.g., Durant et al., 2014), and land degradation (e.g., Ziadat et al., 2022) are examined in detail. Another important aspect of this chapter is the exploration of human interactions with African deserts and the associated conservation efforts. We examined the traditional knowledge and sustainable land-use practices of Indigenous communities living in these regions, highlighting the importance of incorporating their perspectives in conservation strategies. Additionally, we discuss the various human activities that impact African deserts, such as agriculture, mining, grazing, and urbanization, and assess their implications for biodiversity and ecosystem services. The chapter examines case studies of the Sahara Desert, the Namib Desert, and the Kalahari Desert and provides in-depth insights into the environmental challenges, conservation initiatives, and unique ecological characteristics of these distinct desert ecosystems.

2 Geographic location

The African continent is home to several notable deserts (Fig. 7.2), each with its distinct characteristics and ecological significance. Understanding the distribution and extent of these deserts is crucial for comprehending the unique biogeography, ecological dynamics, biodiversity conservation, and the potential impacts of climate change on African deserts. It allows for the identification of priority areas for conservation efforts, the development of sustainable land management strategies, and the preservation of unique desert-adapted species (Izah and Ogwu, 2023).

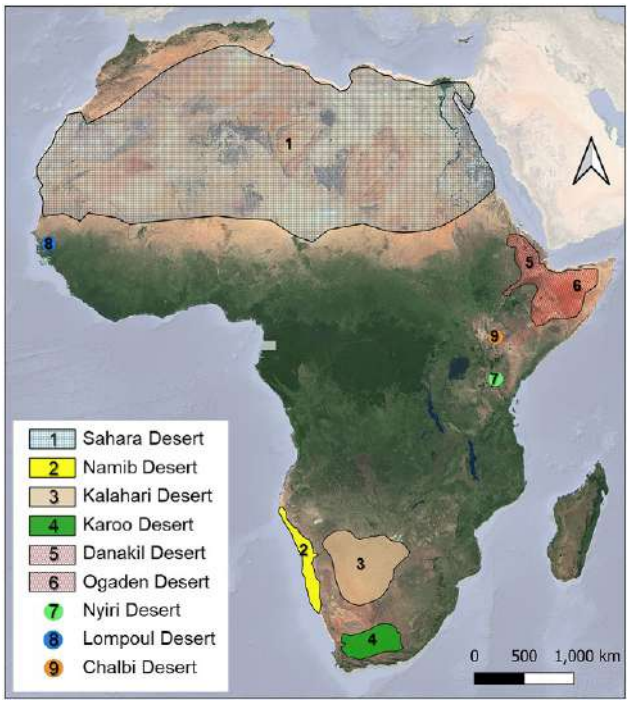

FIG. 7.2 Geographical location of African deserts. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: Original.)*

The Sahara Desert, located in northern Africa, is the largest hot desert globally, covering a vast expanse of approximately 9 million km^2. It spans across multiple countries, including Algeria, Chad, Egypt, Libya, Mali, Mauritania, Morocco, Niger, Sudan, and Tunisia. The Sahara Desert is renowned for its expansive sand dunes, rocky plateaus, and scarce vegetation, showcasing the arid nature of the region (Cloudsley-Thompson, 1984; Laity, 2009; Parsons and Abrahams, 2009). Recent studies have highlighted the increasing southward expansion of the Sahara Desert due to climate change, leading to desertification and land degradation (Terink et al., 2013; Ziadat et al., 2022). These changes have profound implications for local communities, biodiversity, and ecosystem functioning (Ouled Belgacem and Louhaichi, 2013; Whitford and Duval, 2019).

The Namib Desert, situated along the southwestern coast of Africa, is another prominent desert known for its unique coastal fog and ancient sand dunes. It extends across Namibia, Angola, and South Africa, covering an approximate area of 81,000 km^2, with 34,000 km^2 covered by the Namib Sand Sea (Knight and Holmes, 2018). The Namib Desert is considered one of the oldest deserts globally and supports a remarkable array of endemic plant and animal species adapted to its harsh and arid conditions. The fog that rolls in from the Atlantic Ocean provides a crucial source of moisture, sustaining the unique ecosystem

and enabling the survival of specialized organisms (Mitchell et al., 2020; Huntley, 2023).

The Kalahari Desert, located primarily in Botswana but also extending into parts of Namibia and South Africa, is a vast sand basin covering approximately 900,000 km^2 (Knight and Holmes, 2018). Unlike the typical perception of a desert, the Kalahari experiences more rainfall than other African deserts (Porporato et al., 2003), resulting in a diverse mosaic of grasslands, savannas, and dry woodlands (Thomas, 2002; Yu et al., 2017). The intermittent rivers and ephemeral pans found in the Kalahari support a rich diversity of plant and animal life, including unique desert-adapted species such as the Kalahari lion and meerkats (Lovegrove, 2021).

Other smaller African deserts include the Libyan Desert in Egypt and Libya, which can be included with the Sahara, the Danakil Desert located between Ethiopia, Eritrea, and Djibouti, the Ogaden Desert in Ethiopia, and the Karoo Desert in South Africa (Fig. 7.2). Each of these deserts exhibits its own distinct features, ecological communities, and environmental challenges (Laity, 2009). These deserts can be distinguished by the following features:

The Danakil Desert (or Afar Desert), covers an area of 136,956 km^2, including Ethiopia, Eritrea, and Djibouti, and is one of the hottest and most hostile environments on the planet (Butler, 2016; Tesfai, 2020). The local geology is characterized by hot sulfurous springs, salt crusts and multicolored minerals, rifts, faults, black lava, steamy geysers, and active volcanoes (Cieśluk et al., 2014).

The Ogaden region is located on the Somali Plateau, in southeast Ethiopia and encompasses about 300,000 km^2. The region has vast red soil plains. Climatically, it corresponds to the hot arid region of Ethiopian where annual rainfall is less than 400 mm. In geological terms, it is the area where sedimentary rocks occur on or near the surface (Mège et al., 2015).

Karoo Desert: more than 90% of South Africa is either arid or semiarid and drought is a characteristic feature of the climate (Rouault and Richard, 2003; Knight and Holmes, 2018). The Succulent Karoo is associated with drier and warmer conditions, and is found on plains and lower slopes with annual rainfall between 20 and 300 mm and summer temperatures reaching up to 44°C during summer (Esler et al., 2015).

Recent studies utilizing remote sensing data, climate modeling, and geospatial analysis have contributed to a more accurate understanding of the impact of land use land cover (LULC) change on the distribution and extent of African deserts. These studies provided valuable insights into the spatial patterns, environmental dynamics, and potential future changes in desert ecosystems as well as the neighboring arid and semiarid regions (Schultz, 2006; Arar and Chenchouni, 2012; Masih et al., 2014; Rohde et al., 2019; Li et al., 2020; Govender et al., 2022).

3 Major landforms and geomorphic features

The Sahara Desert encompasses a diverse array of landscapes "desertscapes," featuring both active and relict geomorphological formations, some of which are found in other biomes, while others are distinctively Saharan. Among these prominent landforms are the Reg, characterized by rocky landscapes strewn with pebbles and minimal fine elements, covering approximately 80% of the desert's expanse. The hamada is another significant feature, consisting of vast plateaus bounded by steep cliffs, exemplified by formations like the Hammada of Guir and the Hammada of Tindouf along the Algerian-Moroccan borders. The erg represents expansive sandy plains, such as the Saharan Grand Erg. Other notable geomorphic elements include Dayas, closed depressions dotting the landscape, salty soils like Sebkas and Chotts, verdant oases providing pockets of habitation amidst aridity, and towering mountain ranges like Tassili n'Ajjer and the Hoggar (Parsons and Abrahams, 2009; Bouallala et al., 2020; Balbo, 2024).

The deserts of South Africa, including the Kalahari, Namib, and Karoo Deserts, exhibit a diverse array of landforms and geomorphic features shaped by their unique geological histories and environmental conditions. In the Kalahari Desert, vast expanses of red sand characterize the landscape, interspersed with grasslands, savannas, and sparse vegetation. Large salt pans, such as the Makgadikgadi Pan, dot the terrain, formed from ancient lake beds and now serving as critical habitats for wildlife during seasonal floods. The Namib Desert, renowned for its towering sand dunes, boasts some of the highest dunes in the world, including the iconic Dune 45 and Big Daddy. These dunes, sculpted by wind and coastal processes, stretch for hundreds of miles along the Atlantic coastline, creating a dramatic contrast between desert and sea. In the Karoo Desert, rugged mountain ranges, rocky plateaus, and expansive plains dominate the landscape. The region is characterized by unique geological formations, such as the Valley of Desolation, with its towering dolerite columns, and the Great Karoo basin, home to fossil-rich sedimentary deposits dating back millions of years. These diverse landforms not only shape the physical environment but also support rich biodiversity and cultural heritage, making the South African deserts vital ecosystems worthy of conservation and study (Rouault and Richard, 2003; Parsons and Abrahams, 2009; Knight and Holmes, 2018).

4 Climate and climatic variations

Climate and climatic variations play a crucial role in shaping the unique characteristics and dynamics of desert environments. The climatic conditions experienced by

these deserts are influenced by various factors, including atmospheric circulation patterns, proximity to large water bodies, and global climate systems (Williams, 2014; Pausata et al., 2020). Climate and climatic variations in African deserts critically helped to assess the vulnerability of these ecosystems to ongoing and future climate change, especially by providing essential insights into the potential impacts on biodiversity, ecosystem services, and the livelihoods of local communities (Dube et al., 2016; Arar et al., 2019; Bouzekri et al., 2023). The incorporation of climate projections and variability into conservation and land management strategies ensure the long-term resilience and sustainability of African ecosystems (Ackerl et al., 2023).

The Sahara Desert, being the largest hot desert in the world, experiences a hyperarid climate characterized by extremely low annual precipitation and high temperatures (Cloudsley-Thompson, 1984; Laity, 2009). With mean annual temperatures exceeding 30°C, temperatures can rise over 50°C in the hottest months, and temperatures can fall below freezing in the winter (Bradai et al., 2015c). A single daily variation of −0.5°C to 37.5°C has been recorded. The Sahara is also extremely windy. Hot, dust-filled winds create dust devils which can make the temperatures seem even hotter. Studies have shown that the Sahara Desert has undergone significant climatic variations over the past few millennia. For instance, sedimentary records from the region indicate periods of increased aridity known as "megadroughts" during the Holocene, which had substantial impacts on vegetation and ecosystem dynamics (Claussen et al., 2003; van der Meeren et al., 2022). Recent research utilizing climate models and paleoclimate data suggests that the Sahara Desert is expected to experience further aridification and expansion under future climate change scenarios (Pausata et al., 2020).

In the Namib Desert, the climate is influenced by the cold Benguela Current, which runs parallel to the coast. This current creates a unique foggy environment and leads to a distinct coastal desert climate. Recent studies have highlighted the significance of fog as a critical water source for the Namib Desert ecosystem, particularly for desert-adapted organisms and specialized vegetation (Mitchell et al., 2020; Huntley, 2023). Changes in the intensity and frequency of fog events due to climate change can have profound implications for the Namib Desert's biodiversity and ecosystem functioning (Schultz, 2006; Rohde et al., 2019; Liu and Zhou, 2021; Naidoo et al., 2022; Treonis et al., 2022).

The Kalahari Desert experiences a semiarid climate, characterized by higher average annual precipitation compared to other African deserts. However, it still undergoes significant climatic variations (Yu et al., 2017). Batisani and Yarnal (2010) revealed an increasing variability in precipitation patterns within the Kalahari region, with evidence of both prolonged drought periods and extreme rainfall events. These fluctuations in precipitation can lead to fluctuations in vegetation growth, affecting the distribution and abundance of plant and animal species in the region (Mayaud et al., 2017; Yu et al., 2017; Knight and Holmes, 2018).

In recent years, advancements in climate modeling and remote sensing technologies have provided valuable insights into the climatic variations within African deserts. These studies have improved our understanding of the temporal and spatial patterns of rainfall, temperature, and other climate variables especially drought (Fig. 7.3) in these arid regions (Maestre et al., 2021; Masih et al., 2014). Additionally, paleoclimate reconstructions based on sedimentary records and proxy data have contributed to unraveling the past climatic variations and their impacts on African desert ecosystems (Dirmeyer and Shukla, 1996; Claussen et al., 2003; Tierney et al., 2017; van der Meeren et al., 2022).

5 Desertification processes and their drivers

Several factors have contributed to the expansion of dry lands in the African continent and the transformation from the steppe and savanna landscapes to the desert and subdesert ecosystems. These factors directly impact the desertification process or induce other forms of environmental changes that result in land degradation deterioration (Benabderrahmane and Chenchouni, 2010; Vogt et al., 2011; Macaulay, 2014). According to the United Nations Convention to Combat Desertification, land degradation is attributed to both climatic variations and human activities. Climate change is of great importance in the bioclimatic parameter pattern changes that are considered the main biophysical factor that has driven the growth of arid and semiarid regions, as well as have prolonged drought periods (Benaradj et al., 2021; Sivakumar, 2007).

Geomorphological processes, including sand encroachment, wind erosion, siltation, Aeolian deposition, and sandstorms, in sandy deserts, combined with prolonged periods of severe drought and human-induced factors, have significantly contributed to the recent formation of sandy lands (Mainguet and Chemin, 1991). This is evident in the emergence of new Aeolian landforms and geological features such as nabkha, sand dunes, loess deposits, wind fences, and sand sheets (Dougill and Thomas, 2004). Additionally, these processes have led to adverse changes in the chemical and physical properties of soils. This includes an increase in sand content, altering the soil texture (Bouzekri et al., 2023), and a decrease in organic matter and water content (Reich et al., 2001; Schleicher et al., 2011).

Human actions that started 5000 years ago continue to cause the destruction and degradation of African environments (Grove, 1974). The clearance of forests for

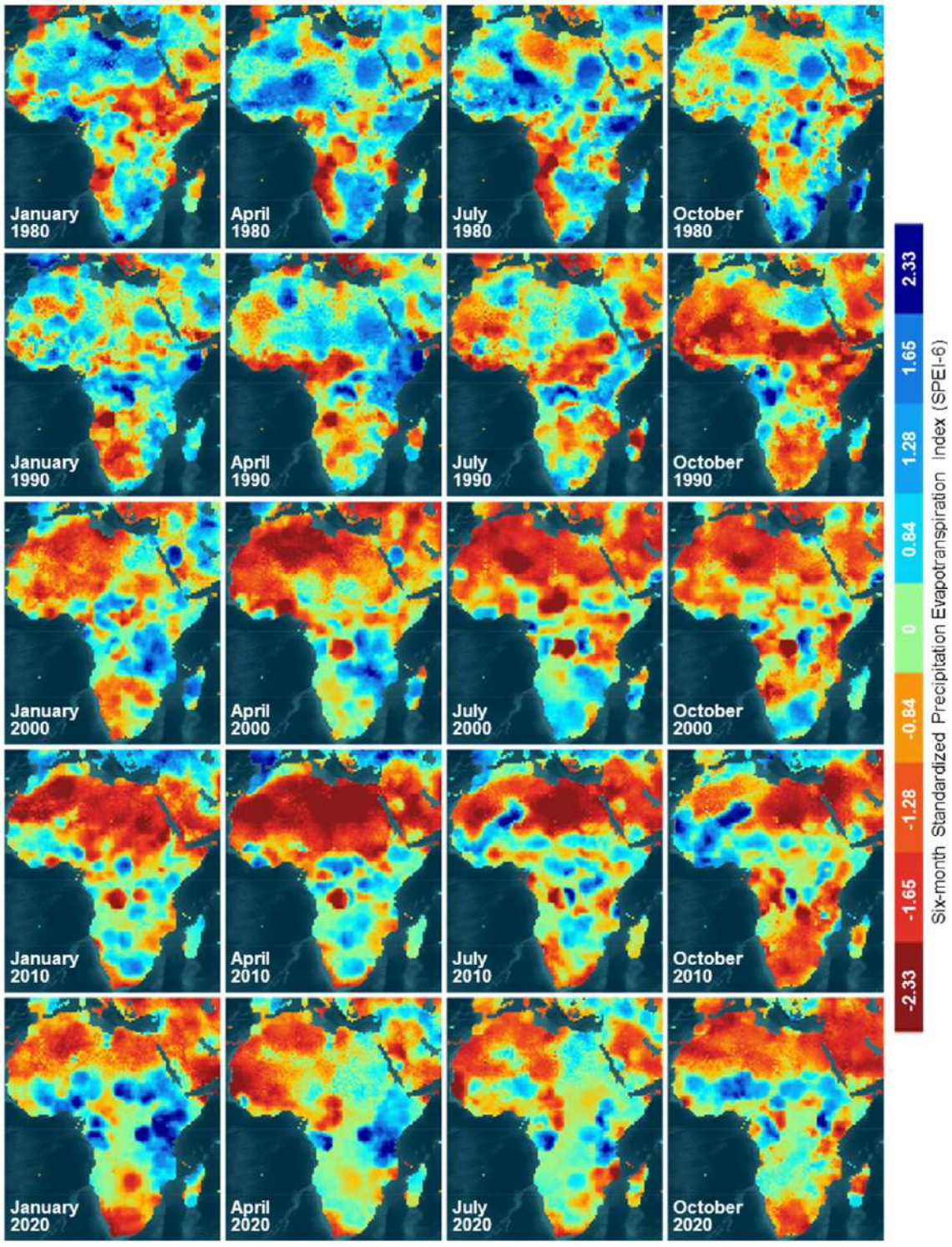

FIG. 7.3 Six-month Standardized Precipitation Evapotranspiration Index (SPEI-6) of the African continent and the Arabian Peninsula for the period 1980–2020. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Data source: Global SPEI database available at https://spei.csic.es/database.html. Photo credits: Original.)*

agriculture, historical and current practices of using fire, and the utilization of trees for charcoal and fuelwood have contributed to an escalating deforestation rate. This reduction in vegetation cover hinders the natural protection of soil against erosion and degradation in one hand. On the other hand, deforestation has converted large areas of forests into shrublands (Timberlake et al., 2010). Overgrazing in arid and semiarid African rangelands drives the expansion of bare land at the expense of shrublands, particularly in North Africa and the Saharan desert. Excessive grazing by livestock has led to the degradation of vegetation cover, as animals consume vegetation at a rate faster than it can regenerate. This process results in soil erosion caused by both wind and water (Dodd, 1994). Rapid population growth in many African countries, combined with poverty and a lack of alternative livelihoods, has necessitated changes in land use. This involves the conversion of natural ecosystems for purposes such as urbanization, infrastructure development, and agriculture. These changes contribute to accelerating the process of desertification (UNCCD, 2022).

6 Biodiversity

Diversity can be approached on many scales: genetically (intrapopulation), specifically (interpopulations) and on a large scale (ecosystems). A temporal scale is also adopted, ranging from a few hours in the case of bacterial diversification to geological timescales for continental differentiation. According to Linder (2014), symmetrically to the equator, Africa is rich in biodiversity paradoxes. The biodiversity within African deserts encompasses a diverse range of species, each playing a crucial role in maintaining the ecological balance of these environments (Ozenda, 1983; Thomas, 2002; Izah and Ogwu, 2023; Li et al., 2024).

6.1 Species richness and endemism

Centers of endemism with concentrations of species found nowhere else are linked to stable climate refugees, insularity, and geographical barriers (Harrison and Noss, 2017), making them particularly vulnerable to habitat loss and environmental change (Myers et al., 2000). Linder (2014) observed a remarkable diversity gradient in Africa, characterized by low species richness combined with low endemism in the central Sahara regions, and the opposite in the south-western tip of the continent, in Cape Town, with species richness reaching 25% of all African species (Manning and Goldblatt, 2013).

The Sahara, Namib, Kalahari, and other African deserts each host a unique assemblage of microbes, plants, and animals adapted to their specific climatic and geological conditions (Huntley, 2023; Izah and Ogwu, 2023; Li et al., 2024). Studying the distribution and abundance of endemic species in African deserts is essential for understanding their evolutionary history and implementing effective conservation measures to safeguard their survival (Brito et al., 2016).

Depending on the rainfall pattern, there are probably two Arid Floras in Africa: a winter-rainfall regime in the south-west African coast and the Namib Desert and a summer rainfall regime along the eastern part of the continent. The last flora is often referred to as succulent thicket, thicket, or seasonally dry vegetation (Schrire et al., 2005). In all cases, in Sub-Saharan Africa, despite the harsh conditions, they harbor a surprising richness of species, including approximately 45,000 species in 29 Mkm^2 compared to 900,000 in South America (17.84 Mkm^2) and 42,000 species in Malesia (3 Mkm^2) (Davis, 1995; Klopper et al., 2007). This flora is predominantly composed of xerophytes and ephemeral plants, locally known as Acheb, with halophytes thriving in moister areas (Bradai et al., 2015a; Bouallala et al., 2020, 2023), the flora boasts one nearly endemic family, along with several isolated monotypic genera with both broad and narrow distribution ranges, and potentially up to 162 endemic species (Zahran and Willis, 1992). The presence of monotypic genera suggests a Tertiary origin, with likely extinction of transitional forms (Cloudsley-Thompson, 1984). Vegetation is sparse along the wadis and dayas, featuring species like *Acacia* sp., *Tamarix* sp., and *Calotropis procera* (Aiton) W.T. Aiton, while hammadas, where groundwater is sufficient, support vegetation such as *Acanthorrhinum ramosissimum* (Coss. & Durieu) Rothm. and *Ononis angustissima* Lam. (Quézel, 1965). Contrary to common belief, the fauna of the Saharan habitats exhibits richness despite its hyperarid conditions (Chenchouni, 2012).

Lewin et al. (2016) reported that reptile, amphibian, bird, and mammal richness is largely congruent and responds similarly to environmental variables (mainly productivity and rainfall). This ecoregion is home to approximately 70 mammal species, including 20 large mammals, around 90 resident bird species, and nearly 100 reptile species (Le Berre, 1989). Arthropods, particularly ants, are also abundant. Endemic fish species to the Sahara Desert include the Desert Pupfish [*Cyprinodon alvarezi* (Miller, 1976)], Sahara Barb [*Enteromius deserti* (Pellegrin, 1909)], Nile Perch [*Lates niloticus* (Linnaeus, 1758)], and Sahara Catfish [*Clarias gariepinus* (Burchell, 1822)]. Among the endemic amphibians and reptiles of the Sahara are the Sahara Toad [*Sclerophrys mauritanica* (Schlegel, 1841)], Sahara Frog [*Pelophylax saharicus* (Boulenger, 1913)], Sahara Sand Viper [*Cerastes vipera* (Linnaeus, 1758)], Sahara Chameleon [*Chamaeleo africanus* (Laurenti, 1768)], and Sahara Desert Monitor [*Varanus griseus* (Daudin, 1803)]. Endemic bird species of the Sahara Desert encompass the Bar-tailed Lark [*Ammomanes*

cincturus (Gould, 1839)], Desert Sparrow [*Passer simplex* (Lichtenstein, 1823)], Tristram's Warbler [*Sylvia deserticola* (Tristram, 1859)], *Saharan olivaceous* Warbler [*Iduna pallida elaeica* (Lindermayer, A, 1843)], and African Silverbill [*Euodice cantans* (Gmelin, 1789)]. Although the number of endemic species remains relatively small given the vast size of the ecoregion (Le Berre, 1989; Cox et al., 2006; Escoriza and Ben Hassine, 2019; Mouane et al., 2024).

Historically, the critically threatened Addax likely inhabited this ecoregion but is now likely extirpated. Small populations of Scimitar-horned Oryx (*Oryx dammah*, EX) may have also been present in the past. Other desert antelope species, albeit in small numbers, may still be found, including the slender-horned gazelle (*Gazella leptoceros*, EN), dama gazelle (*Gazella dama*, EN), and red-fronted gazelle (*Gazella rufifrons*, VU) (Durant et al., 2014; Newby et al., 2016).

The plants and animals of the Sahara face significant threats from desiccation, with plant leaves capable of drying out completely before recovering and animals able to endure the loss of 30%–60% of their body mass and still recover. Many animals in this environment rely solely on metabolic processes for water intake, showcasing remarkable adaptations that enable survival in such harsh conditions (Sánchez-Montoya et al., 2017; Wharton, 2002; Huntley, 2023).

6.2 Adaptations of flora and fauna to desert conditions

Flora and fauna in African deserts have developed extraordinary adaptations to cope with limited water availability, extreme temperatures, and nutrient-poor soils (Neffar et al., 2022; Sánchez-Montoya et al., 2017). Plants in these regions often possess specialized mechanisms to minimize water loss, such as reduced leaf surface area, deep root systems to access underground water sources, and succulent tissues for water storage (Bezzalla et al., 2018; Tariq et al., 2024). Examples include the iconic Baobab tree (*Adansonia digitata* L.), known for its ability to store water in its trunk during dry periods, and the *Welwitschia mirabilis* Hook. f., a unique plant species found in the Namib Desert, which can survive for centuries with minimal water intake (Bombi et al., 2021; Huntley, 2023).

Similarly, desert-adapted animals have evolved physiological and behavioral adaptations to thrive in arid environments. Some species, like the Fennec fox [*Vulpes zerda* (Zimmermann, 1780)] and the Addax antelope [*Addax nasomaculatus* (Blainville, 1816)], have developed large ears or specialized nasal structures to dissipate heat and conserve water (Durant et al., 2014; Karssene et al., 2017). Others, such as the one-humped camel [*Camelus dromedarius* (Linnaeus, 1758)] and the Desert Tortoise [*Gopherus agassizii* (Cooper, 1863)], are capable of storing water for extended periods, allowing them to survive in water-scarce regions (Sánchez-Montoya et al., 2017; Fuller et al., 2021).

According to the phytogeographical classification of White (1983), the ecoregion falls under the Sahara regional transition zone. Perennial vegetation is sparse throughout most of the ecoregion, typically restricted to areas where groundwater reaches the surface or where runoff occurs. Plant diversity is more pronounced in the Western Sahara compared to the eastern region due to lower rainfall. The vegetation exhibits distinct Sahara-Arabian affinities, with remarkable adaptations to arid conditions (World Wildlife Fund and Hogan, 2014). While large expanses of ergs and regs may remain devoid of visible plant life for extended periods, following rainfall, vegetation cover can exceed 50% on sand dunes and 20% on gravel plains (WWF and IUCN, 1994).

6.3 Unique desert ecosystems and their ecological functions

Although 6% of the world's population lives in deserts subject to very hostile environmental conditions, they remain largely marginalized (Mortimore et al., 2009), peripherally perceived and neglected by political and business communities (Middleton et al., 2011).

Desert ecosystems in Africa are not only biologically diverse but also serve essential ecological functions. Despite their seemingly barren appearance, these ecosystems play crucial roles in nutrient cycling, soil stabilization, biodiversity maintaining, and supporting local livelihoods (Li et al., 2024). The unique flora and microorganisms of African deserts contribute to soil fertility through nutrient accumulation and fixation, facilitating the growth of other plant species and providing habitat and food for a variety of animals (Rejili et al., 2012; Hanna et al., 2013; Karabi et al., 2016; Abd El-Gawad, 2021). Additionally, desert ecosystems act as carbon sinks, helping to mitigate climate change by sequestering carbon dioxide from the atmosphere (Loarie et al., 2009; Lal, 2019), containing an estimated one-third of terrestrial global carbon stock (Trumper et al., 2008). These ecological functions of African desert ecosystems are vital for devising sustainable management strategies that balance conservation with human development (Dhawi, 2023; Izah and Ogwu, 2023).

7 Conservation areas and their ecological indicators

Because of their low productivity, deserts have never been considered a conservation priority and have received little attention (Bellard et al., 2014). Yet, these ecosystems have proved to be unique refuges for rare species that survive in

harsh and highly heterogeneous environments (O'Connell and Hallett, 2019).

While global attention has been focused on hotspots, the world's largest tropical desert, the Sahara, has suffered a catastrophic decline in megafauna. Of 14 large vertebrates that have historically occurred in the region, four are now extinct in the wild, including the iconic scimitar-horned oryx (*O. dammah*). The majority has disappeared from more than 90% of their Saharan range, including *Chlamydotis jubatus hecki*—all now on the brink of extinction (Saleh et al., 2001; Durant et al., 2014; Vale and Brito, 2015). Vertebrate species are declining at rates 100–1000 times higher than those in evolutionary history (Dirzo and Raven, 2003), and climate change is projected to increase extinction rates further (Foden et al., 2013).

In particular, desert biodiversity has attracted relatively little conservation finance and action (Davies et al., 2012), although deserts cover 17% of the world's land mass and harbor surprisingly high biodiversity (Safriel et al., 2005), despite their low primary productivity and consequent low biomass. For example, deserts are home to 25% of terrestrial vertebrate species and, combined with xeric shrublands, are among the top three richest biomes for terrestrial vertebrates (Millennium Ecosystem Assessment, 2005).

Conservation efforts in African deserts are concentrated on recognizing and preserving areas with significant ecological value, safeguarding distinctive habitats, and defending endangered species. Grasping the distribution of protected areas and biodiversity hotspots in African deserts offers crucial insights into conservation priorities and indicators for evaluating the efficacy of conservation strategies (Durant et al., 2014; Zhang et al., 2023; Li et al., 2024). Achieving effective conservation and management demands a comprehensive approach that incorporates scientific research, community involvement, policy formulation, and adaptable governance (Brito et al., 2021). Through the utilization of ecological indicators to evaluate conservation results and adjust management tactics, stakeholders can collaborate to guarantee the enduring sustainability and resilience of desert ecosystems for future generations (Goffner et al., 2019; Izah and Ogwu, 2023).

7.1 Protected areas and biodiversity hotspots

Protected areas play a crucial role in conserving biodiversity and maintaining ecosystem functions within African deserts (Dudley et al., 2014; Durant et al., 2014; Izah and Ogwu, 2023; Li et al., 2024). These areas are designated to safeguard key habitats, species, and ecological processes from threats such as habitat destruction, overexploitation, and climate change (Gross et al., 2013; Ogwu et al., 2023). In African deserts, protected areas range from national parks and wildlife reserves to community-managed conservancies, each contributing to biodiversity conservation and sustainable land management.

7.1.1 National parks

Several national parks within African deserts serve as important conservation strongholds, providing refuge to a diverse array of plant and animal species. These parks are often characterized by unique desert landscapes, endemic species, and iconic megafauna. Examples include Tassili n'Ajjer National Park in Algeria (Djouhri, 2012), Namib-Naukluft National Park in Namibia (Dowling and Grünert, 2018), and Aïr and Ténéré Nature Reserve in Niger (Crawhall and Ewangaye, 2017).

7.1.2 Wildlife reserves

Wildlife reserves in African deserts are designated to protect critical habitats and facilitate the conservation of threatened species in their natural habitats. These reserves often collaborate with local communities to promote sustainable land use practices and mitigate human-wildlife conflicts (Durant et al., 2014). Notable examples include the Skeleton Coast Conservancy in Namibia (Wassenaar et al., 2021) and the Termit and Tin-Toumma National Nature Reserve in Niger (Rabeil et al., 2008; Newby et al., 2016).

7.1.3 Community-managed conservancies

Community-managed conservancies are emerging as effective models for conservation and sustainable development in African deserts. These conservancies empower local communities to actively participate in natural resource management, biodiversity conservation, and ecotourism initiatives. Examples include the Kunene Conservancy in Namibia (Störmer et al., 2019) and the Ennedi Natural and Cultural Reserve in Chad (Ghienne et al., 2023).

7.1.4 Biodiversity hotspots

Biodiversity hotspots are regions of high species richness and endemism that face significant threats and require urgent conservation action (Durant et al., 2014; Izah and Ogwu, 2023). Within African deserts, biodiversity hotspots encompass areas with unique ecological features, endemic species, and evolutionary significance. The identification and protection of these hotspots is essential for preserving desert biodiversity and ecosystem resilience. Examples include the Succulent Karoo hotspot in South Africa (Hoffman et al., 2009) and the Mediterranean Basin hotspot in North Africa (Slimani et al., 2022).

7.2 Ecoindicators for conservation assessment

The ecological health and conservation status of African deserts relies on the assessment of a range of indicators that measure habitat integrity, species abundance, and ecosystem functioning. These ecoindicators provide valuable insights into the effectiveness of conservation measures and help guide management decisions for sustainable development (Zhang et al., 2023). Key ecoindicators for conservation assessment in African deserts are as follows:

7.2.1 Species diversity

Monitoring species diversity and composition in different ecosystems, habitats, and land use types provides an indication of ecosystem health and resilience (Zeller et al., 2017; Naidoo et al., 2022). The identification of priority conservation areas and potential ecological disturbances can be achieved by examining changes in species richness, evenness, and distribution patterns (Brito et al., 2016).

7.2.2 Habitat connectivity

Evaluating habitat connectivity and fragmentation allows for the assessment of landscape-scale conservation efforts. Maintaining or restoring habitat connectivity is essential for facilitating species movement, genetic exchange, and ecological processes such as migration and dispersal (Crooks et al., 2011; Wegmann et al., 2014; Sánchez-Montoya et al., 2017).

7.2.3 Population trends

Monitoring population trends of indicator species helps gauge the effectiveness of conservation interventions and detect early warning signs of ecological imbalances. Population assessments provide valuable data on species abundance, distribution, reproductive success, and habitat requirements (Andrews and O'Brien, 2010; Bouallala et al., 2020; Vickery et al., 2023).

7.2.4 Ecosystem services

Assessing ecosystem services provided by African deserts, such as carbon sequestration, soil fertility, and water regulation, offers insights into the socioeconomic benefits of conservation actions (Stringer et al., 2012). Understanding the linkages between ecosystem services and human well-being is essential for promoting sustainable land management and community resilience (Alexander et al., 2016).

7.2.5 Threats and pressures

Identifying and quantifying anthropogenic threats and pressures on African desert ecosystems, including habitat loss, climate change, poaching, and invasive species, informs conservation planning and prioritization. Monitoring changes in land cover, habitat degradation, and species vulnerability helps guide adaptive management strategies and mitigate potential risks (Brito et al., 2021; Ziadat et al., 2022; Li et al., 2024).

7.3 Ecoindicators for assessing desert ecosystem health and resilience

Ecological indicators serve as vital tools for evaluating the status of the environment, tracking changes over time, and guiding conservation efforts (Burke et al., 2008; Rita et al., 2017). To evaluate the health and resilience of desert ecosystems in Africa, the identification and monitoring of key ecological indicators are required (Hiernaux et al., 2016; Measho et al., 2021). By regularly and systematically monitoring these ecological indicators, conservationists and land managers can effectively assess the health and resilience of desert ecosystems, identify emerging threats, and implement targeted conservation interventions to mitigate environmental degradation and promote ecosystem sustainability (Ziadat et al., 2022; Izah and Ogwu, 2023). Some commonly used ecological indicators for assessing desert ecosystem health and resilience are as follows.

7.3.1 Vegetation cover

Monitoring the extent and density of vegetation cover provides insights into the overall health and productivity of desert ecosystems (Almalki et al., 2022). Changes in vegetation cover can indicate shifts in ecological processes, such as desertification or habitat degradation (D'Odorico et al., 2013; Rohde et al., 2019).

7.3.2 Species diversity and abundance

Assessing the abundance, composition, diversity, and functional potential of microbes and plant and animal species within desert environments helps gauge ecosystem stability and resilience. Declines in species diversity or population sizes may signal ecological disturbances or habitat fragmentation (Chenchouni, 2012; Newby et al., 2016; Bouallala et al., 2020, 2023; Treonis et al., 2022; Naidoo et al., 2022).

7.3.3 Water availability

Monitoring water availability, including groundwater levels, surface water flow, and precipitation patterns, is critical for understanding the sustainability of desert ecosystems (Al-Blooshi et al., 2020; Negm et al., 2020). Changes in water availability can impact vegetation growth, wildlife habitat, and ecosystem functioning (Sánchez-Montoya et al., 2017; Kapuka and Hlásny, 2021; Li et al., 2024).

7.3.4 Soil health

Evaluating soil properties such as nutrient content, organic matter, microbial activities, and soil stability provides valuable information on ecosystem health and resilience (Neffar et al., 2013, 2015). Degradation of soil health can lead to decreased productivity, increased erosion, and loss of habitat for desert-adapted species (Safriel, 2009; Brito et al., 2021).

7.3.5 Indicator species

Identifying indicator species that are particularly sensitive to environmental changes can help assess the overall condition of desert ecosystems. These species serve as early warning signs of ecological stressors and provide valuable insights into ecosystem dynamics (Siddig et al., 2016). Brito et al. (2016) underscored the necessity of tailored conservation strategies, noting significant variation in conservation challenges across vertebrate taxa. Referring to their findings, the distribution of IUCN conservation statuses among reptiles, breeding birds, and mammals in the Sahara-Sahel region reveals three reptile species classified as critically endangered (CR), one as endangered (EN), and six as vulnerable (VU). Mammals include 1 species extinct in the wild (EW), 4 classified as CR, 4 as EN, and 15 as VU. Breeding birds comprise four endangered and eight vulnerable species.

7.4 Management strategies for habitat conservation

Effective management of protected areas in African deserts is essential for preserving biodiversity, protecting natural habitats, and promoting sustainable development (Riggio et al., 2019). By implementing these management strategies in conservation areas across African deserts, stakeholders can work toward achieving conservation goals, maintaining ecosystem integrity, and securing the long-term sustainability of desert biodiversity and ecosystem services (Izah and Ogwu, 2023; Zhang et al., 2023). Implementing appropriate management strategies involves a combination of scientific research, stakeholder engagement, policy formulation, and on-the-ground conservation actions (Brito et al., 2021). Some key management strategies for conservation areas in African deserts include:

7.4.1 Habitat protection

Establishing protected areas and wildlife reserves to safeguard critical habitats for desert-adapted species (Brito et al., 2021). Designating buffer zones and implementing land-use zoning regulations to minimize human impacts on sensitive ecosystems (Li et al., 2024). For example, as mountains and plains are potential refuges in the Sahara-Sahel, connectivity between them and the establishment of corridors would increase the area of protection (Loarie et al., 2009; Vale et al., 2015).

7.4.2 Invasive species control

Implementing measures to control and eradicate invasive plant and animal species that threaten native biodiversity and disrupt ecosystem processes (Marchante et al., 2021). Utilizing invasive species monitoring programs and targeted eradication efforts to prevent further and spread minimize ecological damage (Martinez et al., 2020).

7.4.3 Community engagement

Engaging local communities in conservation initiatives through education, capacity building, and sustainable livelihood programs. Involving Indigenous peoples and traditional land users in decision-making processes and resource management activities to ensure cultural sensitivity and promote community ownership of conservation efforts (Durant et al., 2014; Izah and Ogwu, 2023).

7.4.4 Law enforcement and monitoring

Enforcing laws and regulations related to wildlife protection, habitat conservation, and natural resource management. Implementing monitoring and surveillance programs to detect and deter illegal activities such as poaching, habitat destruction, and illegal logging (Brito et al., 2021; Izah and Ogwu, 2023).

7.4.5 Climate change adaptation

Developing adaptation strategies to mitigate the impacts of climate change on desert ecosystems, such as habitat restoration, water conservation measures, and promoting climate-resilient land-use practices (Goffner et al., 2019). Integrating climate change considerations into conservation planning and management to enhance ecosystem resilience and adaptive capacity (Dixon et al., 2003; Armani et al., 2022).

8 Human activities and their impacts

The Sahara, a vast expanse primarily comprising sand and rock, interspersed with small pockets of permanent vegetation, represents a largely undisturbed habitat. Human-induced degradation is most pronounced in areas with access to water, such as oases, where significant alterations to the habitat occur due to human activities (Ziadat et al., 2022). Nomadic pastoralists and traders have historically removed existing tree cover for fuel and fodder, contributing to habitat alteration (Ramprasad et al., 2020). Despite its ecological significance, the Sahara Desert remains inadequately protected, a challenge exacerbated by its low population density and the logistical complexities of delineating borders across its expansive terrain. With fewer than

two million inhabitants scattered throughout the entire desert, the majority, including Tuareg, Tibbu, and Moors, engage in nomadic pastoralism, hunting, and trading, primarily inhabiting the desert margins rather than the central hyperarid region (Abdirahman, 2021).

Human activities significantly impact the delicate habitats of deserts, contributing to habitat degradation, loss of biodiversity, and environmental change. Driven largely by socioeconomic factors, these activities pose significant challenges to the conservation and sustainable management of desert ecosystems (D'Odorico et al., 2013; Li et al., 2024). Pastoralism, a primary human activity in African deserts, involves the grazing of livestock such as camels, goats, and sheep. Overgrazing by domestic animals can lead to soil erosion, depletion of vegetation, and fragmentation of habitats, thereby exacerbating processes of desertification (Le Houérou, 1989; Olagunju, 2015; Doghbage et al., 2024; Neffar et al., 2018; Zerga et al., 2018). Additionally, agricultural practices, including irrigation and cultivation in oasis areas, have the potential to alter natural hydrological cycles, disrupt native plant communities, and contribute to soil salinization (Shin et al., 2022). The expansion of agricultural activities often results in the conversion of natural habitats into croplands, further diminishing biodiversity and worsening habitat loss. Furthermore, water extraction for agricultural and domestic purposes can deplete aquifers, leading to groundwater depletion and desertification in some regions (Goyal et al., 2016; Hadidi et al., 2022; Hartani and Semar, 2024; Negm et al., 2020).

Mining activities, especially for minerals and fossil fuels, also pose significant threats to deserts. Such operations can lead to habitat destruction, soil contamination, and the release of pollutants into the environment, impacting both terrestrial and aquatic species (Magidi and Hlungwani, 2023). The construction of infrastructure, including roads, railways, and urban settlements, further fragments and degrades desert habitats, resulting in habitat loss and increased human-wildlife conflicts (Durant et al., 2014; Li et al., 2024). Moreover, while tourism offers economic opportunities, it can also have adverse effects on desert environments. Unregulated tourism activities, such as off-road driving, littering, and habitat disturbance, can degrade fragile ecosystems and disrupt wildlife behavior (Alrawaf et al., 2023). Climate change exacerbates the impacts of these human activities, leading to rising temperatures, altered precipitation patterns, and more frequent extreme weather events, further straining desert ecosystems and their inhabitants (cf. Section 4). Depending on the species' sensitivity and adaptive capacities, desert-living species could be particularly vulnerable to the magnitude and velocity of climate change as they may already live at their physiological limits (Vale and Brito, 2015). It's the case of diurnal ectotherms particularly sensitive to temperature change and present low adaptive capacity due to relatively sedentary behavior and small home ranges (Barrows, 2011).

Ephemeral habitats in the Sahara, which emerge transiently following rainfall, face fewer threats from human activities. However, areas with permanent water sources, such as oases or locations with shallow groundwater, experience intense local pressures on natural resources. Here, human activities exert significant stress on the remaining populations of large mammals adapted to desert conditions. Hunting, both for sustenance and recreational purposes, has drastically reduced the populations of such species, placing them at risk of extinction. The addax, for instance, is critically threatened due to excessive hunting pressure. Similarly, other desert-adapted antelopes persist in perilously low numbers, classified as endangered due to habitat degradation and hunting (Rabeil et al., 2008; Durant et al., 2014; Newby et al., 2016).

In recent decades, development projects in the deserts of Algeria and Tunisia have seen the implementation of irrigation schemes utilizing water pumped from underground aquifers (Hadidi et al., 2022; Negm et al., 2020). While aimed at bolstering agricultural productivity and supporting local livelihoods, these initiatives have inadvertently triggered adverse environmental consequences. Among the most pressing issues are soil degradation and salinization, predominantly resulting from inadequate drainage systems accompanying the irrigation infrastructure (Hartani and Semar, 2024). As water seeps into the soil, poor drainage exacerbates waterlogging, causing salt accumulation and compromising soil quality. Consequently, agricultural productivity is hampered, exacerbating the challenges faced by local communities reliant on these resources for sustenance. Addressing these issues demands integrated approaches that balance socioeconomic development with environmental sustainability, emphasizing efficient water management practices and holistic land-use planning to mitigate the adverse impacts of irrigation projects on desert ecosystems (Izah and Ogwu, 2023).

Global warming and desertification pose significant challenges to African deserts, exacerbating aridity, altering precipitation patterns, and leading to soil degradation. Human activities such as deforestation, overgrazing, and agriculture contribute to land degradation, disrupting natural ecosystems and threatening biodiversity (Ekka et al., 2023; Abdullahi et al., 2023; Doghbage et al., 2024; Li et al., 2024). Indigenous populations, with their traditional knowledge, offer valuable insights into sustainable land management practices and water resource utilization, which can inform conservation efforts and promote resilience in desert regions. Efforts to conserve African deserts require comprehensive strategies that integrate scientific research, traditional knowledge, and stakeholder collaboration (Brito et al., 2021; Zhang et al., 2023).

Conservation initiatives focus on protecting biodiversity, restoring degraded habitats, and promoting sustainable development. This involves establishing protected areas, implementing adaptive management approaches, and fostering community-based conservation efforts to safeguard desert ecosystems and mitigate the impacts of human activities and climate change (Vale and Brito, 2015; Izah and Ogwu, 2023; Chenchouni and Neffar, 2025).

9 Operational and potential monitoring tools of conservation and restoration

Monitoring tools are indispensable for conserving and restoring desert biomes, offering critical insights into ecosystem health and guiding management strategies. These tools encompass a variety of techniques and technologies tailored to evaluate different aspects of desert ecosystems, including biodiversity, habitat quality, and ecosystem function (Costantini et al., 2016; Izah and Ogwu, 2023). According to Ziadat et al. (2022) and Li et al. (2024), by integrating these tools, conservationists and land managers can efficiently assess, conserve, and rehabilitate desert biomes, safeguarding their long-term sustainability and resilience amidst environmental changes.

9.1 Remote sensing

Remote sensing techniques, including satellite imagery and aerial photography, offer valuable tools for monitoring desert ecosystems over large spatial scales. These methods can provide information on land cover changes, vegetation dynamics, species distribution, and habitat fragmentation, allowing for the identification of areas at risk and the evaluation of conservation interventions (Schultz, 2006; Masih et al., 2014; Brito et al., 2016; Rohde et al., 2019).

9.2 Geographic information systems

Geographic information systems (GIS) technology enables the integration and analysis of spatial data, facilitating the mapping of desert habitats, the delineation of conservation areas, and the assessment of landscape connectivity. By overlaying multiple layers of information, GIS can identify priority areas for conservation and guide land management decisions (Arar and Chenchouni, 2012; Hamadouche et al., 2014; Elbeih, 2021; Bouzekri et al., 2023).

9.3 Biodiversity surveys

Field surveys conducted by ecologists and biologists are fundamental for assessing biodiversity in desert ecosystems. These surveys involve the identification and monitoring of plant and animal species, including rare and endemic taxa, to track population trends, species distributions, and ecosystem health (Brito et al., 2016, 2021; Mouane et al., 2024; Treonis et al., 2022).

9.4 Noninvasive monitoring techniques

Employing technological methods like camera trapping to investigate wildlife populations and behavior in desert landscapes. Camera traps capture images and videos of elusive species, including mammals and birds, thereby yielding valuable data on species abundance, activity patterns, and interactions, which in turn support conservation endeavors (Kingsford et al., 2021; Harkins, 2022).

9.5 Climate monitoring

Continuous monitoring of climate variables, such as temperature, precipitation, and humidity, is crucial for understanding the impacts of climate change on desert ecosystems. Climate monitoring stations and networks provide long-term data that help researchers and managers assess trends, predict future changes, and develop adaptation strategies (Griffis-Kyle, 2016; Dube et al., 2016; Hiernaux et al., 2016).

9.6 Community-based monitoring

Involving local communities in monitoring efforts can enhance conservation outcomes by fostering stewardship and traditional ecological knowledge. Community-based monitoring programs engage residents in data collection, resource management, and decision-making processes, promoting sustainable practices and supporting local livelihoods (Keenan, 2013; Izah and Ogwu, 2023).

9.7 Indicators of ecosystem health

Developing and implementing indicators of ecosystem health, such as vegetation cover, soil moisture, and species diversity, provides a standardized framework for assessing the condition of desert biomes. These indicators serve as early warning signs of environmental degradation and guide management actions aimed at restoring ecosystem function and resilience (Ackerl et al., 2023; Zhang et al., 2023).

10 Conclusions and perspectives

African deserts, vital for global biodiversity and ecosystem stability, harbor diverse plant and animal species uniquely adapted to their harsh conditions. However,

human activities such as overgrazing, agriculture, mining, infrastructure development, and climate change threaten these ecosystems, leading to habitat degradation, biodiversity loss, and aggravating desertification impacts. Safeguarding African deserts is crucial not only for biodiversity conservation but also for supporting the livelihoods of people. Proactive conservation efforts aimed at protecting and restoring desert habitats can mitigate the impacts of climate change, conserve biodiversity, and promote sustainable development in arid regions. Accordingly, swift and decisive action is imperative to preserve these invaluable ecosystems for future generations. Efforts to conserve and manage these deserts require a holistic approach that integrates scientific research, community engagement, policy development, and adaptive governance. Effective conservation strategies should prioritize the protection of key habitats, the restoration of degraded ecosystems, and the sustainable use of natural resources. Additionally, there is a need for increased monitoring and surveillance to track changes in desert ecosystems and assess the effectiveness of conservation interventions. Looking ahead, addressing the conservation challenges facing African deserts will require concerted efforts at local, national, and international levels. Collaborative partnerships between governments, nongovernmental organizations, research institutions, and local communities are essential to develop and implement evidence-based conservation initiatives. Furthermore, raising awareness about the ecological importance of desert ecosystems and promoting sustainable land-use practices is crucial for securing the long-term health and resilience of these vital landscapes.

Future perspectives and challenges for African deserts under global warming entail anticipated shifts in climate patterns and increasing threats of desertification, compounded by human activities such as urbanization and infrastructure development. Sustainable development and conservation opportunities arise from renewable energy adoption, water-saving technologies, and ecotourism initiatives, offering potential solutions to mitigate climate impacts and support local communities. However, addressing research gaps and implementing interdisciplinary approaches are crucial for informing evidence-based conservation strategies and adaptive management practices, emphasizing the importance of integrating traditional ecological knowledge with scientific research to foster resilience and promote environmental stewardship in African deserts.

Conflicts of interests

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Section B

Insular biomes

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Chapter 8

Madagascar and Mauritius as insular biomes

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1 Introduction

Insular biomes are islands or land masses that are isolated from a main land mass. They have certain key features such as biogeographical isolation, endemism, habitat diversity, and low species diversity compared to land masses. They are often studied for conservation purposes. Such biomes are often more vulnerable to global changes including invasive alien species, pollution, habitat destruction, and climate change (Veron et al., 2019).

The ancestors of vascular plants migrated from sea to land during the Silurian period. Mangroves together with seagrasses form part of a unique group of plants that migrated from land to sea during the late cretaceous (100–66 Ma) and recolonized coastal and marine environments (Huntley, 2023). Mangroves are valued globally for their important roles in supporting biodiversity, coastal protection, livelihood support, and climate resilience, globally covering an area of 147,000 km^2 (14,700,000 ha) (Leal and Spalding, 2022).

Mauritius and Madagascar are part of the Western Indian Ocean. The four countries of the Western Indian Ocean, Kenya, Madagascar, Tanzania, and Mozambique, account for 745,518 ha (representing 25% of all mangroves of Africa and 5% of world mangroves), with Madagascar comprising 277,567 ha (37% among the four countries) (Erftemeijer et al., 2022). The mangroves of the insular biome, Mauritius was spread over 145 ha (Bosire et al., 2015) and is currently 181 ha (Republic of Mauritius, 2016). The current chapter focuses on mangroves of Madagascar and Mauritius and their characteristics in these insular countries.

2 Geographic location

The island of Mauritius is located in the southwest of the Indian Ocean and lying about 850 km east of Madagascar (Hammond et al., 2015) (Fig. 8.1) Together with Reunion and Rodrigues, it forms part of the Mascarene Plateau, which is located between latitudes of 21°7 and 19°40′ and longitudes 52°32′ and 63°30′ and it has an area of 1865 km^2 (Bhagooli and Kaullysing, 2019).

The island state of Madagascar is in the Indian Ocean off the eastern coast of southern Africa, east of Mozambique. Madagascar contains extensive mangrove ecosystems which, as of 2020, totaled 277,567 ha (Erftemeijer et al., 2022). This represents 2% of the global distribution and 1.4% of the total forest cover in the country. Madagascar is home to the second largest extent of mangroves in the Western Indian Ocean (WIO) region, representing about 10% of Africa's mangroves and about 37% of the mangroves in the WIO region (Erftemeijer et al., 2022). The vast majority (98%) of these mangroves are situated along the west coast of the country with largest stands occurring around Mahajamba Bay and Ambaro-Ambanja Bays (Jones et al., 2016). The east coast only has about 4260 ha of mangroves (Rakotomavo, 2018).

Mauritius is of volcanic origin with rocks aging from 10 to 0.1 million years (Newsome and Johnson, 2012). It has a coastline of 322 km and is bounded by a fringing reef and barrier reef of 300 km^2, and encloses a lagoon area of around 243 km^2. The coastal zone is characterized by a variety of habitats such as sandy beaches, sand dunes, rocky shores, wetlands, mangroves, corals, and coral reefs (Leckraj, 2021). As per the review and summary given in Bhagooli and Kaullysing (2019), the island is characterized by a central plateau of 400–500 m elevation, a collapsed caldera surrounded by several mountain ranges (the highest peak of 828 m in the southwest region, Black River Mountain range) and plains (mostly dominating north, east, and west parts). The island initially had a native forest, which was reduced to less than 2% in 1997 after 360 years of human colonization (Florens, 2013; Norder et al., 2017).

3 Biodiversity

Mauritius is known to be a hotspot of biodiversity, mainly due to the tropical climate, topography, and isolation, which have

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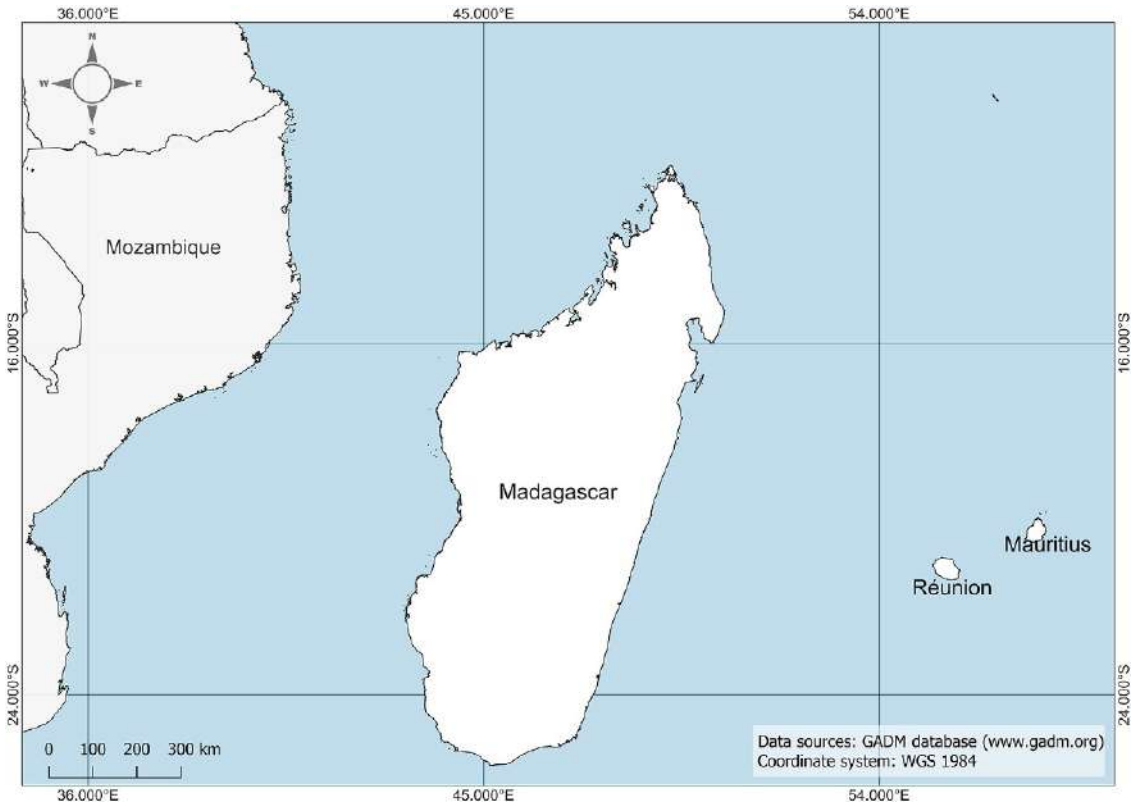

FIG. 8.1 Map to show the location of Madagascar and Mauritius in the Western Indian Ocean. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: Lalao Aigrette.)*

contributed to a diverse biota and a high degree of endemism (Republic of Mauritius, 2017). The terrestrial fauna and floral species diversity is better known compared to marine biodiversity. An overview on the number of native and endemic terrestrial species diversity with 691 angiosperms, 5 mammals, and 125 snails is reported by Florens (2013).

Madagascar is a megadiverse country with a high concentration of endemic species. Its ecosystems include many types of forests, savannah, steppes, rivers, lakes, wetlands, and drylands. With 5600 km of coastline, coastal areas are composed of natural environments that are among the richest and most diversified in the Indian Ocean region, including coral reefs, mangroves, phanerogam seagrass beds, estuaries, and coastal marshes.

In Madagascar, there are eight species of mangroves and in Mauritius two species are known (Bosire et al., 2015). The dominant species in Mauritius is *Rhizophora mucronata* and the other species is *Brugueira gymnorhiza*. As summarized in Appadoo et al. (2016), a number of plant associates such as the fern *Acrosticum aurearum* (Kathiresan and Rajendran, 2005), the shrub *Pemphis acidula*, and the endemic species *Sesuvium ayresi* occur in Mauritian mangroves. A number of salt-tolerant grasses (*Sporobolus virginicus*, *Paspalum vaginatum*, *Paspalidum germinatum*) and woody species can be present such as *Hibiscus tiliaceus*, *Thespesia populnea*, *Dendrolobium umbellatum*, and *Suriana maritima* have been reported by Tatayah (2002).

The mangroves in Mauritius are known to support a rich fungal biodiversity as demonstrated by Poonyth et al. (1999a). She reported on 67 species of fungi of which 61 were new records, collected from intertidal wood and decaying *R. mucronata* prop roots. The most dominant fungi belonged to the genus *Lulworthia*. Poonyth et al. (1999b) also reported three new species of fungi of the genus *Massarina*.

The mangroves of Mauritius support a rich faunal biodiversity (Fig. 8.2). These include resident and migratory birds. Although the mangroves house a diversity of terrestrial organisms, the terrestrial biodiversity (spiders, insects) is not well documented. Some faunal species of molluscs, crabs, and amphipods and fish families are reported in Appadoo et al. (2016).

Eight mangrove species in six families have been inventoried in Madagascar, including Rhizophoraceae (*R. mucronata*, *Bruguiera gymnorrhiza*, *Ceriops tagal*), Avicenniaceae (*Avicennia marina*), Sonneratiaceae (*Sonneratia* alba), Combretaceae (*Lumnitzera racemosa*), Meliaceae (*Xylocarpus granatum*), and Sterculiaceae (*Heritiera littoralis*).

Over 23 lemur species belonging to five lemur families (Cheirogaleidae 7, Lepilemuridae 3, Lemuridae 9, Indriidae 3,

FIG. 8.2 Mangroves of Mauritius. Biodiversity in mangrove habitat (A, B, and C) and Poste de Flacq mangroves (D). *(Photo credit: Jayshnee Mungla (A, B, and C) and Raphael Merven (D).)*

and Daumbentoniidae 1) were observed in Madagascar mangroves (Gardner, 2016). These species forage, sleep, and travel between mangrove forests and terrestrial forest patches.

Gardner et al. (2016) observed more than 73 species of terrestrial birds in Mangrove forests (Fig. 8.3) and over 54.8% of them are terrestrial inhabiting forests, scrublands, or open areas. Furthermore, 24.7% are endemic to Madagascar, including four endemic genera such as *Neomixis tenella* (Common jery), *Neomixis striatigula* (Stripethroat jery), *Hartlaubius auratus* (Madagascar starling), one belonging to an endemic subfamily with *Coua cristata* (Crested coua), and five species belonging to the endemic family Vangidae such as *Newtonia brunneicauda* (common newtonia) and *Leptopterus chabert* (Chabert vanga). Following the classification of IUCN, three recorded species are globally endangered: *Ardea humbloti* (Madagascar heron), *Ardeola idae* (Madagascar pond-heron), and *Anas bernieri* (Madagascar teal). One species, namely *Haliaeetus vociferoides* (Madagascar fish-eagle), is Critically Endangered.

FIG. 8.3 Bird in mangrove in the northwest of Madagascar.

Several commercial fish species, such as mullet (Mugilidae), sicklefish (Drepanidae), and ponyfish (Leignathidae) (Andrianjohany, 1992) are found in mangroves. Temporal patterns of fish assemblages in Toliara mangroves have been documented by Laroche et al. (1997). Large stands of mangrove are associated with a high proportion of organic material in bottom sediments and high productivity of the shrimp *Penaeus indicus* (Razafindrainibe, 1996). Catches of the industrial shrimp fishery correlate positively with mangrove areas (Lhomme, 2001).

4 Conservation areas and their ecological indicators

Mangroves in Mauritius fall mostly under coastal protected areas. Eight marine-protected areas have been proclaimed in the main island. This includes six fishing reserves and two marine parks (Blue Bay Marine Park and Balaclava Marine Park) (Leckraj, 2021). The Blue Bay marine park includes mangrove as habitat. The fishing reserves represent areas where fish resources as well as critical habitats such as mangroves are protected. All the six fishing reserves in Mauritius (Poudre D'or, Poste la Fayette, Grand Port Zone A, Grand Port Zone B, Black River, and Port Louis) are reported to have mangroves. All the protected areas are known to support a high biodiversity.

For mangrove ecosystems, there can be several stressors that can affect them. In a study in a tropical mangrove ecosystem in Indonesia, Tjahjono et al. (2022) suggest that there are a number of indicators such as climate, biophysical setting, and human factors to measure the vulnerability and risk of mangrove ecosystems. In Mauritius, some of these factors may have been studied in isolation and some examples of ecological indicators include forest structure and biophysical setting (Mohit and Appadoo, 2009; Abib and Appadoo, 2012; Ramdhun and Appadoo, 2020), but there is no clearly defined index.

Over 18% of mangroves in Madagascar are currently protected (Shapiro et al., 2000). As an example, mangroves are found in Sahamalaza and Baie de Baly National Park, in the Ramsar site (Ambaro bay, Tsiribihina delta) and within the Locally Managed Marine Area (LMMA) under management transfer contract within the Ministry of Environment and Sustainable Development.

5 Human activities and their impacts

The mangrove ecosystems on islands are subjected to several threats, namely, human-induced and natural ones. Mauritius being a small island, the coastal area is limited and there may be coastal development pressure. As highlighted by Hammond et al. (2015), rapid economic growth can lead to build out due to peri-urban expansion on islands like Mauritius and impact on environment-sensitive areas. This may in turn lead to loss of biodiversity. In recent years, other pressures have been observed in the mangroves of the islands. These include pollution from oil (Seveso et al., 2021), marine anthropogenic debris (Seeruttun et al., 2021), and microplastics (Seeruttun et al., 2023). These potentially impact the fauna leading to damage, long-term effects, and death. Such pressures also reduce the proper functioning of the ecosystem and have a negative impact on its aesthetic value.

Globally, mangroves are vulnerable ecosystems, and they will be affected by climate change. Ellison and Cannicci (2016) point out that ocean warming causes mangrove habitat conversion and affects faunal species. Impacts from sea level rise, increased intensity and frequency of storms, and changes in precipitation are highlighted in a comprehensive global review by Sarkar (2022). Sea level rise may be a very important factor as mangroves are sensitive and the plants as well as the species distribution can be affected by inundation duration, frequency, and salinity levels. This may lead to the migration of mangroves landward, which may however be limited by the slope and available land inward. It can also cause seaward edge mortality and retreat (Ellison and Cannicci, 2016). Such a scenario will be of greater concern, e.g., on small islands like Mauritius. Ragoonaden et al. (2017) showed that the Mean SLR (MSLR) for Mauritius is 3.8 mm yr^{-1} and that negative impacts to be experienced in Mauritius include coastal erosion and saltwater intrusion. The impacts of climate change on coastal erosion on small islands like Mauritius is also highlighted by Doorga et al. (2021). The island of Mauritius has been warming at a rate of 0.0216°C yr^{-1} over the period 1971–2020, with an increase in precipitation averaging 2.29 mm/year over the 1981–2020 period. Such a changing climate over time has had severe impacts on the native ecosystems of the island and freshwater supply systems (Doorga, 2022). Mangroves play a pivotal role as an ecosystem-based solution to reduce disaster risk and adapt to climate change, as highlighted by Sunkur et al. (2023) using Mauritius as a case study.

The total area of mangroves in Madagascar decreased from 284,644 ha in 1996 to 276,118 ha in 2016, representing an overall net loss of 8526 ha (2.3% of the total) over a period of 20 years (Erftemeijer et al., 2022). Mangrove loss have many drivers including unsustainable harvesting due to lack of governance, expansion of agri-/aquaculture areas, increased demand for fuel and timber, and growing demand for lime render (Jones et al., 2016; Scales and Friess, 2019; Rakotomahazo et al., 2019). As the mangrove extent decreases, the trend is not as severe within the protected areas (Shapiro et al., 2000). The mangrove restoration potential in Madagascar is relatively high. It is estimated that 8039 ha are available for restoration, evenly spread along the west coast (Erftemeijer et al., 2022).

6 Operational and potential monitoring tools to conserve or to recover the biome

The mangrove biome in Mauritius is well protected legally under the Fisheries and Marine Protection Act 2007. According to this "No person shall—except with the written approval of the Permanent Secretary, cut, take or remove; (b) damage, a mangrove plant." The biome has been given much attention since 1980 when the coverage was at a critical level of 45 ha. Efforts from all stakeholders has led to an increase to 145 ha in 2009 (Government of Mauritius, 2009) and the latest report is 181 ha (Republic of Mauritius, 2016). The main stakeholders involved in making this change is the Government of Mauritius through the Ministry of Blue Economy, Fisheries and natural resources, together with a number of nongovernmental organizations. Other partners have joined this effort, including the private sector and communities.

In Mauritius, mangrove replanting projects have been done with strong government-driven initiatives (page 39, Chapter 5; UNEP, 2020). Replantation projects are currently carried out by several nongovernmental organizations. Some projects involve public-private and NGO partnership as illustrated by successful replantation projects conducted by Association pour le Developpement Durable at le Morne (see Box 7.1, Chapter 7; Appadoo et al., 2016).

According to the recently published guidelines for mangrove restoration in the Western Indian Ocean region (United Nations Environment Programme, 2020), in order to carry out a successful mangrove restoration, it is important to consider a step-by-step approach where best practices are adopted. It is important to give due attention to the planning phase, the project implementation, and the monitoring and feedback phase. In several restoration projects, monitoring has been neglected and this may lead to failures. Monitoring of vegetation parameters, secondary succession,

and assessment of faunal biodiversity are crucial to show that the ecosystem is restored. Although some NGOs like Eco-Sud do monitor some or the restoration sites where they have done replantation, it is important that the monitoring component (both short term and long term) becomes a crucial component for all projects to ensure successful restoration of the biome.

There are many legal tools applied to the management and use of mangroves in Madagascar. The governance of mangroves often involves multiple levels, ranging from national to subnational and local (Slobodian and Badoz, 2019). Madagascar's national policies prioritize community-led management of natural resources, including mangroves. Although mangroves are considered part of the public domain of the State and are not susceptible to private appropriation, the state can transfer management rights to local communities. This mechanism aims to empower grassroots communities to play an active role in the preservation and restoration of mangrove ecosystems. The Marine Protected Areas and management transfers are the legal frameworks supporting the community-led mangrove management in Madagascar (MIHARI, 2019). The National Strategy for Integrated Mangrove management was launched in 2022 (MEDD, 2022).

In Madagascar, numerous stakeholders including but not limited to the forestry and fishery department, nonprofit organizations, research institutes, private sectors, and community-based organizations are involved and contributed directly or indirectly in the mangrove restoration initiative (MEDD, 2019) (Fig. 8.4). The restoration of mangroves focuses on degraded areas resulting from the overharvesting of mangrove timber. The direct replanting of propagules constituted the techniques adopted to restore mangrove biomes. *R. mucronata*, *C. tagal*, and *B. gymnorrizha* are commonly used in mangrove restoration efforts because they were the most harvested species (Ravaoarinorotsihoarana et al., 2023a). Community-led mangrove restoration initiatives in Madagascar can be considered as successful as the overall survival rate surpassed the 60% target survival (MEDD, 2019; Ravaoarinorotsihoarana et al., 2023a).

The total mangrove cover in Madagascar has increased by some 3000 ha since 2015, which is attributed to restoration efforts over the past decade (Erftemeijer et al., 2022). Continued community participation and the integration of Traditional Ecological Knowledge are reported as critical factors for the success of the initiative to restore the mangrove biome in Madagascar (Ravaoarinorotsihoarana et al., 2023b).

A Payment for Ecosystem Services (PES) focused on mangroves was initiated in southwest Madagascar to promote sustainable financing through the sale of carbon credits and incentivize community-led mangrove conservation and restoration (Blue Ventures, 2019; Rakotomahazo et al., 2019). This is the first blue carbon sequestration project in Madagascar designed to earn over 1300 carbon credits per year through the conservation and restoration of 1200 ha of mangroves (Blue Ventures, 2019). Local communities perceived the PES scheme as a potentially valuable approach for the sustainable management of their mangroves (Rakotomahazo et al., 2021).

7 Conclusions and perspectives

Mangrove forests are ecosystems of global importance due to the multiple goods and services they provide. Despite their importance, mangrove forests face significant threats, primarily driven by anthropogenic activities and climate change. Efforts to address mangrove degradation involve a combination of conservation of the remaining extent through sustainable management practices and restoration of degraded biomes should be taken. It is important to consider restoring degraded mangrove sites and promote ecosystem-based approaches to face global challenges, especially risk from disasters and climate change in an island context.

FIG. 8.4 Fisherman in the southwest of Madagascar.

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Section C

Mediterranean

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Chapter 9

Mediterranean Basin

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1 Introduction

Natural ecosystems serve a variety of functions and provide economic benefits to human beings. Despite this, there is still a rapid depletion of natural resources worldwide. Due to human activities, the global rate of extinction of plants and animals is over 1000 times higher than the average rate observed throughout human existence (Pimm et al., 1995). As a result, several strategies have been developed to sustain the world's critical ecosystems and ecological services, including "biodiversity hotspots" (Myers et al., 2000).

Plant biodiversity hotspots are regions where at least 1500 species of vascular plants are confined and have lost at least 70% of their original natural habitat (Mittermeier et al., 2004). The Mediterranean Basin Hotspot is the second-largest hotspot in the world and the largest of the world's five regions with a Mediterranean climate (Solomou and Sfougaris, 2019; Koulelis et al., 2023). It covers a total area of 2,085,292 km^2, extending from Portugal to Jordan and north to south from northern Italy to Tunisia. In addition to approximately 5000 islands, the Mediterranean Sea contains portions of Spain, France, the Balkan States, Greece, Turkey, Syria, Lebanon, Israel, Palestine 1, Egypt, Libya, Morocco, and Algeria. Several Atlantic islands, including the Canaries, Madeira, the Selvages (Selvagens), the Azores, and Cabo Verde, are located west of the continent (BirdLife International, 2017) (Fig. 9.1).

It is important to be referred, during the midtertiary, the African and Eurasian plates collided in the basin, resulting in a highly variable topographic, climatic, and geographic environment. Consequently, an astonishing diversity of species and habitats has been produced. The Mediterranean Basin Hotspot boasts one of the world's greatest endemic plant diversity areas and is considered to be the third-richest hotspot in terms of plant biodiversity (Mittermeier et al., 2004).

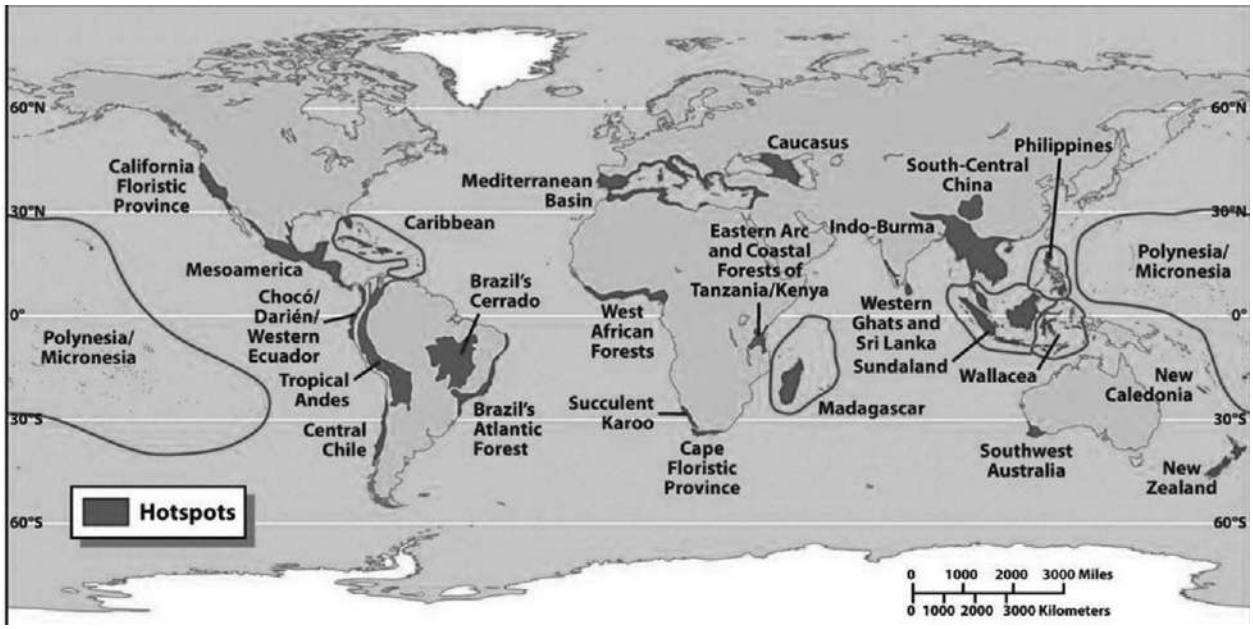

FIG. 9.1 Location of the Mediterranean Basin Hotspot. *(Credit: Dr. A. Solomou.)*

Terrestrial Biomes. https://doi.org/10.1016/B978-0-443-36569-0.00009-4
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Approximately 25,000 species of vascular plants are estimated to occur in the hotspot, approximately half of which are endemic (Blondel et al., 2010).

2 Climate

In general, the Mediterranean Basin Hotspot has a Mediterranean climate, although some islands in the Macaronesian region have environments ranging from Mediterranean to arid and subtropical. During the winter, the Mediterranean climate is characterized by a few cold, rainy days and a few hot, dry days. This region has a wide range of rainfall, from as few as 100 mm to well over 3000 mm every year. There is ample rainfall in the Atlas Mountains and the Macaronesian Islands due to moisture coming from the Atlantic. In contrast, there is very little rainfall on the Cyrenaic Peninsula of Libya. There are periods without significant precipitation for almost 2 months in the western half of the Mediterranean and 5–6 months in the eastern half. The short spring and autumn seasons are crucial to plant growth. In the Mediterranean, snow rarely falls, except for the mountains, but hard frosts occur from time to time (Blondel et al., 2010).

3 Biological history

It is estimated that 10% of the world's plants are found in the Mediterranean Basin, which accounts for approximately 1.6% of the Earth's surface (Blondel et al., 2010). In a surface area one-fourth the size of sub-Saharan Africa, the hotspot is home to approximately the same amount of plant diversity as all of tropical Africa. Diverse factors have contributed to the development of this diversity. Moving earth, volcanic activity, earthquakes, and the near desiccation of the sea during the Messinian Salinity Crisis had a significant impact on living systems and created a mosaic of habitats that are highly heterogeneous in topography, soil type, and microclimate based on altitude, rainfall, and slope exposure (Blondel et al., 2010).

These factors, combined with the region's geographic location at the intersection of three major land masses, Europe, Asia, and Africa, have resulted in fauna and flora that are extraordinarily diverse and highly distinctive. A final factor is the long history of human occupation in the region, which has resulted in a close relationship between the region's flora, its major landscapes, and the human activities that have shaped them for over 10,000 years (Pons and Quézel, 1985). In addition to their particular life traits, Mediterranean endemic plants provide insight into the region's diverse natural habitats, topography, and history. There are important endemism zones in areas exposed to high rates of geological change, where relicts and more recent taxa coexist. Because of its isolation, the Mediterranean region serves as a plant refuge and an area for floral exchanges (Quézel, 1985). It should be noted that in the western basin, high-endemism areas are associated with the southern reaches of the Iberian plate, while in the eastern basin, due to the moderate impact glaciations played and the presence of ultrabasic rocks, vicariant endemism is high (Verlaque et al., 1997). Many species of birds and mammals originate from regions outside of the Mediterranean Basin, particularly Eurasia and Africa. The dispersal abilities of these species are higher than those of the herpetofauna, which show a higher rate of endemism across the basin. There are several ancient lineages and many endemic genera of reptiles, amphibians, and freshwater fish. Large areas of the hotspot are covered by evergreen oak, coniferous, and deciduous forests. However, human settlement and habitat modification have resulted in much of this forest disappearing or undergoing alteration (Tucker and Evans, 1997).

According to Sloan et al. (2014), the Mediterranean Basin Hotspot has one of the lowest percentages of natural vegetation remaining. Throughout history, human pressures have altered Mediterranean ecosystems; however, this long-term "coevolution" between ecosystems and land use practices has continued. The hotspot has shaped several semitransformed habitats, which now host several rare and threatened species (Blondel et al., 2010). There is now a widespread vegetation type called maquis, which consists primarily of hard-leaved or sclerophyllous shrublands maintained through grazing and occasional fires. Several species of endemic and restricted-range plants are at risk due to changes in land use and rural abandonment caused by anthropogenic activity (Sirami et al., 2010).

4 Ecoregions

An ecoregion is a large geographical region characterized by a certain species assemblage, natural communities, and environmental conditions. According to WWF (2006) and The Nature Conservancy (2011–13), 64 ecoregions were identified in the Hotspot: 27 terrestrial, 26 freshwater, and 11 marine (Spalding et al., 2007).

4.1 Terrestrial ecoregions

Six terrestrial biomes are found in the Mediterranean Basin Hotspot, including (1) Mediterranean forests, woodlands, and scrub; (2) tropical and subtropical dry broadleaf forests; (3) temperate broadleaf and mixed forests; (4) temperate coniferous forests; (5) montane grasslands and shrublands; and (6) deserts and xeric shrublands. Among these biomes, 27 terrestrial ecoregions have been identified in the hotspot, and the Mediterranean forests, woodlands, and scrub biome are the most extensive, accounting for 21 ecoregions (Olson et al., 2001) (Fig. 9.2).

FIG. 9.2 Mountain grassland of Mt. Kallidromo in Greece. *(Credit: Dr. A. Solomou.)*

4.2 Freshwater ecoregions

There are 26 freshwater ecoregions within the Mediterranean Basin Hotspot which are divided into 4 types of biomes: (1) temperate coastal rivers; (2) temperate floodplain rivers and wetlands; (3) xeric freshwaters and endorheic basins; and (4) large river deltas (Abell et al., 2008) (Fig. 9.3).

4.3 Marine ecoregions

About 11 marine ecoregions are part of the Mediterranean Basin Hotspot: tropical Atlantic and temperate northern Atlantic. It is important to note that the ecoregions which have been identified are Cabo Verde, Azores, Canaries, Madeira, Saharan Upwelling, South European Atlantic Shelf, Adriatic, Aegean, Levantine, Tunisian Plateau/Gulf of Sidra, Ionian, Western Mediterranean, and Alboran Seas (Spalding et al., 2007).

5 Species diversity and endemism

Even though the biodiversity of this extensive region is exceptionally high, 10 major areas serve as the basin's centers of plant diversity (Médail and Quézel, 1997, 1999). The endemism of this basin can be attributed to these areas in approximately 44%. Mountains and islands make up the majority of these areas. There are 10 components to the project: (1) the High Atlas Mountains in northern Africa; (2) the Betic-Rif range in southern Spain and the coastal regions of Morocco and Algeria; (3) the French-Italian Maritime and Ligurian Alps; (4) the Tyrrhenian Islands; (5) southern and central Greece; (6) Crete; (7) southern Turkey and Cyprus; (8) the Syria-Lebanon-Israel region; (9) Cyrenaica in Libya; and (10) the Canary Islands and Madeira. Excluded from Médail and Quézel's analysis, the Cape Verdean islands are also centers of plant diversity, with endemism rates of 12.5% (Romeiras et al., 2016).

A partial disconnect exists between the Mediterranean and Atlantic Oceans for the marine portion of the hotspot, with Mediterranean taxa primarily deriving from the Atlantic Ocean (Coll et al., 2010). Some groups still have a significant amount of gene flow (Patarnello et al., 2007). It is believed that the high degree of endemism in the basin results from the basin's isolation (Coll et al., 2010). Most biodiversity is concentrated along the coastline, although several biodiversity hotspots are associated with deep and offshore pelagic waters (Danovaro et al., 2010).

The Macaronesian islands are mainly oceanic, with abyssal plains scattered with numerous seamounts (as well as the islands themselves). These islands serve as biodiversity islands for marine life (e.g., deepwater coral reefs) (Mitchell-Thomé, 1976). Biological marine diversity occurs mostly on seamounts and the slopes of the islands, which remain largely isolated. The region is also a stronghold for large pelagic fish, seabirds, and cetaceans. Almost 8% of the world's marine fauna and 18% of marine

FIG. 9.3 Coastal Wetland Schinias Marathon in Greece.

flora are concentrated in this region (Coll et al., 2010; BirdLife International, 2017).

6 Plant and animal diversity

The Mediterranean Basin is a recognized biodiversity hotspot. With 15,000–25,000 species, its flora diversity is outstanding, and 60% are endemic (Quézel, 1985). Approximately one-third of the Mediterranean's fauna is endemic to the region. Due to habitat degradation, many of them are considered vulnerable, endangered, or threatened with extinction. The reduction of Mediterranean biodiversity is also attributed to other significant issues such as pollution, droughts, alien invasive species, and overexploitation.

Islands, which have peculiar and hostile soils or rock types like dolomites, limestones, gypsum, and ophiolites, are especially abundant in species, as are topographic islands surrounded by steep slopes or at the summit of mountains (Blondel et al., 2010). According to Blondel et al. (2010), the proportion of endemic species rises with altitude: on Mediterranean mountain ranges, whether continental (Atlas, Taurus, Lebanon, Anti-Lebanon) or islands (Corsica, Sardinia, Sicily, Crete), the proportion of endemic species can exceed 25%.

Despite widespread recognition of the Mediterranean region as a global hotspot for plants, precise data regarding the distribution and conservation status of plants and habitats in many Mediterranean countries are frequently insufficient, obsolete, or unavailable. This is especially true for countries south and east of the Mediterranean Basin (the subregions of North Africa and the Middle East). It is difficult to monitor the condition of plant diversity in the absence of baseline data on the patterns of plant diversity (Radford et al., 2011). Both species richness and endemism in the Mediterranean region are strongly influenced by the number and patchiness of local plant communities, which are influenced by the history of natural and human disturbance. As a result of human impacts on Mediterranean environments, variety and endemism may be seen as "by-products" (Rackham, 2008).

6.1 Vegetation

Maquis (evergreen shrublands and forests) that are dominated by sclerophyllous oaks such as *Quercus ilex*, *Q. coccifera*, and *Q. suber*, as well as conifer forests that are dominated by *Pinus halepensis*, *P. brutia*, and *Cupressus sempervirens*, are generally regarded as the most complex vegetation types considered in regions of the Southern and Eastern Mediterranean with more arid climatic conditions. In addition, a growing number of recent paleoecological studies indicate that deciduous and semideciduous broadleaf trees played a significant role in Mediterranean ecosystems during the postglacial period, particularly in the northern Mediterranean, and that the dramatic reduction of these forests was primarily the result of human activity and domesticated livestock.

Furthermore, a growing number of recent paleoecological studies show that deciduous and semideciduous broadleaf trees played an important role in Mediterranean ecosystems during the postglacial period, particularly in the northern Mediterranean, and that human activity and

domesticated livestock primarily caused the dramatic reduction of these forests. Many islands in the Mediterranean have distinctive vegetation and landscapes because indigenous or otherwise rare plant species dominate their ecosystems. Open woodlands on Crete feature *Zelkova abelicea*, *Q. coccifera*, and *Acer sempervirens*, whereas mountainous forests in Corsica, Calabria, and Mount Etna (Sicily) feature *P. nigra* subsp. *laricio* (Quézel and Médail, 2003).

In the Mediterranean, a variety of low-growing plant communities have developed as a result of disturbances (mostly wildfires and overgrazing) and climate stress. These are known as phrygana or batha in the eastern Mediterranean, where they are dominated by thorny, often aromatic, and summer-deciduous shrubs and subshrubs, whereas garrigues, tomillares, or matorrals in the central and western northern Mediterranean, depending on the dominant woody species and country. Upper levels of vegetation (supra-Mediterranean to oro-Mediterranean) are only found on the highest points of the largest and most mountainous islands, such as Corsica, Sardinia, Sicily, Crete, and Cyprus. They are made up of discontinuous dwarf shrublands that have adapted to the harsh climate of the high Mediterranean mountains (Guarino et al., 2006).

6.2 Flora

Because of its complicated biogeographical history, the Mediterranean region has played and continues to play the role of a melting pot for plants worldwide. Evergreen woody species such as *Myrtus communis*, *Phillyrea* sp., *Pistacia lentiscus*, and even evergreen oaks like *Q. coccifera* and *Q. ilex* grow together in semiarid areas to form maquis communities. Warm temperate forest communities, possibly including *Taxus baccata*, *Arbutus* sp., *Ilex* sp., *Laurus* sp., *Hedera* sp., *Rhamnus* sp., and *Smilax* sp., were dramatically disrupted by Alpine-Himalayan orogenesis and the onset of glacial events (Box and Fujiwara, 2015).

The physiognomy of Mediterranean landscapes is heavily influenced by thermophilous conifers as well. Throughout large areas of various nations, trees like the cypress (*C. sempervirens*), juniper (*Juniperus phoenicea*, *J. oxycedrus*, *J. foetidissima*, etc.), and pine (*P. halepensis*, *P. brutia*, *P. pinea*) remain dominant (BirdLife International, 2017).

6.3 Birds

There are 534 species of birds in the hotspot, including 63 endemic species. There are three main groups of species: a group of northern, boreal-origin species that inhabit forests, freshwater marshes, and rivers in western Eurasia; a collection of species found on the steppes near the edge of the modern Mediterranean; and a collection of species found in shrubland habitats, including partridges (*Alectoris*) and warblers (*Sylvia*, *Hippolais*). Numerous species migrate from Europe to Africa via the Bosphorus, the Rift Valley, Gibraltar, Sicily, the Balearic Islands, Corsica, Crete, Sardinia, and Cyprus (Blondel et al., 2010).

6.4 Mammals

The fauna of the Mediterranean Basin Hotspot is primarily derived from the Eurasian and African biogeographic zones and, as a result, displays relatively low levels of endemism (Temple and Cuttelod, 2009). There are almost 300 species, and 38 of them are only found on land. Most of them are mice and shrews. The vast majority of mammal species are minuscule. The Muridae is the largest family, with 51 different species of rodents, gerbils, birds, and mice. Vespertilionidae (evening and vesper bats; 38 species) and Cricetidae (hamsters and voles; 23 species) are 2 additional important families in the region. There are eight species associated with environments containing fresh water. None of the hotspot's 31 marine mammals are endemic (BirdLife International, 2017).

6.5 Amphibians

The diversity and abundance patterns of amphibians are opposite to those of reptiles. Overall, species diversity is modest (109 species), and amphibian diversity is greatest in regions with higher precipitation, including western Spain, northern Italy, France, Slovenia, and Croatia. Although species richness is lower, endemism is relatively high, with nearly half of all species (54) endemic to the hotspot.

Most Mediterranean-endemic amphibians adhere to archaic lineages that have remained unchanged since their origins. Examples include two genera of toads, *Pelobates* (one of the four endemic species) and *Discoglossus* (four of the six endemic species), a salamander genus, *Euproctus* (two endemic species), and the genus of olms, *Proteus* (one endemic species) (BirdLife International, 2017).

6.6 Reptiles

Species richness and endemism among reptiles are notably higher when compared with other taxa. It is estimated that there are 299 different species of terrestrial reptiles, with over 40% (117 species) being endemic (Cox et al., 2006; IUCN, 2016). Snakes, lizards, tortoises, and even tropical relics like two species of chameleon (*Chamaeleo chamaeleon* and *C. africanus*) make up the reptile fauna of the Mediterranean Basin. Most Mediterranean reptile species are lizards (67%) and snakes (27%). Many species of reptiles in the genera *Podarcis*, *Lacerta*, *Chalcides*, and *Vipera* evolved in the basin due to intensive adaptive radiation in localized areas. In the Lacertidae, the genera

Algyroides and *Psammodromus* are typical relict Mediterranean endemics. Reptiles in the Macaronesic Islands have high endemism rates, with 90% (38 species) of the species being endemic (Blondel et al., 2010).

6.7 Invertebrates

Invertebrates in the Mediterranean are extraordinarily varied, but little is known about them even though new species are described yearly, similar to other biodiversity hotspots. The hotspot is thought to contain 150,000 insect species alone (Balletto and Casale, 1991). According to estimates made by Coll et al. (2010), the 17,000 species that make up the marine environment of the Mediterranean Sea include 10,000 invertebrate species, of which 1000 are endemic (BirdLife International, 2017).

7 Ecosystem services in the Mediterranean Basin Hotspot

Ecosystem services are the advantages derived from the operation of natural ecosystems. In general, ecosystem services can be divided into four categories: provisioning, regulating, sustaining, and cultural. In the Mediterranean Basin Hotspot, these services include those that are important on a global scale, such as climate mitigation through carbon storage and sequestration, and those that benefit local communities and individuals, such as those that provide essential goods and services for survival, such as food, fuel, and building materials (Millennium Ecosystem Assessment, 2005).

Provisioning services are critical for the livelihoods and economic activity of all human populations in the hotspot. In this severely water-stressed region, water is the most essential ecosystem service. Wetlands, particularly marshes and riparian vegetation, help to filter contaminated water and enhance its quality (Mediterranean Wetlands Observatory, 2012). Utilizing cleaner water for drinking, irrigation, and energy production is simpler and less expensive.

Forests provide wood for construction, furniture, and handicrafts (especially from high-quality woods such as olive and sandarac), as well as firewood and charcoal, which are still indispensable in many rural areas of the region. In the eastern Mediterranean region, cedar wood has been an essential source of high-quality building materials. Nontimber forest products (NTFP) have been used by humans in a sustainable manner for millennia, with cork being the most important NTFP in terms of number of workers employed and revenue generated (Cork Quality Council, 2016). Several woody plants (labdanum, mastic, myrrh, rosin, sandarac, etc.) and Lamiaceae in particular yield resins and essential oils.

The creation of synthetic substitutes diminished their historical significance, but markets for high-quality natural products are now expanding, and the use of medicinal plants remains significant in North Africa and the Middle East. Mushrooms, truffles, fruits, and nuts (such as pine nuts) are also commodities with high added value that is consumed locally or exported.

With the exception of several Balkan regions, subsistence hunting and fishing are now less important sources of animal feed for local populations. It is predicted that the fishing business directly employs between 140,000 and 280,000 people in the area. Commercial fishing, particularly in coastal and marine areas, is a substantial economic activity and an important food supply, directly employing between 140,000 and 280,000 people (Farrugio, 2013; Di Franco et al., 2014).

Regulating services can be expected to become more important as climate change impacts on increasingly densely populated areas. Between 2000 and 2009, more than 2 million individuals in Mediterranean countries were affected by drought and more than 1.1 million by flooding (including more than 2000 deaths). Drought-related disasters cost an estimated $19 billion. Wetlands and other habitats provide crucial protection for mountainous and coastal regions, mitigating the effects of increasingly intense storms and rainfall (Mediterranean Wetlands Observatory, 2012).

Supporting services include the provision of renewable energy from solar and wind power, which will become increasingly essential as energy demand increases and must be met by carbon-neutral sources. Sequestration of carbon dioxide is a crucial auxiliary utility, mitigating the increase in greenhouse gas concentrations in the atmosphere and thereby slowing climate change. The arid climate of the majority of the region restricts the direct carbon sequestration capacity of the forests.

The cultural aspects of ecosystem services include their importance to the tourism industry, one of the three principal service sectors upon which the majority of the hotspot's revenue depends. In addition to this modern economic significance, the landscapes and species of the Mediterranean have contributed to the development of some of the world's most influential civilizations and religions. The wild flora and fauna of the region, as well as the products of traditional farming and livestock, contribute to the distinctiveness and diversity of the region's cuisine, for which it is renowned internationally. In recent decades, hunting has transitioned from a sustenance source to a recreational activity for many people. The intensity of some hunting activities, particularly of migratory birds, makes it a serious environmental concern, despite its close ties to local identity and recreation (BirdLife International, 2016). Despite the critical significance of ecosystem services to the economy and to human well-being, they are frequently undervalued and unrecognized; as a result, they may be harmed or destroyed during economic development.

In other cases, the value of communal resources was recognized. Still, traditional systems for maintaining these services (such as the hima system for managing pasture) have disintegrated due to state-imposed land categories, cultural and economic modernization, and urbanization. There is a spatial or temporal separation between land managers who can influence the quality of ecosystem services and the beneficiaries who may be willing to pay for the service for many services (e.g., water supply).

In other instances, the services (e.g., clean air and beaches) are difficult to quantify or administer and may be perceived differently by locals and foreigners, for example. Frequently, tourists are willing to pay governments directly for investments in natural and cultural ecosystem services (Seidl, 2014). The incorporation of the preservation and management of ecosystem services into government land use and development planning requires data on the values and impacts of these services. In Europe, detailed information is readily available, but in the hotspot countries covered by the ecosystem profile update, it is much less comprehensive. The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) is tasked with producing Regional IPBES assessments that provide a comprehensive analysis of ecosystem services in Europe, Central Asia, and Africa. They are scheduled for completion in 2018. There are also valuable models of participatory, local valuation of ecosystem services from biodiversity protected areas in Madagascar (Neugarten et al., 2016) that could be adapted for implementation in areas where the ecosystem services issue is central to the conservation case. The Mediterranean Basin is one of the most vulnerable regions in the world to climate change, and this will have an impact on the capacity of ecosystems to provide goods and services to human society (Bangash et al., 2013). This is particularly worrisome in light of the growing demands placed on ecosystems. The availability of water for consumption and hydropower generation will decline, while the demand for water for irrigation and tourism will increase. As conditions become drier and fires become more intensive and frequent, the forests of the Mediterranean will diminish. These changes will contribute to an increase in erosion, loss of agricultural potential, and management costs (Bangash et al., 2013; Terrado et al., 2014; BirdLife International, 2017).

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Chapter 10

An Afromontane biome in South Africa: Ecological quality of natural vs transformed habitats

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1 Introduction

In addition to climate change, declining biodiversity, especially in insects, poses a threat to sustainable food production. Insect biodiversity decline can be mainly attributed to the intensification of agriculture, with the main drivers being habitat loss, fragmentation, and use of agrochemicals (Hallmann et al., 2017; Cardoso et al., 2020). Wild populations have been declining rapidly as agriculture expands and intensifies. According to the WWF's Living Planet Report 2022 wildlife populations have declined by an average of 69% over the past 50 years (WWF, 2022). Our attempts to feed the growing human population have left very little space for other species sharing the planet with us and these species cannot survive in the changing environment leading to biodiversity loss.

Healthy soil is crucial for the successful production of food. The health of a soil, for successful plant growth, is a result of intricate interactions between soil, macrofauna, and microorganisms. The role of macrofauna in nutrient cycling includes fragmentation of plant residues and stimulation of microbial activity and this has a direct effect on soil structure through the redistribution of organic material and microorganisms, while increasing soil aeration (González et al., 2001). By burying decomposing organic matter and construction of galleries for nesting within the soil, dung beetles actively contribute to the ecological process of nutrient cycling (Nichols et al., 2007, 2008; Scholtz et al., 2009).

Dung beetles of the subfamily Scarabaeinae occur globally, comprising approximately 7000 species (Daniel and Davis, 2023), they are associated with a great variety of habitats exhibiting significant variation in spatial and temporal characteristics depending on the availability of food (Barbero et al., 1999). The highest diversity of dung beetles is found in tropical forests and savannas (Hanski and Cambefort, 1991; Scholtz et al., 2009). These dung beetles feed mainly on mammal dung during the larval and adult stages, although a few species feed largely on carrion, rotting fruits, fungi, and decaying plant matter (Scholtz et al., 2009). During the process of manipulating these resources, they provide a series of ecological processes such as nutrient cycling, soil aeration, secondary seed burial, and parasite suppression, which are driven by the patterns of consumption and relocation of dung by dung beetles (Raine and Slade, 2019; Nichols et al., 2008). The abundance and biomass of dung beetles are influenced by the physicochemical properties of the soil in agroecosystems (De Farias and Herández, 2017). This information is important to determine the strategies for increasing fertility and for the management of soil conservation in agroecosystems. Several studies indicate that dung beetle assemblages respond negatively to fragmentation and transformation of natural habitats and that larger dung beetles do not survive in these unsuitable habitats (Chandra and Gupta, 2012; Rivera et al., 2021; Noble et al., 2023). To mitigate the pressure of agriculture on biodiversity, we urgently need to prioritize the restoration of land to natural habitats within agricultural landscapes. With that in mind, the objective of this chapter is to provide an overview of how dung beetles respond to habitat transformation in the temperate grasslands of the Afromontane biome in South Africa.

2 Geographic location

Temperate grasslands are high in biodiversity and endemism ensuring important ecosystem services. Unfortunately, these biomes are globally the least protected, with only 4.6% formally protected (Scholtz and Twidwell, 2022).

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The Maloti-Drakensberg of South Africa and Lesotho is one of seven remaining continuous grasslands in the world and their contribution to global grassland conservation is critical (Scholtz and Twidwell, 2022). The likelihood of undescribed species in both alpine and montane zones in the broader South African Great Escarpment is generally very high. The South African portion of the QwaQwa Maloti's comprises the Witsieshoek Community Conservation Area. Although included in the Maloti-Drakensberg Transfrontier Park (MDTP), this area is an imperiled and poorly known biome, not formally protected, and rapidly converted into settlements or degraded by alien plant invasion, bush encroachment, and livestock overgrazing (Shezi et al., 2021), resulting in loss of habitat and biodiversity.

Dung beetle assemblages were monitored at three different localities, a semi-natural (SN) landscape, a conservation agricultural (CA) landscape, and a conventional agricultural landscape (Fig. 10.1). The SN landscape falls in the Witsieshoek CCA within the QwaQwa Maloti's section of the MDTP. There is some human disturbance in this area, mostly by cattle grazing. Two sites were chosen in this landscape at different elevations. Site 1 (S28.6773E28.9019) was at an elevation of 2058 m and falls in the lower subalpine (2000–2400 m). The grazing pressure in this area is moderate to high. This area was dominated by encroaching *Leucosidea sericea* and increaser Eragrostic *plana* plant species, indicating the effects of overgrazing. Site 2 (S28.7273E28.8917) is 5 km from site 1 and at an elevation of 2555 m and falls in the higher subalpine (2400–2800 m). The grazing pressure in this area is low. Because this area is less accessible, it is generally protected from livestock grazing and disturbance. The. CA landscape is 50 km from the SN landscape and falls in an agricultural area near Clarens (Eastern Free State). Two sites were chosen in this landscape at different elevations. Site 1 (28°33′26,45″S28°24′36,11″E) was at an elevation of 1699 m and was in crop fields. The fields were planted with maize and a variety of cover crops by no-till. Agro-chemicals (fertilizer, herbicides, fungicides, and insecticides) were used in this area. Livestock (cattle and sheep) grazed the area after harvest of the maize crop at a high grazing intensity. Site 2 (28°33′55,15″S28°24′48,54″E) was 1 km from site 1 and at an elevation of 1765 m. This site was in a natural pasture that was grazed by cattle at a moderate grazing intensity. The conventional agriculture (CT) landscape is 60 km from the SN landscape and 20 km from the CA landscape. Two sites were chosen in this landscape at different elevations. Site 1 (28° 23′52,13″S28°22′58,27″E) was at an elevation of 1750 m and was in crop fields. Maize and soya were planted with conventional deep tilling. Agro-chemicals (fertilizers, herbicides, fungicides, and insecticides) were used in this area. Site 2 (28.402032°S 28.372308°E) was 1 km from site 1 and at an

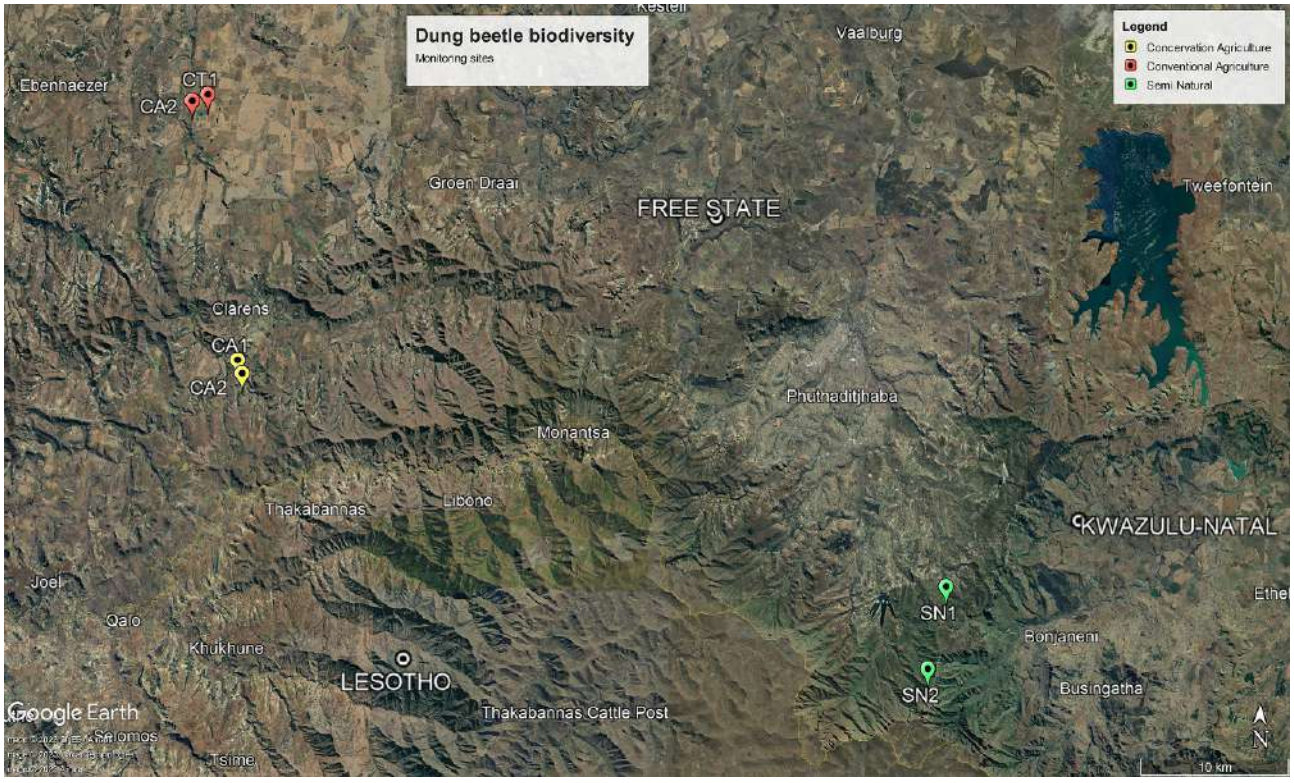

FIG. 10.1 Monitoring sites for dung beetle biodiversity. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: The authors.)*

elevation of 1825 m. This site was in a natural pasture that was grazed by cattle at a moderate grazing intensity.

Dung beetles were monitored in these areas with pitfall traps, baited with pig dung. In each site three plots were chosen, spaced 50 m apart, in each plot we set up four pitfall traps, spaced 5 m apart. The traps were left for 48 h in the field; then' the samples were preserved in ethanol. Voucher specimens of this study are deposited at the National Museum Bloemfontein and Agricultural Research Council.

3 Biodiversity

Biodiversity drives ecological functioning by providing ecosystem functions favored by greater functional diversity in different interacting species. Disturbances in the environment will not only influence individual species, but also these interactions between species and will determine the structure of dung beetle assemblages in an area. Dung beetle assemblages were different in the different landscapes and sites in the study area. The two SN sites were most similar, while there was very little overlap of the two agricultural landscapes with the SN landscape (Table 10.1). There was a higher similarity in dung beetle assemblages between the CA and CT landscapes than the similarity of these landscapes to the SN landscape, with the CT landscape having a greater overlap to the SN landscape than the CA landscape to the SN landscape (Table 10.1). This indicates that despite changes in agricultural practices, from conventional to conservation, the pressure from agricultural activities impacts dung beetle assemblage structure when moving from a habitat with lower overall disturbance to habitats with greater pressure from agriculture.

To ensure the ecological sustainability of rangelands, it is necessary to also consider the ecological functions of dung beetles in addition to diversity (Ardali et al., 2016; Steed et al., 2022; Simba et al., 2022). When determining the effect of habitat disturbance on biodiversity, the functional diversity within a dung beetle assemblage is also an important factor to consider. Regional scarabeinae assemblages in southern Africa may contain a diversity of species, which range in size and live weight from 10 mg to 10 g (Doube, 1990). Different species show a variety of dung-use and reproductive strategies. This was reflected in the dung beetle species collected in the study area. A total of 23 dung beetle species classified in 13 genera were collected in the study area during March 2022 (Table 10.2). These dung beetle species were grouped into seven functional groups based on their size and specific dung-use strategies. Based on their behavior, dung beetles have been placed in four distinctive groups: (Halffter and Edmonds, 1982; Klemperer, 1983; Hanski and Cambefort, 1991) telocoprids are dung beetles which form dung into a ball and roll it away to bury somewhere else; paracoprid dung beetles make nest chambers under the dung pad and transport dung from the pad by tunneling under the dung pad; endocoprids breed inside the dung pad; and kleptocoprids use dung that has already been buried by other dung beetles. There is a clear hierarchy of functional groups in a dung beetle assemblage to compete for dung. FGI (large telocoprids) and FG III (fast-burying paracoprids) are competitively dominant, larger dung beetles which rapidly remove dung from the pad, while the smaller telocoprids (FG II) are also effective competitors (Doube, 1990). FG IV and V (paracoprids) are smaller subordinate groups, which bury dung slowly over many days, while FG VII are endocoprids, which breed inside the dung and FG VI breed in dung buried by other beetles (Doube, 1990). The dominant species in the study area, which occurred at all the sites, was *Onthophagus pilosus* (Table 10.2). This species, occurring more abundantly in the agricultural habitats (Table 10.2), belongs to FG IV, which is a subordinate group, which bury dung slowly over many days. Telocoprid dung beetle, *Scarabaeus basuto* (FGI), occurred only in the two SN sites (SN1 and SN2), while telocoprid dung beetle species, *Epirinus drakomontanus*, *Epirinus gratus*, *Epirinus asper*, *Epirinus punctatus*, and *Epirinus validus* (FGII) only occurred in the SN site at a higher elevation (SN2) (Table 10.2). There was a clear difference in dung beetle species composition and ecosystem function between the SN and agricultural habitats. Telecoprid dung beetles (FGI and II) are larger species and highly effective competitors for dung. They remove the dung from the surface at a fast rate and bury it at a distance, thereby spreading the nutrients in the soil efficiently. These groups were only represented by species present in the SN landscape and were not represented in either of the agricultural landscapes. Telecoprid dung beetles were probably absent in the agricultural areas because they cannot survive the disturbances in these areas. Shahabuddin et al. (2010) found that large-bodied dung beetle species were more sensitive

TABLE 10.1 Sorenson similarity index for dung beetle populations at different sites.

	CA2	CT1	CT2	SN1	SN2
CA1	0.56	0.4	0.57	0.31	0.22
CA2		0.29	0.44	0.35	0.15
CT1			0.6	0.44	0.4
CT2				0.46	0.22
SN1					0.75
SN2					

0–1, no overlap; 1, perfect overlap.

TABLE 10.2 Dung beetle species occurring at different landscapes and sites in study area.

	Species	FG	CA1	CA2	CT1	CT2	SN1	SN2
1	*Scarabaeus basuto*	*I*					3	1
2	*Epirinus drakomontanus*	*II*						5
3	*Epirinus gratus*	*II*						1
4	*Epirinus asper*	*II*						1
5	*Epirinus punctatus*	*II*						1
6	*Epirinus validus*	*II*						8
7	*Xinidium dentilabris*	*IV*			1	20	1	
8	*Onthophagus pilosus*	*IV*	49	89	1	2	19	3
9	*Onthophagus vigens*	*IV*		2				
10	*Onthophagus binodus*	*IV*				1	1	1
11	*Onthophagus cretus*	*IV*					1	1
12	*Euoniticellus africanus*	*IV*	1			1		
13	*Liatongus militarus*	*IV*		1				
14	*Cyptochirus ambiguus*	*IV*		1				
15	*Onitis caffer*	*IV*					1	
16	*Onthophagus venustulus*	*V*	10	42				
17	*Odontoloma peckorum*	*VI*		3				
18	*Aphodius pseudolividus*	*VII*	4	7	1	5		
19	*Aphodius teter* sensu lato	*VII*	1	1		1		
20	*Aphodius laterosetosus*	*VII*	1	1			1	
21	*Harmogaster strydomi*	*VII*	1					
22	*Rhysemus africanus*	*VII*		1				
23	*Teuchestes fossor*	*VII*					1	

CA1, conservation agriculture site 1; CA2, conservation agriculture site 2; CT1, conventional agriculture site 1; CT2, conventional agriculture site 2; FG, functional group; SN1, semi-natural site 1; SN2, semi-natural site 2.

to habitat disturbance and the ratio of large- to small-sized dung beetles declined with land-use intensity. The absence of telocoprid dung beetles in the agricultural areas will have consequences for ecological functioning in the systems since dung beetles from a single functional group can have a major influence on ecosystem function. Slade et al. (2007) found that the absence of large paracoprids (FGIII) reduced dung removal by 75%. The full complement of functional groups is, however, necessary to maximize ecosystem functioning. Menéndez et al. (2016) found complementary effects between dung beetle species with different functional behavior (paracoprid and endocoprid) and that the combined effect of two dung beetle species resulted in the highest soil microbial respiration from the soil. They concluded that the return of C in the form of dung in grasslands, coupled with the activity of a functionally diverse dung beetle assemblage, could result in short-term fluctuations in soil microbial activity with important consequences for soil C cycling.

The SN and CA landscapes supported four functional groups: FG IV, V, VI, and VII, while the conventional agricultural landscape supported only two: functional groups, FG IV and VII (Fig. 10.2). The functional group classification enables us to analyze the structure of different dung beetle assemblages in a way that reflects community function (Doube, 1990). In the present study, community structure was analyzed at two levels: the number of species present (alpha diversity) and the numerical abundance of members of functional groups (Fig. 10.2). The three separate dung beetle communities show different patterns of

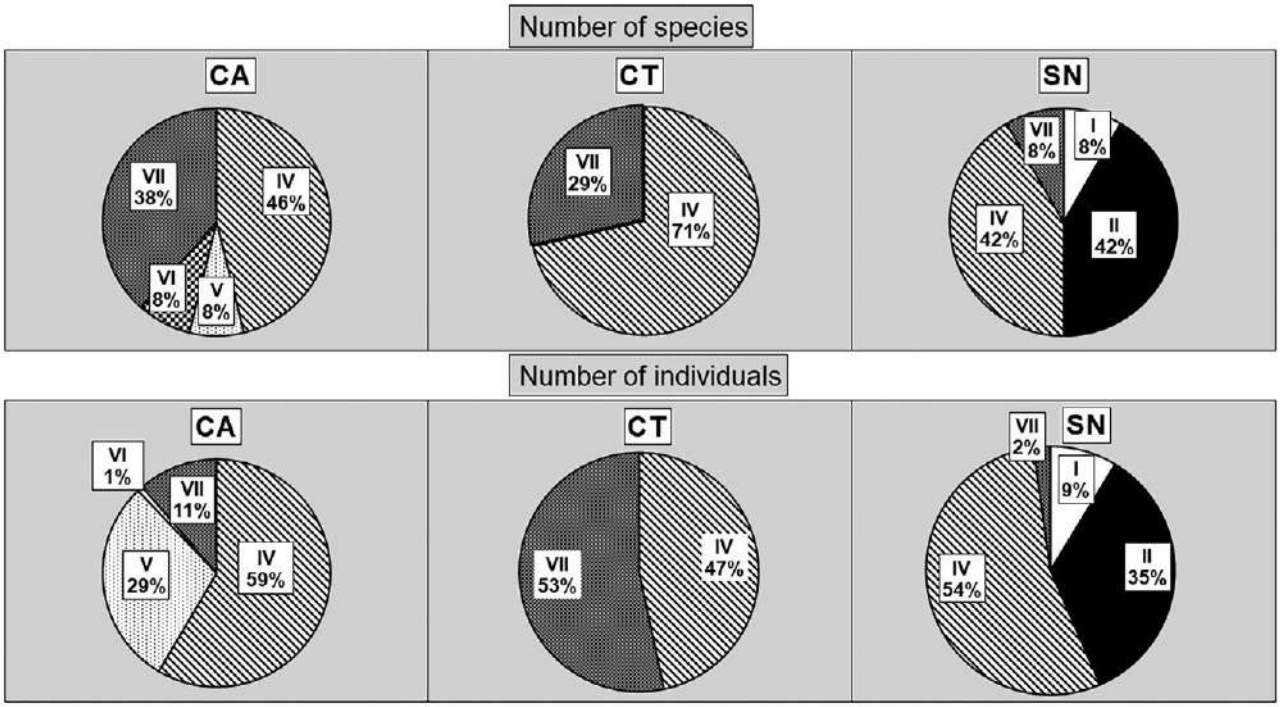

FIG. 10.2 Functional group (FG) classification of dung beetle assemblages in different landscapes, conservation agriculture (CA); conventional agriculture (CT); and semi-natural (SN). *(Credit: The authors.)*

relative abundance of species within functional groups (Fig. 10.2). FG IV was represented in all the communities and was the dominant group in all the communities comprising 46%–71% of species. FGI and FGII were absent from the agricultural (CA and CT) landscapes, while FGV and FGVI were only present in the CA landscape (Fig. 10.2). The use of numerical abundance increases the sensitivity of the functional group analysis. The difference between the three communities became increasingly marked as the levels changed from species to numbers (Fig. 10.2). Noriega et al. (2021) found that dung beetle diversity and functional group richness enhanced dung removal rates and that dung removal performed by telocoprids increased with species richness of telocoprids. In the study area, the dung removal rates can, therefore, be expected to be higher in the SN areas than in the agricultural areas because of the presence of telocoprids in the SN area, while these groups are absent from the agricultural areas.

In the agricultural areas, dung beetle assemblages were impacted by both grazing intensity and change of plant cover in crop fields. Perrin et al. (2020) found that grazing intensity was detrimental to the largest dung beetle species, while the endocoprids were favored. In the present study area, the absence of telocoprids in the agricultural areas may be explained by an increase in grazing intensity in these areas since these groups were also absent from pastures grazed by cattle in the agricultural landscapes (Table 10.2). Simba et al. (2022) found a positive effect of vegetation cover on size distribution in a dung beetle assemblage. The absence of telocoprids in the present study area may also be explained by the changes in vegetation in crop fields where a decrease in vegetation influences the larger telocoprids negatively. Sarmiento-Garcés and Hernández (2021) demonstrated that the ecosystem function of dung removal by dung beetle assemblages was strongly related to the richness and biomass of dung beetles, which were influenced by tree density and air and soil temperatures. They found that areas without tree cover can lose up to 80% of the dung beetle community and up to 90% of the dung removal capacity.

The Shannon diversity (H) was higher for the dung beetle assemblages in the SN landscape and higher at SN2 than SN1 (Fig. 10.3). The dung beetle diversity was therefore highest at the higher elevation in the SN area. In the agricultural landscapes, the dung beetle diversity was higher in the crop fields (CA1) of the CA system than the crop fields (CT1) of the CT system, while there was no significant difference in the diversity of dung beetle assemblages in the pastures grazed by cattle in these two systems (Fig. 10.3). Changes in habitat probably had the principal effect on dung beetle assemblages in the study area. The diversity in the dung beetle assemblages decreased as grazing pressure and agricultural disturbance increased and habitats became more isolated from natural areas. Filgueiras et al. (2011) found fragmentation and isolation to be the most significant variables for changes in dung

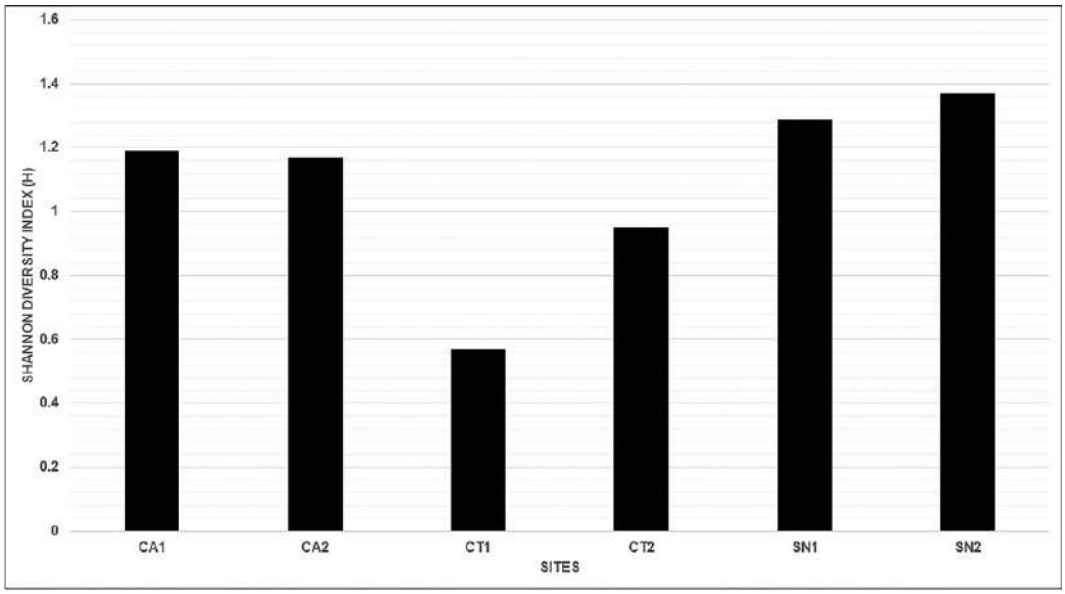

FIG. 10.3 Shannon diversity (H) in dung beetle assemblages at different landscapes and sites in the study area. *(Credit: The authors.)*

beetle species richness, while Shahabuddin et al. (2010) found that dung beetle species richness, abundance as well as species composition, characterized by decreases in mean body size, changed with land-use intensity, indicating that dung type is less important than habitat type for determining assemblage structure of dung beetles.

4 Conservation areas and their ecological indicators

Because of a rapid loss of natural habitats and increasing environmental degradation, there is an urgent need to find a balance between protecting natural habitats and economically viable production in agriculture. The observations in the study area highlight the importance of conservation of natural or SN habitats to protect biodiversity as well as critical ecosystem functions. In this respect, indicators are needed to characterize natural v transformed habitats, and environmentally destructive and environmentally acceptable agricultural practices. Ecological indicators have been widely accepted in conservation as useful tools for monitoring and detecting changes in the environment or habitat conditions. The dung beetle assemblages in the study area showed differences in diversity, functional diversity, as well as different assemblage structure in different habitats. There was a decrease in diversity and functional diversity and change in assemblage structure with functional groups excluded from SN habitats to agricultural habitats with an increase in agricultural intensity and disturbance. The dung beetle assemblages in the study area show the characteristics of a good indicator for habitat degradation and biodiversity decrease. There are several reasons why a dung beetle assemblage is a good indicator of biodiversity loss and ecological deterioration including their fidelity and specificity to a particular type of habitat (Shahabuddin et al., 2014), quick response to habitat degradation, such as destruction, fragmentation, and isolation, (Barlow et al., 2010), sensitivity to regional climatic and ecological conditions, as well as to local edaphic, physiognomic, trophic, and microclimatic factors (Davis et al., 2004). Spector (2006) believe that dung beetles fulfill all the criteria for an effective focal taxon for biodiversity assessment, but standardization and centralization of the methodological, taxonomic, and analytical tools is needed to allow a wider user-base to efficiently collect and interpret data on dung beetle distribution and conservation status.

5 Human activities and their impacts

Dung beetles have wide distribution ranges and a high degree of diversity in many ecosystems. They also play crucial roles in the continued functioning of ecosystems and have been used in biodiversity studies, particularly for assessing the effects of human activities (Davis et al., 2002; Nichols et al., 2007). Observations in the present study showed a change in community structure and function within a dung beetle assemblage, as well as a decrease in diversity, with a change in land use due to agriculture. The impact of land-use change, leading to habitat loss, may have serious implications for ecosystem functioning in these areas. Dung beetle communities are strongly affected by habitat loss and, in areas with agricultural

practices, there is a decrease in the abundance, richness, and total biomass (González et al., 2001), affecting the ecosystem functions, removal and burial of organic material and secondary seed dispersal, they provide (Braga et al., 2013; Noriega et al., 2021).

The response of insect communities to human disturbances is affected by factors at both local and regional levels and the interaction between them. Alonso et al. (2020) observed that factors acting at local and regional scales interact to produce different spatial patterns of dung beetle assemblage response to human land uses. This will have implications for the future of biodiversity and sustainable agriculture. According to Sala et al. (2000), by 2100 the impact of land-use changes on biodiversity is likely to be more significant than that of climate change, nitrogen deposition, species introductions, and changing atmospheric concentrations. It is therefore of utmost importance to determine the impact of land-use changes on the functioning of ecosystems. It is important to recognize the links between ecological functions and biodiversity to evaluate and accurately predict the environmental consequences of human activities (Nichols et al., 2008). Dung beetle assemblages are an integral part of any ecosystem by providing various integral ecosystem functions. This study showed that increased grazing pressure resulted in changes within dung beetle assemblages. The functional structure of dung beetle assemblages changed because of the exclusion of important functional groups as agricultural intensity increased. Perrin et al. (2020) showed that grazing intensity acts as an environmental filter on dung beetle assemblages, selecting species according to traits. The increasing disturbance caused by the expansion of cattle grazing is recognized as exerting one of the greatest effects on biodiversity (Barragán et al., 2014), because livestock activity results in a heterogeneous distribution of defoliation, trampling, and excreta (Perrin et al., 2020).

6 Operational and potential monitoring tools to conserve or to recover the biome

With the impact that agricultural expansion and intensification has on biodiversity and resulting ecosystem function, it is essential to conserve existing natural habitats and restore degraded habitats in agricultural areas. Increasing habitat disturbance in agricultural practices result in changes in species composition, with possible local extinction of some species (Hernández and Vaz-de-Mello, 2009). Landscapes differ and each agricultural area needs to be viewed in the particular landscape to find the best approach for protecting further species loss. This approach will not only be dependent on the specific area but also the specific farming practice. Farm-specific management decisions seem to affect both the composition of local dung beetle assemblages and associated ecological functioning (Piccini et al., 2019). A multiscale approach is needed to understand the consequences of management decisions for a variety of ecosystem functions in agriculture (Beynon et al., 2012). Some agricultural practices seem to support dung beetle assemblages, while others severely alter patterns of dung beetle species diversity and abundance. Hutton and Giller (2003) found that dung beetle species were more abundant on organic than intensive and rough grazing farming sites and suggest that increasing the area of land under an organic or ecological farming regime (a low-input system encouraging diverse ecosystems) might increase regional dung beetle populations, while Nichols et al. (2008) found evidence from temperate and tropical systems indicating that local and regional-scale changes in landuse and mammal faunas can severely alter patterns of dung beetle species diversity and abundance. Although dung beetles can utilize a wide range of dung and will readily colonize cattle dung, Sands et al. (2022) emphasize the importance of conserving areas which maintain indigenous large mammal diversity and are protected from livestock incursions. Most dung beetle species are negatively impacted when nonruminant dung types are absent, as more land is converted to livestock agriculture, and cattle are treated with insecticides with consequent contamination of dung with toxic residues (Sands and Wall, 2018). Considering the important role of dung beetles in an ecosystem, the management strategy in agriculture should be to maintain a variety of mammalian species and ensuring that vegetation cover is not severely reduced through grazing and trampling (Simba et al., 2022). Any agricultural practice that limits use of chemicals, improve the soil, and increase and conserve both the plant diversity and the diversity of herbivores will improve the habitat for dung beetles and increase dung beetle diversity and ecosystem function. It will therefore be important to conserve as much natural habitats in and around agricultural fields. An improved understanding of the ecological importance of dung beetles is necessary to understanding the consequences of diversity loss in natural and human-dominated ecosystems.

7 Concluding remarks

As a result of strong competition, dung beetles evolved different strategies to effectively use dung as food source in both the adult and larval stages. A dung beetle assemblage is made up of a diversity of different species with a variety in size and dung use strategies. Dung beetle assemblages are important in ecosystems to maintain healthy soil and can be used as an indicator of changes in ecosystem health as a result of agricultural pressure. Land-use change from natural to agricultural causes a change in the community structure of dung beetles with some species and functional groups becoming more dominant, while others disappear as

agricultural intensity increases. For effective ecosystem function, the whole assemblage is needed, and the absence of certain functional groups can have drastic consequences for ecosystem function of the dung beetle assemblage in an ecosystem. For efficient ecosystem functioning in agricultural habitats, the protection of natural habitats and the restoration of degraded agricultural habitats is essential.

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Section D

Prairies and steppes

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Chapter 11

Prairies and steppes: Cradles of grassland biodiversity

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1 Introduction

Prairies and steppes are temperate grassland habitats, comprising vast open landscapes of grasses and herbaceous plants or graminoids with little or no trees and shrubs. They hold a large diversity of grass species, mosses, and other plants, dominated by grasses of the Poaceae family. These grassland regions, found in various parts of the world, play a pivotal role in supporting biodiversity, regulating climate changes, and providing ecosystem services. These grasslands provide wildlife habitats including birds, insect, and many other species and food for grazing animals. The deep roots of the grasses hold soil and prevent soil erosion. Besides, they play a key role in reducing greenhouse gas and climate changes by capturing and storing carbon in the soil (Steiner et al., 2019; Bai and Cotrufo, 2022). In addition, prairies support high microbial diversity belowground, which is essential for soil system. High microbial diversity creates resilience against the negative impacts of climate change such as drought (Upton et al., 2018). Therefore, conservation of the grassland ecosystem is crucial for maintaining overall biodiversity (Duchardt et al., 2018).

The grassland ecosystem has been threatened and devastated over time by both physical and environmental factors, including continuous overgrazing, conversion of grassland into commercial areas, climate change, elevated atmospheric carbon dioxide concentrations, historical wildfire occurrences, drought, deterioration of the land, and invasion of nonnative species (Blair et al., 2014; Jamil et al., 2022; Khan et al., 2022; Ali et al., 2023; Kim et al., 2023). These factors effect the grassland biodiversity including flora and fauna; therefore, conservation efforts are of foremost importance.

2 Global distribution of prairies and steppes

Prairies are the grasslands, primarily found in North America, particularly in the great plain region rocky mountain of Mississippi valley and south Saskatchewan to the Gulf of Mexico (Scott, 1995). These prairies covering the region of Canada (Alberta, Saskatchewan, and Manitoba) extending to the mid-west of United States (Minnesota, Wisconsin, Iowa, Illinois, Indiana, Missouri, North and South Dakota, Kansas, and Nebraska), west (Colorado, Wyoming, and Montana) and south (Oklahoma, Texas, and Arkansas), and further extending to some parts of Mexico (Fig. 11.1). These prairies comprise approximately 15% of the continent (Scott, 1995; Anderson, 2006; Smith, 2010). Ecologists have divided the central grasslands of North America into three types of bases on the height of the native grasses, i.e., tallgrass, mixed-, and shortgrass prairies. The tallgrass prairies are in the eastern region with heavy rainfall (625–1200 mm precipitation), comprising C_3 and C_4 species and Forbes. Mixed-grass prairies are situated in the central region with moderate rainfall (375–625 mm precipitation) composed of tall and short grasses, while shortgrass prairies are found in the western part with lower rainfall (260–375 mm precipitation) (Anderson, 2006; Blair et al., 2014).

Steppes, on the other hand, are grasslands with short grasses. Palaearctic steppes form a vast belt with a size range between 8 and 13 million km^2 across the middle of Eurasia, and stretched from Hungary through Ukraine, Russia, Kazakhstan, Mongolia to Northeastern China (Dengler et al., 2014; Wesche et al., 2016). Steppes are mostly hilly regions, foothills, and lowlands, which are too dry to support trees and shrubs but humid enough to

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FIG. 11.1 Geographic distribution of the prairies of North America covering part of Canada, United State, and Mexico. The picture was taken and modified from Google Earth. The redline showed the region covering prairies. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Dr. Zahida Bano.)*

grow vegetation. The ecological importance of the Eurasian steppe is being the home to massive size of biomass. Steppe's biomes are further classified into subtypes, i.e., typical steppe (3.1 million km^2) cover 25%–100% herbaceous plants, desert steppe (1.5 million km^2) cover 10%–25% of mostly dwarf and low-growing shrubs. Forest steppe (2.9 million km^2) is the zone adjacent to the forest and forms a band of macro-mosaic forest covering moist grassland. Meadow steppes are composed of herbs and low drought-tolerant grasses and alpine steppes (2.9 million km^2) are composed of an intermediate type of steppes and arctic-alpine grasslands (Wesche et al., 2016).

3 Biodiversity of prairies and steppes

3.1 Flora

The grassland ecosystem, prairies, and steppe are home to substantial number of grasses and Forbs. The North American prairies are composed of more than 250 species

of tallgrass prairies dominated by C_4 grasses including *Andropogon gerardii* (big bluestem), *Schizachyrium scoparium* (little bluestem), *Spartina pectinate* (prairie cordgrass), *Sorghastrum nutans* (Indian grass), *Sporobolus cryptandrus* (sand dropseed), and *Panicum virgatum* (switchgrass). The mixed grass prairies are dominating by species including *Agropyron smithii* (western wheatgrass), *Nassella viridula* (green needle grass), and short grass prairies composed of species like *Stipa spartea* (porcupine grass), *Bouteloua dactyloides* (buffalo grass), *B. gracilis* (blue grama), *B. curtipendula* (sideoat grama), and *B. hirsuta* (hairy grama) (Anderson, 2006; Sala et al., 2013; Blair et al., 2014).

In Europe, the dry grasslands host a considerable number of vascular plants (Dengler et al., 2014). The dominant plant in steppes is C_3 grasses unlike the tallgrass prairies of North America, which is dominated by C_4 grasses apart from *Cleistogenes* spp. and *Botriochloa ischaemum* in Middle- and Central Asia (Wesche et al., 2016). In Eastern Europe, the steppe consists of perennial graminoids belonging to the genera *Bromus*, *Stipa* (*Stipa pennata*, *Stipa capillata* L.), *Elymus* (*Elymus caninus, Elymus fibrosus*), *Festuca* (*Festucetalia valesiacae*), and *Agropyron* species and Forbes of the genera *Aster*, *Achillea*, *Artemisia*, *Astragalus*, *Centaurea*, *Inula*, *Linum*, and *Salvia* (Hensen et al., 2010; Török et al., 2020). *S. pennata* is also found Mongolia and Mediterranean regions (Hamasha et al., 2012).

The Middle Asian steppes that form a belt from southern-central Serbian to Kazakhstan extend further as a narrow band between the Himalaya Mountains and Central Asian desert. This region is covered by an enormous number of Forbes (*Filipendula vulgaris*, *Fragaria viridis*) and grasses' species (*Phleum phleoides*, *Helictotrichon schellianum*, *Stipa lessingiana*, *Stipa zalesskii*, *Stipa tirsa*, *Koeleria cristata*, and *Tulipa* spp.). The desert steppe covers species like *Stipa glareosa*, *Stipa krylovii*, *Stipa sareptana*, and shrubs of the *Artemisia* species including *Artemisia lessingiana*, *Artemisia semiarida*, and *Artemisia gracilescens*. The tall mountain regions contain species like *Festuca valesiaca*, and *Thymus* species (Smelansky and Tishkov, 2012).

The Mongolian steppe of Central Asia is dominated by feather grasses of genus *Stipa* and wormwood *(Artemisa)* species. The most common species are *Agropyron cristatum*, *Koeleria cristata*, *Carex duriuscula*, and *Cleistogenes squarrosa* (Cheng et al., 2008). The Tibetan steppe is ground to many floras comprising over 2000 plant species of various families. Dominant species of this region includes *Trikeraia hookeri*, *Stipa purpurea*, *Aristida triseta*, *Orinus thoroldii*, *Sophora moorcroftiana*, and many more (Miller, 2005). Besides, *K. pygmaea* is most notable species in this region.

The Mediterranean Steppes form a discontinuos belt starting from Morocco through Antolia and extending to North Iran to Afghanistan. Dominating species of this region includes *Tipa tenacissima*, *Lygeum spartum*, *Artemisia* spp. (*Artemisia lerchiana*, *Artemisia taurica*, *Artemisia aucheri*, *Artemisia sieberi*) *Salsola* spp. and *Poa bulbosa*, etc. (Wesche et al., 2016).

3.2 Fauna

Prairies and steppes are home to a wide range of insects, birds, and animals. Prairies host a variety of fauna including 208 avian species, wildlife, and insects. These include *Cynomys* (prairie dogs), *Bison latifrons* (bison), *Lepus californicus* (jackrabbit), *Sturnella magna* (eastern meadowlark), *Tympanuchus* (prairie chicken), *Eremophila alpestris* (horned lark), *Ammodramus sarannarum* (grasshopper sparrow), *Ammodramus henslowii* (Henslow's sparrow), and *Bartramia longicanda* (upland sandpiper). Insects are an important part of tallgrass prairies' ecosystem, playing roles as herbivores, pollinators, predators, parasites, and decomposers and comprise 50% of species in the terrestrial ecosystem (Risser, 1988; Flombaum et al., 2022).

Steppes are home to many large animal species including *Equus ferus przewalskii* (Mongolia, North China, Russia, and Kazakhstan), *Equus hemionu* (Mongolia), *Equus kiang* and *Bos mutus* (Tibet), *Gazella subgutturosa* and *Procapra gutturosa* (Kazakhstan and Mongolia), and *Camelus bactrianus* (Mongolia and China). Although large grazing animals disappeared from the European steppe, a small amount of *Saiga tartarica* (saiga antelope) still exist in the European, Russian and Kazakhstan steppes (Mallon and Jiang, 2009; Kamp et al., 2016; Mallon, 2016) and *Gazella cuvieri*, *Gazella gazella*, *Nager dama,* and *Ammotragus lervia* in the steppe of the near East. Burrowing herbivore species include *Marmota* spp., nocturnal Jerboas, *Spermophilus* spp. (ground squirrel) and Pikas (Kazakhstan and Serbia), *O. curzoniae* (Tibet), and *O. pallasi* (Mongolia) (Lai and Smith, 2003; Wesche et al., 2007).

Besides animals, steppes host many bird species. Approximately 29% of birds in European fauna are associated with dry grassland or steppe (Dengler et al., 2014). Many birds use steppes for nesting and breeding. The lark is the most abundant species in the steppe (Kamp et al., 2012). Many bird species are distributed across Mongolia, China, and Serbia including *Aquila nipalensis* (steppe eagle), *Aegypius monachus* (cinereous vulture), and *Falco cherrug* (saker falcon), but some species are country-specific, such as *Melanocorypha leucoptera* (white-winged lark) of Kazakhstan and *Melanocorypha mongolica* (Mongolian lark) in Mongolia (Mainjargal et al., 2013), *Vanellus gregarius* (sociable lapwing) and black lark of the Kazakh steppe (Kamp et al., 2009; Lameris et al., 2016).

4 Determinants of biodiversity

The global distribution of the grassland is regulated by climate, temperature, and precipitation, which prevent the formation of a forest or a desert. For example, in prairies

the climate is more favorable for grasses as compared to trees and shrubs. Precipitation plays a significant role in vegetation distribution. In North and South America, there is a slight difference in elevation along with the precipitation gradient. In North America, the eastern part with more rainfall was dominated by tallgrass parries, but they were replaced by mixed- and shortgrass prairies due to differences in the precipitation gradient (Flombaum et al., 2022). Similarly, the growth of a woody plant in steppes is limited by dry climate and occasional frost (Weaver, 1944).

Likewise, soil composition also plays a role in the distribution of the vegetation. Fine-textured soil has more water-holding capacity than the coarse-textured soil. In coarse-textured soil the water penetrates deeper compared to fine-textured soil. Therefore, grasses dominate in fine-textured soil due to their shallow root compared to the wood shrubs bearing deep roots (Flombaum et al., 2022).

Besides precipitation and soil composition, fire plays a key role in determining the grass-forest ecotone. The dominancy of grass or woody vegetation in those regions depends on the frequency and intensity of fire. Fire inhibits woody encroachment but, in the tallgrass prairies of North America, the woody vegetation is increasing due to the intervention of humans in reducing fire frequency (Blair et al., 2014; Flombaum et al., 2022).

Grazing is a key factor that has significant impact on biomass production and the composition of species in the grassland ecosystem because grazing promotes heterogeneity due to selective consuming of species, soil compaction, and through trampling. Light to moderate grazing of grasslands enriches species diversity; however, some grassland can be degraded by overgrazing (Blair et al., 2014).

5 Thread and management of grasslands

During the past centuries, grasslands have undergone a dramatic loss due to the conversion of arable land, infrastructure and, in the former Soviet Union (Ukraine, Russia, and Kazakhstan) and China, the steppes experience a dramatic loss due to the transformation of steppes in arable land. In Ukraine, only 3% of the former steppes are left, and a great loss occurred in Russia and Kazakhstan as well (Hölzel et al., 2002; Kamp et al., 2011; Korotchenko and Peregrym, 2012). In North America, 99% of tallgrass prairies are converted into agriculture land or are lost due to urbanization (Rowe, 2010).

Another thread to the grassland ecosystem is woody encroachment from the adjacent ecosystem or the invasion of nonnative woody plants. The driving force of woody encroachment could be due to many factors including climate change, overgrazing, nitrogen accumulation, elevated atmospheric CO_2, and alteration in the frequency of fire. Additional threads include invasion of nonnative grass species and exotic grasses such as cheat-grass and knapweed in the North American prairies, which shift the native structure and the nitrogen and carbon cycles (Blair et al., 2014; Petermann and Buzhdygan, 2021).

Habitat loss influenced population decline of birds and animals in the prairies and steppes. The bird population exhibited a substantial decline that comprises approximately 74% of the breeding population and the majority of them are terrestrial birds' population of North America (Rosenberg et al., 2019; Vold et al., 2019). *Ovis orientalis* (Mouflon) of western Aisa exhibit strong population decline due to habitat loss and poaching (Wesche et al., 2016). *Alauda arvensis* (skylark) and *Motacilla flava* (yellow wagtail) of the Eurasian steppe also show significant decline (Kamp et al., 2012).

Considering the ecological importance and subsequent loss of grassland, a conservation and restoration effort of the grassland is genuinely needed. Various research works have been carried out focusing on the restoration of tallgrass prairies in North America, which has declined 82%–99% due to the conversion of grassland into agricultural land (Blair et al., 2014). Currently ~4% of the terrestrial biome is underprotected in Central and North America, Sub-Saharan Africa, and the Caribbean (Petermann and Buzhdygan, 2021). The strategy for the restoration of grassland like prairies and steppe includes reintroduction of a native species in formerly cultivated land in a particular region, reduction of woody encroachment, and removal of invasive species.

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Chapter 12

European steppes and forest-steppes

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1 Introduction

Grasslands are dominant landscape elements, covering approximately 40% of the terrestrial land surface of the globe (White et al., 2000). They provide essential ecosystem services, ranging from erosion control and water retention to biological pest control and pollination (Bengtsson et al., 2019). Based on a recent estimate, European grasslands cover more than 650,000 km^2, although it is difficult to assess the extension of grasslands, as there are several other types of habitats that harbor patches of grass dominated vegetation (e.g., shrublands, forest-grassland complexes, marshlands, fens, or heathlands) (Dengler et al., 2020). Grasslands can be classified into two basic categories: intensively managed grasslands are heavily fertilized and intensively used and degraded, while seminatural and natural grasslands possess valuable and diverse vegetation. Seminatural grasslands can be classified further as primary or secondary grasslands: Primary grasslands are treeless because they are too cold (alpine grasslands), too dry (steppes), or otherwise edaphically unfavorable (saline grasslands on salty and compacted soils, rocky grasslands on shallow soils) to sustain continuous tree cover. In contrast, secondary grasslands are man-made habitats mostly created by forest cuts and sustained by regular human interventions in the form of mowing or livestock grazing. Without these types of management, they are subject to natural forest succession and will be encroached upon by shrubs and trees (Dengler et al., 2014).

Among the primary grasslands, steppes belong to the largest continuous grassy biomes on Earth (Wesche et al., 2016). There are various and, in some cases, somewhat contrasting definitions for the 'steppe.' In the broadest view, North American prairies are also considered synonyms of steppes and included into steppe classification. We narrow the definition of steppes to typical treeless vegetation types in cool temperate vegetation zones of Eurasia, characterized by a climate with cold winters, hot summers with a marked dry period but also summer rains (Hurka et al., 2019). Steppes are macroclimate-shaped vegetation types characterized by the dominance of drought-tolerant perennial grasses and forbs, in some types with low shrubs, dwarf shrubs, chamaephytes, and early flowering geophytes (Wesche et al., 2016).

2 Geographic location, origin, and climate

The steppe biome covers a nearly 1000-km wide belt from the eastern part of Europe to the far East of the Eurasian continent, stretching nearly 8000 km. European zonal steppes and zonal forest-steppes covered historically wide areas from the western edge of the Carpathian Basin to the European part of Russia (Fig. 12.1).

Forest-steppes can be considered the natural vegetation in large areas in Bulgaria, Hungary, Moldova, Romania, Russia, Serbia, and Ukraine (Erdős et al., 2022), while European true steppes and desert-steppes can be found in their largest extension in Ukraine and European Russia. There are fragments of steppe-like grasslands in Central Europe, in the dry valleys of the Central Alps, in the upper valleys and in several other parts of central and southern Germany, Lower Austria, and northern Bohemia in the Czech Republic, with a flora characterized by high abundance and richness of steppe elements (Hurka et al., 2019; Kirschner et al., 2020). Many authors question the primary and climatic origin of these grasslands considering them secondary sustained by azonal or extrazonal habitat conditions and regularly managed by mowing and/or grazing (Dengler et al., 2014). In the Iberian Peninsula, there are very special desert grasslands with high cover of dwarf shrubs, called Iberian steppes. These valuable open habitats are sustained by continentality and aridity and characterized by the presence of steppe species

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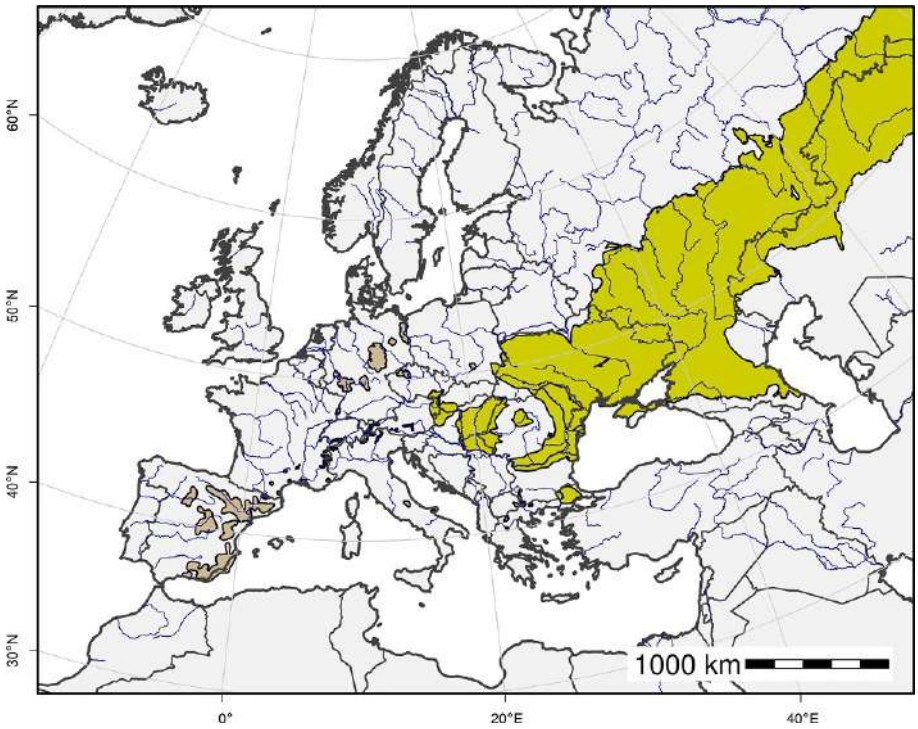

FIG. 12.1 Potential map of zonal (*dirty-yellow color*) and extrazonal (*wheat color*) steppe habitats in Europe. We compiled the zonal steppe distribution from Olson et al. (2001), Erdős et al. (2018), and Bede-Fazekas et al. (2023), while extrazonal steppe distribution was compiled by Kirschner et al. (2020) based on Niklfeld (1974), Meusel et al. (1992), and Braun-Blanquet (1961). Map lines delineate study areas and do not necessarily depict accepted national boundaries.

(Sainz Ollero and van Staalduinen, 2012). The total extent of the azonal or extrazonal steppe habitats is estimated to be approximately 1% of total steppe habitats, but they play a decisive role in the conservation of steppe biodiversity (Kirschner et al., 2020).

Grass-dominated vegetation originated in the late Oligocene and became dominant in Eurasia in the late Neogene coinciding with a cooling trend and increasing seasonality of precipitation and continentality. The first steppic habitats were formed by the retreat of closed forests in the Paleogene to early Miocene in Central Asia. This trend was followed later in Europe by the drying trend starting in the middle Miocene, which favored the extension of steppe habitats. During the cold and dry glacial periods, steppes reached their highest extension in the late Pleistocene and, together with tundra vegetation, dominated Europe from higher northern to middle latitudes (Binney et al., 2017; Hurka et al., 2019). The periodical glacials—interrupted by rather short interglacial periods—caused periodical contraction and extraction of steppes resulting in a complex biota consisting of both cold- and warm-adapted species.

Recent European steppes are characterized by a moderately continental climate with a precipitation median of 520 mm and a spatial median of mean annual temperature of 6.5°C (Wesche et al., 2016). The precipitation is more evenly distributed in the year than in Middle Asian or Tibetan steppe regions and the likeliness of temperature extremes is also lower (Wesche et al., 2016). Typical soils of European steppes are chernozemic soils with deep soil horizon and high humus content; thus, they contribute greatly to carbon sequestration (Lal, 2004). These soils are highly suitable to agriculture, especially for cereal production.

3 Biodiversity and its conservation

Despite their seemingly uniform character, steppes show a remarkable diversity at several scales. At a very coarse scale, the steppe can be divided into three broad latitudinal belts (Walter and Breckle, 1989; Chibilyov, 2002; Wesche et al., 2016): (1) forest-steppes as a transition zone toward closed forests (= meadow steppes with forest patches) (see also Erdős et al., 2022; Loidi et al., 2023), (2) true (grass) steppes classified into various subtypes mostly based on the dominant grass species, and (3) desert-steppes, a transition zone toward semideserts.

The northernmost parts feature a mosaic of forest and grassland patches and are referred to as forest-steppes (Fig. 12.2). The grassland component of this type is sometimes called meadow steppe. It is a dense and rather high sward with many forbs. Typical graminoid species include *Bromus erectus*, *Carex humilis*, *C. montana*, *Festuca rupicola*, *Poa angustifolia*, *Stipa capillata*, and *S. pennata*, while frequent forb species are *Adonis vernalis*, *Anemone*

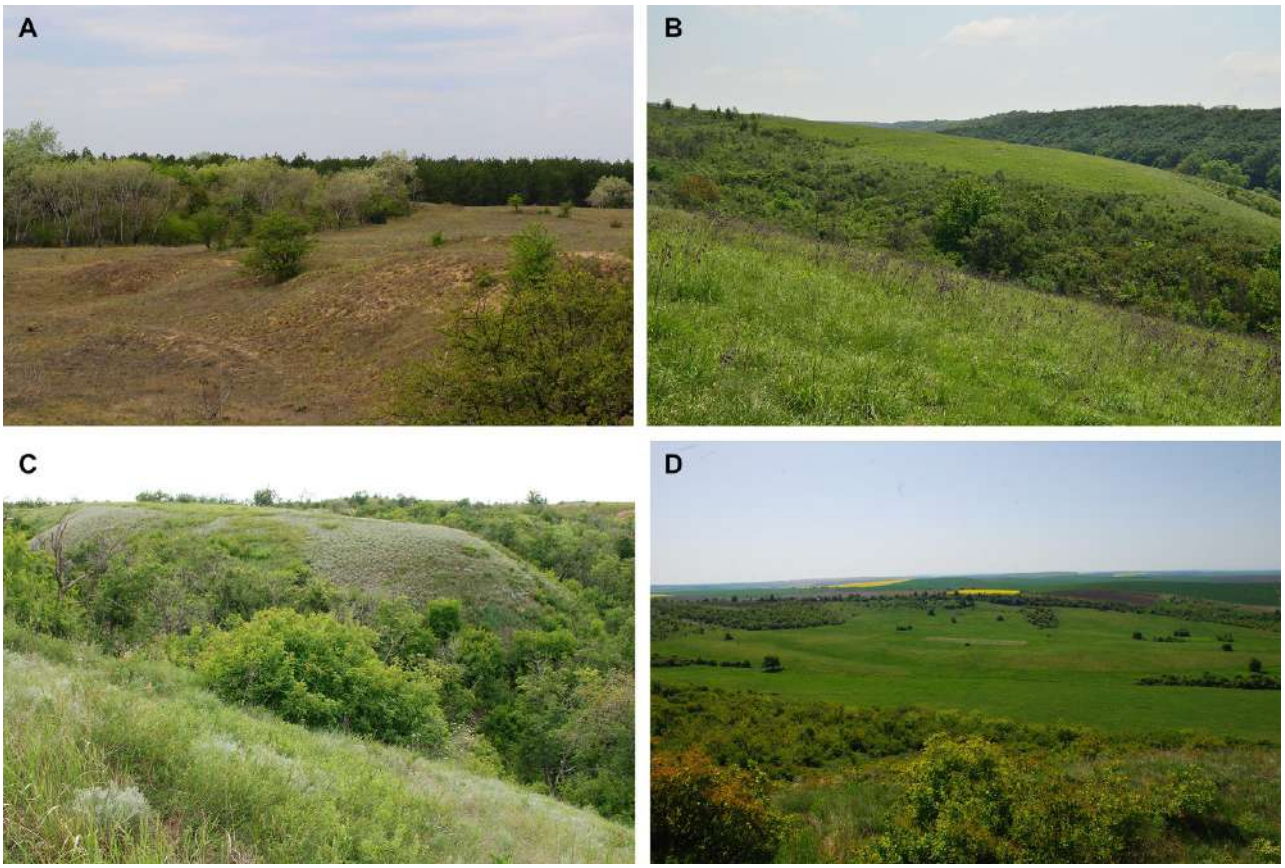

FIG. 12.2 Examples of forest-steppes: (A) sandy forest-steppes in Hungary (Kiskunság region, Tázlár), (B) forest-steppe vegetation with *Galatella villosa* in Northern Bulgaria (Sovata, Veliko Tarnovo), (C) extrazonal forest-steppe vegetation in European Russia (Kalach-na-Donu, Volgograd), and (D) view of the "Bozhurluka" Steppe Reserve in Northern Bulgaria with extrazonal true steppe in a forest-steppe matrix (Gorna Studena, Veliko Tarnovo). *(Photo credits: Csaba Tölgyesi (A) and Gábor Sramkó (B, C, D).)*

sylvestris, *Filipendula vulgaris*, *Fragaria viridis*, *Galium verum*, *Iris aphylla*, *Leucanthemum vulgare*, *Ranunculus polyanthemos*, *Salvia pratensis*, *Trifolium alpestre*, and *T. montanum* (Berg, 1958; Walter and Breckle, 1989; Korotchenko and Peregrym, 2012). Meadow steppes are reported to host more than 1000 vascular plant species per 100 km^2 (Zlotin, 2002). These communities are polydominant and thus have a large plot-level diversity. Walter (1968) lists 115 vascular plant species from a 100 m^2 plot near Kursk, while the number of plant species per 1 m^2 varied between 37 and 56 in the regions around Kursk and Voronezh (Chibilyov, 2002). Very high per plot species richness values were reported from the northwestern parts of the steppe biome (Czech Republic: Klimes et al., 2001; Romania: Dengler et al., 2009, 2012; Ukraine: Kuzemko et al., 2016; Roleček et al., 2019), but due to their transitional position toward the closed forest biome, it is possible that these communities are anthropogenically created secondary grasslands.

Toward the south, the forest-steppe is replaced by the true steppe (Fig. 12.3), which is less high, less dense, poorer in herb species, while *Stipa* species are more abundant here (*S. capillata*, *S. pennata*, *S. lessingiana*, and *S. ucrainica*; Walter and Breckle, 1989; Korotchenko and Peregrym, 2012). Other important grass species include *Bromus inermis* and *Koeleria pyramidata*. Annuals such as *Draba verna* and *Holosteum umbellatum* become more typical, as they can establish in the open gaps within the sward. Geophytes, including species of the genera *Crocus*, *Gagea*, *Ornithogalum*, and *Tulipa* are also important elements of this zone. Other typical species include *Falcaria vulgaris*, *Paeonia tenuifolia*, *Salvia nutans*, and *S. nemorosa*. As there is a general decrease in plot-level species richness toward the south, the true steppe is less diverse at the local scale than the meadow steppe (Walter and Breckle, 1989).

The southernmost parts of the steppe biome are called desert-steppes (Fig. 12.4). They are characterized by the decreasing dominance of *Festuca* (e.g., *F. valesiaca*) and *Stipa* species (e.g., *S. lessingiana*), and the increasing importance of *Agropyron cristatum*, *Artemisia taurica*, and *A. maritima* (Walter and Breckle, 1989; Wesche et al., 2016). The vegetation cover is small. The three main latitudinal zones of the steppe biome may further be subdivided. For example, Chibilyov (2002) distinguishes three subzones within the forest-steppe, the true-steppe, and the desert-steppe zones.

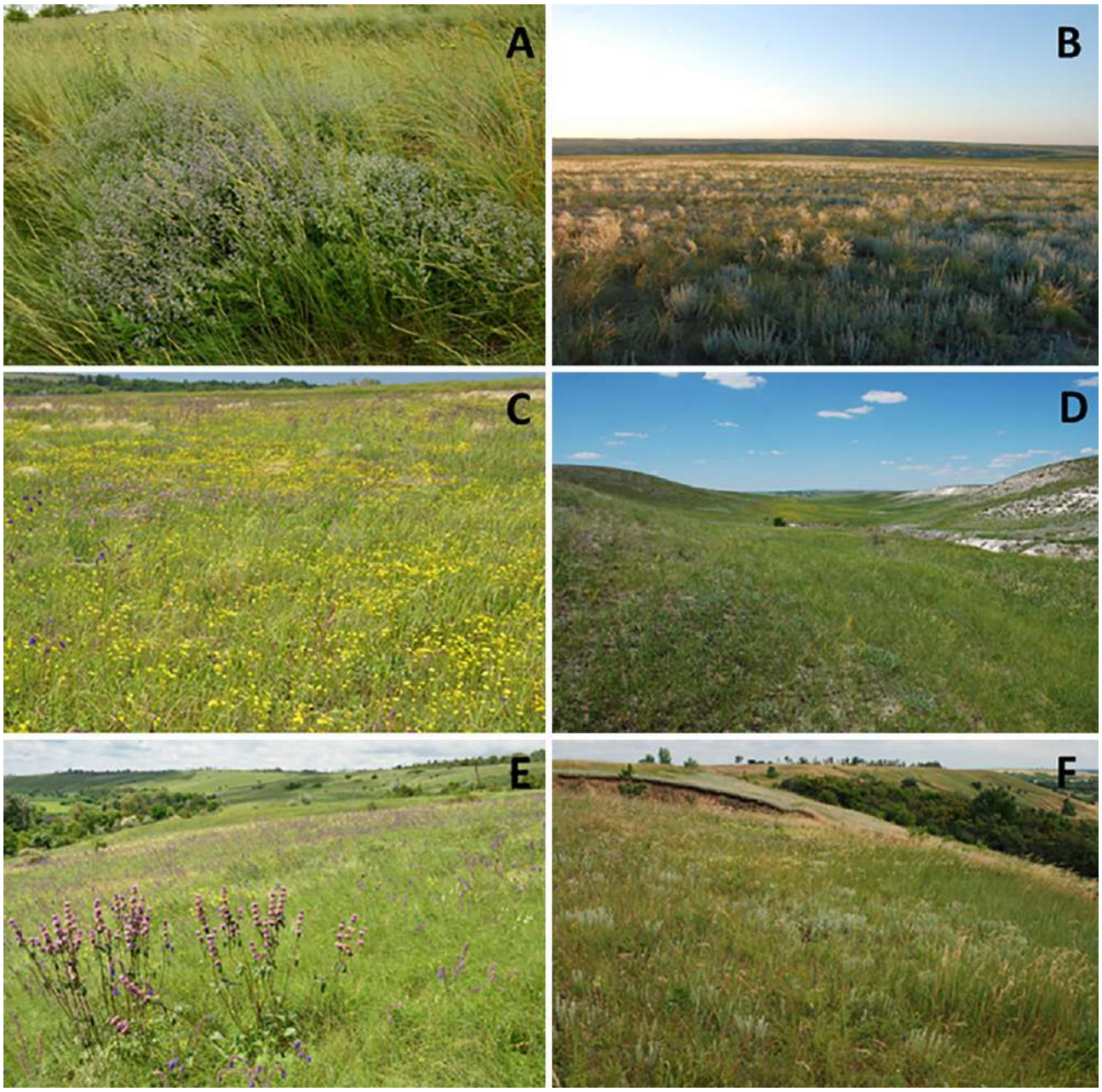

FIG. 12.3 Examples of true steppes: (A) A flowering individual of *Nepeta parviflora* in a forb-rich true steppe in Southern Ukraine (Burhunka, Kerson), (B) *Stipa artemisia* true steppe in European Russia (Volgograd District), (C) true steppe with the massive flowering of *Linum flavum* in southern Ukraine (Troitske, Zaporizhzhia), (D) true steppe in European Russia (Bolshaya Golubaya River, Volgograd District), (E) true steppe vegetation on south-facing slope of the valley of Samara River in Southern Ukraine (Malomykolayivka, Dnipropetrovsk), and (F) true steppe on a steep slope in European Russia in the valley of Savala River (Krasnoe village, Novohopyorsk, Voronezh). *(Photo credits: Gábor Sramkó (A–F).)*

The landscape-scale diversity of the steppe is increased by the fact that minor differences in topography or soil result in the emergence of different plant communities, usually within close proximity (Chibilyov, 2002). Specific locations with saline or stony soils even gave rise to the formation of endemic species (Chibilyov, 2002; Kricsfalusy, 2012). Korotchenko and Peregrym (2012) mention 11 plant alliances and 15 plant associations from the Ukrainian steppes, although the total number of dry grassland associations reported from Ukraine is as high as 126 (Kuzemko et al., 2014). In the forest-steppes of the Carpathian Basin, Molnár et al. (2012) distinguished nine very rough grassland habitat categories, each of which may contain several plant associations.

At a fine scale, the activity of burrowing animals further enhances diversity. These animals create new microhabitats

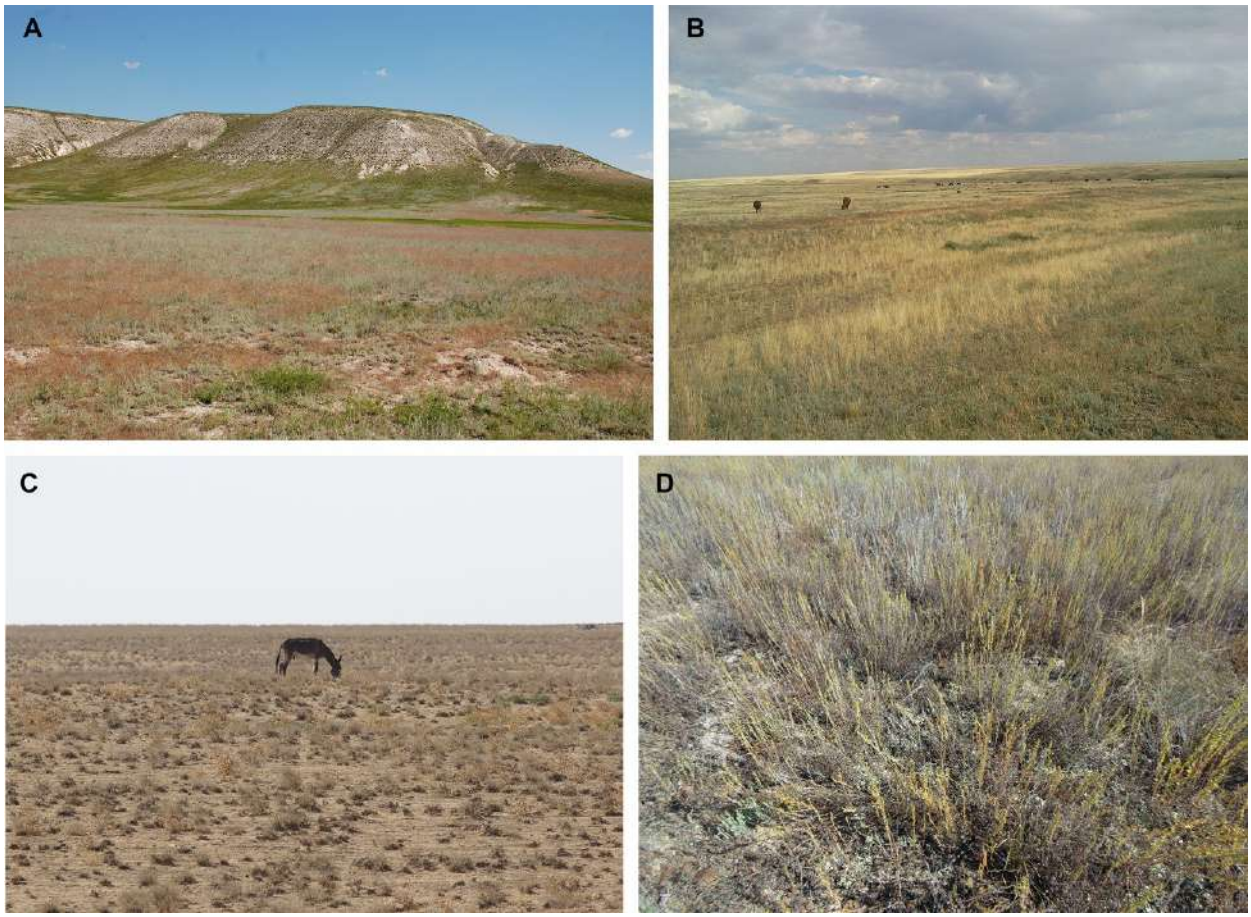

FIG. 12.4 Examples of desert-steppes: (A) edaphic semidesert-steppe in European Russia (Bolshaya Golubaya River, Volgograd District) with *Artemisa* steppe in the foreground and calcareous steppe in the background, (B) zonal semidesert-steppe in European Russia (south of Elista, Kalmykia), (C) donkey grazing in the desert-steppe in the European part of Kazakhstan (Caspian lowlands), and (D) closer look of the desert-steppe with the dominance of *Artemisia* spp. in the European part of Kazakhstan (Caspian lowlands). *(Photo credits: Gábor Sramkó (A, B) and Csaba Tölgyesi (C, D).)*

around their burrows by significantly altering abiotic site factors. For example, Walter and Breckle (1989) note that the soil near the opening of the burrows of *Marmota bobak* and *Spermophilus citellus* is poor in humus and rich in calcium, while *Microtus* species and *Lagurus lagurus* accumulate urine and feces near the burrow. The soil of fox dens was shown to have increased nitrogen content, while the amount of litter and the cover of the vegetation was lower on the den than in the neighboring area (Godó et al., 2018). Plant species richness was significantly lower on fox dens than in the surrounding matrix, but dens provide habitats for weedy species and annuals and biennials, which have few opportunities to survive in the closed grassland (Godó et al., 2018). Thus, foxes enhance vegetation patchiness and increase community-scale plant diversity. The mounds of *Spalax leucodon* proved compositionally different from and significantly more diverse than the surrounding vegetation (Valkó et al., 2022), although differences in diversity may not always be significant (Zimmermann et al., 2014).

Functional diversity of European steppes is assumed to be high (Erdős et al., 2018; also see Walter and Breckle, 1989), although studies in the topic are extremely scarce (Erdős et al., 2023). Within the *Poaceae* family, bunch grasses and rhizomatous, narrow-leaved and broad-leaved grasses are typical in the steppe biome, while the size range includes minute annuals like *Poa bulbosa* and large perennials such as *Chrysopogon gryllus*. The diversity of other graminoids, especially *Carex* species, is also notable. Although European steppes are dominated by C3 species, the C4 strategy is also represented, primarily among the grasses (e.g., *Bothriochloa ischaemum* and *Chrysopogon gryllus*) and in the family *Amaranthaceae*. Species using the CAM photosynthetic pathway (e.g., in the genera *Sedum*) can also be found in steppes. Nitrogen-fixing legumes (e.g., *Astragalus*, *Medicago*, *Lathyrus*, and *Vicia* species) are typical throughout the biome. Hemicryptophytes and geophytes also have an important role among steppe plants, while annuals and biennials become numerous mainly in disturbed sites. Dwarf shrubs and

nanophanaerophytes such as *Fumana procumbens*, and *Thymus* and *Chamaecytisus* species add to the functional diversity of steppes, while small shrubs like *Prunus fruticosa* and *P. tenella* may be abundant locally. Larger shrubs and trees are only present along rivers, in ravines, and in other topographic depressions, as well as in the forest-steppe zone (Chibilyov, 2002). Tumbleweeds (e.g., *Crambe tatarica*, *Gypsophila paniculata*, and *Salsola kali*) make use of the winds sweeping through the open area. Mosses, lichens, and algae are also typical in many steppic environments and can reach very high cover values in certain regions.

The number of animal species restricted to the steppe biome is rather low, as most species also occur in other biomes (Chibilyov, 2002). For example, only ca. 30% of the mammals living in the steppe are restricted to this biome (this figure is ca. 70% in both the boreal forest and the desert) (Chibilyov, 2002). Ungulates are among the most characteristic animals of the European steppe, but their diversity and abundance have been severely reduced during the last few centuries. *Saiga tatarica* has been driven to extinction from most of the European steppe in the 19th century (Berg, 1958; Wilhelmy, 1943; Marcuzzi, 1979); currently, it is critically endangered in the continent. Similarly, *Equus ferus* (Russian wild horse or tarpan) vanished from the European steppe and became extinct globally in the 20th century (Wilhelmy, 1943). *Equus hemionus*, which used to live in the steppes of Ukraine and European Russia (Chibilyov, 2002), is now found only in Asia. Herds of *Bison bonasus* and *Bos taurus primigenius* once roamed the steppes of Europe (Török et al., 2020); the former narrowly escaped extinction (and its sole native population is outside the steppe biome), while the latter became extinct in 1627. *Alces alces* is currently a rare visitor in steppes.

In contrast to ungulates, rodents continue to have large diversity in the steppe. Burrowing species are especially typical. *Spalax* is an obligatory subterranean genus; some of its taxa, such as *Spalax microphthalmus* have stable population numbers, while others, including the taxa of the superspecies *Spalax leucodon*, form highly isolated and extremely small populations (www.iucnredlist.org). In the Carpathian Basin alone, five endemic *Spalax* taxa evolved (Németh and Csorba, 2014). Other typical burrowing steppe rodents are, among others, *Allactaga major*, *Cricetus cricetus*, *Ellobius talpinus*, *Lagurus lagurus*, *Marmota bobak*, *Spermophilus citellus*, and *S. suslicus* (Berg, 1958, 1959; Marcuzzi, 1979; Lavrenko and Karamysheva, 1993; Chibilyov, 2002; Tishkov et al., 2020). Among carnivores, *Mustela eversmannii* is a typical steppe inhabitant.

In the avian group, birds of prey form a diverse group in European steppes, as they benefit from the abundance of small rodents and the carrion of large-bodied domestic ungulates (Wesche et al., 2016). Some of them, including *Aquila nipalensis* and *Falco cherrug* are endangered (www.iucnredlist.org). Relatively large-bodied ground-nesting birds such as *Otis tarda* and *Tetrax tetrax* are another typical group of steppe birds (Berg, 1959; Marcuzzi, 1979; Tishkov et al., 2020). In addition, larks (e.g., *Alauda arvensis*, *Melanocorypha leucoptera*, *M. yeltoniensis*, and *M. calandra*) and buntings (e.g., *Emberiza hortulana* and *E. calandra*) are typical inhabitants of the European steppe (Berg, 1959; Tishkov et al., 2020).

The diversity of reptiles in Europe increases toward the south; thus, steppes harbor more reptile species than closed forests but much less species than deserts (Zlotin, 2002). *Dolichophis caspius*, *Eremias arguta*, and *Podarcis tauricus* can be mentioned as typical inhabitants of the European steppe (Berg, 1959; Marcuzzi, 1979).

Among the insects of the steppe, the following taxa are the most important: Orthoptera, Homoptera, Heteroptera, Coleoptera, Lepidoptera, Hymenoptera, and Diptera (Berg, 1959; Tishkov et al., 2020). According to Sergeev (2021), the orthopteran fauna of the European steppe contains more than 240 species, including some endemics and subendemics. Some insect taxa reach very high levels of diversity at the local scale. For example, in the Karadagh Nature Reserve in the Crimean steppe, 1516 species of Lepidoptera have been found (Konstantinov et al., 2009). Konstantinov et al. (2009) note that, due to the strong winds of the steppe, two major insect groups have considerable advantage here: those with excellent flight skills and those that are flightless. Coprophagous beetles are especially typical. As a considerable proportion of the plant biomass is in the soil in steppe ecosystems, insects utilizing the underground parts of plants are also numerous (Konstantinov et al., 2009). Earthworms constitute an important group of steppe animals, and they play a major role in soil-forming processes; they favor moist environments and get rarer toward the south (Marcuzzi, 1979). In the steppes east of the River Volga, there may be four earthworm species below a $1\,m^2$ area (Marcuzzi, 1979).

According to a global analysis, temperate grasslands are the most threatened ecosystems on earth (Hoekstra et al., 2005). This is particularly true for the European steppe. In Central Europe and Ukraine, based on recent estimates, more than 90% of former true-steppe and forest-steppe vegetation has been destroyed in the last century (van der Sluis et al., 2009; Török et al., 2016). Currently, about 1.4 million km^2 of Eurasian steppes are situated in protected areas, which accounts for about 14.1% of the whole area of the steppe biome (Wesche et al., 2016). This score is higher than the overall percentage of protected temperate grassland habitats of the world (Henwood, 1998); but, if we do not consider the large protected areas in the Tibetan plateau, this goes down to 6.5% (Wesche et al., 2016). Calculated exclusively for the western Palaearctic, these scores are even lower, as the area of protected sites shows an increasing trend from the West to the East (Wesche et al.,

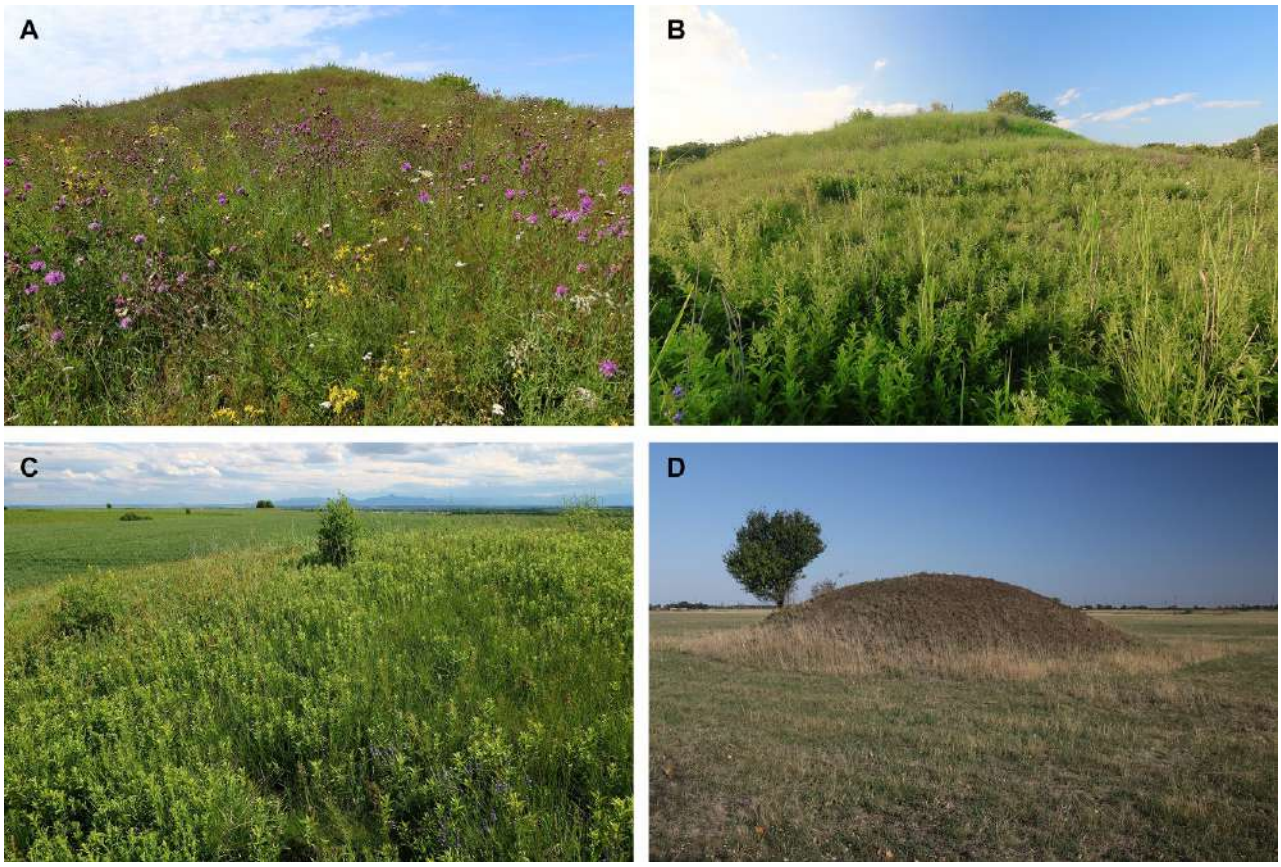

FIG. 12.5 Islands of biodiversity: Steppe remnants on kurgans. (A) 'Erdő-halom' kurgan covered with forb-rich steppe vegetation in East Hungary (Bihar County), characterized by the high cover of *Centaurea scabiosa*. (B) 'Mondró-halom' kurgan characterized by the high cover of *Inula germanica*. (C) Kurgan in the foothills of the Rodope mountains, Bulgaria, covered by a steppe grassland fragment with a high abundance of *Inula germanica*. And (D) 'Sárga-halom' small kurgan embedded in alkaline grasslands in East Hungary (Nagykunság region). *(Photo credits: Balázs Deák (A–D).)*

2016). In Europe, those steppe grassland fragments are preserved that are unsuitable for agricultural production (very dry sand steppes, saline grasslands; Roleček et al., 2019; Molnár et al., 2012; Eliáš et al., 2013) or have a special position or functional role in the landscape (road verges, relict grassland fragments on steep slopes of loess valleys), or situated on so called 'sacred' places like burial mounds called also kurgans (Deák et al., 2020).

Kurgans are man-made landscape elements—mostly burial mounds—from the pre-Roman times distributed from the eastern part of Hungary to Mongolia (Deák et al., 2023). Kurgans were formed from the top-soil material of their surroundings and are characterized by a hemispherical shape with a height of half to 15–20 m and a diameter ranging from a few meters up to a hundred (Deák et al., 2023) (Fig. 12.5).

They were built for diverse purposes, mostly as burial mounds, but also, they have acted as border marks between different tribes or regions and also served as excellent watchpoints in plains (Deák et al., 2016). Several millions of kurgans have been destroyed in the past century, but a couple of hundreds of thousands still prevail and many of them harbor more or less intact steppe grassland vegetation (Deák et al., 2019, 2023). Kurgans support variable habitat conditions because of their topographic heterogeneity, e.g., different vertical positioning or slope aspects provide different levels and timing of irradiation and humidity (Sudnik-Wójcikowska et al., 2011; Deák et al., 2020; Apostolova et al., 2022). Due to this habitat heterogeneity and to the moderate level of anthropogenic impact, kurgans host exceptionally high diversity. For example, 721 plant species were found on 106 Ukrainian kurgans, the total area of which was a mere 100.7 ha (Sudnik-Wójcikowska et al., 2011). A similar study of 138 Hungarian kurgans, with a total area of 38.9 ha, revealed 446 vascular plant species (Deák et al., 2020).

4 Human activities and their impacts

Humans have a long history in the steppe biome, where they practiced a hunter-gatherer lifestyle (Khotinsky, 1993; Kitagawa et al., 2018). Their effect on the steppe, apart from a few possible exceptions, must have been local and of limited intensity. Major human impact in the steppe biome

appeared only in the Neolithic period ca. 7500 years ago, when Europeans began to practice agriculture (Chibilyov, 2002; Novenko et al., 2009; Korotchenko and Peregrym, 2012; Feurdean et al., 2015; Pokorný et al., 2015). Until the beginning of the 19th century, nomadic grazing was the most typical land use in most of the eastern European steppe. As the steppe had been grazed by native ungulates earlier, moderate grazing by domestic animals is usually considered a continuation of this natural disturbance and thus regarded as being capable of preserving a near-natural condition (Molnár and Borhidi, 2003; Smelansky and Tishkov, 2012). Due to the favorable properties of chernozem soils, plowing was also present in the European steppe. However, the human population was so low that most of the steppe remained in an almost natural condition for a very long time (Korotchenko and Peregrym, 2012). In most of the region, the area of arable lands became considerable only during the 16th and 17th centuries, while the process became especially severe in the 19th century (Chibilyov, 2002; Novenko et al., 2009; Korotchenko and Peregrym, 2012; Smelansky and Tishkov, 2012). The forest-steppes and the northern parts of the typical steppes suffered the heaviest losses during this time (Smelansky and Tishkov, 2012). Erosion due to inadequate farming practices caused the destruction of considerable agricultural areas in what is now Moldova, Ukraine, and Russia (Chibilyov, 2002). A second wave of destruction happened between 1917 and 1941, when huge steppe areas were plowed (Chibilyov, 2002; Vasyliuk et al., 2017). The last dramatic transformation of the steppe biome happened during the 'Virgin Lands Campaign' after the Second World War in the early 50s to the early 60s, when the remaining steppe areas were brought under cultivation, especially east of the River Volga (Chibilyov, 2002; Hurka et al., 2019). Most of the typical steppes were destroyed in this period (Smelansky and Tishkov, 2012).

Most of the large ungulate fauna was extirpated by the beginning of the 20th century: *Equus hemionus* became extinct in Ukraine in the 16th century and disappeared from the region between the Volga and the Ural Mountains in the 19th century. *Equus ferus* became extinct in Ukraine in the 19th century (Chibilyov, 2002).

Today, larger and more or less continuous steppe fragments in Europe are found almost exclusively in nature reserves. In Ukraine, it is estimated that only 3% of the original steppes have survived up to now (Korotchenko and Peregrym, 2012). Tishkov et al. (2020) estimate that ca. 5% of the original steppes of European Russia have survived west of the River Volga; the few remnants are mostly small and isolated (Smelansky and Tishkov, 2012). The situation is somewhat better only in the region east of the River Volga, where intact or almost intact steppes cover between 10% and 23% of the area (Chibilyov, 2002; Smelansky and Tishkov, 2012).

After the collapse of the socialist regimes in Central and Eastern Europe, the transition from collectivized, state-run agriculture to a market-driven economy caused significant changes in the extent and intensity of agricultural land use. As a result, in the last few decades, the area of plowed land has dramatically decreased, and vast arable areas, including large areas in Central and Eastern Europe as well as temperate regions of Russia, have become abandoned (Schierhorn et al., 2013; Hurka et al., 2019). This offers a unique opportunity for passive restoration through the spontaneous recovery of steppe habitats or facilitates the intentional restoration of these habitats (Korotchenko and Peregrym, 2012; Smelansky and Tishkov, 2012).

Currently, afforestation is among the most serious threats facing steppes, except in Russia, where it is not a widespread phenomenon (Tishkov et al., 2020; Török et al., 2020; Tölgyesi et al., 2022; Dubyna et al., 2023). The fight against climate change may include afforestation, but it has been shown that afforestation in the steppe biome causes more harm than good. Artificial tree plantations, especially those of nonnative species, support very low taxonomic and functional diversity and have negligible ecological value (e.g., Ho et al., 2023). Moreover, tree plantations in steppe and forest-steppe environments desiccate the lower soil layers and create regional water shortage (Walter and Breckle, 1989; Tölgyesi et al., 2020). This is especially severe if *Pinus nigra* or *P. sylvestris* is used for afforestation, as they transpire water throughout the year. In Ukraine, 80% of tree plantations established in the steppe zone either burn down or dry out in the first 2 years (Dubyna et al., 2023). Despite scientific evidence, afforestation is still advocated by decision-makers, partly as green-washing (Korotchenko and Peregrym, 2012; Smelansky and Tishkov, 2012; Kuzemko et al., 2014; Tölgyesi et al., 2021).

Due to global climate change, European steppes are experiencing increasing temperatures currently. Though the trends for precipitation are less clear, summer precipitation seems to decrease and is projected to continue to do so in the upcoming decades (Dubyna et al., 2023). Studies on the history of the European steppe show that the position of the biome has changed according to modifications in climate (e.g., Novenko et al., 2016; Shumilovskikh et al., 2018). Thus, it is reasonable to assume that ongoing climate change will result in a northward shift of the steppe, meaning that the species composition of a given site will change toward more dry-adapted species (Török et al., 2020; Dubyna et al., 2023). In addition, a general decrease in species richness can be expected, which is also suggested by space-for-time studies (e.g., Kovács-Láng et al., 2000; Erdős et al., 2019). Ecosystems' resistance and the nonlinear nature of ecological responses to climate make predictions extremely difficult.

One of the most serious threats facing the European steppe is the spread of invasive species (e.g., Molnár et al., 2012; Erdős et al., 2018). In Russian steppes, *Chaerophyllum aureum* and *Galega orientalis* are among the most successful herbaceous invaders (Sukhorukov, 2011). *Asclepias syriaca* is an additional dangerous invasive species (Holoborodko et al., 2016). One of the most interesting features of steppes, which distinguishes them, for example, from prairies of the North-American continent, is the dominance of C3 grasses (Hurka et al., 2019). Because of the ongoing trends of climate change, the spread of invasive species, especially drought stress-adapted C4 grasses, poses a recent threat. The spread of invasive North American C4 prairie grass, *Sporobolus cryptandrus,* was reported from large areas of Hungary, Ukraine, and Russia (e.g., Alekseev et al., 1996; Kuvaev and Stepanova, 2014; Gouz and Timoshenkova, 2017; Maltsev and Sagalaev, 2018; Török et al., 2021). The species was proven to be intensively spreading and highly drought tolerant, with a very effective dispersal strategy both in space and time, and currently covers large areas with more or less monodominant stands (Török et al., 2021).

Among woody species, *Ailanthus altissima* and *Robinia pseudoacacia* are the most dangerous ones (Protopopova et al., 2006; Holoborodko et al., 2016), while *Elaeagnus angustifolia* threatens the integrity of desert-steppes in southern Ukraine (Protopopova et al., 2006; Sudnik et al., 2009). The majority of invasive plant species originate from North America (Kucher, 2015; Holoborodko et al., 2016), where many of them grow in prairies, the North American equivalents of steppes (e.g., *A. syriaca*, *Erigeron canadensis*, and *S. cryptandrus*). Invasive animals in the European steppe include *Diaspidiotus perniciosus* (Holoborodko et al., 2016).

Steppe plants are adapted to grazing (Wesche et al., 2016), but over- and under-grazing threatens the ecological integrity of steppes. Over-grazing is a serious threat primarily in the southern, arid steppes, while under-grazing is more problematic in the northern and northwestern parts (forest-steppes) (Tishkov et al., 2020).

Fires represent a natural disturbance in the steppe biome, but anthropogenic burning that is too frequent or happens at the wrong season can negatively influence the steppe, leading to a decrease in species richness and the destruction of threatened species (Korotchenko and Peregrym, 2012).

The most recent threat to some European steppe areas is the armed conflict in the eastern part of Ukraine, which contains some of the oldest and most valuable steppe reserves. Beginning with 2014, the movement of military vehicles, explosions, fires, and the construction of fortifications have done considerable damage to the vegetation and the soil, while explosions also cause sulfur and heavy metal contamination (Vasyliuk et al., 2017; Dubyna et al., 2023). Moreover, in the chaotic situation, conservation legislation cannot be enforced, which results in illegal mining operations, logging, and poaching. There is no doubt that the situation has become even worse since 2022.

Other human activities such as mining, off-road racing, and overexploitation are less conspicuous but have nevertheless considerable destructive effects (Korotchenko and Peregrym, 2012; Smelansky and Tishkov, 2012; Dubyna et al., 2023). While most of these effects are local, this is not always the case. For example, many showy plants became endangered because they have been collected by the local human population (e.g., *Adonis volgensis*, *Crocus reticulatus*, *Hyacinthella pallasiana*, and *Paeonia tenuifolia*). Mining (including oil and gas extraction) is particularly destructive in the easternmost parts of the European steppe (Tishkov et al., 2020).

5 Operational and potential monitoring tools to conserve or to recover the biome

Monitoring the vegetation of steppes has a long history in Europe (Walter and Breckle, 1989). Most studies focused on how vegetation develops under different grazing regimes. For example, early investigations in Askania Nova, a Ukrainian steppe reserve founded in the 19th century, revealed that over-grazing results in major compositional changes, including the disappearance of several forbs and the increasing dominance of annuals and grazing-resistant species. In contrast, the absence of grazing leads to the formation of a thick layer from dead plant material, which eventually results in patches devoid of living plants (Wilhelmy, 1943; Walter and Breckle, 1989). The restoration actions subjecting grasslands have a rich literature in Europe and encompass various grassland types and techniques from wet to dry grasslands restored completely by technical reclamation to passive restoration (e.g., reviews of Hedberg and Kotowski, 2010; Kiehl, 2010; Török et al., 2011). While restoration actions and conservation measures are urged or frequently implicated with the support of the European Union in steppe-like grasslands (e.g., Cremene et al., 2005; Török et al., 2010) or some forest-steppe habitats (e.g., OAKEYLIFE project, LIFE16 NAT/HU/000599), much less is known about such actions outside of the European Union. Knowledge on the restoration of steppes is mostly scattered in the published scientific literature, and heavily focused on the spontaneous regeneration of steppe habitats is Russia and Ukraine following the large-scale abandonment of agricultural activities after the collapse of state-run agriculture (van der Sluis et al., 2009; Nguyen et al., 2018). In South-Eastern Ukraine, military activities from 2014 onward lead also to increased cropland abandonment (Skakun et al., 2019). Authors argue that the spontaneous succession of croplands seems to be a promising restoration tool, especially if the spontaneous recovery processes might be supported by

well-planned livestock grazing, which can provide an enhanced seed transport forming existing steppe patches (proposed by Stroh et al., 2002). However, cropland abandonment cannot be considered a very effective restoration measure in the steppe region: recovered grasslands have no protection from being reused again as croplands. Also, such agricultural fallow lands are frequently targeted by tree plantation campaigns and the awareness of open habitat restoration is generally low compared to afforestation activities, worldwide (Prishchepov et al., 2021; Tölgyesi et al., 2022).

Together with the huge level of cropland abandonment, there has been a dramatic drop in grazing livestock; the complete abandonment of grazing led to increased litter accumulation in large areas (Hölzel et al., 2002; Wesche et al., 2016). This litter accumulation often leads to an increased likelihood of wildfires in the affected regions (e.g., Dubinin et al., 2011).

6 Conclusions

Temperate grasslands are among the most threatened ecosystems in the world, and this is especially true for European steppes, where much of the biome has been lost to agriculture. The European steppe has high taxonomic, functional, and habitat diversity and harbors many species of special conservation relevance. Although only a very small fraction of steppes has survived on the continent, and the last remnants are usually severely fragmented, the last few decades have brought a new, unique opportunity to restore steppes on abandoned arable fields. Moreover, the steppe biome is adapted to grazing by large ungulates, and this means that grazing by domestic animals represents a sustainable practice. Indeed, using steppes as pastures can effectively contribute to their conservation by preserving or even enhancing biodiversity, while also providing economic benefits.

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Chapter 13

American

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1 Introduction

Grasslands are the most important ecosystem for human survival, because we depend on their fertile soils for ecosystem services and for producing food, fibers, fuel, animal feed, and pharmaceuticals. Human activities and global warming threaten grassland health, requiring aggressive land management strategies to conserve and restore Earth's least protected biome (Scholtz and Twidwell, 2022). Grasslands are found in temperate and subtropical regions of North and South America, supporting the growth of diverse plant communities that include numerous grasses and forbs that, in turn, support higher trophic levels and contribute positively to carbon sequestration through high levels of soil organic matter. Biodiversity is higher than might be expected from such consistent niches across the landscape (Petermann and Buzhdygan, 2021). Birds represent the most diverse group of vertebrates, while mammals are well-represented by small rodents and bats, and grasslands host a striking diversity of sunflower, legume, and graminoid plant species (Andrade et al., 2023). Despite their importance, only 2%–3% of grasslands in the Americas are protected (Commission for Environmental Cooperation, 2011–2012; Modernel et al., 2016), requiring determined efforts to assess, monitor, restore, and manage the remaining grasslands.

Ecological indicators are used to provide an overall assessment of ecosystem health without conducting exhaustive, time-consuming, and expensive surveys of every organism and abiotic process in the ecosystem. They involve measuring specific features of a region that act as surrogates that can indicate the conservation value of an area, identify threats to ecosystem health, and characterize land management and restoration progress, among others (Dale and Beyeler, 2001; Niemi and McDonald, 2004; Rákosya and Schmitt, 2011; Gerlach et al., 2013; Wills and Landis, 2018; Wilsey et al., 2019; Czúcz et al., 2021). Ecological indicators may include an evaluation of species richness, diversity, distribution, abundance, reproductive success, fire regimes, lichen health, etc., and investigations may focus on a single species, a group of species, community-level metrics, or a combination of these, depending on the questions being asked (Schiller et al., 2001; Orlofske et al., 2010; Gerlach et al., 2013; Elliott and Johnson, 2018; Czúcz et al., 2021; Solascasas et al., 2022). In grasslands, individual bird species have been assessed as indicators of overall diversity (Wells and Rosenberg, 1999), and groups of birds have been used to indicate ecosystem health and biological integrity (Hudson et al., 2017; Browder et al., 2002). Ecological indicators also include various insects as indicators of remnant health (Orlofske et al., 2010; Fenner et al., 2018), abiotic conditions (Gerlach et al., 2013), and responses to management practices (Solascasas et al., 2022), and other animals can reveal the quality of ecosystem functioning (Orlofske et al., 2010). Ultimately, these indicators are used to communicate ecosystem status to land managers, conservation organizations, stakeholders, policymakers, and the public, anyone who understands the importance of protecting, restoring, and conserving grasslands that have been disturbed or nearly destroyed by human actions.

Humans have contributed to destruction, fragmentation, alteration, and degradation of grassland habitats. Due to their fertility, most grassland ecosystems have been converted into row crop agriculture or grazing lands (Best et al., 2001; Modernel et al., 2016; Tack et al., 2017; de Rito et al., 2022; Ríos et al., 2022). Historically dominant grazing herbivores have been replaced by livestock domesticated for human needs, changing the physical and chemical structure of the landscape and the functions of the ecosystem. Historical fire frequency and patterns that are imperative for soil health and plant reproduction have been disrupted by human activities and urbanization, resulting in the invasion of nonnative species, changes in the structure of the landscape, and reduced availability of appropriate habitats for insects, birds, and small mammals (Herkert, 1994; Samson and Knopf, 1996; Murphy, 2003; Stoner and Joern, 2004; Baldi et al., 2006; Allen et al., 2011; Rahmig et al., 2009; Drummond et al., 2012; Begosh et al., 2020; Gaskin et al., 2021; Rutledge et al., 2023). Pesticide and herbicide use has harmed habitats and species and contaminated soils and water sources (Bartuszevige et al., 2002; Fuhlendorf et al., 2009; Goulson, 2013; Mineau

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and Whiteside, 2013; Pleasants and Oberhauser, 2013; Rosenberg et al., 2019; Li et al., 2020). Species ranges are shifting toward the poles as climate change involves increased temperatures, drier air, and more intense storms (Peterson, 2003; Hufkens et al., 2016; Edwards et al., 2019). As a result of human-modified landscapes, humans are tasked with developing and implementing operational and monitoring tools to restore and protect American grasslands.

Effective land management strategies include disturbance regimes such as grazing, prescribed fire, and sometimes mowing/haying as well as seeding, spraying herbicides, and mechanical removal of undesired species (Owensby et al., 1973; Wilson and Schmidt, 1990; Erhardt, 1995; Feber et al., 1996; Pollak and Kan, 1998; Fuhlendorf et al., 2002a; Towne et al., 2005; Rudolph et al., 2006; Cummings et al., 2007; Freese et al., 2007; Guyette et al., 2012; Grman et al., 2013; Kohl et al., 2013; McMillan et al., 2022). These operational tools are evaluated using both short- and long-term monitoring strategies including the evaluation of ecological indicators, the choice of which depend on overall management goals (Tester and Marshall, 1962; Bureau of Land Management, 1996; Lindenmayer and Likens, 2010; Gerlach et al., 2013). These management strategies or combinations thereof are required for the preservation of American grasslands.

2 Geographic location

2.1 Grasslands

The Earth's grassland biome has historically consisted of vast swaths of relatively flat landscapes containing rolling hills and fertile soils dominated by grasses, forbs, and occasional trees and small shrubs (Rutledge et al., 2023). Grasslands have temperate to subtropical climates with moderate temperatures and rainfall and a history of grazing and periodic fire (Allen et al., 2011; Rutledge et al., 2023). Although other small patches of grasslands exist in North America, such as the Palouse prairie in Washington State (Davis, 2019) and the California Central Valley Grasslands (Scholtz and Twidwell, 2022), here we focus on the grasslands of central North America called the Great Plains (Fig. 13.1). Likewise, though a broader definition of grasslands in South America may include regions such as Chaco and mountain grasslands or savannas (Bioregions 2020, 2020), we focus on the Río de la Plata Grasslands of the southeast (Fig. 13.1; Soriano et al., 1992).

Grasslands may experience temperature extremes but average around −20°C to 30°C (−4°F to 86°F) with rainfall amounts varying greatly across the region (Earth Observatory and NASA, 2024). Perennial, long-lived grasses dominate this landscape, and in undisturbed areas, they reduce erosion because their roots can reach depths of over 1 m (Rutledge et al., 2023). Grassland biomes are maintained by fires, grazing, and other natural and anthropogenic disturbance regimes (i.e., disturbance events that impact temporal and spatial characteristics of an ecosystem; Keane, 2017). Fires are important to the health, function, and sustainability of grasslands, in that fires can release minerals from leaf litter and expose soil, impacting competition among plants by killing woody species while enhancing growth of native grasses and forbs, which can attract large herbivores with new plant growth and can alter the structure of habitats (Helzer, 2021). Additionally, prescribed burns (i.e., controlled vegetation fires set by experts in specific weather conditions) can benefit soil organisms and impact other biological, physical, and chemical properties of the soil (Zhu et al., 2020). Herbivory and grazing also sustain grassland ecosystems by limiting woody growth that may otherwise become established and drive out native herbaceous plants (Petermann and Buzhdygan, 2021). Most soils of native grassland ecosystems are classified as Mollisols, which are dark-colored and rich in humus, making them highly fertile and productive (Liu et al., 2012). Mollisols cover nearly 300 million ha in North America and over 100 million ha in South America (Liu et al., 2012).

2.2 North America

The Great Plains of North America cover more than 300 million ha of the continental interior, extending from the coastal plains of northern Mexico to the southern provinces of Manitoba, Saskatchewan, and Alberta in Canada (Fig. 13.1; Samson et al., 2004). The Rocky Mountains form a stable western border for the region and create a rain shadow effect, from dryer climates in the western Great Plains to wetter climates in the east; the eastern extent of the Great Plains, near the Mississippi River, has historically been a dynamic interface with hardwood forests governed by climate and fire regimes (Engle et al., 2008). Vegetation communities reflect the climate gradient, with shortgrass prairies in the western High Plains, tallgrass prairies in the eastern extent of the Great Plains, and mixed grass communities between these two regions. The Great Plains covers a broad area from north to south as well, resulting in a latitudinal gradient of warmer climates with longer growing seasons in lower latitudes and colder climates with shorter growing seasons to the north (Samson et al., 2004). Within these general climatic trends, mixed soil types can result in unique plant communities that do not fit the typical expectations for the region. In the Sandhills region of north-central Nebraska, which covers over 6 million ha (>61,000 km^2) and features the deepest region of the Ogallala aquifer (Scholtz and Twidwell, 2022), sandy soils lead to greater water retention at lower soil depths, allowing for a unique plant community that more resembles the tallgrass prairie community than would be expected for the longitude (Barnes et al., 1984). In contrast, the plant communities of the Loess

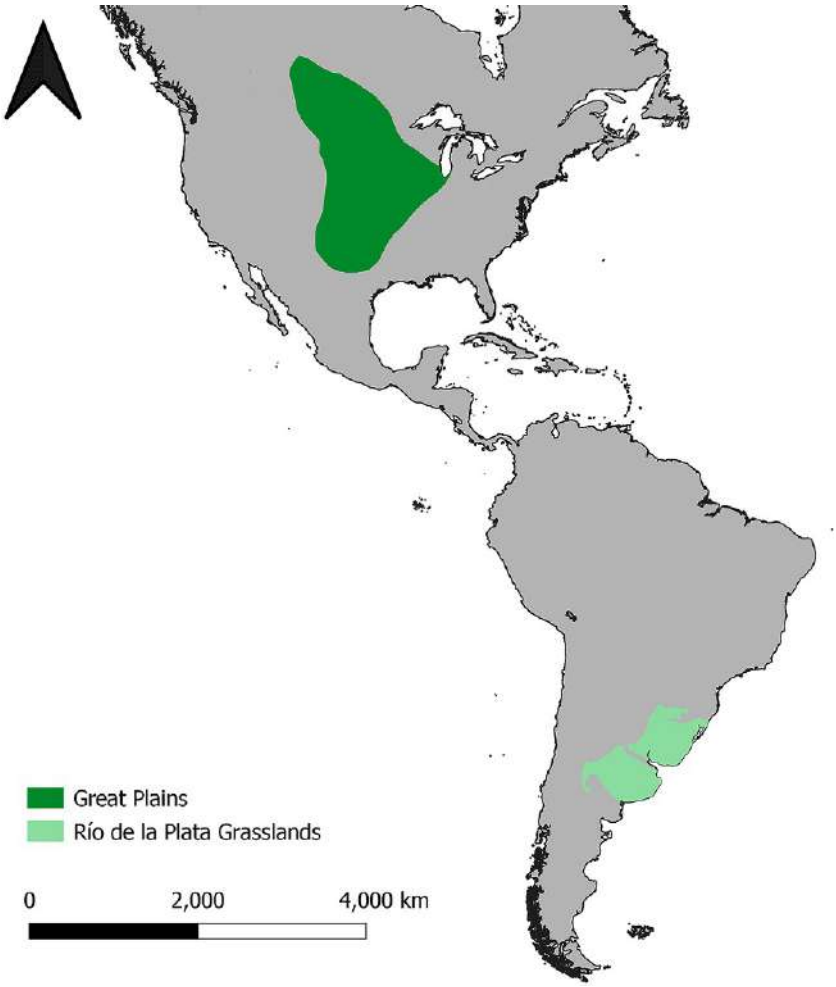

FIG. 13.1 Map of the Americas highlighting the grassland regions discussed in the text. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Modified from Anderson, R.C., 2006. Evolution and origin of the Central Grassland of North America: climate, fire, and mammalian grazers. J. Torrey Bot. Soc. 133, 626–647; Locklear, J.H., 2017. Endemic plants of the central grassland of North America: distribution, ecology, and conservation status. J. Bot. Res. Inst. Texas, 11, 193–234; Andrade, B.O., Marchesi, E., Burkart, S., Setubal, R.B., Lezama, F., Perelman, S., et al., 2018. Vascular plant species richness and distribution in the Río de la Plata grasslands. Bot. J. Linn. Soc. 188, 250–256.)*

Hills (where soils are made up of fine, wind-blown silt) in western Iowa are more similar to mixed grass communities to the west than to the surrounding tallgrass communities typical of the region (Novacek et al., 1985).

2.3 South America

In South America, the watershed of the Río de la Plata supports large areas of grassland, most notably the subtropical Campos grasslands of Brazil, Uruguay, and Argentina and the temperate Pampas grasslands of Argentina to the south (Fig. 13.1; Di Giacomo and Krapovickas, 2005; Overbeck et al., 2007; Modernel et al., 2016; Jaurena et al., 2021). The Río de la Plata grasslands cover over 70 million ha (Soriano et al., 1992; Baeza and Paruelo, 2018). Grasses form the dominant vegetation type throughout the Pampas and Campos, with forests occurring in isolated areas and riparian habitats; the Campos region receives more rainfall than the Pampas and has a warmer climate overall (Di Giacomo and Krapovickas, 2005; Bocalon et al., 2023). Average precipitation in the region follows a southwest to northeast gradient ranging from 700 to 1600 mm (27″–63″) per year; temperatures follow the same gradient ranging from 14°C to 22°C (57°F–71.6°F) (Modernel et al., 2016).

3 History

3.1 North America

Large areas of the Great Plains grasslands have been converted to row crop fields—a highly altered ecosystem compared to the grasslands they replaced—with hectares of a single agricultural plant species bordered by marginal field-edge habitats such as fencerows and shelterbelts (Best et al., 2001; Tack et al., 2017). In the 19th and early 20th centuries, Euro-American settlers on the Great Plains exterminated almost all wild herds of the American bison (*Bison bison*) and transformed large areas of mixed grass and short grass prairie into cattle (*Bos taurus*) pastures; outside of pastures, many grassland ecosystems lost grazing

as a disturbance regime, allowing ecological succession into shrub and forest habitats (Anderson, 2006; Engle et al., 2008; Derner et al., 2009; Allred et al., 2011a; Kennedy and Burbach, 2020). Regions unsuitable for row-crop agriculture, such as the Flint Hills of Kansas and the Sandhills of Nebraska, and dominated by cattle grazing provide remnant unplowed tallgrass prairie habitats (Wang et al., 2013). Agricultural policies and land use have driven the conversion of native habitats to row crop agriculture in the Great Plains throughout the last century, leading to a loss of about 70% of native habitats region-wide (Samson et al., 2004). Farming on fertile prairie soils requires tillage and other ground disturbances that leave soils bare such that atmospheric events like wind and rain lead to soil erosion. It takes approximately 1000 years for 25.4 mm (1″) of soil to accumulate but a very short time for it to erode (Arsenault, 2014). The Dust Bowl that impacted the Central United States from 1931 to 1939 was caused by poor agricultural practices and unusually high temperatures; extended drought resulted in erosion losses of 480 tons of soil per acre (Benjamin, 2004). Modern farming practices such as no-till planting, terracing, rotating crops, and planting winter cover crops have reduced soil loss in the Great Plains (Borowski, 2021).

3.2 South America

Humans have altered grasslands to access the goods provided by nature at the expense of biodiversity and ecosystem function (Baeza and Paruelo, 2020; Andrade et al., 2023). Following European colonization, much of the Río de la Plata grasslands became pasture for cattle; however, these pastureland typically retained native vegetation community and supported high levels of biodiversity (Modernel et al., 2016). In recent decades, many native pastures have been converted to row crop agriculture, with soybeans (*Glycine* max) replacing many graminoid crops and rice (*Oryza sativa*) cultivation spreading broadly (de Rito et al., 2022; Ríos et al., 2022) in a process that parallels habitat conversion in the North American Great Plains (Baldi et al., 2006; Modernel et al., 2016; Baeza and Paruelo, 2020; de Freitas et al., 2020). Baeza and Paruelo (2020) investigated the rates of change in land use/land cover (LULC) in the Río de la Plata grasslands over the first decade and a half of the 21st century, documenting a 23% increase in farmed area. Overbeck et al. (2007) found a similar loss (25%) of grassland area in the South Brazilian Campos, also due to land-use changes. Baeza and Paruelo (2018) modeled remotely sensed data to calculate Human Appropriation of Net Primary Production (HANPP), which quantifies the amount of biomass used by humans, revealing a decrease in energy available for other organisms in an ecosystem. Their results highlight a greater human impact on energy flow in the Río de la Plata grasslands than in other ecosystems around the world.

4 Biodiversity

Grasslands constitute nearly 40% of Earth's terrestrial area and contain diverse assemblages of biological organisms such that their species diversity may rival that of tropical rainforests, driving investigations into the coexistence of so many species in such similar niches (Petermann and Buzhdygan, 2021). Grasses make up the greatest biomass of grassland vegetation, while forbs (or flowering non-grasses) make up the greatest diversity of plant species (Turner and Knapp, 1996). High plant diversity positively impacts diversity at higher trophic levels. Predators, herbivores, omnivores, and detritivores also contribute to healthy ecosystem diversity. Invasive species may contribute to increased local species richness; however they can have detrimental effects on native plant species and community structure (Petermann and Buzhdygan, 2021). Grazing and fire are used by land managers to reduce shrub cover, control aggressive invasive plant species, and curtail forest expansion (Overbeck et al., 2007, 2015), thus increasing biodiversity in grasslands.

4.1 North America

Approximately 12,000 years ago, the last glaciers retreated from Canada and the northern United States, resulting in climate shifts and the evolution of present-day fauna and flora, such as development of mixed C_3 and C_4 grasslands in the north and drought-tolerant C_4 grasses that dominate the south (Augustine et al., 2021). This evolution resulted in large herds of migratory herbivores such as the iconic American bison (Fig. 13.2A), elk (*Cervus canadensis*), deer (Family: Cervidae), and pronghorn (*Antilocapra americana*) that created heterogeneous landscapes that land managers today try to mimic with livestock; in some fragmented areas, limited numbers of bison have been restored (Augustine et al., 2021). Many grassland birds that are endemic to the Great Plains occurred there 26,000 years ago, including upland sandpipers (*Bartramia longicauda*), lark buntings (*Calamospiza melanocorys*), dickcissels (*Spiza americana*), and western meadowlarks (*Sturnella neglecta*; Augustine et al., 2021).

Grassland birds are uniquely adapted to relatively dry landscapes and low-profile, herbaceous vegetation with well-camouflaged plumages, nesting on the ground, and eating seeds (Johnsgard, 2011). Lekking birds, those species in which the males attract females with impressive plumage and elaborate courtship displays, include the sharp-tailed grouse (*Tympanuchus phasianellus*) and the greater prairie-chicken (*Tympanuchus cupido*; Fig. 13.2B). The sharp-tailed grouse's mating display also includes rapid stamping of the feet, rattling of the tail feathers, and squawking sounds as it inflates and deflates an air sac on its neck (Helzer, 2022). Smaller-bodied bird species such as songbirds and shorebirds

FIG. 13.2 Representative species from the American Grasslands. (A) American bison (*Bison bison*); (B) greater prairie-chicken (*Tympanuchus cupido*); (C) big brown bat (*Eptesicus fuscus*), (D) black-footed ferret (*M. nigripes*), (E) blowout penstemon (*Penstemon haydenii*); and (F) Pampas fox (*Lycalopex gymnocercus*). *((A) Photo: Chauncey Kellar/United States; (B) Photo: Ron Knight/United Kingdom; (C) Photo: Dayna Berry/Univ. of Nebraska at Omaha; (D) Photo: Elisa Dahlberg/USFWS; (E) Photo: Mark Bellis/USFWS; (F) Photo: iNaturalist, pablo_gfs/Uruguay.)*

that build their nests on the ground concealed in grassland vegetation are a crucial part of grassland biodiversity. They connect trophic levels by consuming plant matter and arthropods (Whelan et al., 2008) and becoming prey for a diverse community of mammals and reptiles (Klug et al., 2009). The songs of grassland birds such as western meadowlarks and bobolinks (*Dolichonyx oryzivorus*) contribute to the soundscape of Great Plains ecosystems, helping create a unique sonic profile for the bioregion.

The diversity of mammals in the Great Plains may be surprising because, in addition to large grazing animals, it includes many species of rodents (order Rodentia) and bats (order Chiroptera; Fig. 13.2C), which are the second largest group of mammals in the region; bats are the only true flying mammals, eating insects at night in quantities up to 67% of their body weight (Bats: Order Chiroptera, 2017). While most grassland animals are adapted to the wide-open landscapes, bats prefer areas of high tree cover (Trubitt et al., 2019). The activities of European settlers such as shelterbelt plantings resulted in afforestation and the emergence of human structures such as bridges and buildings that benefit bats but may be at odds with conservation efforts focused on other grassland species (Coleman and Barclay, 2012; Trubitt et al., 2019). The critically endangered black-footed ferret (*Mustela nigripes*; Fig. 13.2D) was thought to be extinct, but over 300 individuals have been restored to their native habitats through various strategies including cloning (Bly, 2024). One strategy for successful release of black-footed ferrets is to rear them in large cages that are placed in colonies of prairie dogs, which are their primary prey source (Dobson and Lyles, 2000). Small mammals such as mice (Mus sp.) and shrews (Family: Soricidae) are also important members of prairie ecosystems; they alter soil structure with tunnels that provide habitats for other species, store or distribute seeds, and serve as a food source for many predators (Heisler et al., 2013).

Ecosystems are often defined by their vegetation. In the Great Plains, vascular plants total over 2900 species, including 730 genera in 159 families, all but 81 of which are flowering plants (Keeler, 2011). Asteraceae (sunflower family, 430 species) and Fabaceae (legume family, 178 species) are the largest forb families, while Poaceae (grass family, 260 species) is the largest graminoid family, but the composition of species varies among communities and over the course of the growing season, and plant species diversity can serve as a proxy for overall biodiversity in a community (Keeler, 2011; Roxanne Steele Kellar et al., 2015). Despite historical misconceptions about the paucity of endemic species in the Great Plains, the Central Grasslands (inclusive of all tall, mixed, and shortgrass prairies) of North America house 382 endemic plant species (an estimated 100 in the Great Plains alone; Keeler, 2011; Locklear, 2017). The critically imperiled blowout penstemon (*Penstemon haydenii*; Fig. 13.2E) is the rarest and one of the most unique angiosperms in the Great Plains, as it grows only in the Sandhills region of Nebraska and the Great Divide Basin in Wyoming in landscape features called blowouts where sands often shift (Stubbendieck et al., 1993).

4.2 South America

The Río de la Plata grasslands of Argentina, Brazil, and Uruguay contain both subtropical and temperate climate zones; therefore, they also contain high species diversity. Common mammals include the Andes skunk (*Conepatus chinga*), the flagship Pampas fox (*Lycalopex gymnocercus*; Fig. 13.2F), plains viscacha (*Lagostomus maximus*), South American cougar (*Puma concolor concolor*), and guanacos (*Lama guanicoe*), which are close relatives of the llama (Schipper, 2023), while common birds include the double-collared seedeater (*Sporophila caerulescens*), the great Pampa finch (*Embernagra platensis*), the grassland yellow finch (*Sicalis luteola*), the long-tailed reed finch (*Donacospiza albifrons*), and the greater rhea (*Rhea americana*; Claire, 2002). Andrade et al. (2018) compiled a list of 4864 vascular plant species for the Río de la Plata grasslands, and presented them in a Venn diagram highlighting the species that are shared among Argentina, Brazil, and Uruguay. Approximately 8% of these plant species are endemic to this region, and similar to the North American Great Plains, the three plant families with the greatest numbers of species are Asteraceae (659 spp.), Poaceae (645 spp.), and Fabaceae (399 spp.; Andrade et al., 2018).

The Río de la Plata grasslands of Brazil cover an estimated 13.7 million ha (Overbeck et al., 2007) and contain very high levels of biodiversity—one square meter of native area may contain more than 50 species (de Freitas et al., 2020). Andrade et al. (2023) compiled lists of plant, animal, bacteria, and fungal species along with their geographic ranges, primary habitats, and conservation status. They listed a total of 12,503 species, which included as many as 692 operational taxonomic units (OTUs) of soil microbes identified from five sampling sites. Their list also included 5358 animals, of which Aves was the largest vertebrate group with 567 species, and vertebrates had the greatest number of Critically Endangered, Endangered, or Vulnerable species according to three lists of endangered species (Andrade et al., 2023). Pampas cat (*Leopardus colocola*) is an example of a species that is endemic to this region, but with little ecological, life history, and distribution data available, its conservation status is unclear, and its habitat is unprotected or fragmented (Tirelli et al., 2021).

Most studies of biodiversity in the Río de la Plata grasslands focus on the subtropical Campos of Brazil, but in 2018, Florencia Grattarola initiated the Uruguayan Biodiversity Data Consortium and developed an online database of the biodiversity of Uruguay called *Biodiversidata* (https://biodiversidata.org/). The first phase of the project focuses on vertebrate diversity, and the second phase focuses on plants (Grattarola et al., 2020). The database contains species records for 1648 vascular plants and 673 tetrapods (Grattarola et al., 2019, 2020).

5 Conservation areas and their ecological indicators

5.1 Conservation areas

Healthy grassland ecosystems are important not only for supporting diverse wildlife, microbes, and plants but also important for providing ecosystem services, such as nutrient cycling, water filtration, erosion reduction, flooding restriction, and carbon sequestration, as well as providing sites of recreational, cultural, and spiritual activities (MDNR, 2024). However, given the vegetative history and fertile soil composition, the area of native prairie in North America has declined by over 99% since European settlement (Samson and Knopf, 1994) due to anthropogenic disturbance, primarily agriculture (Rutledge et al., 2023). Less than 3% of North America's native grasslands are protected (Commission for Environmental Cooperation, 2011–2012). Similarly, in South America, the area of the Río de la Plata grasslands has declined by 2.4 million ha in the last 20 years (Baeza et al., 2022), while less than 2% of the biome has some form of conservation protection (Modernel et al., 2016). Global climate change is also impacting grasslands. Droughts and increasing temperatures have led to the loss of wetlands (called prairie potholes), the spread of invasive species, an increased risk of severe wildfires, survival of pathogens and insect pests that previously could not thrive in these areas, and reduced crop yields and livestock production (Conservation in a Changing Climate, 2021). These dangers, together with

the scarce fraction that is conserved, make prairies the "most threatened and least protected biome" (Scholtz and Twidwell, 2022, p. 1).

5.1.1 North America

Created in 1985 in the United States, the Conservation Reserve Program (CRP) provides economic incentives for landowners to plant agricultural fields with native and non-native grasses and forbs, with the goal of promoting erosion control on marginal lands and benefiting wildlife (Johnson and Schwartz, 1993). Conversion of lands used for production into CRP grasslands has increased the availability of grassland habitat region-wide; however, the amount of land enrolled in the CRP is affected by commodity prices for crops such as corn (*Zea mays*) and soy (Drummond et al., 2012).

5.1.2 South America

In South America, only 1% of grasslands are on public conservation land, underlining the importance of conservation on private lands (Azpiroz et al., 2012). Efforts such as the Alliance for the Grasslands aim to certify beef produced on natural grasslands that also serves as wildlife habitat (Azpiroz et al., 2012). Uruguay created the Responsible Rural Production Program to provide a financial incentive to ranchers managing pastures in ways that preserve habitat features used by vulnerable wildlife, while Argentina and Brazil have programs that promote wildlife-friendly grazing practices (Azpiroz et al., 2012).

6 Ecological indicators

Indicators of the ecological condition of an area or its inhabitants are defined in various ways and include a range of metrics spanning abiotic and biotic components (Czúcz et al., 2021); these indicators are necessary for effective communication among scientists, the public, stakeholders, and policymakers. Ecological indicators are measurable features of an area that serve as surrogates for the exhaustive documentation of every organism or abiotic measurement in a community or ecosystem, which would be both time consuming and cost prohibitive. However, as these characteristics are vast in nature, results based on differing foci can be ambiguous (Heink and Kowarik, 2010). Even the varying definitions of the term "indicator" can be mutually exclusive (Heink and Kowarik, 2010). Nevertheless, ecological indicators are used for a multitude of reasons, from assessing current environmental conditions to predicting future changes in those conditions (Dale and Beyeler, 2001; Niemi and McDonald, 2004; Gerlach et al., 2013). In American grasslands, birds, plants, and insects are most often used as ecological indicators, while mammals and soil organisms are used less frequently. The indicator species is primarily selected based on the research or management questions asked (Gerlach et al., 2013), and ultimately a picture of ecosystem health must be developed for effective use and management of conservation areas (Schiller et al., 2001).

Ecosystems are complex, and ecological indicators target useful aspects of these ecosystems by measuring, summarizing, or isolating key pieces of information that can be easily understood while characterizing the larger system (Bertollo, 1998; Schiller et al., 2001; Niemi and McDonald, 2004; Lindenmayer et al., 2015; Hudson et al., 2017). These characteristics include patterns of diversity and structure among genes, species, populations, and landscapes, as well as the functional contributions of those components (Niemi and McDonald, 2004; Heink and Kowarik, 2010; and see Heink and Kowarik 2010, Table 1 for definitions and features of the term "indicator" that reveal the varying uses of the word in ecology and environmental planning). For example, in Canadian grasslands, Houston and Schmutz (1999) found that the western kingbird (*Tyrannus verticalis*) indicated tree growth in the early 20th century, while the southern dogface butterfly (*Zerene cesonia*) may be used to identify functional remnant patches in prairies (Fenner et al., 2018). However, since individual indicators are not equivalent in form and content, they limit the ability to draw broad conclusions about ecosystem health; thus, multiple indicators are often combined to address larger questions such as the condition of ecosystems and the extent of ecosystem services (Schiller et al., 2001; Czúcz et al., 2021). Because they exemplify important components of functional ecosystems, the diversity and abundance of a conglomerate of terrestrial arthropods, including ants, long-horned grasshoppers, plant hoppers, picture-wing flies, and crab spiders, were measured to indicate biotic integrity of tallgrass prairie restoration progress in Iowa, United States (Orlofske et al., 2010). Andrade et al. (2018) assessed the species richness and distribution of vascular plants to support conservation planning in the Río de la Plata grasslands of South Brazil. Because of the broad range of ecological indicator definitions, applications, and interpretations, it is imperative to communicate the objectives of their use and recognize the scale at which they are measured (Niemi and McDonald, 2004).

Ecological indicators are used to assess the condition, quality, or conservation value of an ecosystem, monitor conservation, management, restoration practices, assess ecosystem services, identify changes or problems over time, and potentially identify remediation strategies or conservation actions (Dale and Beyeler, 2001; Niemi and McDonald, 2004; Rákosya and Schmitt, 2011; Gerlach et al., 2013; Wills and Landis, 2018; Wilsey et al., 2019; Czúcz et al., 2021). In addition, ecological indicators can be used to characterize existing or potential ecosystem services that can inform managers and assist in communicating

conservation value to the public (Peters et al., 2016; Wills and Landis, 2018; Antonella et al., 2020). Edaphic arthropods play an important role in temperate grasslands and can be good indicators of ecosystem changes due to varying land management practices (Solascasas et al., 2022). Other studies evaluate changes over time (Hope et al., 2017) to identify trends or predict potential changes to the environment in the future (Niemi and McDonald, 2004). Many studies focus on the biodiversity of one taxonomic group in order to indicate the levels of biodiversity in a site, taxa, or communities (Niemi and McDonald, 2004; Steele and Pires, 2011; Gerlach et al., 2013; Aust et al., 2015; Roxanne Steele Kellar et al., 2015; Ahrendsen et al., 2016; Peters et al., 2016). Although many studies focus on broader questions about ecosystem health, such as soil function (Teague and Dowhower, 2022) and maximizing the conservation value of grasslands (Wills and Landis, 2018), others focus on more specific questions, such as forage quality (Ranglack and du Toit, 2015) or identification of prairie remnant patches (Fenner et al., 2018). A number of grassland species provide the basis for both qualitative and quantitative ecological indicators that are used to communicate the properties of ecosystem condition.

Indicator species are used to describe the diversity, condition, or distribution of other species with similar ecological requirements or to measure ecosystem status, environmental changes, specific ecological factors, or stress responses that may pertain to other species (Ricketts et al., 1999; Dale and Beyeler, 2001; Niemi and McDonald, 2004; Gerlach et al., 2013). While using species as ecological indicators can be both cost-effective and informative, Gerlach et al. (2013) warn that species richness metrics are more useful at larger (continental) spatial scales, and the selected taxa must reflect the particular ecosystem of interest. Nonetheless, a large number of studies have identified species that can be useful ecological indicators in American grasslands.

6.1 Birds

Single bird species and groupings of species have been used as indicators of a wide range of ecosystem characteristics, ranging from overall health to the status of other species. Established in 1966, the North American Breeding Bird Survey (BBS) is a collaboration between the Canadian Wildlife Service (part of Environment Canada) and the Eastern Ecological Science Center (part of the US Geological Survey; USGS, 2023). As part of the BBS, volunteers and other dedicated data collectors located across the North American continent gather population data on over 400 bird species (USGS, 2023). These data are used to provide a quantitative foundation for bird status reports, multispecies indicators, and ecosystem health not only in North American prairies but also across the North American continent, motivating conservation actions (Hudson et al., 2017). The array of BBS contributions includes but is not limited to: (1) the short-term trend metric for 60%–66% of breeding bird species in the prairies of Alberta, Saskatchewan, and Manitoba, Canada, (2) the synthesis of land cover features and BBS data that led to distribution models for the Sprague's pipit (*Anthus spragueii*), resulting in maps used to inform the acquisition of prairie pothole lands for waterfowl conservation, (3) the integration of BBS data and environmental data to reveal how environmental factors may impact breeding area survival and productivity, and (4) the provision of BBS population trend data for 347 bird species to the US Environmental Protection Agency's Report on the Environment, which describes the health of the US environment (Hudson et al., 2017). Various studies have identified grassland bird species that can serve as indicators of American prairie ecosystem health. For example, Wells and Rosenberg (1999) identified upland sandpipers as indicators of diverse and intact grassland communities in the Northeast. Browder et al. (2002) studied prairie pothole habitat types in North Dakota, USA, and formulated a metric for assessing the biological integrity of grasslands using assemblages of breeding birds. Their study focused on both the presence of grassland birds in high-quality habitats and the absence of bird species in areas with woody vegetation or human disturbance.

Studies involving bird species have also focused on their more specific roles as ecological indicators. The degree and range of the major threats to grassland birds of South America have been assessed by studying seedeaters because they are ubiquitous in grasslands and are sensitive to human disturbance (da Silva, 1999). Seed-eating finches (*Sporophila* sp.) migrate within and among major ecological regions in South America, and their seasonal movements help to identify areas that may be critical for preserving the populations of over 23 species (da Silva, 1999). The grasshopper sparrow (*Ammodramus savannarum*) is an obligate bird species found in North American prairies with medium height vegetation and moderately deep litter (Shaffer et al., 2021). In a study aimed at identifying a single species that could serve as an indicator of the effectiveness of management strategies, Elliott and Johnson (2018) investigated densities of grasshopper sparrows and six other grassland bird species that rely on upland tallgrass prairie. They found that the grasshopper sparrow was the strongest predictor of species density for the other six species studied in southwestern Minnesota, United States. However, they also concluded that because vegetation structure strongly influences grassland bird behavior, it would be better to utilize multiple, complementary indicator species to assess management and monitoring practices in this ecosystem (Elliott and Johnson, 2018).

6.2 Insects

Arthropods can be indicators of ecosystem processes and function (Orlofske et al., 2010). Gerlach et al. (2013) provide a general overview of many different invertebrates and their potentials and limitations as ecological indicators. Their review revealed that Orthoptera (e.g., grasshoppers and crickets) can be particularly useful in grasslands due to their high sensitivity to changes in grassland condition. In addition to orthopterans, ants and butterflies have been suggested as good indicators (Gerlach et al., 2013). This conclusion is supported in a review by Wills and Landis (2018) in which ants are discussed as ecosystem engineers in North American grasslands; therefore, impacting prairie productivity may impact biodiversity of soil microbes, plants, and other invertebrates. Because of this role, ants can serve as important indicator species of land management practices and conservation efforts (Wills and Landis, 2018). Several studies have identified butterflies as ecological indicator groups, including one by Fenner et al. (2018) in which sulfur butterflies (Family: Pieridae), among the most abundant butterflies in North American prairies, may be good ecological indicators in these endangered grasslands. Wallner et al. (2013) focused on Hemiptera, suborder Auchenorrhyncha, which includes treehoppers, planthoppers, leafhoppers, and spittlebugs, to develop a North American tallgrass prairie quality index called the Auchenorrhyncha quality index (AQI). Species abundance and richness values were combined with six life history traits to develop the AQI, which was then tested using a vacuum apparatus or sweep nets at 10 sites; ultimately the authors concluded that the AQI would best be used in conjunction with additional metrics gathered from diverse organisms (Wallner et al., 2013).

Comprising over 80% of the soil-dwelling fauna, edaphic arthropods play important roles in temperate grassland ecosystems and have been investigated for their potential use as ecological indicators (Solascasas et al., 2022). Although the review by Solascasas et al. (2022) identified more studies focused on macroarthropods (specifically, beetles, spiders, and ants) than on microarthropods, few studies focused on grasshoppers as grassland indicators, which is contrary to the review by Gerlach et al. (2013). They also concluded that species richness and abundance indices based on different arthropods varied with different management strategies (Solascasas et al., 2022). Many of the studies that are focused on arthropods suggest there is great potential for the identification of additional, perhaps better, ecological indicators among the multitude of insects yet to be investigated (e.g., Solascasas et al., 2022).

6.3 Plants

Plant characteristics and metrics such as plant species richness, diversity, distribution, and abundance have long been used to assess ecosystem diversity, function, and quality. The Floristic Quality Index (FQI) is an ecological indicator used by conservation organizations, land managers, and government agencies to quantify plant community health (Swink and Wilhelm, 1979). FQI combines plant species richness with the mean coefficient of conservatism, which is a value assigned to a plant species based on its tolerance to anthropogenic disturbance (Taft et al., 1997; Spyreas, 2019). Higgins et al. (2001) used the FQI to assess the quality of 63 tallgrass prairie remnants that had differing land management practices in eastern South Dakota, United States. In a study designed to compare metrics that could quantify the impacts of conservation, restoration, and reconstruction efforts in tallgrass prairies, Taft et al. (2006) calculated classic diversity metrics and vegetation integrity indices, including FQI. The results revealed that FQI was one of the most effective metrics for distinguishing qualitative differences in floristic integrity among plant communities in north-central Illinois, USA, though some caveats were discussed. (Taft et al., 2006). Spyreas (2019) reviewed the FQI, noting that this metric has continued to be popular for assessing conservation value and ecological integrity, despite a scarcity of studies on its philosophical basis and proper application. He suggests that more regional studies incorporating large datasets will allow users to better interpret the meaning of FQI scores (Spyreas, 2019). Peters et al. (2016) combined Simpson's Diversity Index (SDI; Simpson, 1949) with ant and plant species richness to indicate species richness and abundance of bees, beetles, butterflies, and birds in constructed grasslands in southwest Ohio, United States. They found that plant SDI was the best predictor of bee and beetle abundance, particularly for bees that provide ecosystem services (Peters et al., 2016).

Multiple studies in South America have been conducted to assess plant distribution patterns and classify vegetation groups. In the South Brazilian grasslands, Andrade et al. (2019) identified indicator species associated with each grassland assemblage (Dufrêne and Legendre, 1997). In the pampa biome, where exotic species had the greatest abundance, Bahia grass (*Paspalum notatum*) was identified as the most important species and clearly distinguished this biome from other grasslands (Andrade et al., 2019). Contributing to the knowledge needed for the design of conservation plans, Bueno et al. (2017) investigated the floristic components that define the different ecoregions in Paraguay and identified the environmental forces driving the patterns of species distribution. They performed dendrogram analyses using tree species to characterize vegetation types and conducted indicator species analysis to identify species with a greater affinity for certain vegetation types. The results of a cluster analysis revealed floristic variation among eight phytogeographic regions, including the Pampa for which *Salix humboldtiana* (Humboldt's willow) showed the greatest

indicator value (Bueno et al., 2017). The nonmetric multidimensional scaling (NMDS) analysis showed that aridity was the most significant factor delineating the Pampa where drought occurs at least a few months every year (Bueno et al., 2017).

6.4 Mammals

Mammals are not typically used as ecological indicators in American grasslands like they are in the tundra biome in Alaska, North America, for example, where small mammals with fast generation times serve as indicators of the change in biodiversity over time (Hope et al., 2017). One exception is the use of American bison as indicators of forage quality (Ranglack and du Toit, 2015). Ranglack and du Toit (2015) monitored free-ranging bison following mechanical or prescribed burn management to combat woody encroachment (e.g., *Juniperus osteosperma*, Utah juniper, and *Pinus edulis*, piñon pine) in prairies in southern Utah. The results revealed that burned habitats provided greater foraging opportunities than either mechanically manipulated or undisturbed habitats. Focusing on the Pampas region of Argentina, Antonella et al. (2020) conducted a review of scientific articles reporting on bird and mammal ecological functions and their negative interactions with humans and human activities. Results of their investigation may lead to conservation strategies in which humans and wildlife may best coexist (Antonella et al., 2020). Studies that focus on defining and characterizing the nuances of varying ecosystems will continue to aid conservation planners, land managers, and policymakers.

7 Human activities and their impacts

Human activities that influence biotic communities have impacts ranging from outright habitat loss to the less-obvious effects of human presence, fragmentation, or structures in the landscape on particularly sensitive species (Samson and Knopf, 1996). While not all human activities result in negative effects on wildlife, many of the key drivers of change in American grasslands are anthropogenic, making them key to conservation planning and implementation (Samson et al., 2004). The wildlife communities found today in American grasslands are the result of both evolutionary history and more recent anthropogenic effects, such as the introduction of nonnative species and changes to historic disturbance regimes (Engle et al., 2008). This section focuses on the most severe negative impacts of human activity on wildlife, while human activities aimed at conservation and management are covered in the following section.

7.1 Habitat loss

The most severe habitat losses in the North American Great Plains have occurred in the tallgrass prairie, reaching over 99% loss in some areas (Samson and Knopf, 1996). Greater proportions of native prairie remain to the west, with about 30% of mixed grass prairie and 50% of shortgrass prairie remaining (Samson et al., 2004); however, some areas have seen almost total loss of prairie habitat, such as in Texas, where only 20% of native shortgrass habitat persists (Samson and Knopf, 1996). Loss of breeding habitat has contributed to steep declines in some species. Grassland birds have shown the greatest magnitude of loss out of all bird groups in North America, with 74% of species in decline, primarily due to habitat loss (Rosenberg et al., 2019).

South American grasslands have seen steep losses in recent decades due to the expansion of row-crop agriculture; for example, between 2000 and 2010, the acreage of soybeans more than doubled in the Pampas and Campos ecoregions of Brazil, Uruguay, and Argentina (Modernel et al., 2016). Less than 1% of Pampas and Campos habitat is conserved as protected areas, meaning that the persistence of native grasslands is tied to the ranching industry, and that habitat quality of those grasslands is largely dependent on ranching practices (Krapovickas and Di Giacomo, 1998).

7.2 Habitat conversion

While some animals such as a few mammals (Beatty et al., 2014) and generalist bird species (Baldi et al., 2006) may benefit from anthropogenic habitats such as row crop agricultural fields, these fields have only remnant native plant communities on field margins and offer few resources, particularly breeding habitat, for many grassland-obligate species (Samson and Knopf, 1996; Murphy, 2003). Pesticide use on agricultural lands has also harmed species such as insects and birds that persist in agricultural landscapes (Bartuszevige et al., 2002; Mineau and Whiteside, 2013; Rosenberg et al., 2019).

Some agricultural habitats such as pastures, hayfields, and crops such as alfalfa and grains provide altered or temporary grassland habitats of varying quality and temporal availability (Bollinger, 1995; Best et al., 2001; Samson et al., 2004; Drummond, 2007; Engle et al., 2008; Rahmig et al., 2009). Pastures are suitable as breeding habitat for some grassland obligate species such as the grasshopper sparrow in the Great Plains (Rahmig et al., 2009) and as wintering habitat for migratory shorebirds such as the buff-breasted sandpiper (*Calidris subruficollis*) in the Río de la Plata Grasslands (Lanctot et al., 2004). Lands used for agricultural purposes offer different resources and habitat availability throughout the year; for example, Great Plains hayfields are often harvested in May, at the start of the nesting season for many grassland birds, meaning that nesting sites are no longer available (Luscier and Thompson, 2009; Allen et al., 2019). Alfalfa (*Medicago sativa*) fields allowed to grow during the summer breeding season following a spring harvest can provide nesting habitat for dickcissels (Frawley and Best, 1991).

Incentives for producing corn for use as biofuel have reduced the amount of CRP acreage in North America (Wright and Wimberly, 2013). CRP lands can provide habitat for grassland-obligate species of birds and insects, and have ecosystem-wide benefits on water quality (Hartzler and Buhler, 2000; Ribic et al., 2009; Johnson et al., 2016; Quinlan et al., 2021; Pavlacky et al., 2022). Increased enrollment in the CRP can help mitigate the negative effects of landscape change on grassland biodiversity (Pavlacky et al., 2022). When CRP fields lack disturbance regimes such as fire or grazing or when they are too small for the habitat size requirements of some species, they tend to form dense grasslands with low diversity and their value as habitat is limited (Rahmig et al., 2009; Begosh et al., 2020). However, the density of grassland obligate species is higher on CRP lands than on unmanaged marginal grasslands such as field edges, grass-covered terraces, and waterways bordered by grass, which feature bird communities dominated by generalist species, suggesting that CRP lands offer benefits to grassland obligates compared to alternate land uses (Cox et al., 2014).

7.3 Invasive species

7.3.1 Graminoids

Introduced plant species have transformed American grassland ecosystems. In North America, exotic grasses such as brome (*Bromus* sp.) and Kentucky bluegrass (*Poa pratensis*), originally introduced for livestock pasture and erosion control, form dense stands and compete with native plant species for habitat (Grace et al., 2000; Azpiroz et al., 2012). In areas of South America historically dominated by tallgrass communities, cattle grazing can make ecosystems more vulnerable to invasive plants when native grass species are heavily grazed, opening up the canopy and allowing invasive plants to establish themselves (Krapovickas and Di Giacomo, 1998). Introduced *Cyperus* species of sedges have become invasive in parts of the South American grassland biome (Fonseca et al., 2013). Differences in biology between native and introduced grass species have enabled some introduced species to become invasive. Nonnative cool-season grasses begin their active growth period earlier in the spring and for longer in the fall than native warm-season grasses, letting them exploit times of the year for growth and reproduction when native grasses are dormant (Ellis-Felege et al., 2013). Turfgrasses limit the floristic and structural diversity of the vegetation community, thereby limiting opportunities for animals including grassland-nesting birds and insects dependent on native plant species for nectar, pollen, or habitats for larvae (Ellis-Felege et al., 2013).

7.3.2 Forbs

Some invasive forbs have the potential to form rank growths on disturbed soil and outcompete native plants for space and resources, such as bull thistle (*Cirsium vulgare*), sweet clover (*Melilotus* sp.), and Chinese bushclover (*Lespedeza cuneata*) in North America (Silliman and Maccarone, 2005; Cummings et al., 2007; Nemec et al., 2013). In South America, highly invasive forbs include *Kali tragus* (prickly Russian thistle) and *Diplotaxis tenuifolia* (perennial wall-rocket; Fonseca et al., 2013).

7.3.3 Woody species

While eastern red cedar (*Juniperus virginiana*) is native to North America, propagation of this species for windbreaks and other agricultural uses, along with fire suppression, has allowed the species to invade many grassland habitats (Gaskin et al., 2021). In South America, woody species such as *Acacia mearnsii* (black wattle), *Gleditsia triacanthos* (honey locust), and *Genista monspessulana* (French broom) have become invasive in Pampas grasslands (Fonseca et al., 2013).

7.4 Habitat fragmentation and landscape change

Remaining grassland habitats in the Americas are fragmented, meaning that they are smaller than their historical counterparts, divided by features such as roads and fencerows and bordered by anthropogenic habitats (Samson and Knopf, 1996). The effects of grassland fragmentation on biota depend on the biology of the particular organism. Smaller-bodied, shorter-lived species tend to have decreased species richness/abundance in fragments, but connectivity between fragments can benefit species richness (Debinski and Holt, 2000). Larger-bodied and highly mobile species have home ranges that may extend over many fragments, but reaching different parts of the home range can be dangerous when crossing obstacles such as roads, fencerows, and human-populated areas (Debinski and Holt, 2000; Gehring and Swihart, 2003). Coyotes (*Canis latrans*) showed a greater tolerance for the presence of crop fields in the landscape than smaller-bodied mammals (Gehring and Swihart, 2003). The effects of decreasing habitat size in grasslands can also depend on the trophic level of the species, with higher trophic levels more negatively affected by fragmentation (Stoner and Joern, 2004). Social structure can play a role as well (Debinski and Holt, 2000). Solitary, low-density species could have a harder time successfully finding mates in fragmented landscapes (Jager et al., 2006). The effects of habitat fragmentation and landscape change on different parts of the biotic community are discussed below.

7.4.1 Plants

While habitat loss has the greatest effect on plant species richness in the short term, fragmentation has a larger effect in the long term (Alofs et al., 2014). Woody encroachment in grassland remnants reduces the diversity of herbaceous

species, and fragmented habitats facilitate the further spread of woody species (Alofs et al., 2014; Scholtz et al., 2018). In pastures, targeted herbicide use can help control invasive, grazing-tolerant plant species and increase overall plant diversity (Smart et al., 2011); however, intense herbicide use can limit floristic diversity (Fuhlendorf et al., 2009).

7.4.2 Insects

Grassland disturbance regimes have strong indirect effects on the insect community through their effects on the plant community (Stoner and Joern, 2004). Widespread herbicide use in agricultural landscapes has harmed the insect community, with particularly severe effects on species such as monarch butterflies (*Danaus plexippus*) that rely on milkweed (*Asclepias* sp.) plants found on field margins and remnant grasslands (Pleasants and Oberhauser, 2013). Broad-spectrum pesticides that target broad-leafed forbs can reduce forb cover and decrease the diversity of the arthropod community in pastures (Fuhlendorf et al., 2009). Insecticides can also have harmful bottom-up trophic effects on grassland ecosystems through their impacts on insects; in recent decades, the use of neonicotinoid pesticides has increased, making it the most widely used agricultural pesticide globally (Goulson, 2013). Li et al. (2020) found that neonicotinoid use reduced grassland bird diversity. Neonicotinoids can affect birds when they eat crop plant matter and through their impacts on the insect community, a key food source for grassland birds (Li et al., 2020).

7.4.3 Birds

Some grassland-obligate birds have lower densities in smaller habitat fragments that are otherwise suitable in quality, a phenomenon known as area sensitivity (Herkert, 1994). Woody encroachment decreases the habitat quality of fragments for birds that breed in grasslands, as many grassland birds avoid areas with high shrub or tree cover. Over time, grasslands invaded by woody plants develop a bird community more similar to that of shrubland or forest habitats (Coppedge et al., 2001a).

Woody vegetation at the edges of remnant grasslands can have important effects on community dynamics. Wooded edges are linked to greater nest parasitism by brown-headed cowbirds (*Molothrus ater*; Patten et al., 2006), and increased nest predation by mesopredators that travel through woody habitats to seek out prey within grassland fragments (Winter et al., 2000). Species respond differently to the landscape at different scales (Johnson, 2001). Some species such as bobolinks and eastern meadowlarks respond to the presence of grassland habitat in the surrounding landscape, while other species are less affected by landscape variables (Ribic et al., 2009). Birds with larger home range sizes, such as lesser prairie-chickens (*Tympanuchus pallidicinctus*), are sensitive to landscape-level effects of fragmentation such as increased woody cover, reduced connectivity between habitats, and a higher percentage of cropland in the landscape (Fuhlendorf et al., 2002b).

7.4.4 Stream fish

Stream ecosystems in the Great Plains are at even higher risk than terrestrial ecosystems (Dodds et al., 2004). In the western Great Plains, extraction of water resources for agriculture has greatly decreased aquatic habitat availability, while in the eastern Great Plains, barriers to movement such as impoundments have led to habitat fragmentation and isolated populations (Perkin et al., 2015). The depletion of groundwater and its effects on surface water availability has led to broad community changes in many stream communities, with the populations of large-bodied fish species declining steeply and small-bodied species expanding in highly altered landscapes (Perkin et al., 2017). Remnant stream habitats are often highly altered, with linear channelized courses through cropland replacing meandering courses with many microenvironments; increased flow rates and high sediment loads; and diminished water quality due to fertilizers, herbicides, and pesticides from the surrounding landscape (Dodds et al., 2004).

7.4.5 Herpetofauna

The loss of ephemeral wetlands such as bison wallows and small wetlands drained for agriculture negatively affect amphibians (Scott, 1996). Most species of reptiles in the Great Plains remain widespread, but estimating population sizes and understanding responses to habitat variables has proved challenging, as they are cryptic and difficult to study (Scott, 1996; Wilgers and Horne, 2006). Fire has a strong effect on reptile species occurrence, with some species becoming more abundant in recently burned areas and others more abundant in unburned areas, resulting in community composition shifting with the frequency of burning (Wilgers and Horne, 2006). Amphibians are driven more by watershed dynamics than fire (Larson, 2014).

7.4.6 Mammals

Woody encroachment can cause species turnover among the small mammal community and may disrupt relationships between seed dispersers of grassland plant species (Hope et al., 2021). For some mesopredator mammal species (i.e., those that occupy mid-range trophic levels) such as the Virginia opossum (*Didelphis virginiana*), anthropogenic habitats such as agricultural fields and landscape features such as fencerows and roads have provided new resources and enabled range expansions (Beatty et al., 2014). Large predators such as wolves (*Canis lupus*) and mountain lions (*Puma concolor*) are absent from many areas in the Great

Plains due to hunting and persecution. Ecosystems without large carnivores had fewer hardwood trees associated with Great Plains ecosystems, such as cottonwoods (*Populus* sp.) and bur oaks (*Quercus macrocarpa*), due to increased browsing by deer (Ripple and Beschta, 2007).

7.5 Changes to disturbance regimes

In parts of the Great Plains dominated by ranching, grassland habitats of variable quality persist with cattle replacing bison as a disturbance regime (Wang et al., 2013). However, in areas where grazing pressure is high, bird diversity is lower (Toombs et al., 2010). Prescribed burns in the Flint Hills tend to be frequent, with the goal of encouraging the growth of grasses for cattle forage (With et al., 2008). Overall, bird populations in the Flint Hills are declining, suggesting land use with cattle grazing as the primary goal, without effective strategies to balance wildlife use of the land, is not enough to slow population declines (With et al., 2008).

Fire suppression has allowed fire-intolerant species such as eastern red cedar to invade grassland habitats, further degrading habitat quality (Coppedge et al., 2001a,b; Anderson, 2006; Engle et al., 2008; Alofs et al., 2014; Scholtz et al., 2018). Fragmentation and fire suppression may have a synergistic effect in promoting the spread of woody cover (Scholtz et al., 2018).

7.6 Urbanization

Historically, conversion to agriculture has affected a much larger area of the Great Plains than urbanization (Drummond et al., 2012). In South America, agricultural land use has been a much larger contributor to grassland fragmentation than urbanization (Baldi et al., 2006). Some species, such as habitat-sensitive insects, have lower abundances near urban landscape features (Davros et al., 2006). However, urban grasslands can still have considerable habitat value, as many species persist in grassland habitats within urbanizing landscapes (McLaughlin et al., 2014). However, anthropogenic structures have contributed to the fragmentation of aquatic ecosystems (Perkin et al., 2015).

7.7 Climatic and environmental changes

The effects of global climate change on Great Plains grassland ecosystems include increased aridity, earlier initiation of spring phenology, greater storm intensity, and higher average temperatures; this suite of changes could lead to northward range shifts as the southern areas of species' ranges become inhospitable (Peterson, 2003; Hufkens et al., 2016; Edwards et al., 2019). Some climatic effects differ among Great Plains regions; for example, predicted average wind speeds are higher in the southern Great Plains and lower in most of the northern and western Great Plains (Edwards et al., 2019). Overall grassland productivity may increase as climate change intensifies if a longer growing season mitigates the effects of greater aridity (Hufkens et al., 2016). Northward shifts in the distribution of some Great Plains species indicates that the historic boundaries of ecoregions are changing, creating a challenge for land management targeting a historical ecological community at a fixed location (Roberts et al., 2019). Within a species, subspecies and populations may respond differently to climate change, affecting intraspecific genetic diversity (Giudicelli et al., 2019).

In South American grasslands, climate change will likely lead to changes in plant communities, with higher average temperatures and decreased soil availability leading to C_4 grasses replacing C_3 species and leading to reduced cycling of nitrogen between the plant community and the soil (Epstein et al., 2002). Water availability will be lower and more variable throughout the year, with severe droughts and storms of greater intensity, all of which could pose challenges to the economic feasibility of raising cattle (Yahdjian and Sala, 2008). Many species ranges will shift, some with broader effects on ecosystem services; Maldaner et al. (2021) predicted that dung beetle (various families within the superfamily Scarabaeoidea) ranges in South American grasslands would become much smaller by 2050, meaning that many areas would lack dung beetles that help incorporate organic matter into the soil and reduce the abundance of parasites that affect cattle and other host species.

Global climate change interacts with more local effects such as landscape change, habitat fragmentation, and invasive species. Fragmented ecosystems such as grasslands have lower diversity in microclimates than larger, less fragmented ecosystems, resulting in a lower capacity to buffer the effects of climate change (Jarzyna et al., 2016). Grassland fragmentation can facilitate the spread of disease, as previously isolated populations living in separate habitats come into contact with each other more frequently (Nuñez et al., 2020). Warmer climates with increased aridity could benefit nonnative species of grasses in competition with native species due to greater tolerance for extreme environmental conditions (Duell et al., 2021).

Changes to land cover and resource utilization can have local effects on climate, and in some regions, these local effects could be more pronounced than those of global climate change, or have interacting effects (Sohl et al., 2012). Depletion of groundwater, such as the use of the High Plains Aquifer for agriculture, has had ecological consequences such as decreased surface water availability and changes in the assemblages of stream fish (Perkin et al., 2017). Erosion due to tillage will compound erosion impacts due to wind and water, which is predicted to be higher in some regions as climate change intensifies (Edwards et al., 2019).

Nutrient cycles have been greatly affected by Euro-American settlement and agriculture, with C and N soil concentrations reaching their lowest levels during the Dust Bowl period due to intensive cultivation and overgrazing without soil conservation practices (Wang et al., 2014). While increased attention to soil conservation and restoring soil nutrients has slowed the loss of C and N, higher average temperatures and aridity are linked to increased rates of nutrient loss from soils, particularly on cultivated land (Follett et al., 2012).

8 Operational and potential monitoring tools to conserve or recover the biome

Grasslands are one of the most threatened ecosystems in the world; therefore, it is imperative to utilize management as a method to maintain and conserve remaining grassland remnants and marginal grasslands as well as recover and restore degraded habitats (Cottam and Wilson, 1966; Tester and Marshall, 1962). The three major operational tools used for grassland management include grazing, fire, and mowing regimens (Erhardt, 1995; Feber et al., 1996; Pollak and Kan, 1998; Towne et al., 2005; Rudolph et al., 2006; Freese et al., 2007; Guyette et al., 2012; Kohl et al., 2013). Other operational tools used, to a lesser extent, include herbicidal spraying, mechanical removal, and seeding (Owensby et al., 1973; Wilson and Schmidt, 1990; Fuhlendorf et al., 2002a; Cummings et al., 2007; Grman et al., 2013; McMillan et al., 2022).

In order to evaluate the impacts of operational tools utilized within a habitat, it is necessary to conduct short-term and long-term monitoring of fauna and flora and adjust the frequency, intensity, and choice of tools as needed in order to meet management goals (Tester and Marshall, 1962; Bureau of Land Management, 1996; Lindenmayer and Likens, 2010). Both the type and frequency of monitoring tools needed is dependent on overall management goals. Wildlife surveillance can include but is not limited to surveys of birds (Rosenstock et al., 2002), herpetofauna (Ali et al., 2018), mammals (Silveira et al., 2003), and insects (Kuno, 1991). Vegetation surveys can include, but are not limited to, those of forb attributes (Bureau of Land Management, 1996), litter (Tester and Marshall, 1962), nectar (Geest and Baum, 2022), and cover class and canopy (Daubenmire, 1959). Invasive plant species monitoring is also needed to determine appropriate control strategies (Prior et al., 2018). This monitoring includes utilizing remote sensing and GIS (Joshi et al., 2004; Filippelli et al., 2020), community scientist data collection (Crall et al., 2011; Gallo and Waitt, 2011), or on-ground vegetative sampling (Grant et al., 2009).

8.1 Grazing as an operational tool

North American prairies evolved with dominant grazing ungulates, including the American bison (Anderson, 2006; Knapp et al., 1999); however, today, the dominant grazers are domestic cattle (Anderson, 2006; Allred et al., 2011b). While it is unclear how cattle grazing compares to historic bison grazing (Towne et al., 2005; Freese et al., 2007; Kohl et al., 2013), it is known that fencing reduces the movement of grazers and may increase grazing pressure in smaller habitats (Kohl et al., 2013). Additionally, grazing pressure can change depending on season of grazing, grazing intensity, and livestock stocking rates (Blüthgen et al., 2012; Manley et al., 1997), which can alter vegetation structure, canopy cover, and bare ground availability (Gardner et al., 1997; Vogel et al., 2007; Teague et al., 2004; Towne et al., 2005).

To a lesser extent, grazing and browsing species such as domestic sheep (*Ovis aries*) and goats (*Capra hircus*) are used for weed management in North American prairies (Frost and Launchbaugh, 2003; Schieltz and Rubenstein, 2016). Goats are particularly useful for removing woody species and plant species with allelochemical properties because they have large livers (Frost and Launchbaugh, 2003). In comparison, sheep are more selective in their grazing choices, but they can tolerate high-fiber diets that come mainly from forbs (Frost and Launchbaugh, 2003). Selecting a grazing species also depends on ensuring that the animal's nutritional needs can be met within a landscape (Frost and Launchbaugh, 2003).

Cattle and sheep grazing is a common grassland management technique in the South American Pampas and Campos, in countries like Brazil (Allen et al., 2011), Uruguay (Sollenberger et al., 2005), Bolivia (Bogino et al., 2015), and Argentina (McDaniel et al., 2005) among others. Studies in South America have found that higher herbage allowance, or the amount of vegetation cattle are allowed to consume in a specific area, is linked to higher species diversity (Carvalho et al., 2003; Altesor et al., 2006; Overbeck et al., 2006; Soca et al., 2008; Modernel et al., 2019). Fire may also be used in conjunction with cattle (McDaniel et al., 2005) or sheep (Bogino et al., 2015) grazing in South American grasslands.

8.2 Fire as an operational tool

Grasslands are adapted to fire, with fire being necessary to maintain native vegetation and prevent encroachment of woody species and nonnative fire-intolerant species (Pollak and Kan, 1998; Rudolph et al., 2006; Guyette et al., 2012). Prescribed fire or prescribed burning is utilized by land managers to mimic the natural effects of wildfire (Huntzinger, 2003). While the importance of fire in North American grasslands is well known, less is known about the use of fire in South America, especially in South Brazilian grasslands (Overbeck et al., 2018). However, indigenous peoples in Argentina (Bogino et al., 2015) and Bolivia (McDaniel et al., 2005) have long histories of using fire as a management method in grasslands.

8.2.1 Temporal

How often fire occurs (fire return interval), varies by biome but tends to be lower in grasslands (Dickmann and Cleland, 2002). However, the frequency of historically occurring wildfires and induced prescribed fires may not be the same in some areas (Swengel, 2001), with fire either occurring too frequently or too infrequently. Historically, wildfires occurred most often during hot and dry weather, however, presently burn bans are often in effect for prescribed fire during similar weather (Westerling et al., 2003; Wonkka et al., 2015). The timing of fire such as during the growing season or during the nongrowing season can change plant composition (Dickson et al., 2019). Choosing the frequency of prescribed burns is an important consideration; some wildlife and floral species are more fire-tolerant than others and occur in higher numbers in more frequently burned areas (Driscoll and Henderson, 2008; Moranz et al., 2012; Griffiths and Brook, 2014; Caven et al., 2017; DeSha et al., 2017; Dickson et al., 2019). The timing of fire is also an important consideration relative to flora and fauna life stages, as fires during the life stages of less mobile species will have more negative consequences than fires during the life stages of more mobile species (Swengel, 2001; Kral et al., 2018).

8.2.2 Spatial

Beyond temporal considerations of prescribed fire, the spatial configuration of fire is also important (Fuhlendorf and Engle, 2001). Fire can be detrimental to wildlife, particularly insects, if it occurs in small grassland fragments, due to the inability of surrounding populations to recolonize areas post fire (Swengel, 2001; Swengel and Swengel, 2007; Black et al., 2011). Maintaining heterogeneity in grasslands is imperative in order to support a community of species that may respond differently to different times and seasons since fire (Fuhlendorf and Engle, 2001; Augustine and Derner, 2015; Geest et al., 2023).

8.2.3 Habitat impacts

Fire can affect forb structure, height, composition, morphology, abundance, phenology, nectar quality, and nectar quantity (Bulan and Barrett, 1971; Haysom and Coulson, 1998; Pollak and Kan, 1998; Wrobleski and Kauffman, 2003; Collins and Calabrese, 2012; Moranz et al., 2014; Adamidis et al., 2019; Geest and Baum, 2022). Fire impacts grasses as well, with C_4 grasses responding more favorably to fire than C_3 grasses (Collins and Calabrese, 2012). Changes in the vegetation community can affect wildlife, particularly invertebrates (Haysom and Coulson, 1998), by encouraging host plant growth (Adamidis et al., 2019), providing more bare ground for host plants to grow (Baum and Sharber, 2012), providing more bare ground for nests (Brokaw et al., 2023), or affecting nectar quality and quantity for pollinators (Geest and Baum, 2022).

8.3 Mowing as an operational tool

In North America, mowing (or haying) is an additional management technique utilized to prevent woody species encroachment in grasslands and can be used by itself or in combination with fire and grazing (Erhardt, 1995; Feber et al., 1996; Swengel, 1996, 2001). Mowing frequently occurs on roadsides and other smaller marginal grasslands (Halbritter et al., 2015; Steidle et al., 2022) due to the ease of utilizing mowers in comparison to other management techniques. However, as mowing can be detrimental to invertebrates (Swengel, 2001; Steidle et al., 2022), partial mowing (Black et al., 2011) or infrequent mowing is recommended (Halbritter et al., 2015) to allow time for populations to rebound post mowing. Little is known on the frequency or effects of haying in South American grasslands.

8.4 Alternative operational tools

8.4.1 Herbicide

Grasslands have suffered from invasion by undesirable plant species (Owensby et al., 1973; Silliman and Maccarone, 2005; Cummings et al., 2007; Grant et al., 2009; Pierce and Reich, 2010). Controlling for these invasive species can involve herbicide treatments (Fuhlendorf et al., 2002a; Cummings et al., 2007; McMillan et al., 2022), as well as introducing species to act as biological control agents (Eddy et al., 2003).

8.4.2 Mechanical removal

Mechanical removal is often necessary when grasslands have high densities of woody species encroachment, such as eastern red cedar (Zhong et al., 2020). Mechanical removal quickly removes trees and opens the canopy (Morton et al., 2010). However, the machinery used to remove trees needs to be used carefully in order to prevent excessive soil erosion (Zhong et al., 2020).

8.4.3 Seeding

Seeding can increase biodiversity in a grassland and is frequently used to restore grassland habitat (Grman et al., 2013; Hansen and Gibson, 2014; Barak et al., 2022; Larson et al., 2022). The selection of seed mix chosen is often determined by a variety of factors outside of ecological choice including social and economic factors (Rowe, 2010). Beyond the type of seed mix chosen, timing of seeding and method of seeding are also important considerations when using seed mixes to restore grassland habitat (Larson et al., 2011).

9 Conclusions

Some species are considered conservation-dependent, meaning that in the absence of active habitat management, invasive species control, and/or reintroduction of

captive-bred individuals, the species would soon go extinct. In the Americas, grasslands are effectively a conservation-dependent ecosystem, primarily due to their reliance on relatively frequent (every 2–5 years) disturbance regimes such as fire or grazing to reset the ecosystem to an earlier phase of succession, as well as trigger growth and seed release in fire-dependent plant species. Allowing ecological succession to proceed unchecked (i.e., without active management through disturbance regimes) would likely result in the invasion of woody species and invasive herbaceous species into most grasslands, resulting in ecosystems more similar to biomes outside of the historic grasslands, such as the expansion of hardwood forests westward in the Great Plains. This would probably lead to the extinction of many species and greatly decrease abundance of others. Besides the intrinsic value of grassland biodiversity, many of the management actions required for grassland persistence have secondary benefits. A grassland maintained in the long term is a carbon sink, as mature grasslands have high levels of soil organic matter. Grass-dominated and forb-diverse plant communities can reduce soil erosion in marginal areas unsuitable for growing crops and in climates marked by frequent drought and occasional severe storms that lead to runoff. At the landscape level, managing grassland habitats contributes to local habitat diversity, providing benefits to species with large home ranges that inhabit multiple habitat types.

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Section E

Shrublands

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Chapter 14

Biodiversity and ecological dynamics of the Fynbos biome in South Africa

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1 Introduction

Fynbos is a highly diverse biome known for its rich biodiversity, with approximately 6000 plant species that are endemic to the region. This biome is globally recognized for its remarkable plant species diversity and higher levels of endemism, and it coinc ides with the Cape Floristic Region (CFR) (Cox, 2001). This unique ecosystem is endemic to South Africa (Rebelo et al., 2006), specifically found in the Western Cape and Eastern Cape Provinces (Fig. 14.1A and B). Spanning about 6% of southern Africa's land area, it serves as a vital habitat for over 50% of the region's species and constitutes a significant portion (80%) of the widespread Cape floral kingdom (Cowling, 1992; Linder, 2003). The Fynbos biome experiences a Mediterranean climate characterized by an extended dry season, occasionally interrupted by short winter rainfall. This climatic condition aligns it with other Mediterranean vegetation biomes found across the globe, including North American chaparral, Chilean matorral, Mediterranean maquis, and Australian mallee (Ackerly et al., 2014; Finch and Meadows, 2018). Fynbos is divided into two distinct subecoregions: montane Fynbos and renosterveld, and lowland Fynbos and renosterveld. Vegetation in Fynbos is primarily characterized by dense shrubland consisting of evergreen sclerophyllous plants (Fig. 14.2), particularly those belonging to the Proteaceae family, which constitutes the majority of plant species in this biome (Midgley and Rebelo, 2008). This biome is associated with nutrient-poor soils, strong winds, and short winter rainfall, and predominantly contains plants that have adapted to frequent fires, such as shallow-rooted proteoids, heath-like ericoids, reed-like restioids, and bulbous geophytes (Cowling and Richardson, 1995; Moran and Hoffmann, 2012).

Fynbos holds great importance within South Africa and globally (Finch and Meadows, 2018). It stands as one of the world's biodiversity hotspots, nurturing a diverse range of vegetation types such as heathlands, shrublands, and grasslands, each housing its unique assortment of plant species. Moreover, Fynbos serves as a habitat for numerous mammals, reptiles, and insects, including notable species like the *Cape Sugarbird*, Bontebok, *Cape Mole rat*, *Cape Gerbil*, *scatter-hoarding rodents*, *four-striped field mouse* (Holmes et al., 2022; Fig. 14.3 A and B), and various butterflies and ant species (such as *Anoplolepis* and *Pheidole*) (Slingsby, 2017). The intricate ecological dynamics of Fynbos further underline their significance in global conservation efforts to maintain the sustainability, existence, and flow of ecosystem services under this threatened biome.

2 Fire adaptability

Fynbos is a fire-driven ecosystem that relies heavily on fire for stimulating seedling recruitment (Kraaij and Van Wilgen, 2014). Therefore, it is crucial to either burn the area or provide fire-related cues for successful seedling recruitment (Holmes and Richardson, 1999). In the southwestern Cape of South Africa, fires typically occur during the hot and dry summer months (Kraaij and Van Wilgen, 2014). Fynbos vegetation has evolved various adaptations to cope with fire (Finch and Meadows, 2018), and fire intervals in the ecosystem range from 5 to 50 years (Rebelo et al., 2006). More than half of Fynbos species are capable of resprouting after fire from epicormic buds, lignotubers, or geophytic bulbs, corms, and rhizomes (van Wilgen and Forsyth, 1992). For successful recruitment of the entire Fynbos species diversity, it is important to consider spatial and temporal variations in fire intensity. However, it is crucial to avoid extremely high-intensity fires, often caused by the increased fuel load from alien

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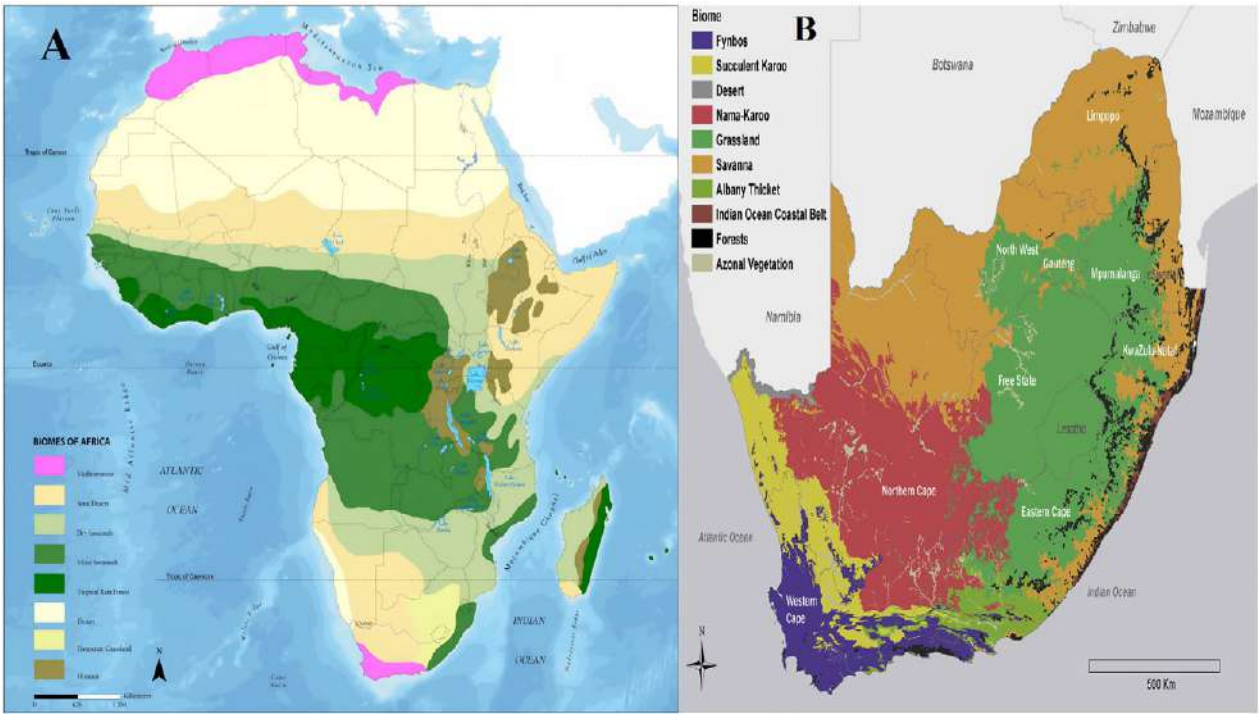

FIG. 14.1 (A) Fynbos on the African map and (B) biomes of South Africa. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: Finch, J.M., Meadows, M.E., 2018. South African biomes and their changes over time. The Geography of South Africa: Contemporary Changes and New Directions World Regional Geography Book Series. https://doi.org/10.1007/978-3-319-94974-1_7.)*

FIG. 14.2 Peninsula Sandstone Fynbos in Table Mountain National Park in Cape Town, South Africa. *(Photo credit: https://www.goodfreephotos.com/.)*

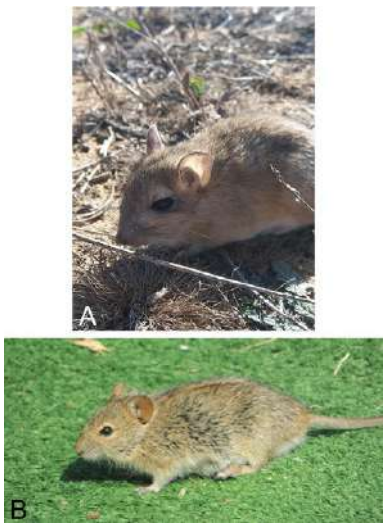

FIG. 14.3 (A) Cape Gerbil and (B) striped field mouse. *(Photo credits: (A, B) Jacques vd Merwe.)*

trees, as they may eliminate sprouts and small-seeded species (Holmes and Richardson, 1999).

To accommodate important germination cues and maintain long-term biodiversity, burning a site during late summer or early autumn is essential (Holmes and Richardson, 1999). Fire plays a critical role in nutrient cycling within the Fynbos ecosystem by preventing the accumulation of excessive pathogens and rejuvenating older vegetation. Some Fynbos plant species rely on fire for their growth and flowering immediately after the fire event, and the chemicals present in the smoke during burns play a vital role in postburn seed germination in Fynbos (Holmes and Richardson, 1999).

3 Features and dispersal pattern

Fynbos communities are characterized as evergreen, sclerophyllous shrublands that thrive in nutrient-poor soils (Holmes and Richardson, 1999). They are recognized as Mediterranean-type biomes due to their flora's physiognomy and functional ecology, as well as their association with winter rainfall and hot, dry summers (Ackerly et al., 2014). The dominant features of Fynbos include the prevalence of sclerophyllous shrubs and the crucial role of fire, which plays a significant role in maintaining biodiversity and ecological processes within most Mediterranean-type ecosystems. Fynbos is localized between two plant communities that are intolerant to fire. Toward the more humid southeastern boundaries, it is bordered by afro-temperate forests, known as the temperate boundary (MacPherson et al., 2019). On its northwestern fringe, characterized by semiarid conditions, Fynbos is adjacent to the Succulent Karoo (Gillson et al., 2020).

Notably, the dispersal of most Fynbos species occurs over short distances through mechanisms such as ant dispersal, ballistic dispersal, or passive means. Long-distance dispersal by vertebrates is rare within Fynbos vegetation (Holmes and Richardson, 1999).

4 Significance of Fynbos

A pristine particularly mountain Fynbos ecosystem that remains largely untouched offers numerous valuable ecosystem services, including the provision of water, the availability of harvestable wildflowers, attraction for tourists, genetic storage, and the preservation of biodiversity (Higgins et al., 1997). The presence of diverse species within these ecosystems plays a vital role in regulating various ecological processes such as the cycling of nutrients, water purification, and the sequestration of carbon. These processes contribute significantly to the overall well-being and stability of the environment. In 2005, experts in natural resource economics estimated that the total economic worth of the biodiversity found in the Fynbos region, which includes plants, animals, scenic beauty, ecosystems, as well as services like water purification and erosion control, exceeded R10 billion per year. This impressive figure accounted for more than 10% of the gross geographic product of the Western Cape Province during that time (SANBI, 2018).

5 Nutrient cycling and ecological stability

Shrublands within the Fynbos biome play a crucial role in maintaining ecological stability. An example of this is the *Elytropappus rhinocerotis* plant, which can extract nutrients from deep within the soil and concentrate them in the topsoil (Cramer, 2014). These nutrients are then utilized by other species, contributing to the overall health of the ecosystem. Additionally, the occurrence of fire, although more common in Fynbos compared to other vegetation types in the CFR, plays a significant role in nutrient redistribution (Cramer, 2014). During fires, volatile compounds are released and nutrients are deposited, leading to a large-scale redistribution of nutrients. This process results in the immediate increase of nutrient levels in the topsoil, including nitrogen (N), which then percolates into deeper soil layers. Furthermore, the presence of insects, particularly butterflies, within the Fynbos biome is crucial as they provide important pollination services to nearby farms and agricultural areas.

6 Genetic storage

The Fynbos ecosystem is highly valued for its significant contribution to floral genetic diversity and the presence of unique plant species that are endemic to the region (Higgins et al., 1997). Since 1970, the Agricultural Research Council (ARC) has actively engaged in a comprehensive breeding and research program to commercialize South African Proteaceae, with a specific focus on the cultivation of Protea, *Leucospermum*, and *Leucadendron*. An integral part of this program is the Fynbos Genebank, especially for use in the breeding program to conserve the genetic diversity of Fynbos plants with floriculture potential, and is, therefore, considered a national asset, as it is used in the breeding program for the development of new cultivars while supporting crop science and plant protection researches activities (Matlhako et al., 2010).

7 Cultural and research value

A substantial segment of the CFR's population, estimated to be around 6 million individuals, relies heavily on natural resources to sustain their livelihoods. This biologically diverse region is primarily located in the Western Cape, with smaller portions extending into the Northern and Eastern Cape provinces. Apart from the direct utilization of natural resources, economic benefits are also derived from the Fynbos through the cultivation and harvesting of indigenous rooibos tea, wildflowers including proteas, aromatic oils extracted from buchu, thatching reeds, and several traditional as well as commercially marketed medicinal plants (SANBI, 2018).

8 Water regulation and purification

The Fynbos biome in the Western Cape is characterized by numerous small wetlands, which play a crucial role in preserving the quality of water from subcatchments within this

biome (Turpie et al., 2010). These wetlands have significant ecological and practical benefits for both the environment and the downstream human communities. For instance, they serve as a barrier against contamination, safeguarding fisheries from harmful pollutants and mitigating the potential adverse effects on human health. In particular, the proliferation of algae or aquatic macrophytes in response to nutrient loading is a concern, given the concentration of urban settlements along the coastline. Furthermore, the movement of water within this region is effectively regulated by the mountain biomes, following a process that involves precipitation, interception, filtration, runoff, evapotranspiration, and groundwater flow (Higgins et al., 1997).

9 Threats and conservation challenges

The transformation of natural habitats due to agricultural practices, the introduction of invasive alien plant (IAP) species, and the expansion of urban areas are recognized as major threats to biodiversity (Rouget et al., 2003a; Latimer et al., 2004). Various studies have underscored the persistent and future risks posed by these factors (Rouget et al., 2003a; Latimer et al., 2004). Fynbos, a highly diverse Mediterranean-type biome, is particularly vulnerable to a combination of drivers, including climate change, alterations in land use, shifts in fire patterns, pollution, and the proliferation of nonnative plant species (Bomhard et al., 2005; Midgley et al., 2006; Driver et al., 2012; Rebelo et al., 2011; Altwegg et al., 2014; Kraaij and Van Wilgen, 2014; Slingsby et al., 2017; Gillson et al., 2020). Within South Africa's terrestrial ecosystem, the Fynbos biome faces the highest level of threat, with only fragmented remnants of its natural habitats remaining (Driver et al., 2012).

Despite the implementation of measures to protect this biome, the safeguarding of the Fynbos mountains has been more successful compared to the lowland ecosystems within the biome (Rouget et al., 2003b; Driver et al., 2012). Unfortunately, a significant number of Fynbos plant species, totaling 1736, are now classified as critically endangered (CR), endangered, or vulnerable according to the South African National Biodiversity Institute (SANBI, 2018). Additionally, the extinction of several animal species within this ecosystem, such as the black rhino, cape buffalo, hippopotamus, elephant, and other once-abundant animals in the Fynbos, is a direct consequence of urbanization and other environmental factors that have displaced them. While the Cape leopard, as well as a few other wild cats, antelopes, the cape porcupine, and mole snake, have managed to survive in this biome, their visibility and the presence have also declined over time.

10 Habitat loss

The CFR experiences substantial changes due to the impacts of agriculture, invasions by alien plant species, and urban development (Rebelo, 1992; Lombard et al., 1997; Rouget et al., 2003c). Over time, the Fynbos ecosystem within the CFR has been subject to degradation, loss of species, and transformation due to agricultural practices, forestry activities, and urbanization (Moran and Hoffmann, 2012). Presently, only approximately 15% of the original lowland Fynbos remains preserved, with more than 60% persisting in less accessible mountainous regions (Moran and Hoffmann, 2012; Rebelo et al., 2011). Land cover analyses showed that in 1990, the Fynbos biome had a natural vegetation cover of approximately 71%. However, over the span of 28 years, this coverage experienced a decline of 5% (Dayaram et al., 2021). Consequently, this has led to the fragmentation and degradation of Fynbos areas, posing significant challenges for numerous plant and animal species dependent on this specific habitat.

11 Climate change

The winter-wet, summer-dry ecosystem has been identified as highly vulnerable to future climate change, particularly due to the projected increase in intense and localized drought events (IPCC, 2007; West et al., 2012). The effects of climate change have the potential to significantly alter the Fynbos biome. This includes rising temperatures, changes in precipitation patterns, and a higher frequency of extreme events like drought, all of which can disrupt species distributions, impact pollination processes, and pose a long-term threat to the survival of certain Fynbos plant species (West et al., 2012). Research conducted by Driver et al. (2012) indicates that the eastern and northern regions of the Fynbos are already experiencing climate stress, with the climate conditions in these areas resembling those found in the Succulent Karoo or Albany Thicket. These shifts in climate have resulted in range shifts, population declines, and in some cases, extinctions of species (Hughes, 2000; McCarty, 2001; Parmesan and Yohe, 2003; Root et al., 2003). For instance, *Protea longifolia* and *Serruria linearis* are classified as CR due to their susceptibility to climate change impacts (Bomhard et al., 2005).

12 Invasive alien species

IAPs pose a significant threat to biodiversity worldwide and are recognized as the second largest global threat to biodiversity after direct habitat destruction (Latimer et al., 2004). Within the context of this global hotspot, IAPs are particularly detrimental to biodiversity, emerging as one of the most critical challenges (Rouget et al., 2003a). By the mid-1990s, an estimated 10 million hectares of land in

South Africa had fallen victim to invasive woody plants, covering approximately 16% of the country's total land area (Maitre et al., 2000; Kotzé et al., 2010). Extensive research reveals that the Fynbos biome in South Africa endured significant damage from invasive wood plants in 1997 (van Wilgen et al., 2001). Among the most pervasive and harmful alien, plant species are trees and shrubs belonging to the *Acacia*, *Hakea*, and *Pinus* genera (Parker-Allie et al., 2004; De Lange and van Wilgen, 2010; Driver et al., 2012). Extending their reach into both mountainous and low-lying regions of the Fynbos biome, these species have colonized excessive ranges of land (Rouget et al., 2003a). Consequently, Fynbos has a long-established history of alien plant introductions, with projections indicating that their density and land occupation will peak by approximately 2150 (De Lange and van Wilgen, 2010).

Particularly notable invasive species in the area encompass *Acacia cyclops*, *Acacia longifolia*, *Acacia saligna*, *Eucalyptus* spp., *Leptospermum laevigatum*, *Paraserianthes lophantha* (Rouget et al., 2003a), and *Pinus radiata* (Galloway et al., 2017). These nonnative plants tend to outperform the native vegetation, consequently altering the natural fire patterns, diminishing biodiversity, surpassing indigenous flora in resource competition (including water, nutrients, and sunlight), and amplifying vulnerability to disturbances, specifically fire-adapted invasive plant species such as pines (Driver et al., 2012). Currently, control programs aimed at combating these major invaders in the Fynbos region incorporate biological control methods that employ insects and diseases that specifically target the alien plants from their region of origin (Olckers, 2004). Extensive studies have shown that the utilization of biological control, either independently or in conjunction with other strategies, has resulted in a significant reduction in the invasiveness of certain targeted IAPs (Wood and Morris, 2007; Gordon and Fourie, 2011). Introduced biological control agents, such as weevil species, seed-feeding moth species, and gall-forming rust fungus (Moran and Hoffmann, 2012), have been utilized for this purpose. Mechanical and chemical control methods, on the other hand, are required less frequently and result in cost savings (Marais et al., 2004; Moran et al., 2004; Le Maitre et al., 2008).

13 Agriculture and forest plantations

The Fynbos biome is negatively impacted by intensive agricultural practices, such as crop cultivation and livestock farming. Additionally, inappropriate land management practices, such as monocultures and forestry, further disrupt the delicate balance of this ecosystem. Agricultural activities in the area encompass various practices, including cultivation, orchards, vineyards, bush clearing, grazing, and forestry plantations predominantly consisting of *Pinus* and *Eucalyptus* stands (Rouget et al., 2003a). Notably, the primary agricultural crop is dryland winter wheat, accompanied by substantial areas of irrigated orchards and vineyards (Turpie et al., 2010). The conversion of land for irrigation purposes, including the establishment of orchards, vineyards, pastures, parks, and golf courses, as well as dryland agriculture, has been identified as a factor contributing to increased nitrogen concentrations (ammonium, nitrates, and nitrites), likely due to fertilizer application (Turpie et al., 2010). There is a promotion of pine plantation's forestry industry as a means to ensure the economic viability of this pant (Van Wilgen, 2015), there is also an alarming expansion of vineyards at unprecedented rates, these pose a threat to the biodiversity hotspots long-term sustainability (Fairbanks et al., 2004).

14 Urbanization

The Mediterranean Biome in the CFR experiences the second-highest rate of population growth in the area, following Chile (Underwood et al., 2009). The process of urbanization primarily takes place in the lowland regions surrounding coastal Cape Town (Underwood et al., 2009; Rebelo et al., 2011). According to the United Nations, it is estimated that by 2030, approximately 71.3% of South Africa's population will reside in urban areas, with this number expected to increase to nearly 80% by 2050. Projections also indicate that by 2030, approximately 75% of the South African population will be living in urban areas (Cooperative Governance and Traditional Affairs, 2016). Therefore, it is crucial for conservation efforts within the CFR to establish connections between human activities and ecological and evolutionary dynamics (Rebelo et al., 2011; Alberti et al., 2020).

In urban areas, certain challenges related to the Fynbos ecosystem arise, including the improper disposal of garden waste, unauthorized picking of flowers, and the presence of vagrants sleeping over (Rebelo et al., 2011). Urbanization directly impacts the edge effect of the remaining Fynbos fragments by encouraging the proliferation of nonnative annual grasses. Moreover, the discharge of sewage and the utilization of lawn fertilizers contribute to the eutrophication of shrub vegetation in these areas (Rebelo et al., 2011).

15 Pollution

Increased nitrogen emissions from vehicle exhaust and other forms of pollution have profound impacts on ecosystems (Rebelo et al., 2011). These impacts include changes in vegetation composition, such as an increase in grassiness and a higher frequency of fires (Wilson et al., 2009). Additionally, the presence of sewage spills in nature reserves, especially when left unaddressed, can lead to the depletion of numerous

plant species within Fynbos ecosystems (Rebelo et al., 2011). It is essential to note that Fynbos ecosystems are naturally low in nutrients, and any form of pollution that enriches the soil, be it from sewage spills, fertilizers, drift from surrounding fields, or enriched groundwater, directly influences the vegetation in this particular ecosystem (Holmes et al., 2022). Furthermore, pollutants such as pesticides, herbicides, and industrial emissions have the potential to adversely affect both the flora and fauna within these ecosystems, further highlighting the detrimental consequences of pollution (Holmes et al., 2022).

16 Conclusion

The sustainability and preservation of the Fynbos biome, a unique hotspot that supports a high level of endemic plant diversity, is threatened by ongoing global climate change and land use changes. To address this challenge, it is crucial to enhance the resilience of the Fynbos biome by expanding existing protected areas, as highlighted by Holmes et al. (2022). Efforts should also be focused on restoring degraded patches within the biome. This can be achieved by implementing measures to increase and improve the prevention, control, and eradication of invasive species. These measures may include the use of biological, chemical, and mechanical methods (Marais et al., 2004; Moran et al., 2004; Olckers, 2004; Le Maitre et al., 2008).

Recent studies indicate that climate change contributes to the increased invasion of alien or weedy species, as well as an elevated frequency of extreme events such as storms (Holmes et al., 2022). Furthermore, it is predicted that the West Coast forelands area will face higher temperatures and reduced winter rainfall, which will have implications for fire return intervals and postfire recruitment, particularly in low sand Fynbos (Holmes et al., 2022). Consequently, consecutive years of drought may hinder seedling establishment, as they may survive the first year but face mortality in the second year. Therefore, it is essential to consider these effects on seedling survival and adjust conservation strategies accordingly.

Promoting alternative sources of income for local communities within this biome will have the potential to substantially decrease the direct utilization of natural resources, such as the extraction of wildflowers and reeds for thatching purposes (SANBI, 2018). Additionally, special attention should be given to the lowland Fynbos region, where the depletion of vegetation has been progressively driven by agricultural practices, the establishment of resorts, and urban expansion. Consequently, any development of urban settlements, along with their associated infrastructure, must prioritize the preservation of the local biome, while effectively managing waste disposal, preventing oil spills, and implementing well-planned construction activities.

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Chapter 15

Mediterranean shrublands

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1 Introduction

Climate change caused by human activities, anthropogenic and natural disturbances, and the synergies between climate change and disturbances, along with the exponential increase in the global population, menace the integrity of all the world's biomes, including Mediterranean shrublands. This jeopardizes their perpetuation and therefore, the ecosystem services they provide to the populations inhabiting them are also not guaranteed. In this sense, ecology has become an indispensable tool for understanding the functioning of ecosystems, the degree of impact they undergo, and to what extent they are resilient (Sala et al., 2000). Ecological resilience is the ability of a system to absorb disturbances and reorganize while maintaining the same function, structure, and identity (Walker et al., 2004). On the other hand, socio-ecological resilience is the capacity to adapt to unexpected changes without significantly altering human well-being (Chapin et al., 2010). These conceptualizations involve a paradigm shift from the extractive model of natural resource exploitation to a realistic view of the long-term negative consequences of this model.

According to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change (Stocker et al., 2013), the intensity and duration of drought have likely increased in many regions of the world since 1970, and the loss of aboveground plant growth is 60% higher during extreme short-term droughts, the frequency of which has increased in the last fewdecades (Smith et al., 2024).

Mediterranean shrublands are primarily found in Mediterranean climate zones within the mid-latitudes. These zones encompass the Mediterranean Basin, central Chile, Western Cape of South Africa, and southwest and southern Australia (Keeley et al., 2012). Nevertheless, this biome is not limited to the more typical Mediterranean climate zones. Peel et al. (2007) updated the world climate map and expanded the csb climate type (which corresponds with the Mediterranean climate, according to the Koppen-Geiger climate classification) into an extended area (clearly showed in the fig. 7 of Peel et al., 2007) at a latitude similar to that of central Chile, crossing the Andes mountain range, in Argentina.

Even without considering drastic changes, such as those brought about by the expansion of the agricultural frontier, shrublands do not consistently maintain stability and can either increase or decrease their presence in the landscape. This variability is influenced by factors such as grazing, fire, and the interactions of these disturbances with the changing climate. Shrubby grasslands worldwide are experiencing significant changes, with a notable shift from herbaceous to woody plant dominance in some areas, and the opposite transition from woodland to grassland in others, leading to what is termed as grass-woodland transitions (Sala and Maestre, 2014). This concept is not novel; as far back as 1989, Westoby et al. introduced a model called "states and transitions" for managing rangelands not in equilibrium.

The study was conducted at San Ramón ranch, located 30 km east of Bariloche in the Río Negro province, northwestern Patagonia, Argentina (41°03′19″S, 71°01′50″W) (Fig. 15.1).

The climate is Mediterranean and temperate, with 60% of the precipitation concentrated in autumn and winter. The mean annual precipitation is 578 mm, and the mean annual temperature is 8.6°C. Strong W-NW winds frequently blow throughout the year, accentuating water stress during the summer (based on 1928–2024 records from the meteorological station of San Ramón ranch, unpublished data). The soils are Haploxerolls with sandy-loam texture, and superficial horizons contain low concentrations of organic matter (Gaitán et al., 2004). Shrublands of *Mulinum spinosum* (Cav.) Persoon, *Senecio bracteolatus* Hook. & Arn., and *Acaena splendens* Hook. & Arn. dominated different sectors. Monospecific shrublands of *Fabiana imbricata* Ruiz & Pav. are found in the middle slopes (Fig. 15.2).

Before the first vegetation monitoring (1999), the region experienced a severe drought (363 mm in 1998 vs 578 mm, the historical mean annual precipitation) (see Ghermandi

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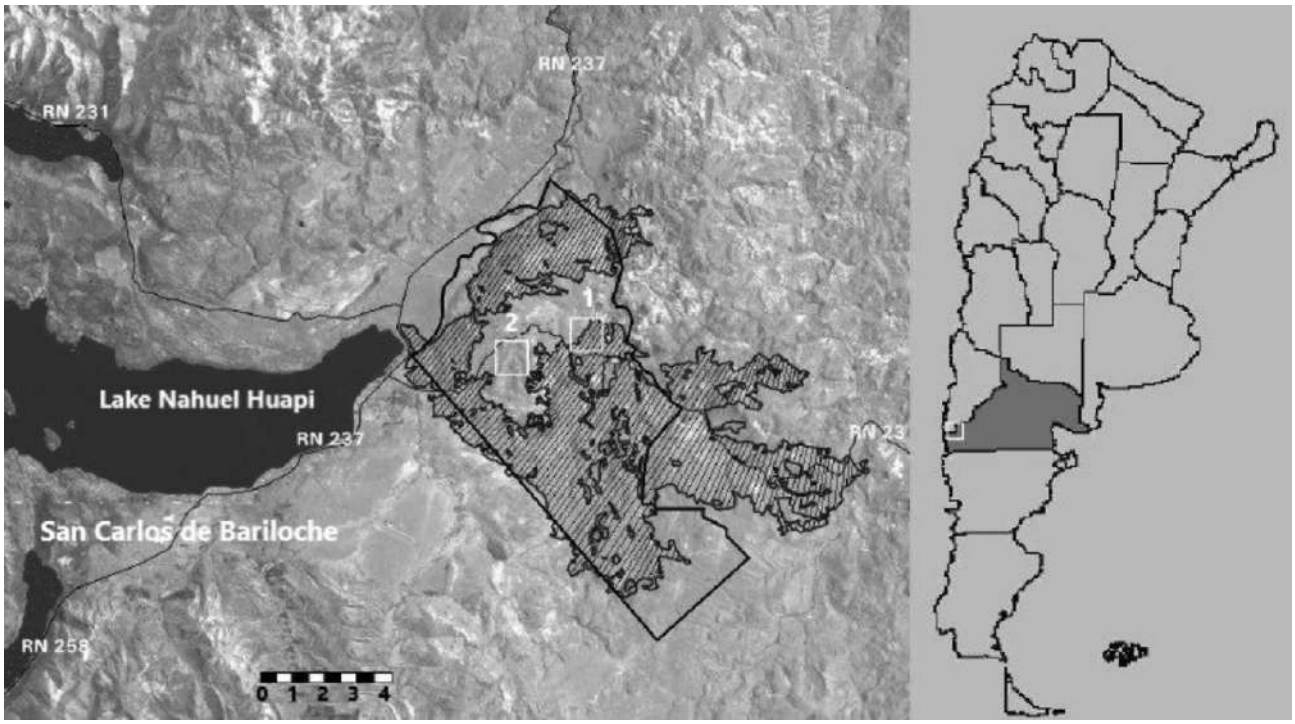

FIG. 15.1 Study area. Shaded area indicates the 1999 fire. (1) Burned site and (2) overgrazed site. *(Credits: Authors.)*

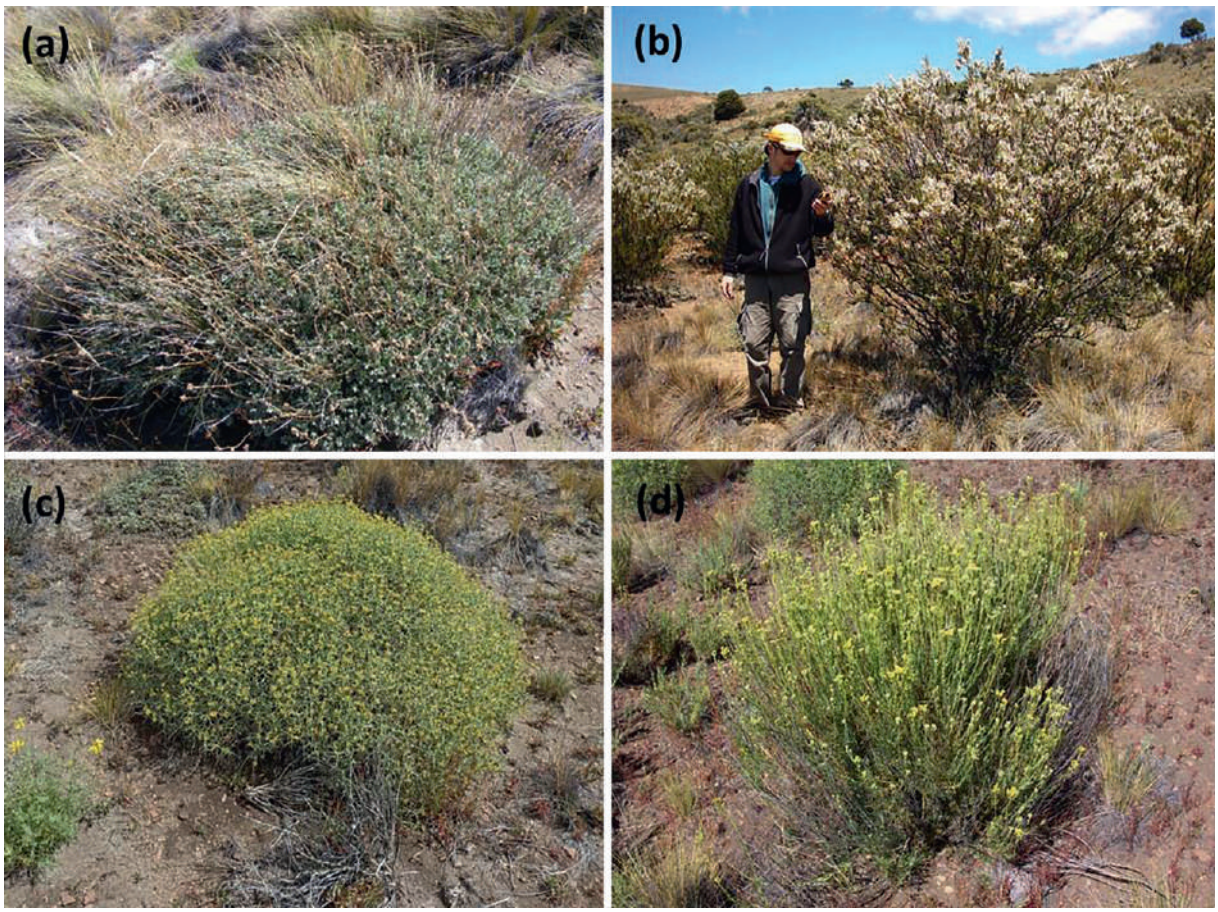

FIG. 15.2 Dominant shrubs of the northwestern Patagonian steppe: (A) *Acaena splendens*, (B) *Fabiana imbricata*, (C) *Mulinum spinosum*, and (D) *Senecio bracteolatus*. *(Photo credits: The authors.)*

FIG. 15.3 Sheep and cattle in northwestern Patagonia. *(Photo credits: The authors.)*

and Gonzalez, 2009). Following that, a wildfire occurred (Ghermandi et al., 2004), and in 2011, a volcanic eruption deposited tephra (Gaitán et al., 2011; Ghermandi et al., 2015). The land use in the region is almost exclusively dedicated to extensive livestock production, involving sheep and cattle (Fig. 15.3). Some areas of the ranch are degraded due to overgrazing.

Our main objective has been to describe the postfire behavior of shrubs, and elucidate the mechanisms behind their establishment, growth, and senescence. These insights are derived from long-term monitoring, including the evaluation of the nurse effect. In this chapter, we describe the knowledge acquired during over 20 years of field observation, monitoring, and experimentation in the northwestern Patagonian semiarid steppe. Throughout this period, the climate underwent notable changes, and starting in 1991, a drought trend became established (Fig. 15.4).

We will conclude with recommendations related to land use and climate change for the conservation of the integrity of the ecosystem of the Mediterranean shrublands of northwestern Patagonia.

2 Postfire dynamics of shrubs: How do shrubs recover lost space?

In the northwest Patagonia steppe, a recurrent fire regime exists and the postfire advancement of shrubs can be visualized at the landscape scale (Fig. 15.5).

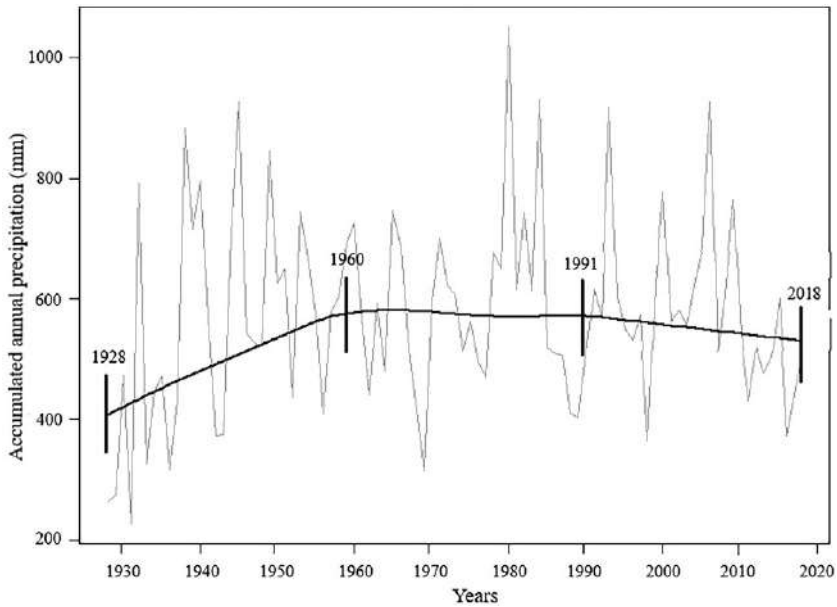

FIG. 15.4 Accumulated annual precipitation (mm) for the period 1928–2018 (San Ramón meteorological station, unpublished data). *Vertical lines* delimit the periods with positive trends (1928–1960) and negative trends (1991–2018). *(Credit: The authors.)*

In January 1999, a severe wildfire occurred and affected 13,280 ha, equivalent to 60% of the ranch. From January 2001 to 6 years later (2001–2006), the recruitment of seeder shrubs and the growth of resprouter shrubs were monitored in five plots of 10 m in diameter each (78.5 m^2, total sampled area: 392.5 m^2). Shrubs were counted and two widths from each one were measured to calculate the area occupied by each plant. Then, we calculated the space recover of each species of shrubs (as the percentage of the total sampled area) to estimate whether resprouting shrubs are disadvantaged or favored by fire compared to seeders. Percentage of space recover was calculated each year, and with respect to the prefire area only in the case of *Acaena splendens*, which dead plants leave postfire imprints. All individuals of *F. imbricata* germinated from seeds originating from a nearby unburned shrubland. The seeds were carried from the slope to the plain during the early postfire period when the burned vegetation did not intercept the flow of soil and seeds were dragged by the rains.

No differences were found in the occupancy percentages of *A. splendens* when comparing postfire years. The occupancy percentage of *F. imbricata* and *Mulinum spinosum* significantly increased in 2006 compared to previous years (2001–2004). The value in 2005 did not significantly differ from that of 2006. The occupancy percentage of *Senecio bracteolatus* was significantly higher in 2005 and 2006 compared to the early years (2001−2003). The occupancy percentage in 2004 did not differ from the value in 2005 but was significantly lower than that in 2006 (Fig. 15.6).

With respect to the postfire strategies, having vegetative regeneration from protected meristems (typical of the resprouter species such as *M. spinosum* and *S. bracteolatus*) constituted a competitive advantage over shrub species that are killed by fire and recover only through seeds (seeders such as *A. splendens* and *F.imbricata*). The growth rate of the shrubs was different for this reason. *Acaena splendens* is a cushion subshrub that barely branches out and grows more in two dimensions than in height. This results in slower growth compared to the other three species that have a clearly shrubby habit. After the fire, *A. splendens* leaves a very clear trail on the soil, from which the prefire occupancy surface could be measured. After 7 postfire years *A. splendens* reached only 52% of prefire cover (Fig. 15.6).

Fabiana imbricata suffered from browsing by European hares, limiting its growth, whereas *S. bracteolatus* and especially *M. spinosum*, shown exponential growth due to their numerous branches. It is interesting to observe how the architecture of plants strongly influences their growth. The structural characteristics, such as their branching patterns, leaf arrangement, and overall form, play a crucial role in determining how they respond to their environment. This adaptation is often a result of evolutionary processes that optimize the ability of species to capture sunlight, absorb nutrients, and reproduce efficiently. *Fabiana imbricata* forms coetaneous shrublands on mid-slope sites. In the study area, it was not present before the fire. It represents an advancement of the unburned shrubland. *Mulinum spinosum* and *S. bracteolatus* are adapted to fire, regaining its initial size in a few years. Postfire mortality of *M. spinosum* is practically zero and it is a highly persistent species in the community (Gonzalez et al., 2015). Being thorny, it is not preferred by livestock and is superficially consumed by sheep only during the flowering season.

In conclusion, the native shrubs not only recover postfire but can also form new shrubland foci, taking advantage of the favorable conditions generated postfire by resource release.

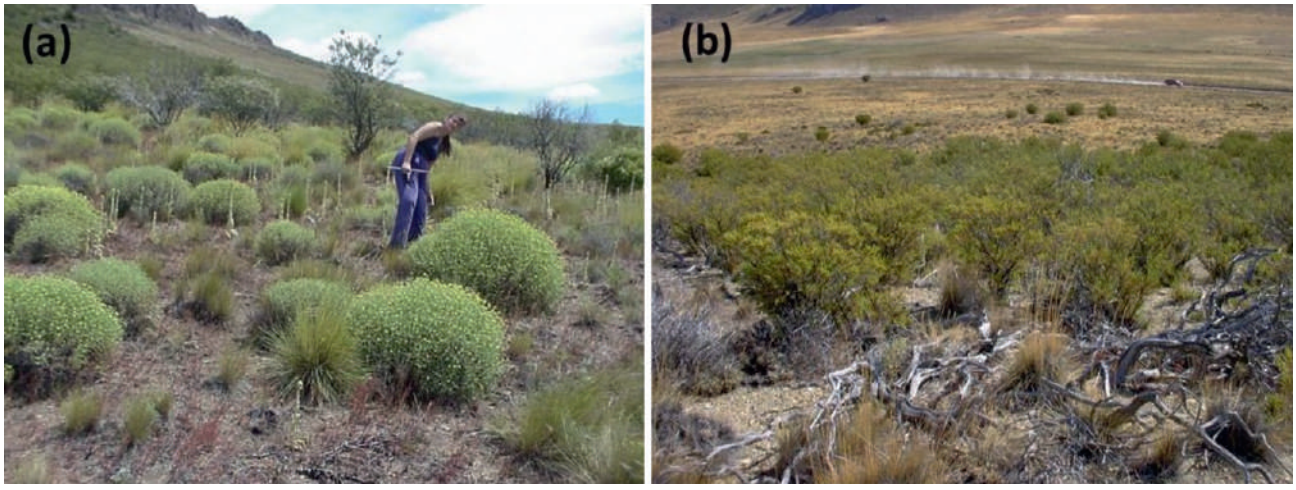

FIG. 15.5 Shrublands of (A) *Mulinum spinosum* and (B) *Fabiana imbricata. (Photo credits: The authors.)*

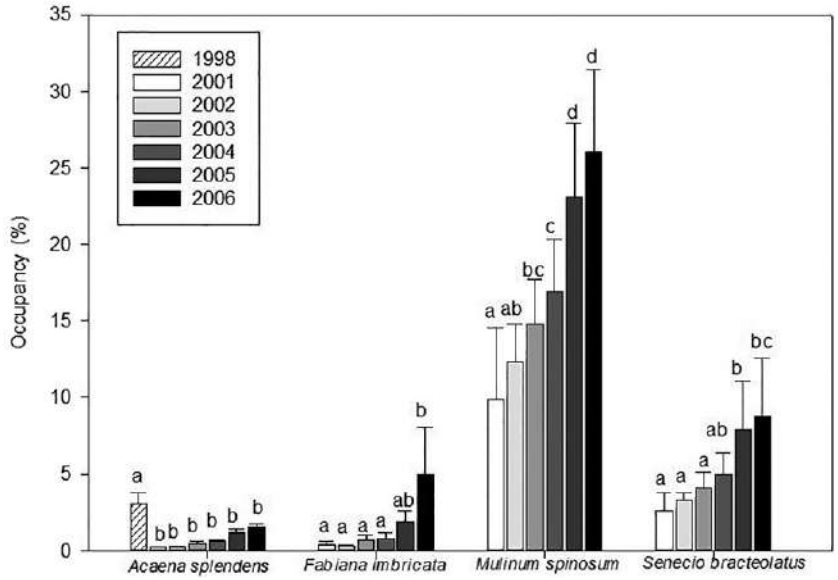

FIG. 15.6 Postfire percentage of shrub occupancy (mean ± SE) in grasslands of northwestern Patagonia. The prefire percentage of *Acaena splendens* occupancy (1998) is presented. Different lowercase letters indicate significant differences ($P < 0.05$). *(Credit: The authors.)*

Thereafter, we will describe in detail our knowledge about the eco-physiology of the shrub species that we have studied, grouping them into seeders and resprouters.

3 The seeder shrubs: *Fabiana imbricata* and *Acaena splendens*

Fabiana imbricata (Solanaceae) is a shrub adapted to a semiarid environment, where annual rainfall ranges from 300 to 600 mm. Using dendrochronological techniques, we inferred that this species establishes itself in coetaneous groups of plants, principally in response to fire (since it is a seeder shrub), but also after soil movement, as occurs on roadsides or in strips of gas pipelines (Ghermandi et al., 2010, 2013; Oddi and Ghermandi, 2015).

F. imbricata can reach up to 3 m in height, is long-lived (living at least 150 years), and reproduces from the age of 5. It flowers in summer and produces fruit in autumn, yielding many small and lightweight seeds (200,000 per plant) measuring 1 mm in length and 0.1 mg in weight (Ghermandi et al., 2013).

F. imbricata seeds form persistent soil seed banks, remaining viable for a long time, probably more than 50 years. We deduce the longevity of seeds from their extremely small size. It is known that very small seeds are buried and dispersed over time. For example, very small seeds of *Verbascum* spp. have been shown to remain viable for more than 80 years (Kivilaan and Bandurski, 1981).

Seed scarification enhances *F. imbricata* germination, indicating the presence of some type of dormancy mechanisms: mechanical restriction of germination of the embryo, presence of chemical inhibitors, and restricted oxygen uptake. Dormancy in *F. imbricata* is crucial to prevent germination in unfavorable conditions, particularly in harsh and variable environments. This trait becomes significant for the species, ensuring that germination occurs during favorable conditions, such as after a fire when gaps in vegetation open up (Dudinsky and Ghermandi, 2013). In fact, *F. imbricata* belongs to the Solanaceae family, like *Nicotiana tabacum*, whose seed dormancy was extensively studied by Finch-Savage and Leubner-Metzger (2006) and exhibits very similar characteristics.

We studied the role of fire on *F. imbricata* seed germination. In order to achieve this, a factorial experiment was designed under laboratory conditions to study the effect of heat, charcoal, smoke, scarification, and their interactions on *F. imbricata* seed germination. Seeds treated with the higher temperatures required a longer period of time to germinate, significantly affecting the mean germination time. Total germination percentages in *F. imbricata* were significantly enhanced by scarification, and the interaction of scarification, heat, and smoke was more important than the effect of each fire factor alone (Fig. 15.7). The positive response to fire cues indicates that this species would have an adaptive advantage to colonize postfire (Dudinsky and Ghermandi, 2013).

Mathematical models of population dynamics of the seeder *F. imbricata* at different fire frequencies and at different precipitation values indicate that the shrublands advance in the steppe reaching its maximum with a fire frequency of once every 6–10 years (de Torres Curth et al., 2012; Ghermandi et al., 2010).

F. imbricata forms monospecific, coetaneous, and dense shrublands, which can occupy several hectares. In

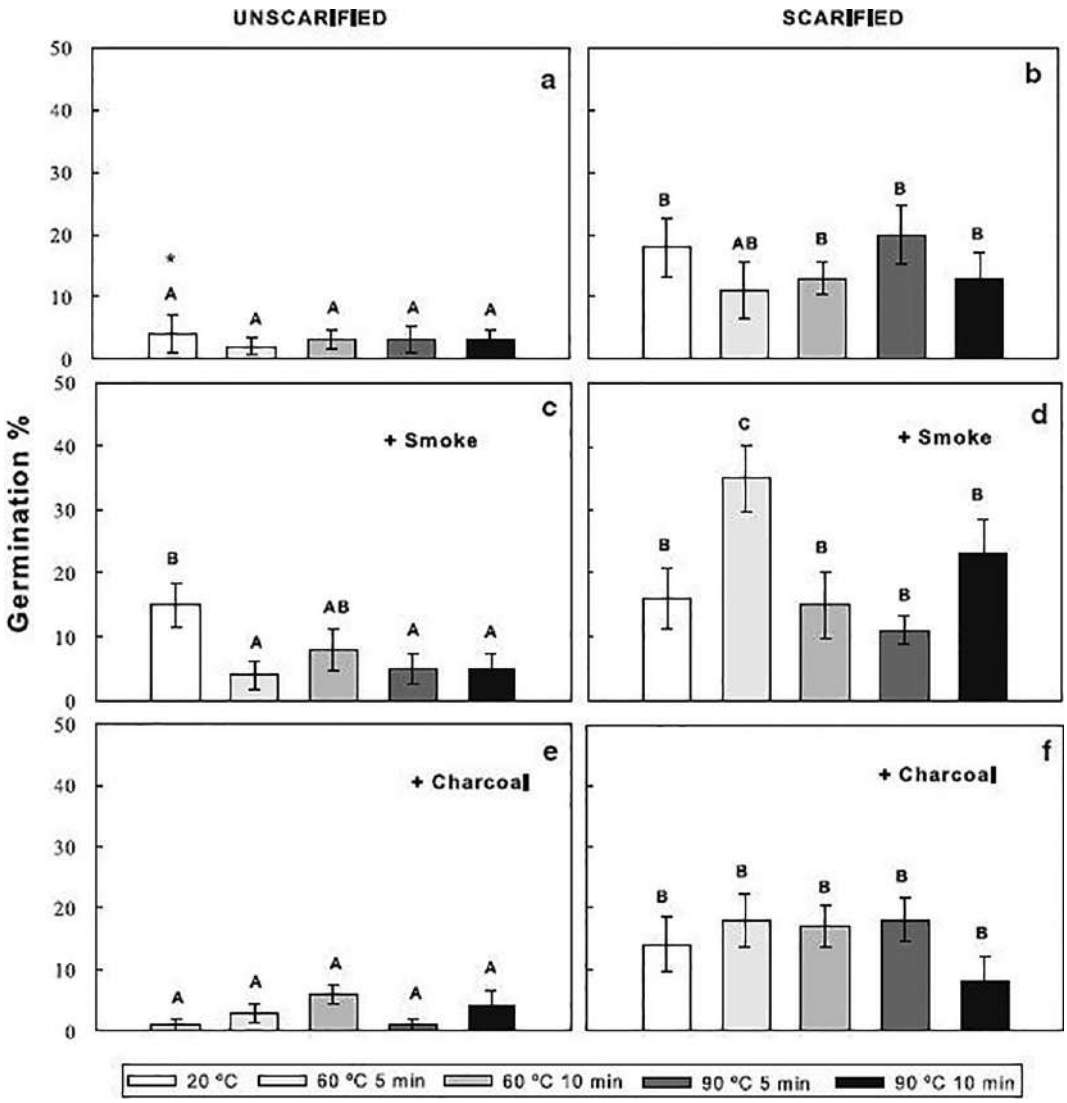

FIG. 15.7 Effect of (A) heat, (B) heat + scarification, (C) heat + smoke, (D) heat + smoke + scarification, (E) heat + charcoal, and (F) heat + charcoal + scarification on seed germination (Mean ± SE) of *Fabiana imbricata*. Different uppercase letters indicate significant differences. *(Credit: The authors.)*

shrublands composed of sexually mature plants, there is no establishment of new individuals, even though the seed bank in the soil is abundant (more than 2000 seeds/m^2 up to 1 cm depth). If the seeds are too deeply buried, they will find it nearly impossible to germinate, even under suitable rainfall and temperature conditions. They need to reach the surface, or to arrive very close to it, in order to emerge as seedlings. However, being close to the surface also entails risks because superficial soil layers dry out rapidly during the summer, inhibiting seed germination (Ghermandi et al., 2013).

After the fire at San Ramón ranch in January 1999, the formation of incipient shrublands was observed. Similarly, the formation of a shrubland can be observed perfectly mimicking a section of the Pilcaniyeu-Bariloche gas pipeline (Fig. 15.8), where the plants were found to have the same age as the gas pipeline construction. The presence of the new postfire foci of shrublands therefore, responds to a chain of factors: fire followed by autumn-winter exposure of deep seeds, combined with rainy springs allowing seed imbibition and germination (Ghermandi et al., 2010; Oddi et al., 2010). It is possible that inter-annual climatic variability also plays a significant role. For instance, the El Niño-Southern Oscillation (ENSO) phenomenon involves the alternation of rainy years (during the El Niño phase) followed by dry years (during the La Niña phase) in northern Patagonia (Veblen and Kitzberger, 2002).

The ENSO is attributed to an oscillation in the equatorial Pacific Ocean temperature that influences large-scale weather patterns of air masses circulation and, consequently, influences the redistribution of energy (expressed by temperatures) and water (manifested as changes in precipitation) globally. Rainy year in semiarid ecosystems lead

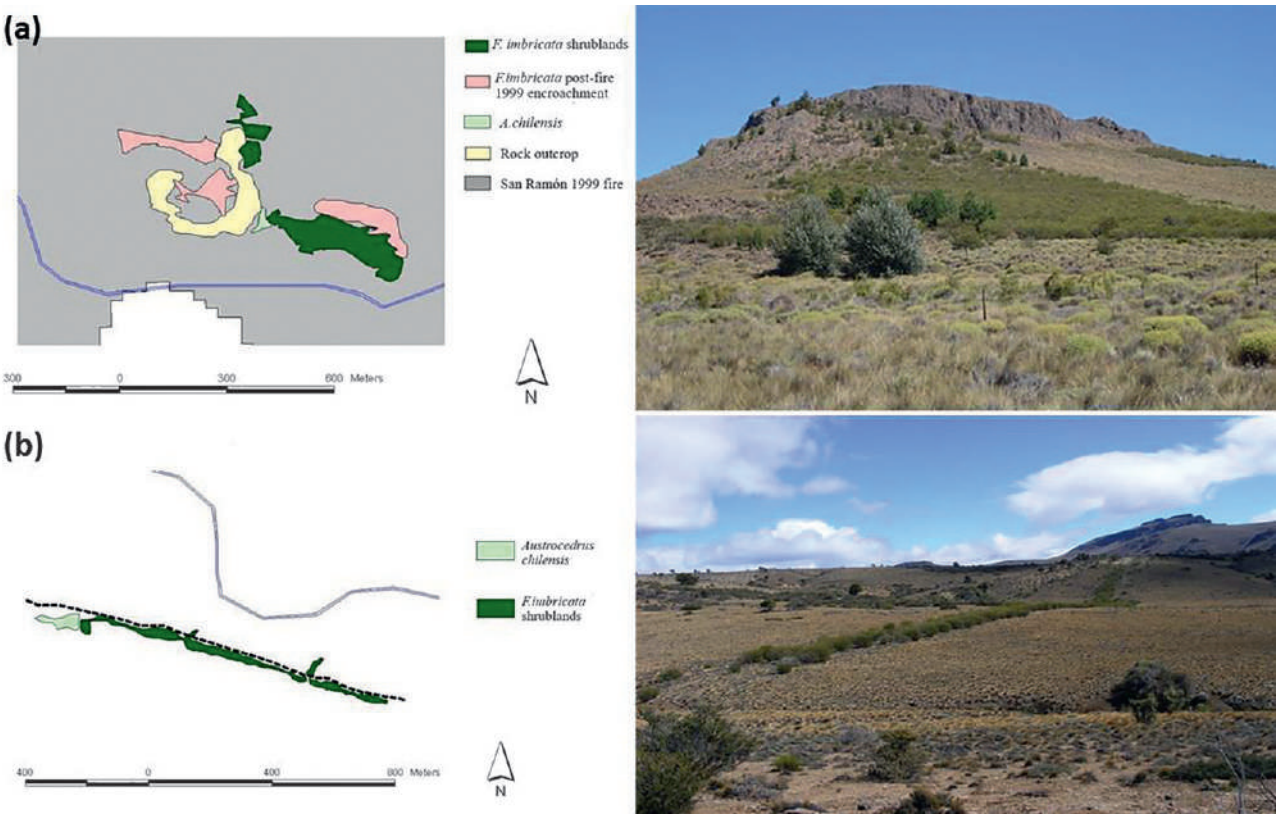

FIG. 15.8 *Fabiana imbricata* shrubland encroachment (A) after 1999 fire and (B) after construction of gaspipeline. *(Photo credits: The authors.)*

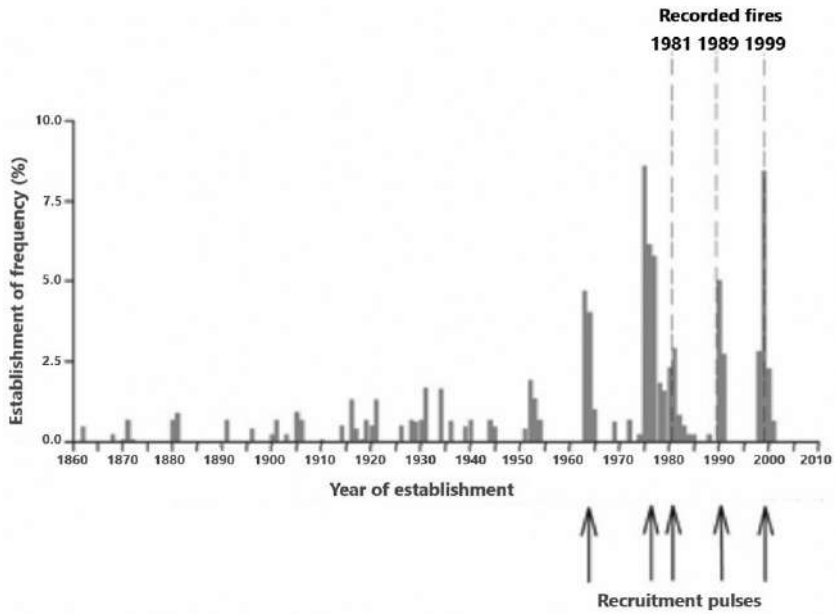

FIG. 15.9 Relationship between fires and years of massive seedling recruitment of *Fabiana imbricata*. *(Credit: The authors.)*

to an increase in biomass. If this year is followed by a drought, the biomass will dry, become prone to burning and, if postfire spring is wet, rains will promote the *F. imbricata* establishment (Ghermandi et al., 2013) (Fig. 15.9).

Acaena splendens (Rosaceae) is a ruderal cushion shrub of 40–60 cm high (Correa, 1971) (Fig. 15.2A). It reproduces exclusively by seeds contained in thorny and large achenes (8 mm length, 33 mg), which are dispersed through ectozoochory (Marticorena, 2006). *Acaena splendens* colonizes

sandy soils and areas degraded by overgrazing (Gonzalez and Ghermandi, 2021) as well as disturbed areas like roadsides (Chichizola et al., 2021).

As mentioned earlier, *A. splendens* plants damaged by fire die. Consequently, this species recolonizes the postfire environment by seed germination in areas near the burned mother plant. We named this mechanism self-replacement (Ghermandi et al., 2004) (Fig. 15.10A). The achenes do not bury themselves due to their large size but they can survive fire because of two factors: tolerance to high temperatures due to their hard cover (Gonzalez et al., 2010) and/or burial by the granivorous rodents *Akodon xanthorinus* and *Eligmodontia morgani* (Ghermandi, 1995).

We monitored the growth of *A. splendens* plants postfire, and we have observed that they often establish very close together, and as they grow, they often merge, forming an apparent single plant. We call this effect coalescence (Fig. 15.10B). Additionally, during this coalescence process they can engulf postfire juveniles of other species and we call this mechanism phagocytosis (Fig. 15.10C). In fact, the individuals of *A. splendens* are almost creeping and grow like an oil stain in two dimensions.

It is not possible to determine the age of the plants, as they lack growth rings. However, in monitored plants recruit postfire (1999), we have observed that the senescence stage initiates at around 20 years (2019). During this stage, the canopy begins to die centrifugally (from the center to edge) forming an internal area partially protected by the 20 cm high canopy of the same plant (Fig. 15.10D).

The internal area functions as a microsite with microenvironmental conditions (soil fertility, temperature, and humidity) that significantly differ from those found in adjacent gaps (i.e., areas between tussocks and shrubs). These conditions facilitate the colonization of *A. splendens* seedlings (another type of self-replacement) and other species in the internal senescent areas (Gonzalez and Ghermandi, 2019). Therefore, senescent *A. splendens* plants act as nurse plants (sensu Niering et al., 1963),

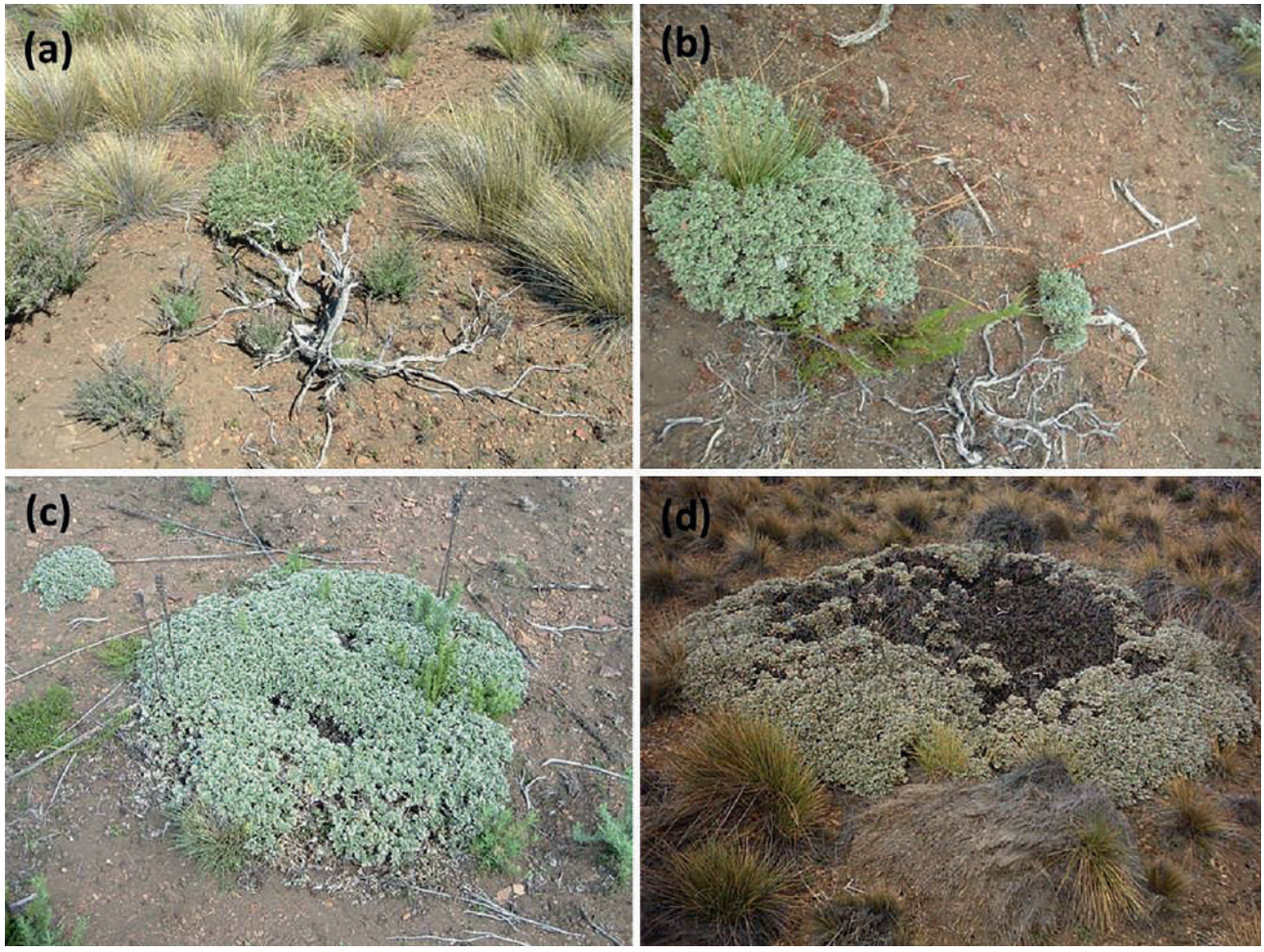

FIG. 15.10 Different processes in *Acaena splendens* plants (A) self-replacement, (B) coalescence, (C) phagocytosis, and (D) senescence. *(Photo credits: The authors.)*

playing a crucial role in enhancing species richness and diversity in degraded grasslands (Gonzalez and Ghermandi, 2019).

The microsites of senescent *A. splendens* plants play an important role in influencing soil characteristics. These microsites contain decomposing organic matter that contributes with essential nutrients to soil fertility and maintains moisture levels. The values of C, N, and the C:N ratio were higher in *Acaena* plants compared to adjacent gaps. Specifically, the organic matter content was significantly higher (2.75% vs 1%) (Gonzalez and Ghermandi, 2019) and exceeded the typical value (2%) of the Haploxeroll soil, which includes a mollic epipedon in the surface layer (Gaitán et al., 2004). The presence of this organic matter increased soil particle aggregation and enhanced water retention capacity (Pugnaire et al., 2004). Furthermore, microsites of senescent *Acaena* plants exhibit lower temperatures and higher humidity compared to adjacent gaps, during the stress produced by the hydric deficit (spring–summer; Gonzalez and Ghermandi, 2019) (Fig. 15.11). Shrubs are widely used as nurse plants in restoration programs in semiarid regions (Maestre et al., 2001; Soliveres et al., 2011). In several degraded grassland areas within the study site, *A. splendens* dominates the vegetation cover (approximately 25% of a total cover of 40%–50%), while the cover of the native grass *Festuca pallescens*, the main forage, is very low or nonexistent (Gonzalez and Ghermandi, 2021). In order to facilitate the reintroduction of *F. pallescens* into the degraded grassland, an assessment was conducted to evaluate the performance of the senescent *A. splendens* plant as a nurse plant (Gonzalez and Ghermandi, 2019).

The recruitment and survival of *F. pallescens* seedlings from seed sown in senescent *A. splendens* plants and in adjacent gaps were assessed over 1 year. In the *A. splendens* microsites, the seed germination of *F. pallescens* was notably higher compared to the gaps (38.5% vs 8%). Additionally, *A. splendens* microsites provided favorable conditions (i.e., low temperature and high humidity) during spring months (September-December) that facilitated the survival of *F. pallescens* seedlings compared to gaps (12% vs 3%). Despite these initial positive trends, no seedlings survived 1 year after the initial field sowing. The high mortality observed was attributed to the unusually hot and dry spring-summer, characterized by a 77% lower accumulated precipitation than the historical mean (September-March: 47 mm vs 171 mm). It was concluded, that the mechanism of facilitation tends to fail during years of extreme drought conditions, when the water deficit exceeds the favorable conditions that *A. splendens* plants can typically provide.

Nevertheless, it is important to note that the observed exceptionally dry conditions in this particular year should not discount the possibility that, under normal meteorological conditions, the buffering effect of *A. splendens* could benefit the survival of *F. pallescens* seedlings. The use of *A. splendens* plants is desirable over other nurse plants as they gradually lose competitiveness as they senesce. This has implications at the community level, indicating a positive shift in species composition, transitioning from an undesirable shrub to economically significant species.

4 Resprouter shrubs: *Mulinum spinosum* and *Senecio bracteolatus*

Mulinum spinosum (Apiaceae) is a cushion shrub 30–150 cm tall with spiny deciduous leaves(Fig. 15.2C). This shrub contributes low to forage biomass as only new shoots and flowers are consumed by sheep (Siffredi et al., 2015). It flowers between November to January and produces seeds in late summer, which are dispersed by anemochory

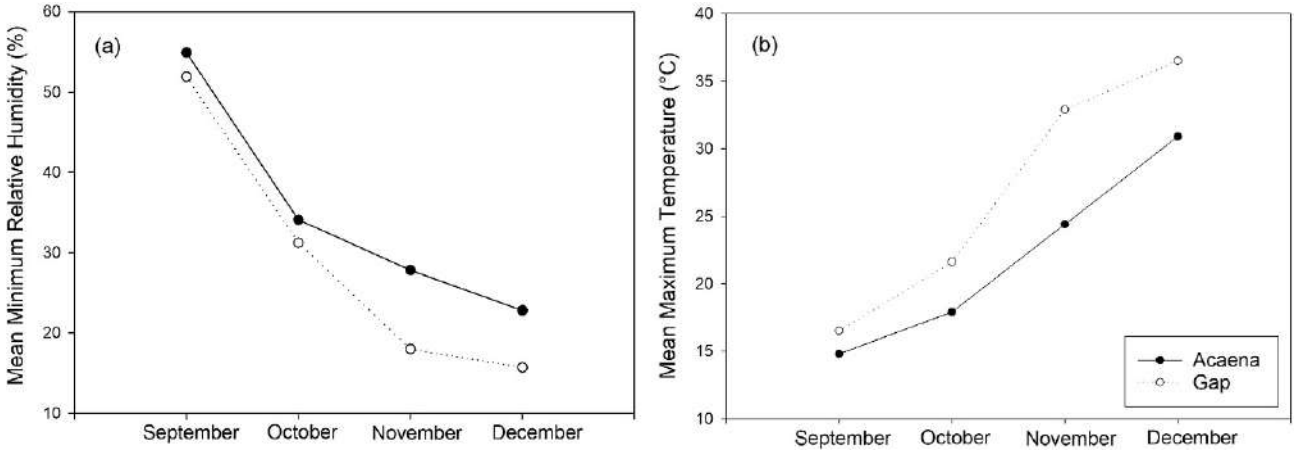

FIG. 15.11 (A) Mean minimum relative humidity and (B) maximum temperature recorded in *Acaena* and gap microsites from September to December. *(Credit: The authors.)*

(Gonzalez and Ghermandi, 2012). The seeds are large (5.6 × 4.6 mm, 4.8 mg) and flat forming transient seed banks (Gonzalez and Ghermandi, 2008). These seeds exhibit morphophysiological dormancy, which means that the embryo requires an after-ripening period of about 6 months to germinate (Gonzalez and Ghermandi, 2012).

In experimental burns simulating the impact of both low and high fire intensities on germination of sown seeds of *M. spinosum*, findings revealed that seeds partially tolerate temperatures of 80°C (Control: 58%, high temperature treatment: 24%, low temperature treatment: 20%)(Gonzalez et al., 2022). This susceptibility was attributed to the basal position of the embryo, which reduces its insulation from elevated temperatures.

Plants grow during spring and early summer and branch at the end of summer. The new branches further extend in the following spring (Damascos and Ghermandi, 2007). The annual growth of the plant is restricted to the external layer (<12 cm thick) while the inner part of the cushion is composed of dry branches. This architecture provides a fine fuel material that facilitates the circulation of oxygen, increasing the flammability and promoting combustion. Besides this architecture, the presence of secondary metabolites (terpenes and phenols) in the leaves increases their flammability. Results from leaf flammability experiments, indicating a short time to ignition and high heat released (Blackhall and Raffaele, 2019), along with measurements at the plant level showing a high proportion of fine dead fuel and low bulk density (Gonzalez et al., 2022), confirm the high flammability of this shrub. In particular, the bulk density (dry biomass per unit volume) is related to plant compaction; the lower the compaction, the lower the bulk density (Pausas and Moreno, 2012). In less compact plants, the residence time of fire is reduced, allowing buds to survive and resprout after fires.

The characteristics related to flammability are complemented by a high postfire regrowth capacity, rendering this species well-adapted to fire (Ghermandi et al., 2004; Gonzalez et al., 2015). In experimental burns of individual plants, the survival of *M. spinosum* was 100%, even when exposed to high fire intensity (Gonzalez et al., 2015). This shrub has underground buds located at a depth of 5 cm, which survive due to the buffering effect of the soil. Heat transfer in the soil is limited, even during high-intensity fires; for instance, at a depth of 2.5 cm, the soil temperature does not exceed 100°C (Whelan, 1995; Carrington, 2010; Franzese and Ghermandi, 2012). *Mulinum spinosum* plants exhibited vigorous resprouting after burns, rapidly recovering their volume (23% of their preburn volume 1 year after the burn and 40% after 2 years.

A stochastic matrix model was developed to investigate the impact of fire frequency and fire size on long-term population dynamics of *M. spinosum* (de Torres Curth et al., 2020). Their findings indicate that for fire return intervals exceeding 50 years, the population becomes independent of fires, displaying an annual population growth rate of 5.6%. It was suggested that postfire resprouting would allow *M. spinosum* to maintain population growth even under high fire frequencies, excluding annual fires. Under frequent fire conditions, resprouting plays a relatively more significant role than sexual reproduction, although this importance diminishes with decreasing fire frequency. Climate models forecast an increase in temperature and a decrease in precipitation (10%–30%) for northwestern Patagonia (Pessacg et al., 2020), both conditions favorable for enhancing fire frequency. In this scenario, the *M. spinosum* population is expected to persist in the community due to its high resprouting capacity.

Finally, the role of *M. spinosum* as a nurse plant in the dynamics of seedling recruitment in more xeric steppe conditions (250 mm) was assessed (López and Cavallero, 2017). This shrub facilitated seedling emergence and survival of perennial species (grasses and shrubs) even in a highly degraded state. In the overgrazed grasslands of our study area, we observed the native forage grasses *Bromus setifolius* and *Hordeum comosum* sheltering in *M. spinosum* plants, where they resist overgrazing by producing propagules, avoiding local extinction, and consequently enhancing the resilience of the grassland (Fig. 15.12).

FIG. 15.12 *Bromus setifolius* refugees on *Mulinum spinosum* plant. *(Photo credits: The authors.)*

Senecio bracteolatus (Asteraceae) is a densely multibranched shrub 40–100 cm high (Correa, 1971) (Fig. 15.2D). The fruits are medium-sized achenes (1 mg) (hereafter seeds). The seeds have a pappus and are dispersed by anemochory, germinating in autumn and spring (Franzese et al., 2009). This species produces abundant seeds (between 8000 and 80,000), although a high percentage of them are nonviable (69%) and suffer predispersal predation (22%), mainly caused by a dipteran (Order Brachycera) (Franzese, 2006). Both factors would significantly reduce effective seed production and thus limit seedling recruitment. Seeds are poorly represented in the transient seed bank (<10%) (Gonzalez and Ghermandi, 2008). Despite the absence of seeds in the banks of burned areas, laboratory experiments and experimental burns on buried seeds found that they can tolerate temperatures of 80°C (Gonzalez and Ghermandi, 2012; Gonzalez et al., 2022).

Senecio bracteolatus resprouts after fire from aboveground basal buds of the root crown. Moreover, it recruits from seeds of nearby unburned areas and resprouting burned plants (Ghermandi et al., 2004; Franzese et al., 2009). Survival and growth of *S. bracteolatus* plants postfire depends on fire intensity. Experimental burns on individual plants, simulating both high and low intensity, revealed lower survival in plants subjected to high temperatures compared to those exposed to low temperatures (63% vs 33%) (Gonzalez et al., 2015). The lower survival and growth were attributed to plant architecture and nonburied buds exposed to fire, lacking the buffering effect provided by soil. *Senecio bracteolatus* exhibits numerous woody basal stems, creating a dense fuel that burns slowly, allowing it to sustain high temperatures for a longer time. This prolonged heat exposure damages the buds, reducing the number and size of resprouts (Whelan, 1995).

During the first growing season, low fire temperatures enhanced abundant seed production of burned *S. bracteolatus* plants (Gonzalez et al., 2015). Moreover, the burned plants exhibited larger seeds (0.75 vs 0.59 mg) and a higher germination (100% vs 80%) compared to those of unburned plants (Gonzalez et al., 2015). This difference was attributed to an increase in postfire resource and nutrient availability. Early postfire seed production can promote early seedling recruitment, allowing them to benefit from the resource release and reduced competition after fire. Additionally, seed size has important ecological implications for germination and seedling recruitment, growth, and survival. Large seeds confer advantages during establishment, exhibiting better tolerance to water stress and competitive edge due to rapid root development (Leishman et al., 2000; Westoby et al., 2002).

Postfire recruitment of *S. bracteolatus* seedlings occurs in gaps and under the canopy of tussock grasses (Franzese et al., 2009) and other shrubs (Gonzalez, personal observation). We assessed the microsite conditions and the recruitment and establishment of *S. bracteolatus* seedlings under the canopy of the tussock grass *Pappostipa speciosa* and in gaps between vegetation 6 years after fire (Franzese et al., 2009). The canopy of the tussock grass reduced light intensity (95% less radiation than in gaps) and soil temperature (24°C lower than in gaps), and the presence of litter maintained humidity. The number of *S. bracteolatus* seedlings was higher under tussocks than in gaps, but the number of juveniles and adults was higher in gaps. Therefore, the tussock grass acts as a nurse species for the recruitment of *S. bracteolatus*, providing an optimal refuge protecting them from the harsh conditions of summer. Nevertheless, as the seedlings grow, their roots become deeper and begin to compete with those of the tussock grasses. Therefore, the initially favorable conditions for *S. bracteolatus* seedlings changes over time turning unfavorable due to competition with tussock grasses for the resources (e.g., light and water). However, gaps present more stressful situations. Thus, for a successful establishment, a combination of favorable meteorological conditions during the two recruitment seasons of autumn and spring (moderate temperatures and abundant rainfall) followed by a mild summer is required, and such conditions are not very common. In conclusion, fires create an opportunity for a significant recruitment of *S. bracteolatus*, driven by the increase in suitable microsites. Postfire interactions, involving both facilitation and competition, function as regulators of the abundance and distribution of this species (Gonzalez et al., 2010).

5 Conclusions

In Northwestern Patagonia steppe, we have carried out various studies on postfire strategies of shrubs (Franzese et al., 2009; Ghermandi et al., 2013; Dudinsky and Ghermandi, 2013; Oddi and Ghermandi, 2015; Gonzalez et al., 2015). The advance of shrubs, which in our case are all native, means different things depending on the more or less anthropocentric approach: on the one hand, it responds to natural processes of postfire secondary succession; on the other hand, it decreases the quality of environments dedicated to livestock breeding because shrubs are woody and less palatable for sheep and cows. From a more classic ecological perspective, we could see the potential postfire shrub advance as something inevitable and of a neutral nature, but that would be true only if humans had not strongly intervened in climate change that leads to droughts and modifies the fire regime. In this context, many of the wildfires are not natural, even if they are triggered by lightning.

The postfire sprouting strategy is, in the short term, more effective in terms of space occupancy than recolonizing from seeds, as shrub seeders do. However, evolutionarily, sexual reproduction is advantageous, because it

allows for genetic exchange and greater plasticity. In the long term, new stands of *F. imbricata* with plants that reach a height of 2–3 m change the physiognomy of the community from shrubby steppe to monospecific shrubland. This is a radical change that does not occur with the auto-perpetuation through resprouting of *M. spinosum* and *S. bracteolatus*.

The new shrublands of *F. imbricata* locally change the physiognomy of the steppe, and we have demonstrated that they have two effects: the first is to decrease the productivity of ecosystem because *F. imbricata* is not eaten by livestock, and the second is to decrease plant diversity due to the allelopathic substances that this species accumulates in soil (Dudinsky and Ghermandi, 2013). However, the long-term dynamics of *F. imbricata* indicate that fires often eliminate the shrublands in flat sectors of the steppe, pushing them toward the middle slope. This limits the negative effects of the postfire advance of this shrub (Oddi, 2013).

The steppe is resilient to fire; however, the system does not return to the same prefire situation due to fire-induced changes, some of which are irreversible and potentially harmful in terms of productivity loss, as shrubs are less palatable than grasses. Rapid changes in the fire regime, related to climate change, are likely to significantly alter species proportions and modify the structure of the ecosystem. Some consequences of these changes can be foreseen and taken into account in future land use plans, aiming to improve and align them more closely with the socioeconomic development of the region.

Acknowledgments

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Chapter 16

Xeric shrublands

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1 Introduction

Globally, there are 846 terrestrial ecoregions spread across 14 biomes and 8 realms (Dinerstein et al., 2017), encompassing 41% of the Earth's drylands surface area, with xeric shrublands being distributed within these regions. Located in arid and semiarid regions worldwide, they constitute environments conducive to sparse and open vegetation, with shrubs as the tallest plants (Frost et al., 2018). These areas account for 35% of global diversity (Cherlet et al., 2018), influenced by various factors, including water scarcity, topographical variations, geology, soil type and quality, seasonal rainfall patterns, fire, herbivore pressure, and the impacts of human management over centuries, and in some cases, millennia (Davies et al., 2012).

Shrublands are defined as areas where shrubs exceed a height of 0.5 m, and shrub cover is above 25% (or, if the cover is less than 25%, shrubs have a cover of at least 10% and exceed herbaceous cover), while tree cover remains below 10% (Faber-Langendoen et al., 2012). These are characterized by woody perennial plants, often lacking a defined crown, typically ranging in height from 0.5 to 5 m (FAO, 2020). Some species may exhibit a shrub form in exposed conditions but transform into small trees in moist and sheltered environments. Shrub types can be deciduous or evergreen (Ellsworth and Sternberg, 2015). Alternative terms such as scrubs, scrubland, and shrubby formations are also used. Shrub height and density may vary, ranging from low-growing, widely spaced configurations in open shrublands to taller, denser shrubs in closed shrublands (Richardson and Rejmanek, 2011; Wassilieff, 2023). In specific regions, this shrub formation may constitute stable vegetation or represent a transitional stage in ecological succession following fire, deforestation, or agricultural abandonment (Kycko et al., 2022).

It is essential to note that primary shrubland areas are predominantly found in Mediterranean climates characterized by wet, mild, long winters, and hot, dry summers (Mevy et al., 2022). Plant species in these areas typically consume less water and exhibit greater resistance to aridification (Casermeiro et al., 2004). Climatically, xeric shrublands receive more rainfall than deserts and grasslands but less than forested areas (Chenchouni et al., 2025), with annual precipitation ranging between 200 and 1000 mm. Summers are characterized by hot and dry conditions, with temperatures exceeding 38°C, while winters are cold and wet, often dipping below 1°C for extended periods (https://earthobservatory.nasa.gov/biome/bioshrubland.php).

In general, shrublands consist of shrubs or short trees, typically with an open structure that allows grasses and other short plants to grow between them. The spatial heterogeneity of water, sediments, and nutrients (Yang et al., 2011; Fernández, 2015; Frost et al., 2018) results in a discontinuous, sparse, and open vegetation cover forming patches (Kröpfl et al., 2013), surrounded by a matrix of denuded soils. In these ecosystems, shrub species often exhibit adaptations to xeric conditions and specific characteristics such as small leaves, thorns, or succulent tissues (Joffre and Rambal, 2001). These ecosystems are well adapted to disturbances such as fire, which becomes a necessary and inevitable driving force in their ecological dynamics, even serving as a stimulus for specific diversification in the Mediterranean region, as observed in the case of fynbos (van Wilgen, 2013; Rundel et al., 2016).

Being commonly associated with deserts as a general biome, xeric shrublands are characterized by distinctive biogeographical and ecological features, along with unique resources and attributes. Despite the increasing interest in these environments, a comprehensive biogeographical synthesis of their ecosystem attributes is still lacking. This literature review aims to examine the definition of shrublands, their geographic distribution, biodiversity, human impacts, and the conservation tools employed for their preservation. This assessment of the current state of knowledge could

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prove instrumental in developing more effective strategies for the planning and design of restoration and conservation actions.

2 Geographic distribution of xeric shrublands

Xeric shrublands are primarily located in western coastal regions between 30° and 40° latitude both north and south. These encompass areas such as southern California, Chile, Mexico, the regions surrounding the Mediterranean Sea, and the southwestern regions of Africa and Australia (Vallejo et al., 2012). While the boundaries of xeric shrublands may vary between regions, they persist predominantly within arid and semiarid zones globally (Fig. 16.1).

3 Biodiversity

Despite being considered inhospitable environments, xeric shrublands exhibit spatial variability that provides a range of resources, microclimatic refuges, and ecological niches (Yang et al., 2011), contributing to various ecosystem services (ES) (Ribeiro and Ribeiro, 2016; Rodary et al., 2016; Wangai et al., 2016; Hao and Yu, 2018). These services include soil erosion prevention and optimization of carbon exchange, thus mitigating carbon emissions into the atmosphere (Scheffer et al., 2014; Carrión-Prieto et al., 2017; Farguell et al., 2022). Such conditions enable the maintenance of ecological processes during periods of environmental stress (Loreau et al., 2001; Thuiller et al., 2005) and enhance resilience to adverse environmental conditions (Malika et al., 2009).

There are five Mediterranean climate regions globally (the Mediterranean Basin, California, central Chile, the Cape region of South Africa, and the southern regions of Australia) that share a Mediterranean-type (warm-temperate) climate characterized by warm, dry summers and cool, wet winters, supporting evergreen sclerophyllous shrublands as the dominant vegetation complex (Rebelo et al., 2006; Rundel et al., 2016). Collectively, these regions account for 2% of the Earth's surface, host 20% of the planet's plant species, are considered biodiversity hotspots, with fire regimes stimulating species diversification (Rundel et al., 2016), and are inhabited by approximately 5% of the world's population (del Pozo et al., 2019). They include:

3.1 Mediterranean shrublands

Mediterranean shrublands are predominantly situated in the xerothermic zone of the Mediterranean region, constituting permanent formations where soil or climate characteristics prevent the establishment of a dense tree cover (Casals et al., 2023). Also known as Mediterranean scrublands, these areas feature plant formations commonly referred to as "garrigue" or "maquis" in France or "macchia" in Italy, depending on the nature of the soil substrate (limestone or siliceous). Found in regions bordering the Mediterranean Sea, particularly in southern Europe (Spain, Italy, Greece, Portugal), North Africa (Morocco, Algeria, Tunisia), and certain parts of the Middle East (Lebanon, Palestine, Turkey).

Typical Mediterranean communities are predominantly composed of woody plants with conservative characteristics, including slow growth, high sclerophyllicity associated with low water and nutrient availability, small leathery leaves, thick cuticles, and deep root systems (Sardans and Peñuelas, 2013). Common shrub species include Mediterranean scrub oak (*Quercus coccifera* L.),

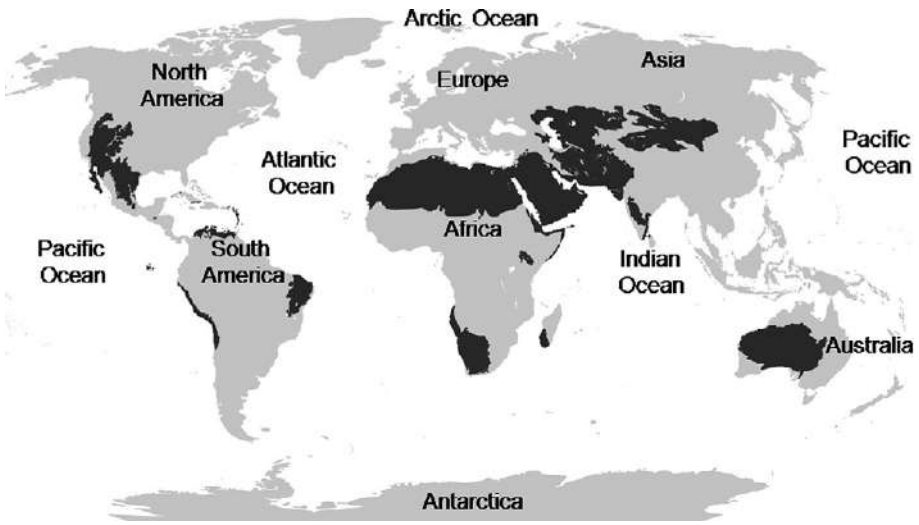

FIG. 16.1 Distribution of xeric shrublands on the earth. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: https://commons.wikimedia.org/wiki/File: Biome_map_13.svg (Accessed 1 January 2024).)*

mastic (*Pistacia lentiscus* L.), rockrose (*Cistus* spp.), lavender (*Lavandula* spp.), rosemary (*Salvia rosmarinus* L. syn. *Rosmarinus officinalis*), myrtle (*Myrtus communis* L.), and juniper (*Juniperus* spp.).

Mediterranean shrublands are among the most fire-prone vegetation types due to their high fuel load, elevated fine particle content, low humidity in summer (Gabriel et al., 2021), and high concentration of volatile organic compounds (Santana et al., 2011). These habitats collectively harbor 10% of the world's species (Kycko et al., 2022).

3.2 Sahelian shrublands

Sahelian xeric shrublands represent an arid ecosystem located in the Sahelian region of Africa, spanning several West African countries, including Senegal, Mauritania, Guinea-Bissau, Gambia, Mali, Burkina Faso, Niger, Chad, Sudan, and Eritrea. The exact demarcation between the Sahara and the Sahel is challenging, but generally accepted rainfall contours of 100–200 and 400–600 mm/yr serve as the northern and southern limits of the Sahel, respectively (Anyamba and Tucker, 2005).

According to Begué et al. (2011), the Sudanian zone in this region is characterized by various natural vegetation forms, including shrubs, wooded savannas, and forests. Broad valleys predominantly feature tree parks and cultivated fields growing crops such as sorghum, maize, and cotton. Common shrubs and small trees in this ecosystem include acacias [e.g., *Senegalia senegal* (L.) Britton], baobabs (*Adansonia digitata* L.), *Calotropis procera* (Aiton) W.T. Aiton, and thorn bushes (Ozer et al., 2010). Described as a degradation hotspot (Nicholson, 2002), this zone's condition is significantly influenced by the rainfall regime (Fig. 16.2).

3.3 California Chapparal

The predominant vegetation type in southern California is a diverse assembly of Mediterranean species characterized by drought-tolerant evergreen woody shrubs known as chaparral (Syphard et al., 2019). Threatened by frequent fires (Franklin et al., 2014), the current chaparral flora includes several palaeo-endemic taxa specific to chaparral, such as chamise (*Adenostoma fasciculatum* Hook and Am), scrub oak (*Quercus berberidifolia* Liebm), and species within the *Ceanothus* genus, and manzanita (*Arctostaphylos* spp.), along with other families like Cactaceae (*Bergerocactus* spp.), Rutaceae (*Cneoridium dumosum* (Nutt) Hook.f), Papaveraceae (*Dendromecon* spp.), Anacardiaceae (*Malosma* spp.), and Hydrangeaceae (*Carpenteria* spp.) (Rundell, 2018).

3.4 Fynbos

Fynbos corresponds geographically to the Capensis, as delimited by Werger, representing one of the vegetation biogeographical regions of southern Africa, extending between 31°S and 35°S and 18°E to 27°E (Moll and Jarman, 1984). Dominated by shrubs with small evergreen leaves, fynbos encompasses three dominant vegetation types: fynbos, renosterveld, and strandveld, with regeneration closely linked to fire (Rebelo et al., 2006). Fynbos

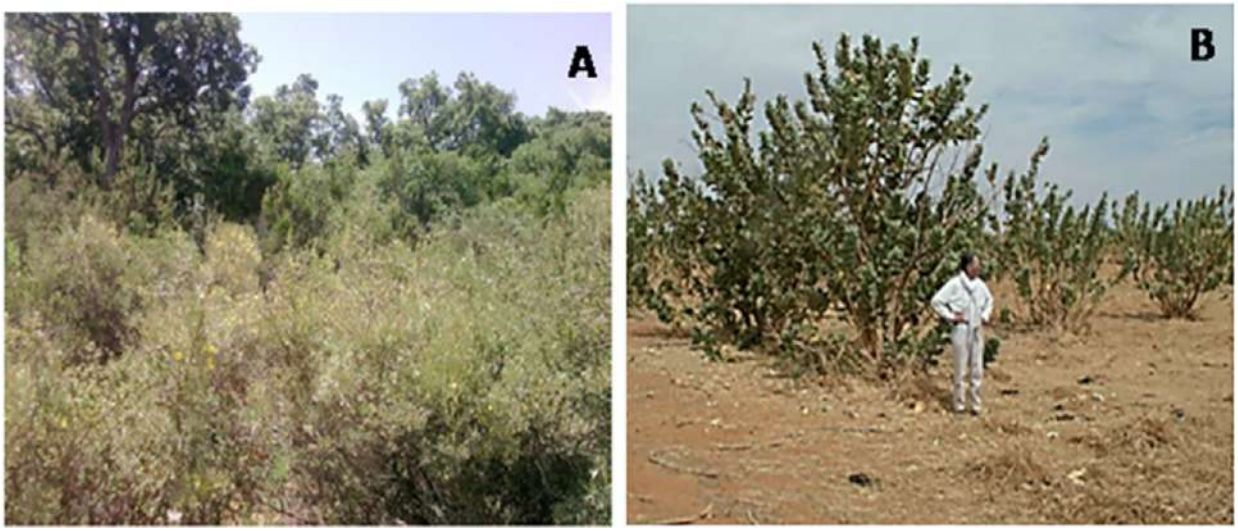

FIG. 16.2 (A) Algerian cork oak forest burned in 1983, 1993, and 2000. (B) Sahelian shrublands (Typical vegetation typical of Sahelian shrublands: extension of *Calotropis procera* colonizing heavily degraded former fallow land (southern Mauritania, February 2005)). *(Photo credits: (A) Chouahda, S., 2016. Résilience des écosystèmes forestiers du nord-est algérien après incendie: cas des subéraies. Doctoral Thesis, University of Annaba, Algeria and (B) Ozer, P., Hountondji, Y.C., Niang, A.J., Karimoune, S., Laminou Manzo, O., Salmon, M., 2010. Désertification au Sahel: historique et perspectives. Bull. Soc. Géogr. Liège 54, 69–84. https://hdl.handle.net/2268/10347.)*

has long been associated with oligotrophic soils (Coetsee et al., 2015). Speciose families in inland fynbos floras include Ericaceae, Restionaceae, and Proteaceae (Grobler and Cowling, 2021). Only one fynbos species, *Protea nitida* (Mill.) (waboom), regularly attains a tree-like form and is confined to the lower (richer) scree soils of the fynbos.

3.5 Karoo shrublands

The Karoo, an arid and semiarid region in southwestern southern Africa, is characterized by significant geological, climatic, and floristic diversity, particularly featuring low shrub vegetation on shallow soils (Cowling, 1986). The vegetation includes various species of shrubs, succulents, and low trees. Common plant species in the karoo include spekboom (*Portulacaria afra* L. Jacq.), karroid shrubs (e.g., *Euphorbia* spp., *Aloe* spp.), and various grasses, with dwarf succulent shrubs occurring along the west coast at rainfall levels below 200 mm per year (Milton et al., 1998).

3.6 Australian shrublands

Australia's shrublands exhibit diverse climates, ranging from hot, dry conditions to more moderate temperatures with varying amounts of rainfall. Characterized by the presence of Acacia and Eucalyptus species, including *Acacia aneura* (F. Muell), *Acacia tetragonophylla* (F. Muell), *Acacia victoriae* (Benth), *Eucalyptus* spp., *Atriplex bunburyana* (F. Muell), *Eremophila cuneifolia* (Kraenzl), etc. (Enright et al., 2007).

3.7 Patagonian shrublands

As described by Kröpfl et al. (2007), the Patagonia region of South America, extending over parts of Argentina and Chile, features shrublands known as the Patagonian steppe or Monte. These shrublands are characterized by low-growing shrubs, grasses, and hardy plant species adapted to the windy, semiarid conditions of the region, with daily and seasonal temperature variations and seasonal rainfall of no more than 200 mm per year, generally concentrated in winter and/or spring. Chile's Mediterranean ecosystems are among the most productive of the Mediterranean biomes, with soils derived from basic eruptive rocks, particularly basalt, which is rich in phosphorus (average phosphorus content of 0.08%, twice that of the Mediterranean Basin and California and three times that of the Mediterranean area of South Africa and Australia (Le Houérou, 2009). Common shrub families include evergreens like Berberidaceae (*Berberis* sp.), Proteaceae (*Lomatia hirsuta* Lam), Celastraceae (*Maytenus boaria* Molina), Aquifoliaceae (*Schinus patagonicus* Phil.), and deciduous varieties like Grossulariaceae (*Ribes cucullatum* Hooker and Arnott), Nothofagaceae (*Nothofagus antarctica* G. Forst.), Verbenaceae (*Diostea juncea* Gillies and Hook), and others (de Paz et al., 2021).

3.8 Chaco shrublands

One subtropical ecosystem is the Grand chaco, an extensive region of almost 1,000,000 km^2 that spans the plains of northern and central Argentina, western Paraguay, and southeastern Bolivia, as well as the extreme west of the state of Mato Grosso do Sul in Brazil (Pennington et al., 2000). The shrublands of the chaco have a semiarid to dry, subhumid climate, with distinct wet and dry seasons, and most rainfall occurs in the summer months. Winters can be relatively cold. Formerly dominated by xerophilous semideciduous forests (Cotroneo et al., 2018), the main physiognomic type is now composed of succulent shrubs (mainly Chenopodiaceae, Cactaceae, and Portulacaceae) and Acacia (Zak and Cabido, 2002), as well as other tree species such as quebracho (*Schinopsis* spp.), carob (*Prosopis* spp.), and algarrobo (*Prosopis alba* Griseb), Table 16.1).

TABLE 16.1 Biogeographic and plant diversity traits of the five Mediterranean-type ecosystems of the world.

Region	Area (10^6 km^2)	Native plant species	Endemism (%)	Modal natural fire frequency (years)	Topographic heterogeneity	Climatic heterogeneity
Southwestern Australia	0.31	8000	70	10–15	Low	Moderate
Cape Region	0.09	9000	68	10–20	Moderate	High
Mediterranean Basin	2.30	25,000	50	30–100	High	Very high
California	0.32	5000	60	30–100	Very high	Very high
Central Chile	0.16	2900	30	Fire absent	Very high	Very high

Table adapted from Cowling, R.M., Rundel, P.W., Lamont, B.B., Arroyo, M.K., Arianoutsou, M., 1996. Plant diversity in Mediterranean-climate regions. Trends Ecol. Evol. 11(9), 362–366. https://doi.org/10.1016/0169-5347(96)10044-6.

4 Conservation areas and their ecological indicators

Anthropogenic activities have extensively impacted terrestrial ecosystems, leaving few natural areas protected (Cardillo et al., 2006). Current biodiversity extinction rates are hundreds or even thousands of times higher than initial rates (Ceballos et al., 2015), underscoring the critical role of protected areas (PAs) as refuges for biodiversity (Hoffmann et al., 2019). These areas deliver diverse ES with socioeconomic and ecological significance, including environmental education, recreation, aesthetic values, cultural heritage (Ribeiro and Ribeiro, 2016; Vlami et al., 2017), and ecological functions like nitrogen fixation and carbon sequestration (Piao et al., 2009; Díaz-Delgado and Mücher, 2019; Jelen et al., 2020). Importantly, they possess the capacity to autonomously conserve biodiversity without the need for management interventions (Leroux et al., 2010).

Governments and private actors consider PAs a key element of conservation efforts (Strassburg et al., 2020), proving effective against biodiversity loss (Waldron et al., 2020). Prioritization of conservation areas is essential, emphasizing the value of new PAs (Venter et al., 2016) and guiding the allocation of conservation efforts (Hofstad, 2005), particularly where the lack of a functional classification of national ecosystems poses a serious challenge (Martínez-Tilleria et al., 2017).

According to Dudley (2008), PAs are defined as "*a clearly defined geographical space, recognized, dedicated, and managed, through legal or other effective means, to achieve the long-term conservation of nature with associated ES and cultural values*." These areas can be wholly or partially protected, with varying forms such as natural monuments, wildlife sanctuaries, biosphere reserves, Ramsar sites, botanical gardens, protected landscapes, nature reserves, national parks, world heritage sites, etc. (Jelen et al., 2020).

Juffe-Bignoli et al. (2014) report a global database comprising 2363 PAs recognized under international conventions, including 228 natural and mixed World Heritage sites and 2135 Ramsar sites. The global surface area of PAs has nearly doubled since the early 1990s, covering more than 242,400 PAs, amounting to 15% (20.6 million km^2) of the world's land area (Díaz-Delgado and Mücher, 2019; UNEP-WCMC and IUCN, 2024), up from 13.4% (Coad et al., 2009).

However, Jenkins and Joppa (2009) note that half of the ecoregions fail to reach the 10% protection target set by the Convention on Biological Diversity. Over three-quarters, covering most of the globe, have less than 10% of their area under strict protection, with 4% having no PAs at all, and 13% lacking any strict PAs. Of 846 terrestrial ecoregions, 98 ecoregions (12%) exceed half the PAs, 313 ecoregions (37%) fall short of half the PAs but still have enough intact habitat to meet the target, and 207 ecoregions (24%) are at risk, where only 4% of the natural habitat remains on average (Dinerstein et al., 2017).

In northern Ethiopia, Belay et al. (2019) reveal a steady decline in land cover in xeric shrublands from 1973 to 2015, with the dominant vegetation cover decreasing from 59.9% in 1973 to 22.9% in 1995 and 9.4% in 2015. According to Juffe-Bignoli et al. (2014), although deserts and xeric shrublands represent 18.9% of the Earth's surface, protection only covers 12.2% (Table 16.2).

Biodiversity is a "multidimensional" concept encompassing genetic, species-specific, and ecosystemic or landscape aspects. It cannot be adequately assessed by a single indicator to evaluate the state of ecosystems in conservation areas, monitor the effectiveness of conservation efforts, and identify potential threats to the environment (Chevassus-Au-Louis et al., 2009). Ideally, a multitude of indicators should be used to provide a comprehensive understanding of ecosystem health and PAs effectiveness. Focusing exclusively on a single species or taxon, for example, can lead to taxonomic bias and reduced biodiversity (Anderson, 2022). Key indicators include:

- Species richness.
- Abundance of populations of key species.
- Presence of invasive species.
- Land connectivity: well-connected networks of PAs create corridors that allow mobile species to adapt to pressures through migration (Saura et al., 2017).
- Endemicity: biodiversity hotspots cover 17.3% of the planet's land surface (excluding Antarctica) (Sloan et al., 2014) (Fig. 16.3) and present both exceptional endemism and extreme threats to the integrity of their vegetation, making them global conservation priorities (Mittermeier et al., 2011). For instance, fynbos alone represents 1.4% of the world's land surface (Myers et al., 2000). Hotspots are home to 77% of all mammal, bird, reptile, and amphibian species, including 50% of all plant species and 42% of endemic terrestrial vertebrate species, as well as three-quarters of all threatened terrestrial vertebrates (Mittermeier et al., 2011; Gorenflo et al., 2012). Cultural diversity is also high in hotspots, where half of all indigenous languages are found.
- Functional diversity and phylogenetic diversity (Waldron et al., 2020).

5 Human activities and their impacts

The benefits of xeric shrublands are indisputable, providing grazing for livestock, flood control, soil erosion mitigation (Hao and Yu, 2018), and contributing to carbon

TABLE 16.2 Percent of terrestrial, marine and freshwater ecoregions reaching 30% coverage by protected areas and other effective area-based conservation measures (OECMs).

Biogeographical unit	Realm	PA (%)	OECM addition (%)	Year
Ecoregions	Terrestrial	27.51	1.65	2024
Ecoregions	Marine	24.57	0	2024
Ecoregions	Freshwater	20.94	0.67	2024
Biomes	Marine	0	0	2024
Biomes	Freshwater	0	0	2024
Biomes	Transitional	0	0	2024
Provinces	Marine	22.58	0	2024
Provinces	Pelagic	2.70	0	2024

Source: UNEP-WCMC and IUCN, 2024. Protected Planet Report 2024. UNEP-WCMC and IUCN, Cambridge, United Kingdom; Gland, Switzerland. https://livereport.protectedplanet.net/.

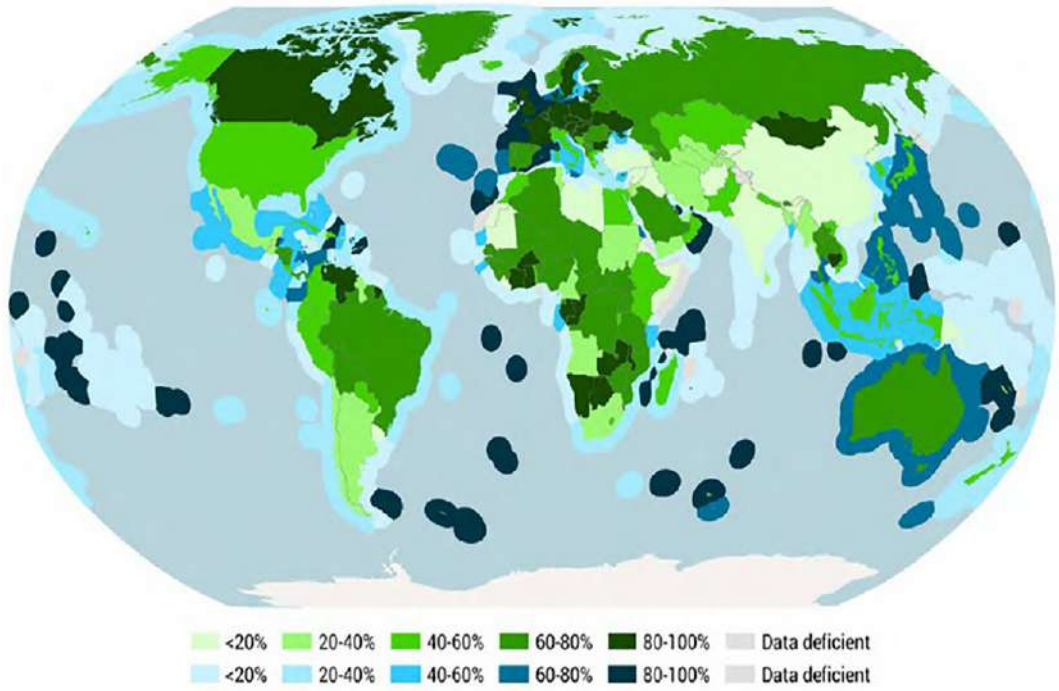

FIG. 16.3 Species protection index (SPI) at the national level. The index is shown as a percentage of the habitat-suitable ranges necessary to safeguard assessed species that is within protected areas and other effective area-based conservation measures (OECMs). SPI is shown for the terrestrial and inland waters realm and for the marine realm. *Source: UNEP-WCMC and IUCN, 2024. Protected Planet Report 2024. UNEP-WCMC and IUCN, Cambridge, United Kingdom; Gland, Switzerland. https://livereport.protectedplanet.net/. Calculated by Map of Life (mol.org) in association with the GEO Biodiversity Observation Network and the Half-Earth Project.*

sequestration for global warming mitigation (Mills et al., 2005). Understanding the pattern of human impacts on the natural environment is crucial for targeting management strategies and biodiversity conservation actions (Venter et al., 2016).

The ecological integrity of xeric shrublands is threatened by various interrelated factors, particularly due to population growth (resulting in increased demand for resources), increased fire frequency, habitat loss and fragmentation, disturbance of shrub-influenced communities, invasive species,

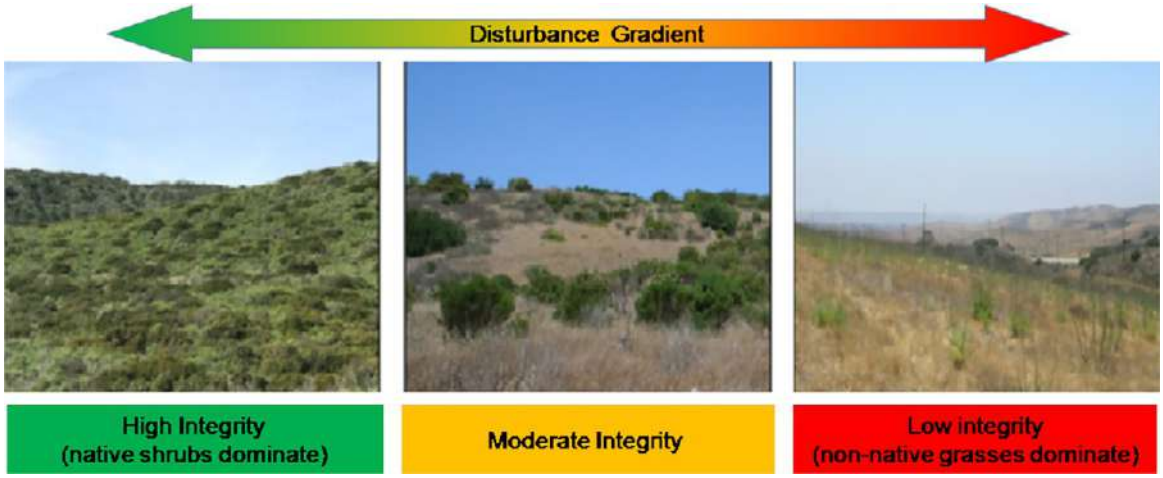

FIG. 16.4 Integrity classes are defined by plant functional group composition (proportion of shrub and annual grass). Visual distinction between these functional groups makes them readily understood by nonspecialists. *(Photo credits: Dawn M. Lawson in Lawson, D.M., Keeley, J.E., 2019. Framework for monitoring shrubland community integrity in California Mediterranean type ecosystems: information for policy makers and land managers. Conserv. Sci. Pract. 1(11), e109. https://doi.org/10.1111/csp2.109.)*

land clearing, uncontrolled grazing, and climate change (Stylinski and Allen, 1999; Thuiller et al., 2005; Butchart et al., 2010; Ceballos et al., 2015; Kobayashi et al., 2019; Salgado-Luarte et al., 2019; Bär Lamas et al., 2021; Zubkova et al., 2022; Macheroum and Chenchouni, 2022). These threats compromise the ability of shrublands to provide essential ES. The temperate grasslands, savannas, and shrublands biome is described as the most threatened in the world (Hoekstra et al., 2005) (Fig. 16.4).

6 Operational and potential monitoring tools to conserve or to recover xeric shrublands

A strategic plan adopted during the 10th Conference of the Parties to the Convention on Biological Diversity in Nagoya in 2010 stipulated that "by 2050, biological diversity must be valued, conserved, restored, and wisely used, sustaining ES, maintaining a healthy planet, and providing essential benefits for all people" (Rodary et al., 2016).

Similar to other biomes, each with its unique characteristics, the conservation of xeric shrublands necessitates continuous monitoring for more effective measures. According to Venter et al. (2016), considerable efforts have been made over the last two decades to assess human impact on the environment by mapping and measuring their influence on species and habitats at different spatial scales. In an analysis of 52 African studies published between 2005 and 2014, Wangai et al. (2016) found that 62% of the work in Africa targeted wetlands and water catchments, excluding biodiversity-rich terrestrial ecosystems. There are several monitoring tools for conserving xeric shrublands, with the most crucial ones including:

6.1 The world database on protected areas (WDPA)

This database is a fundamental tool for monitoring progress toward conservation goals, containing spatial and attribute information from governments and nongovernmental organizations on over 120,000 national and international PAs (Coad et al., 2009). The expansion of PAs is a primary strategy and an essential tool for achieving international biodiversity targets (Watson et al., 2014). Iannuzzi et al. (2020) propose private PAs as a promising conservation tool for governance to complement state-managed PAs.

6.2 Tourism and ecotourism

Ecotourism is a potentially powerful tool for biodiversity conservation in PAs and beyond. According to Halpenny (2010), in addition to tourism's contributions to conservation finances and the local economy, transformative visitor experiences in PAs can cultivate proenvironmental behaviors and garner public support for conservation. This study was conducted on the Galapagos National Park (Cajiao et al., 2020), employing terms like "monitoring" or "monitoring program," where systematic data collection is carried on the characteristics of tourist numbers (quantity and distribution of use), tourist behavior, and the conditions of natural resources on or near tourist sites where the pressure of use is likely to compromise resource conditions.

6.3 Promoting conservation agriculture

In arid regions, conservation strategies should advocate for ecologically sensitive agricultural practices such as

agroforestry, pastoralism, and conservation agriculture as alternatives to intensive agricultural practices (Chenchouni and Neffar, 2025). Local knowledge and indigenous institutions can play a crucial role in effectively implementing these practices. In arid landscapes, managing biodiversity at a landscape scale is only feasible when farming systems are regarded as conservation tools. In this context, conservation becomes a tool for the sustainable development of agriculture (Davies et al., 2012; Zhang et al., 2023). The use of mobile technology and online tools has facilitated the timely acquisition of relevant data, mitigating constraints associated with data collection and systematization, which are essential in monitoring implementation.

6.4 Remote sensing and satellite imagery

Rapid technological advances in remote sensing have expanded its application across various scientific disciplines related to the assessment of natural resources. In recent years, numerous remote sensing satellites and sensors with varying spatial, spectral, and temporal resolutions have been developed, offering high-quality spatial and temporal data at different scales (Assmann et al., 2019). In comparison with traditional methods, remote sensing enables the estimation of vegetation state over larger areas with increased accuracy and efficiency (Stojanova et al., 2010), optimizing the assessment of vegetation response to practices such as prescribed burning (Chuvieco et al., 2019).

Li et al. (2017) employed airborne light detection and ranging (Lidar) to model aboveground biomass in a xeric shrub-steppe landscape in the Great Basin, USA, which is crucial for estimating fuel loads and measuring carbon storage in dryland ecosystems (Angell et al., 2001; Shrestha and Stahl, 2008). According to Lippitt et al. (2018), the use of remote sensing with moderate spatial resolution imagery has facilitated a spatially comprehensive, cost-effective, and repeatable approach to monitoring changes in plant communities and the establishment and expansion of nonnative herbaceous cover in sensitive habitats in Californian shrublands.

Leveraging these advantages, many studies have utilized spectral indices (e.g., NDVI) derived from remote sensing to estimate vegetation cover (Blanco et al., 2016; Marie et al., 2022), monitor disturbances (e.g., forest fires, soil degradation), and identify potential habitat losses (Mermoz et al., 2005). Satellite imagery provides a large-scale perspective of xeric shrublands, enabling the monitoring of changes in vegetation cover, land use, landscape dynamics, and the spread of fires.

While methods for collecting data on species distribution in the field are time-consuming and influenced by sampling design, satellite remote sensing is considered suitable for mapping and monitoring species distribution (Rocchini et al., 2016). However, it is important to note that these tools have limitations in spatial and spectral resolution (Gillespie et al., 2008).

6.5 Geographic information system (GIS)

GIS integrates spatial data with ecological information to generate detailed maps, enabling the analysis of the distribution and condition of xeric shrublands. It aids in identifying areas at risk, assessing fragmentation, and devising effective conservation strategies. Ribeiro and Ribeiro (2016) utilized participatory GIS procedures (PGIS) to map the perceptions of visitors, residents, and managers in an urban park in Rio De Janeiro.

6.6 Use of drones

Multispectral images acquired through drones play a crucial and complementary role in vegetation monitoring, particularly in heterogeneous areas like semiarid shrublands (Díaz-Delgado and Mücher, 2019). The application of remotely piloted aircraft systems (RPAS) or drones extends to ecological studies, forestry, ecosystem restoration, and environmental resource assessment research, driven by the ability to obtain extensive data at high spatial resolution for detailed mapping of the structural and functional attributes of vegetation (Chen et al., 2018; Assmann et al., 2019; Díaz-Delgado and Mücher, 2019; Gonzalez-Musso et al., 2020).

Using drones for field surveys and vegetation monitoring yields precise data on the species composition of xeric shrublands, population trends, and habitat conditions. Despite its quick and easy acquisition of large datasets at high spatial resolution, there are instances, such as in the assessment of vegetation cover in low-cover shrub areas after prescribed burns in semiarid Mediterranean shrublands (Sierra de Filabres; Almería; southern Spain), where drones seem to underestimate plant cover. This discrepancy may arise from the challenge of detecting very small and scattered plants by drone sensors, which are more effectively identified through visual assessment (Pérez-Luque et al., 2022). As noted by van Blerk et al. (2022), the use of drones may not be conducive to detecting population-level changes at an ultrafine scale and details at the species level.

6.7 Monitoring of climate

Monitoring weather patterns, temperature, and precipitation trends is imperative for understanding how climate change impacts xeric shrublands. Long-term climate data assists in predicting future impacts and guiding conservation efforts. Addressing the effects of climate change, including extreme weather events like storms, floods, droughts, and sea-level

rise, can reduce greenhouse gas emissions from deforestation and land degradation. Protected areas play a crucial role in enhancing the long-term resilience of ecosystems (Juffe-Bignoli et al., 2014; Zhang et al., 2023).

6.8 Biodiversity assessment

Biodiversity assessments are crucial for unveiling changes in habitat conditions and ecosystem functioning in Mediterranean regions, resulting from anthropogenic or natural processes. One of the most widely adopted approaches to assess biodiversity patterns and processes is examining species diversity at both biogeographic and ecological scales (Chiarucci et al., 2017).

The relationships between species diversity and climatic factors such as temperature and precipitation along gradients have been extensively studied in various regions, including Algeria, Spain (Baudena et al., 2015), and West African countries bordering the Sahel (Da et al., 2018). Overgrazing impacts on species diversity have also been explored (De Bello et al., 2006; Macheroum and Chenchouni, 2022).

Conservation action priorities are often determined using tools like the International Union for Conservation of Nature (IUCN) Red List (Duffus et al., 2023). Employing techniques such as camera traps, acoustic monitoring, and DNA analysis (Yin et al., 2022; Mugerwa et al., 2024), researchers can assess species richness, population size, and distribution in xeric shrublands. These data contribute to developing targeted conservation plans for specific species of concern.

6.9 Involving communities in monitoring

Local communities, armed with indigenous knowledge, play a pivotal role in decisions contributing to conservation initiatives (Zhang et al., 2023). They can complement scientific monitoring efforts for conserving xeric shrublands through citizen science programs, enhancing their environmental knowledge and emphasizing the value of their resources (Hart and Sumner, 2020). Understanding the red list for each region is an effective means of conservation, offering an opportunity to inform people about the status of species and prioritize those at risk for conservation (Rodrigues et al., 2006; Meuser et al., 2009).

7 Conclusions

Globally, biodiversity faces numerous threats, primarily of human origin. However, arid zones exhibit a distinctive characteristic as they crucially support the livelihoods of a significant portion of the global human population (more than 38%). These populations rely on arid zones for food security, emphasizing the imperative need for biodiversity conservation and sustainable resource management. Given the undeniable role of these areas in providing ES, their development should be well-informed, considering precise geographical boundaries and acknowledging their specific and endemic biodiversity.

The conservation areas and tools elucidated in this document serve as a roadmap, guiding us toward more effective actions. These actions include fostering good governance and legislation, promoting environmental education, and adopting a participatory approach that involves local populations in socioeconomic development while safeguarding their environmental resources. Recognizing the substantial challenges faced by arid zones, such as high poverty rates and low investment, addressing these challenges necessitates concerted efforts at both local and international levels. The integration of these approaches is pivotal for the successful conservation and sustainable development of arid zones.

Conflicts of interests

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Section F

Savannas

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Chapter 17

Land degradation and its associated effects on dung beetle species in African Savanna

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1 Introduction

The African savanna is a highly important ecosystem that encompasses a significant portion of the Earth's land surface. It consists primarily of a continuous grass layer (Fig. 17.1), interspersed with woodland fragments and single trees (Sankaran et al., 2005). The structure of the savanna is influenced by rainfall patterns, soil fertility, and disturbance regimes, such as fire and herbivory (Sankaran et al., 2005). This ecosystem is crucial for human economies, as it supports a large proportion of the world's human population, livestock, and wild herbivore biomass. Additionally, savannas play a key role in carbon and nutrient storage, as well as regulating hydrological cycles (Scholes and Archer, 1997; Jackson et al., 2002; House et al., 2003; Beerling and Osborne, 2006). The African savanna biome includes various tropical ecosystems characterized by a continuous grass layer alongside trees, under distinct wet and dry seasons (Justice et al., 1994). These diverse ecosystems cover approximately half of sub-Saharan Africa (Fig. 17.2) and are broadly categorized into moist-dystrophic/broad-leaved savannas and arid-eutrophic/fine-leaved savannas (Justice et al., 1994). The former occurs in plateau areas with infertile soils derived from igneous rocks, under rainfall regimes above 600 mm. The latter is found in lower lying regions with nutrient-rich soils and rainfall ranging from 400 to 800 mm. Savannas in southern and central Africa are typically characterized by woodland interspersed with grassland along drainage lines, while drier acacia-studded plains dominate the fine-leaved savannas (Du Toit and Cumming, 1999).

African savannas support a higher diversity of ungulate species, including mega-herbivores, compared to other biomes (Du Toit and Cumming, 1999). The ecosystem is also home to various carnivores, rodents, birds, and approximately 1750 plant species (Sankaran and Ratnam, 2013). Among the notable species found within the savanna biome are wildebeest, zebra, buffalo, warthog, elephant, rhino, lion, and giraffe (Du Toit and Cumming, 1999; Fryxell et al., 2004; Shorrocks, 2007; Sankaran and Ratnam, 2013; Hempson et al., 2015).

The vegetation structure of the savanna is strongly influenced by fire, herbivory, soil, and rainfall (Fig. 17.3). Consequently, changes in land use and climate pose significant threats to this dynamic ecosystem. Therefore, it is crucial to prioritize the preservation, existence, and distribution of savannas. Sustainable conservation and restoration programs should be implemented in disturbed areas to ensure the continuous provision of ecosystem services to both the human population and wildlife.

2 Biodiversity

The African savanna is renowned for its diverse range of wildlife, from insects to large animals, and boasts a greater variety of ungulate species than any other biome (Du Toit and Cumming, 1999). The mammal population in the African savanna includes both small herbivores such as dik-dik (*Madogua* spp.), impala (*Aepyceros melampus*), and rodents like the springhare (*Pedetes capensis*), as well as larger herbivores like the African elephant (*Loxodonta africana*), African buffalo (*Syncerus caffer*), hippopotamus (*Hippopotamus amphibius*), and giraffe (*Giraffa camelopardalis*). Predatory carnivores in the savanna include the African wildcat (*Felis lybica*), genet (*Genetta* spp.), civet (*Civettictis civetta*), leopard (*Panthera pardus*), spotted hyena (*Crocuta crocuta*), lion (*Panthera leo*), African wild dog (*Lycaon pictus*), and cheetah (*Acinonyx jubatus*).

A wide variety of bird species also found in the savanna, such as weavers, vultures, ostriches, and hornbills, along with

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FIG. 17.1 Variation in tree cover across different savanna sites: (A) open savanna with a short-grass understory in Samburu National Park, Kenya; (B) open savanna with a mid-to-tall-grass understory in Kruger National Park, South Africa; (C) wooded savanna in Kruger National Park; and (D) wooded savanna with tall-grass understory in Mudumalai Tiger Reserve, India. *(Photo credits: Courtesy of Edward February.)*

notable reptiles and amphibians like snakes, tortoises, and lizards (Scholes and Walker, 1993; Du Toit and Cumming, 1999; Shorrocks, 2007; Sankaran and Ratnam, 2013; Hempson et al., 2015). Insects, including butterflies, ants, beetles, termites, grasshoppers, aphids, and bees, are present in large numbers throughout the savanna. The average plant richness in African savannas is estimated to be around 1750 species, with significant variation across different types of savannas. Plant richness tends to increase from arid to mesic systems (Sankaran and Ratnam, 2013). The miombo savannas of East Africa stand out with a staggering 43,000 species, while the Somalia Maasai region is home to approximately 2500 species, half of which are endemic, making it one of the most biodiverse grassland areas in the world (World Conservation Monitoring Centre, 1992; Sankaran and Ratnam, 2013). Tragically, the African savanna ecosystem is currently facing severe degradation, habitat loss, and climate change, which have direct consequences for the mammals and insects that rely on it. Dung beetle insects, for example, play a crucial role in the biodegradation of organic materials through their feeding activities. These beetles primarily consume mammalian dung but are also capable of utilizing carrion, rotting fruits, fungi, and decaying plant matter. By engaging in feeding activities, dung beetles contribute significantly to nutrient cycling in the savanna ecosystem, particularly during the growing season when nutrients are rapidly recycled and leached from the rhizosphere. This vital recycling process facilitates the germination and growth of plants, including grasses and trees, through the dispersal of seeds. Dung beetles have been recognized as valuable indicators of diversity, endemism, and environmental quality in various biogeographical, conservation, and agro-ecosystem studies. They are instrumental in burying dung, recycling nutrients, controlling dung-breeding pests, bioturbating soils, and dispersing seeds (Bornemissza, 1976; Halffter and Favila, 1993; Andresen, 2002; McGeoch et al., 2002; Davis et al., 2004; Spector, 2006; Nichols et al., 2007, 2008; Brown et al., 2010).

3 Role of dung beetles in the African savanna ecosystem

The African savanna biome is recognized as the sole habitat where a diverse range of mammal species continues to thrive (Hanski and Cambefort, 1991). These mammals serve as a crucial resource for dung beetles and exhibit a close

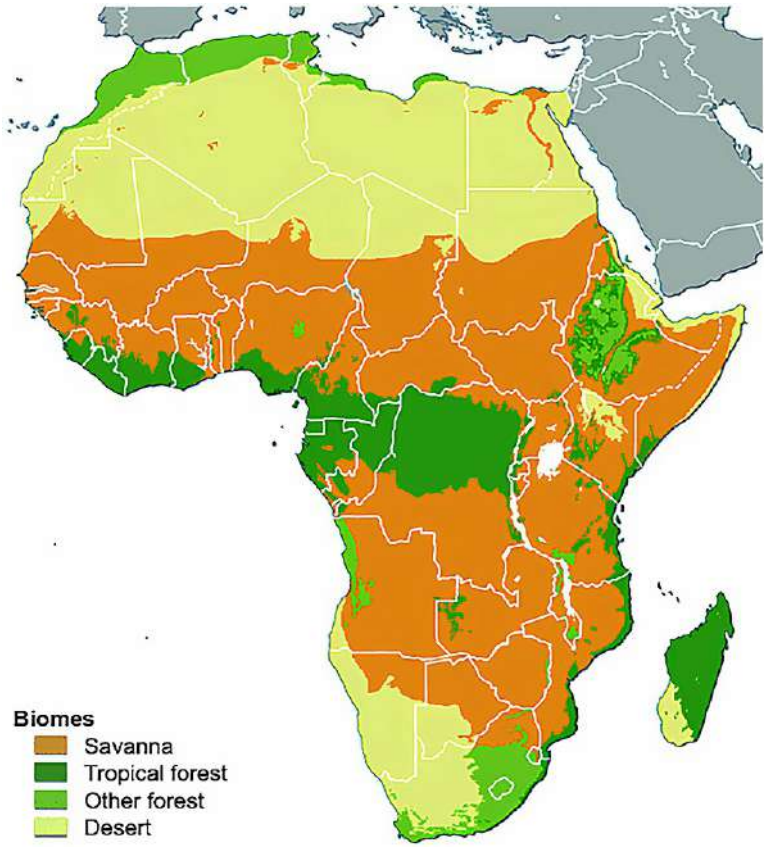

FIG. 17.2 A map of African biomes indicating savanna coverage. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: Falkenmark, 2018)*

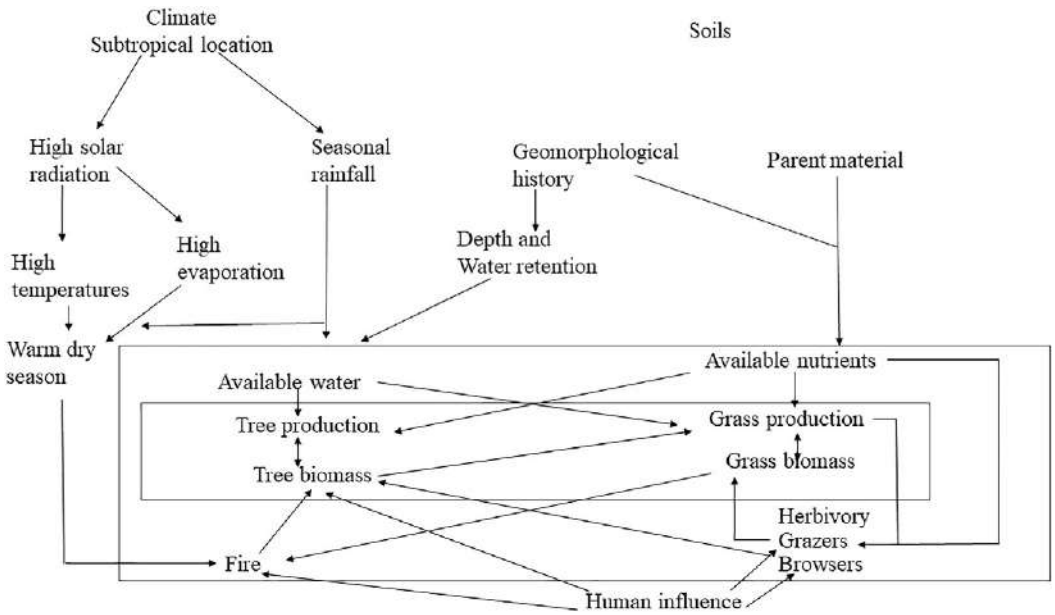

FIG. 17.3 Key driving variables for savanna structure. *(Photo credits: Scholes, R.J., Walker, B., 1993. An African Savanna: An Overview of Savanna Ecology. Cambridge University Press. https://doi.org/10.1017/CBO9780511565472.017.)*

association with one another. The African savanna has the highest diversity of dung beetles (Davis et al., 2001; Kunz and Krell, 2011), with an estimated 75 genera and 1500 species (Hanski and Cambefort, 1991). These beetles play a significant role in the functioning of the African savannas; it is estimated that beetles bury a metric ton of dung per hectare per year in African savannas (Hanski and Cambefort, 1991). By feeding on dung, these beetles support seed dispersal, nutrient recycling, parasite control, greenhouse gas reduction, increased grass growth, and soil fertilization (Nichols et al., 2008; Scholtz et al., 2009; Raine and Slade, 2019; Andresen and Urrea-Galeano, 2022). Research conducted by Williams and Haynes (1990) indicates that herbivores utilize only a small portion of the nutrients they consume, with 60%–99% of ingested nutrients being returned to the soil through dung and urine. Consequently, dung beetles play a crucial role in the removal of herbivore-accumulated dung from soil surfaces (Stanbrook et al., 2021a, b). The presence of dung beetles in African savannas primarily depends on rainfall patterns. In regions experiencing two rainy seasons, there are two phenological types of beetles, one with restricted occurrence early in the year and the other with a more continuous presence during both rainy seasons. However, in regions with only one rainy season, the seasonal occurrence of all species is more limited (Hanski and Cambefort, 1991). Another significant factor affecting the occurrence of dung beetles is the presence of large herbivorous mammals, particularly for the larger species of dung beetles. The size of beetles and their species diversity increase in areas with a richer mammalian fauna. For example, large beetles like *Heliocopris*, which are the world's largest dung beetles (Pokorný et al., 2009), demonstrate a positive correlation with the presence of dung from large herbivores such as elephants and rhinos. These large beetles also tend to occur at relatively low population densities (Stanbrook et al., 2021a, b).

Dung beetles have been categorized into four functional guilds based on characteristics such as body size, reproductive strategy, flight activity patterns, and dung removal behavior. These guilds include telecoprids or rollers, paracoprids or tunnelers, endocoprids or dwellers, and kleptocoprids (Doube, 1990; Cambefort and Walter, 1991; Kunz and Krell, 2011). "Rollers" form dung balls and roll them horizontally on the soil surface away from the dung pile then bury it after a certain distance, "Tunnelers" pull dung portions into tunnels dug vertically under or near the dung pat. "Dwellers" do not remove dung from the site of deposition, they live inside it, and "kleptoparasites" steal nests of other beetles, mainly paracoprids and telecoprids (Hanski and Cambefort, 1991).

4 Conservation challenge

Savannas have been identified as ecosystems that are highly susceptible to the impacts of changes in land use, climate, human population pressure, fire regime, habitat loss, and fragmentations (Andersen et al., 2003; Sankaran et al., 2005; IPCC, 2007; Sankaran and Ratnam, 2013; Ford et al., 2016). These changes in land use have primarily stemmed from population growth, urbanization, resource exploitation, and the conversion of land to agro-ecosystems (Fairbanks et al., 2000; Davis et al., 2020). Such changes have a direct effect on the dung beetle community, particularly species with limited ranges and those that exhibit specialized ecological associations (Hanski and Cambefort, 1991). Over time, dung beetles have experienced a rapid decline in these ecosystems due to alterations in agricultural practices, including intensification and reduced pasture grazing (Jankielsohn et al., 2001; Hutton and Giller, 2003), as well as the loss of their natural habitats (Roslin and Koivunen, 2001; Nichols et al., 2007). Furthermore, the excessive use of anthelmintics, such as ivermectins, has further contributed to the decline of dung beetles (Beynon et al., 2012; Manning et al., 2018; Lumaret et al., 2022). Additional factors impacting dung beetles include fire (Nyundo and Ngongolo, 2015; Smith et al., 2019) and climate change (Davis et al., 2020). It is important to note that the sensitivity of dung beetles to land-use change varies among species, with different levels of tolerance exhibited (Scheffler, 2005; Nichols et al., 2007; Louzada et al., 2010).

5 Agriculture

The majority of savanna regions across the globe have undergone considerable alteration as a result of subsistence agriculture (Andersen et al., 2003). Agricultural production has been identified as a primary contributor to the loss of biodiversity (Foley et al., 2005). This is particularly evident in the conversion of native vegetation communities to intensive agricultural systems, which has rapidly transformed landscapes and accelerated the decline of biodiversity (Foley et al., 2005). For instance, growth in agricultural activities can be observed through the notable expansion of cultivated land surrounding Kenya's Masai Mara Reserve. This expansion is evident as the cultivated land area has substantially increased from 4875 ha in the mid-1970s to over 50,000 ha two decades later, as documented by Serneels et al. (2001); this expansion has been linked with decreases in the populations of large mammals, as reported by Ogutu et al. (2011).

Global food demand is projected to increase in the coming years, and the conversion of land for agricultural production will likely continue to have significant implications for biodiversity, particularly in developing countries that experience high population growth rates, increasing access to markets, and expanding agricultural activities (Geist and Lambin, 2001; Godfray et al., 2010; Kastner et al., 2014). The conversion of landscapes to agriculture also plays a substantial role in the loss of dung beetle

diversity (Jankielsohn et al., 2001; Numa et al., 2012). In savanna ecosystems, the richness of dung beetles is positively influenced by the percentage cover of native vegetation, with the amount of savanna in the landscape serving as a key driver of dung beetle richness (Lascaleia et al., 2018). By reducing the area of native vegetation, the number of dung producers and available resources diminish, thereby limiting the presence of dung beetles that rely on dung for food and reproduction (Hanski and Cambefort, 1991). Additionally, studies have demonstrated that dung beetle richness increases in areas with diverse ungulate populations, which are primarily found in larger patches of savanna supported by cattle ranching and protected areas (Lascaleia et al., 2018). Conversely, agricultural areas offer minimal resources and are thus of little value to dung beetles (Lascaleia et al., 2018). Moreover, the use of fertilizers and pesticides in intensive agricultural production is likely to further reduce the diversity of dung beetles (Hutton and Giller, 2003; Scholtz et al., 2009). A recent study by Nependa et al. (2021) found that dung beetle biomass is lower on farms compared to nonagricultural areas, as farms tend to be dominated by smaller generalist species while larger-bodied dung specialist species, which are known to be functionally superior, are absent (Fig. 17.4). Additionally, the proportion of individuals belonging to species commonly found in both farms and protected areas was physically smaller on farms, indicating reduced body condition. The low biomass of dung beetles in farm areas is influenced by the exclusion of several large-bodied tunneling species, such as *Catharsius ulysses* and *Heliocopris atropos*, as well as a lower abundance of large rolling species like *Pachylomera femoralis* and *Scarabaeus goryi* (Nependa et al., 2021). On the other hand, protected areas harbor a greater number of unique dung beetle species compared to farmlands, particularly larger species such as *Heliocopris andersoni* and *Heliocopris atropos*, which are typically absent on farms (Nependa et al., 2021). However, there are a few dung specialist species, such as *Onitis bilobatus* and *Phalops pyroides*, which do prefer native dung and can still be found on farms (Du Toit and Cumming, 1999; Tshikae et al., 2008).

6 Fire

Fire plays a crucial role in the functioning of savanna ecosystems (Fig. 17.5), not only shaping their structure but also preserving the distinctive biodiversity that characterizes them (Ratnam et al., 2011). However, in recent times, savannas have become the most fire-prone biome worldwide, accounting for approximately 85% of the annual global land area burned (Mouillot and Field, 2005). It is estimated that nearly 35 million ha of savanna vegetation burn each year in Africa alone (Mouillot and Field, 2005). These savanna fires contribute significantly to the release of carbon emissions, emitting approximately 1.67 Gtons of carbon per year, accounting for half of the global carbon emissions resulting from biomass burning (Mouillot et al., 2006). Furthermore, savanna fires are significant sources of atmospheric nitrogen oxides (NO_x) and account for over half of the NO_x emissions released globally by wildfires (Beerling and Osborne, 2006). The loss of savanna habitats has been estimated to be twice as high as that of rainforests (Grace et al., 2006). The heightened incidence of fires in African ecosystems appears to be driven by poachers, who set fires to facilitate access to tall grass and attract wildlife to the resulting green flushes; in addition to that, land clearing and livestock management are also responsible for igniting the vast majority of savanna fires. The increased frequency of fires may contribute to the loss of essential elements in forage, such as nitrogen and phosphorus, and promote the growth of low-nutritive-quality forage species, such as *Themeda triandra*, which could have detrimental effects on herbivore biomass (Anderson et al., 2007). Moreover, burning significantly impacts the abundance of beetles in grasslands (Nyundo and Ngongolo, 2015) since they are affected by direct heat and ash-related mortality (Smith et al., 2019). In addition, fire intensifies competition for forage among grazers, including wild ungulates and cattle, and indirectly affects dung beetles through variations in dung quality resulting from the quantity and quality of forage consumed by these animals (Odadi et al., 2011).

7 Climate change

The relationship between savanna vegetation structure and water availability is particularly strong. Therefore, future changes in rainfall, both in terms of total amount and seasonality, are likely to have significant effects on the dynamics of woody vegetation in savannas (Sankaran and Ratnam, 2013). Climate models predict that future rainfall patterns in many savanna regions around the world may either increase or decrease (IPCC, 2007). Furthermore, there is also a likelihood of changes in rainfall distribution throughout the season, with fewer but more intense rainfall events and longer dry periods in between (IPCC, 2007).

As a result of these changes, savannas may undergo shifts toward more wooded states in areas experiencing increased precipitation, and toward more open states in regions with decreased precipitation or more intense rainfall events (Sankaran et al., 2005, 2008; Good and Caylor, 2011). Factors such as increased rainfall, elevated CO_2 levels, and nitrogen deposition are expected to promote shifts toward more wooded states, whereas lower rainfall, more intense rainfall events, and longer intraseasonal droughts may cause shifts toward more open states. Moreover, the increased concentration of atmospheric CO_2 is likely to provide a competitive advantage to C3 trees and shrubs over C4 grasses. This could further contribute to a shift toward more wooded states in savanna (Sankaran and Ratnam, 2013).

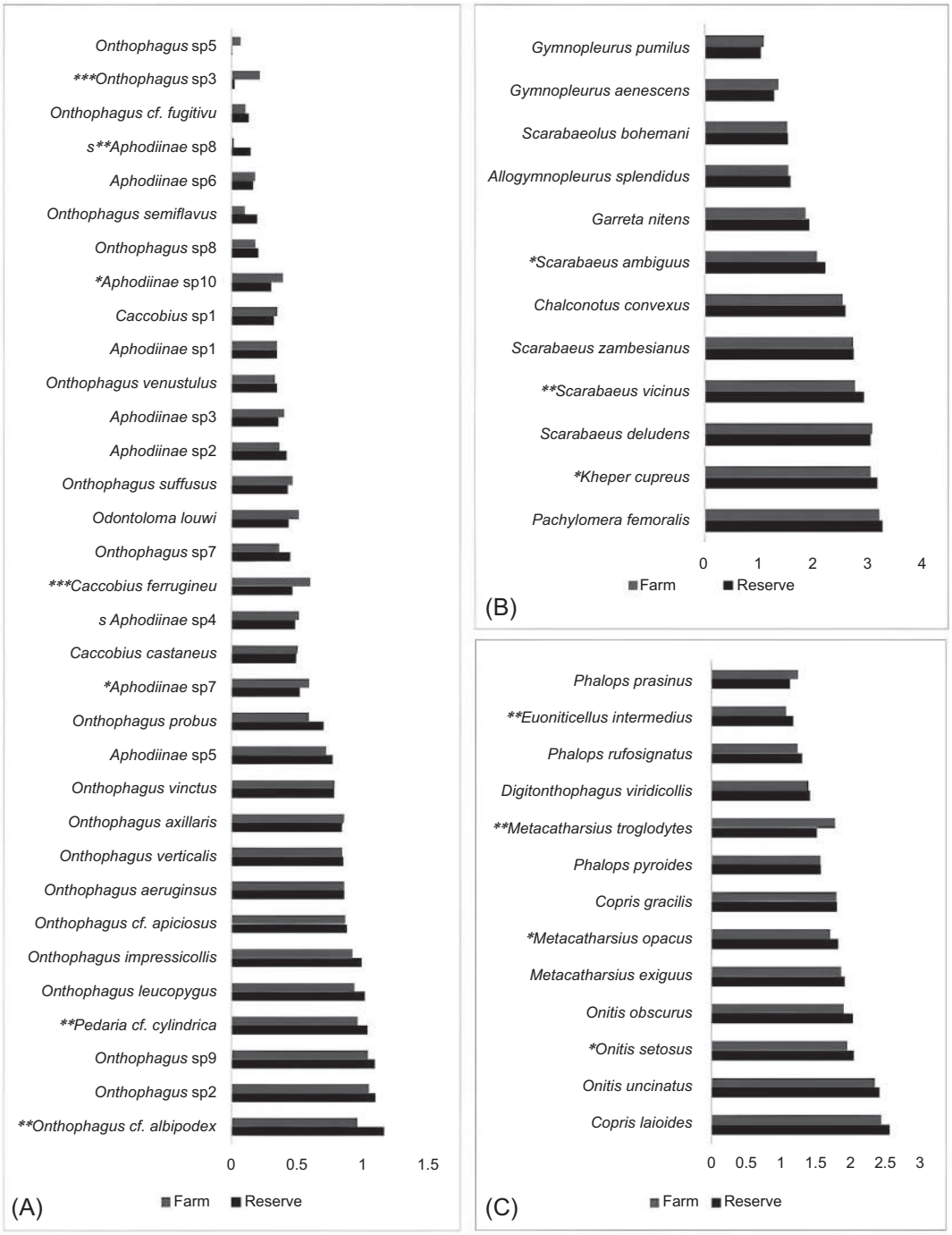

FIG. 17.4 Bar graphs displaying the mean difference in dung beetle mass on farms (gray) and PAs (black). The *x*-axis represents the log of the mean mass (mg) of the species and the dung beetle species are listed on the *y*-axis. Only those species that were present in both farms and PAs are shown here. Grouping of species for bar graphs is according to nesting type: a = dwellers, b = rollers, and c = tunnelers. Those species with significant differences are marked with "*." *(Photo credits: Nependa, H.U., Pryke, J.S., Roets, F., 2021. Replacing native mammal assemblages with livestock in African savannahs, impacts dung beetle diversity and reduces body size. Biol. Conserv. 260, 109211. https://doi.org/10.1016/j.biocon.2021.109211.)*

It is worth noting that temperature, rainfall, and rainfall seasonality are the primary drivers influencing the occurrence of dung beetle species. A recent study conducted in South Africa by Davis et al. (2020) predicts that rising temperatures due to climate change could favor the expansion of dung beetle species adapted to northeast regions, while potentially negatively impacting cooler-adapted species found in the southwest. This shift in species

FIG. 17.5 A ranger lights a management fire in the Serengeti ecosystem, Tanzania, to control the growth of the bush. *(Photo credits: Colin Beale/The Conversation.)*

abundance and dung removal is expected to be more pronounced in southwest regions with a greater reliance on summer rainfall.

8 Habitat loss and fragmentation

Land cover change in sub-Saharan Africa is currently experiencing rapid transformations. Desertification is occurring primarily in the northern and more arid areas of savannas, while urbanization is expanding along transportation networks (Western et al., 2009; Vuorio et al., 2014). Additionally, there is a shift from dispersed ranching and pastoralism to more intensive agriculture (Western et al., 2009; Vuorio et al., 2014). These changes have profound effects on wildlife in the savannas. The loss of habitat is contributing to the decline of herbivore populations, including antelope (Ogutu et al., 2011). This decline has further consequences for other species that rely on herbivore dung for their survival and reproduction, such as dung beetles. The expansion and establishment of roads, railways, and pipeline schemes proposed across various African countries (Beale et al., 2013; Caro et al., 2014) have also played a significant role in fragmenting the savanna landscape. Furthermore, these infrastructure developments alter the quality of essential habitat for wildlife. These ongoing changes pose challenges to the preservation and conservation of biodiversity in sub-Saharan Africa.

9 Bush encroachment

Bush encroachment has been acknowledged as a significant issue for various savanna vegetation types since the early 1900s. This problem has become increasingly recognized as a threat to rangelands (Hoffman and O'Connor, 1999; O'Connor et al., 2014). Moreover, it is predicted that the impact of bush encroachment may worsen in the future (Higgins and Scheiter, 2012; Moncrieff et al., 2014) due to changes in fire patterns, grazing pressure from livestock, climate change, loss of browsing herbivores (including megaherbivores), and outbreaks of diseases affecting wildlife and livestock, such as rinderpest. Additionally, extensive harvesting of woody plants for fuel and shifts in human settlement patterns in response to diseases like sleeping sickness contribute to the intensification of this issue (Spinage, 2012; O'Connor et al., 2014). The species primarily responsible for bush encroachment and invasion include *Vachellia hebeclada*, *V. karroo*, *V. nilotica*, *V. tortilis*, *Salvia mellifera*, *Dichrostachys cinerea*, *Rhigozum trichotomum*, and *Terminalia sericea* (O'Connor et al., 2014). Encroachment occurs most rapidly in protected areas compared to commercial ranches and communal properties. For example, Lake Manyara National Park, Tanzania, experienced the highest increase in woody cover between 1985 and 1991, which was attributed mainly to the decline of elephants caused by poaching activities, reducing their population from 6 to 1 elephant per square kilometer in the region (O'Connor et al., 2014).

Consequently, the reduction in the elephant population also impacts the dung beetle community, which relies on dung pats for feeding and reproduction. Jankielsohn et al. (2001) observed that overgrazed habitats with decreased grass cover and increased woody shrub cover can hinder the rolling behavior of small telecoprids and impair their ability to detect dung due to odor attenuation.

10 Poaching

There is a concerning trend of declining wildlife species, and the protected areas established to safeguard these species for future generations are currently facing severe financial challenges (Bruner et al., 2001; Tear et al., 2021). While predators have traditionally been considered to be at the greatest risk of extinction, recent studies have revealed that herbivores exhibit the highest rates of extinction among mammals, birds, and reptiles (Ripple et al., 2015; Atwood et al., 2020). It is estimated that in 2011, more than 25,000 elephants were slaughtered by poachers in Africa (AFC, 2016); in addition to that there is a rapid decline in rhino and buffalo populations in countries like South Africa, Namibia, Kenya, Tanzania, Botswana, and Zimbabwe (Hilborn et al., 2006; Sinclair et al., 2007; Metzger et al., 2010; Herbig and Minnaar, 2018). The loss of large wild herbivores potentially has far-reaching consequences for other species, such as dung beetles that rely on herbivores' excrement for their survival and ecological function (Nichols et al., 2009).

11 Livestock keeping

The transformation of land for livestock grazing has a significant impact on the world's agricultural land, accounting for almost 60% of it. This practice has severe consequences for native biodiversity, particularly in savanna ecosystems where the majority of forage is obtained. Interestingly, these savanna ecosystems support a disproportionately high diversity of native mammal herbivores (Alkemade et al., 2013; Newbold et al., 2015; Veblen et al., 2016; Cardoso et al., 2020; Samways et al., 2020). The abundance of livestock now surpasses that of wildlife in Africa, which has profound implications for both the continent as a whole and the specific savanna habitats (Hempson et al., 2017). The replacement of native mammal populations with higher grazing livestock disrupts the normal functioning of ecosystems, especially in relation to the roles played by arthropod taxa like dung beetles (Nichols et al., 2009; Young et al., 2018; Filazzola et al., 2020). These arthropods have a long coevolutionary history with mammals and are sensitive to changes in both biotic and abiotic conditions (Davis et al., 2008; Nichols et al., 2009; Sole and Scholtz, 2010). The increased use of veterinary parasiticides, which can retain their insecticidal properties after being excreted, poses a significant threat to dung beetles. These residues can cause immediate death or varying degrees of paralysis, leading to reduced feces location ability, decreased female fertility, and substantial mortality in larvae. Consequently, the long-term use of parasiticides results in the decimation of dung beetle populations and an overall decline in their functional efficiency. This decline ultimately leads to the accumulation of dry dung on the ground (Jean-Pierre et al., 2020). Furthermore, the use of veterinary medical products further diminishes the quality of dung produced by livestock (Hammer et al., 2016; Verdú et al., 2018), impacting dung beetle communities (Scholtz et al., 2009). Additionally, livestock trampling negatively affects vegetation and soil, further reducing the richness of dung beetles (Vohland et al., 2005; Tonelli et al., 2017).

12 Fire suppression

Fire has traditionally been a common occurrence in savanna ecosystems, primarily resulting from human activities and practices of indigenous peoples (O'Connor et al., 2014). However, colonial governments have historically held a negative view on fire and enacted legislation against its use. This has remained relevant in present-day savanna countries such as South Africa and Botswana, where prescribed burning is encouraged, especially in protected areas (Jacobs, 2000). Exclusion of fire has significant implications for savanna ecosystems. It contributes to an increase in woody density and can lead to bush encroachment in mesic and moist savannas, as well as being important in arid savannas (Hoffmann et al., 2012). An example of this is observed in the invasion of grassland near Pietermaritzburg, South Africa, following 50 years of fire protection, resulting in the proliferation of both indigenous *Vachellia* and alien acacia species (Titshall et al., 2000). Fire exclusion studies conducted in tropical African savannas have shown that the complete absence of fire can lead to the development of rainforest within just a few decades. Fire plays a crucial role as a buffering mechanism in savannas, preventing the transition to woodland or forest by destroying juvenile trees and shrubs and suppressing the development of taller, fire-resistant plants (Jeltsch et al., 2000). The presence of savannas in southern Africa is strongly linked to the occurrence of fire (Jeltsch et al., 1996, 1998). Transitional changes from savanna to woodland due to fire suppression will favor certain species of dung beetles that are habitat generalists and forest specialists.

13 Pastoralism

Several large herbivore species, such as *Equus quagga* and *Hippotragus leucophaeus*, went extinct in Africa during historical times. These species had limited distributions on the periphery of the savanna biome (Du Toit and Cumming, 1999). The African savannas have experienced significant declines in geographical distribution, population size, and genetic diversity among large herbivores. The primary factor behind these changes is the rapid increase in human populations. This has had a direct impact on animal communities, as indigenous herbivore biomass has been replaced by domestic livestock. Furthermore, the transformation of land through subsistence cropping and desertification has resulted in the loss of spatial heterogeneity and plant cover, which has affected savanna

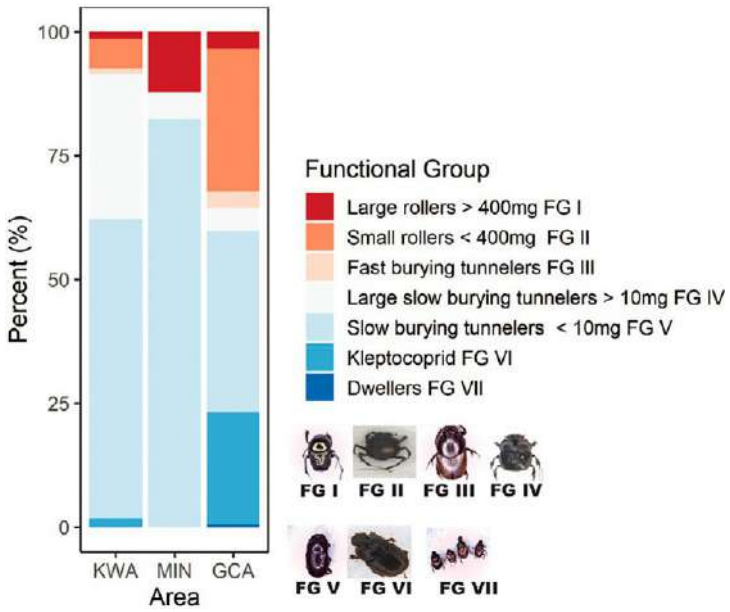

FIG. 17.6 Percentage of each functional group across land-use degradation gradient: Kwakuchinja (KWA) → Minjingu (MIN) → Lokisale Game Controlled Area (Kwakuchincha is a wildlife corridor; Mijingu is a grazing and agricultural area). *(Photo credits: Stanbrook, R., Harris, E., Jones, M., Wheater, C.P., 2021a. The effect of dung beetle size on soil nutrient mobilization in an afrotropical forest. Insects 21(12) 141. https://doi.org/10.3390/insects1202014. Stanbrook, R., Norrey, J., Kisingo, A.W., Jones, M., 2021b. Dung beetle diversity and community composition along a land use gradient in a savannah ecosystem of North Western Tanzania. Trop. Conserv. Sci. 1–15. https://doi.org/10.1177/19400829211008756.)*

vegetation and soils. These disturbances have also led to changes in the structure of dung beetle species assemblages (Fig. 17.6), particularly in areas where there is direct interaction between pastoral communities and wildlife (Stanbrook et al., 2021a, b). The presence of high concentrations of livestock increases the availability of dung resources, leading to closer spatial aggregations of dung and reduced temporal variation in food availability. As a result, there is uneven species abundance, low diversity, and communities dominated by a few species, especially in disturbed areas compared to protected areas. Interestingly, protected areas tend to have a greater abundance of dung beetles, possibly because the presence of various dung types allows for a greater diversity of species (Nependa et al., 2021).

14 Rangeland extension

African savannas exhibit a higher concentration of large herbivores, such as cattle and goats, which are distributed more extensively across various habitats compared to indigenous herbivore communities. This phenomenon is primarily driven by human perception, wherein pastoralists consider stocking rates achieved during above-average rainfall years as optimal and suitable for the system. Consequently, water provisioning infrastructure extends grazing pressure into previously inaccessible areas that lack natural water sources. However, during periods of drought, both livestock and wildlife populations experience more severe and dramatic declines compared to unmanaged systems due to the depletion of "reserve" grazing areas. As landscapes transition from natural systems to farming rangelands, there is a shift from diverse wild mammal species to a limited number of domestic, usually nonnative species (Young et al., 2005; Whipple and Hoback, 2012; Culot et al., 2013; Nependa et al., 2021; Pryke et al., 2022). This replacement process leads to niche simplification (Pryke et al., 2016; Nependa et al., 2021), which in turn can be expected to negatively impact dung beetle diversity (Steed et al., 2022).

15 Conclusion

The decline of dung beetles, particularly large-bodied beetles, in the African savanna is directly attributed to various human activities such as agriculture intensification, poaching, and the current global climate change. This decline will have a significant impact on the ecological functions and ecosystem services provided by dung beetles. A reduction in diversity of dung beetle communities can result in dramatic effects on decomposition rates and nutrient cycling (Scholtz et al., 2009; Lumaret et al., 2022).

Consequently, before making any decisions regarding the realignment of protected areas or proposing alternative protection strategies, it is crucial to have detailed maps of key wildlife habitats both inside and outside protected areas. Ideally, these maps should also consider the calculations of minimum viable areas (Beale et al., 2013). Furthermore, it is important to identify and secure buffer areas that can be utilized during extreme conditions (Beale et al., 2013). This approach would effectively protect wildlife during transitions between the dry seasons and wet seasons, while also

conserving their migratory routes and seasonal ranges outside the protected areas. Enhancing relationships with local communities is crucial to fostering the perception of wildlife as a valuable economic resource. This can be achieved through various initiatives such as organizing school visits to protected areas, implementing reasonable entrance fees, and providing accommodations suitable for different income groups. Similar approaches have been successfully implemented in countries like South Africa through organizations like SANParks (2012). Additionally, allowing adjacent communities to sustainably harvest selected plant resources within designated parks, such as thatching grass, firewood, reeds, and medicinal plants, can further incentivize their involvement in conservation efforts. Moreover, allocating a portion of gate fees to community projects, establishing comanagement agreements, and implementing poverty alleviation projects can contribute to the overall well-being of local communities. Educational programs and employment opportunities can also play a vital role in reducing poaching activities, stabilizing wild animal populations, and increasing the desire to conserve and protect ecosystems, as highlighted by Mbaiwa and Stronza (2011). Enhancements to the control measures against commercial hunting for valuable commodities, which often involve poachers equipped with advanced technology and helicopters from distant locations, are imperative. Although military-style operations can temporarily deter such incursions, they alone cannot prevent their recurrence. To achieve sustainable control over poaching, it is crucial to enforce stringent poaching laws, demonstrate political commitment to allocate sufficient resources, and provide training for rangers (Beale et al., 2013). Furthermore, it is essential to prioritize budget allocation for the funding of patrols and other activities conducted by wildlife wardens (Hilborn et al., 2006; Knapp, 2007; Brashares et al., 2011).

Therefore, it is crucial to promote structural heterogeneity through a controlled restoration program, managing invasive species to prevent bush encroachment, implementing planned prescribed and rotational burning practices, restoring habitats, and effective management of livestock is essential for the sustainable conservation of savanna dung beetles.

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Chapter 18

Savannas in Asia

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1 Introduction

Savanna is a vegetation type that grows under hot, seasonally dry climatic conditions and is characterized by an open tree canopy above a continuous tall grass understory (Bhardwaj, 2019).

The term savanna typically evokes imagery of vast stretches of open grasslands, interspersed with sparse, flat-crowned trees, and grazed by numerous herds of roaming hoofed animals in the minds of many individuals (Huntley, 1982). While the iconic imagery of nutrient-rich East African savannas remains amongst the most enduring in the public consciousness, they fail to capture the diverse array of physical, structural, and functional characteristics that define various savanna ecosystems worldwide (Sankaran and Ratnam, 2013). The term savanna is thought to be derived from a Carib Amerindian word meaning "treeless plain" (Scholes and Walker, 1993; Shorrocks, 2007). Initially documented in Spanish by G.F. de Oviedo Valdes in 1535, as "sabana," the term described land devoid of trees but abundant in the grass of varying heights (Bourlière and Hadley, 1970, 1983). Since then, the term has been widely and diversely used, and extended to include trees (Bourlière and Hadley, 1983). In its modern context, this term describes environments characterized by a consistent layer of grass and scattered patches of trees (Scholes and Archer, 1997).

As a biome, tropical savannas are relatively young when compared to other forested biomes of the tropics (Bourlière and Hadley, 1983). Savannas currently ranks as the Earth's second-largest biome, encompassing approximately 33 million km^2 or nearly 20% of the planet's land area (Scholes and Walker, 1993; Ramankutty and Foley, 1999; Beerling and Osborne, 2006). These ecosystems are widespread across Africa, Asia, South America, and Australia, encompassing over half of the land area of the southern continents (Sankaran and Ratnam, 2013). Savannas play a crucial role in supporting a significant portion of the global human population, as well as serving as vital rangeland for livestock biomass, and are crucial ecosystems in supporting essential ecological functions (Scholes and Archer, 1997; Beerling and Osborne, 2006). Additionally, savannas harbor exceptional diversity and densities of wild herbivores and carnivores compared to other ecosystems on Earth (Sankaran and Ratnam, 2013).

2 Geographic location

Biogeographers have traditionally mapped the occurrence and distribution of biomes across the globe as a function of temperature and precipitation (Whittaker, 1975). In these frameworks, savannas occur between deserts and tropical dry forests, but the transitions between grassland, savanna, and woodland have not been easy to classify. This reflects a long-standing unresolved issue in the understanding of the savanna biome. The limits to its distribution and the nature of the biome boundaries at both the arid and the mesic end are highly variable across the globe, and the dynamics underlying these patterns remain poorly understood. Many landscapes in the mesic end of the savanna biome, for example, southern peninsular India, mainland Southeast Asia from Myanmar through northern Thailand, and the Guinean and Congolian savanna-forest regions in Africa are characterized by a dynamic mosaic of vegetation with open savannas, woodland savannas, and tropical dry forests occurring in a patchwork within the same climatic envelope (White, 1983; Stott, 1991; Ratnam et al., 2011). Thus, climate alone appears insufficient to explain the distribution of the savanna biome (Bond, 2008). Further, the physiognomy of anthropogenic savannas is very often similar to that of natural ones, complicating efforts to map the "natural" occurrence of savannas (Bourlière and Hadley, 1970).

Extensive tropical savannas, largely characterized by C_4 grasses, are prevalent within the latitudinal range of 0°–30° north and south of the equator (Bond, 2008; Edwards et al., 2010). These ecosystems are distributed across areas with diverse soil compositions and precipitation patterns (Hunyley, 1982; Shorrocks, 2007). Savannas typically experience a pronounced dry season lasting between 2 and 9 months each year, during which fires are

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commonly observed (Stott, 1988; Scholes and Walker, 1993; Lloyd et al., 2008). With this caveat, depending on the continent, rainfall in the wet season can range from 200 to 2500 mm (Stott, 1988; Lehmann et al., 2011).

Savannas exhibit significant variability in their physical and structural characteristics throughout their geographical distribution. They encompass a gradient from nearly pure grasslands to closed woodlands (Fig. 18.1), exhibit differences in the characteristics of dominant trees (fine-leaved vs broad-leaved), herbaceous vegetation (tall vs short grasses, vegetated vs bare patches), plant life-history characteristics (deciduous vs evergreen trees, annual vs perennial grasses), tree spatial patterns (random, regular, or clumped), and plant and soil nutrient status (nutrient-poor or dystrophic savannas vs nutrient-rich or eutrophic savannas) (House et al., 2003).

This inherent variability in savannas across their distributional range can pose problems when it comes to characterizing and mapping savannas and developing a synthetic understanding of their functioning. As Lamotte and Bourlière (1983) noted: "… there is no such thing as a "typical" savanna ecosystem. Rather, there is a gradient of related ecosystems ranging from open woodlands to almost treeless "steppes." However, these systems all share several common features which make them similar to each other rather than different." In the following sections, we focus on the commonalities of these systems in terms of the different driver variables, the interactions amongst which lead to the observed variability in their structure, composition, and functioning.

3 Biodiversity

Savannas are typically not recognized as ecosystems with high biodiversity compared to rainforests and coral reefs; however, these uncomplicated environments are unique structures supporting numerous distinct species and communities (Sankaran and Ratnam, 2013). Although comprehensive estimates of biodiversity at the biome level are currently not readily available for Asian mixed tree-grass ecosystems, an analysis of localized studies reveals notable patterns in floral diversity (Sankaran and Ratnam, 2013). Similar to the trend observed in African savannas, the diversity of woody plants in the Indian subcontinent appears to have escalated from arid to mesic environments.

FIG. 18.1 The Kaziranga National Park in India. *(Photo credits: Matyas Rehak.)*

The species richness of trees and shrubs varies from 20 to 35 in dry thorn forests, expanding to approximately 60–80 species in dry deciduous forests (Sukumar et al., 1992). Compared to woody plant species, the herb and grass layers in these ecosystems exhibit significantly higher species richness. For instance, a study on 40 savanna-grassland communities in southern India by Sankaran (2009) identified 278 herb and grass species in the understory, with a majority of species having narrow distribution ranges. Similarly, Singh et al. (2010) found 1011 species along the tropical dry scrub forests-grassland continuum in central India, with the *Poaceae* family, consisting of 112 species of grasses, being the most diverse group. In the dipterocarp savanna forests of Thailand, Stott (1990) documented a notable diversity of understory species, including numerous geophytes.

In the realm of mammalian fauna, Asian savanna habitats exhibit a less diverse array of ungulates and carnivores compared to their African counterparts. However, species such as the Greater one-horned rhinoceros and tiger, exclusive to the Asian region, face significant conservation challenges due to the diminishing size of their habitats (Sankaran and Ratnam, 2013). Furthermore, the avian population in Asian savannas, though distinct, is relatively less diverse in species richness in comparison to African savannas (Fry, 1983).

4 Conservation areas and their ecological indicators

Unlike the extensively researched African savannas, Asian savannas have received comparatively less attention, resulting in a limited understanding of the distribution and dynamics of the various savanna types in the area. This knowledge gap can be attributed to various contributing factors.

In contrast to Africa, where nearly half of the land is arid to semiarid and characterized by savanna landscapes, a significant portion of continental South and Southeast Asia experiences an annual rainfall exceeding 800 mm, resulting in forests and savanna woodlands dominating this region (Sankaran and Ratnam, 2013). Additionally, historically, the primary focus on the vegetation in this area stemmed from timber and forestry activities during the early colonial era. Biogeography and historical factors have combined to create a legacy of predominantly focusing on forests to understand and explore vegetation within this area. Most early regional vegetation classification schemes such as those by Puri (1960) and Champion and Seth (1968), which continue to be widely cited today, primarily focus on categorizing the "forest types" of South and Southeast Asia, largely neglecting the inclusion of savanna ecosystems. Numerous vegetative formations found in the Indian subcontinent exhibit characteristics of savannas, featuring a predominance of C_4 grasses in the understory alongside woody tree or shrub elements. Despite these attributes, they are commonly classified as forests (Sankaran and Ratnam, 2013). Examples of such formations include the sparsely vegetated "thorn and scrub forests" observed in the arid and semiarid zones of western India, as well as the denser "mixed and dry deciduous forests" present in the more humid regions of peninsular India as described by Champion and Seth (1968). Similarly, in the classification systems of Southeast Asia, the predominant vegetation type in the lowland areas stretching from northern Burma to Thailand, Laos, Cambodia, and Vietnam is a dry deciduous dipterocarp forest with sparse ground cover of grasses (Sankaran and Ratnam, 2013). This ecosystem is commonly known as "forests claire," "open forest," or "savanna forest" (Stott, 1990). Stott (1991) pointed out that despite the striking resemblance of these dry dipterocarp forests to Africa's miombo woodlands they have been largely overlooked by scholars studying savanna vegetation and they remain a neglected subject, even in modern texts.

4.1 Dry thorn forests

These are open tree-grass ecosystems characterized by trees of height ranging from 6 to 9 m. Associated with the trees, there is a diverse lower story composed of small trees and shrubs, while the presence of thin grass is predominant, particularly flourishing during the wet season (Sankaran and Ratnam, 2013). However, a substantial portion of the ground is left exposed during the dry season. The southern forms of this vegetation type extend across the dry tracts of peninsular India and along the lee of the Western Ghats, experiencing an average annual rainfall between 500 and 800 mm (Sankaran and Ratnam, 2013). The *Acacia* genus, known for its diverse species, is notably prevalent in association with plants such as *Ziziphus*, *Capparis*, and succulent *Euphorbia* species. In the understory, perennial grasses such as *Themeda quadrivivalis*, *Themeda triandra*, *Thelepogon elegans*, and *Sehima nervosum* are frequently found. In the northern regions, which stretch across the semiarid areas of central India to the east of the Thar Desert, the vegetation is characterized by shorter vegetation compared to the southern thorn forests, where species like *Acacia*, *Mimosa*, and *Prosopis* dominate. The grasses in these areas are primarily annual rather than perennial, with *Lasiurus hirsutus*, *Panicum turgidum*, *Sporobolus marginatus*, and various *Cenchrus* species being commonly observed (Sankarn and Ratnam, 2013). These ecosystems bear structural and compositional similarities to the "grass and shrub" savannas found in arid regions of Africa.

4.2 Dry deciduous forests

These woodlands, many of which have grassy understories, occur throughout the Indian peninsula with typical mean

annual rainfall ranging between 1000 and 1400 mm, accompanied by prolonged dry periods lasting 5–7 months (Sankaran and Ratnam, 2013). While they can thrive on various well-drained soils, these woodlands are particularly notable on rocky or shallow soil types. These areas are characterized by dominant tree species such as *Tectona grandis*, *Anogeissus latifolia*, as well as species from *Terminalia*, *Boswellia*, *Diospyros*, and *Sterculea*. Grasses in these regions are typically of moderate height, with prevalent varieties including *Heteropogon*, *Themeda*, and *Saccharum* (Sankaran and Ratnam, 2013). Additionally, the bamboo species *Dendrocalamus strictus* is commonly found throughout these woodlands. Both fire and grazing play crucial roles in these landscapes, which have historically coexisted with some of the highest human population densities in the peninsula. In terms of structure, they bear the closest resemblance to the "tree and shrub" savannas found in Africa. Yet, the specific factors that distinguish the "savanna" formation from the "forest" formation (without grassy understories) in this habitat type remain uncertain.

4.3 Moist Sal savanna

These are broadleaf forests with dense grass cover that are found sporadically throughout the central plains of the Indian subcontinent. The average yearly precipitation in these areas can vary between 1400 and 1900 mm (Sankaran and Ratnam, 2013). The prevailing tree species in this area are Sal or *Shorea robusta*, along with *Careya arborea*, *Emblica officinalis*, and *Wrightea tomentosa*. The primary grasses found in the understorey consist of *Imperata cylindrica*, *T. triandra*, *Cymbopogon*, and various *Apluda* species. As these plant formations thrive within a moist forest zone, Champion and Seth (1968) hypothesize that frequent fires and winter frosts play pivotal roles in shaping these ecosystems.

4.4 Dipterocarp savanna forests

These arid woodlands extend across the seasonal mainland of Southeast Asia, spanning from eastern Vietnam to Cambodia, Laos, Thailand, and Myanmar, and reaching the Manipur state in Northeast India (Sankaran and Ratnam, 2013). These areas typically receive an average annual precipitation of 800–2000 mm, characterized by dry periods lasting between 5 and 7 months. The widespread tree species found in the area include *Dipterocarpus intricatus*, *Dipterocarpus tuberculatus*, *Dipterocarpus obtusifolius*, *S. robusta*, *Shorea roxburghi*, and *Shorea siamensis*. Additionally, notable tree species encompass *Dillenia varieties*, *Pinus merkusii*, *Pterocarpus macrocarpus*, *Terminalia species*, and *Xylia kerrii*. The primary grasses in the region comprise genera such as *Arundinella*, *Capellipedium*, *Heteropogon*, *Polytoca*, *Imperata*, *Themeda*, and pygmy bamboos of the *Arundinera* genus. Characteristic flora also includes dwarf palms of the *Phoenix* genus and cycad members of the *Cycas* genus. Structurally, these ecosystems bear similarities to the miombo woodlands of Africa and the eucalypt woodland savannas of Northern Australia, as noted by Stott (1990).

4.5 Savannas in China

4.5.1 Savannas Hun Shan Dak

Departing from Tiananmen Square in Beijing and driving northward for 2 h, there is an extremely unique sparse tree grassland in the world (Fig. 18.2). In the middle of the Xilingol Plateau in Inner Mongolia, there's a sandy area called Hun Shan Dak. It rises from Darinall, the western foot of the southern section of the Great Xing'an Mountains in the east, and extends to the west along the railway from Jining to Erenhot. The total area is 52,000 km^2, equivalent to the size of Croatia. This is one of the four major sandy lands in China and is also a savanna. The water and grass of Hun Shan Dak are rich and beautiful (Fig. 18.3). People call it "south of the Yangtze River," while others call it a "garden desert," which is a paradise for cross-country self-driving. It is a relatively rare temperate savanna.

The longitude has played a decisive role in the formation of this savanna. The regions of Gansu and Xinjiang province in China, located at the same latitude, primarily consist of desert grasslands. Similarly, the temperate areas in the United States at the corresponding latitude also exhibit desolate characteristics.

The influence of latitude zonality cannot be ignored either. There is no distribution of elm savanna in other deserts in Inner Mongolia at the same longitude, like Kubuqi, Tengri, and Badan Jilin Desert, because of the insufficient precipitation.

Hun Shan Dak has abundant precipitation. There are many large gaps in sandy soil, therefore, the rainwater is not lost. It can be stored smoothly underground. Some low-lying areas turned into a lot of small bubbles gradually. The older Mongolian language is called "Nur." For example, Darinur (Fig. 18.4).

Why the savanna is attractive? Because many large wild animals inhabit there. The same is true of Hun Shan Dak. It is one of the largest wildlife habitats in northern China. People can see the yellow sheep, sand fox, red deer, and hare, and of course, there are domestic camels (Fig. 18.5). Hun Shan Dak is also a breeding place for migratory birds such as black face split, swans, larks, shrikes, and falcons.

4.5.2 Savannas in Hainan Island

Hainan Island not only has sunny coastlines but also sparse grasslands. In the western part of Hainan Island, stretching from Yangpu and Changjiang to Yinggehai, along the

FIG. 18.2 The elm tree is the dominant tree species in Hun Shan Dak, accounting for 95% of the sand trees. *(Photo credits: Guan Yongqing.)*

FIG. 18.3 The annual precipitation of Hun Shan Dak can reach 400 mm, which is a famous watery desert in China. Most of these small rivers flow into the Gogustai River. *(Photo credits: Guan Yongqing.)*

FIG. 18.4 Darinur is located in Hexigten Banner, Chifeng City, Inner Mongolia, with an area of about 237 km^2, which is equivalent to the size of the Cook Islands. *(Photo credits: Badarilatu.)*

coastal plains, river deltas, and some sandy areas, there is a strip of land approximately 170 km long and 3–20 km wide. This is the only true tropical sparse grassland in China. One of the most typical areas is The Daitan National Nature Reserve. Upon entering the scenic area, one might even feel as if they have been transported to Africa. Here, one can find the "critically endangered" Hainan deer (*Cervus eldii*) (Fig. 18.6), living in the area.

In the early 1960s, the Hainan deer was on the brink of extinction, with only a few remaining in the communes of Dongfang and Baisha. By 1976, the deer was on the verge of extinction, with only 44 individuals left. In response, the government established the Daitan National Nature Reserve in Hainan (Fig. 18.7), to protect this unique species native to Hainan.

Hainan Island has a tropical monsoon climate, and one would expect to see lush tropical rainforest scenery. Why then, in the western part of the island, has a nonzonal sparse shrub grassland formed? Scholars classify global sparse shrub grasslands into three types based on their causes: soil, volcanic or anthropogenic, and climate. Amongst these, climate is the most important factor.

The western part of Hainan Island also experiences this typical climate pattern: high temperatures throughout the year, distinct wet and dry seasons, and particularly dry conditions during the dry season. The reasons behind this are the interactions of the East Asian monsoon, the South Asian monsoon, and the Northwest Pacific monsoon. Their combined effects shape the tropical sparse tree grassland of western Hainan Island.

During the winter and spring seasons, the clash between polar continental air masses and maritime air masses brings rare dry-season rainfall to the eastern part of Hainan Island. Unfortunately, the Qiongzhong mountains block this precipitation, creating a "rain shadow zone" and making the dry season in the western part of Hainan even drier.

The prevailing summer southwest and southeast monsoons are respectively blocked by the Changshan

FIG. 18.5 The average biomass of sparse forest grassland is 90% higher than that of temperate grassland, so it can carry more large animals. *(Photo credits: Guan Yongqing.)*

FIG. 18.6 Datian is the most typical tropical sparse shrub grassland reserve in Hainan. *(Photo credits: Li Yunbin.)*

FIG. 18.7 Today, the population of Hainan deer in this area has exceeded 2000. *(Photo credits: Li Yunbin.)*

Mountains and the Qiongzhong Mountains creating a "rain shadow zone" in the western part of Qiongzhong where summer rainfall is significantly less than in other areas of Hainan Island. The "burning wind" in spring exacerbates the dryness of the dry season in the west of Qiongzhong which lasts for 6 months and is also a prolonged windy season. The prevailing southwest winds in this period are warm and hot creating a "burning wind effect" resulting in high temperatures. Furthermore, the quaternary sedimentary layers in the western part of Hainan Island are

highly permeable leading to a "karst" effect where precipitation is difficult to retain making the already dry ground even drier.

4.5.3 Savanna in Yun Nan Grand Canyon

The Foehn wind also brought savanna to Yunnan, Hengduan Mountain, which runs north and south by the Lancang River, Jinsha River, Nu River, and Yuan River, waiting for several major rivers to cut deep. The airflow crosses the mountains and faces the windward slope, bringing abundant rainfall and climbing to the leeward slope. For every 1000 m of sinking, the temperature increases by 6.5°C. When it descends to the bottom of the Grand Canyon, which is 4 or 5 km away, it's already a high temperature of over 20°. Water vapor also decreases sharply and the air becomes hot and dry. This is the so-called dry and hot valley (Fig. 18.8).

In these hot and dry regions, a valley-shaped Savanna is academically known as dry and hot sparse shrubs and grasslands.

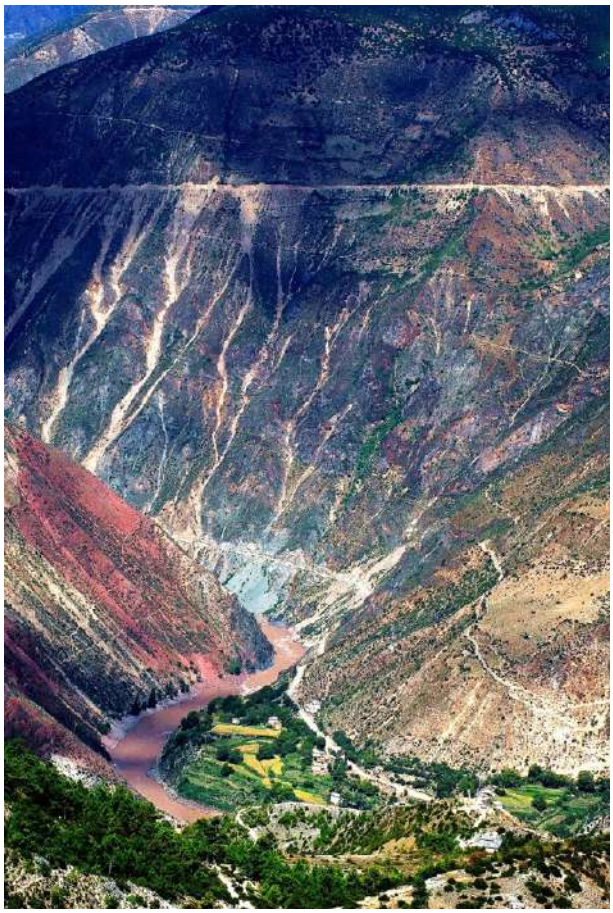

FIG. 18.8 The Grand Canyon of the Lancang River on the eastern slope of Meili snow mountain, the snow mountain meltwater forms a valley oasis. *(Photo credits: He Yihong.)*

FIG. 18.9 Shrimp flower. *(Photo credits: Wu Jianmei.)*

Although the savanna in these canyons looks different from the African savanna, it actually possess an extremely similar community appearance. That is to say that they have phylogenetic relationships with species or they could be species of African savanna itself. For example, in the dry and hot valleys of the middle and lower reaches of the Lancang River, kapok, emblic leaf flower fruit, and shrimp flower (Fig. 18.9) have a close relationship with the vegetation in the African savanna.

There are two species of shrimp flower in the world. One species is distributed in African savannas, while the other originates from Madagascar in Africa and travels through India to the Yunnan savanna. This distribution often supports the statement that the separation of the Indian Plate from Madagascar, along with northward drift and collision and fusion with the Asian continent, has transformed the vegetation of the original African savanna into that of the Asian continent.

Savannas in Yunnan are mainly distributed on low hills and plateaus in the canyons of the Yuan River, Lancang River, and Nujiang River. Some places are filled with cacti (Fig. 18.10), which gives people the illusion of being in South America.

This is a huge contrast from those Yunnan's signature attractions in ShangriLa. However, in these deep canyons of Yunnan, most of them are like this kind of landscape baptized by the foehn wind.

Although it looks a bit barren, researchers conducted research and set up 44 infrared cameras in the dry and hot valley of the Jinsha River. They snapped gorals, blue sheep (Fig. 18.11), and other wild animals. This indicates that there is a high level of biodiversity maintained here, even higher than subtropical mountain forests. The area offers shade, food, and shelter for wild animals, livestock, and human beings. Also became a paradise for early humans to complete evolution.

FIG. 18.10 The altitude of hot and dry river valleys in Yunnan is usually below 1000 m, with abundant solar and thermal resources. *(Photo credits: Xie Gang.)*

FIG. 18.11 China's national second-level protected wild animal blue sheep. *(Photo credits: Geng Dong.)*

5 Conclusions

Asia's savannas have existed for millions of years and are considered to be sustained as successional disclimax ecosystems as a result of human activities and natural disturbances such as fire, floods, and grazing. These ecosystems provide habitats for a range of faunal species, including species that are now adapted to these ecosystems and have even evolved into endemic species. However, intensive agriculture practices, livestock grazing, human settlement, and invasion by exotic invasive species are rapidly changing natural savannas, which will eventually place the survival of grassland-adapted species, and the ecosystem services provided by savannas at risk. Further, shifts in the monsoon seasons, reflected by intense rainfall events and long periods of droughts, are likely to have impacts on the structure and dynamics of savannas across Asia.

Finally, it is important to note that the impacts of various disturbances in driving community and ecosystem dynamics within South Asian savannas are largely understudied, and the current knowledge of the functional role of herbivores and fires remains mostly limited. Thus, if these important ecosystems are to be sustained, especially under uncertain weather patterns and expanding anthropogenic influences, a better understanding of these ecosystems should be gained to formulate effective savanna management strategies.

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Chapter 19

Australian Savannas

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1 Introduction

Apart from the Polar Regions, savanna formations occur on all continents of the world, with the largest extent found in the wet-dry tropical regions of Africa, South America, and Australia. There is growing appreciation that tropical savanna and grasslands are as biodiverse as tropical forests, particularly for mammals, birds, and amphibians (Murphy et al., 2016). Australian savanna biota have a high degree of endemism, a function of long periods of geographic isolation punctuated by periods of exposure to Malesian floristics and faunal elements (Barlow, 1994). These processes have shaped northern Australian biodiversity, as did the arrival of Aboriginal peoples some 60,000 years ago with their customary land management practices, particularly the use of fire (Kershaw, 1986), especially prominent during the Holocene (Bird et al., 2024).

Australian vegetation is dominated by *Eucalyptus*, *Corymbia*, and *Acacia* woodlands and shrublands, and their distribution spans cool temperate, warm temperate, Mediterranean, semiarid, and tropical climate zones of Australia (Gillison, 1994). In the northern quarter of the continent, the climate is monsoon driven, with highly seasonal rainfall (Suppiah, 1992; Bowman et al., 2010), and here tropical savanna ecosystems have evolved (Fig. 19.1). Northern Australian savannas occur across ancient (Precambrian basement rock) weathered landscapes of low topographic relief and soils are thus leached and low in nutrient status (McKenzie et al., 2004). Alternating warm wet and cooler dry periods, with conditions favorable for the oxidation and reduction of metal-rich soils (Fe, Al), have produced duricrusts and acidic texture-contrast soils that limit plant growth. These climatic and soil characteristics, plus a frequent fire regime due to an annual dry season of 4–6 months (Russell-Smith and Yates, 2007), have driven the extensive diversification of stress tolerant and resilient flora with *Sorghum*, *Aristida* and *Triodia* species of the Poaceae, *Grevillea* in the Proteaceae, *Eucalyptus* and *Corymbia* of the Myrtaceae, and *Acacia* in the Mimosaceae all dominant plant groups (Fox et al., 2001).

Savannas of the rest of the world tend to support a higher population density where subsistence agriculture and livestock management are major land uses, whereas northern Australian savanna is remote with a low population density (0.01–0.3 persons km^2), both historically and contemporarily, and as such it is a largely intact and continuous landscape containing a number of Australia's biodiversity hotspots (Woinarski et al., 2007; Murphy et al., 2016). However, over the last four decades it is apparent that there are changes in ecological processes driven by climate change (Allen et al., 2021) and anthropogenic disturbances via shifts in fire regimes (Einoder et al., 2023), and invasive species, be it plant (Setterfield et al., 2013a,b; Rossiter-Rachor et al., 2023) or animal (Mihailou and Massaro, 2021; Radford et al., 2021). These impacts are potentially amplified by increasing land and water resource use pressures (King et al., 2015; Bristow et al., 2016; Rhebergen and Yeates, 2023). This chapter will outline the distribution and characteristics of savanna landscapes in Australia, the biodiversity and conservation areas and describe recent impacts from land use and climate change on biodiversity and savanna function. Finally, we outline current management options to maintain the natural capital of this unique region.

2 Geographic location

Tropical savanna is the predominant vegetation type across the northern quarter of Australia where rainfall is seasonal and mean annual rainfall is above 500 mm per year (Fig. 19.2A and B). This biome occurs across three state and territory jurisdictions in northern Australia, namely northwest Western Australia, the Northern Territory (NT) and northern Queensland (Bowman et al., 2010; Fig. 19.2). Northern coastal regions receive rain

Terrestrial Biomes. https://doi.org/10.1016/B978-0-443-36569-0.00019-7
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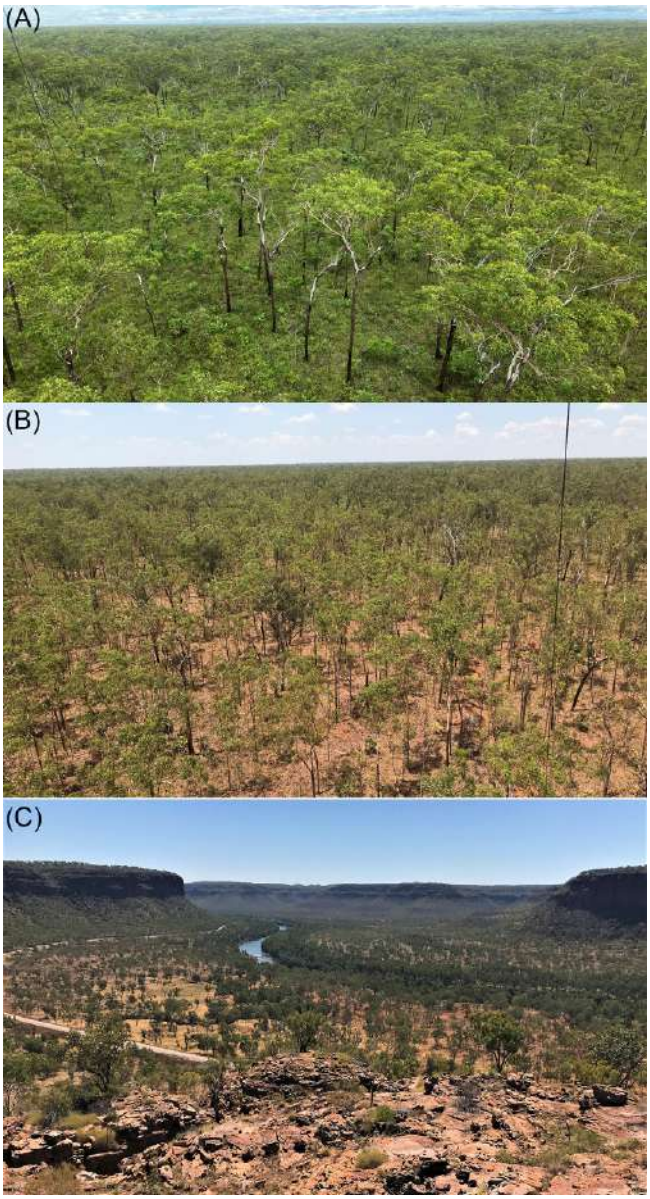

FIG. 19.1 Tropical savannas of northern Australia. High rainfall (~1900 mm pa) and frequently burnt savanna (2 in 3 years) dominated by *E. tetrodonta* and *E. miniata* is shown during (A) the wet, and (B) the dry season. Panel (C) shows open-woodland savanna from the Victoria River region of western NT, an area of lower rainfall (~650 mm) resulting in reduced woody cover compared to (A) and (B). The Victoria River and dissected escarpment environs provide a degree of fire protection and support a distinct flora and fauna from the lowland savanna habitats. *(Images courtesy of the authors.)*

approaching 2000 mm per annum, with 80%–90% of this rain falling between September to April with high inter-annual variability of rainfall and runoff, a feature of the northern Australia's hydro-climatology (Petheram et al., 2008). This region extends from latitudes 11 to 25°S covering ~1.91 million km^2, about 12% of the world's tropical savanna (Beringer et al., 2015), with a Köppen-Geiger *Aw tropical savanna* climatic class (Peel et al., 2007). Floristically, these savannas are considered as megathermal (thermal optima at ~28°C) open-woodlands to open-forests (after Gillison, 1994), with tree height (8–20 m) and woody cover generally declining as rainfall decreases with distance from the coast (Bowman, 1996; Williams et al., 1996). The overstory flora is dominated *Eucalyptus* and *Corymbia* tree species, an understory of near-continuous C4 grass cover (Fig. 19.1) and a unique fauna (Woinarski et al., 2007).

3 Biodiversity

The faunal and floristic assemblages of Australia's tropical savanna have been shaped by the proximity to the Malesia region, geographic isolation, the monsoonal climate, low

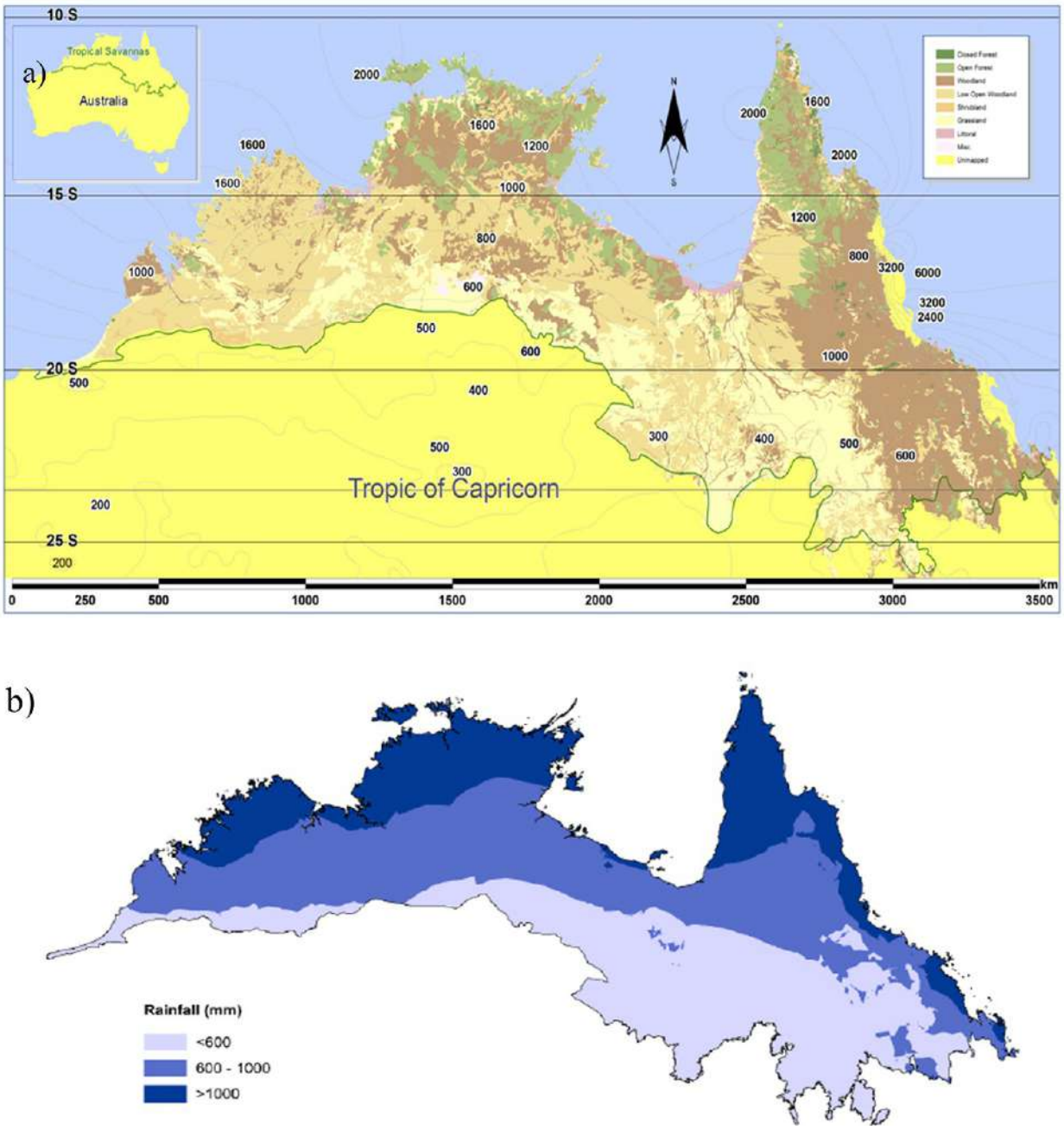

FIG. 19.2 The savanna biome of northern Australia as defined by Fox et al. (2001) with broad vegetation types shown in (A), and (B) distribution of broad rainfall bands.

fertility soils driving the evolution of scleromorphy, and a regime of frequent fire (Barlow, 1994; Bowman et al., 2010). The dominant *Eucalyptus-Corymbia* savanna woodlands have phytogeographic origins from the iconic Australian autochthonous Gondwanan floristic element; vegetation that evolved following Australia's separation from Gondwanaland and the northward tectonic drift, resulting in geographic isolation, increasing aridity and fire frequency during the Pliocene and Quaternary eras. This evolutionary history and the dominance of Myrtaceous flora in contemporary Australian savannas reflects the unique features of Australian savanna relative to South American and African savanna biomes in terms of relationships to climatic drivers, soils, and fire (Lehmann et al., 2014).

Australian savannas are relatively intact, and they differ from other savanna regions of the world in that the woody vegetation is predominantly evergreen despite occurring in

one of the world's most seasonal and high-temperature savanna climates (Bowman and Prior, 2005). Typically, the plant canopy consists of two or three strata depending on site features (rainfall, fire regime). Below the overstory of *Eucalyptus* and *Corymbia* species is a mid-canopy stratum of juveniles of these species, plus a range of non-eucalypt taxa plus a third strata, the understory which is dominated by C_4 grasses, regenerating saplings, resprouts, and other shrubs species (Williams et al., 1997). At lower rainfalls (<500 mm), *Acacia*-dominated savanna communities occur and include extensive areas of brigalow (*A. harpophylla*), lancewood (*A. shirleyi*), and gidgee (*A. cambegei*, *A. georginae*). While *Eucalyptus*/*Corymbia* species are evergreen, at the edge of their biogeographic ranges, these tree species can lose significant leaf area late in the dry season (Bowman and Prior, 2005). The mid-strata features pan-tropical tree and shrub species with a range of phenologies from semi- to fully-deciduousness (Williams et al., 1997), a biodiverse strata especially rich in locations with low fire frequency (Bond et al., 2012). A variety of tall annual and perennial grasses occur, including *Sorghum*, *Heteropogon, Chrysopogon, Triodia* and *Schizachrium*. The perennial grass *Heteropogon contortus* (black speargrass) dominates the understory in eastern Queensland, with *Themeda triandra*, *Aristida*, *Bothriochloa*, and *Chrysopogon bladhii* becoming more dominant as rainfall declines.

Embedded within the vast savanna matrix across northern Australia are other important ecosystems; extensive floodplains, associated seasonal wetlands, *Melaleuca* swamp forests, and scattered spring-fed and/or fire protected closed forest patches with Malesian or intrusive phytogeographic origins (Malaysian Peninsula, Papua New Guinea, Russell-Smith, 1991). It is evident Australian savanna flora and fauna was influenced by the arrival of humans some 60,000 years ago via their use of fire and hunting pressure (Bird et al., 2013; Johnson, 2016), and again with European settlement some 150 years ago, who brought extensive pastoralism and changed fire regimes (Lewis, 2002), mining, urban settlements, and invasive species (Setterfield et al., 2013a,b; Frank et al., 2014; Woinarski et al., 2015).

Several major Australian biodiversity hotspots occur in the tropical savannas including the sandstone escarpment vegetation of western Arhemland (Woinarski et al., 2009), the Cape York savannas of north Queensland (Wannan, 2014), and the Kimberley region of northwest WA (Bowman et al., 2010). These are some of the most species-rich tropical savanna and grassland ecoregions globally (Murphy et al., 2016), and each is characterized by significant topographic relief and escarpments that are distinct from the surrounding lowland savannas (Fig. 19.1C). These escarpments provide a diversity of environments, including rock pavements, deeply dissected gorges, wetland systems, and deep sandy plains. Rock pavements and sand sheets support diverse heathlands, while sheltered gorges provide moisture through the dry season and fire protection, enabling the survival of relictual rainforest species (Prior et al., 2007) as well as open-woodland savanna communities.

Savanna landscapes support a wide range of iconic birds, mammals, reptiles, amphibians (Woinarski et al., 2007) plus a remarkable range of freshwater fish species that occur across the extensive northern river systems (Pusey and Kennard, 2009). There is also a rich invertebrate fauna, in particular ants, termite, and grasshopper groups who play a key role in nutrient cycling given the absence of large herbivores in Australian savanna (Andersen et al., 2018).

As is the case for all savanna landscapes, seasonal rainfall and the presence of frequent fire are pervasive drivers of vegetation structure and biodiversity (Scholes and Archer, 1997). High rates of wet season primary production followed by the rapid onset of the lengthy dry season (6–8 months) results in a highly flammable landscape (Liedloff and Cook, 2007) and fire is frequent and occurs right across the savanna domain (Fig. 19.3). Fire burns up to 50% of the region annually, contributing 3% to Australia's reportable greenhouse gas emissions under the Kyoto protocol (Maraseni et al., 2016) and this landscape can be considered one of the most fire-prone on earth (Edwards et al., 2021).

Almost a century of research on African savannas has highlighted the fundamental importance of fire as a driver of savanna productivity, vegetation physiognomy, livelihoods, and biodiversity (Scholes and Walker, 1993; Biggs et al., 2003; Sankaran et al., 2005; Higgins et al., 2007; Archibald et al., 2009; Buitenwerf et al., 2012; Stevens et al., 2016). Similarly, over the last 50 years, a number of manipulative fire experiments have been designed and implemented in Australian savannas to also examine impacts of fire properties (seasonal timing and severity, frequency, and recurrence) on floristic and faunal assemblages (Russell-Smith et al., 2003; Williams et al., 2003a,b; Andersen, 2021). From such experiments, ecosystem-scale resilience to fire can be assessed, as well as understanding the responses of specific taxa to fire.

Major experiments include CSIRO's subcatchment scale Kapalga experiment, implemented from 1989 to 1996 in Kakadu National Park (Andersen et al., 2003). This experiment featured replicated subcatchments subjected to contrasting fire regimes of early and late dry season fires to

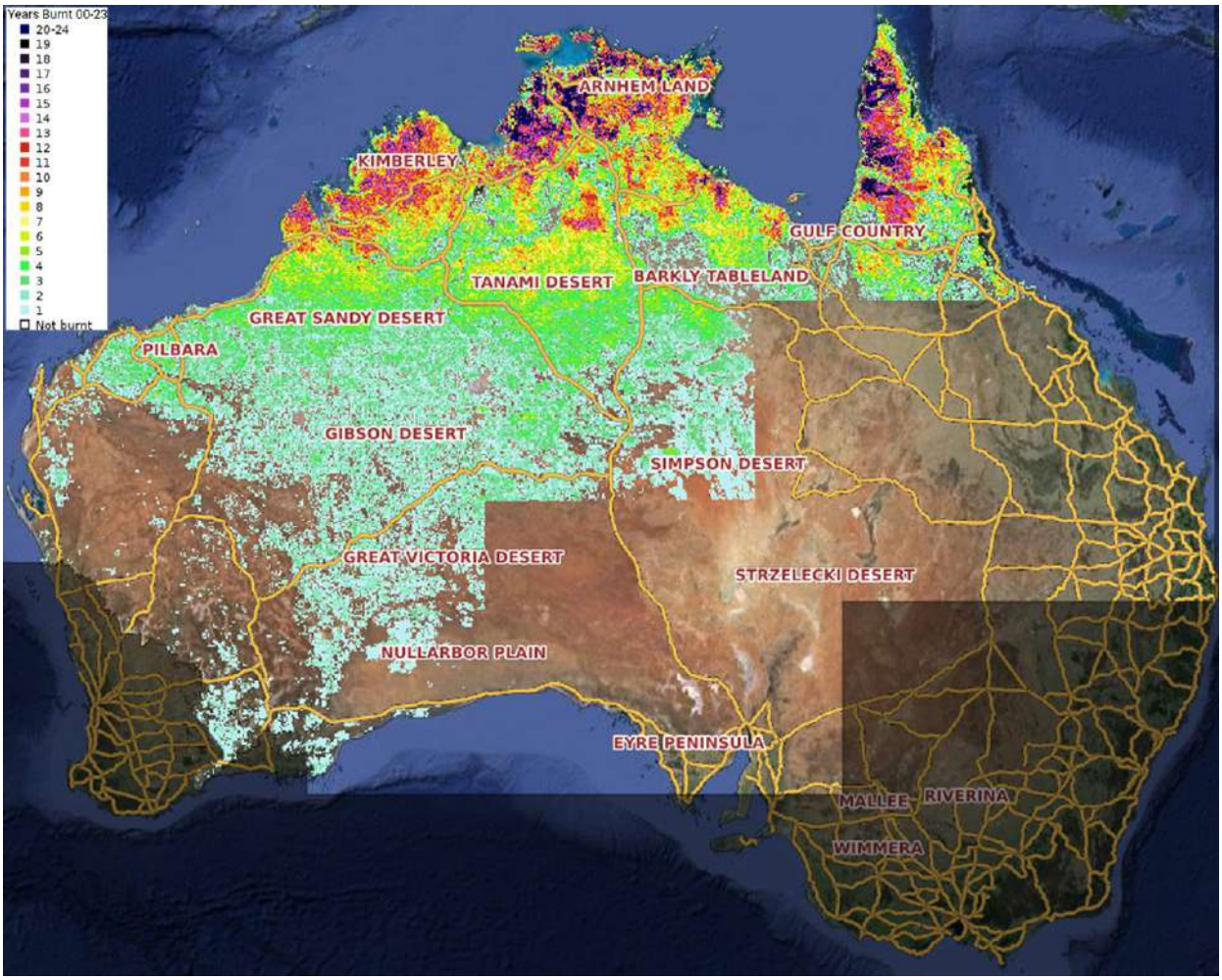

FIG. 19.3 Fire frequency map (years burnt, 2000–23) for the northern Australian savanna region. The map was generated by the North Australian Fire Information system (NAFI, firenorth.org.au/nafi3). Burnt area maps are derived from MODIS satellites (250 m pixels, Russell-Smith et al., 2009a). Map lines delineate study areas and do not necessarily depict accepted national boundaries.

compare the impacts of maintaining unburnt savanna to subcatchments burnt with lower intensity early dry season fires and subcatchments burnt with higher intensity, late dry season fires (Fig. 19.4). A progressive regime was also implemented where fires were lit throughout the dry season. Also operational in Kakadu National Park was the long-term (1973–96) plot-scale (1 ha) Munmarlary experiment, also with early, late and unburnt fire treatments with similar goals (Russell-Smith et al., 2003, 2010a). Ongoing is the Territory Wildlife Park fire experiment (2003-present) also at a replicated plot-scale (1 ha) assessing early dry season fire recurrence (annual fire, 1 in 2, 3, and 5 years), plus a late dry season treatment burnt every 2 years and an unburnt treatment (Levick et al., 2019; Andersen, 2021). The Tiwi Island Fire Experiment (2008-present) assesses vegetation biomass, structure and fauna in 50–100 ha blocks subjected to early and unburnt treatments on the Tiwi Islands, a high rainfall savanna (>1600 mm per annum) north of Darwin, NT (Richards et al., 2012). The Kidman Springs Fire experiment (Shurburn, 1993-present) is an important long-term study examining interactions between fire and grazing on differing soils types (Cowley et al., 2014) located in lower annual rainfall (740 mm), semiarid grazed savannas of the NT. Woody and native pasture species dynamics are assessed with early and late dry season fire treatments recurring at 2-, 4-, and 6-year intervals with unburnt "control" plots. This design is repeated on two common soil types supporting savanna woodland (calcarosol red earth

FIG. 19.4 Typical fire severities in mesic savannas of the NT: (A) low severity, early dry season fire (late April), (B) moderate severity dry season fire (June), and (C) high severity and fire (September) fueled by highly invasive African Andropogon grass, *A. gayanus*. *(Images courtesy of the authors.)*

soil) and grassland (vertosol gray cracking clay) ecosystems with all plots subjected to grazing (Woinarski et al., 1999; Cowley et al., 2014, 2021; Walsh et al., 2014).

These experiments have provided key insights to wide range of responses of savanna flora and fauna to differing fire regimes; a key finding was the importance of periods with fire and the impact of frequent, high-severity late dry season fires on both flora (tree growth, seed production, recruitment, and mortality, Williams et al., 2003a) and fauna (small mammals) while some taxa such as ants, frilled lizards, and predatory birds were abundant in late dry season treatments (Corbett et al., 2003). In short, a range of responses to fire regime was observed and Williams et al. (2003b) concluded no single fire regime will deliver all conservation goals to maximize biodiversity given such complexity—adaptive management is required that is underpinned by ongoing research linked to effective long-term monitoring of both habitat metrics and targeting particular species.

Complementing these manipulative fire experiments, plot based, long-term biodiversity monitoring was established in the mid-1990s (1994–97) via the Three Parks Fire Plot Network (Russell-Smith et al., 2009b, 2010b) with 220 plots (20×40 m) established within the three major national parks of the NT (Litchfield, Kakadu, and Nitmiluk). From 2006, an addition 110 plots were established in lower rainfall zones of the Gulf of Carpentaria, central Arhemland, and the Kimbery region of WA (Murphy et al., 2023). These long-term plots have been used to examine changes to tree biodiversity, demographics, growth, cover change, and carbon sequestration modeling (Lehmann et al., 2008, 2009a,b; Murphy et al., 2010, 2023). A major biodiversity concern has been the loss of small mammals across northern Australia (Woinarski et al., 2011) and this plot network was used to assess spatial and temporal trends in mammal richness and abundance and how fire impacts are moderated by environmental gradients (fire-protection and terrain ruggedness, landscape position, and productivity) (Stobo-Wilson et al., 2020; Einoder et al., 2023). Fauna monitoring over 24 years showed major declines in mammal richness and abundance where plots were subjected to large fires with fire events reoccurring within a 5-year threshold. Habitats of low topographic relief and at the lower end of the rainfall gradient were the most susceptible to mammal loss (Einoder et al., 2023).

4 Conservation areas and their ecological indicators

The current protected area estate across northern Australian savanna comprises a range of conservation entities, including state- and territory-managed national parks and environmental and conservation reserves, the World Heritage Listed parks Purnululu National Park (WA), and Kakadu National Park (NT) managed by the Australian Government, private conservation area managed by organizations such as the Australian Wildlife Conservancy (AWC) and Bush Heritage Australia, Indigenous Protected Areas, Defense land (under the EPBC Act, 1999), and RAMSAR-listed wetlands. These conservation areas (Fig. 19.5) are significant in size and account for 12.3% of the savanna area of northern Australia, an area of ~23 million ha, and collectively form a crucial network for biodiversity conservation (Perry et al., 2021).

5 Human activities and their impacts

Humans arrived in northern Australia some 65,000 years ago (Clarkson et al., 2017) via land bridges to New Guinea and Malesia, likely dispersing through northern Australia to the eastern semiarid and arid zones, and then to southeastern Australia and into the arid interior (Bird et al.,

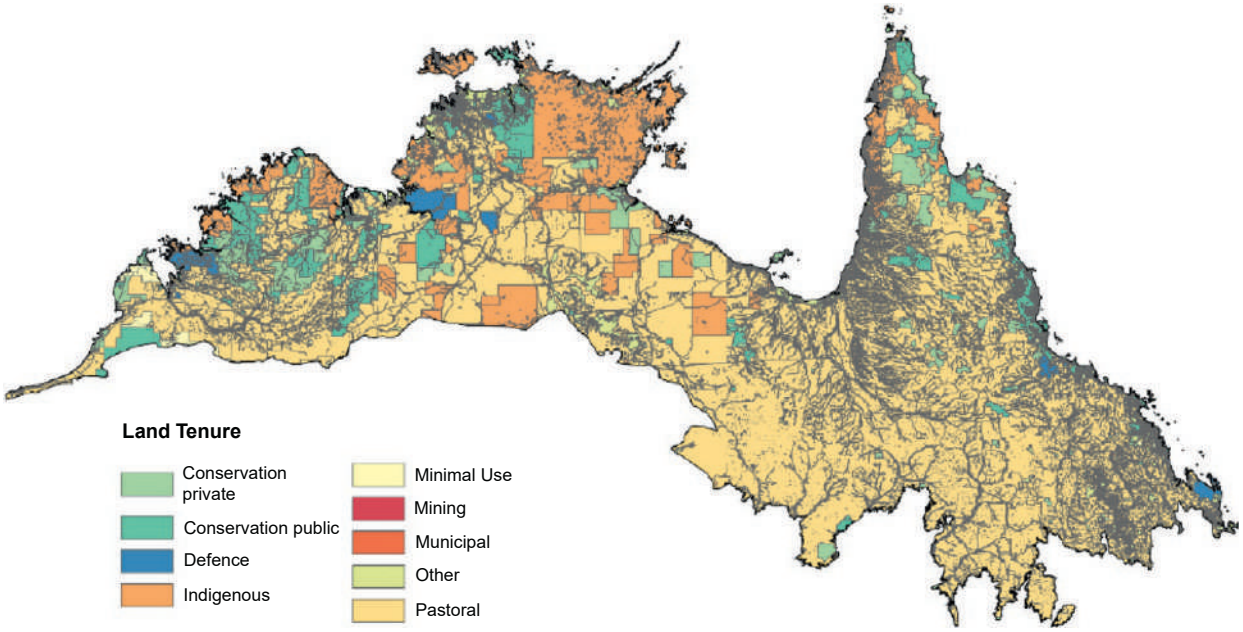

FIG. 19.5 Land tenure across the northern Australian savannas. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Redrawn from Perry, J., Patykowski, J., Murphy, H., Richards, A., Gillespie, G., Vanderduys, E., Douglas, M., Setterfield, S., Somaweera, R., 2021. Savanna Fire Regimes and Biodiversity Impacts in Northern Australia (Northern Australia Environmental Resources (NAER) Hub, CSIRO, p. 69. https://nesplandscapes.edu.au/wp-content/uploads/2021/11/Savanna-fire-regimes-and-biodiversity-impacts-final-report.pdf (Creative Commons Attribution 4.0 Australia license).)*

2016). Fig. 19.3 provides a contemporary map of fire occurrence and frequency, although prehuman and pre-European fire regimes would have looked significantly different. Prehuman fire regimes presumably comprised infrequent but extensive and severe fire events. The arrival of Aboriginal peoples brought active management, which reduced the extent and frequency of these large fires and was replaced by the lighting of smaller, localized fires for hunting, travel, ceremony, and protection of resources (Yibarbuk et al., 2001), this shift in fire scale resulted in changes to geographic range and demographic structure of vegetation, and created habitats that favored the abundance of particular faunal species (Bowman, 1998; Bird et al., 2024).

While the current savanna estate of northern Australia can be described as intact, since the arrival of Europeans 150 years ago, a number of threats to the region's biodiversity and ecosystem resilience is apparent. On arrival of Europeans, de-population of Aboriginal managed landscapes occurred to make way for pastoral activities and urban settlements. The cessation of Aboriginal land management changed fire regimes, with contemporary fire regimes featuring a degree of suppression for pastoral management, as well as uncontrolled, frequent, and damaging wildfires that has changed vegetation structure, floristic mix with a loss of fire-intolerant species and fuel characteristics (Woinarski et al., 2007). Frequent fire results in declines in fire-sensitive biota such as non-eucalypt woody taxa (e.g., *Callitris intratropica*), granivorous birds, small mammals (Woinarski et al., 2011), engineers recruitment bottlenecks as frequent fire suppresses seed, and woody regeneration (Setterfield, 1997; Bond et al., 2012; Werner and Prior, 2013), and reduces savanna carbon sequestration potential (Beringer et al., 2007).

Adding to fire threats to biodiversity is the introduction of exotic fauna, such as cats, associated with the growing pastoral industry. Animals such as Asian buffalos, donkeys, horses, and pigs have multiple destructive impacts on floodplain and wetlands water quality and biodiversity across northern Australia (Ens et al., 2016; Mihailou and Massaro, 2021). Domestic cats have also become feral animals impacting savanna ecosystems and are highly effective predictors of native rodent fauna. Predation is especially severe in savannas subjected to high severity fire and disturbance from feral herbivore activity (Davies et al., 2017, 2020).

Exotics grasses were also introduced to improve pastoral production, but a number of species have escaped the pastoral estate and have invaded natural and

conservation areas. Introduced pasture species have invaded floodplains (Setterfield et al., 2013a) and savanna ecosystems, particularly in high-rainfall (>1000 mm) landscapes (Rossiter-Rachor et al., 2008; Setterfield et al., 2013b, 2018). A particularly invasive tropical C4 grass species, *Andropogon gayanus*, is highly productive and generates fuel loads up to five times that found in native Australian savannas (Rossiter et al., 2003). Once cured, this fuel load supports a cycle of high-severity fires (Fig. 19.4C) and results in significant tree mortality and a major loss of biodiversity. The loss of tree cover promotes further growth of the shade-intolerant C_4 grass and over subsequent wet seasons, and in patches with extreme infestations, this grass can initiate a feedback loop that drives savanna stands on a trajectory toward a simple grassland, in effect an ecosystem collapse (Bergstrom et al., 2021; Rossiter-Rachor et al., 2023).

6 Outlook

The above discussion highlights contemporary threats to biodiversity across Australia's savannas, and while they can still be classified as relatively intact landscapes, there are clearly growing pressures on ecosystem biodiversity from fire regimes that are at odds with conservation goals (Stobo-Wilson et al., 2020; Einoder et al., 2023) and impacts from feral animals and exotic plant species. Fig. 19.6 is a modeled distribution of threatened species density for northern Australia; the savanna region has a relatively high density of threatened species, e.g., the Arhemland escarpment (circled) and the frequently burnt coastal savannas. For the tropical savannas, the mean is 4.4 species (range: 0–35) and across the Australian continent, including the far more disturbed southern Australian landscapes, the mean is 5.1 species (range: 0–71).

In addition, northern Australia is perceived as a region that could support an expanded agricultural estate (Petheram et al., 2018) such as genetically modified and high water-efficient cotton variants (Rhebergen and Yeates, 2023). To support a viable cropping industry, up to 100,000 ha which will include previously cleared savanna woodland. Any additional clearing would release in the order of 148 Mg CO_2-e ha^{-1} based on estimates of Bristow et al. (2016). Other land and water resources pressures may come for expanded horticulture and the expansion of an onshore hydraulic fracturing of gas reserves in the NT (Pepper, 2018).

Climate change impacts are likely to be complex and difficult to predict with both positive and negative feedbacks interactions between climate change, biodiversity, and fire regimes. Changes in biodiversity composition resulting from the spread of exotic grasses, shifts in species distributions, and the effect of elevated atmospheric CO_2 on vegetation will affect the key drivers of fire regimes in Australia. Williams et al. (2009) reviewed likely impacts of climate change and fire regimes on Australian biodiversity. The study featured four regional case studies, one being the tropical savannas and concluded that effective management of biodiversity in a changing climate requires an increase savanna areas that are unburnt for up to 4–5 years, rather than the current 1–3 year cycle. This will be challenging with elevated temperatures (Fig. 19.7) and an increased incidence of days >35°C (Fig. 19.8) that will drive a more flammable landscape. Savanna fuel production is likely to be maintained at current levels given the observed increasing rainfall in northern Australian (Australian Bureau of Meteorology and CSIRO, 2024) and elevated CO_2 driving increases in savanna productivity and water use efficiency (Hutley et al., 2022).

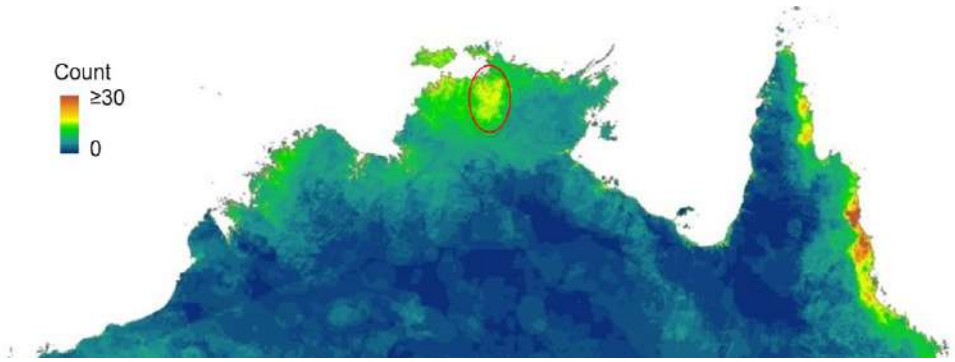

FIG. 19.6 Modeled distribution of threatened species density (number per 1 × 1 km grid) for northern Australia. The map includes all species (and subspecies of plants and animals) listed as threatened under the Australian Government's Environment Protection and Biodiversity Conservation Act, 1999. The Arhemland escarpment is *circled*. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Modeled distributions of the threatened species were downloaded from https://fed.dcceew.gov.au/datasets/erin::australia-species-of-national-environmental-significance-distributions-public-grids/about (Assessed January 14, 2024).)*

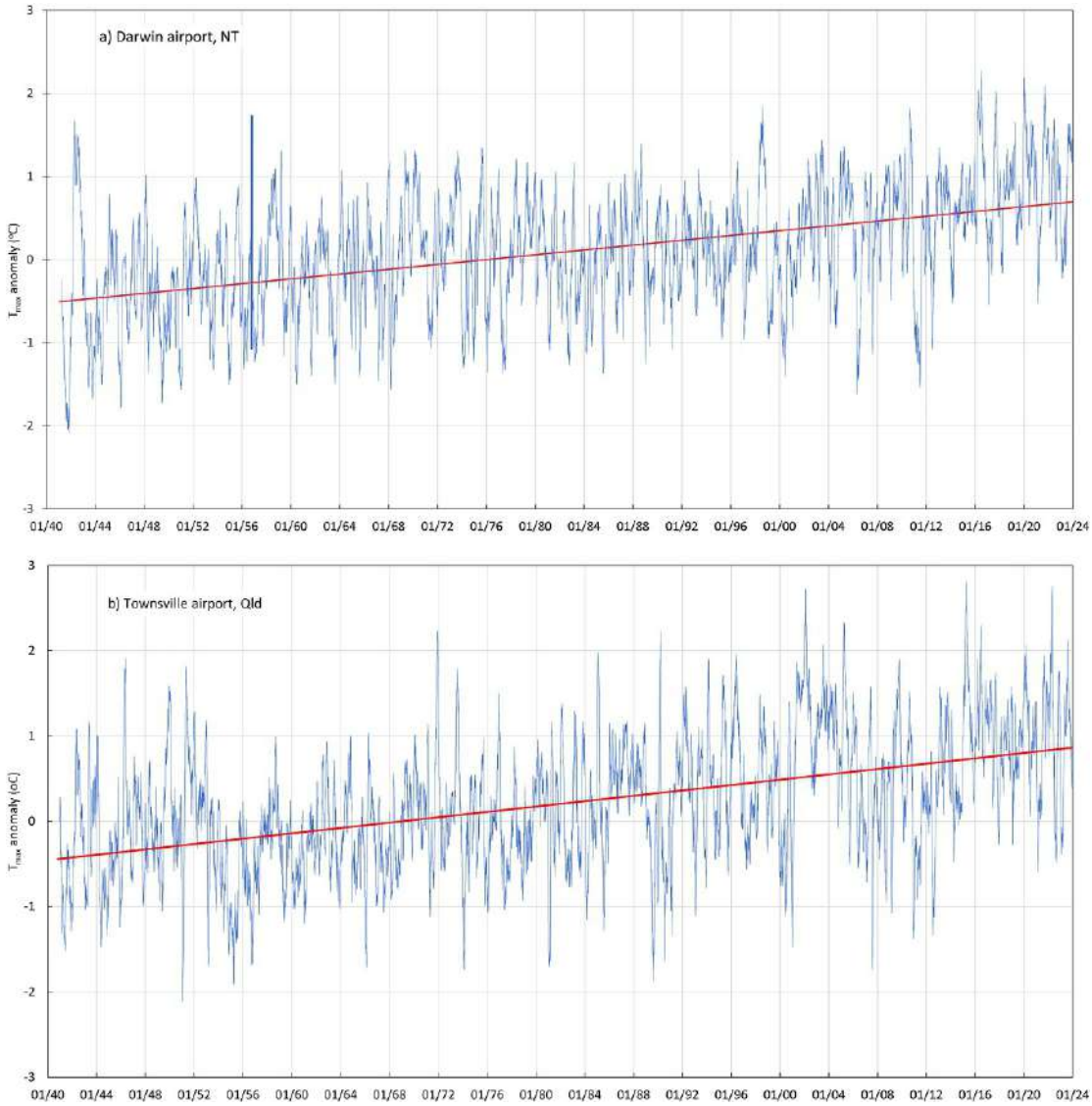

FIG. 19.7 Daily maximum air temperature (T_{max}) anomalies for two long-term weather stations in northern Australia, Darwin in the Northern Territory, and Townsville, Queensland. Data were taken from Bureau of Meteorology stations with anomalies calculated using mean monthly T_{max} for 1960–90. *Trends lines* are all significant ($P < .001$) with temperature increases in T_{max} of 1.2°C and 0.9°C year^{-1} across these two sites.

Climate modeling (CMIP6, http://pcmdi.llnl.gov/CMIP6) suggests rainfall across northern Australian may experience little change, with no significant trends evident (Gutiérrez et al., 2021), although observation suggest an increasing level of climatic variability (State of the Climate, 2022). CMIP6 modeling of maximum surface air temperature projects an increase across the northern Australian domain of ~1.6°C by 2050 under an SPCC 1-2.6 scenario relative to 1850–1990. This increases to ~2.4°C under an SSP5-8.5 scenario (IPCC Atlas, Gutiérrez et al., 2021). This is a projection of high confidence, supported by historic observations from three long-term climate stations in the NT and Queensland (Fig. 19.7), with temperature increases in T_{max} to date of 1.2°C, 1.3°C, and 0.9°C year^{-1} across these two sites, respectively. These rates of change are lower than Australia's current continental average warming of 1.47 ± 0.24°C since national records began in 1910 (State of the Climate, 2022).

Fig. 19.8 shows the seasonal breakdown of the fraction of days experiencing >35°C maximum temperatures. This thermal threshold is being exceeded with increasing frequency over the last three decades, with the 2020s experiencing a fivefold increase from August to December. This significant warming occurs during the late dry season and into the premonsoon months of October to December. In the highly seasonal climate of the NT region, this period is one of high stress, given that there has not been significant rainfall for some 6 months during a typical seasonal cycle. With climate change, this suggests the savanna flora

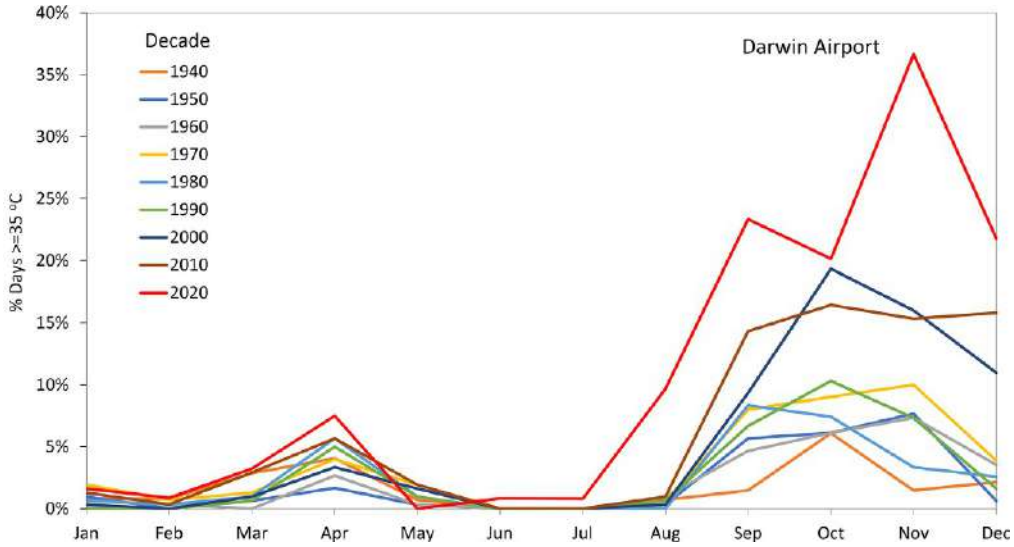

FIG. 19.8 Seasonal pattern of warming over time from 1941 to 2023. For each month, the number of days when T_{max} equaled or exceeded 35°C were averaged using data for each decade since 1940 for the Darwin airport station (Bureau of Meteorology, station 014015), one of the longer meteorological records in northern Australia.

and fauna need to cope with an increasing trend in maximal temperatures coinciding with the period of minimal available moisture, be it surface water, a critical resource for fauna in the late dry season, or soil moisture, a critical resource for vegetation in the late dry season, stressors will be amplified by the increasing climatic variability.

7 Conclusions

Australian savannas are a globally unique biome shaped by the evolutionary processes particular to the Australian continent, that of geographic isolation and seasonal aridity for the tropical savanna. The biota has experienced some millions of years of climate change and human influences, although the last 150 years of change are unprecedented with European settlement in an era of now rapidly changing climate. While intact and supporting a significant tourist industry, there is evidence that savanna landscape quality and biodiversity are declining, including within the conservation estate. Conservation management can address feral animals and threats posed by weed species plus unsuitable fire regimes, but there is a clear need to reform funding and approaches to biodiversity management (Legge et al., 2023). There is evidence that growing Aboriginal leadership via caring for Country programs facilitates far greater engagement of Indigenous communities in biodiversity conservation. Incentivized savanna burning programs undertaken by Aboriginal fire managers is reducing greenhouse gas emissions from savanna fires by adopting Indigenous-informed fire practices, which has now been operational for over a decade (Edwards et al., 2021) and fine scale, low severity fire likely has potential biodiversity benefits if long unburnt patches can be maintained (Perry et al., 2016). There is also evidence of slowly changing community attitudes toward a "nature-positive" world that values ecosystem services provided by natural landscapes, and using already cleared land for newer "smart" agricultural development. Such approaches could be beneficial in the Australia savannas given land clearing and degradation for much of the region is currently at management levels.

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Chapter 20

American Savanna: A brief review

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1 Introduction

This chapter deals with the American Cerrado, and the vegetation characteristics of each region of the Americas will be presented, as well as their biodiversity, conservation units, use and occupation, and monitoring strategies.

The terms Cerrado and Savanna will sometimes appear in the text as synonyms; however, it is necessary to separate their particular features for a complete understanding of each region, as there are several local and regional variations and peculiarities.

In the literature, more than 200 technical terms related to the word savanna are found, as many typologies are classified as such. Depending on the region, on the geographical distribution, and on the extension of the area, the quantification of biodiversity changes drastically and changes the concept of savanna as a consequence.

The term "savanna" originates from the American continent and was cited for the first time, in 1535, by Gonzalo Fernández de Oviedo y Valdés when describing savanna as "land without trees, but with a lot of tall and short grass" (Cole, 1986).

According to Walter (2006), the term savanna was used to describe vegetation types in the Caribbean and South America until the mid-19th century. Only much later the term savanna was associated to a vegetation type on the African continent.

According to de Gois Aquino et al. (2009), currently the term savanna has been widely used to designate different vegetations whose concepts are divided into two schools, the European and the American. For the European school, savannas only occur in the tropical zone and for the American school, savannas occur beyond the tropical zone, extending to the subtropical zone, including part of the steppe of the American continent.

According to the modern and broader concept, savanna can be defined as a vegetation covered by a continuous or discontinuous grassy layer with the presence of scattered trees and shrubs that can be found in South America, Africa, Oceania, and Asia. In this chapter, we will deal with the American savanna and its variations, according to the concept of the American ecological school.

Furthermore, in de Gois Aquino et al. (2009), Cerrado is defined as a vegetation complex that has ecological and physiognomic relationships with other savannas in tropical America and in other continents such as Africa and Australia. Cerrado is composed of a phytophysiognomic mosaic that includes grassland formations (e.g., Campo Limpo), savanna formations (e.g., Cerrado restricted sense), and forest formations (e.g., Matas de Galeria) (Ribeiro and Walter, 2008). Within this context, the vegetation classified as savanna presents intermediate structural characteristics between the grassland formations and the forest formations of Cerrado, for example, Campo Limpo, and Cerradão, respectively.

In this chapter, we will treat the Cerrado according to the definition of Ribeiro and Walter (2008).

2 Geographic location

In South America, the savanna is present in different countries, such as Brazil, where it is called "Cerrado," Colombia, Venezuela, and Guyana, also known as "Llanos" and "Rupununi Savannas," respectively. In Brazil, it occupies an area of 2,036,448 km^2, around 22% of the national territory. Its continuous area covers the states of Goiás, Tocantins, Mato Grosso, Mato Grosso do Sul, Minas Gerais, Bahia, Maranhão, Piauí, Rondônia, Paraná, São Paulo, and the Federal District. In addition, there are enclaves in Amapá, Roraima, and Amazonas. This territorial space contains the sources of the three largest hydrografical basins in South America (Amazônica/Tocantins, São Francisco, and Prata) and is the largest biome in South America.

In Colombia, Cerrado occurs in a region known as "Los Llanos Orientales," which extends across the east of the country, covering the departments of Meta, Casanare, Arauca, Vichada, Guainía, and Guaviare. This region borders Venezuela and is characterized by vast flooded plains and savannas, where the Cerrado develops and is not as extensive as in other countries.

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The main plant formation in Venezuela that resembles savannas is known as "Llanos," which are extensive flooded and grassland plains, with vegetation consisting of grasses and scattered shrubs. This region is characterized by a seasonal climate, with periods of drought and flooding.

In Central America, savannas are found in Costa Rica, Panama, and Honduras. In North America, although they are not so extensive, there are savanna areas in southern Florida, in the United States, such as the Everglades. In Costa Rica, savannas are mainly found in the northwestern region of the country known as "Guanacaste." The Guanacaste area is home to the Palo Verde National Park and Lomas Barbudal Biological Reserve, which are known for their vast floodplains, grasslands, and wooded savannas. These areas are important ecosystems for the conservation of biodiversity and are home to a variety of animal and plant species adapted to this type of environment.

In Panama, savannas occur mainly in the region known as the "Azuerol Peninsula" or "Península de Azuero." This area is located on the Pacific coast, in the southwest of the country. Panama's savannas are characterized by low vegetation, grasses, and shrubs, and are often used for livestock grazing.

The savannas found in Honduras are in some areas along the Caribbean coast, especially in the regions of Gracias a Dios and Colón. These areas are known for their floodplains and savanna vegetation.

In the United States, the American Cerrado is known as "prairie" and is found mainly in the Great Plains, a vast central region of the country from Texas to Canada. This area includes states of Kansas, Nebraska, Oklahoma, Iowa, Missouri, and South Dakota, among others. The American prairies are characterized by vast expanses of grasses, with few trees and scattered shrubs. The vegetation is adapted to continental climate conditions, with hot summers and cold winters and is an ecosystem rich in biodiversity, housing a variety of native plants, such as buffalo grass, wheatgrass, mugwort, and cacti. Additionally, these areas are home to a wide variety of animals, including bison, antelope, coyotes, birds of prey, and small rodents.

However, it is important to highlight that the American prairies have been significantly affected by conversion to agriculture and urbanization. Many areas have been converted into agricultural land or pastures for livestock, which has also happened in the case of the Brazilian Cerrado.

The Brazilian Cerrado, according to data from the Mapbiomas database, a collaborative network, formed by NGOs, universities, and technology startups, already lost approximately 2.91 million hectares between the years 1985 and 2022, corresponding to 25.2% of net loss, with 47.9% of native vegetation remaining and 50.1% occupied by agriculture and 0.7% by water bodies (Mapbiomas, 2022).

3 Biodiversity

In terms of fauna and flora, the Colombian and Venezuelan Los Llanos occupy a vast region of floodplains in South America, and are known for their incredible biodiversity. In terms of fauna, Los Llanos is the region where the jaguar, one of the largest felines in Americas, can be found. Other felines, such as spotted jaguar and puma, can also be found in these vast plains.

In addition to felines, the capybara and the marsh deer roam the flooded plains searching for food; the birdlife of the Llanos is also significant, with more than 300 species of birds, including the "tuiuiú," some eagles species, the jabiru, and a variety of herons and flamingos. These migratory birds take advantage of rivers and lakes to feed and reproduce, making Los Llanos an important stopping point along their migratory routes.

The Llanos are also home to a rich diversity of mammals, as large herbivores such as capybara and marsh deer are common in the region and can be spotted while grazing on the floodplains. River dolphins, such as the pink dolphin and the tucuxi, can also be found in the rivers and lakes of the Llanos, adding even more to the diversity of the local fauna.

Reptiles are found in the Venezuelan Llanos, and several kinds of alligators, and the Pantanal caiman, are commonly found in the waters and banks of rivers, as well as turtles, such as the Amazon turtle and the soft-shelled turtle. Snakes such as the "sucuri" and rattlesnake are also found in the region.

Los Llanos is characterized by extensive areas of savannas and riverine vegetation. In the flooded areas, there are dense gallery forests, mainly composed of species such as "buriti," "acacia," and "samaneiro." These forests provide shelter and food for many animals, as well as playing a vital role in protecting river banks from erosion.

More than 300 species of birds can be found in the Guanacaste region, making it a paradise for bird watchers from all over the world. Among the species are the royal heron, the tiger heron, the harpy eagle, the "tuiuiú" and a wide variety of ducks, geese, and herons. Mammals are also representatives of the local fauna and the anteater, white-tailed deer, and spider monkey can be found.

The region has several humid areas and mangroves, where several species of fish and crustaceans can be found, and the waters are home to sea turtles, such as the green turtle and the hawksbill turtle, as well as several species of tropical fish and mollusks. The regional flora is vast and abundant and "Guanacaste," "caroba," "cedar," and "catuaba" can be found, as along with red mangroves and white mangroves.

The Panama savanna is a unique and fascinating region known for its impressive biodiversity. The fauna of the Panama savanna is diverse and includes a variety of

mammals, birds, reptiles, and insects. Among the notable mammals are the anteater, sloth, and white-tailed deer, which are adapted to the seasonal conditions and undergrowth of the savanna. Furthermore, the region is also home to feline species such as the ocelot and the Moorish cat, which have adapted to life in this open environment.

The birdlife of the Panamanian savanna is equally rich, with several species found in the region, including the caracara, the yellow-headed vulture, and the snail hawk. Additionally, the open plains provide an ideal habitat for birds of prey such as the peregrine falcon and golden eagle, which can be seen gliding through the skies in search of prey. Reptiles such as iguanas and snakes are also present in the Panamanian savanna. The black iguana, a species endemic to the country, is a notable example of the region's reptilian biodiversity. Furthermore, species of snakes such as the *boa constrictor* and coral snake are found in the most wooded areas and along river banks.

Grasses, such as macega grass and fat grass, are dominant in the open landscape. These types of plants play a vital role in maintaining savanna structure, by providing food for the herbivores that inhabit the region. Trees and shrubs like white oak, "ipê," and "cedar" can also be found, mainly in more humid areas, such as along watercourses.

The fauna of the Honduran savanna is adapted to the specific conditions of this open environment. Among the notable animals are birds, such as the yellow-headed vulture, the "snail hawk," and the "caracará," which are often seen hovering over the landscape Looking for food. Furthermore, several species of migratory birds use the savanna as a stopping point during their migratory routes.

Mammals such as the tamandua, white-tailed deer, and armadillo are found in the savanna of Honduras, in addition to the spider monkey. Reptiles are also an important part of the biodiversity of the Honduras savanna, where lizards, such as the green iguana and black iguana, can be seen along river banks and in more wooded areas. Furthermore, several species of snakes, such as the rattlesnake, are also found in the region, playing their role in the food chain and maintaining ecological balance.

The flora of the Honduras savanna is adapted to seasonal conditions and water availability, with the dominant vegetation made up of tall grasses, such as fat grass and star grass, as well as trees and shrubs, such as white oak, "guayabo," and juniper.

The American prairie is an ecosystem that stretches across central North America, known for its vast biodiversity and natural beauty. This open landscape, characterized by extensive areas of grasses and wildflowers, is home to an impressive variety of species adapted to this unique environment. The fauna of the American prairie is diverse, herbivores play an important role in this ecosystem, and species such as bison, antelope, and rabbits are commonly found.

The American prairie is also home to birds, including hawks, prairie's rooster, American larks, and prairie's grouses. These birds specialize in open habitats and use the vast expanses of grasses to breed and feed. Additionally, many migratory birds use the prairie as an important stopping place American prairie during their journeys. Large mammals such as the gray wolf, coyote, and puma are common on the prairies.

The flora in the American prairie is composed by grasses like "capim-buffalo," Indiangrass, and andropogon grass dominating the landscape, with extensive green carpets. Additionally, a variety of colorful wildflowers, such as wild sunflower, aster, and coneflower, are present on the prairie, attracting important pollinators such as butterflies and bees.

The Brazilian Cerrado is considered the richest savanna in biodiversity in the world, home to an impressive variety of fauna and flora. This unique and diverse biome is home to thousands of species adapted to their specific characteristics, playing fundamental roles in ecosystems and contributing to the dynamics in the different physiognomies of the biome.

The Cerrado flora is marked by a great plant species diversity. From the point of view of biological diversity, the Brazilian Cerrado is recognized as the richest savanna in the world, housing 11,627 species of native plants already cataloged. There is a great diversity of habitats, which determines a notable alternation of species among different phytophysiognomies. About 199 species of mammals are known, and the rich birdlife comprises about 837 species. The numbers of fish species (1200), reptiles (180 species), and amphibians (150 species) are high. The number of endemic fish is not known, but the values are quite high for amphibians and reptiles: 28% and 17%, respectively. According to recent estimates, the Cerrado is home to 13% of butterflies, 35% of bees, and 23% of termites in the tropics. Trees like "pequi," "baru," "Jatobá," and "Ipê" are emblematic of the biome and play an important role in feeding various animals, as well as providing shade and shelter.

The Cerrado's fauna is very important, with a wealth of animal species that have adapted to the challenging conditions of the biome. Among mammals, we can find the spotted jaguar, the maned wolf, the giant anteater, and the giant armadillo. The diversity of birds is also notable, with species such as the green-billed toucan, the king vulture, the rhea, the "carcará," and the woodpecker.

Reptiles such as the broad-snouted caiman, the anaconda, the "jararaca," and the rattlesnake are also part of the Cerrado's rich fauna, as are amphibians such as "rãzinha do Cerrado" and the "perereca verde." Furthermore, the biome is home to a variety of insects, fish, and small vertebrates that play important roles in the food chain and ecosystem dynamics.

In the Cerrado, it is possible to observe a significant interaction between animals and plants. An example of this interaction is the relationship between the "lobeira" species (*Solanum lycocarpum* A. St.) and the maned wolf, whose main food, or favorite, is the fruit of "lobeira." The relationship between them is beneficial for both, as one can obtain food and allows the seeds of the tree species to be dispersed throughout the fields and open areas of the Cerrado.

4 Preservation areas and their ecological indicators

Conservation units' main function is to preserve species "in situ," avoiding land use and occupation from contributing to the extinction and/or reduction of the species' population.

Thirty conservation units can be found in the American prairies, which stand out for carrying out fauna and flora monitoring research.

In Brazil, Cerrado is preserved in 388 conservation units, corresponding to only 8.3% of its total area, being 3% of full protection and 5.1% of sustainable use.

5 Antropic activities and their impacts (urban, infrastructure, agriculture, and/or production activities)

The Brazilian Cerrado was occupied gradually and constantly, in a complex process, influenced by several historical, economic, and social factors.

In addition to human occupation and occupation with infrastructure, agriculture was and has been an important factor in the opening of new areas, leading to the removal of native vegetation from the Cerrado.

According the Mapbiomas database (2022), 50.1% of the Cerrado area is occupied by agriculture, which corresponded to 99 million hectares in 2022. This exponential growth in agriculture has led to the conversion of natural areas into pastures and agricultural plantations, resulting in environmental impacts such as deforestation, leading to the loss of biodiversity and soil degradation, the fragmentation of habitats and a decrease in the quality and quantity of water resources, in addition to increased greenhouse gas emissions, which in 2020 reached values around 100 $MtCO_2$/year.

With the expansion of agriculture in Cerrado, there is also the local removal of traditional and Indigenous communities from their lands, the loss of access to natural resources essential for the subsistence of families, and the intensification of land conflicts in the region.

The expansion into areas of native vegetation is continuous and the region with the greatest occupation today is the region called MATOPIBA, a region formed by the state of Tocantins and parts of the states of Maranhão, Piauí, and Bahia, where strong agricultural expansion occurred from of the second half of the 1980s, especially with grain cultivation. The name is an acronym formed by the acronyms of the four states: Maranhão, Tocantins, Piauí, and Bahia.

Matopiba's agricultural production is marked by large grain harvests, especially soybeans, corn, and cotton. There are around 4,800,000 ha planted with soybeans, which resulted in a total production of 18.5 million tons in the 2022/23 harvest, which represents around 12.3% of the total Brazilian production.

However, the conversion of these areas causes changes in ecosystems, contributing to climate change and consequently affecting water availability, which can bring irreversible impacts to regions that have few areas covered with residual Cerrado vegetation.

There are many challenges enforced by the occupation of Cerrado, which makes it essential that sustainable policies and practices be adopted that reconcile agricultural and livestock production with environmental conservation and social development. To achieve this, strategies for forest management, low-carbon agriculture, strengthening the rights of traditional and Indigenous communities are necessary, in addition to encouraging research and innovation focused on the countryside.

6 Operational monitoring tools and potential to conserve or recover the biome

Although the savanna is widespread in the Americas, all areas have already suffered a process of degradation and vegetation suppression. Both degradation and suppression, which can also be reported as deforestation, have been monitored through local tools in the field and through remote sensing tools that track and assess conditions and changes in ecosystems. These tools are important for providing up-to-date data and information that assists in the management and conservation of the American Cerrado.

Remote sensing is a technique that uses images captured by satellites, aircraft, and drones to obtain information about vegetation cover, species distribution, and other characteristics of the Cerrado. These images are processed and analyzed to identify patterns and changes over time, which can help monitor the expansion of agriculture, deforestation, and changes in vegetation.

Weather stations are equipment that collect climate data, such as temperature, humidity, precipitation, and solar radiation. This data is essential for understanding climate patterns and environmental conditions in the American Cerrado. They help to monitor seasonal variations, extreme events, and climate change that can affect the ecosystem.

Using data collected at weather stations and through remote sensing, mathematical models can be established to simulate and predict patterns and processes in the

American Cerrado. Models help to understand how environmental changes and human interventions affect ecosystem dynamics. They can also be used to assess future scenarios and support decision-making related to management and conservation.

Around the world, the Global Forest Watch (GFW) platform provides data and tools for monitoring forests using cutting-edge technology that allows anyone to access information about where and how forests are changing around the world.

In terms of information and to illustrate the power and ability to join information, one of the platform's tools shows the loss of vegetation across countries. In Colombia, the loss was 258 kha of natural forest, equivalent to 160 Mt of emissions of CO_2 in 2022. At GFW, it is possible to find data for all countries, including information on climate, fires, and biodiversity.

In Brazil, the Mapbiomas platform stands out, as it specifically monitors all six biomes, including the Cerrado.

From monitoring, it is possible to observe areas where vegetation was removed, either authorized by public bodies or illegally. However, in order to advance in the preservation of the Brazilian biome, it is important to plan the ecological restoration of ecosystems, so that their functioning is reestablished.

For the processes to be reestablished, it is important to know the landscape and especially the level of resilience of the place, which is the capacity that a certain ecosystem has to return to conditions close to the original ones, before the disturbance, alone and without the help of human activity. The greater the local resilience, the lower the degree of human intervention for restoration.

Among the techniques, the best known and most widespread is the planting of seedlings, which depends on material of excellent origin so that they can survive in a new location, even when they are still small. However, although it is a widely used technique, it depends on the area to be restored.

In the Brazilian Cerrado, as well as in other American savannas, the vegetation is not always arboreal; sometimes it is low, with a predominance of grasses, in which the technique for planting seedlings will not be appropriate. To restore these environments, there is the possibility of using direct sowing of seeds of herbaceous species, grasses, which have the potential to germinate directly in the soil. In these formations, the introduction of tree species, whether by planting seedlings or direct sowing, is not recommended, due to the high probability of failure and the future decharacterization of the areas.

Conservation of the biome also has great potential, mainly through the creation and maintenance of Conservation Units, the UCs. UCs are, in many countries in America, regulated by law and maintained by the state, as a way of conserving flora, fauna, and ecosystem services, in addition to protecting archeological sites.

7 Conclusions

This chapter addressed the Cerrado in the Americas and presented the unique and important biodiversity and ecosystems found there. The richness found in the different countries where the Cerrado/savanna occurs, both in fauna and flora, is incomparable, housing endemic species and a high number of species.

However, natural or anthropogenic factors have exerted strong pressure on the Cerrado/savanna areas, resulting in significant environmental impacts, such as deforestation, habitat loss, and ecosystem fragmentation.

We conclude, therefore, that the conservation and sustainable management of areas are imperative not only for the preservation of biodiversity, but also to guarantee the well-being of the human communities that depend on these areas and to ensure life on the planet.

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Section G

Subterranean

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Chapter 21

Subterranean ecosystems

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1 Introduction

Subterranean biomes include a network of interconnected voids that are filled with air or water and that vary in size from large caverns to capillaries. Biologically, these spaces can be roughly divided into three size classes: (1) caves and caverns, which are accessible to humans and other larger animals, (2) intermediate-sized voids or mesocaverns, which support invertebrates and smaller vertebrates, and (3) capillaries or microcaverns, which mostly house aquatic meiofauna and microbes. Most cavernous substrates include a continuum of all three size classes in an anastomosing system. Organisms occupying these voids vary in their relationship to life underground: (1) species that are restricted to living entirely underground (troglobionts), (2) species that can complete their life cycle underground but also survive in similar surface habitats (troglophiles), and (3) species that regularly enter caves for shelter or food (trogloxenes) (Fig. 21.1). Most interesting are the obligatory, subterranean-adapted species, as these often display unique morphological, physiological, and behavioral traits that improve their ability to exploit underground spaces and resources. The food energy supporting subterranean communities is mostly allochthonous, that is, produced on the surface and brought underground by water, gravity, air currents, animals (e.g., bats), and plants (e.g., roots). Therefore, nearly all subterranean habitats heavily depend on surface ecosystems. External ecosystems not only supply organic resources to subterranean environments (e.g., organic detritus), but also support species of troglophiles and trogloxenes. Additional energy is supplied by chemoautotrophic microorganisms that rely on minerals as their energy source. In fact, a few subterranean food webs are based almost entirely on chemoautotrophic microbes (Sarbu et al., 1996; Frumkin et al., 2023).

2 Geographic setting

The subterranean biome comprises one of the most abundant and widespread biomes on land, covering at least 20% of the Earth (Sánchez-Fernández et al., 2021). Examples occur from frozen high latitudes to tropical lowlands, and within many different substrates. In concert, they contain the largest reservoir of liquid freshwater on Earth (Retter et al., 2021). The majority of caves and other subterranean habitats that have been studied occur in more soluble rock formations, particularly carbonate rocks. These types of rocks are frequently associated with a unique landscape known as karst. Karst systems are widespread across the globe, and while the physicochemical processes governing the dynamics of these systems remain relatively uniform, the appearance of karstic landscapes can vary significantly due to factors such as latitude, climate, precipitation, vegetation type, and topographic relief (Fig. 21.2).

Subterranean habitats also commonly form in many other rock types and substrates, including ice, lava, salt, iron ore, hyporheic, and poorly consolidated material, such as talus slopes and riverine deposits, and on all continents and most islands (Fig. 21.3) (Howarth and Wynne, 2022). There are undoubtably 100s of thousands of caves, and even a greater number of networks of mesocaverns supporting subterranean ecosystems. As an example, Brazil currently has around 24,000 registered caves, although some projections suggest the existence of more than 300,000 caves in the country (Cruz and Piló, 2019). These cavernous areas are clustered in separate isolated island-like ecosystems. Each cavernous area contains numerous more or less interconnected subterranean habitats, which are separated by noncavernous substrates (see Fig. 2 in Mammola, 2019).

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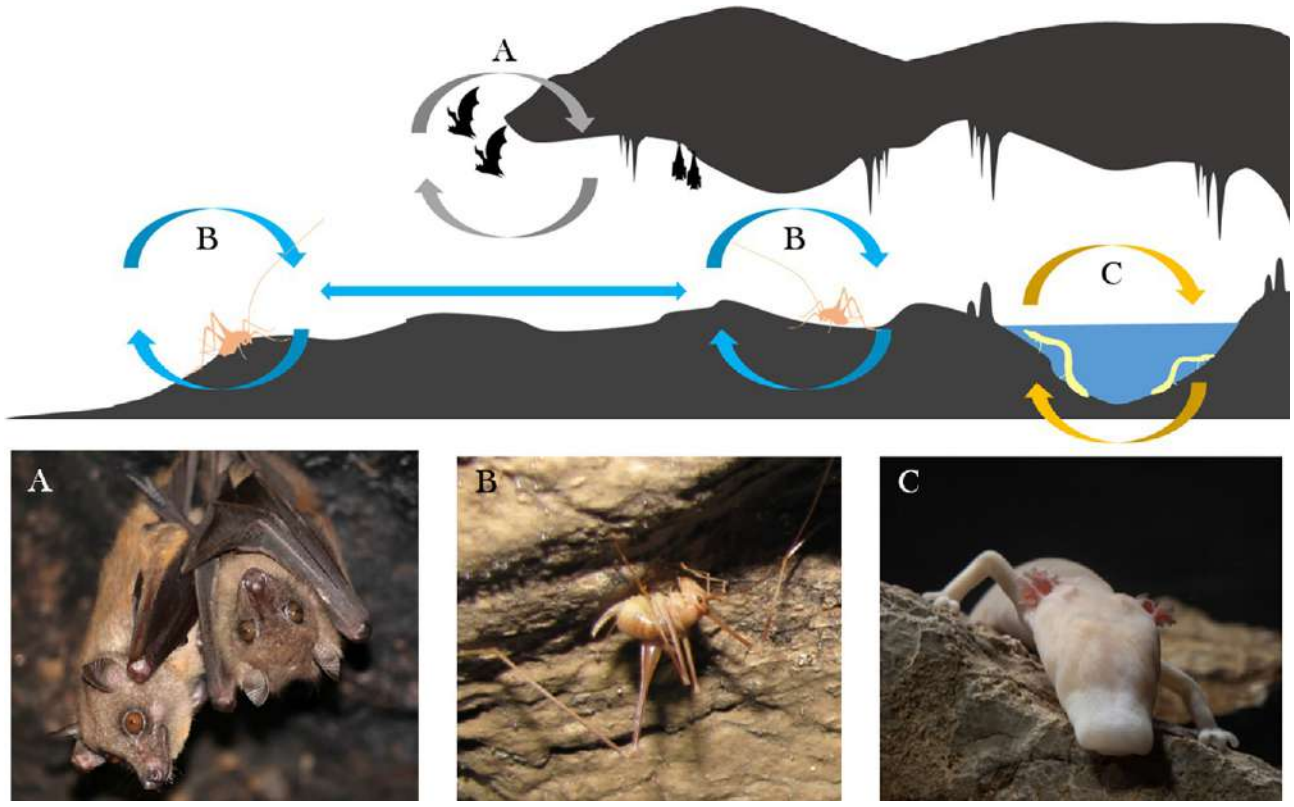

FIG. 21.1 Examples of cave fauna. (A) Trogloxenes, which use cave habitats but depend on external environments to complete their life cycle; (B) troglophiles, which can complete their life cycle in both external and subterranean environments; and (C) troglobionts, which are exclusively found in subterranean habitats, thus being unable to complete their life cycles in surface habitats. *(Photo credits: (A) Marconi Sousa Silva; (B) Rodrigo Lopes Ferreira; and (C) Teo Delic.)*

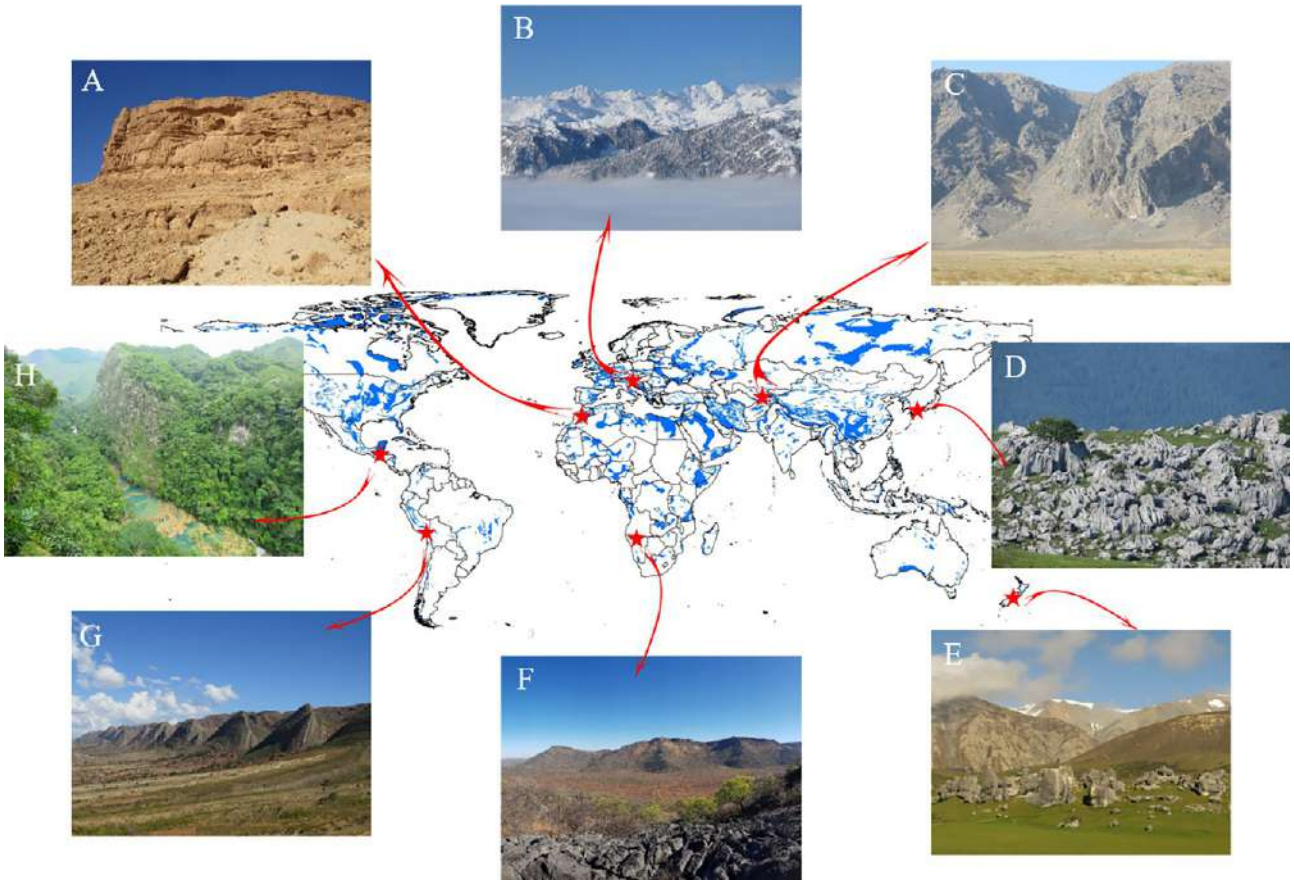

FIG. 21.2 Distribution of karst landscapes across the globe. Blue areas in maps are karst patches. (A) Desert karst landscape in Morocco; (B) alpine karst in Slovenia; (C) steppe karst in Uzbekistan; (D) subtropical karst in Japan; (E) subalpine karts in New Zealand; (F) semiarid karst in Namibia; (G) Andean karst in Bolivia; and (H) tropical karst in Guatemala. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: Rodrigo Lopes Ferreira.)*

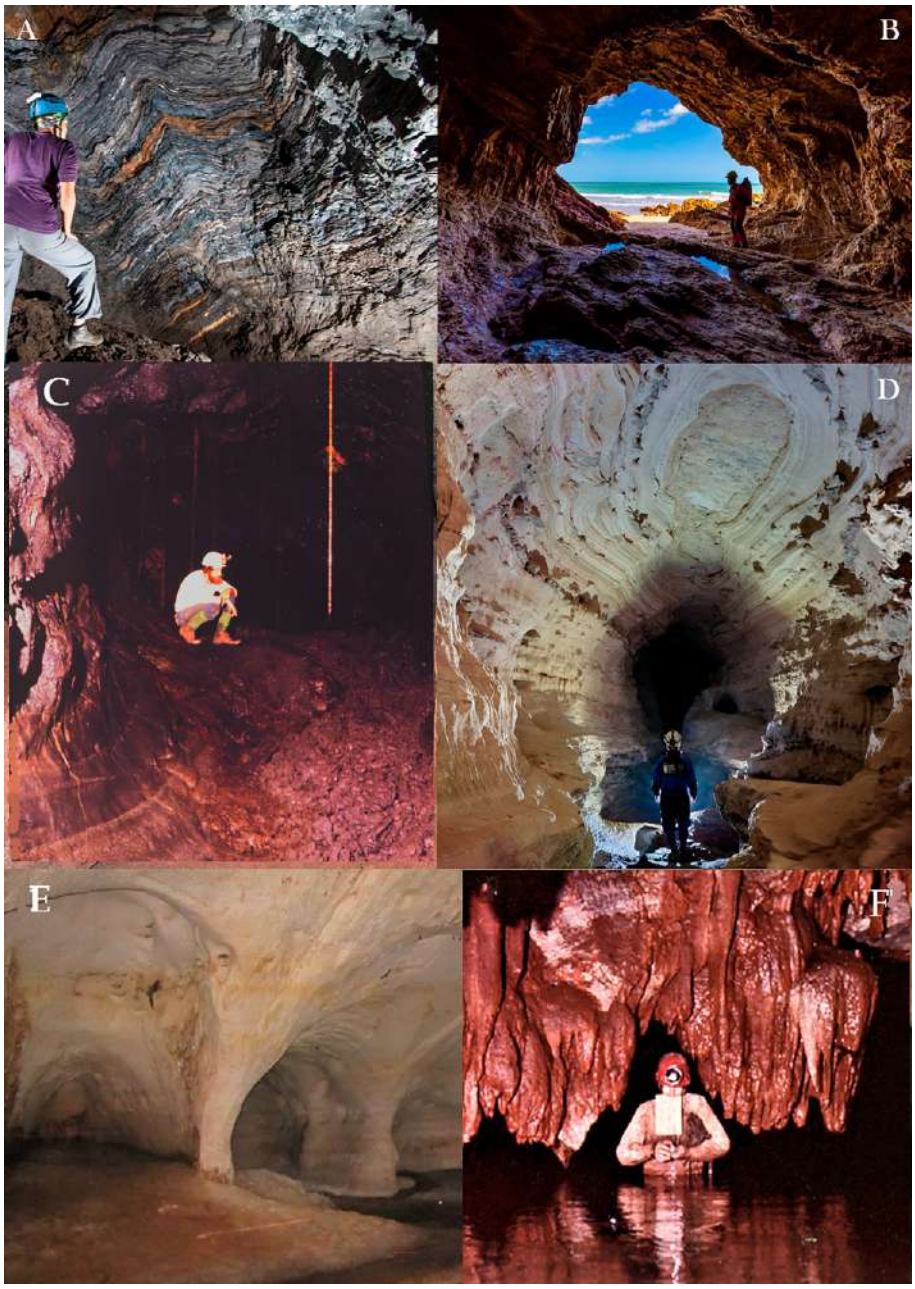

FIG. 21.3 Interiors of caves in different substrates. (A) Iron ore cave in southeastern Brazil; (B) granite cave in Northeastern Brazil; (C) lava tube in Hawaii; (D) quartzite cave in southeastern Brazil; (E) sandstone cave in northern Brazil; and (F) limestone cave in Northern Thailand. *(Photo credits: (A) Augusto Gomes; (B) Diego Bento; (C) Francis G. Howarth; (D) Rodrigo Lopes Ferreira; (E) Rodrigo Lopes Ferreira; and (F) Nancy C. and Francis G. Howarth.)*

Note that surficial soil habitats are not discussed in this chapter, as soil ecosystems are closely associated with the overlying surface environments, and soil science is considered a separate discipline from (bio)speleology.

3 Biodiversity

Subterranean environments harbor a diverse array of organisms, which display different levels of specialization to live underground. Generally, surface taxa that exhibit preferences for moist and shaded habitats, along with a generalist diet, have a greater potential to successfully colonize and establish populations underground (Culver and Pipan, 2009). Furthermore, subterranean habitats harbor a high diversity of microorganisms (Retter et al., 2021). Examples of a few of the many distinct subterranean habitats are shown in Fig. 21.4. Even though each individual subterranean ecosystem supports relatively few species of

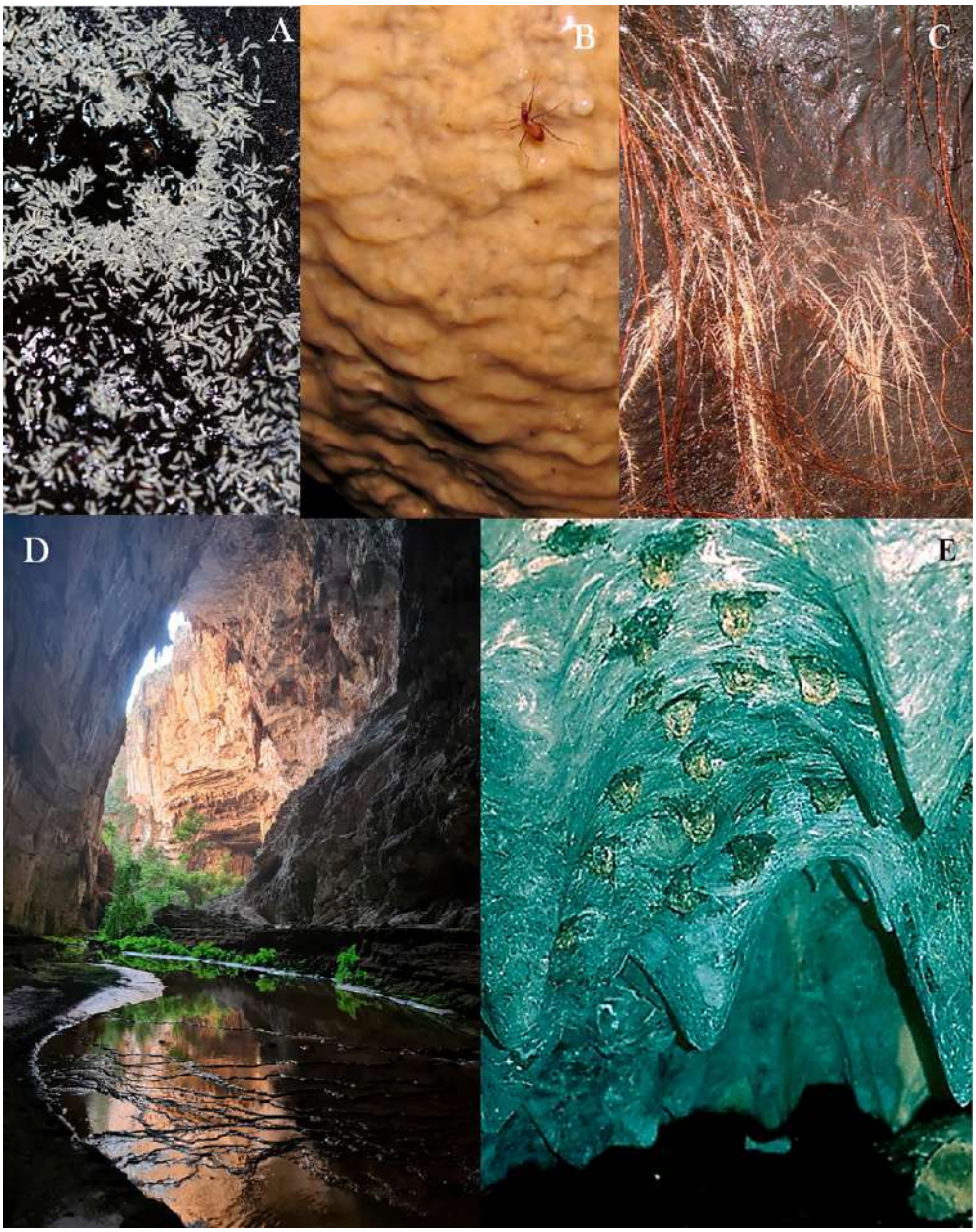

FIG. 21.4 Examples of habitats and microhabitats inside caves. (A) Guano pile produced by hematophagous bats with troglobiontic springtails (*Acherontides* sp.) in southern Brazil; (B) dripstone with a troglobiontic beetle (*Hadesia vasiceki*) in Bosnia-Herzegovina; (C) root mats in a lava tube in Hawaii with roots of the nonnative tree (*Myrica faya*) (white) and the native tree *Metrosideros polymorpha*) (brown); (D) cave stream in Brazil; and (E) cave swiftlet nests (*Aerodramus* sp.) on ceiling of limestone cave in Chillagoe, Queensland, Australia. *(Photo credits: (A) Rodrigo Lopes Ferreira; (B) Rodrigo Lopes Ferreira; (C) Francis G. Howarth; (D) Rodrigo Lopes Ferreira; and (E) Francis G. Howarth.)*

multicellular organisms compared to numbers of species in overlying surface habitats, the high numbers of local endemic species underground represent a large portion of local biodiversity (Vermeulen and Whitten, 1999; Gibert and Deharveng, 2002).

Only a tiny percentage of caves and mesocaverns have been surveyed, even within relatively well-known regions (Culver et al., 2013; Ficetola et al., 2019). For example, in well studied karst regions, such as Slovenia and West Virginia, less than 20% of the known caves have been investigated biologically, even superficially (Culver et al., 2013), and the accumulation curves for the number of species have not reached an asymptote (Culver et al., 2004, 2006). Culver and Holsinger (1992) estimated that there should be from 50,000 to 100,000 obligate cave-dwelling species worldwide, and indeed, hundreds of new species are discovered each year. Furthermore, there are many regions that remain virtually unsampled, especially

in the tropics (Deharveng and Bedos, 2019; Howarth, 2022). An even greater number of facultative cavernicoles and periodic inhabitants of caves are known. Although the biodiversity of subterranean habitats remains poorly known, even to an order of magnitude, the existing data are sufficient to allow a qualitative assessment of the environmental impacts of climate change.

4 Conservation areas and their ecological indicators

Although buffered by the surrounding rock, subterranean ecosystems are vulnerable and threatened by novel environmental stressors, that is, events with which the community has had no prior experience. These include overexploitation, pollution, habitat destruction, nonnative species, and climate change (Petty, 1976; Clements et al., 2006; Sánchez-Fernández et al., 2016; Mammola et al., 2018, 2019a, 2020, 2022; Niemiller and Taylor, 2019; Wynne et al., 2021). In the presence of these threats, many subterranean species have been recently assessed as endangered according to criteria of the International Union for Conservation of Nature (IUCN) "Redlist of Threatened Species" (e.g., Borges et al., 2019; Reboleira et al., 2022; Milano et al., 2022) as well as by regional assessments, such as the Endangered Species Act in the United States (Elliott, 2000). Ominously, a few subterranean species are believed to be extinct (Elliott, 2000; Rodrigues et al., 2012). Because the habitat is discrete and island-like, populations of subterranean-restricted species are often small and can be severely affected by human activities on both the surface and underground. This chapter focuses on the potential effects of climate change on the subterranean biome, a topic which is recognized as a critical question in subterranean biology (Mammola et al., 2020). We also include a review of temperature regimes within subterranean environments to help better understand the effects of climate change.

5 Temperature of subterranean habitats

The environment of caverns is zonal, with five zones recognized: entrance, twilight, transition, deep, and stagnant air. The entrance zone represents the boundary between the surface and underground. The twilight zone extends from the limit of vascular plant growth to total darkness. The transition zone is dark but still affected by weather events on the surface. The deep zone lies beyond the transition zone and is characterized by a nearly constant temperature and negligible or no evaporation. In long, complex caves, a stagnant air zone may occur and is distinguishable by calm, stagnant air and condensing water vapor (see Fig. 3.3 in Howarth and Moldovan, 2018). The boundaries between each of the three dark zones are often dynamic and dictated by passage shape and climate.

The temperature deep within caves often approximates the mean annual surface temperature over the cave, but several factors can affect cave temperatures in different passages (Badino, 2010). For example, the temperature in the entrance, twilight, and transition zones, as well as in shallow subterranean habitats, fluctuates in response to weather and diurnal and seasonal changes (Howarth, 1982, 1983; Tobin et al., 2013; Mammola et al., 2016). Temperatures in the deep zone are relatively constant but may change slowly due to weather events on the surface. The deep zone is often warmer than expected, due in part to the retention of heat by passage shape, such as constrictions limiting air circulation and dead-end upper-level passages which trap heat. Dead-end passages that slope downward trap cooler air (Williams, 1954; Cigna, 2004; Wind et al., 2022). In addition, caves with high concentrations of CO_2 are usually warmer than expected since CO_2 is a greenhouse gas, which absorbs and stores heat.

Excess heat enters subterranean environments with incoming water and air, respiration by resident organisms (including plant roots), by composting of organic material, and by geothermal energy. In addition, caves in iron ore deposits can be heated by exothermic reactions between the ore and incoming water (Piló et al., 2022). Large deposits of flood debris or guano can provide a significant amount of heat by respiration and composting (Harris, 1970; Moulds, 2006; Ladle et al., 2012; Piló et al., 2022).

The increase in temperature by geothermal heat is small and difficult to detect near the surface (Badino, 2005; Lucazeau, 2019). However, in the cave deep zone and mesocaverns, in which the atmosphere is stable for prolonged periods, the small input of heat from below would increase the temperature. Since groundwater absorbs geothermal energy flux, the temperature change is dampened in overlying voids (Badino, 2005; Luhmann et al., 2015). In addition, some roosting bats rely on geothermal heat (Levin et al., 2015).

Incoming water and moist air can significantly change the deep zone temperature (Badino, 2005; Brookfield et al., 2016; Saccò et al., 2021), but temperature change is attenuated toward the mean annual surface temperature with increasing distance from the input (Kambesis et al., 2013; Howarth, 1982). Both liquid water and water vapor have such a high specific heat capacity compared to dry air and the substrate that pulses of incoming water from moist air, rain, and sinking streams have a much greater effect on cave temperatures than incoming dry air. In addition, because of its high specific heat, the presence of water buffers the cave temperature. The large conduits present in soluble rock, (e.g., karst) allow rapid influx of surface water, which can significantly affect groundwater temperature (Brookfield et al., 2016).

Solar radiation increases the surface temperature, especially of exposed rock; however, this heat is not conducted

directly into deep underlying voids, because land surfaces are poor heat conductors and because heat is generally conducted upward. However, this surficial heat can warm the air and water that subsequently enter caves. Heat leaves caves by airflow and exiting water, especially in more open caves and those with several entrances or with exiting streams. Heat also leaves caves by slow conduction upward to the surface.

Importantly, temperature dynamics in caves can be intricately linked to biological processes. For example, large colonies of roosting bats and birds can increase cave temperatures near their roosts (Lundberg and McFarlane, 2009). Indeed, many cave chambers, especially in the tropics and subtropics, have gained recognition for their remarkable ability to attain exceptionally elevated temperatures, reaching approximately 40°C. These caves (known as "hot caves" Tejedor et al., 2005; Ladle et al., 2012) are characterized by distinct features. Notably, the elevated temperatures are attributed to respiration and thermal radiation emitted by the densely packed bat and bird colonies residing in the chambers, as well as to the decomposition of their guano (Peck et al., 1998). Development of hot caves is contingent on specific geomorphological attributes that include a single, relatively small entrance, limited airflow, a high concentration of bats or birds (often numbering in the tens to hundreds of thousands of individuals), consistently warm ambient temperatures ranging from 28°C to 40°C throughout the year, and relative humidity (RH) exceeding 90% (Silva-Taboada, 1979; De La Cruz, 1992; Rodriguez-Duran, 2009).

6 Direct and indirect impacts of climate change on subterranean habitats

As the planet warms, organisms physiologically and behaviorally adapt to the new conditions, shift their distribution toward new climatic optima, or become extinct (Bellard et al., 2012). Species restricted to subterranean habitats cannot migrate beyond the boundary of the terrain that encloses them (Sánchez-Fernández et al., 2016). Aquatic species are often more widely distributed because aquifers can connect cavernous terrains. However, in both aquatic and terrestrial systems, dispersal is slow, resulting in many isolated genetically distinct populations (e.g., Kane et al., 1992; Snowman et al., 2010; Wessel et al., 2013). Thus, climate change may extirpate the more vulnerable populations resulting in loss of taxonomic, phylogenetic, and functional diversity along with critical evidence to decipher the factors involved in the evolutionary ecology of obligatory cavernicoles.

With the climate becoming more unpredictable, the frequency, severity, and impacts of extreme meteorological and ecological events are increasing. Some of these phenomena affect subterranean systems, but most of these impacts have been rarely quantified due to the stochastic and unpredictable nature of these episodes, as well as the difficulty of studying their impacts in complex and largely inaccessible subterranean voids (see Vaccarelli et al., 2023 for a metaanalysis of available studies). Yet accumulating evidence reinforces the premise that subterranean climates are changing (Badino, 2004; Domínguez-Villar et al., 2015; Pipan et al., 2018; Retter et al., 2021; Sánchez-Fernández et al., 2021; Benz et al., 2022; Turner et al., 2022; Vaccarelli et al., 2023). Therefore, we predict that subterranean habitats are likely to be severely impacted (Mammola et al., 2018, 2019b). The more vulnerable subterranean communities include the fauna of glacier caves (Howarth, 2021), cave systems with seasonal or perennial ice (Colucci et al., 2016; Iepure, 2018; Wind et al., 2022; Reboleira et al., 2022), and other sensitive systems, such as small, isolated caverns, that have little topographic relief and caverns with few associated mesocavernous voids (Howarth and Wynne, 2022). Furthermore, "superficial" caves, such as iron ore caves, epikarst, and other shallow subterranean habitats, can be highly vulnerable to the impacts of climate change, due to their proximity to external environments (Mammola et al., 2016).

Negative impacts are both direct (i.e., changes in the subterranean environment that directly affect biota) and indirect (i.e., changes on the surface that impact subterranean habitats). However, both direct and indirect risks cascade jointly in difficult-to-predict ways, often acting synergistically, cumulatively, or even antagonistically. For example, the impacts of droughts, extreme and unseasonable storms, floods, and wildfires often reinforce each other and enhance their effects on water and nutrient inputs, erosion, vegetation cover, nonnative species invasions, and populations of animals roosting in caves. Increasing temperatures in bat hibernacula in temperate North America will make the caves more favorable for the recently established nonnative fungal disease (*Pseudogymnoascus destructans*), which will further threaten populations of vulnerable species of bats (Turner et al., 2022). The relative importance of these threats varies and depends on several factors including region, climate, geography, substrate type, and biotic factors, as well as land management, and laws and regulations and their enforcement (Vaccarelli et al., 2023). The main risks are summarized in Table 21.1, and the interrelationships diagrammed in Fig. 21.5.

6.1 Increasing temperature

Physiological tests suggest that obligatory subterranean animals have limited capacity to withstand temperature fluctuations (i.e., stenothermic), since they evolved in confined habitats that maintain a relatively constant temperature (reviewed by Mammola et al., 2019b; Pallarés et al., 2019). However, most studies have reported thermal

TABLE 21.1 Principal risks of climate change affecting subterranean habitats.

Climatic hazard	Environmental effects	Potential impacts to subterranean biome
Warming	Increased temperature, decreased relative humidity, changes in physics and chemistry of groundwaters	Physiological stress, desiccation of habitat, species disperse, or population extinction
Droughts	Decreased water, increased severity and frequency of floods and wildfires, increased groundwater extraction	Reduced water and nutrient input, desiccation of habitat
Wildfires	Increased severity and frequency of fire impacts	Reduced nutrient input, increased risk of floods, erosion, and sedimentation
Floods	Increased severity and frequency of erosion and sedimentation	Increased water infiltration, habitat disturbance and reduction by increased erosion, solution, and sedimentation, increased toxins
Storms	Increased severity and frequency of local squalls	Habitat disturbance and reduction by flash floods, increased erosion, solution, and sedimentation
Nonnative species invasions	Increased movement and colonization by nonnative species	Negative interactions with native species (e.g., competition, predation, parasitism), change in nutrient inputs
Sea level rise	Inundation of coastal habitats, increased storms, erosion, and toxic algal blooms	Inundation of subterranean habitats, habitat disturbance, increased erosion and sedimentation, saltwater intrusion into freshwater aquifers
Timing of disturbance events	Increased frequency and unpredictability of disturbances	Interruption of behavior, e.g., life cycle, reproduction, molting, roosting
Climate change mitigation programs	Adding soil on surface, groundwater extraction, alternative energy programs, e.g., dams and windmills	Habitat reduced, disturbed, and less suitable, reduced nutrient input. See Chapter 18 of Volume 3 for details

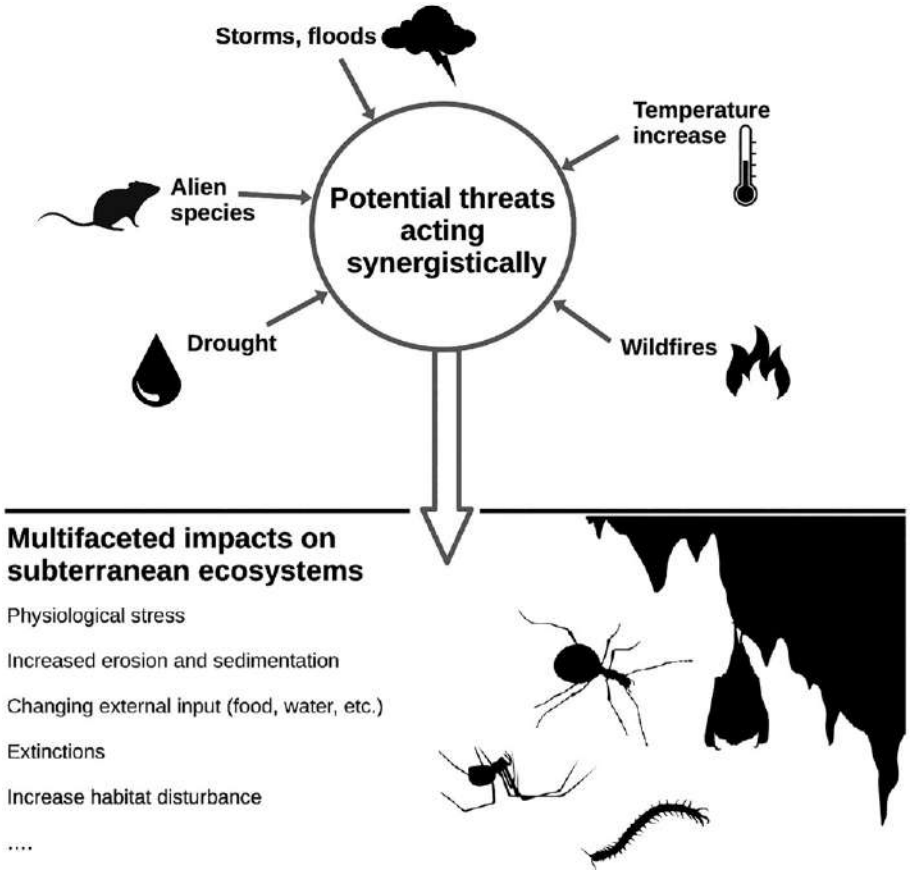

FIG. 21.5 The breadth of impacts of climate change on subterranean ecosystems. *(Credit: Stefano Mammola.)*

limits for single populations, and the results have been mixed (Vaccarelli et al., 2023). Some species can tolerate a narrow range of temperatures close to that of their subterranean habitat (Mermillod-Blondin et al., 2013; Rizzo et al., 2015), whereas others can tolerate a broader temperature range than the temperatures they experience in their habitat (e.g., Pallarés et al., 2020). The variable response may be attributed to different factors, including both the parameters and setup of the physiological experiments and biotic features (phylogenetic history of the taxa considered, their degree of subterranean specialization, and the extent and structure of the subterranean habitat). Concerning the latter, animals living in large caves that connect to extensive mesocavernous voids and encompass some topographic relief are likely to be more tolerant of temperature change, since mesocaverns would be further buffered and since cavernicoles can track changing temperatures by moving in three dimensions to a more suitable environment (Howarth and Wynne, 2022). For example, in the tropics, temperatures in the deep cave zone maintain a nearly constant temperature within individual passages (Mejía-Ortíz et al., 2020), but temperatures in individual passages in the same cave may differ by >5°C depending on passage shape, CO_2 concentration, and presence of a heat source or sink (Howarth, 2022).

The variable response to temperature tolerance also may be due to the study design; that is, in many studies, the animals were exposed to higher temperatures at rates faster than they would experience underground, giving them insufficient time to acclimate (Castaño-Sánchez et al., 2020). Additionally, only a single population was studied; e.g., Colado et al. (2021) examined the thermal response of a population of a pseudoscorpion in a cave at 10 °C in northern Spain, whereas the species occurs in a karst terrain with an elevation range of 25 to >1100 m a.s.l. and a temperature range of 10–14°C. Conversely, Pallarés et al. (2020) and Beasley-Hall et al. (2022) showed that cave-adapted animals are sensitive to sublethal physiological effects at above normal temperatures; thus, cavernicoles may be more vulnerable to climate change than indicated by thermal tolerance studies. In addition, relative humidity has rarely been adequately controlled in thermal tolerance studies, whereas our studies in Hawaii have indicated that relative humidity has a much greater influence on the behavior and survival of obligate cave species than does temperature (Howarth, 1980, 1983; Ahearn and Howarth, 1982; Hadley et al., 1981).

Because of the high thermal inertia of water, average temperatures in groundwater and deep cave zones would change slowly, perhaps only a fraction of a degree per year (Castaño-Sánchez et al., 2020; Domínguez-Villar et al., 2015) and even slower in deep mesocaverns. Unlike terrestrial animals in most surface environments, cave animals would rarely be exposed to heat shock from a sudden change in temperature, except beneath urban and industrial areas where temperatures may be elevated >5°C by infiltrating warm water from storm runoff and the urban heat island effect (Castaño-Sánchez et al., 2020). Also, because of thermal inertia, temperatures of aquatic habitats would usually be more stable than terrestrial habitats, and thus aquatic species might be more stenothermic than terrestrial species (Mermillod-Blondin et al., 2013; Castaño-Sánchez et al., 2020). However, even a small increase in groundwater temperature can have major negative consequences to water quality and populations of microorganisms with cascading effects on associated biota (Retter et al., 2021). Although most studies suggest many specialized cave species can tolerate a moderate rise in temperature, the long lifespan and low reproduction rates make recovery from disturbances problematic (Castaño-Sánchez et al., 2020).

Alternatively, subterranean habitats may function as thermal refugia for surface species that habitually shelter in caves (Moog et al., 2021), screes (Ledesma et al., 2020), and dolines (Marcin et al., 2022) during periods of unfavorable weather or other disturbances (de Fraga et al., 2023). Such refuges will become more important as extreme climate-caused events (e.g., heat waves, droughts, wildfires) become more frequent and severe (Mammola et al., 2019b). As an example, Castro-Souza et al. (2023) used species distribution modeling techniques to investigate the biogeographic history of Neotropical cave-dwelling crickets belonging to the genus *Endecous*. Their objective was to identify potential long-term environmental refuges in South America. Their findings revealed a projected significant reduction in the spatial distribution of this genus in the near future, suggesting an increased likelihood of isolation within subterranean ecosystems. Considering the ongoing impacts on surface environments and the anticipated effects of future climate change, the authors emphasized that conservation of caves and karst landscapes is expected to play a crucial role in safeguarding not only these crickets but also the associated subterranean communities. However, severe events at an unusual time may not provide appropriate cues to warn these animals to seek or exit shelters. Also, additional surface species entering caves will create new biotic associations with possible negative effects on resident species (de Fraga et al., 2023).

6.2 Droughts

The combined effects of added heat and reduced rainfall caused by climate change will increase the frequency and severity of droughts, which will have far-reaching impacts on a variety of habitats (Berg et al., 2016). In temperate regions, the increase in temperatures has a greater role in soil moisture than reduced rainfall (Schumacher et al., 2022). Sinking surface water carries nutrients and moisture underground; thus, droughts will decrease the amount of

water and nutrients entering subterranean habitats (Retter et al., 2021). Additionally, droughts reduce productivity on the surface, which affects the amount of available nutrients underground and populations of animals that enter caves. Prolonged droughts would desiccate subterranean habitats, posing a significant threat to cave-dwelling species that exhibit high sensitivity to dehydration (Ahearn and Howarth, 1982).

The effect of droughts on subterranean ecosystems is exacerbated by the extraction of groundwater. Depletion of groundwater directly impacts aquatic ecosystems by reducing the size of the habitat, altering nutrient availability, and heightening the risk of infiltration by contaminants (e.g., Castaño-Sánchez et al., 2020; Cavelan et al., 2021), as well as by causing sea level rise along coasts. Withdrawal of groundwater is a major cause of sea level rise (Seo et al., 2023). Increasing the frequency and severity of droughts will intensify the demand for groundwater (Castaño-Sánchez et al., 2020; Seo et al., 2023), further magnifying the impacts on subterranean biomes. For example, Castaño-Sánchez et al. (2020) studied the potential impacts of increasing salinity by groundwater extraction on three aquatic species of cave crustaceans in arid Australia. They concluded that increasing salinity threatened the integrity of groundwater ecosystems.

6.3 Wildfires

The increased frequency and severity of droughts have made the affected regions more susceptible to wildfires (Jolly et al., 2015). Areas especially impacted include Western North America, Australia, and parts of Europe, Asia, South America, and Africa (Brooks et al., 2004). The severity and frequency of wildfires is also enhanced by the presence of fire-tolerant invasive nonnative species (Brooks et al., 2004).

Wildfires impact subterranean communities by reducing food energy entering caves, for example, by reducing populations of cave visitors and colonies of cave-roosting species and by increasing soil depth by ash deposition. Soil captures water and nutrients, thereby favoring shallow rooted plants and limiting the amount and numbers of native species of deeply rooted plants (Howarth and Wynne, 2022). One of the few studies comparing populations of cavernicoles before and after wildfires was conducted at Lava Beds National Monument in California, USA. That study found a potentially significant drop in biodiversity in the caves under surface habitats burned by fires. Loss of vegetation changed groundwater flow, especially how water percolated down through the soil, as well as reduced the amount of nutrients entering the caves (Geological Society of America, 2021). Conversely, fires can lead to a temporary increase in species richness within caves. This phenomenon was observed by Ferreira (unpublished data) in a quartzite cave located in southeastern Brazil where, shortly after a fire event, the cave exhibited a significant rise in species diversity. This was attributed to several surface species seeking refuge within the cave to escape the fire (see also de Fraga et al., 2023). However, the subsequent increase in species richness was followed by substantial local extinction of external species within the cave, particularly those lacking preadapted traits for cave survival.

Sinkholes with a cave entrance at the bottom can act as refuges for vegetation and associated fauna, since the heat from a passing wildfire will draw the cool moist air out of the cave. The cool air and strong convection prevent the flames from affecting life in the sinkhole (F.G. Howarth and F.D. Stone, personal observations).

6.4 Floods

Floods play an important role in shaping the ecology of caves (Hawes, 1939; Pacioglu et al., 2019). Floods transport significant amounts of water and nutrients underground, which affect populations and life history events of subterranean species. For example, many caves in temperate regions experience cold water floods from melting snow. The predictable influx of water and nutrients transported by seasonal floods provides cues for the timing of behaviors of many subterranean animals (e.g., feeding, mating, molting, dispersal) (Lisowski, 1982; Leitheuser and Holsinger, 1985). Climate change will change the frequency and intensity of flooding events, with potentially severe impacts on subterranean communities. Flooding by warm water later in the season in temperate caves can be especially harmful by raising the temperature and upsetting normal cues (Lisowski, 1982). Floods also increase the rates of erosion and sedimentation, which can fill and disturb habitats (US Fish and Wildlife Service, 1988; Rabelo et al., 2018). Although flooding can carry additional nutrients underground, some of the flotsam may pollute caves.

Although more organic nutrients would be carried into subterranean habitats during wet seasons than during drier seasons, a study conducted by Souza-Silva et al. (2011) revealed that stochastic pulse flows, occurring sporadically throughout rainy periods, often disturb and remove the previously accumulated resources. Consequently, they reported that the true sustenance utilized by subterranean communities, particularly in Neotropical caves, derived from organic matter transported near the end of the rainy season and remained available throughout the subsequent dry season. Additionally, unpredictable floods caused by climate changes during dry periods may disrupt and severely alter the trophic dynamics of many caves, thus directly impacting subterranean communities.

6.5 Storms

Recent research has revealed that within severe storms, several brief squalls of exceptionally heavy rainfall occur over a limited period (<1 h) and area (<1 km^2). These squalls are the major cause of local flooding, landslides, and related damages within the larger storm. Their intensity and frequency are increasing by climate change; yet their occurrence is infrequently detected due to the limited number of rain gauges deployed and obscured by averaging of rainfall data over longer timescales (Ayat et al., 2022). As large storms intensify due to climate change, the number and severity of these squalls will increase. Major damage associated with large storms is generated by these squalls. Cavernous terrains may be especially vulnerable to these brief heavy rainfall events. Floods can carry sediment, nutrients, and toxins underground, or alternatively flush out nutrients and life from affected areas (Retter et al., 2021). Tarburton and Tarburton (2013) found that flooding and heavy rain can severely disrupt reproduction of the Australian Swiftlets (*Aerodramus terraereginae*) by washing away and causing premature disintegration of nests. In addition, heavy rain sinking underground can significantly alter the cave temperature.

6.5.1 Erosion and sedimentation

Intense storms and floods significantly increase the impacts of erosion, collapse, and sedimentation, which can severely affect subterranean communities, particularly following wildfires that strip away surface vegetation and droughts that desiccate the land. External environments surrounding caves have undergone historical modifications worldwide, largely driven by logging, agriculture, livestock, and urbanization. One of the direct consequences of the removal of native vegetation is an increased vulnerability of exposed or inadequately protected soils to erosional processes (Simões et al., 2014; Souza et al., 2021). During intense storms, erosion is greatly amplified, resulting in the transport and deposition of soil and sediment to lower areas of the landscape. Given that many caves are situated in topographically depressed regions, such as sinkhole bottoms, water serves as a conduit for transporting external materials underground.

Due to the erosional processes occurring over subterranean habitats, a high sediment influx can cause siltation of water courses and conduits, thereby reducing and harming the available habitats for subterranean species (Rabelo et al., 2018). Siltation can harm both entire subterranean communities as well as specific species. As an example, the obligatory cave isopod *Iuiuniscus iuiuensis* can be particularly sensitive to silting since this species is the only known isopod to build mud shelters in which to molt (Souza et al., 2015). Hence, any disturbances in the sediment type inside the single cave where it is found can be harmful to this endangered species. Furthermore, the diversity of microhabitats directly determines the number of niches available for cave species, which can be dependent on distinct microhabitats (Souza-Silva et al., 2021).

6.6 Invasive nonnative species

Often overlooked in cave conservation work are the impacts of invasive nonnative species on cave resources, especially on vulnerable species and communities (Howarth, 1981; Howarth and Moore, 1984; US Fish and Wildlife Service, 1988; Reeves, 1999; Moseley, 2009; Tarburton and Tarburton, 2013; Wynne et al., 2014; Howarth and Stone, 2020; Gerovasileiou et al., 2022; Nicolosi et al., 2023). Climate change will exacerbate the problem (Vaccarelli et al., 2023). As the planet warms, many species are shifting their distribution poleward or upward in elevation (Bellard et al., 2012). Both native and nonnative species that shift their distribution will make new biotic associations, which potentially would affect cave communities by changing the composition and relative abundance of resident species. Climate change may facilitate harmful nonnative species transported via human activities to colonize new areas. Surface-dwelling invasive nonnative species can impact subterranean communities, for example, by interacting with native species (e.g., competition, predation), thereby altering species composition, as well as by reducing food energy entering caves, such as by decreasing populations of species roosting in caves (Elliott, 2000; Taylor et al., 2005; Cornwall, 2022). For example, reducing populations of roosting bats can have devastating effects on cave ecosystems (Hamilton-Smith and Eberhard, 2000; Deharveng and Bedos, 2019). Nonnative plants can send roots deep into caves and disrupt host plant communities (e.g., Fig. 21.4C) (Howarth et al., 2007), which can impact nutrient and water resources. Nonnative diseases reduce or extirpate populations of cave species with dire effects on affected communities (Camp et al., 2019; Turner et al., 2022). For example, rapid ohia death caused by the fungi *Ceratocystis lukuohia* and *C. huliohia* are depressing populations of the tree *Metrosideros polymorpha,* which is a keystone species in caves in Hawaii (Stone et al., 2012; Camp et al., 2019).

6.7 Sea level rise

Coastal habitats are increasingly impacted by rising and warming seas, intensifying storms and droughts, and acidification resulting from climate change stressors (He and Silliman, 2019). Sea levels are rising by melting of land-based glaciers, extraction of ground water, and expansion by warming (Seo et al., 2023). Exacerbating the impacts of rising seas is that, in many regions, coasts are sinking from water extraction and the weight of urbanization

(Scavia et al., 2002; Ferguson and Gleeson, 2012). Warmer water and atmosphere acting together increase evaporation, which will intensify the frequency and severity of coastal storms. The major physical impacts of rising sea levels result from storm surges, which increase erosion of coastlines, inundation of deltas, as well as flooding and loss of many marshes and wetlands. Saltwater intrusions threaten coastal aquifers and estuarine systems (Scavia et al., 2002; Castaño-Sánchez et al., 2020; Retter et al., 2021). Warmer seas will also intensify the frequency and severity of algal blooms and anoxia. Furthermore, species moving poleward and colonization by nonnative species will cause new biotic associations, some predictably detrimental to native species (Bellard et al., 2012).

Subterranean habitats will also be severely impacted by sea level rise, but the risks have rarely been assessed in global change studies (Sánchez-Fernández et al., 2021; Wynne et al., 2021). However, many of the same stressors affecting surface habitats will also impact subterranean environments. For example, storm surges will cause greater erosion, sedimentation, inundation, saltwater intrusions, and disturbance of subterranean habitats.

Coastal cavernous habitats include sea caves, caverns, and aquifers. Sea caves are created by surf pounding on and eroding a shoreline cliff. They form in both cavernous and impervious rock and may be flooded or dry. Sea caves offer refuge for many coastal marine animals (e.g., Karamanlidis et al., 2021) and are home to many unique invertebrates and microorganisms (Parravicini et al., 2010; Gerovasileiou et al., 2022). These coastal caves are threatened by rising temperatures, inundation, increased erosion, sedimentation, collapse, and toxic algal blooms. Already, sea cave communities are imperiled by warm water intrusions. Parravicini et al. (2010) surveyed a sea cave on the NW Mediterranean coast before and after two summer heat waves had raised the local seawater temperature by 4 °C. They reported that macroinvertebrates disappeared after the heat waves and were replaced by crustose microorganisms (Parravicini et al., 2010).

Coastal caverns and aquifers occur where cavernous rock substrates meet the sea. They are numerous and widespread along the shores of all continents and most islands. Aquifers often form a continuum from marine (if flooded by salt water), anchialine (if mixohaline), and freshwater environments (van Hengstum et al., 2019). The fauna and flora also track this gradient with specialized species inhabiting each zone. Marine species may advance during sea level rise, but freshwater and mixohaline species must migrate, adapt to the changing habitat, or be regionally extirpated (van Hengstum et al., 2019). Terrestrial species may also relocate, but only if a suitable habitat is available. Storm surges may disrupt the salt-freshwater zonation, further harming resident communities. Recent studies suggest that large coastal anchialine systems, which have sufficient topographic relief, are relatively resistant to sea-level rise (e.g., Yucatan Peninsula); however, low-level platforms may be severely affected or entirely disappear (e.g., many islands) (van Hengstum et al., 2019).

6.8 Disruption of behavior

The increase in frequency and unpredictability of extreme weather events, as well as other novel stressors caused by climate change, may disrupt the behavior and life cycles of subterranean populations. For example, floods occurring at the normal time bring in food resources which provide cues for feeding, mating, and reproduction among cavernicoles. Floods at anomalous times or temperatures can interrupt normal behavior. Anomalous storm surges and flooding may disrupt breeding of the endangered Mediterranean Monk Seal, *Monarchus monarchus* (Karamanlidis et al., 2021); flood control programs increase sedimentation and reduce reproductive success of the endangered Kentucky cave shrimp, *Palaemonias ganteri* (US Fish and Wildlife Service, 1988); and abnormal silting can inhibit molting by the cave isopod, *I. iuiuensis* (Souza et al., 2015).

6.9 Impacts caused by mitigation programs

Because the effects of climate change on subterranean habitats are rarely considered, many programs designed to adapt to or mitigate the impacts on surface ecosystems may harm the subterranean biome. These projects include habitat restoration, extraction of groundwater, adding barriers to protect coastlines, alternative energy projects, and mining projects (Richardson et al., 2021). Additionally, mitigation programs targeting subterranean ecosystems can be detrimental for certain ecosystem components (Meierhofer et al., 2022). A more complete analysis and specific examples will be presented in Chapter 18 of Volume 3.

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Section H

Taiga

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Chapter 22

Asia

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1 Introduction

Boreal forests, or taiga, are the second largest terrestrial biome on Earth after deserts and xeric shrublands, accounting for 30% of the world's forest cover (Gauthier et al., 2023; Kayes and Mallik, 2020). Russian scientist A. Beketov introduced the term "taiga" in 1886 to define the boreal forest, while P. Krylov provided a more specific definition as a dark coniferous boreal forest, distinct from the Scots pine and larch forests (Ermakov, 2003; Sochava, 1980). The word *taiga* (*taika*) originates from Old Turkic origin, meaning "mountainous area" (Murzaev, 1996; Shumilova, 1962). Today, the term taiga refers to all coniferous boreal forests, not just dark coniferous forests.

Dense forests consisting of shade-tolerant spruce, fir, and Siberian pine are commonly referred to as "dark taiga." Conversely, the more open forests and woodlands comprising light-demanding pine and larch are recognized as "light taiga" (Box and Fujiwara, 2013). Taiga forests contain more than 55% of the world's conifers and 11% of the world's biomass (Russia's Boreal Forests, 2023, wwfeu.awsassets.panda.org). The forest includes bogs, swamps, marshes, shallow lakes, rivers, and wetlands that retain significant amounts of water. Wetlands account for approximately 30% of the world's boreal forest area, making it one of the most important sources of freshwater globally (Benoy et al., 2007).

The taiga is characterized by several distinct features: closed canopy, stable humidity, the absence of wind, moss cover, abundance of mushrooms, the presence of shade-tolerant plants in the undergrowth, and abundance of spore plants (e.g., ferns, horsetails, and lycopods). The stable environment of the taiga has ensured that ancient groups of plants and animals are preserved. The region is also abundant in evergreen plants, lianas, orchids, numerous epiphytes, and saprophytes. The conservation of the taiga is of paramount importance in preserving the planet's biodiversity.

The human impact on the taiga biome began with the extraction of natural resources. Specifically, the Siberian taiga possesses a high concentration of minerals, including the West Siberian oil and gas fields, associated with Cretaceous and Jurassic sediments (Sochava, 1980), the largest Lensky and Tungussky coal basins, the Yakutian diamond deposits, the Angara region's iron ore basins, and the most important peat bogs (Bezrukikh et al., 2018). Geological surveys, well drilling, equipment transportation, and construction of worker settlements have commenced in the area, leading to a highly unfavorable impact on the taiga ecosystem.

Taiga plays an important role in the global carbon cycle and in the regulation of climate change, acting as a giant reservoir of carbon emissions (Pappas et al., 2023; Pan et al., 2013). Global warming, melting permafrost, and extensive wildfires in the Siberian taiga have emitted significant greenhouse gases into the atmosphere, leading to a tipping point in the Earth system (Lenton et al., 2008).

2 Geographic location

The taiga is situated in the moderately humid geographical zone of the Northern Hemisphere and spans an area of 17 million km^2 (Kayes and Mallik, 2020). The taiga is one of the world's longest climate zones, stretching a great distance. The taiga in North Asia is bigger than the rainforests in the Amazon. The southern end of the taiga is at the 42nd parallel in northern Hokkaido, Japan, while the northern end stretches all the way to the 72nd parallel on the Taimyr Peninsula in Russia (Fig. 22.1). The most significant waterways in the Asian taiga are the Ob River, the Yenisei River, and the Lena River.

The taiga has a pronounced continental climate, moderately warm in summer and harsh in winter. The snow stays for an extended period, and precipitation is higher than evaporation. Permafrost is widespread across a significant part of this area. The primary types of soil are podzolic, permafrost-taiga, and bog soils (Bezrukih et al., 2019; Parmuzin, 1985). The taiga is divided into three subregions: southern, middle, and northern, based on the vegetation

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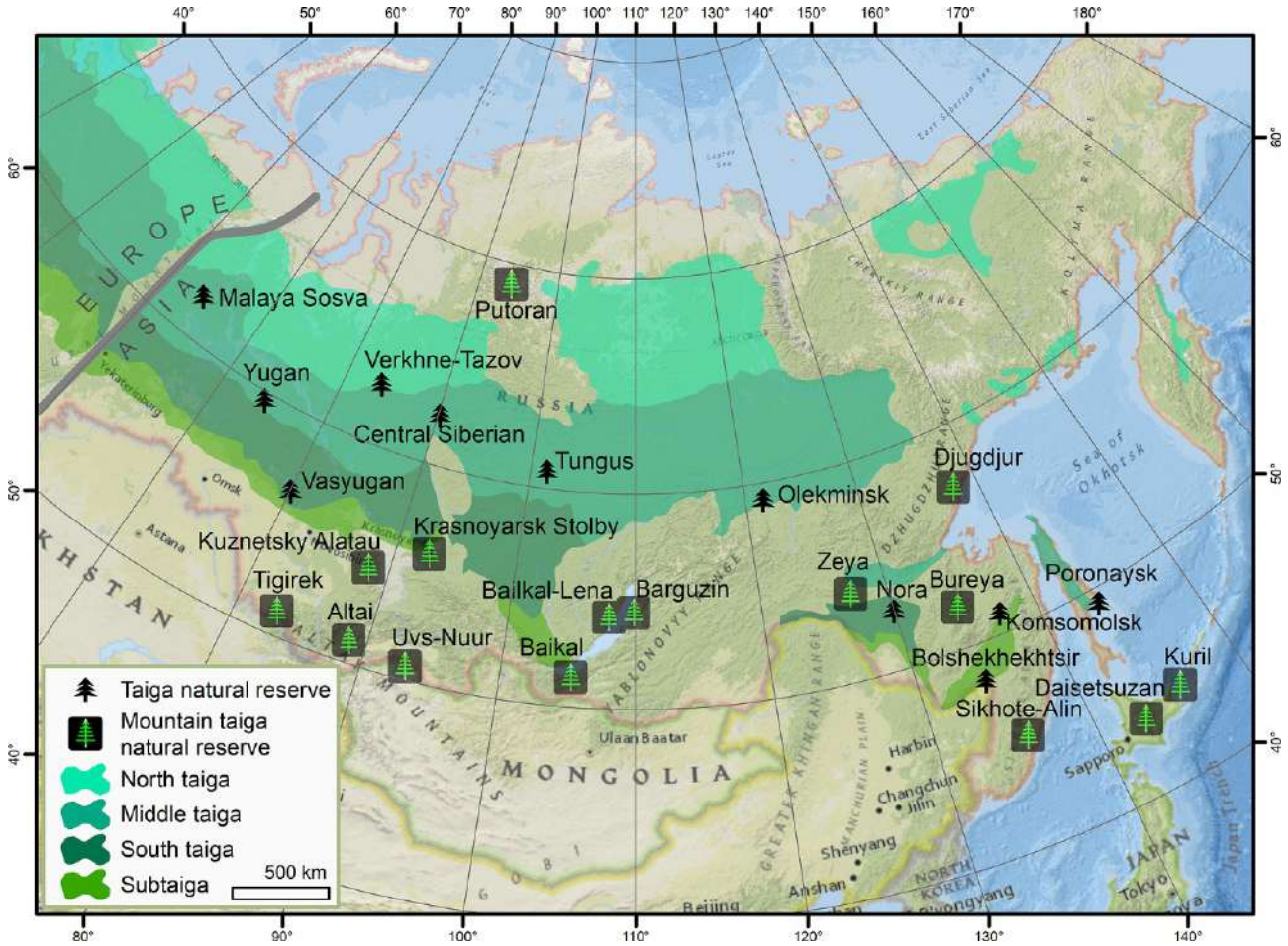

FIG. 22.1 Asian taiga and natural protected areas. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Tatiana Semina.)*

structure from south to north (Fig. 22.1). The Asian taiga is also classified into geographical provinces (ecoregions): the Northeast Siberian, the West Siberian, the Central Siberian, the East Siberian, and the Far Eastern provinces (Vegetation Cover, 1985; Sochava, 1980).

The main tree species of the northern taiga is larch (*Larix* spp.), the only deciduous conifer surviving on permafrost. Other indicators of the northern taiga are *Rhododendron tomentosum* (Stokes) Harmaja and *Vaccinium uliginosum* L. At the southern border of the northern taiga, the dark conifers *Picea obovata* Ledeb. and *Abies sibirica* Ledeb. occur. The northern boundary of the northern taiga is marked by the disappearance of zonal shrub tundra, an increase in the area of closed forest, a decrease in the presence of hypoarctic species in communities, up to their complete disappearance. The southern boundary is determined by the change of dominant tree species in zonal communities. In Western Siberia, it coincides with the southern limit of permafrost distribution at the 63rd parallel, while in Eastern Siberia, it extends northwards to the 66th parallel (Modern Russia, 2020).

The middle taiga is located between 59° and 63°–66° north and the forest covers more than half of the territory (Modern Russia, 2020). In areas of loamy rocks on the watersheds and ridges, the vegetation is mainly dominated by *Pinus sibirica* Du Tour, mixed with *P. obovata*, *A. sibirica*, and *Betula* and with a dense cover of gypsum moss (Fig. 22.2).

Dark coniferous forests with a relatively complex structure dominate the southern taiga. *P. obovata*, *A. sibirica*, and *P. sibirica* grow in the swampy areas of the southern taiga, the so-called *urmans*. Birch forests constitute a substantial portion, almost 40%, of the forested region. Moreover, forests of *P. sylvestris* L. are abundant on the river terraces (Bezrukikh et al., 2018).

The Northeastern Siberian taiga includes the Verkhoyansk, Kolyma, and Dzhugdzhur mountains. The climate is extremely continental, with long winters and short summers. The average annual temperature here is between −3.5°C and −14.5°C and the average annual precipitation is only 175 mm (Shlotgauer and Kryukova, 2005). Near the Sea of Okhotsk, conditions are milder, with an

FIG. 22.2 Middle taiga of Western Siberia. *(Credit: Nikolay Lashchinskiy.)*

average annual temperature of −2.8°C and annual precipitation of about 456 mm. Permafrost covers the entire Northeastern Siberian taiga. (https://www.oneearth.org: Climate—Northeast Siberian taiga, 2023).

The West Siberian taiga stretches from the Ural Mountains in the west to the Yenisei River in the east. It borders tundra in the north and forest-steppe in the south. The climate is continental, subarctic in the north and temperate in the south. The West Siberian taiga is characterized by moderately severe, cloudy, and snowy winters (average January temperature—18–29°C), followed by relatively warm and humid summers after a short spring. Precipitation averages 400–550 mm per year, and snow cover reaches 90–100 cm in February–March in the eastern regions, where cyclonic activity is intense in winter. Average July temperatures in the north are 13°–14°, and near the southern border of the zone—about 18°–19°. This is a vast lowland area of coniferous forests and wetlands, including the most extensive peatlands in the world (Gvozdetsky and Mikhailov, 1978; https://www.oneearth.org: West Siberian Taiga, 2023).

In Eastern Siberia, the Yenisey River to the west and the Verkhoyansk, Kolyma, and Dzhugdzhur mountain ranges to the east delineate the boundary of the taiga region. To the north, the Putorana and Anabar plateaus form the boundary with the Taimyr tundra. To the south, the East Siberian taiga is bordered by the Baikal boreal forest. The region experiences a pronouncedly continental climate with the Siberian anticyclone causing severe cold and drought conditions from September to April. In the southwestern part of the Eastern Siberian taiga, the average annual temperature is −1.7°C and the average annual precipitation is about 356 mm. In the northeastern part, the average annual temperature drops to −11.2°C and the average annual precipitation is 288 mm. Permanent permafrost covers most of this area (Gvozdetsky and Mikhailov, 1978; https://www.oneearth.org: Eastern Siberian Taiga, 2023).

The southernmost taiga grows around Lake Baikal. The southernmost forested region of the taiga biome is located around Lake Baikal. This area is focused on the Yablonoi and Khentii mountain ranges, which cross the boundary between Russia and Mongolia. The northern part of the region is covered by permafrost. In the southwestern part, the average annual temperature is −1.7°C and the annual precipitation is about 356 mm. The average annual temperature is −11.2°C and the average annual precipitation is 288 mm in the northeastern part (Gvozdetsky and Mikhailov, 1978; https://www.oneearth.org: Eastern Siberian Taiga, 2023).

In the southeastern most region of the taiga lies the coniferous forest of Hokkaido, which consists mainly of Sakhalin fir-Mongolian oak forest and boreal and subalpine forests of spruce and larch. This region represents the warmest and most humid area within the taiga zone. The average annual temperatures are 6–8°C and the average annual precipitation is 1120 mm (Climate of Hokkaido, 1982; https://www.oneearth.org: Hokkaido Montane Conifer Forests, 2023).

3 Biodiversity

Despite a low level of biodiversity in the taiga, its ecological significance is very high (Gauthier et al., 2023; Burton et al., 2010; Dezhkin, 1989). There is a noticeable gradient of increasing floristic diversity in the boreal biome from north to south (Leskinen et al., 2020), e.g., in Eastern Siberia the southern taiga is represented by the highest plant species diversity (472 species), while in northern open larch forests only 253 species were found (Zyryanova et al., 2010). The number of animals also increases from north to south (Modern Russia, 2020).

The Northeastern Siberian taiga, one of the coldest places on earth, has vast areas of pristine larch forest with *Larix gmelinii* (Rupr.) Kuzen. with some dwarf *Pinus pumila* (Pall.) Regel and *Betula fruticosa* Pall. (Khokhryakov, 1989), tundra bogs, and wild shorelines that are home to a variety of wildlife. In the mountain-forest belt of the Dzhugdzhur Mountains dark coniferous with Yezo spruce [*Picea jezoensis* (Siebold & Zucc.) Carrière] and larch light coniferous forests are widespread (Shlotgauer and Kryukova, 2005; Sochava, 1980). Within the forests, there are shallow depressions known as *alasy*, which form due to seasonal thawing and freezing of the soil surface. These depressions typically feature meadow or steppe vegetation along their edges, and frequently have a pond in their center. Fragments of steppe vegetation can be found in the Yana and Indigirka river basins, though only on south-facing slopes. To the northeast of the region, larch forests are intermixed with polygonal tundra bogs and wooded ridges. *Betula ermanii* Cham. is found as an admixture in submontane spruce forests and along the southern coast (Shlotgauer and Kryukova, 2005).

The fauna of the Northeastern taiga is quite diverse. The endangered great knot [*Calidris tenuirostris* (Horsfield, 1821)] and Blakiston's fish-owl (*Bubo blakistoni* Seebohm, 1884) breed in the sparse taiga. Steller's sea eagle (*Haliaeetus pelagicus* Pallas) patrols the coast and estuaries, and the black-billed capercaillie (*Tetrao urogalloides* Middendorff, 1853) inhabits the forest undergrowth. Reindeer (*Rangifer tarandus* L.), moose (*Alces alces* L.), brown bear (*Ursus arctos* L.), wolves (*Canis lupus* L.), and wolverines (*Gulo gulo* L.) inhabit the interior of the Northeastern Siberian taiga (https://www.oneearth.org: Northeastern Siberian Taiga, 2023).

The West Siberian lowland taiga comprises vast forests and colossal peat bogs (Fig. 22.3). The flat terrain, water-absorbing clay rocks, high precipitation, and low evaporation cause such an abundance of bogs. These bogs form enormous complexes, with the largest in Eurasia being the Vasyugan Bog.

Permafrost is widespread in the northern taiga of Western Siberia. The largest areas are occupied by light coniferous forests of larch and larch-pine (*Larix sibirica*, *P. sylvestris*) with shrub cover (*R. tomentosum*, *Vaccinium vitis-idaea* L., *V. uliginosum*, *Vaccinium myrtillus* L., *Empetrum nigrum* L., *Betula nana* L.) (Yurkovskaya and Safronova, 2019). Dark coniferous forests with green moss, often found in combination with peat bogs in wetlands, are the most typical of the West Siberian middle taiga (Fig. 22.4). In the western part of the middle taiga, lichen-pine forests on outwash plains and green-moss pine forests on dry slopes are common. Larch-pine and pine forests dominate the northern half of the middle taiga. The spruce-fir-Siberian pine forests and spruce-pine forests with *V. vitis-idaea*, *V. myrtillus*, *R. tomentosum*, and *Linnaea borealis* L. cover the southern and southeastern middle taiga. Upland bogs with *Chamaedaphne calyculata* (L.) Moench, *E. nigrum*, *V. vitis-idaea*, *Sphagnum* and lichens predominate over forests (Yurkovskaya and Safronova, 2019; Yurkovskaya, 2012).

A notable characteristic of Western Siberia's southern taiga is the prevalence or substantial role of the Siberian fir (Fig. 22.4). In contrast, the Siberian larch is prevalent solely in the northern part (Sochava, 1980). Forests with *A. sibirica*, *P. obovata*, and *P. sibirica* with nemoral herbs [*Oxalis acetosella* L., *Carex pediformis* var. *macroura* (Meinsh.) Kük., *Stellaria bungeana* Fenzl, and *Circae alpine* L.] prevail. In the southwest, large areas are occupied by green moss and lichen pine forests. Primary and secondary birch forests include *Betula pendula*, *B. pubescens* Ehrh., and *Populus tremula* L. Almost half of the subzone is occupied by peat bogs, in some places with *P. sibirica* or *P. sylvestris* (Yurkovskaya and Safronova, 2019).

The fauna is represented by brown bear, gray wolf, *Eurasian lynx* (*Lynx lynx* L.), wolverine, and reindeer, as well as smaller predators such as Eurasian otter (*Lutra lutra* L.), sable (*Martes zibellina* L.), Asian badger (*Meles leucurus* Hodgson), and Siberian weasel (*Mustela sibirica* Pallas). Small mammals are particularly abundant, including the tundra shrew (*Sorex tundrensis* Merriam), Siberian flying squirrel (*Pteromys volans* L.), red squirrel (*Sciurus vulgaris* L.), and birch mouse (*Sicista* sp.) (https://www.oneearth.org: West Siberian Taiga, 2023).

The wetlands attract many migratory and breeding birds because of the many insects that live there. The critically endangered yellow-breasted bunting (*Emberiza aureola*

FIG. 22.3 Peat bog located within the taiga of Western Siberia. *(Credit: Oleg Kardash.)*

FIG. 22.4 Dark coniferous taiga with *Picea obovata* in Western Siberia. *(Credit: Oleg Kardash.)*

Pallas, 1773) and Siberian crane, the endangered white-headed duck [*Oxyura leucocephala* (Scopoli, 1769)] as well as numerous vulnerable species such as the greater spotted eagle (*Aquila clanga* Pallas, 1811), red-breasted goose [*Branta ruficollis* (Pallas, 1769)], horned grebe [*Podiceps auritus* (Linnaeus, 1758)], and rustic bunting (*Emberiza rustica* Pallas, 1776) depend on the undisturbed habitat (https://www.oneearth.org: West Siberian Taiga, 2023).

The East Siberian Taiga forest represents the world's most extensive intact boreal forest. The western and southern areas are dominated by the deciduous *Larix sibirca*, with the northern and eastern parts home to *L. gmelinii*. Hybridization of the two species occurs in the areas where their ranges overlap. The understorey is sparse, with *R. tomentosum, V. uliginosum, Vaccinium oxycoccos,* mosses, lichens, and fungi. In the south and west, there are mixed forests—Siberian larch, dark conifers (*P. obovata*, *P. sibirica*, and *A. sibirica*) and broadleaf trees (*B. pendula*, *B. pubescens*, and *P. tremula*). The forest floor is a rich mixture of sedges, lingoberry, and rhododendron. Various unique species, including *Adenophora jacutica* Fed., *Koenigia amgensis* (Michaleva & Perfil.), and *Artemisia czekanowskiana* Trautv., are endemic to this area (Box and Fujiwara, 2013).

Yezo spruce is found in the montane submarine and maritime regions of northeastern Asia. Spruce forests are considered a zonal vegetation type in the lower Amur Basin, Sakhalin, the southern Kurils (Iturup), and northeastern Hokkaido. Important companion species in the stands are *Abies nephrolepis* (Trautv. ex Maxim.) Maxim. (continental areas) and *A. sachalinensis* (F. Schmidt) Mast. (island areas). In addition to firs, other species of dark conifers may be mixed in with the Yezo spruce stands, such as *Picea koraiensis* Nakai in the southern areas (Krestov, 2003).

The fauna of Eastern Siberia is richer than that of Western Siberia. Russia's largest populations of brown bear, moose, wolf, red fox [*Vulpes vulpes* (Linnaeus, 1758)], reindeer, and wolverine are found here (Solovyev, 2000). To the east of the Yenisei River live the musk deer (*Moschus moschiferus* Linnaeus, 1758), the red mouse [*Apodemus speciosus* (Temminck, 1845)], the moose, the European roe deer [*Capreolus capreolus* (Linnaeus, 1758)], the Eurasian wild boar (*Sus scrofa* Linnaeus, 1758), and the red deer (*Cervus elaphus* Linnaeus, 1758). Amur tigers (*Panthera tigris* subsp. *altaica* Temminck, 1844) and oriental leopards [*P. pardus* subsp. *orientalis* (Schlegel, 1857)] are also found in far eastern areas (Kitchener et al., 2017).

Typical bird species include the black-billed capercaillie (*T. urogalloides*), Siberian spruce grouse [*Falcipennis falcipennis* (Hartlaub, 1855)], Siberian accentor [*Prunella montanella* (Pallas, 1776)], great gray owl (*Strix nebulosi* J.R. Forster, 1772), and Naumann's thrush (*Turdus naumanni* Temminck, 1820), Siberian blue robin [*Luscinia cyane* (Pallas, 1776)], rufous-tailed robin [*Luscinia sibilans* (Swinhoe, 1863)], Siberian lentil [*Carpodacus roseus* (Pallas, 1776)], white-backed swift [*Apus pacificus* (Latham, 1801)], great turtle dove [*Streptopelia orientalis* (Latham, 1790)], black-billed capercaillie, Baikal teal [*Sibirionetta formosa* (Georgi, 1775)], and black crow (*Corvus corone* Linnaeus, 1758). Berkut [*Aquila chrysaetos* (Linnaeus, 1758)], osprey [*Pandion haliaetus* (Linnaeus, 1758)], peregrine falcon (*Falco peregrinus* Tunstall, 1771), black stork [*Ciconia nigra* (Linnaeus, 1758)], and black crane (*Grus monacha* Temminck, 1835) are listed in the Red Book of Russia (The Red Book, 2001; Soloviev, 2000).

The mountainous taiga surrounding Lake Baikal represents the southernmost region of the Siberian taiga, where dark taiga from the west meets light taiga from the east. The forest belt in the mountains of southern Baikal Siberia consists of *L. sibirica*, *A. sibirica*, *P. obovata*, *P. sibirica*, and *P. sylvestris*. The lower part of the forest belt is occupied by light coniferous taiga dominated by Scots pine and larch, while the upper part is occupied by dark coniferous taiga (Peshkova, 1985). Forest undergrowth includes species such as *Rhododendron dauricum* L., *V. myrtillus*, and *Alnus hirsuta* (Spach) Rupr. Mosses and lichens are abundant. Steppe vegetation occupies the lower mountain belts, while a subalpine belt of meadows, shrubs, and sparse trees is present at elevations above 1500 m. Above 1700 m, tundra communities are observed.

Baikal forests are inhabited by large mammals such as *A. alces*, *C. elaphus*, *U. arctos*, *C. lupus*, *Nyctereutes procyonoides* (Gray, 1834), *M. moschiferus*, and L. *lynx*. The white-naped crane [*Antigone vipio* (Pallas, 1811)] nests in the humid upland areas, and the endangered Tarbagan marmot [*Marmota sibirica* (Radde, 1862)] and great bustard (*Otis tarda* Linnaeus, 1758) breed in the steppe areas.

The most southeastern taiga in the world is located on Hokkaido, which is Japan's second-largest volcanic island. The forest on the mountain slopes is dominated by Sakhalin fir (*Abies sachalinensis*) with Mongolian oak (*Quercus mongolica* Fisch. ex Ledeb.), painted maple (*Acer pictum* Thunb.), Japanese lime [*Tilia japonica* (Miq.) Simonk.], Erman's birch (*B. ermanii*), and Yezo spruce (*P. jezoensis*). There are three major climax forest types in the boreal forest zone: *P. jezoensis-A. sachalinensis* forest, forest with *Picea glehnii* (F. Schmidt) Mast., and *L. gmelinii* forest. The subalpine forest belt includes three main forest types: with *Abies mariesii* Mast., with *B. ermanii* and *P. pumila* thicket (Okitsu, 2003). Japanese alder [*Alnus japonica* (Thunb.) Steud.] and wetland species such as water chestnut (*Trapa natans* L.) and yellow pond lily (*Nuphar japonica* DC.) are common in the marshes of Kushiroshitsugen National Park (Krestov, 2003; Miyawaki, 1984).

The fauna is rich and includes large populations of brown bear, Yezo sika deer [*Cervus nippon* subsp. *yesoensis* (Heude, 1884)], and northern pika [*Ochotona hyperborea* (Pallas, 1811)]. Hokkaido is home to birds with diverse niches, such as the endangered *E. aureola* and the critically endangered *B. blakistoni*, which has a unique and large population in Hokkaido. The Kushiro marshes are also home to the endangered red-crowned crane [*Grus japonensis* (Statius Muller, 1776)], which has its most important nesting and wintering grounds here (Hokkaido Montane Conifer Forests: https://www.oneearth.org).

4 Conservation areas and their ecological indicators

There are various categories of protected areas in the Asian taiga. Natural reserves are areas of land or water where the whole ecosystem is preserved in its natural state. Typically, these areas are inaccessible to tourists, although some natural reserves may allow entry with a special permit. National parks are institutions dedicated to nature protection and research. Their territories encompass natural complexes and objects of special ecological, historical, and aesthetic value, and are intended for use in nature protection, education, science, culture, and regulated tourism. A brief overview of the major protected areas of the Asian taiga includes the largest natural reserves in Russia and national parks in Japan (Fig. 22.1).

Malaya Sosva (224,270 ha) is situated on the West Siberian Plain within the Khanty-Mansiysk Autonomous District—Yugra. The forest covers over 84% of the territory, with approximately 14.5% consisting of peat bogs. The dominant tree species is *Pinus sylverstris*. *P. obovata* prevails in the dark coniferous taiga (Reserves of Siberia, 1999). The fauna comprises typical species of the middle taiga, but its originality lies in the presence of many mammal species with a more northern range and birds with a more southern one (Rajewski, 1982). Common species in the reserve include squirrels, chipmunks, hares, sables, ermines, otters, foxes, several insectivores, and mouse-like rodents. Among the birds observed are tits, nuthatches, woodpeckers, finches, chaffinches, thrushes, certain species of owls and raptors, waders, and ducks. Grouse, black grouse, hazel grouse, and white partridge are also frequently spotted (Dezhkin, 1989).

Yugan (648,636 ha) is situated in the middle taiga in the north of Priobie, the Khanty-Mansiysk Autonomous District—Yugra. The forested area comprises 64% of the total area and the nonforested area is 36%, mainly bogs (Reserves of Siberia, 1999; Baikalova et al., 1998). The river valleys are covered by dark coniferous forests: fir-spruce, fir-spruce-Siberian pine. The hills are dominated by spruce, while the flat ridges and slopes are dominated by Siberian pine. Coniferous forests with Scots pine are common, as well as peat bogs. These forests frequently feature a mixture of birches and aspens. Plants and animals that require special attention are the yellow lady's slipper orchid, slipper orchid, lily of the valley, calypso bulb, brown bear, lynx, sable, wolverine, black stork, osprey, white-tailed eagle, golden eagle, peregrine falcon, Siberian crane, and red-breasted goose during migration. These species should be prioritized for conservation efforts.

Verkhne-Tazov (534,955 ha) is located in the north taiga in Yamal-Nenets Autonomous District on the upper reaches of the Taz River in the Verkhne-Tazov Highlands, a part of the Siberian Hills (Reserves of Siberia, 2000). Dark coniferous forests consist of pure Siberian pine, spruce, and fir, as well as mixed forests. Large areas are occupied by suppressed Scots pine forests on peat bogs and shrub associations. The most common mammals are wolves, moose, foxes, squirrels, chipmunks, and brown bears. Birds include the bean goose, oystercatcher, gyrfalcon, eagle, and white-tailed sea eagle. The Siberian taiga bean goose, red-breasted goose, peregrine falcon, golden eagle, osprey, and white-tailed eagle are listed in the Red Book of Russia (The Red Book, 2001).

Vasyugan (614,803 ha) is located on the border of the Tomsk and Novosibirsk regions in Western Siberia in the south taiga. The whole territory of the Vasyugan reserve is a complex of forest and bog ecosystems. The forest and swamp communities are dominated by primeval communities with Scots pine. The ridges and more drained areas are occupied by old forests with the prevalence or significant participation of Siberian pine, spruce, aspen, fir, and birch. The Red Book lists 41 animal species, including reindeer, river otter, and Russian desman, and two plant species (The Red Book, 2001).

Central Siberian (1,019,899 ha) is located in the middle taiga of Krasnoyarsk region on the right bank of the Yenisey River. Dark coniferous taiga is the main forest community of the reserve. The dominant species is the Siberian pine, and less often the Siberian spruce. Scots pine and Siberian larch forests occur in fragments, mostly associated with light sandy and sandy loam soils. The unique feature of the reserve is that the Yenisei River is the biogeographical boundary, the area where the western and eastern faunal complexes meet. Squirrels, chipmunks, brown bears, ermines, sables, and moose are common, while wood grouse and black grouse are abundant. During the migration season, many ducks visit the reserve, some of which stay to nest (Reserves of Siberia, 2000).

Poronaysk (567,000 ha) is situated in the eastern part of Sakhalin Island. Over 70% of the region is forested. The vegetation of the reserve consists of coniferous and deciduous formations with Yezo spruce, Sakhalin fir, stone birch, and Japanese pine (*Pinus thunbergii* Parl.), as well as swamp and meadow vegetation found along the seashore. Five species of plants are included in the Red Book of

Russia (The Red Book, 2001). The reserve is home to various animal species such as the viviparous lizard, Siberian frog, grouse, nutcracker, mallard, Asian chipmunk, river otter, and brown bear.

Olekminsk (847,108 ha) is located in the south of the Republic of Sakha (Yakutia), where the Prilena Plateau borders the Aldan Plateau (Reserves of Siberia, 1999). The entire reserve is situated within an area of permafrost that is 100–200 m thick. The area forms a connection between the taiga of the South Siberian mountains and the taiga of the middle region. The main forest species are *L. gmelinii* and *B. pendula*, which cover 53.4% of the forested area. *P. sylvestris* and *A. sibirica* forests occupy 28.9%. Six plant species are included in the Red Book of the Russian Federation (The Red Book, 2001). The basic components of the mammal fauna are sable, wolverine, chipmunk, wood lemming, red and red-gray voles, Siberian weasel, flying fox, brown bear, and moose. Southern taiga species (Manchurian deer, some hand-wings) and typical mountain taiga species (northern pika, musk deer) are also common. Taiga birds (grouse, bearded owl, hawk owl, Tengmalm's owl, three-toed and black woodpecker) live in the reserve together with East Siberian species (black-billed capercaillie, red-necked nightingale, Siberian flycatcher, Siberian thrush, stonechat duck).

There are two national parks on Hokkaido Island, Japan—Daisetsuzan and Shiretoko. These areas also protect taiga ecosystems.

Daisetsuzan (226,700 ha) is located in the mountainous area of Hokkaido. Daisetsuzan National Park comprises the Daisetsuzan, Mt. Tokachi, and Mt. Shikaribetsu volcanic groups alongside the more ancient geological strata found in the Hidaka Mountain Range. The foothills are a vast coniferous forest zone consisting mainly of *P. jezoensis* and *A. sachalinensis* and deciduous trees, but as the elevation increases the appearance changes to a coniferous forest and Erman's birch forest, and further into the forest boundary the dwarf pine communities. The mountain's forests are rich in mammals, including brown bear, Yezo sika deer, Yezo red fox, Yezo stoat, and Hokkaido squirrel, and birds, including the rare Blakiston's fish owl and black woodpecker, as well as the rare Eurasian three-toed woodpecker. There are a number of species found only in a limited area of Mt. Daisetsu, such as the northern pika. The subspecies of *Parnassius eversmanni* Ménétriés and *Boloria freija* Thunberg, the alpine butterflies, are only found in Mt. Daisetsu and are relics.

Shiretoko (38,600 ha) covers most of the Shiretoko Peninsula at the northeastern tip of Hokkaido Island. The area is covered by temperate and subalpine mixed forests. The main trees are *A. sachalinensis*, *B. ermanii*, and *Q. mongolica*. Above the forest line, there are dense thickets of *P. pumila*. The area hosts numerous birds, such as Steller's and white-tailed eagles, and marine life, including spotted seals, killer whales, and sperm whales.

There are nature reserves located to the south of the taiga that protect the mountain forests. These reserves include Krasnoyarsk Stolby, Kuznetsky Alatau, Sikhote-Alin, Altai, Tigirek, Uvs-Nuur, Baikal, Bailkal-Lena, Barguzin, Bureinskiy, Djugdjur, and Zeya. Putoran Nature Reserve is situated on the Putorana mountain plateau in the north-west of the Central Siberian Plateau (Fig. 22.1).

5 Human activities and their impacts

Human economic activity leads to changes in the main characteristic of the taiga—the change of forest cover, in particular the destruction of the forest. The forest is being reduced as a result of ever-increasing timber harvesting in recent years, cleared to make space for settlements (Girona et al., 2023), arable land and pastures, and destroyed by forest fires (Fig. 22.5). Taiga is also partially fragmented by roads, railways, and infrastructure development (Martin et al., 2023).

The destruction of woody vegetation causes changes in the regime of surface and subsurface runoff, local water rotation, and soil formation conditions, leading to the transformation of plant communities and the disappearance of many animals. In Siberia, the locals rely on the forest for their livelihoods—timber harvesting, gathering berries, mushrooms and medicinal plants, traditional agriculture (such as grazing livestock and making hay), and hunting (Leskinen et al., 2020; Bezrukikh et al., 2018; Aksenov et al., 2002).

According to the state statistics, damage to Russia's Boreal Forests in 2014–17 was caused by fires (63%), insects (15%), weather conditions (11%), diseases (10%), and other factors such as industrial pollution (~1%) (Leskinen et al., 2020; Forest Fund of Russia, 2004). The predominant cause of fires in Siberia is attributable to human activity (Achard et al., 2008).

Forests in Siberia have always burned, but in the 21st century the frequency and areas of forest fires increased (Gauthier et al., 2023): in some years, the areas covered by fire reached 10–12 million ha (Kharuk et al., 2021). Ongoing climate change has increased the area burned and the frequency and severity of fires, leading to significant changes in forest ecosystems (Gauthier et al., 2023). In the northern taiga, approximately 1.5% of the entire cryolite forested area sustains damage from forest fires annually, and the burned areas cannot recover their productive potential and forest cover for decades or even centuries without human assistance (Gauthier et al., 2023; Abaimov et al., 2000). The average yearly loss of taiga due to fire ranges from one to three million hectares (Krylov et al., 2014).

FIG. 22.5 Logging and forest fires occur frequently in the taiga of Eastern Siberia near the Angara River. *(Credit: Natalia Rudaya.)*

Larch and Scots pine are pyrophytes, so their evolution took place in conditions of regular forest fires, but the dark coniferous taiga suffers from fires. Fires cause the most damage to the dark coniferous forests formed by Siberian pine, fir, and spruce. Birch and aspen spread over the burned areas, and dark conifers that tolerate shade grow under their canopies. Slowly they move into the upper layers of the forest. In a hundred years, the deciduous trees are replaced by dark conifers, but part of the territory can be permanently occupied by grasses, shrubs, and aspen or birch. The reason is that the overgrown dry grass and shrub cover in spring provoke fires that destroy the dark coniferous undergrowth. Human help in restoring the dark coniferous forest is essential in such habitats (Kharuk and Ponomarev, 2020).

Deforestation and forest fires have serious consequences associated with significant carbon emissions. The vast wetlands of the taiga zone are important reservoirs of greenhouse gases. They are trapped in peat, much of which is currently frozen in permafrost. Global warming and associated thawing of permafrost and forest fires could increase emissions by releasing huge amounts of carbon dioxide (Pugh et al., 2019; Seidl et al., 2017; Bradshaw and Warkentin, 2015).

In Western Siberia alone, for example, the permafrost of peatlands contains 70 billion tons of methane, a quarter of all the methane stored on the Earth's surface. Its release could lead to a significant warming of the Earth's climate, comparable to decades of anthropogenic carbon dioxide emissions (Mohajan, 2012).

While climate cycles alter fire regimes (Groisman and Soja, 2009; Kharuk et al., 2005), there are other factors that influence the frequency and severity of fires in the taiga. The Siberian silk moth (*Dendrolimus superans sibiricus* Tscetv.) regularly infests forests in Siberia and in most cases leads to large and severe fires due to increased fuel loads (Krylov et al., 2014). From 1880 to 1969, about 13 million ha of East Siberian forests were destroyed by the Siberian silk moth, representing a loss of 2 billion m^3 of growing stock. The topography of Siberian silk moth foci in coniferous forests coincides with forests disturbed by anthropogenic impacts (Bakhvalov et al., 2010).

In addition, many insect pests of forests are expanding their range north and east due to climate change (Gauthier et al., 2023). Insect outbreaks heavily damaged more than 1 million ha in the mid-1990s and affected an area of more than 10 million ha of northern larch forests in 2000–01 in

latitudes where this insect had rarely been observed before (Leskinen et al., 2020; Shvidenko et al., 2013).

The North Siberian taiga remains intact, although areas that are more accessible have been subject to intensive forestry operations. Between 1965 and 1990, over 15% of the forest in the Magadan region was cleared. Livestock and horse breeding are carried out in the river basins and in the Alasian areas. Until the 1990s, the forests of the North Siberian taiga were logged by the state, but then it decreased significantly. Today, illegal logging and export to China, Japan, and South Korea is a serious problem, especially in the more accessible southern areas. Climate change will have a profound impact, with forests predicted to expand north and east into tundra areas. The competitive advantage of larch over other species may decrease, changing the composition of forests. Climate change also increases the risk of catastrophic forest fires (https://www.oneearth.org: Northeastern Siberian Taiga, 2023).

The specific problems of anthropogenic impact on the taiga of Western Siberia, in addition to the general ones, are that 70% of Russia's oil and natural gas extraction occurs there, and is associated with habitat destruction and pollution incidents. As a result, temperatures are rising faster than the global average and the effects of thawing permafrost, associated changes in plant productivity and hydrology, species range shifts, droughts, increased forest fires, and tree disease outbreaks could be enormous (https://www.oneearth.org: West Siberian Taiga, 2023).

The specific problems of Eastern Siberian Taiga, whose nature is generally intact, are the hydroelectric power plants on the Angara, Zeya, and Vilyuy Rivers, which form huge reservoirs upstream. The electricity generated by the Vilyuy dam is used at the Mir diamond mine. Due to diamond mining, the Sakha region suffers from heavy metal and mineralized brine water pollution (Aksenov et al., 2002).

The taiga areas around Lake Baikal are largely protected, but climate change is threatening the trees of the Baikal taiga (Roe, 2020). Siberian pine and birch stands are dying, and this is associated with drought and reduced soil moisture. One of the most common problems threatening the taiga in this region is fire. In the southeastern parts of the region, it is difficult to find forests that have not burned for more than 25 years (Morozova et al., 2023).

The first half of the 20th century marked the beginning of massive human intervention in the ecosystem surrounding Lake Baikal. This period was marked by large-scale deforestation and extensive cultivation of virgin lands, as well as industrial-scale fishing of valuable fish species. Logging is prevalent near the Chinese border, where the illegal sale of timber takes place. Poaching also poses a persistent threat to forest wildlife (Brown et al., 2021).

Hokkaido's taiga forests are the most undisturbed and over 25% of them are protected. The island of Hokkaido, formerly called Yezo or Ezochi, was inhabited by the Ainu until the late 19th century, who used natural resources for their economy (Kato, 2020). It has suffered from overgrazing, unsustainable use of water resources for irrigation, and pollution of streams from gold mining.

Hokkaido has now become a "hotbed of sustainable tourism." The existence of a well-managed system of protected areas has also brought problems to the region, particularly the increased demand for more tourist infrastructure such as roads, lodges, and trails. Hiking trails are a cause of severe soil erosion, especially because of the volcanic nature of these areas (Jones, 2015).

The remoteness of some regions and the lack of natural predators have led to a dramatic increase in the number of Yezo sika deer, which have caused severe damage to agriculture and forest trees (McCullough et al., 2009). Brown bear overpopulation is also an issue, with conflicts with human populations in cities such as Sapporo (Yoshikazu, 2017).

6 Operational and potential monitoring tools to conserve or to recover the biome

Scientists are focused on predicting future taiga landscapes in the long term. It is important to take into account the environmental conditions of forest formation. The conditions in various parts of the taiga region differ, and to enhance forest quality and productivity, it's important to determine the optimal forest cover for each region, the most appropriate species composition, land reclamation systems, and forestry practices. This requires not only the conservation but also the restoration of forest ecosystems, taking into account their resilience to global change. To achieve this, remote sensing, modeling of natural and anthropogenic disturbances in boreal forests, and forest restoration must be prioritized (Gauthier et al., 2023; Seidl et al., 2023; Senf and Seidl, 2021; Hansen et al., 2013).

Priority actions for the conservation of the taiga biome in the coming decades should be identified and include mitigating the effects of climate change, preventing human-caused forest fires, controlling habitat destruction and pollution from oil and gas extraction, and stopping illegal logging and poaching (https://www.oneearth.org/).

Specific issues in various parts of the taiga require special efforts to conserve the ecosystem. For the northernmost taiga, priority conservation actions will include increasing connectivity between protected areas with a focus on climate change adaptation, protecting breeding seabird colonies, and improving enforcement to prevent and stop illegal logging (https://www.oneearth.org: Northeastern Siberian Taiga, 2023).

In the taiga of Western Siberia, where oil and gas companies are concentrated, it is particularly important to prevent the impact of oil and gas exploration on plant and animal habitats. Taiga bogs, which occupy large areas

in Siberia, especially in the West Siberian Plain, can be transformed into more valuable land through drainage. Forests, extensive grasslands, and even arable land will take their place. However, the implementation of such reclamation must take into account the geographical consequences that will always accompany it. Bogs are very dynamic natural complexes, and changes in the conditions of their formation cause a drastic reorganization of their structure, the limits and final results of which are not yet clear (Kovda, 1991; https://www.oneearth.org: West Siberian Taiga, 2023).

For Eastern Siberia, climate change will affect the distribution of species, the risk of forest fires, and the response of forests to dry conditions and increased wind speeds. Human activity is also destroying the taiga. The Mir diamond mine was closed in 2001, but has since been reopened as an underground mine, with its attendant pollution problems. The exploitation of gas and coal fields in southern Sakha could be extremely damaging. The Angara region west of Lake Baikal is experiencing intensive logging, some of which is illegal.

Specific priority conservation actions for the Eastern Siberian taiga include monitoring and minimizing pollution at mining sites, controlling and managing fire regimes, and preventing and stopping illegal logging (https://www.oneearth.org: East Siberian Taiga).

It is very important for the southern parts of taiga to find a scientifically sound solution to the problem of maintaining the balance between different types of agricultural land—forests, arable land, pastures, and meadows. It is a fact that the landscapes of this subzone, especially the forest ones, have a great economic value. Under a competent regime, they can be used more economically than unproductive pasture and arable land created to replace depleted forests (Bezrukikh et al., 2018).

7 Conclusions

The taiga is one of the largest and most vulnerable biomes on the planet. At the same time, the taiga is home to a large number of animals and plants and is also the "lungs of our planet." Taiga breathes, and every year it absorbs huge amounts of carbon dioxide, a major contributor to global warming, and reduces its concentration in the atmosphere. Currently, the taiga is under active anthropogenic influence, which leads to the destruction of forests. At the same time, climate and anthropogenic-induced degradation of permafrost in the northern taiga leads to large emissions of greenhouse gases. In order to preserve the taiga biome of northern Asia, a number of protected areas have been established in Russia and Japan, where human presence is limited and biodiversity is preserved. The main factors that destroy the forest are increased timber harvesting, cropland and pasture, and destruction by forest fires. Taiga is fragmented by roads, railways, and infrastructure development. The destruction of woody vegetation causes changes in the regime of surface and underground runoff, local water rotation, and soil formation conditions. It leads to the transformation of phytocenoses and the disappearance of many animals. In the forthcoming decades, key actions to conserve the taiga biome should include mitigating the effects of climate change, preventing human-caused forest fires, controlling habitat destruction and pollution from oil and gas extraction, and stopping illegal logging and poaching.

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Chapter 23

North Europe

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1 Introduction

One of the most important conditions for the sustainable development of mankind is the preservation of biodiversity on our planet. A sufficient level of natural diversity is a prerequisite for the normal functioning of ecosystems and the biosphere as a whole. In the Russian Federation, the implementation of the decisions from the world summits on sustainable development begins with the development of regional biodiversity conservation strategies (National Strategy for Biodiversity Conservation of Russia, 1998). Biodiversity refers to the diversity of living organisms in all areas of their existence: terrestrial, marine, and other aquatic ecosystems and communities of organisms; it is the diversity of species, diversity within species, and diversity of communities of organisms and ecosystems (Ecological Encyclopaedic Dictionary, 2002). The very phrase "biodiversity" was first used by G. Bates in 1863 in his work "Naturalist on the Amazon River," when he described his impressions of encountering about 700 different species of butterflies during an hour-long excursion (Bates, 1958).

The term includes all species of plants, animals, and microorganisms and the ecosystems of which they are a part. The term encompasses varying degrees of natural biodiversity, including both the number of species and their frequency of occurrence. The concept of "biodiversity" only came into wide circulation in 1972 at the UN Stockholm Conference on the Environment, where ecologists managed to convince political leaders of the world community that the protection of wildlife should be a priority in all human activities on Earth.

There are three main types of biodiversity:

1. Genetic diversity (genetic polymorphism)—Diversity of populations according to traits or markers of genetic nature

This type of biodiversity is caused by adaptive necessity in natural populations and represents inherited diversity within and between populations of organisms (Johannsen, 1935; Altukhov, 2003; Lebedeva et al., 2004).

2. Species diversity—diversity of species of living organisms (plants, animals, fungi, and microorganisms) in a particular ecosystem or specific area (Odum, 1986; Yurtsev, 1992).
3. Ecosystem diversity—based on differences between ecosystem types and diversity of habitats and ecological processes. The diversity of ecosystems is noted not only in terms of different components (structural, functional), but also in terms of scale—from microbiogeocoenosis to biosphere (Pianka, 1981; Tereshchenko et al., 1994; Lebedeva et al., 2004).

In the frames of this study, different reforestation methods produce phytocenoses with different levels of ecosystem biodiversity. And the greatest number of species of flora and fauna will be characterized by the cenosis formed by biogroups, as only in this case the typical forest environment is preserved as much as possible (when reforestation by rows, the common pine is displaced by aspen) (Koldanov, 1966). The diversity of landscapes, reflecting the peculiarities of territorial structure and the influence of local, regional, and national cultures of the society, is considered a separate category. One of the most important conditions for the sustainable development of mankind is the conservation of biodiversity on our planet (Programme of Action, 1993). Despite the wide coverage of the concept in the literature, biodiversity is, to some extent, difficult to define. The most straightforward way relates to the definition of species richness and variability within a community (Pielou, 1975). Measuring diversity can provide information on the mechanisms that influence community structure and the way in which environmental change affects the ecosystem (Pielou, 1975).

It is important to investigate how ecological diversity changes in response to disturbance to understand how

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ecosystems respond to different disturbances. This can be useful for managing human-induced disturbances, as ecosystems respond to them in the same way as to natural disturbances (Dennis and Patil, 1988; Främbs, 1990). Species diversity can be used as an indicator of ecosystem health (Pielou, 1975).

No single environmental factor taken in isolation can explain the cause of species diversity in a particular landscape zone of the globe. At the end of the 20th century, the links between climate and diversity were discussed in a paper based on a comparison of the number of species of some insect groups in the Russian Plain. The authors concluded that the problem of correlation between climate and biodiversity is still at the descriptive stage of study. Due to the existence of numerous hypotheses, "diversity" is the result of contradictions, a compromise between the genetically inherent potential of form formation and the resources of the environment. Thus, evolution is directed toward "increasing diversity" (Chernov and Penev, 1993). The formation of diversity is a self-moving process; it creates the preconditions for its further evolution; so, it can be argued that diversity generates diversity according to the feedback principle (Whittaker, 1980).

The relevance of the conservation of rare and endangered species of vegetation and fauna is determined by the importance of the task of their conservation as elements of biological diversity, which is the basis for the integrity of ecosystems and the Earth's biosphere. Currently, the main threat to the existence of rare plant habitats in Northern European forests is recognized as unsustainable forest management. The consequences of this are manifested in the reduction of undisturbed forests, loss of typical landscapes, disturbance of the natural balance of ecosystems, and reduction of biodiversity. The most effective method of preserving rare plant species and plant communities is the designation of specially protected natural territories and specially protected forest areas.

2 Materials and methods

The theoretical basis of this study was formed by fundamental and applied studies of domestic and foreign scientists devoted to the issues of economics of nature management, in particular, the issues of biodiversity conservation and various aspects of protected areas; forest management and reforestation; and climatic changes and their impact on forest ecosystems. Methods of statistical, comparative analysis, synthesis, grouping of actual data, and sample surveys were used. The information base of the work was statistical data of the Ministry of Natural Resources of the Russian Federation, Forest Plans of the subjects of the Russian Federation, data from annual reports of nature reserves, as well as data obtained by the author in the course of independent research.

The object of the study is the Scandinavian and Russian taiga.

3 Geographic location

The Scandinavian and Russian taiga is an ecoregion within the taiga and boreal forest biome. It is located in northern Europe between tundra in the north and temperate mixed forests in the south and covers about 2.16 million km^2 (832.8 thousand square miles) in Norway, Sweden, Finland, and northern European Russia, being the largest ecoregion in Europe. In Sweden, the taiga is primarily associated with the Norrland area.

Northern Europe borders the countries of Western and Eastern Europe with access to the seas of the Atlantic Ocean (Baltic, North, and Norwegian) and two seas of the Arctic Ocean (Greenland, Barents). It is located at the intersection of important sea and air routes from Europe to North America and the harsh climate of the northern parts of the region (Bohn et al., 2000; Sporrong, 2003).

Northern part of the European part of Russia: Arkhangelsk region, Nenets Autonomous Okrug (Arkhangelsk region), Murmansk region, Komi Republic; Vologda region; and Republic of Karelia. The Arkhangelsk region borders with the Republic of Karelia in the west, with the Komi Republic in the east, with the Vologda and Kirov regions in the south, and with the Murmansk region in the north (the border passes through the White Sea). The Nenets Autonomous Okrug is located in the northeast of the European part of the Russian Federation.

The Murmansk region borders on the Republic of Karelia in the southwest, and Finland and Norway in the west and northwest. In the west, northwest and north, the Komi Republic borders with the Arkhangelsk region and the Nenets Autonomous Okrug, in the east with the Yamalo—Nenets Autonomous Okrug and the Khanty-Mansi Autonomous Okrug—Yugra, in the southeast with the Sverdlovsk region, in the south with the Perm Krai, and in the southwest with the Kirov region.

The Vologda region borders the Republic of Karelia in the northwest, the Arkhangelsk region in the north, and the Kirov region in the east. In the south, it borders with the Kostroma and Yaroslavl region, in the southwest and west with the Tver, Novgorod, and Leningrad regions.

The Republic of Karelia borders Finland in the west, Leningrad and Vologda regions in the south, Murmansk region in the north, and Arkhangelsk region in the east.

4 Biodiversity

4.1 Forests of Norway

According to 2020 data, the forested area is 12.8 million ha. The total forests stock of 512 million m^3, where coniferous

species account for 425 million m^3 (82.8%). Norway's forest cover is 27%. The stock is distributed by individual tree species: spruce—42%, pine—28%, birch—18%, oak, beech, and the remaining 10%. Only 65%–70% of the annual timber growth is cut in the country. Forest cultures are created by sowing and planting, 99% of forests artificially created (Nordic Statistics Database, 2024, https://www.nordicstatistics.org).

4.2 Forests of Sweden

As of 2021, the total area of forest land is 27.9 million hectares. Growing wood stock is 3 billion m^3; annual forest growth is124 million m^3; and timber removal per year is 84 million m^3. The volume of logging reaches 70% of the annual growth. The composition of forest species is homogeneous. Spruce (40%) and pine (38%) predominate. Deciduous species (mainly birch) account for 17%. In the country, 10% of forests are naturally regenerated and 82% are artificially regenerated.

Forestry products of Sweden provide raw materials for one of the leading branches of the country—the woodworking industry. Sweden is the fifth largest producer of paper and cardboard in the capitalist world (after the United States and Canada). Every year, Sweden imports wood pulp from Finland and Norway.

4.3 Forests of Finland

The total forest area is 22.4 million hectares; growing wood stock is 1.9 billion m^3. The annual forest growth is 120 million m^3, and annual wood removal is 69 million m^3. The main tree species are pine (over 50% of the forested area) and spruce (about 30%). In the country, 16% of forests are naturally regenerated and 72% are artificially regenerated. About 1/3 of the country's area is occupied by swamps.

The average forest cover in Finland is 65%, and in Sweden 62%. These are the largest forest exporters in Europe (European Union, 2024, https://www.ec.europa.eu).

4.4 Forests of Arkhangelsk region

The total land area of the Arkhangelsk region is 29.3 million hectares. The forest area is 22.3 million hectares, 42.2% of which is covered by stands of spruce, 20.6% pine, 30.1% birch, 4.3%aspen. The forests of the Arkhangelsk region are characterized by an average species biodiversity of tree species. Mixed-species stands (mixed stands of 2–3 species) there are located in 72.3% of the territory. The total stock in the forests of the Arkhangelsk region amounted to 2.7 billion m^3. The annual forest growth is 30 million m^3. Percentage of an area covered by forests is 54%. The forests of the Arkhangelsk region are old-growth forests. Mature, overmature trees dominate in forest stands of all species (42.9%). In the forest fund of the region, exploitation forests prevail—69.2%; protective forests make up 30.8%. The main factors causing the weakening of forest stands are disease damage, mechanical damage, frost damage, and damage by lowland fires. The average stock is 93 m^3/ha. The estimated cutting area is used by 50% (Forest Plan of Arkhangelsk Region, 2018).

4.5 Forests of Nenets autonomous Okrug

The forests are characterized by an average species biodiversity of tree species. Mixed stands of 2–3 species are found throughout the forest area. The main cause of damage to forest stands is windfall. Permafrost and cold soils prevent deep penetration of roots into dense mineral horizons. The total forest area is 446.8 thousand hectares. The forested area of the Nenets Autonomous Okrug is 190.5 thousand hectares, 43.7% of which is covered by stands of spruce, 26.5% pine, 28.1% birch, and 1.7% aspen. The total forests stock is 18 million m^3. Annual growth amounted to 109 thousand m^3 (Forest Plan of Nenets Autonomous Okrug, 2019). Mature and overmature trees dominate in stands of all species (52.7%). The average stock per unit area (1 ha) is insignificant and amounts to 96 m^3/ha. The estimated harvesting area is used by 41%.

4.6 The forests of the Murmansk region

The total land area of the Murmansk region is 10 million hectares. The area covered by forests is 5 million hectares. Young forest stands of natural and artificial origin occupy 6.1% of the forest area of the Murmansk region (Forest Plan of Murmansk Region, 2019a, b). The estimated cutting area on the lands of the forest fund, except for the lands of defense and security, amounted to 5.3 million m^3. Utilization of the estimated cut is no more than 16%. The total stock in forests is 262 million m^3. The annual forest growth is 3.6 million m^3. The average stock per unit area (1 ha) is 48 m^3/ha. Coniferous plantations prevail in the forest fund of the Murmansk region by area, 59.5% and soft-leaved plantations are 37.4%, respectively. Pine, spruce, and birch are the main forest forming species of the region and occupy 96.1% of the forest area of the subject.

A special place in the forests is occupied by territories where vegetation has been destroyed as a result of atmospheric pollution by smoke and gas emissions from nonferrous metallurgy enterprises. The main impact of emissions here is borne by forest vegetation. The destruction of vegetation entails mechanical disturbance and acceleration of erosion processes. Negative changes in climatic characteristics occur in such areas: temperature differences increase, soil freezing intensifies, wind speed and the number of storm weather days increase.

4.7 The forests of the Komi Republic

The total land area of the Komi Republic is 34.3 million hectares, 43.4% of which is covered by stands of spruce, 18.4% of pine, 26.5% of birch, and 3.4% of aspen. The forested area is 25.8 million hectares. The total forest reserve of the Komi Republic is 2.5 billion m^3. Annual growth is 20 million m^3. Of the total wood stock of spruce is 41%, pine is 21.5%, birch is 25.1%, aspen is 6.1%. The role of other tree species in the formation of the total wood stock is not significant. In the age structure of forests of the Komi Republic, mature and overmature forest stands (44.6%) prevail. The average stock per unit area (1 ha) is 100 m^3/hectares (Forest Plan of Komi Republic, 2019).

The total area of forest land in the Vologda region is 11.6 million ha, on 31.9% of which are stands of spruce, pine is 13.8%, birch is 30.2%, aspen is13.4%, gray alder is 5.5%, tree willow is 2.9%, and other species grow. The forested area is 9.8 million hectares. Annual forest growth is 2.2 million m^3. The total stock of trees in the Vologda region forests amounted to 1.6 billion m^3. The area is dominated by mature and overmature forest stands of all species (35.7%). The average stock per unit area (1 ha) is 163 m^3/ha. The forests of the Vologda region play a major role in the preservation of unique flora, conservation, and reproduction of fauna (Forest Plan of Vologda Region, 2019).

4.8 Forests of the Republic Karelia

The total land area of the Republic of Karelia is 14.8 million ha. The forested area of the Republic of Karelia is 9.5 million hectares. Total forest stock is 1 billion m^3. Annual growth is 15 million m^3. Average forest stock is 105 m^3/ha. The annual volume of timber removal by all types of felling on leased forest areas is set at 8.2 million m^3; actual timber harvesting amounted to 6.41 million m^3, or 78% of the established volume. One-third of the territory of the Republic of Karelia is occupied by mature and overmature forest stands (32.2%). This indicates the intensity of logging operations in the republic as a whole. A significant part of mature, overmature forests is located in the category of protective forests, i.e., forests with limited use. The main species—pine (44.2%) and spruce (22.2%) occupy up to 66.4% of the territory of the republic. Birch prevails among deciduous species (26.7%).

More than half of the total timber stock (48.8%) is located in mature, overmature forest stands, 23.9% of the stock is represented by middle-aged forest stands, the share of old-growth forest stands is 10.2%, and the share of maturing forest stands is 17.1% of the total stock (Forest Plan Karelia Republic, 2019).

Distribution of forest types of forests by ecological series (in represented regions) is displayed in Table 23.1.

TABLE 23.1 Distribution of forest types of forests by ecological series (in represented regions).

Pine forests
I Ecological series—reduction of soil richness. Group 1 of forest types—complex pine forests. Group 2 of forest types—green-moss pine forests. Forest types—sagebrush pine forests, grass pine forests, fresh blueberry pine forests.
II Ecological series—waterlogging of soils by stagnant waters. Group 3 forest types—long-moss pine forests. Forest types: bilberry wet pine forests; Longhorn pine forests; pine forests of birdseed; wet shrubby-lichen pine forests. Group 4 of forest types—sphagnum pine forests. Forest types: pine forests: pine forests of sod and sphagnum, pine forests of shrub and sphagnum, pine forests of sphagnum, pine forests of bogs (pine on bogs).
III Ecological series—decreasing soil moisture and richness (originating from the forest type (bilberry pine forest, fresh), completing the II ecological series. Group 2 of forest types—green-moss pine forests. Forest type—lingon berry pine forests. Group 5 of forest types—lichen pine forests. Forest types—moss-lichen pine forests, lichen pine forests; dry shrub-lichen pine forests, rocky lichen pine forests, heath pine forests.
IV Ecological series—waterlogging of soils by flowing waters with strengthening of their flow ability Group 6 of forest types—grass-bog pine forests. Forest types: sedge-reed pine forests, grass-sphagnum pine forests.
V Ecological series—waterlogging of soils by flowing waters with weakening of their flow ability. Group 6 of forest types—grass-bog pine forests. Forest types: pine forests of watch of plant—sphagnum, pine forests of sedge-sphagnum.
Spruce forests
I Ecological series—reduction of soil richness. Group 1 of forest types—complex spruce forests. They are relatively rare in the north-eastern forest taxation area. During taxation they are usually referred to sagebrush spruce forests. Group 2 of forest types—green-moss spruce forests. Forest types: wood sorrel spruce forest types; blueberry fresh spruce forests; herbaceous spruce forests.
II Ecological series—waterlogging of soils by poorly flowing waters. Group 3 forest types—long-moss spruce forests. Forest types: blueberry wet spruce forests, long-moss spruce forests, spruce dwarf forests, willow spruce forests. Group 4 of forest types—spruce sphagnum forest types: spruce shrub-sphagnum forests, spruce sphagnum forests.
III Ecological series—decreasing soil moisture and fertility. The III series originates from the bilberry spruce, fresh forest type, which completes the II ecological series. Group 2 forest types—green-moss spruce forests. Forest type — lingon berry spruce forests. Group 5 of forest types—lichen spruce forests. Forest type—lichen spruce forests.

TABLE 23.1 Distribution of forest types of forests by ecological series (in represented regions)—cont'd

IV ecological series—waterlogging of soils by flowing waters with gradual weakening of their flow ability depending on the reduction of terrain slope, steepness of slopes and hollows. Group 6 of forest types—grass-bog spruce forests. Forest types: lodge pole spruce forests, meadow spruce forests, wet valley spruce forests, grass spruce forest, grass-sphagnum spruce forests, sedge-sphagnum spruce forests, sedge-reed spruce forests.
Larch forests
Larch, unlike pine and spruce, forms a very small number of forest types—wood sorrel forest types, blueberry fresh and lingo berry forests, belonging to the ecological series of decreasing richness of soils underlain by limestone, dolomite and gypsum. As a rule, fir is found only within spruce forests on fresh soils in forest types belonging to the ecological series of decreasing soil richness.

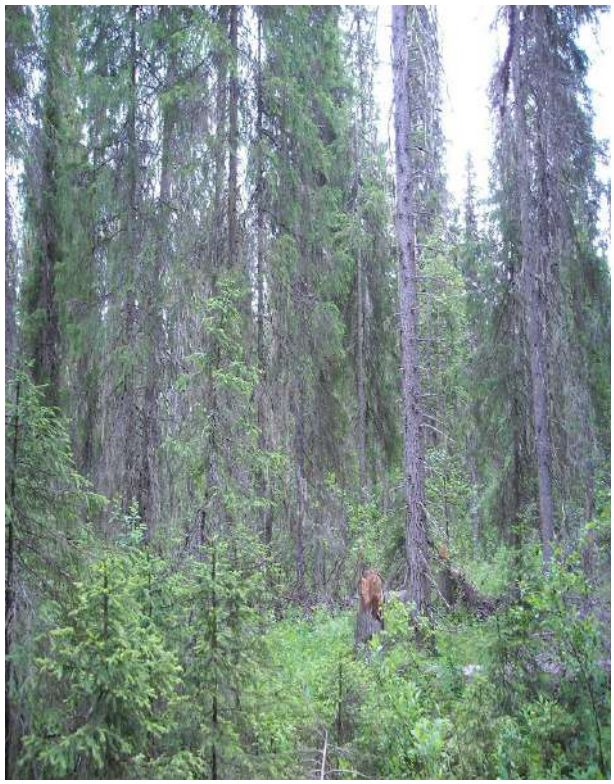

FIG. 23.1 Bilberry spruce fresh forest type (Arkhangelsk region). *(Photo credits: Photo by Surina E.A. (from personal archive).)*

Spruce forests are the most widespread in the North Europe. The most productive stands (green-moss forest types with richly drained soils and pronounced relief: blueberry, lingon berry, and sorrel types of pine and spruce forests) were mainly harvested. Bilberry forest types occupy a significant participation (to 50% of forest covered area) in represented territory of forest fund (Fig. 23.1).

To date, more than 10,000 species have been assessed on the European Red List (The European Red List of Threatened Species, 2024, www.environment.ec.europa.eu). The Swedish Red List 2020 of protected species are: 43 orchid species (all of Sweden's orchid species); 232 other vascular plant species; 12 moss species; 8 lichen species; 5 fungi species; 1 algae species; 250 bird species (all of Sweden's bird species); 27 mammal species; 7 reptile species (all of Sweden's reptile species); 13 amphibian species (all of Sweden's amphibian species); 31 species of invertebrates (Protected Species, 2024, www.naturvardsverket.se).

The 2021 Norwegian Red List for Species contains 4957 species, of which 2.8 thousand species are classified as threatened. This constitutes 21% and 12%, respectively, of the 23.4 thousand species that were assessed for the Red List. Of the threatened species, 289 are categorized as critically endangered (CR), 959 are endangered (EN), and 1504 are vulnerable (VU) (Norwegian Red List for Species, 2021, www.biodiversity.no).

A new assessment of threatened species indicates an increasing loss of biodiversity in Finland (Hyvärinen et al., 2019). Of the 22,000 species assessed, 11.9% were classified as threatened. Almost half (48%) of the nearly 400 habitat types were assessed as threatened. In Finland, 45.3% of species assessments indicate a good conservation status, while the European Union average corresponds to 27.5%. The proportion of assessments indicating bad conservation status is 13.2%, compared to the EU average of 20.6%. Finland hosts 130 protected under the Habitats Directive, which is approximately 9% of the total of the Habitats Directive's species and 250 protected birds, which is approximately 54% of the species protected under the Birds Directive (Finland, n.d., www.biodiversity.europa.eu).

About protected areas, percent (of total land/exclusive economic area (EEZ) in 1950: Finland was 0%; Norway was13.3%; Sweden was 0.1%; in 2022: Finland was 13.3%, Norway—17.7%, and Sweden was 15.3% (Organisation for Economic Co-operation and Development, 2024, www.stats.oecd.org).

Old-aged coniferous forests, which occupy a specific weight, are characterized by relatively low species richness of rare species. In plant communities of bilberry and longhorn groups of coniferous forests, seven species of rare and endangered plants have been described, including: calypso orchid (*Calypso bulbosa* (L.) Oakes), dark-red helleborine (*Epipactis atrorubens* (Hoffm.)), fragrant orchid (*Gymnadenia conopsea* (L.) R. Br.), white Adder's mouth (*Malaxis monophyllos* (L.) R. Br.), which are recorded

mainly in bilberry fresh and wet spruce forests, sometimes long mossy, on carbonate loams and gypsum outcrops: small white orchid *Leucorchis albida* (L.) E. Mey., lesser butterfly-orchid (*Platanthera bifolia* C. Rich), and *Viola mauritii* Tepl. in spruce forests and in pine forests of similar types. Bristly bellflower (*Campanula cervicaria*), peach bells (*Campanula persicifolia*), *V. mauritii* Tepl., male fern (*Dryopteris filix-mas* (L.) Schott) occur in wood sorrel spruce forest types. Ghost orchid *Epipogium aphyllum* (F. W. Schmidt) occur in wet valley spruce forests. Broad-leaved helleborine (*Epipactis helleborine* (L.) Crantz), and marsh helleborine (*Epipactis palustris* (L.) Crantz) were described in the grass-bog group of spruce forests.

In the second half of the 20th century, due to the constant growth of logging operations, there was a sharp increase in the volume of logging, and continuous concentrated logging began to be used in forest practices. The formation of deciduous and mixed forests began on the clear-cutting and burned areas of primary coniferous forests. Logging, significantly changing the appearance of landscapes and causing changes in the abundance of many species, practically does not affect the diversity of local flora. This is due to the fact which habitats with maximum species diversity and habitats of rare and most vulnerable species (wetland and water protection forests, rocky habitats) are little affected by logging (due to existing restrictions and unattractiveness for loggers).

The total species composition of vascular plants, including rare and endangered species, is usually richer than in forest communities prior to harvesting. The species that lived under the stand canopy remain at the clear-cut, finding refuge in clumps of undergrowth, in shrubs, and at stumps; and a significant species group are spread to the clear-cutting from neighboring communities and some species introduced by humans during forestry operations. Secondary forests of small-leaved species, mainly aspen, to a lesser extent, birch, and in river valleys, gray alder have formed everywhere in place of native bilberry-green-moss spruce forests. The forests are predominantly herbaceous. The data obtained indicate that massifs of deciduous and mixed forests are habitats of rare and endangered plant species, primarily in sparse grass, floodplain, and wetland forests. Rare species of vascular plants (21 species) are most represented in coarse-grass forest types. These are primarily the orchid family: lady's slipper [*Cypripedium calceolus* (L.)], [*Dactylorhiza fuchsii* (Druce) Soo], and heath spotted orchid [*Dactylorhiza maculata* (L.) Soo], broad-leaved helleborine [*E. helleborine* (L.) Crantz], small white orchid [*L. albida* (L.) E.Mey], ghost orchid (*E. aphyllum* Sw.), lesser butterfly-orchid [*P. bifolia* (L.) Rich.], calypso orchid [*C. bulbosa* (L.) Oakes], bird's-nest orchid [*Neottia nidus-avis* (L.) Rich.], and military orchid [*Orchis militaris* (L.) Rich.]; furthermore, the following rare plant species can be mentioned: bristly bellflower [*C. cervicaria* (L.) Rich.], peach-leaved bellflower [*C. persicifolia* (L.)], peach-bells (*Conioselinum boreale* Schischk), flat pea [*Lathyrus sylvestris* (L.)], bird-in-a-bush [*Corydalis solida* (L.) Clairv.], Siberian iris [*Iris sibirica* (L.)], Siberian adonis (*Adonis sibirica* Patrin ex Ledeb.), anomalous peony [*Paeonia anomala* (L.)], eastern pasqueflower [*Pulsatilla patens* (L.) Mill.], cowslip [*Primula veris* (L.)], and male fern [*D. filix-mas* (L.) Schott].

Of the wetland forest types in deciduous and mixed forests, the grass-sphagnum forest type should be noted for the highest occurrence of rare species. The following species can grow here: [*C. calceolus* (L.)], spotted lady's slipper (*Cypripedium guttatum* Sw.), Baltic orchid [*Dactylorhiza baltica* (Klinge) Orlova], [*Dactylorhiza fuchsia* (Druce) Soo)], early marsh orchid [*Dactylorhiza incarnate* (L.)], heath spotted orchid [*Dactylorhiza maculate* (L.)], narrow-leaved marsh orchid [*Dactylorhiza traunsteineri* (Saut.)], dark-red helleborine [*E. atrorubens* (Hoffm. ex Bernh.)], broad-leaved helleborine [*E. helleborine* (L.) Crantz.], marsh helleborine [*Epipactic palustris* (L.) Crantz], ghost orchid [*E. aphyllum* (F.W.Schmidt)], fragrant orchid [*G. conopsea* (L.) R. Br], white adder's mouth [*M. monophyllos* (L.) Sw], and lesser butterfly-orchid [*P. bifolia* (L.) L. C. Rich.]. Special attention among vulnerable species should be paid to the [*C. calceolus* (L.)], which inhabits waterlogged birch forests, developing in ecotopes with key nutrition.

To date, one of the main tasks for the protection of forest ecosystems in the taiga zone has been the preservation of native coniferous forests. The Red Book of the Arkhangelsk region contains information about 360 species (63 species of fungi, 58 species of lichens, 53 species of mosses, 16 species of algae, 101 species of vascular plants, 8 species of invertebrates, and 61 species of vertebrates).

Currently (third edition), the Red Data Book of the Komi Republic includes 532 taxa: 150 fungi (including lichens), 314 plants, and 68 animals. The third edition of the regional Red Data Book and its Annex 1 include 19 animal species listed in the Red Data Book of the Russian Federation (Dronova, 2001; Kamelin, 2008). The Red Book of the Republic of Karelia contains 774 species, including 442 species of plants and 332 species of animals.

The fauna of the Murmansk region is relatively poor. No species peculiar only to the Kola Peninsula been found. Many mammals of the North, forced to make constant migrations in search of food are characterized by the gregarious way of life. There are 32 species of terrestrial mammals registered in the region. Among terrestrial mammals, the most numerous are rodents (13 species).

The Red Book of the Vologda region includes 202 species of vascular plants, 36 species of mosses, 4 species of algae, 31 species of lichens, and 21 species of fungi.

5 Conservation area and their ecological indicators

At present, there are 111 specially protected natural territories in the territory of the Arkhangelsk region (8 of federal significance, 100 of regional significance, and 3 of local significance). The total area of specially protected natural territories located in the Arkhangelsk region without the sea water area is a million hectares.

There are three state reserves in the Murmansk region: the Pasvik State Nature Reserve in the Pechenga District, on the border with Norway; the Laplandsky State Nature Reserve near the city of Monchegorsk, and the Kandalaksha State Nature Reserve on the islands of the White and Barents Seas. The territories of the reserves contain unique fragments of pristine spruce taiga, massifs of typical northern taiga native pine forests.

Vologda region: Russky Sever National Park, Darvinsky State Reserve.

The nature reserve fund of the Republic of Karelia consists of 150 with a total area of 1 million hectares, which is 5.6% of the republic's area. On the territory of the Komi Republic there are nine of federal significance with a total area of 564.8 thousand hectares, including 368.8 thousand hectares, on the lands of protected areas, 196,000 ha on the lands of the forest fund, water fund, and lands of other categories.

The most effective method of conservation of rare plant species and plant communities is the designation of specially protected natural territories and specially protected forest areas. However, information on their location is insufficient; so, a multilevel system of identification and allocation of habitats of rare, threatened, and endangered species is necessary. This requires the ability to recognize them and properly allocate them in all types of economic activities.

Main indicators: attitude to soil moisture; attitude to fertility; light conditions; landscape region, landscape group, landscape type; distribution by climatic zones; habitat, forest type.

6 Human activities and their impacts

One of the predominant uses of forests is timber harvesting. In Finland, Norway, and Sweden, timber processing is the leading branch of the forest industry. Most of the products are exported to different countries. Rational use of forest resources provides for a whole complex of measures on reforestation, forestry operations, utilization of forest raw material resources, and forest protection and conservation.

Forests are intensively used for forestry industry, agriculture, primarily for haying and grazing of farm animals, as well as beekeeping. According to the forest registry, the total area of hayfields is 175,000 ha; and the area of forest pastures is 397.5 thousand hectares. Biological stock of the main types of food forest resources includes: wild berries, 923.5 thousand tons and mushrooms, 497.4 thousand tons. The main species of honey-bearing plants of the tree and shrub layer of Russian forests are linden trees (honey productivity up to 1000 kg/ha), willows (up to 150 kg/ha), and others. The areas occupied by these species are rather significant at 107,000 ha. The total area of reindeer pastures is more than 300 million hectares.

One of the main types of raw material resources in forests is wood—an indispensable raw material for many sectors of the economy. The demand for it is constantly increasing.

Transportation inaccessibility of plantations for small consumers and the resulting increase in the cost of harvesting also serves as a barrier to increasing logging volumes. The high cost of timber products is the main reason for their uncompetitiveness compared to other more southern regions. This is due to low exploitable timber reserves in mature stands and poor technical equipment of enterprises for timber harvesting.

Due to broad, wide-ranging changes in environmental conditions, it is quite common for changes in plant distributions to occur that are presented as range shifts. Range shifts are the result of two parallel, independent processes: the dispersal of a species in one direction, leading to an expansion of its range, and the extinction of a species at the opposite edge of its range, causing a contraction of the latter in that part of the range. Depending on either the course of both processes, progressive (settlement of new space) and regressive (reduction of previously occupied territory) features may prevail in range shifts.

Given this assumption, for each plant species it is necessary to establish, if possible, the reasons for its rare occurrence, which may be the following:

1. Progressive deterioration of climatic conditions for species that were widespread and adapted to these conditions in the previous period of the postglacial (after the Valdai glaciation) with climatic conditions favorable for them. Further changes in climatic conditions may lead to complete extinction of this category of species.
2. Rarity of plant species adapted to modern climatic conditions due to changes in the species composition and structure of forests during the postglacial period. At first, the melted glaciers were replaced by treeless areas covered with herbaceous vegetation—characteristic of tundra, bogs and steppes. Probably, plants that have become rare nowadays also took part in the composition of this vegetation. Subsequent primary forests on these expanses were formed mainly of deciduous species, mainly aspen and birch, and then of oak and its companions. The primary forests were sparse and light,

favorable for the settlement and widespread distribution of the now rare, light-loving, lower tier plant species. Gradually, deciduous forests became complex, more closed, increasingly shading the ground cover with their upper canopy. Favorable habitats for most of the light-loving plants became less and less. They began to occur less and less frequently. Especially unfavorable for many of them, ecological conditions were formed under the canopy of later settled conifers, mainly dark conifers—spruce and fir. The centuries-long predominance of dark conifers in taiga forests resulted in a significant reduction in the habitats of many now-rare plants. Strong forest fires in coniferous forests, which destroyed not only seeds, but also roots, rhizomes, bulbs, and tubers of plants, contributed to this. It should be emphasized that climatic conditions for the category of plants in question remained favorable, not preventing their wider distribution under certain groundwater conditions, composition of tree species in forest stands, and species composition of vegetation on burned areas and clear-cutting.

That is, they can grow indefinitely in certain forest types without the threat of their disappearance or extinction, and they can find favorable ecological conditions and expand their distribution and frequency of occurrence in burned areas and clear-cutting after the death of natural coniferous forests. Forest management measures can contribute to this process.

3. Some rare plant species are not widespread due to the incomplete process of their dispersal from neighboring regions—western, eastern, northern, and southern. This category of species finds favorable climatic and soil conditions in the region, and there is no real threat of extinction or extinction for them.
4. The reason for the rare occurrence of some species is their ecological adaptation to specific soil and subsoil conditions, especially those of low distribution. These are soils underlain by limestone, dolomite, gypsum, marl, etc., as well as habitats confined to outcrops of these rocks on the Earth's surface. Therefore, the nature of distribution of this category of plants is limited. Some of the rare species growing under ecological conditions that are not widespread are classified as relict or endemic.
5. A specific reason for the rare occurrence of some plant species is their occurrence in transitional ecological conditions, such as tundra-taiga. For this category of species, there seems to be no threat of extinction.
6. Currently, one of the main reasons for changes in the ranges of rare, threatened, and endangered plant species is human activity in the forest and their numbers and proportion of their presence in different forest biocenosis (not only decreasing, but also increasing) is human economic activity in the forest. As a result of clear-cutting, natural coniferous forests disappear and are replaced by secondary deciduous or mixed deciduous-coniferous forests. Ecological conditions change accordingly, first in clear-cutting and then under the canopy of secondary forests. Changing ecological conditions may, in some cases, worsen for some rare plant species, leading to a decrease in their occurrence and distribution, as well as to an increased risk of their complete disappearance or extinction; in other cases, ecological conditions for a number of species may improve, contributing to an increase in their occurrence and distribution.

Studies of forest vegetation changes triggered by human impact in the Arkhangelsk Region have been a major field of research for many years (Chiviksina et al., 2000; Tsvetkov et al., 2000). Of special interest are vegetation changes and phytocenosis composition (succession) under different levels of human (anthropogenic) impact within Kargopolsk-Plesetskaya forest area (for example).

Biodiversity has been a popular field of research ever since the early 1990s, reflecting growing concerns due to a worldwide deterioration of natural resources (Rio Conference, 1992). Forestry in countries is under increasing pressure from environmental protection groups and ever-broader parts of the society to steer forest management in a more sustainable way. Mapping biodiversity indicators scheme (methodologies) and assessment of biodiversity in forest areas.

A core indicator of target system sustainable forest management in Russia calls for the "Diversity of species groups and ecosystems." The target system defines sustainability via different criteria and indicators in the comprehensive sense by taking into account economic, ecological, and social aspects. Assessment is on the bases: five indicators for species (organismal) diversity and nine indicators for ecological diversity. Investigations are carried out on three different categories (subunits) of the Kargopolsk-Plesetskaya forest area (Arkhangelsk region) (main unit), with each subunit categorized by differing degrees of vegetation changes caused by human impact: unchanged forestlands; slightly changed forestlands; and strongly changed forestlands. Criteria for categorization of Kargopolsko-Plesetskaya forest territory by human impact were: degree of species changing; share of young forests; degree of changing of natural complexes; development of road network; and distance from industrial centers.

The investigated subunits are comparable in terms of the potential variety and structure of forest phytocenosis, climate, and soil. Data describing alterations in forest phytocenosis taking place in these subunits stem from representative sample plots.

Traditional methods of geo-botanic description are used. The changes of forest cover are: from forest area to nonforest area; structural redistribution of forest area by planting different predominant forest species (afforestation); change of the predominant phytocenosis of the forests; the composition and structural diversity of phytocenosis; the composition and organization of forest parcels; and development of different types of succession.

The criteria of referring the forest territory to one of the categories of human impact were: degree of species changing; increasing of share of young forests; degree of changing of natural complexes; development of road network; and distance from industrial centers. Distribution of vegetation in ecosystems of all three categories of the territory was taken into account on several levels:

1. Macrohomogenous—forced integration of areas with different types of biogeocenosis to one forest site (10 ha and more).
2. Mesomosaic of sites—natural combinations of close-to-nature types of biogeosenosis (associations) in the volume of practically understood forest types defined by Sukachev (up to 10 ha).
3. Meso-micromosaic—natural structure of biogeocenosis (within 1 ha).

While two first levels can be registered with the help of air photos, the third one requires fieldwork.

Unchanged forest territory (38% of forest fund) are mainly covered by forest ecosystems of the mature types with average composition of 42% pine, 28% spruce, and 30% birch. The forests on slightly drained and wet habitats are dominating (*Sphagnosum*, *Ledum palustre,* and *Sphagnosum* types—39%; *Polytrichosum* and wet *Myrtillosum* types—28%). Among nonforest ecosystems (14%), swamps (comprising 65%–78% of nonforest area), as well as hayfields are dominating.

The hayfield ecosystems are mostly occupied by forest communities of drained, swampy, and alluvial condition types. The forest and nonforest ecosystems are represented by complex communities with typical taiga, swampy taiga, and forest-hayfield flora. The changed forest areas (about 34% of area) the same as complicated groups: formations of mature forests suffering systematic load by human, old secondary, as well as young and middle-aged forests with dominant main tree species. Generalized composition of the forests is 38% pine, 32% spruce, and 30% birch (aspen). The forest territory is represented by approximately the same types of habitat: wet *Myrtillosum* (24%), *Polytrichosum*, *Shagnosum,* and involving wild rosemary *L. palustre* (L.) plants are dominating (31% in total). *Myrtillosum* forest type (fresh) (22%) are spreading forests type. The vegetation changes have different natures. The old forests are characterized by processes such as impoverishment of species list, oppression, degradation, and even elimination of some species. In secondary forests, along with reduction of representatives of some species of primary vegetation, the appearance of hayfield representatives and weed flora is noted. In connection with temporary processes swamping the forest territory of *Hylocomiosa* and *Herbosa* habitat types after cutting the temporary settling of *Polytrihum* and *Sphagnum* species is possible, as well as some species of sedge and horsetails. The total amount of vegetation species in mature secondary forests does not decrease. At the same time, the changing of structure of sinusia of moss-lichen and subshrub-grass layers is taking place. To the age of ripeness of secondary forests under weak influence, the species structure lower layers restored. Strongly changed lands form 28% of forest territory of provenance. This includes all young and middle-aged forests, as well as forest cultures in the forests of green zones and protective belts.

The open forest areas (cut areas, burnt places, wastelands) are referred to this category as well as no forest lands (homesteads, roads, careers, garbage dumps, etc.). The share of forest-covered area is approximately 88%–90%. The average composition consists of 13% pine, 43% spruce, and 44% birch (aspen). The forest ecosystems of former fresh and wet *Myrtillosum* types are dominant. Currently, these ecosystems are at different stages of deviations from their original appearance. They are characterized by different correlations of primary and secondary hayfield and weed vegetation types. The condition of primary and secondary vegetation differs greatly on different areas. The communities of primary vegetation exhibit greater variability of condition. Weed and ruderal vegetation on urban territories, in some cases, can substitute for forest species, as well as crowd out representatives of hayfield and margin communities.

According to the inventory data from 2021, total anthropogenic greenhouse gas emissions in Russia amounted to 2156.6 million tons of CO_2-eq. excluding the land use, land-use change and forestry (LULUCF) sector, and 1671.8 million tons of CO_2-eq. including LULUCF. Compared to 1990, the base year for the UNFCCC and the Kyoto Protocol, in 2021, total anthropogenic greenhouse gas emissions decreased by 45.9% when the LULUCF sector is taken into account, and by 31.9% without it. In 2021, anthropogenic sequestration of greenhouse gases in the LULUCF sector offset approximately 22.5% of total anthropogenic greenhouse gas emissions from industrial sources. The significant forest area of the Arkhangelsk region has been declining for the past 30 years. According to the field results (2009, 2011–2012-ss), it was determined that the main factor is aged forest stands, forest sites, and climatic conditions. According to our experiments, the main cause of spruce forest mortality is their age limit; therefore, both factors (insects and decay) are not the major initial causes of spruce forest die-back, but they are the

consequences of the situation when spruce forest stands attain their maximum natural age when mortality occurs and the old trees are replaced by younger poles. At the same time, findings from other studies on spruce stands located in the Arkhangelsk region point to other factors reducing the age limit of spruce forests, such as groundwater-level change, climate change, including climate anomalies, nutrient-poor soils and deep-freezing soils, recreation, root and butt root, and insects. According to A.V. Zhigunov, T. A. Semakova, D.A. Shabunin (from St. Peterburg Research Institute of Forestry), the phenomenon of mass die-back of forests spread throughout the European territory of Russia is in the context of the mass die-back of forests growing within the postglacial areas of the entire northern hemisphere. The main cause of mass die-back of forests in Northwest Russia is now recognized as a biotic factor—root rot. Forests of northern taiga with their low productivity are very sensitive to anthropogenic impacts. The influence of climate on the forests die-back has not been sufficiently studied, and no models have been found for the investigation area. However, there is an evidence of air temperature rise as compared to the average long-term data, and of irregularity of precipitation. These factors cause three times and more variation in Selyaninov hydrothermal coefficient (HTC) from year to year. Strong winds and extreme winter climatic conditions favor windfall and windbreak. The process of ash-gray soil small depth adversely affects the water-air regime of soils. Under such conditions, the soil layer available to plants above the layer hinders the availability of soil moisture from the lower horizons. On the contrary, in wet years water logging is probable.

In Kargopolsky forest division where the research was conducted, the air temperature rise during the vegetation period as compared to the long-term average data (1881–1960) was up to 1.9–6.1°C during the years when the forests die-back was intensive. Amount of precipitation varies greatly from year to year. As a result, the Selyaninov HTC changes by three or more times. In 1997, the amount of precipitation was the least (116 mm during the vegetation period). This was the year when die-back of spruce forests started in the Dvina and the Pinera interfluve. According to most researchers, the expected increase in summer climatic parameters' difference have even more negative effects than just change in absolute indices. Maximum wind speed during the research period ranged from 12 to 23 m/s and exceeded the critical values. Recurrence rate of winter months with thawing weather is 82%–91%. At the same time, thawing weather and strong frosts often interchange and the temperature can drop to −50°C. Based on the foregoing, the following facts typical for the research area can be noted: the lack of precipitation in combination with relatively high air temperatures at the beginning of the vegetation period, strong winds and dry years, three or more times difference in annual Selyaninov HTC, and average air temperature anomalies increase. All these factors could affect the resistance of spruce stands to windfalls and snow breaks and they could not but cause the consequences. We found damage caused by fungal diseases. The mottled wood rot is caused by the root sponge [*Heterobasidion annosum* (Fr.) Bref], wound rot is caused by bleeding conifer crust [*Stereum sanguinoletum* (Alb. & Schwein.)], mixed rot is caused jointly by the honey fungus and root sponge (Fig. 23.2).

FIG. 23.2 Root sponge *Heterobasidion annosum* (Fr.) Bref. *(Photo credits: Photo by Surina E.A. (from personal archive).)*

The pine sponge *Phellinus pini* (Brot) A. Ames causes the variegated core rot in spruce *Phellinus chrysoloma* (Fr.) Donk.

White core rot of deciduous trees is caused by the false rot fungus *Phellinus igniarius* (L. ex. Fr.) Quel and the closely related rot fungus *Phellinus nigricans* (Fr.) P. Karst.—birch, *Phellinus tremulae* (Bond.) Bond. et. Boriss.—aspen, white rot caused by *Fomes fomentarius* (L.) Fr. Brown rot of conifers is caused by the fringed fungus *Fomitopsis pinicola* (Fr.) Karst., in birch—*Piptoporus betulinus* (Bull.) P. Karst.

We took the differences between the trends of regional anomalies of mean annual air temperature calculated from 39 CMIP–5 models and from observational data (for 1940–1969 and 1976–2005), which are called model errors in the figure presenting the results of comparison of precipitation trends (Fig. 23.3).

It follows from the data that the scatter of indicators between the models is quite large.

For the average annual temperature, the differences between the observed and model trends range from 9.4 to −8°C/100 years, and the differences between the observed and model trends of annual precipitation amount are

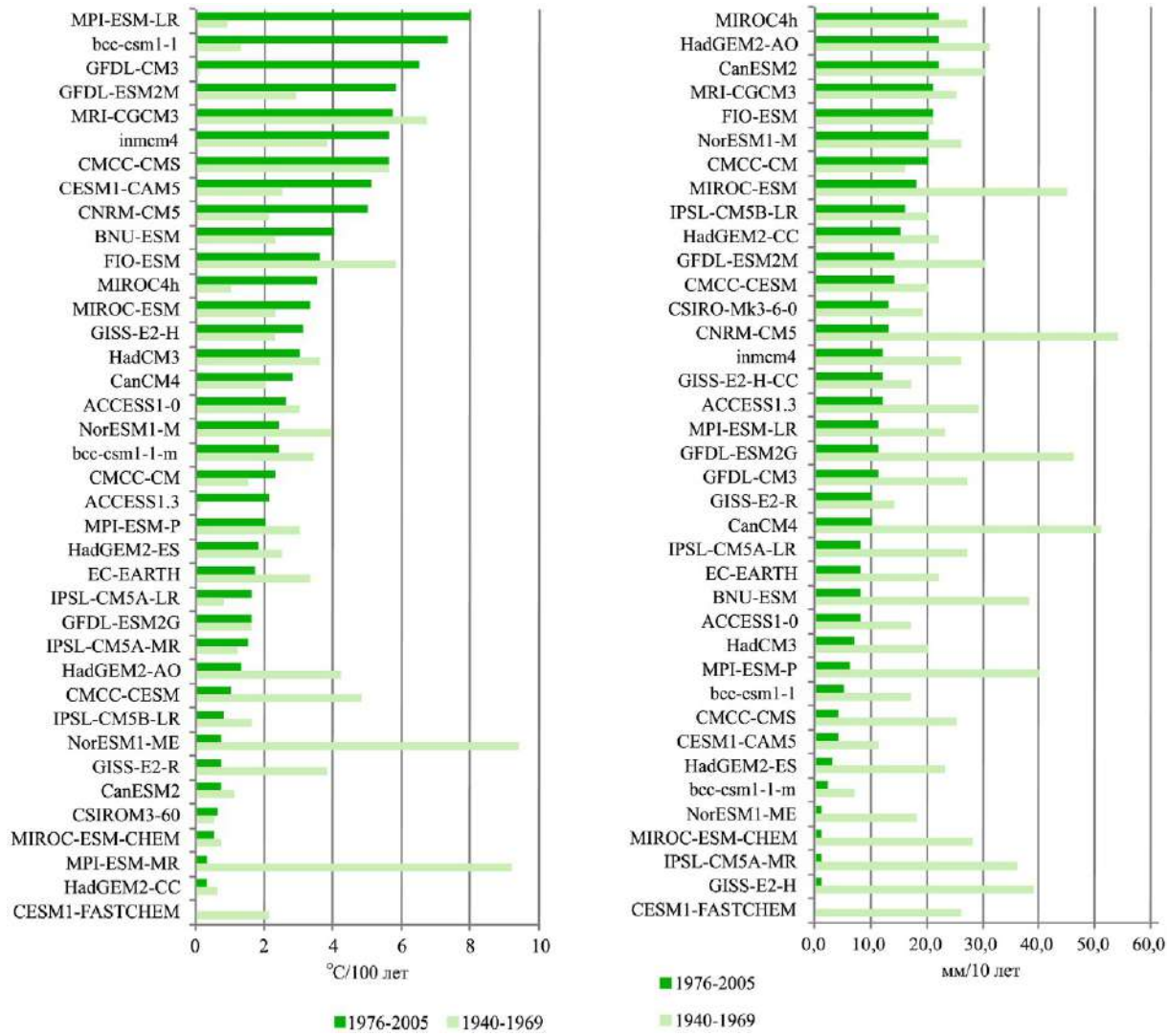

FIG. 23.3 Error of reproduction of the trend of air temperature (A) and annual precipitation (B) by CMIP-5 models for two periods (modulo) for the study area. *(Photo credits: by Surina E.A., leader NIOKTR No. 114091270038 "Assess the impact of forest management activities on the carbon cycle and the development of adaptation scenarios; the system of forest management in managed forests of northern and middle taiga of the European-Ural part of Russian Federation connection with the expected changes in climate" (2014–2016-s) (www.rosrid.ru).)*

22–54 mm/10 years. Forecast of changes in the sum of temperatures and duration of the growing season, including those with active temperatures (above 10°C) for the study area, is given in Table 23.2.

Taking into account the fact that the duration of the growing season is predicted to be 161–182 days, and with active temperatures 115–132 days, we can consider the difference in the change of indicators between the middle and northern taiga to be insignificant: for the duration of the growing season, it is 1–3.5 days, and for the sum of temperatures the difference will be only a couple of tenths of a degree per day, which can be attributed to modeling error.

The magnitude of the predicted absolute values of the sum of temperatures above 10°C and above 5°C, as well as the duration of the growing season, including active temperatures, are given in the tables. For the period 2046–2064, the average annual sum of active temperatures is forecast to be 1803–1881°C, and temperatures above 5°C are forecast to be 2154–2259°C. The duration of the growing season will be 161–167 days, and the duration of the active growing season with temperatures above 10°C will be 114–117 days.

The figure shows that the predicted HTC lies outside the norm of 0.38–0.57 practically in all years in the northern taiga and throughout the projected period in the middle

TABLE 23.2 Change in the average annual sum of surface air temperatures and duration of the growing season in the northern and middle taiga of the European-Ural part of Russia for 2046–2064 in relation to the base period 1981–2000.

Forest district	Change in the sum of temperatures (°C) and duration of the growing season (days) under SRES scenarios											
	A2				A1B				B1			
	Sums of temperatures		Vegetative period with		Sums of temperatures		Vegetative period with		Sums of temperatures		Vegetative period with	
	$\Sigma t>10$	$\Sigma t>5$	$t>10$	$t>5$	$\Sigma t>10$	$\Sigma t>5$	$t>10$	$t>5$	$\Sigma t>10$	$\Sigma t>5$	$t>10$	$t>5$
Northern taiga	631.1	613.9	34.9	33.2	671.7	664.4	36.5	36.3	593.9	560.2	33.9	30.0
Middle taiga	639.7	654.1	32.4	34.5	684.5	704.2	34.3	37.3	582.5	587.3	30.4	31.3
Different taiga	−8.6	−40.2	2.5	−1.3	−12.8	−39.7	2.2	−1.1	11.4	−27.0	3.5	−1.4

taiga of the study area. Consequently, the projected climate change for the area of spruce forests distribution is beyond the natural climatic limits of their distribution area, according to the required (HTC) (Figs. 23.4 and 23.5).

Thus, an increase in temperature and precipitation is predicted for the study area. The amount of precipitation prevails over evaporation, which causes an increase in runoff. Climate warming, especially in winter, leads to a weakening of the influence of permafrost processes and phenomena, to changes in the existing drainage capacity of the hydrographic network, which may lead to a decrease in the groundwater level in the watershed zones and to an increase in the lower parts of the watersheds. This will negatively affect the established plant communities adapted to a certain water and hydrochemical regime. The prevalence of spruce plantations in the study area will only increase the negative effect due to the specific location of the root system of trees and soil structure.

Climate modeling data show that changes in the indicators are identical, but absolute values differ significantly. The growing forest stands are characterized by a rather wide range of ecological conditions with certain climatic indicators. Exceeding the limit of climatic norms of the indicators of the area of predominant species distribution may indicate a threat to the equilibrium of their condition, up to the complete disappearance of formations. Analysis of the average annual surface air temperature shows that, in the northern taiga, the excess is predicted under two scenarios (A2, A1B) in the second half of the century (2060–2063), as well as under the sum of temperatures above 5°C (2051–2058) and the sum of temperatures above 10°C (2050–2057). Under scenario B1, the indicators do not reach the maximum permissible values here. The duration of the projected growing season in the northern taiga does not exceed the available limits. In the middle taiga, exceeding of climatic norms for the area of dominant species is predicted much earlier and under all scenarios: in terms of mean annual surface air temperature in 2047–2053 under A2 and B1 scenarios. Even earlier than 2046 under the A1B scenario; in terms of the sum of temperatures above 5°C and the sum of temperatures above

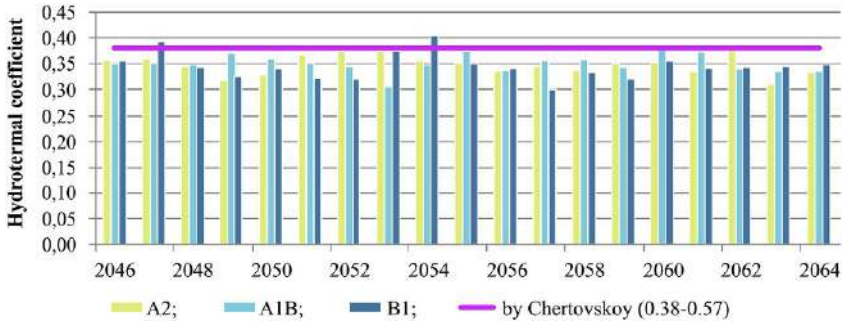

FIG. 23.4 Hydrothermal coefficient in the northern taiga of the European-Ural taiga region under three SRES scenarios. *(Photo credits: Surina E. A.)*

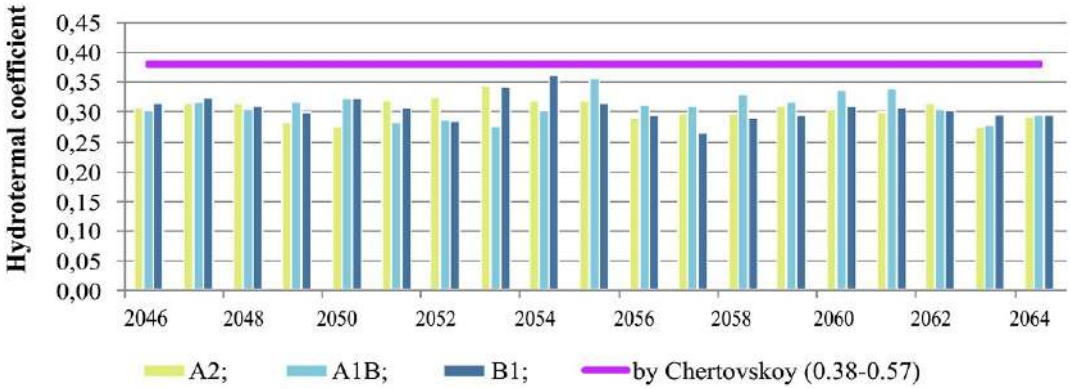

FIG. 23.5 Hydrothermal coefficient in the middle taiga of the European-Ural taiga region under the three SRES scenarios. *(Photo credits: Surina E. A.)*

10°C; duration of the growing season—from 2049 (A2 and B1) and earlier than 2046 (A1B). The predicted HTC (2046–2064) also lies outside the natural range indicators (Tarakanov et al., 2017; Surina and Senkov, 2018).

Consequently, with the exception of scenario B1 for the northern taiga, we should expect from the middle of this century to exceed some climatic indicators characteristic of the area of the predominant formation of spruce forests, which will entail deterioration of sanitary conditions, high susceptibility of forests to any negative influences (snow, snowfall, windfall, insect damage, forest diseases, changes in groundwater levels), as well as a significant increase in fire danger.

The changes will be expressed in the growth of the area of dead forests, primarily from fires and unfavorable weather conditions and soil and climatic factors.

Accordingly, scenarios for adaptation of forest management in managed forests based on the studied new generation models for greenhouse gas scenarios growth should be based on measures aimed at reducing the area of dead forest stands.

Comparing the tabulated values with the predicted data from the ensemble of CMIP models, we obtained the following results for the study area (Fig. 23.6).

7 Operational and potential tools to conserve or to recover the biome

It has been found that the carbon balance is strongly influenced by the forms of forest management. Since climate change mitigation opportunities are largely related to forests (reduction of forest losses from fires, pests, diseases, other disasters, intensive reforestation and afforestation, etc.), the organization and planning of forest management activities is an important step toward adaptation and mitigation of climate change.

Analysis of the results of simulation experiments shows that, for all objects in different natural conditions, the greatest accumulation of carbon in the forest ecosystem occurs in the scenario without impacts. The most preferable scenario of carbon conservation (60–80 years for harvesting cycling): forest measures (soil mineralization and thinning and sanitary cuttings).

The increase in mean annual temperatures leads to an increase in phytomass and a decrease in carbon stocks in forest litter, while relatively increasing carbon accumulation in the organic matter of soil mineral horizons.

By increasing harvesting intensity and reducing harvesting turnover, it is possible to increase the participation

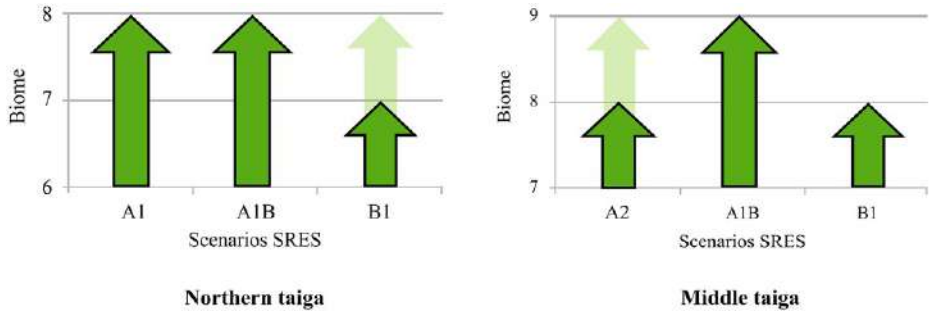

FIG. 23.6 Change of vegetation zones for the period 2046–2064 (6, northern taiga; 7, middle taiga; 8, southern taiga; 9, subtaiga (mixed forest); the light outline shows possible variants of events). *(Photo credits: by Surina E.A., leader NIOKTR No. 114091270038 "Assess the impact of forest management activities on the carbon cycle and the development of adaptation scenarios the system of forest management in managed forests of northern and middle taiga of the European-Ural part of Russian Federation connection with the expected changes in climate" (2014–2016) (www.rosrid.ru).)*

of coniferous species in the species composition. This will only contribute to an increase in productivity and timber quality, as well as in reducing the risks of fires and forest diseases. Further, utilization of more wood from thinning operations contributes to forest conservation.

It is difficult to overestimate the impact of forest fire protection on carbon storage. Other measures also play a significant role: insect and disease control, measures to promote natural regeneration, and forestry work. They contribute to increasing the sustainability and productivity of forests and, consequently, to enhanced carbon storage. The higher the level of management, the stronger its impact on the carbon cycle.

When identifying places of growth of rare and endangered species and populations of taiga plants with the purpose of their conservation or restoration, it is necessary to take into account that ecological conditions (climatic and groundwater) are continuously changing in the geographical space of the region. Certain combinations of ecological conditions and biogeocenotic factors favorable for a particular plant species make its growth in one or another part of its range probable.

In the process of identifying locations of rare and endangered plant species within their ranges, climatic conditions, which have already influenced the size and shape of the ranges during their formation, can be neglected and all attention can be paid to groundwater conditions and biogeocenotic (typological) composition of forests. Groundwater conditions at the macro level are characterized in geographical atlases by landscape areas and landscape types, and at the meso-level, by forest types in typological classification schemes. Forest types are also characterized in typological classification schemes.

Geographical landscapes and landscape types are distinguished within landscape areas. Geographical landscape is an ecological system (geosystem, natural territorial complex) with predominance of genetically homogeneous, interrelated, and mutually conditioned by structure and metabolism combinations of relief forms, geological basement, quaternary deposits, soils, microclimate, hydrological network, and phyto- and zoocenosis, located in the same climatic conditions. Landscapes that are similar in terms of the structure of their components and other features are grouped into landscape types. Each landscape type is characterized by specific interrelationships among the ecosystems at the lowest taxonomic level that make it up, and within these ecosystems—of their constituent components. Each type of geographic landscape is characterized by the prevalence and regular spatial distribution of certain forms of relief and quaternary deposits, types, genera and species of soils, hydrological network of certain composition, structure and density, certain types of microclimate, and certain types of communities, including plant communities.

Forest biogeocenosis are called forest stands in silviculture. Their elementary classification unit is the forest biogeocoenosis type, or forest type. Forest biogeocenosis serve as elementary structural systems that make up a forest geographic landscape. The terms “forest biogeocenosis” and “facies” are synonyms. Forest biogeocenosis type or forest type is an elementary classification unit of forest landscape type.

The following concepts should be used consistently when identifying the locations of rare, threatened, and endangered forest plant species: geographic range of the plant species, landscape area, geographic landscape type, and forest type. These concepts reflect with exhaustive completeness the relative homogeneity of climatic conditions, relative homogeneity of soil conditions (underlying and soil-forming rocks, i.e., geological basement and quaternary deposits on it) within the landscape area; and higher homogeneity of climatic, soil conditions, and forest vegetation cover (e.g., pine forests). The relative homogeneity of climatic conditions (underlying and soil-forming rocks, i.e., geological basement and quaternary deposits on it) within the territory of a landscape area; higher homogeneity of climatic, soil-groundwater conditions and forest vegetation cover (e.g., pine forests or spruce forests) within a set of landscapes of the same type; high homogeneity of ecological conditions (climatic and soil).

The complex of all landscape areas, landscape types, and forest types can represent the entire diversity of ecological conditions within a region and the entire diversity of taiga vegetation, the species and population composition of which, including rare and endangered plant species, is closely related to the specific ecological conditions of a particular landscape area, landscape type, and forest type.

Allocation and preservation of habitats of rare flora species in the course of forestry operations and forest management, along with the existing protected areas and special protection area, to create conditions for conservation important rare forest communities, increase, in general, the species diversity of taiga ecosystems.

The location of rare, threatened, and endangered rare plant species may be identified and delineated in the process of forest inventory, harvesting, survey and design work before, also during implementation of various measures, both as key habitats in large areas within a taxation unit or several units, and as key habitats within a taxation unit.

Key habitats are elements of geographical landscapes that are of special value for biodiversity conservation. They are characterized by a high level of species diversity of forest communities. They include river valleys, large streams, lakeshores, bog edges, steep slopes, forest islands among treeless bogs, high-age mixed forests with predominance of overmature aspen, areas with a distinctly mosaic forest environment (different-aged stands, “windows” in the canopy, presence of deadwood, tall stumps, dead wood,

deciduous trees, etc.), which creates conditions for the existence of many rare plant species.

Key habitats within a forest taxation unit represent elements of forest environment, mesorelief and microrelief with the most valuable natural characteristics, where rare plant species exist or are expected to exist. They are characterized by a spatially limited set of environmental conditions that support the entire development cycle of individuals, populations, or species.

Key habitats include small watercourses, waterlogged depressions, springs, small lakes that are sources of streams, rocky areas and rock outcrops, karst sinkholes, accumulations of large deadwood in late stages of decomposition with biogroups of natural reforestation, individual high-growth trees of pine, aspen, willow, and birch.

In key biotopes and habitats of rare plant species, different restrictions of economic activities are envisaged, depending on their requirement for preservation of the forest environment:

1. Research of a new special protection area with a complete prohibition of economic activities, if the species is demanding to preserve the forest environment or is located within a rare vegetation community.
2. Stopping of main-use logging if the species is located within an existing special protection area.
3. Limitation of logging in the part of the allotment where a population of a rare species is found.
4. Stopping of laying trails and crossing of machinery through watercourses, waterlogged depressions, near springs, springs, sources of streams and rivers, where rare species are found or may grow.

Preserving the integrity of habitats of rare species confined to low lying areas and watercourses will allow, in addition to the main objective, not to disturb the hydrological regime of the territory and avoid waterlogging in large areas of clear-cutting, not to pollute water sources and, in general, to improve environmental protection activities.

The Red Book of the Russian Federation, Red Books of the regions of the Russian Federation and the approved list of protected plants serve as the basis for the conservation of rare species.

For a preliminary study of forest areas in order to identify the most probable habitats of rare species, schemes of their distribution areas, maps of geographical landscapes, forest plantation plans, aerial photographs, or high-resolution space images can be used. The latter can also be used for preparation of outlines, projects of forest management measures, technological maps for various types of works, and for making current changes in forest inventory materials.

When certification is carried out in the lease bases of logging enterprises for identification of rare species, special field studies are conducted by experts with the issuance of an appropriate conclusion.

Prior to forest management activities, the most characteristic habitats of rare species are the basis of available information (taxation descriptions, cartographic materials, forest plans, aero survey, space survey, etc.). During fieldwork of the site, a detailed study was undertaken, surveying them to identify key habitats and habitats of rare species. The identified key habitats and habitats have been mapped on the outline of the area of the projected forest management measures and plans with the indication of the type of special protection area. In special protection areas, seed clumps projected on the harvesting areas have been identified. Pine forests area on poorly drained soils located within 0.25–0.5 ha, for spruce forests—at least 0.5 ha. Measures for the preservation of key habitats and habitats of rare species to be ensured in the course of works are given.

During the acceptance of the completed works, environmental protection measures are assessed, including the conservation of key habitats and habitats of rare species. In case of violation of the requirements for the preservation of key habitats, the perpetrators are held liable in accordance with the applicable forestry and environmental legislation.

8 Conclusion

The research has established that the natural biological diversity of taiga vegetation, including rare and endangered plant species, is the main natural value of northern European forests, which determines their ecological, raw material, and social value.

The main natural value of rare and endangered plant species is their biological value, consisting in the role of these species as repositories of the unique species and population genetic fund necessary for the conservation of biological diversity on the planet.

Anthropogenic transformation of landscapes is mainly caused by logging operations. On significant territories, there is a change of indigenous coniferous plantations with deciduous ones, which are less valuable in economic and silvicultural terms. In the vicinity of large industrial centers, as a result of aerotechnogenic emissions, significant damage to forest ecosystems is observed.

The processes of restoration of the original composition of herbaceous-shrubby stands and lichenoflora in derived coniferous forests typical for green-moss forest types of the taiga zone takes up to 50 years.

The species composition of plant communities in derived deciduous forests is somewhat different from that in native coniferous forests. Changes in forest management methods, the use of clear-cutting and thinning systems and modern aggregate machinery in native forests will help to

reduce or prevent the degradation of the natural habitat of rare species.

At the same time, habitats with maximum species diversity and habitats of rare and most vulnerable species in forest areas that have not been logged due to the low attractiveness for loggers or protection by existing logging regulations have remained virtually unchanged. They fulfill the role of key habitats and habitats of rare, threatened, and endangered plant species.

The challenge is to balance measures to conserve biodiversity with the increased intensity of forest management that may be needed to meet the higher demand for wood, including for renewable energy production.

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Section I

Temperate forests

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Chapter 24

Temperate forest of Asia in the wake of climate change

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1 Introduction

In East Asia, from Nepal to the western Pacific coast, the shift from tropical to subtropical (warm temperate) evergreen forests is more noticeable than it is elsewhere in the tropics. It stretches continuously for nearly 2000 km, from 85°E longitude in east-central Nepal to 122°E in Guangxi, Hainan, and southern Guangdong provinces in southern China. The northern boundary of the Indo-Burmese tropical and East Asian-Chinese subtropical biogeographic provinces, where it contributes to the largest floristic diversity hotspot, lies south of the Qinling Mountains' east-west climatic barrier. The northern edge of the Asian tropics increases in richness eastward (Wu and Wu, 1996; Axelrod et al., 1998; Wu et al., 2011; Qian and Ricklefs, 1999; Zhu, 2013). These ecosystems are essential for controlling the climate locally and globally (Bala et al., 2007). By stabilizing temperatures, reducing the frequency of extreme weather events, and preserving hydrological cycles, the effects of climate change can be lessened. Examples of these processes include carbon sequestration and water cycling (Field and Barros, 2014).

By influencing temperature regimes and precipitation patterns, the intricate relationships between the flora, soil, and atmosphere found in temperate forests regulate regional climates (Christensen et al., 2013). Moreover, the dense canopy cover of these forests inhibits soil erosion, reducing the likelihood of landslides and floods in the surrounding areas. The harvest of timber and nontimber forest products, which support customs and lifestyles, is the main source of revenue for many communities.

In order to increase local and global trade, these forests are exploited for their game animals, edible fungi, and medicinal plants. In addition to cultural sites, recreational hubs, and spiritual retreats, these ecosystems attract tourists and foster a sense of interconnectedness with the natural world (Watson et al., 2018).

Due to variables such as increasing temperatures, shifting precipitation patterns, and a rise in the frequency of extreme weather events, the extent and composition of forest ecosystems are altering (Seneviratne et al., 2016). Endemic species face extinction when their habitats become untenable due to their adaptation to a limited range of temperatures (Thomas et al., 2004). According to Foley et al. (2005), human-induced stressors such as pollution, urbanization, and deforestation further threaten the health and viability of temperate forest ecosystems.

2 Impact of climate change

The resilience of Asia's temperate forests is seriously threatened by changes in temperature and precipitation patterns (Christensen et al., 2013). Between 17% and 30% of the world's land area is made up of semiarid ecosystems, which are distinguished by climatic regimes with low mean annual precipitation ($<650\,mm\,year^{-1}$) and vegetation tailored to dry soil conditions (Poulter et al., 2013).

The hydrology and ecosystem characteristics of Central Asia are greatly influenced by low-regional annual precipitation and the climatic temperature gradient, as evidenced by the taiga and boreal forest vegetation in the north and shrub-desert vegetation in the south. The north's forest productivity may benefit from projected increases in air temperature brought about by climate change, whereas the south will have greater constraints on soil moisture. Hence, the relationships between temperature and precipitation as well as potential gains in water-use efficiency due to higher CO_2 levels point to a complicated response of Inner Asian vegetation dynamics to climate change.

The livelihoods of communities dependent on freshwater supplies are at risk due to these changes, which also increase the vulnerability of mountain ecosystems (Watson et al., 2018). The effects of climate change on temperate forests are exacerbated by an increase in the frequency

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and severity of extreme weather events, such as storms and droughts (Seneviratne et al., 2016). Tree mortality, habitat fragmentation, and heightened vulnerability to pest outbreaks and wildfires are all consequences of these disturbances, which upset the dynamics of the forest (Parmesan and Yohe, 2003).

The increasing frequency, severity, and scope of disturbances are causing forest communities to shift beyond their natural bounds, according to Usbeck et al. (2010) and Weed et al. (2013). Because of the current high rate of environmental change occurring worldwide, it is anticipated that disruptions to forests will intensify. This could pose a challenge to species trying to advance through their evolutionary adaptation processes (Hartmann et al., 2015).

According to Trumbore et al. (2015), there is therefore ambiguity over the future of global forests, the biodiversity they sustain, and the capacity of ecosystem services to sustain human society. Aliens pose a special hazard to native biodiversity and ecological function through their introduction, as noted by Thomas et al. (2004).

3 Vulnerabilities of Asia's temperate forests

Alpine ecosystems, alpine livelihoods, and regional economics will all be impacted by the continued melting of alpine permafrost and the subsequent reduction in the area of frozen ground due to rising air temperatures and increased insolation (Huss et al., 2017).

Thus, it is anticipated that climate change will have a substantial effect on the ecology, geomorphology, and hydrology of the high mountains of Asia. Fossil fuel consumption, deforestation, and other human-caused processes are the main causes of climate change, which raises atmospheric emissions of greenhouse gases (GHGs), primarily carbon dioxide (CO_2) (IPCC, 2014; Scott et al., 2016).

Along with changes to ecosystems and the distribution of flora and fauna, climate change has far-reaching effects, such as an increase in sea level, more frequent and extreme heatwaves, flooding, droughts, and cyclones (IPCC, 2014). Climate change has significant social and economic ramifications, particularly for vulnerable populations in developing countries where they may experience migration, a shortage of food and water, and higher healthcare risks (IPCC, 2014).

Adger et al. (2005) claim that the socioeconomic vulnerability of those who depend on forests exacerbates the problems brought on by climate change. Because they depend on forest resources for their survival, marginalized and indigenous groups are particularly vulnerable to changes in land-use patterns and disruptions in ecosystem services (Watson et al., 2018). Comprehensive strategies that incorporate scientific understanding, common sense, and community-based management practices are required to tackle these complex issues.

4 Adaptation strategies

4.1 Conservation and restoration

Protected zones that cross international borders, such as those that separate China and Nepal, can be created to preserve migratory species and their migration routes (Aryal et al., 2018; Xiong et al., 2022). Initiatives for conservation and restoration are essential to the resilience and health of temperate forests. Planting native species in newly formed forests, protecting existing forests, and rehabilitating damaged regions are some of these tactics. Reforestation projects mitigate the consequences of climate change by increasing biodiversity and sequestering carbon (IPCC, 2022; Xu et al., 2021). Furthermore, it is crucial to safeguard old-growth forests because they retain high biodiversity and store significant amounts of carbon (Smith et al., 2019).

4.2 Enhancing biodiversity resilience

In temperate forests, increasing genetic, species, and ecosystem variety contributes to biodiversity resilience. Given that varied forests are less likely to experience catastrophic losses, planting a variety of tree species can increase resilience to pests, diseases, and extreme weather events (Jones et al., 2018). According to Lindner et al. (2018), fostering biological corridors throughout a landscape can help species migrate and adapt to shifting climatic circumstances.

4.3 Community engagement and livelihood diversification

Climate adaptation and sustainable forest management depend on varied sources of income and community involvement. Encouraging local communities to engage in conservation efforts guarantees that they have a stake in the well-being of forest ecosystems. In order to diversify revenue streams and lessen dependency on deforestation, this can entail training in ecotourism, nontimber forest product harvesting, and sustainable forestry methods (Ford et al., 2020; Hassan and Nhemachena, 2021). According to Liu et al. (2017), involving local communities in the monitoring and management of forests can improve their ability to adapt locally to the effects of climate change.

4.4 Policy and governance interventions

Effective adaptation techniques are made possible by initiatives in policy and governance. It is essential to create and

implement laws that prevent illicit logging and land conversion in forests and to provide incentives for sustainable forest management techniques (Brown et al., 2018).

To enable well-informed decision-making, policies should support research and monitoring initiatives that collect information on the health of forests and the effects of climate change (Keskitalo et al., 2020). Enhancing institutional capabilities and cultivating cross-sectoral partnerships can enhance the execution and efficacy of adaption strategies (IPCC, 2022).

5 Challenges and opportunities in adaptation strategies for Asia's temperate forests

5.1 Challenges climate variability and extremes

Impact on Ecosystems: More frequent and intense storms, heat waves, and droughts can exacerbate soil moisture deficits and increase the rate at which trees die. The disturbance of forest ecosystems caused by these extreme weather events makes conservation and restoration efforts more challenging (Xu et al., 2021).

Resource Allocation: Lack of resources, both financial and human, could make it more difficult to carry out extensive conservation and restoration projects. Obtaining adequate funding and skilled labor is a significant challenge for many Asian locations (Ford et al., 2015).

Pest and Disease Outbreaks: The state of forests is further stressed by the possibility that rising temperatures would lead to more insect and disease outbreaks. Higher temperatures and prolonged droughts are two examples of variables that can drastically lower tree mortality and promote the spread of pests like bark beetles (Anderegg et al., 2015).

5.2 Opportunities

Reforestation and Afforestation: Large-scale afforestation and reforestation projects can improve ecosystem services, give habitat for species, boost carbon sequestration, and rehabilitate degraded areas (Xu et al., 2021).

Ecological Corridors: Ecological corridors that link fragmented habitats can support species migration and adaptation, enhancing the resilience of biodiversity overall (Lindner et al., 2018).

Diversified Planting Strategies: Implementing varied planting tactics that include a range of native species can help strengthen the resilience of forests to climate change. According to Jones et al. (2018), these strategies can reduce the likelihood that pests and diseases will result in a significant loss of forests.

6 Conclusion

The temperate forests of Asia are facing a crisis moment due to climate change. The resilience of these ecosystems is facing previously unheard-of difficulties due to the intensifying effects of warming temperatures, altering precipitation patterns, and an increase in the frequency of extreme weather events. To protect temperate forests and the priceless benefits they offer to civilization, immediate action is required to increase their capacity for adaptation. The sustainability of Asia's temperate forests depends on their ability to adapt to the effects of climate change. We can increase the resilience of these important ecosystems by enlarging protected areas, putting adaptive forest management methods into place, involving local communities, carrying out research and monitoring, and incorporating climate change into policy. To preserve these forests and the numerous advantages they offer, sustained efforts and international cooperation are crucial.

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Chapter 25

Temperate forests at southern South America: Challenges for management and conservation to face climate change

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1 Patagonian forests

The temperate forests of southern South America extend from 36.8° to 55.0° SL in a narrow strip extending more than 2000 km in length along the Patagonian Andes (Dimitri, 1982; Matteucci et al., 2021). Several geological processes and glaciation events generate significant geomorphological differences that influence ecosystem formations and their genetic diversity (Mattera et al., 2020; Soliani et al., 2020; Díaz et al., 2022). The Andean-Patagonian forests constitute a unique vegetation (Donoso, 1994), with different forest types growing from the sea level at Cape Horn to more than 2000 m.a.s.l. in northern Patagonia (Fig. 25.1). These forests are geographically isolated from other subtropical forest formations in the continent, separated by extensive arid and subarid formations (Zeballos et al., 2020). For this reason, it is considered a biogeographic island, with a high number of endemic species (Armesto et al., 1998; Martinuzzi et al., 2023; Rosas et al., 2023), including many unique genera and even families with a purely southern distribution (Speziale et al., 2010). The dominant forest formation is the humid temperate deciduous or evergreen type, which varies in dominant species with altitude, aspect, and latitude (Matteucci et al., 2021). This forest region presents a significant cultural heritage due to its archaeological, historical, and cultural values (Martínez Pastur et al., 2016; Nahuelhual et al., 2021), and 34% of their area is under protection (Brown and Pacheco, 2006), although the degrees of implementation and effectiveness may vary depending on the area (Rosas et al., 2018, 2021).

2 Regional climate characterization

The past and present climate in Patagonia is controlled by the southern westerlies (Mansilla et al., 2016; Álvarez-Barra et al., 2022). Westerly airflow is blocked by the north-south orientation of the Andes mountains, resulting in a sharp west-east precipitation gradient, that determines the forest formations (Iglesias et al., 2014; Viale and Garreaud, 2015; Álvarez-Barra et al., 2022). Contrasting landmass extents at higher latitudes lead to distinctive climates in the northern and southern hemispheres (McCulloch et al., 2020). The dominating feature of the southern mid-high latitudes is the Southern Ocean that allows the prevailing southern westerly winds responsible for driving the circulation of the Southern Ocean (e.g., Antarctic Circumpolar Current) (Lamy et al., 2015). This is an important factor affecting primary net productivity (Bakun et al., 2010) and the global carbon cycle (Lovenduski and Gruber, 2005; Landschützer et al., 2015; McCulloch et al., 2020).

In general, the climate is temperate to cold and humid, with abundant snowfall during autumn and winter, and frost occurs almost year-round with strong winds predominantly from the west. There is a climate gradient associated with topography, e.g., temperature decreases from north to south, and precipitation from west to east. The climate of the region also presents great variability due to latitudinal and altitudinal variations and the local relief heterogeneity (De Fina, 1972), influencing over ecological processes of different forest ecosystem types (Matteucci et al., 2021; Toro-Manríquez et al., 2023). Some differences are evident

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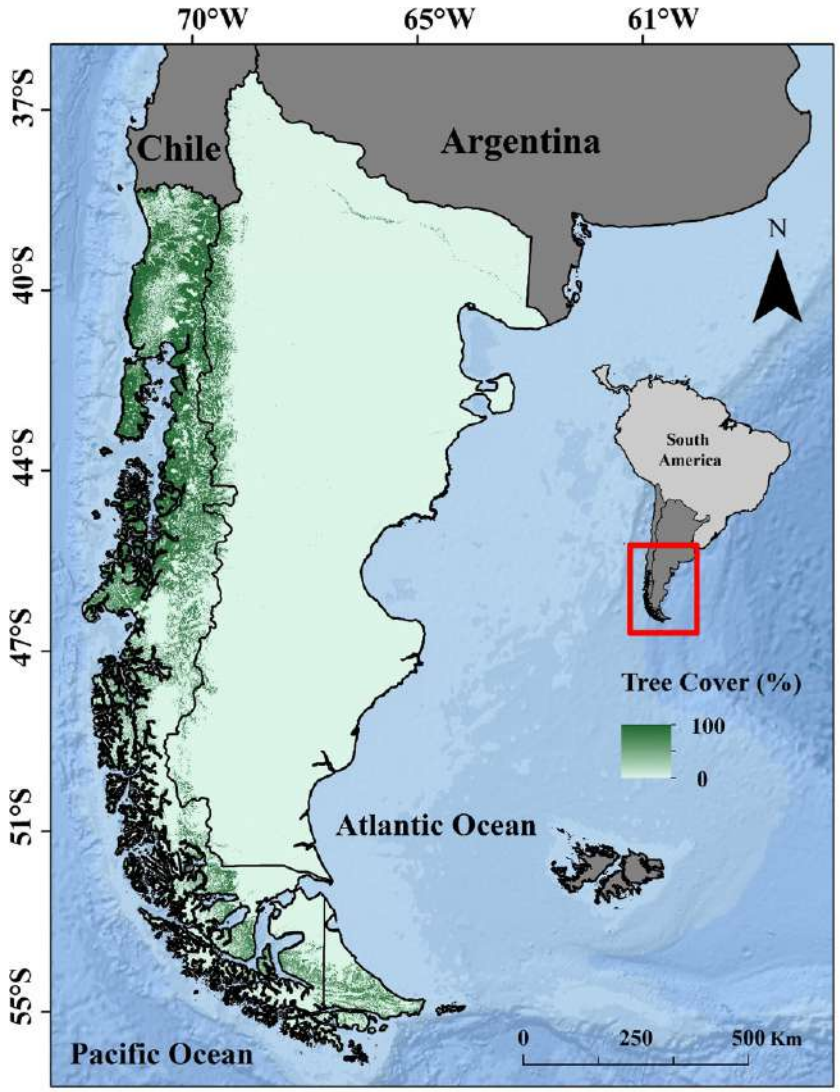

FIG. 25.1 Boundaries of Patagonia region in Chile and Argentina, and the distribution of natural forests. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Modified from www.globalforestwatch.org.)*

between the western (Chile) and eastern sectors (Argentina) of the Patagonian Andes (Jara et al., 2019). While precipitation from the south-west provides abundant moisture to sustain evergreen and rainforest communities on the western side, the Andes Mountains represent an orographic barrier to the eastward advection of westerly flow, generating a marked precipitation reduction on the eastern side (Agosta et al., 2015; Viale et al., 2019).

3 Forest natural dynamics

In addition to climate, natural disturbance regimes and anthropogenic impacts play an important role in shaping landscape patterns and vegetation composition (Johnstone et al., 2016). In forests, these factors create spatial and temporal patches that modify successional pathways (Álvarez-Barra et al., 2022) by affecting regeneration patterns and species composition. Fires are the most important factor in Patagonia (Holz et al., 2012), where the magnitude and frequency of fire events are linked to changes in temperature and moisture during the Holocene (Moreno and Videla, 2016). Another major disturbance factor is related to volcanic activity (e.g., ash deposition), e.g., tree mortality during volcanic events drives different natural dynamic patterns (Swanson et al., 2013, 2016; Álvarez-Barra et al., 2020; Moreno-González et al., 2020).

Other factors that influence vegetation characteristics are associated with the herbivory by native (e.g., *Lama guanicoe*) or exotic mammal species (e.g., sheep, cattle, horses, *Cervus elaphus* and *Castor canadensis*, among others)

(Relva and Veblen, 1998; Martínez Pastur et al., 2021, 2022), insects (Lencinas et al., 2017), exotic plant species invasion (Lencinas et al., 2019; Soler et al., 2019; Iglesias et al., 2022), and logging (Martínez Pastur et al., 2019), diseases (Amoroso et al., 2015), and extreme climate events (Soto-Rogel et al., 2020; Díaz et al., 2022). The latitudinal forest extension with contrasting microclimates and human occupation history determines that the current disturbance factors must be analyzed separately for northern and southern Patagonia (Rovere et al., 2014). In northern areas, the most relevant factors are associated with fires, afforestation of exotic species, urbanization, and herbivory of domestic animals. The southern areas have better conservation status; however, severe harvesting or exotic species (e.g., *C. canadensis*) strongly degrades the forest (Henn et al., 2016). The great spatial heterogeneity at the landscape scale is partly due to topography, which modified local microclimates and the soil evolution and, in consequence, defines the species distribution and vegetation types (Huertas Herrera et al., 2018). Another factor of heterogeneity is the dynamics of patches modeled by the wind or snow avalanches (Rebertus et al., 1997), causing various successional states from small stands (e.g., wind shifts, pests, etc.) to large sectors of the forest (Amoroso and Blazina, 2020; Soto et al., 2022).

4 Management and conservation

The Patagonian forests have been used by first inhabitants for a long time (Orquera et al., 2012), but the large-scale forest use began with European settlements (Gea et al., 2004). In this context, large areas were converted into pasture for livestock, with the consequent loss of ecosystem functions and biodiversity conservation (e.g., Lencinas et al., 2019; Martínez Pastur et al., 2019). Many silvicultural proposals have been used (e.g., clear-cuts, selective cuts, shelterwood cuts) (Peri et al., 2022). These methods are based on economic values (e.g., harvesting costs, yield, growth) (Martínez Pastur et al., 2000), converting primary forests into homogeneous managed secondary forests. Recently, society changed its perception of the forestry proposals and claimed the need for biodiversity conservation in forest management (Gamondes Moyano et al., 2016). These new silvicultural alternatives must consider other ecosystem functions in long-term decision-making (Peri et al., 2016). This new forest management paradigm includes economic and noneconomic parameters to reach an environmental sustainability of the ecosystems under management (Martínez Pastur et al., 2017; Peri et al., 2022).

Variable retention forestry is one alternative that combines economic and ecological values applied in southern Patagonia during the last decades (Martínez Pastur et al., 2019) by considering: (i) the maintenance of protection forests such as riparian forests, edge forests (forest-grassland or peatlands), sloping forests or low-quality sites, (ii) the conservation of primary forest as aggregates (30%), (iii) 10–15 m^2 of basal area as dispersed retention among aggregates, and (iv) the retention of deadwood, stumps, roots, and nontimber remains from the crowns. This proposal has yields comparable to shelterwood cuts (Martínez Pastur et al., 2000), reducing felling and harvesting costs (Martínez Pastur et al., 2007). The efficiency in conserving biodiversity and natural cycles was evaluated in forest insects (Lencinas et al., 2017), lentic insects (Simanonok et al., 2011), spiders (Argañaraz et al., 2020), birds (Lencinas et al., 2018), mosses (Lencinas et al., 2008), fungi (Hewitt et al., 2018), and understory plants (Soler et al., 2019), as well as maintenance of the natural regeneration and nutrient cycles (Martínez Pastur et al., 2019).

5 Climate change and extreme events

Pollen records in Patagonia identified major changes in the influence of the southern westerly winds ranging from multimillennial to centennial timescales over the last 25,000 years (Moreno, 2004; Moreno and Videla, 2016; Jara et al., 2019). Climate changes, in conjunction with other nonclimatic disturbance factors (e.g., fires and volcanism), modified the structure, composition, and dynamics of the regional vegetation, driven by changes since the last glacial termination (near 11,000–18,000 years) (Iglesias et al., 2014; Iglesias and Whitlock, 2014). The magnitude of climate warming in midlatitude regions is greater than that in low-latitude regions, which means that forests in these areas are more likely to be affected by climate change (IPCC, 2022), e.g., annual rainfall has decreased and mean temperature has increased by 0.1°C per decade since 1960 (Villalba et al., 2003; Magrin et al., 2014). There is evidence of extreme climate phenomena such as the hemispheric modes (Southern Annular Mode—SAM and El Niño-Southern Oscillation—ENSO). These events generate significant variability in temperature and precipitation, affecting forest dynamics (Gong and Wang, 1999; Garreaud et al., 2009; Petrie et al., 2016; Tovar et al., 2022), which might lead to modifications in climate resilience thresholds (Peri et al., 2022). These effects might be stronger and variable at extreme latitudes (e.g., Tierra del Fuego), where the climate is affected by both the Atlantic and Pacific Oceans, as well as the Antarctic climate and Andes Mountains (Berman et al., 2013; Garreaud et al., 2013). The combination of these interacting factors suggests that the growth of tree species in the study region may respond to climate change differently compared to trees in other regions of the world (Villalba et al., 2012). Also, within the region, contrasting responses may be

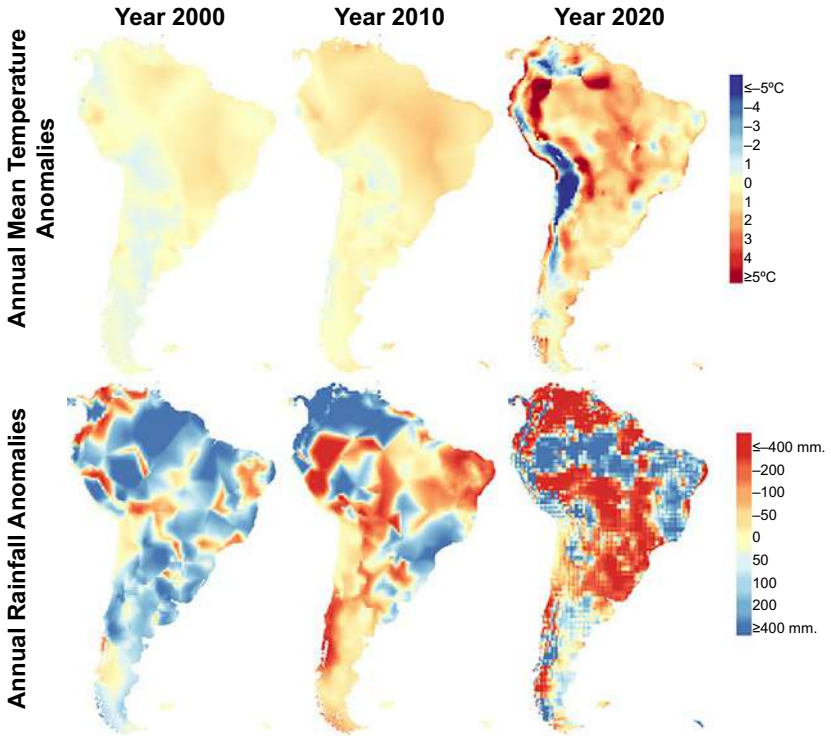

FIG. 25.2 Climate anomalies (annual mean temperature and annual rainfall) for South America during the last decades (years 2000, 2010, and 2020). Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Modified from www.globalclimatemonitor.org.)*

observed as each species responds to different limiting factors (Wright et al., 2004; Tomlinson et al., 2014; Castellano et al., 2019).

Climate change has become more evident in the last decades, e.g., 2000–20 (Fig. 25.2) (Walther et al., 2002; Barros et al., 2015; Camarillo-Naranjo et al., 2018). The anomalies of temperature and rainfall in South America showed extreme values across the territory. However, these trends are not the same for all the countries and regions (Latif and Keenlyside, 2009), e.g., some areas in Patagonia are experiencing cooling while others are warming, and rainfall has decreased toward the north. This is the main factor that influences the development of different ecosystems over time (Hoegh-Guldberg and Bruno, 2010; Guerra et al., 2017) due to human-derived impacts (e.g., greenhouse gases emission) (Gifford et al., 2011). This increased ecosystem vulnerability affects humans (Millar and Stephenson, 2015) and influences over ecosystem functions of natural and managed systems (e.g., Zhao and Running, 2010).

Climate change alters disturbance regimes, both natural and anthropogenic, which modulate forest structure and species composition (Pickett and White, 1985). The climate also indirectly affects the frequency, magnitude, type, and extent of these disturbances (Overpeck et al., 1990; Veblen et al., 2005), and the consequences of these changes are linked to social-ecological impacts (Turner, 2010; Seidl et al., 2011). These changes drive new climate scenarios and lead to new spatial species and community patterns, altering the natural disturbance regimes and ecosystem services. Among the most important changes related to the disturbance regimes are the increases in the frequency and severity of fires (Pechony and Shindell, 2010; Westerling, 2016) and episodic events of insect attacks (Paritsis and Veblen, 2011; Kautz et al., 2017).

6 Impact of climate change on forest processes

The silviculture needs to incorporate resilient proposals that can adapt to the potential climate change scenarios, including extreme events (e.g., greater frequency of droughts, heat waves or extreme frost episodes) (e.g., Suárez et al., 2015; Spinoni et al., 2018). It is necessary to consider the response of trees to these climatic variations, not only due to temperature or precipitation, but also in relation to the interaction with the stand conditions, disturbance types, species competition, or the resource availability (Nitschke et al., 2012). Srur et al. (2008) have reported significant increases in the radial growth of Patagonian forests near the tree line, mainly related to the increase of temperature, while forests growing at lower altitudes presented a negative trend due to the water deficit. These changes affect the reproduction cycles of natural

forests and should be considered in future management planning (Millar et al., 2007; Hallegatte, 2009; Carilla and Grau, 2010). The effects of climate must be considered together with the anthropic effects, which can enhance the negative effects (Crausbay et al., 2017). In *Araucaria araucana* forests of northwest Patagonia, a lack of trees below 100 years of age has been observed in the driest sectors of its distribution due to a combination of the drier local conditions and fruits consumption (Roig et al., 2014). In this context, the influence of climate change must be analyzed at different scales. Here we present two case studies, one at the regional scale and the other at the stand scale, where the results of the impact of these climatic variations are shown and discussed.

6.1 Biodiversity concern due to climate change at landscape level

Biodiversity is strongly influenced by the regional climate and the environment (Rosas et al., 2023). Climate change can induce alterations in phenology (Ma et al., 2013) and temperature patterns in the landscape across the growing seasons (Mina et al., 2021), resulting in variations in the forest stand values over time (e.g., nutrient cycles in soils) (Sardans and Peñuelas, 2013). When the species are not able to adjust to these changes, populations can decline over time due to temporal mismatches (e.g., food and reproduction) (Silveira et al., 2021). High spatial variability (SV) in phenology (proxy: vegetation greenness) and climate (proxy: soil surface temperature) can reduce biodiversity loss by increasing ecosystem resilience when high interannual variability (IAV) occurs (Peaucelle et al., 2019). Thus, high SV can be related to higher food resources, habitat diversity, and climatic refugia, enhancing species survival (Oliver et al., 2010).

To determine the biodiversity conservation concerns under high IAV, we employed the spatiotemporal remotely sensed indices determined by Silveira et al. (2021) for the whole of Argentina based on the Enhanced Vegetation Index (EVI) and land surface temperature (LST) images from Landsat imagery. We used the final products (GRIDS at 250-m resolution), including: (i) interannual variability in phenology based on land surface temperature (IAV-LST), (ii) spatial variability based on land surface temperature (SV-LST), (iii) interannual variability in phenology based on vegetation greenness (IAV-PHE), and (iv) spatial variability based on vegetation greenness (SV-PHE). Silveira et al. (2021) assumed that: (i) High IAV in the seasonality of LST and phenology of vegetation greenness is a threat for biodiversity. This high variability can exceed the rate at which organisms are able to adapt, disrupting the synchrony of ecological interactions, and thus survival, of the species. (ii) High SV in LST and vegetation greenness enhance resilience to high IAV. SV provides a higher variety of resources in close proximity, increasing suitable conditions available during times of climate extremes. Biodiversity conservation concerns due to threats from high interannual climate variability was considered: (i) low concern when IAV is low and SV is high (L), (ii) medium concern when both IAV and SV are high, or both IAV and SV are low (M), and (iii) high concern when IAV is high and SV is low (H). With these layers, we also calculated two indexes to synthetize the biodiversity conservation concerns due to threats from high interannual climate variability: (i) I-LST = integration between IAV-LST and SV-LST, where I-LST = (IAV-LST − SV-LST)/2; and (ii) I-PHE = integration between IAV-PHE and SV-PHE, where I-PHE = (IAV-PHE − SV-PHE)/2. For the maps, we extract the data using the hexagonal binning technique, a spatial methodology which offers the advantage of integrating different pixels (e.g., averaging values for each pixel) within polygonal regions to effectively capture spatial patterns (Battersby et al., 2017; Rosas et al., 2023).

IAV and SV presented the same trend (higher biodiversity conservation concern and higher landscape resilience) (Fig. 25.3). The lowest values of IAV-LST were found in the driest forested areas of Argentina (lowest threat for biodiversity conservation concern), while the biggest values were found in the rainforests (highest threat for biodiversity conservation concern), while Patagonian forests presented intermediate values. The lowest values of SV-LST were found in the central area of Argentina (lower landscape resilience), while the biggest values were found in central and north Patagonian forests (higher landscape resilience). The lowest values of IAV-PHE were found in the rainforests (lowest threat for biodiversity conservation concern), while the biggest values were found in the most austral forests of Argentina (Tierra del Fuego) (highest threat for biodiversity conservation concern). Finally, the lowest values of SV-PHE were found in the driest forest regions and central areas of Argentina (lower landscape resilience), while the biggest values were found across all of Patagonia (higher landscape resilience). These results indicated that the trends for biodiversity conservation concern varied depending on the studied indicator (LST or PHE). The driest forest areas showed different trends compared to rainforests when IAV was considered (LST and PHE). However, the SV of both variables (LST and PHE) showed similar trends, where the driest forest areas presented lower landscape resilience compared to Patagonian southern forests, which presented the maximum values.

Climate change constitutes one of the greatest threats to forest biodiversity and ecosystem services for the next decades (Seidl et al., 2014; Thom et al., 2019). It was found that climate change impacts will be spatially heterogeneous and nonlinear due to differences in climate, edaphic conditions, and competitive interactions (Creutzburg

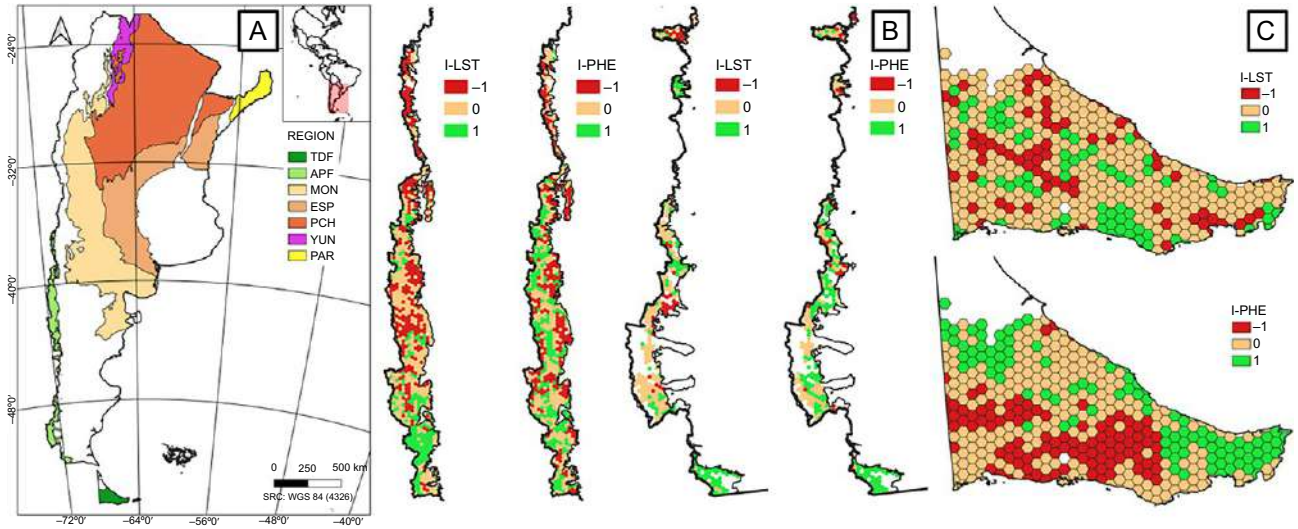

FIG. 25.3 Biodiversity concern due to climate change at the landscape level in Patagonia. (A) forest regions of Argentina (*APF*, Andean Patagonian forests; *ESP*, Espinal forests; *MON*, Monte forests; *PAR*, Paranaense rainforests; *PCH*, Parque Chaqueño forests; *TDF*, Tierra del Fuego forests; *YUN*, Yunga rainforests), (B and C) biodiversity concern integration indexes (i) I-LST = integration between interannual variability in phenology based on land surface temperature and spatial variability based on land surface temperature, and (ii) I-PHE = integration between interannual variability in phenology based on vegetation greenness and spatial variability based on vegetation greenness. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: To the authors.)*

et al., 2017). Defining this spatial heterogeneity can assist land managers and policy makers to prioritize areas for climate adaptation (Seddon et al., 2016). Identifying spatio-temporal patterns of LST and vegetation greenness (PHE) at the landscape level are important for decision-making in forest management and biodiversity conservation (Silveira et al., 2021), e.g., identification of areas to implement multiobjective adaptive management (e.g., land-sharing strategies) or prioritizing areas for protection efforts (e.g., creation of new natural reserves) (Peri et al., 2022; Rosas et al., 2021, 2023). Forests with high seasonality of temperature or IAV in the phenology may require management actions that enhance SV (LST and PHE) to enhance resilience to avoid biodiversity loss from high IAV (Silveira et al., 2021). In contrast, areas with high SV are more adequate for protection because species can persist if IAV increases due to climate change (Martínez Pastur et al., 2019; Ye et al., 2023).

6.2 Climate change influence over the reproduction processes in managed and unmanaged forests at the stand level

The climate variations worldwide affect forest dynamics making the processes of seeding and natural regeneration essential, for maintaining the structure of forests over time (Peri et al., 2016; LaMontagne et al., 2020), mainly in the context of climate change (IPCC, 2022). Recent research has emphasized the significance of woodland effects on regeneration at a landscape level (Pereira et al., 2013), where these effects include regulating species richness and abundance as influencing seedling growth and survival over time. Filotas et al. (2014) characterized forest ecosystems as dynamic systems that continuously adapt to environmental conditions by impacting their resilience and adaptability. These attributes are key objectives in forest management practices. Regeneration is recognized as a factor contributing to forest resilience (Rist and Moen, 2013) as it greatly influences disturbance recovery processes (Albrich et al., 2020) leading to different forest types or nonforest states (Johnstone et al., 2016).

During the 20th century, the main silvicultural practices in *Nothofagus pumilio* forests of Tierra del Fuego were selective and clear-cuttings (Mutarelli and Orfila, 1973), or shelterwood cuts (Martínez Pastur et al., 2000). Recently, variable retention harvesting became a new silvicultural alternative to achieve multiple uses of forest management (Martínez Pastur et al., 2019). This practice combines economic and conservation objectives, where a portion of the structural and compositional diversity of the original primary forest is maintained in the harvested stands, with the objective of retaining it for more than a typical rotation length period (Gustafsson et al., 2012). In harvested areas through variable retention harvesting, the different retention types generate different microenvironments that modify the microclimate (solar radiation and water availability) and soil properties, playing a decisive role in seedlings germination, growth, reproduction, and mortality of

plant species. Variable retention harvesting combines aggregated and dispersed retention and greatly influences seed production (SP) (Rodríguez-Souilla et al., 2023a), recruitment, survival, and growth of the future plants (Torres et al., 2015; Rodríguez-Souilla et al., 2023b), where the first years after harvesting are critical for recruitment to recover the impacted forest structure. This tree species abundantly regenerates with seeds after natural and human disturbances (e.g., harvesting), following natural dynamics based on gaps formation (Alan and Veblen, 1993). Therefore, seeding is also an essential process to guarantee forest continuity (Rosenfeld et al., 2006) and conservation of the natural genetic resources (Premoli and Kitzberger, 2005). This regeneration process is mainly regulated by biological and environmental factors (e.g., topography, soil conditions, and changes in the forest structure) (Hacket-Pain et al., 2019; Koyejo et al., 2020), and also is influenced by the regional climate during primordia development and flowering (McKee and Richards, 1996; Martínez Pastur et al., 2019). In this context, we describe two of these ecological processes: (i) annual SP related to the activity of climatic modes over a 17-year period, and (ii) effect of climate on annual height growth of the regeneration.

SP (million ha^{-1} $year^{-1}$) was measured during 2006–22 in three types of managed stand retention (aggregated retention—AR, dispersed retention with aggregate protection—DRI, dispersed retention—DR), as well as in primary unmanaged forests (PF). Additionally, the annual height of natural regeneration was also measured at each plant. The annual growth rate (cm $year^{-1}$) was derived from these measurements, with a further distinction made among plants in age class categories (1: 1–5 years old, 2: 6–10 years old, 3: 11–15 years old). The studied postharvest period was characterized for the occurrence of hemispheric modes (ENSO and SAM in their joint response) on the saplings height growth (September to January). Values of ENSO and SAM were obtained from public data (https://www.weather.gov/fwd/indices and https://www.stateoftheocean.osmc.noaa.gov/atm/sam.php), and were classified according to its prevalence of positive (+) or negative (−) ENSO/SAM values (ENSO−/SAM−, ENSO−/SAM+, ENSO+/SAM−, ENSO+/SAM−).

Linked to SP, the results indicate that the magnitude of this production was influenced by the combined occurrence of both events, which potentially affects the silvicultural treatments. This differential influence can be explained by the physiological responses of the remaining trees. PF exhibited a higher seed quantity under the ENSO+/SAM+ combination, followed by ENSO−/SAM+. However, the latter combination resulted in the lowest SP in AR, DRI, and DR, indicating differences in behavior between harvested and unharvested stands, and can be associated with a modification in the SP dynamics of the studied stands (Fig. 25.4).

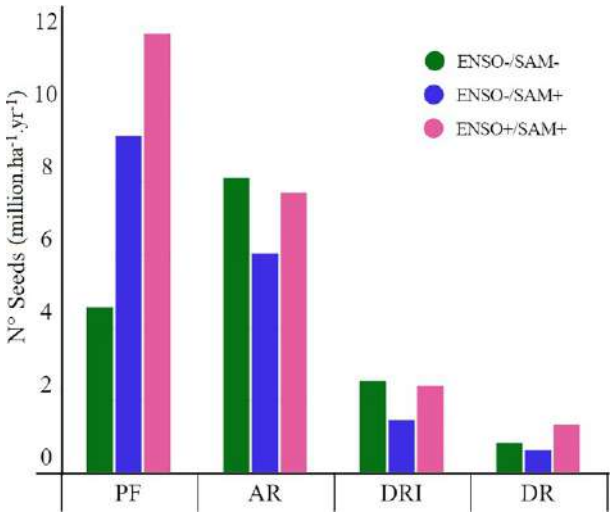

FIG. 25.4 Annual seed production (million ha^{-1} $year^{-1}$) for different forest treatments (*AR*, aggregated retention; *DR*, dispersed retention; *DRI*, dispersed retention with aggregate protection; *PF*, primary unmanaged forests) considering three climate event (ENSO or SAM) combinations. *(Credit: To the authors.)*

In the same way, the height growth of the natural regeneration presented different growth rates among treatments (DR > DRI > AR > PF) and among age classes (3 > 2 > 1 in harvested stands), where differences are related to the combined occurrence of ENSO and SAM phenomena, as was described for SP (Fig. 25.5). PF and AR showed similar behaviors in response to the combination of events, with ENSO−/SAM− being the most beneficial for age categories 1 and 3, showing different responses for the age of the plants. Besides, DRI and DR showed higher annual growth rates with the occurrence of ENSO+/SAM+ for all age categories except from DR at category three where ENSO−/SAM− was the highest one. For all age categories, the least beneficial situation was ENSO−/SAM. Finally, ENSO+/SAM− did not occur during the studied period.

Nothofagus forests have been extensively studied in relation to climate (Aschero et al., 2022; Brand et al., 2022). However, few works explored the connection between the climatic context and the dynamics of native forests in Patagonia (Rodríguez-Souilla et al., 2023b). When analyzing climate change, variations in the percentage of occurrence of hemispheric modes between periods (1950–2006 and 2007–22) were registered, denoting an increase in the occurrence of ENSO+/SAM+ for both seasons to the detriment of other combinations, evidence of increment in temperatures. In the context of climate change with periodic increases in temperature and variations in annual rainfall (IPCC, 2022) and the occurrence of climatic extreme events, it is likely that annual SP could be altered (Monks et al., 2016). Differences on SP linked to climate variations could be explained by differences in the allocation of resources at different times

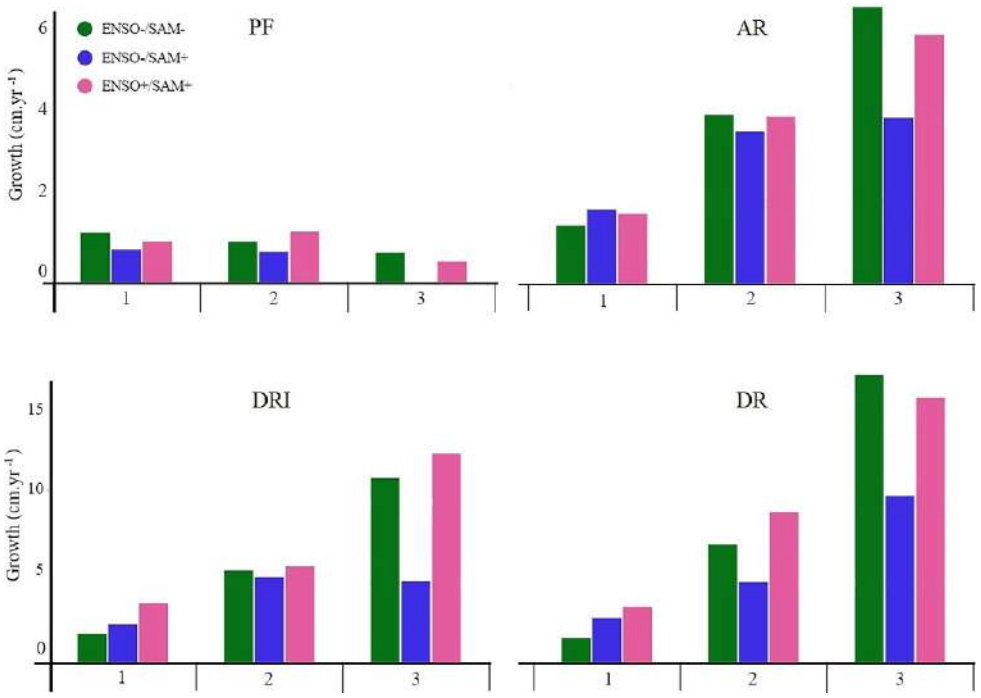

FIG. 25.5 Natural regeneration height growth (cm year^{-1}) for different forest treatments (*AR*, aggregated retention; *DR*, dispersed retention; *DRI*, dispersed retention with aggregate protection; *PF*, primary unmanaged forests) during three climate event combinations linked with the age class category of the regeneration (1 = 1–5 years-old, 2 = 6–10 years-old, 3 = 11–15 years-old). *(Credit: To the authors.)*

of the year (e.g., carbon and other nutrients), as well as differences in the growing season length and modification of tree phenological stages (Torres et al., 2015; Bogdziewicz et al., 2019). Warm springs may lead to more uniform and abundant flowering and pollination success (Allen and Platt, 1990), and therefore higher SP, associated with it to the critical phase of floral primordial development (Martínez Pastur et al., 2019). Also, natural regeneration is susceptible to climatic variations (Torres et al., 2015), depending on their age, where years of low temperatures (ENSO−/SAM+) limited seedlings development and increased dry apices due to frost mainly in the spring season (Suárez et al., 2015). However, growth values are not critical in terms of height growth, being below the thresholds of points of no return for mortality. This is consistent with Srur et al. (2018) who reported that higher temperatures may generate greater establishment and growth of seedlings in tree lines. Practitioners need to take into account the occurrence of extreme climatic events (extreme ENSO or SAM), as well as the durability or frequency of these processes on the ecological processes. It potentially might impact the establishment and mortality of forest stands, by determining the climatic threshold of native forest development.

We conclude that the extreme events significantly affected natural processes related to seeding and regeneration, by altering the resilience capacity of the unmanaged and managed stands. These results at the stand level support the idea that the new forest management proposals must be based on potential climate variations, including the influence of climate events (SAM and ENSO), and not only the average climate data of the last decades. Some of these unusual events can lead to failures in the success of the management proposals, generating negative synergies that affect the sustainability of ecosystem planning (Peri et al., 2022).

7 Challenges for management and conservation to face climate change

Forest cover and composition are naturally modified over time, depending on climatic barriers (Zak et al., 2008), e.g., documented advance of *Austrocedrus chilensis* in northern Patagonia (Veblen and Lorenz, 1988; Veblen et al., 2005), or the recent tree line advance of *N. pumilio* (Ivancich et al., 2012; Srur et al., 2008, 2018). The forest dynamics and growth are also affected by these changes (Amoroso et al., 2015; Rodríguez-Catón et al., 2016), and lead to modifications in the assemblage of species in the different forest types (Wolters et al., 2000) affecting primary and secondary productivity (e.g., bird communities). Besides, we have already discussed the incidence of ENSO phenomena, where the exceptional warming of the Pacific Ocean produces greater evaporation, leading to heavy winter precipitation across the Andes Mountains, but also the increase of air temperatures in northern

Patagonia (Aceituno, 1988; Kiladis and Diaz, 1989). These warming processes increase drought events in Patagonian forests, e.g., in *A. araucana* forests (Roig et al., 2014). For example, when comparing tree growth during El Niño events, it was observed that they are more sensitive to this variable, especially during the months before sprouting (Hadad et al., 2014). As previously mentioned, these environmental changes also alter the natural cycles by modifying their ecosystem functions, e.g., from decomposition rates, immobilization and/or release of nutrients, changes in water cycles (Hopkins et al., 2012). These alterations had an economic impact over the forests under management, e.g., losses associated with insect attacks (Volney and Fleming, 2000; Allen et al., 2010). These attacks multiplied as a result of variations in the climate, modifying the habitat of these species, increasing the number of annual reproductions, and even potential evolutionary patterns (Ryan et al., 2018). Lastly, the indirect effects of climate change, such as the increasing fire events (Allen et al., 2010; Westerling, 2016), have been well documented for Patagonia during the last decades (Kitzberger et al., 1997; Veblen et al., 2005; Holz et al., 2012; Mundo et al., 2017).

To face the uncertainty and evidence of modifications due to climate change, it is essential to expand the studies about ecology, forest management, and conservation, increasing the studied environmental gradients in order to define accurate baselines for decision-making, e.g., to define mitigation and adaptation schemes to face climate change. These studies should include temporal analyses of forest dynamics across different vegetation types, as it constitutes a useful indicator to establish new conservation strategies at the landscape level. In the literature, many recommendations on adaptations for agriculture and livestock exist (Magrin et al., 2005; Gauly et al., 2013), including biodiversity conservation (Wolters et al., 2000; Walther et al., 2002; Ryan et al., 2018). However, the literature on silvicultural practices for native forests to face climate change is much less abundant (Allen et al., 2010; Keenan, 2015).

In this context, climate change brings new challenges to define new conservation strategies, due to the survival thresholds of several species having changed, e.g., the design of current natural reserves no longer covers all the species for which they were designed (Swetnam et al., 1999; Hallegatte, 2009; Gifford et al., 2011). Numerous studies propose different alternatives to improve these conservation strategies to face climate change (Heller and Zavaleta, 2009), including: (i) designing biologically complex reserves to better capture the unpredictable changes over time (Bartlein et al., 1997), (ii) increasing the size of the main reserves to protect greater diversity of environments across climatic and environmental gradients (Hartig et al., 1997; Millar et al., 2007), (iii) increasing connectivity (e.g., corridors) (Halpin, 1997), (iv) conserving species in more than one reserve (Halpin, 1997; Millar et al., 2007), (v) mitigating other impacts that may generate negative synergies with the climate (e.g., invasive species) (Chornesky et al., 2005), (vi) relocating species with high conservation values among the different reserves (Hannah et al., 2007), (vii) implementing less intensive management practices in areas of high conservation value (Franklin et al., 1992), and (viii) implementing broad-spectrum monitoring programs to promote adaptive management (Millar et al., 2007), increasing the collaborations between different institutions (Grumbine, 1991).

There are also different potential strategies in the silvicultural management to face climate change (Millar et al., 2007; Rist and Moen, 2013). These are based on building resistance in stands under management to face potential environmental changes that may occur, promoting greater resilience by changing thinning or harvesting practices and allowing stands to continue different dynamics, so that they can respond favorably to these changes (e.g., postdisturbance dynamics must follow different pathways). Millar et al. (2007) suggest several strategies: (a) assist transitions to face climate change, favoring the directions of change to support greater biodiversity in the managed forests, e.g., promote the migration of species. These objectives can be achieved by modifying harvesting, altering thinning objectives, or favoring different specific forest species over time (Halpin, 1997). (b) Increase the duplication of environments in the landscape in sensitive areas to climate change. (c) Introduce germplasm management for local populations, modifying the assumptions that environments and climate do not change over time. Within the new paradigms, changes in local genomes that migrate according to the necessary adaptations of the species involved should be favored. (d) Improve ecosystem connections and avoid fragmentation, where the key to success is migration of species due to climate change. It is necessary for more flexible forestry proposals to promote connectivity, e.g., variable retention systems generate greater connectivity in managed forests than traditional harvesting proposals (e.g., Lencinas et al., 2017; Martínez Pastur et al., 2019). (e) Include the conservation of high conservation value environments, not only those that host greater richness of species but also those environments that present exceptional conditions to act as a refuge for specific species in the long term (Huntley and Webb, 1989), and must be identified to ensure their conservation within the managed landscape (Peri et al., 2022).

8 Final remarks

Climate change is a phenomenon that presents significant variations at the planetary, regional, and local scales, but

with different magnitudes and directions. The existence of tools based on remote sensing at the landscape level may assist decision-making processes in management and conservation to face climate change. Using these tools, it is possible to identify differential spatial trends in temperature and precipitation during the last decades affecting the ecosystems and net primary productivity. Nowadays, it is possible to quantify the changes that have occurred in natural ecosystems on a small scale, in relation to the current native forest cover, and their interaction with the main socioeconomic factors (e.g., fires events). In this context, it is necessary to rethink about the forestry practices that must be adapted to each region and that must be flexible enough to adapt to a wide range of possible climate and anthropogenic scenarios. The recommendations for the development of new management and conservation strategies must be designed at a regional, provincial, or ranch scale, in relation to the involved ecosystems (e.g., assemblage of species or the main uses by the society). Monitoring must be the base for the new management paradigms, analyzing the change trends and relationships with climate using remote sensor tools. Climate change must be incorporated in the planning for the use and conservation of our native temperate forests of Argentina.

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Chapter 26

Europe temperate forests

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1 Introduction

The most ancient temperate forests, characterized by minimal influence from the last glacial period, are situated in southern Europe. Historically, Europe was blanketed by forests; however, the continent's colonization precipitated widespread deforestation to obtain timber for myriad purposes including construction, energy generation, heating, furniture crafting, and shipbuilding, among others (CEJA, 2002).

In the initial stages of the Roman conquest, for instance, France boasted approximately 80% of its territory covered by temperate forests, dwindling to 15% by 1800 (currently exceeding 30%). Italy lays claim to just about 1% of the total forested area untouched, while Portugal has 0.67%, and Spain, along with Malta (the four countries predominantly Mediterranean forests), and Germany, register 0% (temperate forest). Conversely, Liechtenstein, Bulgaria, Georgia, and Montenegro emerge as countries (excluding Russia) with the highest percentages of intact temperate, mountain, and Mediterranean forest areas, with figures such as 21.7%, 18.1%, 17.7%, and 11.1%, respectively. These pristine primary forests owe their preservation to feudal lords who utilized them for hunting, often making access challenging (CEJA, 2002).

As the availability of wood dwindled, with less than 3% of native forests remaining today, efforts were directed toward both conserving Europe's remaining original forests and implementing strategic afforestation initiatives to meet increasing demand. Notably, between 1750 and 1850, the European forest area contracted by approximately 190,000 km^2, subsequently expanding by over 386,000 km^2 since the latter date (>10% pre-Industrial Revolution). In the past 34 years alone, data from the European Environment Agency (EEA) indicate a net increase of at least 170,000 km^2 in forested areas across Europe, equivalent to the landmass of Uruguay.

Europe has witnessed a notable increase in forest coverage, now comprising over 35% of the continent, with a breakdown of 46% conifers, 37% broadleaves, and 17% mixed forests (Forest Europe, 2020; European Commission, 2020). A prominent illustration is the Landes Forest in France, the largest human-caused forest in Western Europe spanning 14,000 km^2, established in the last century primarily for timber and pine resin production (CEJA, 2002). However, the connectivity of Europe's forests remains regionally constrained (Wolseley et al., 2017), with certain areas facing economic conflicts (Blicharska et al., 2020). Inadequate connectivity adversely affects biodiversity, notably epiphytic lichens, due to microclimate alterations, edge effects (Kiebacher et al., 2017), and hindered gene flow and foraging activities by fauna. Effective biodiversity conservation necessitates robust connectivity and landscape continuity, fostering diverse environmental conditions conducive to various biotic communities (Świerkosz et al., 2017).

Furthermore, forests play pivotal roles in society, contributing to human well-being and recreation, while also serving as a significant economic resource by providing timber and other ecologically valuable ecosystem goods and services such as carbon sequestration, nutrient cycling, and natural hazard mitigation (e.g., flood control) (Ammer et al., 2018; Forest Europe, 2020). Despite the acknowledged importance of primary forests (Bruun and Heilmann-Clausen, 2021; Schall et al., 2021), their limited extent underscores the critical significance of close-to-nature forest management and forest restoration in safeguarding Europe's terrestrial biodiversity (Bauhus et al., 2013) (Fig. 26.1).

In addition to deliberate tree planting efforts, natural forest expansion has notably occurred, particularly over the past 24 years, attributed to several factors: (i) a reduction in wood consumption facilitated by the adoption of electricity and fossil fuels; (ii) rural-to-urban migration, particularly among youth, resulting in the abandonment of arable land (e.g., Finland, Boreal forest) and deserted villages (e.g., Portugal, Mediterranean forest); and (iii) the proliferation of green spaces, including forests, in and around urban areas, especially smaller cities, catering to recreational

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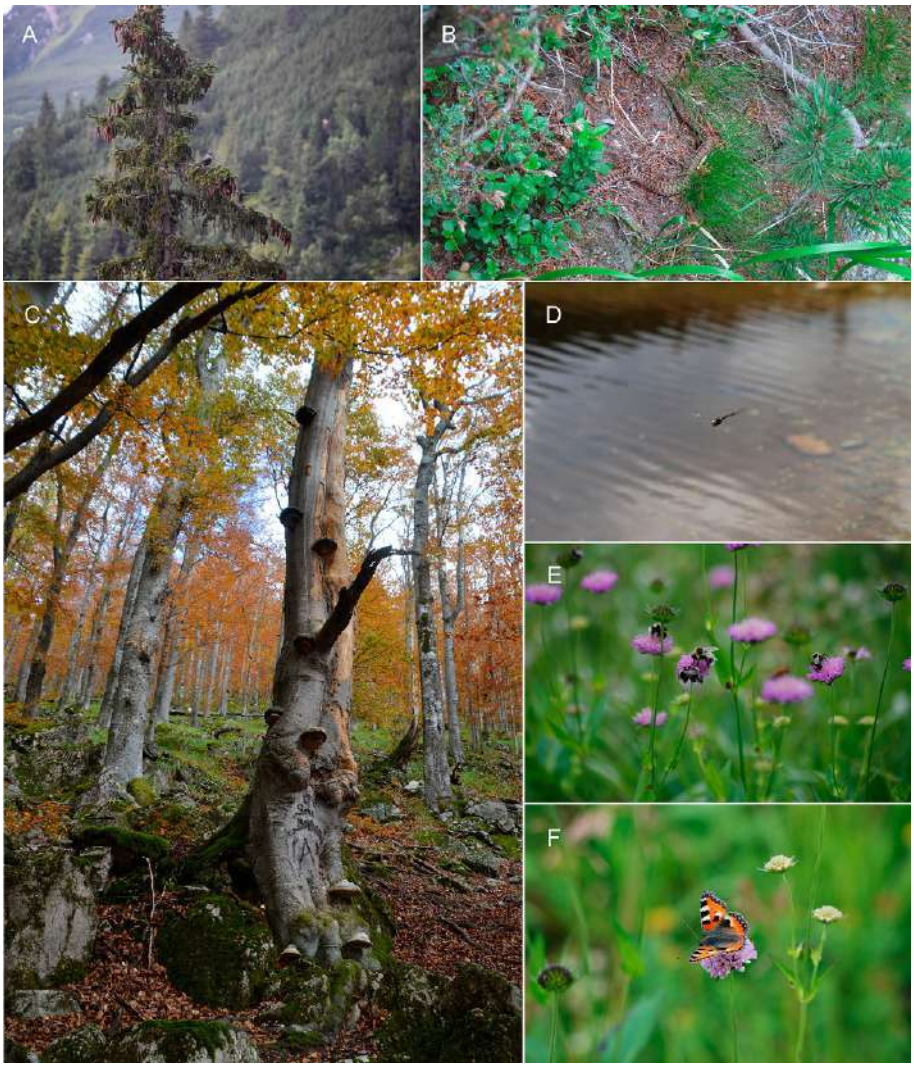

FIG. 26.1 (A) Bird perched on a conifer tree; (B) *Vipera berus*; (C) mushrooms, bryophytes, and lichens in the forest; (D) Odonata; (E) Apidae; and (F) *Aglais urticae*. Retezat Mountains (Southern Carpathians), Romania. *(Photo credits: Dan Cogalniceanu.)*

activities such as hiking and sports for residents and tourists (e.g., Germany, Temperate forest). The status of forests, biodiversity, and ecosystem services are intricately interconnected, with biodiversity levels reliant on the integrity, health, and vitality of forests. Conversely, declines in forest biodiversity precipitate diminished productivity and sustainability (Marín et al., 2021).

The escalating global rate of species extinction and human-induced impacts on biodiversity underscore the imperative to conserve, restore, and sustainably manage ecosystems (Oettel and Lapin, 2021; Ette et al., 2023). In this chapter, a comprehensive exploration will be undertaken to encompass biodiversity dynamics, anthropogenic influences and their repercussions, operational methodologies, monitoring tools, and the ramifications of global warming within Europe's temperate forests.

2 Biodiversity

The biodiversity of Central European temperate forests, including epiphytic lichens (Vicol and Sahlean, 2023), has been significantly shaped by human activities since the postglacial era when forests first formed (Norton, 1996). Over time, these activities have led to the extinction of numerous forest species and animals, such as *Bos primigenius* (Grove, 2002). The expansion of forested areas, particularly with the proliferation of pine forests, has emerged as a significant concern for biodiversity in Europe, with 42% of native tree species facing extinction out of a total of 454 species. Endemic species are even more threatened, with 58% at risk. This alarming situation is not solely attributed to the planting of pine trees; invasive species and the impacts of climate change have also played significant roles (Langmaier and Lapin, 2020).

Moreover, the substantial human population across Europe, along with its urban centers, has fostered the hybridization of wild (e.g., *Felis silvestris* and *Canis lupus*) and domesticated (e.g., *Felis catus* and *Canis familiaris*) species (Salvatori et al., 2020; Matthias et al., 2022). Hybridization events between gray wolves and domestic dogs are widespread across the continent, facilitated by wolves venturing into human-inhabited landscapes where free-living dogs are abundant. Additionally, areas with high wolf mortality due to human activities can disrupt pack structures, increasing the likelihood of encounters and interspecific mating. This phenomenon poses a significant threat to wildlife conservation, as human-induced hybridization, often linked to species introduction and habitat degradation, can create reproductive opportunities between species that would not typically interbreed in natural settings (Salvatori et al., 2020; Matthias et al., 2022).

The decline in European biodiversity has intensified over the past two centuries due to human activities, significantly altering the landscapes of Central Europe (Eckelt et al., 2018). This reality has prompted the development of conservation strategies, exemplified by the EU Biodiversity Strategy for 2030, implemented by the European Commission (2020)).

Romanian forests, characterized by diverse soils, climates, and rich postglacial vegetation history, highly represent species-rich and ecologically diverse ecosystems (Reif et al., 2022) (Fig. 26.2). Recent inventory data, including satellite imagery and historical aerial photos, reveal that the Fagaras Mountains in Romania harbor between 27,000 and 61,846 ha of primary forests and old-growth forests within the total forest area of 274,000 ha (Schickhofer and Schwarz, 2019). Consequently, the Fagaras Mountains host the largest cluster of temperate primary forests. While smaller in scale, the Carpathian Mountains also contain temperate primary forests (Kameniar et al., 2023). Additionally, other regions in Central and Eastern Europe contribute to this forest type, including Uholka-Shyrokyi Luh (Ukraine, 9800 ha), Valea Cernei (Romania, 5153 ha), and Izvoarele Nerei (Romania, 4677 ha). Isolated pockets of temperate primary forests persist in the Balkans (e.g., Stara Planina in Bulgaria), Lumi and Gashit (Albania), Biogradska Gora (Montenegro), and Perucica (Bosnia and Herzegovina) (Kameniar et al., 2023).

Epiphytic lichens exhibit varying richness across different forest types. In Central Europe, mixed forests and forested steppes have the lowest richness, while broadleaf forests in the mountainous Carpathian and Caledonian regions exhibit the highest richness (Vicol and Sahlean, 2023). Notably, unmanaged forests support greater epiphytic lichen richness compared to managed forests due to the adverse effects of forest management practices (Hämäläinen et al., 2021; Vicol and Sahlean, 2023). Consequently, the mountainous forests of Europe, with less anthropogenic impact, play a crucial role in maintaining high epiphytic lichen diversity. These forests warrant

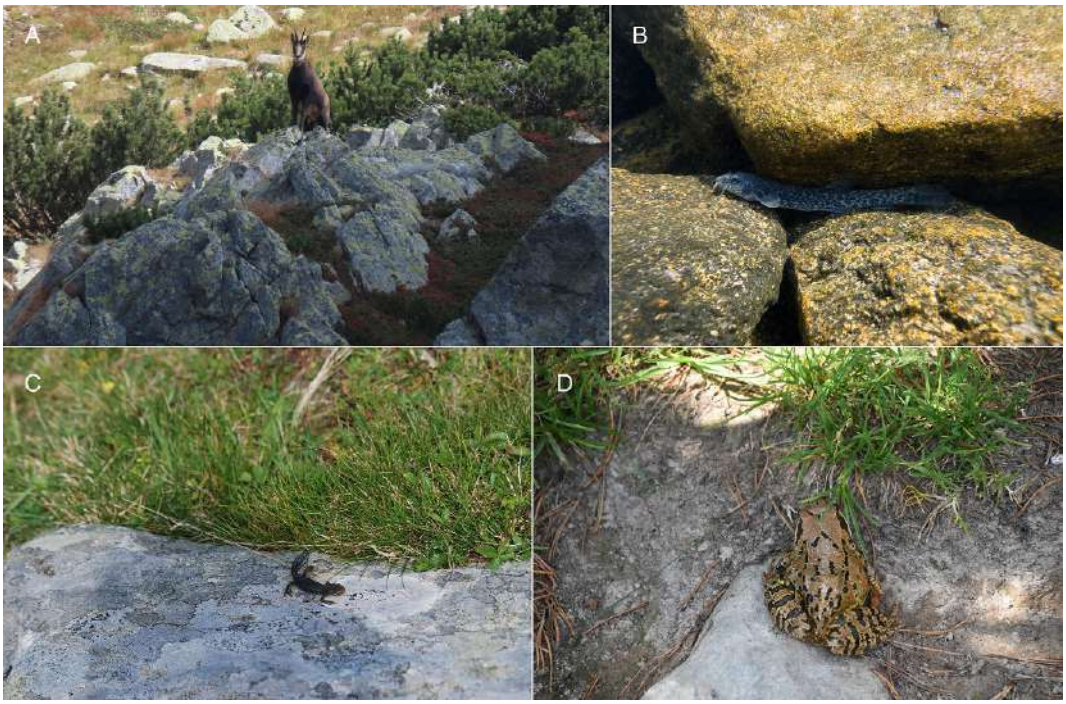

FIG. 26.2 (A) *Rupicapra rupicapra* (Bovidae); (B) *Noemachilus barbatulus* (Cobitidae); (C) *Ichthyosaura alpestris* (Salamandridae); and (D) *Rana temporaria* (Ranidae). Retezat Mountains (Southern Carpathians), Romania. *(Photo credits: Ovidiu Dragan (A–C) and Dan Cogalniceanu (D).)*

protection and conservation due to their biological and ecological significance (Vicol and Sahlean, 2023).

Furthermore, maintaining connectivity between forests through ecological corridors and minimizing edge effects are essential for conserving epiphytic lichens and reducing the risk of species extinction (Lindenmayer et al., 2000; Niculae et al., 2017; Palmero-Iniesta et al., 2020).

Approximately 150 types of natural forest ecosystems have been described in Romania and categorized into seven zonal forest formations: (i) forest steppes and dry oak forests; (ii) forests with *Carpinus orientalis*; (iii) forests with *Quercus* spp. and *Carpinus betulus*; (iv) forests of *Fagus sylvatica* and mixed forests of *F. sylvatica*; (v) mixed mountain forests of *Abies alba* and *Picea abies*; (vi) fir forests; and (vii) subalpine scrubland with *Pinus mugo* subsp. *mugo* (Reif et al., 2022). In extreme locations, azonal forests occur, dominated by (i) *Pinus nigra* subsp. *banatica*; (ii) *Pinus sylvestris*; (iii) *Larix decidua* subsp. *carpatica* and *Pinus cembra*; and (iv) valuable broad-leaved species, including *Acer* spp., *Fraxinus* spp., *Ulmus* spp., *Tilia spec* (Reif et al., 2022). Along watercourses with periodic flooding, riparian forests and shrublands occur, dominated by (i) *Alnus glutinosa*; (ii) *A. incana*; (iii) pioneer woodland of *Myricaria germanica*; (iv) pioneer forest with *Populus* spp. and *Salix* spp.; and (v) noble wood riparian forest with oaks, *Ulmus* spp., and *Fraxinus* spp. (Reif et al., 2022).

Poland's Białowieża National Park (BNP) stands as the last extensive vestige of primeval temperate forest across the European plain. Recognized for its significance, this forest is included on the World Heritage List of the United Nations Educational, Scientific, and Cultural Organization (UNESCO), marking it as the first cross-border site in Europe, spanning both Polish and Belarusian territories (Wojnicka-Półtorak et al., 2023). Within this forest, the natural formation of tree cavities occurs, providing essential nesting sites for various bird species, particularly in areas relatively untouched by human interference (Broughton et al., 2022). Many songbirds native to European forests rely on these tree cavities for nesting, which are not typically created through tree excavation. Instead, they are formed through the natural processes of fungal decomposition and compartmentalization following tree damage, such as the loss of branches or stem breakage (Broughton et al., 2022).

The density and distribution of wild ungulate species have been on the rise across European landscapes since the latter half of the 20th century. This trend can be attributed to various factors, such as the decline in natural predator populations, hunters seeking higher game densities (e.g., traveling to Africa), hunting regulations, land abandonment, and deliberate reintroduction efforts. As these ungulate populations continue to grow in Europe, they bring about significant consequences for forest plant communities.

Bark removal by large herbivores (e.g., *Bison bonasus* and *Cervus elaphus*) could be another important, long-forgotten mechanism for initiating the cavities, which are paramount for these birds (Broughton et al., 2022). The reintroduction of such animals into European forests, where they have become extinct, could aid in the restoration of the abundance and diversity of animals living in tree cavities (Broughton et al., 2022). Future research is needed to investigate the proposal of large herbivores being important cavity-formation agents in trees that could increase biodiversity (Broughton et al., 2022). There is already interest in the reintroduction of large native mammals and the restoration of natural processes and ecological complexity, as addressed within the framework of rewilding (Perino et al., 2019).

The reintroduction of European bison in areas like Damerower Werder (Germany), the Rothaar Mountains (Germany), and the Kraansvlak project (Netherlands) has led to a fascinating phenomenon: frequent debarking of forest trees (Schröder et al., 2019). These impressive animals, once extinct in certain regions, are now contributing to the shaping of forest ecosystems. Similarly, the reintroduction of Red Deer in Knepp Wildland (England) has also resulted in debarking and early cavity formation, akin to what is observed in Białowieża National Park (BNP) (Broughton et al., 2022). Ungulate Pressure and Understory Diversity: Chevaux et al. (2022) emphasize the importance of vertical stand structure and ungulate pressure in shaping the diversity of understory plants. However, careful management of hunting practices is crucial. Cooperation between foresters and hunters is essential to balance ungulate attributes and pressure, ultimately benefiting understory diversity and ecosystem functions in managed forests.

To achieve a rich mosaic of landscapes at a larger scale, a variety of active management tools are required to protect biodiversity, along with the designation of unmanaged areas with minimal human intervention (Kjucukov et al., 2022). Changes in tree species composition and structural forms are crucial. For beech and oak forests, a set of indicator species host different communities, highlighting that the presence of diverse forest types at the landscape scale additionally benefits overall biodiversity conservation (Leidinger et al., 2020). The diversity of commercial European forests is relatively low due to canopy structures tending to be species-poor or dominated by a single species (Vild et al., 2024). Consequently, there is a general lack of microhabitats and large deadwood pieces, while the understory may harbor hundreds of different herbaceous species (Vild et al., 2024). Moreover, entire habitat types are threatened with extinction, underscoring the need for regeneration and aging of open forests with the presence of old trees (e.g., seed bank) (Kjucukov et al., 2022).

To counter this trend, large-scale nonintervention within protected areas becomes essential to uphold the existence of

old-growth forests (Kjucukov et al., 2022). Simultaneously, urgent efforts are needed to restore open forests, a task that requires—not exclusively—active human intervention. These dual actions are critical for diversifying the landscape, achieved by synergistically combining segregative and integrative forest management tools aimed at bolstering biodiversity (Hrivnák et al., 2022; Kjucukov et al., 2022).

Long-term investigations into plant dynamics within temperate forest communities have revealed trends of biotic homogenization and an increase in nutrient-demanding species, albeit with localized variations. While mesophilic or nutrient-rich sites exhibit minor compositional changes, nutrient-poor sites experience more pronounced shifts. Therefore, comprehensive, long-term surveys spanning diverse community and environmental gradients play a pivotal role in unraveling and addressing factors such as soil conditions, climate dynamics, and human impacts on biodiversity (Lynn et al., 2021).

For instance, unaltered beech forests are absent in the lowlands of central Europe (Meyer et al., 2021), underscoring the necessity of preserving a portion of managed beech forests for natural succession (Meyer, 2020). German old-growth forests, now recognized as national treasures, hold significant potential to become future biodiversity hotspots. These beech forests provide the spatial breadth required to facilitate the natural spatiotemporal dynamics inherent to these ecosystems (Meyer et al., 2021). Nevertheless, various factors, such as the abandonment of traditional forest uses and nitrogen deposition, pose threats to the plant biodiversity sheltered by beech forests, leading to reduced vegetation diversity and homogeneity observed over decades.

Conversely, beech forests (*F. sylvatica*) constitute the primary temperate forest ecosystem in the western part of the Palearctic, yet most forests have undergone centuries of intensive management (Hrivnák et al., 2022; Mamadashvili et al., 2023). Consequently, the few remaining natural beech forests are recognized as crucial conservation targets in Europe and are integrated into the Natura 2000 network of protected areas. In the Caucasus region, particularly in Georgia and Armenia, beech forests harbor numerous endemic plants and animals (Aliev, 2021), and are included in the Emerald Network of Areas of Special Interest for Conservation. Recognized for their global significance in conserving temperate forest habitats in Europe, several remnants of natural beech forests have received the UNESCO World Heritage designation. Old-growth beech forests, particularly those at higher altitudes, host the greatest diversity of tree-related microhabitats (TreMs) (Mamadashvili et al., 2023). TreMs represent vital structural components of forests, exerting significant influence on biodiversity and ecosystem functioning (Przepióra and Ciach, 2022).

Zachar et al. (2022) investigated changes in vegetation within temperate forests (Fagetum) in the Matra Mountains of Hungary using three management methods: abandonment, mowing, and grazing. The authors concluded that systematic cutting and less intensive grazing (nonovergrazing) had a positive effect on biodiversity and species coverage. As a result, there was an increase in the naturalness of grassland habitats, while wild game was able to act as ecological engineers (Zachar et al., 2022).

As the majority of European forests are used for timber production and given the limited extent of unmanaged forests (especially primary ones), it is essential to include commercial forests in forest biodiversity conservation (Tinya et al., 2023). To develop ecologically sustainable forest management practices, it is important to understand the impacts of management on the organisms living in forests (Tinya et al., 2023). In Europe, particularly in Central Europe, the main conservation strategy for forests is integrative management or sustainable forest management (SFM), attempting to balance economic interests with biodiversity conservation objectives (Leidinger et al., 2020; Burrascano et al., 2023). This is complemented by strictly unmanaged forest reserves, often small and with a history of forest management. It is still not clear how these reserves contribute to conservation when the surrounding forest is under progressive integrative management (Leidinger et al., 2020).

Old deciduous forests, i.e., those with old trees (e.g., *Quercus robus/petraea* and F. sylvatica) in Europe are home to endangered species such as the bat Myotis bechsteinii (Singer et al., 2021). This bat species prefers woodpecker cavities to roost, i.e., woodpeckers are keystone species for this bat. Both woodpeckers and *M. bechsteinii* prefer forest patches because they have patterns similar to primeval forests. Therefore, it is important to maintain and increase forest patches, providing the habitat characteristics—limit of 20 (DBH > 50 cm) or 10 (DBH > 70 cm) of large deciduous trees with deadwood structures per hectare—within the managed forests (Singer et al., 2021).

Large areas in Europe consists of small-scale private forests, which often contain structures and habitats of high value for nature conservation. However, these small forests are under increasing pressure due to the cumulative need for firewood and wood products (Hansen et al., 2023). Hansen et al. (2023) studied microhabitat diversity and density, distribution of live tree and deadwood volumes, and management intensity in private (small-scale) forests in the hills of Lower Saxony (Germany). These researchers observed that there are two types of conservation value structures in these forests: (i) slow-evolving structures (type A)—such as large-diameter living trees and tree-related microhabitats, with longer timescales; and (ii) rapidly evolving structures (type B)—such as deadwood, which are influenced by both owner attitudes and landscape parameters.

Hansen et al. (2023) report that long-term (generational) commitments to conservation legislation, education of forest owners, and financial incentives are needed to conserve type A structures (slow evolution). On the other hand, efforts to promote type B structures (faster evolution) proved to be more advantageous in private forests due to the structural diversity of the stands, with strong identification of the owners with their land (Hansen et al., 2023).

In Germany, half of the forest area is privately owned (≈20 ha) (BMEL, 2014). While forestry on larger farms (≥20 ha) has become more mechanized and commercial, smallholders often extract little timber from their forests (Hansen et al., 2023). In addition, many of these private properties are approximately 3 ha (BMEL, 2014), often precluding economic use (e.g., heavy machinery). Because of this, they have greater structural diversity, biodiversity, TreMs, and deadwood. Deadwood also provides the substrate and habitat for many groups of organisms such as birds, invertebrates, bryophytes, lichens, fungi, and small mammals, many of which are endangered (Hansen et al., 2023). Similar to TreMs, the supply of deadwood on these properties and related biodiversity can be put under pressure with increasing forest management intensity (Hansen et al., 2023).

European feral cats exhibit better-preserved genetic integrity in forests located in the continental and temperate parts of Europe compared to those in the Mediterranean and Temperate Insular regions (Matthias et al., 2022). To ensure the conservation of these felines, efforts should focus on reducing mortality resulting from human activities, revitalizing and restoring their populations and breeding habitats, and managing prey populations (e.g., European rabbits). Additionally, measures should be implemented to minimize interactions between wild and domestic cat populations (Matthias et al., 2022). The presence of domestic carnivores (e.g., dogs and cats) in natural landscapes not only introduces domestic genes into wild populations but also poses risks such as disease transmission and predation on wild prey. Therefore, in the management plans of protected areas, careful consideration of the role of domestic carnivores is essential for achieving effective conservation objectives for biomes (Matthias et al., 2022).

Despite the high genetic integrity of feral cats throughout temperate continental biomes, continuous monitoring and research targeting the main feral cat populations must always be up to date, thus allowing the detection of hybridization problems that arise and contributing to a deeper understanding of the complex ecological and demographic dynamics that drive hybridization (Matthias et al., 2022). Feral cats are protected by the Bern Convention, which says that each signatory country must take appropriate and necessary legislative and administrative measures to ensure special protection (Matthias et al., 2022). However, while the threat to conservation due to dog-wolf hybridization is recognized by the Standing Committee of the Berne Convention, the same is not true for feral cats (Matthias et al., 2022).

In addition to temperate upland forests, floodplain forests, which are periodically flooded, are considered the most complex and biologically rich habitats in the temperate zone (Przepióra and Ciach, 2022). Softwood riparian forests are classified as a near-threatened habitat, while hardwood forests are considered threatened throughout Europe (Janssen et al., 2016), emphasizing the urgent need for their restoration (Geist, 2016) (Fig. 26.3). Among the inhabitants of these ecosystems is the *Castor fiber* beaver, which once faced near extinction due to extensive hunting but has since rebounded across much of its former range,

FIG. 26.3 River Nerl, Ivanovo region, Central Russia. *(Photo credits: Surina Elena.)*

thanks to active conservation efforts (Dalbeck et al., 2020). Beavers play a significant role in shaping their environment by constructing dams, engaging in digging activities, and logging. These activities have notable effects on small, low-order streams and their floodplains, influencing the distributions, richness, and abundance of amphibian species (Dalbeck et al., 2020), thereby enhancing local biodiversity. Indeed, Dalbeck et al. (2020) observed that all 19 amphibian species occurred in beaver ponds.

Beavers tend to be the main providers of essential habitat for amphibians at the headwaters of small streams, which account for 60%–80% of water bodies in temperate European river basins (Dalbeck et al., 2020) (Fig. 26.4). Beavers and their ability to modify or create habitats for endangered amphibians help to implement some European guidelines on water management, contributing to their long-term conservation (Dalbeck et al., 2020). In addition to fungal diseases (e.g., Bd and Bsal chytrids), invasive predators, and pollution (e.g., greenhouse gases), habitat loss is the greatest threat to amphibians, and beavers can help reverse this situation (Dalbeck et al., 2020). This occurs, among other reasons, because beavers increase availability and reduce habitat fragmentation, i.e., they increase the areas that can be colonized by endangered amphibian species (Dalbeck et al., 2020).

3 Human activities and their impacts

From the Neolithic period (5000 years) to the present day, the diversity and composition of European forests have been and are influenced by human activities, which have led to this continent having around 2%–3% of native forests (= 4.68 million ha, excluding Russia) (Kaplan et al., 2009; FAO, 2020). The destruction of forests, waters, and habitat

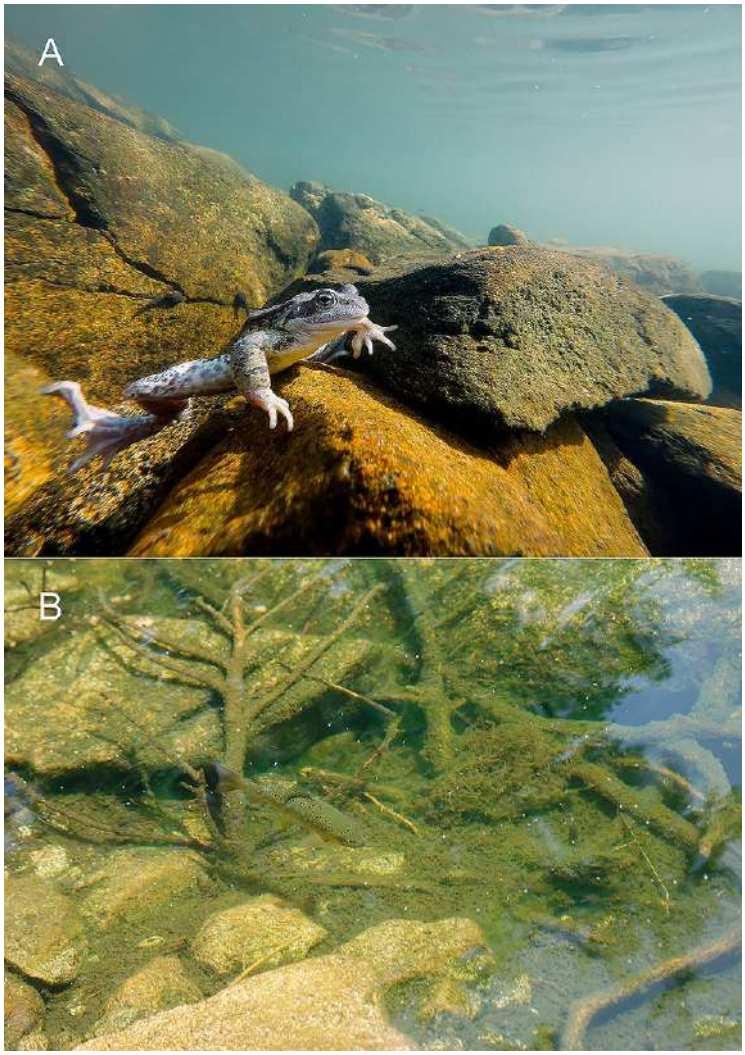

FIG. 26.4 (A) Rana temporaria (Ranidae) and (B) Salmo trutta (Salmonidae). Retezat Mountains (Southern Carpathians), Romania. *(Photo credits: Ovidiu Dragan.)*

fragmentation, especially through agricultural intensification, profoundly impacts biodiversity (e.g., amphibians) (Dalbeck et al., 2020). In addition to deforestation, hunting in the past has had a strong impact on European animals, such as *B. bonasus*. This animal was slaughtered due to the value of its skin and horns and was once considered an endangered species, but due to its protection, it is currently classified as a vulnerable species.

While 24% of Europe's forests are protected (Forest Europe, 2020), the majority of these areas are also managed for timber production (Fig. 26.5). Europe has long been characterized by forests subjected to relatively intensive management practices, and the combined pressures of global biodiversity loss and climate change have underscored a silvicultural crisis, particularly evident in extensive disturbances within commercial monocultures, such as those found in Central Europe (e.g., Czech Republic) (Kjucukov et al., 2022). However, a significant portion of European forests has historically undergone intensive forest management (e.g., Romania), and restoring these areas to their primary forest characteristics would require centuries (Albrich et al., 2021).

There are few remaining complex primary forest landscapes in temperate Europe, with approximately 80% of them located in the Carpathian Mountains, characterized predominantly by beech, fir, and spruce trees (Reif et al., 2022). One notable example is found in the Fagaras Mountains of Romania (Kameniar et al., 2023). The entire primary temperate forest landscape has undergone a diverse array of historical disturbances in terms of severity and timing (Trotsiuk et al., 2014). High-severity disturbances have led to periods of synchronized disturbance activity at the landscape scale. In contrast, moderate and low-severity disturbances have been asynchronous and sporadic in primary forests dominated by *P. abies* and *F. sylvatica* (Kameniar et al., 2023). A peak in canopy disturbance occurred in the late 19th century, particularly between 1890 and 1910, across the region (Kameniar et al., 2023). At the stand scale, there have been periods of synchronized disturbances with varying severities for forests dominated by *P. abies* and *F. sylvatica*, with the former exhibiting moderate synchrony and greater variability than the latter (Kameniar et al., 2023).

Estimates of virgin, near virgin, and ancient forests in Romania range from 150,000 to 200,000 ha. However, due to illegal and legal logging, around 350,000 ha disappeared between 2001 and 2019 (Reif et al., 2022). Legislation in Romania requires virgin forests to be protected and production forests to be managed sustainably, and these forests are also subject to European legislation (Reif et al., 2022). However, there is a serious lack of oversight at all levels of administration, even in National Parks (some dependent on logging for maintenance budgets). According to Reif et al. (2022), large-scale clear-cutting (>of 3 ha) is

FIG. 26.5 (A) Production of seedlings and (B) processing of tree trunks. Germany. *(Photo credits: Surina Elena.)*

only legally allowed in exceptional situations (e.g., for safety reasons). Nevertheless, in practice, large-scale clear-cutting is the normal situation and is declared as a "permitted" intervention.

The Białowieża Forest in Poland stands as a unique bastion of natural sciences and educational opportunities in Europe, boasting a conservation tradition spanning over 500 years and harboring trees that have stood for more than two centuries (Wojnicka-Półtorak et al., 2023). Many of its best-preserved stands can be deemed primitive due to their possession of essential characteristics indicative of such status (Wojnicka-Półtorak et al., 2023). Its worth noting that there are no entirely pristine forests in Europe untouched by human influence. Throughout its history, the Białowieża Forest has witnessed various forms of human utilization. However, its designation as a royal forest conferred a degree of protection, limiting exploitation and agricultural expansion from the 15th to the early 20th century (Samojlik et al., 2013). Nevertheless, the onset of World War I marked the beginning of large-scale timber exploitation. In the aftermath of the war, from 1924 to 1929, The Century European Timber Corporation felled a staggering 2.5 million m^3 of timber using meticulous harvesting methods (Wojnicka-Półtorak et al., 2023). These areas were left devoid of trees except for designated seed trees and natural monuments, with no reforestation efforts undertaken. Naturally, the deforested areas regenerated primarily through the dispersal of light-seeded species (Wojnicka-Półtorak et al., 2023).

The adoption of drought-tolerant exotic trees in response to climate change in Central Europe has gained traction (Kriegel et al., 2021; Bärmann et al., 2023). One such species is *Pseudotsuga menziesii*, native to western North America and adapted to temperate climates with summer droughts, it is renowned for its rapid growth and marketable wood (Bärmann et al., 2023). However, the potential ecological repercussions of its introduction have yet to be comprehensively assessed. In light of this gap, Bärmann et al. (2023) investigated the impact of introducing this exotic species alongside *F. sylvatica*, a native of Central Europe. Their findings revealed significant alterations in the undergrowth, particularly when *P. menziesii* accounted for 40%–50% of the canopy, leading to a complete loss of the original character of the undergrowth when it exceeded 75% (with *F. sylvatica* comprising less than 25% of the canopy) (Bärmann et al., 2023). Consequently, Bärmann et al. (2023) caution against high proportions of this exotic species in conjunction with *F. sylvatica*, especially in areas prioritizing nature conservation efforts.

Douglas fir monocultures often have negative effects on forest biota, while the effects of mixtures with native tree species on forest ecosystems are less studied (Kriegel et al., 2021). Kriegel et al. (2021) investigated the effects of three tree species (Douglas fir, Norwegian spruce, native European beech) on land beetles in temperate forests of Germany. These authors noted that terrestrial beetle abundance and functional diversity were higher in pure Douglas fir stands, while tree mixtures showed intermediate values between pure conifer and pure beech stands. This variability likely stems from factors beyond the mere identity of tree species, including structural characteristics within forest stands (Kriegel et al., 2021). The authors noted that while mixed planting partially mitigated the distinct changes in assemblage structure observed between pure beech and pure Douglas fir stands, these differences persisted to some extent.

The promotion of beneficial forest structures can alter the biodiversity associated with the planting of nonnative trees (Kriegel et al., 2021). However, conservation efforts targeting typical species composition may be compromised by the integration of nonnative tree species, even in mixtures with native trees, due to changes in the clustering structures of the associated biota (Kriegel et al., 2021). The contrasting results of the work of Kriegel et al. (2021) and other studies on the influence of Douglas fir on the associated biota highlight the need to understand the ecological consequences of planting nonnative tree species deeply and broadly. These results, in part, are due to taxon-specific responses and changes in environmental effects throughout forest development. Therefore, more research is needed to integrate multiple taxa and multiple stages of forest development (Kriegel et al., 2021).

Despite accounting for more than 80% (<1% of forest biomass) of plant diversity in temperate forests and being considered an indicator for sustainable and disturbance-sensitive forest management, the understory remains neglected (Blondeel et al., 2021; Cutini et al., 2021; Frati et al., 2022). However, the practice of coppice forest management has been revived by various European countries. Coppice forests have experienced extensive historical management, resulting in a cyclical succession of phases (abandonment and overexploitation) driven by changes in human population density and activities such as wood, firewood, and charcoal harvesting, as well as grazing (Mattioli et al., 2015). For example, coppice forests represent 42.3% of forest cover in Italy, prompting the establishment of new requirements for their restoration (Chelli et al., 2023).

Understory species richness and changes in species and functional composition are decreased during tree regeneration after felling (Chelli et al., 2023). Coppice management affects understory plants: (i) directly through forest roads, harvesting machines, and/or charcoal kilns and (ii) indirectly by modifying soil conditions such as soil erosion, moisture, nutrients, light, and temperature, among others (Cervellini et al., 2017; Douda et al., 2017). Even old-growth coppice forests have impoverished understory. In addition, there are other problems such as

uncertainty about the effective impact of coppice on the understory as well as cross-sectional effects and complex responses of different facets of diversity (Chelli et al., 2023). Therefore, coppice management can have an impact on the landscape by producing forests with less recreational value.

Herbivory by ungulates, and at the same time the deposition of atmospheric N, continues to eutrophicate forests and threaten many rare plant species. These pressures can interact critically to shape biodiversity (Segar et al., 2022). Increases in herbivory by ungulates were associated with high temporal species turnover. However, the identities of the winning and losing species depended on the levels of N. Herbivory reduced the proportion of nonnative and nutrient-demanding species and favored threatened and small-distributed species, under conditions of low levels of N deposition (Segar et al., 2022). However, all of these results were reversed at high levels of N deposition. Herbivores also decreased shrub cover, perhaps increasing the effects of N due to increased light levels in the understory. Eutrophication levels can determine whether herbivory acts as a catalyst for the "N time bomb" or as a conservation tool in temperate forests (Segar et al., 2022).

Invasive alien species (IAS) pose a significant threat to both global and local biodiversity, leading to issues such as hybridization, disease transmission, and interspecies competition, particularly within forest ecosystems (Langmaier and Lapin, 2020). Human activities are mainly responsible for introducing and disseminating IAS in forests, and these are: (i) clearing can alter lighting conditions and resource availability favoring IAS (e.g., *Impatiens parviflora*); (ii) the construction of forest roads can promote the spread of seeds and specimens of exotic plants through the movement of contaminated soil and construction material; and (iii) highways near massive forests (Langmaier and Lapin, 2020).

Langmaier and Lapin (2020) analyzed the impact of exotic plant species on forest regeneration, which they consider to be one of the important steps for the survival of forest ecosystems (Langmaier and Lapin, 2020). These authors noted, in their review of the results of 3825 studies, that 53 species of vascular plants negatively influenced forest regeneration in Central European forests. In total, 21 tree species were affected by the invasive alien species in 24 studies. The main negative mechanisms of IAS on the success of native tree regeneration were competition for resources and physicochemical and structural impacts on regeneration, in addition to indirect impact through interaction with other species (Langmaier and Lapin, 2020).

And finally, megaprojects are projects that enormously impact the landscape (natural or seminatural) quickly and radically, and despite efforts to reduce and/or mitigate the negative effects, it is difficult to avoid their detrimental effects (Piccini et al., 2023). To develop conservation plans in forests threatened by human activities (e.g., construction sites), it is key to characterize forest structure and its bioindicators (e.g., saproxylic beetles) (Piccini et al., 2023). To sustain the abundance and diversity of saproxylic beetles locally, it is necessary to increase the volume of deadwood and enrich the diversity of living trees, which are threatened by the expansion of the construction site, such as that of the Turin-Lyon high-speed rail line (Piccini et al., 2023).

4 Operational and monitoring tools

4.1 Interviews and questionnaires

The application of appropriate management measures or changes in forest management in specific areas has faced difficulties in proving relationships between a specific type of forest management and the presence or absence of specific species (Lindenmayer and Laurance, 2012). The conflicting beliefs between forest management that is deleterious to biodiversity and forest management considered sustainable (e.g., ecological functions) and, therefore, negatively impacting its productivity, have been intensely discussed.

Due to the above facts, a survey was conducted among Czech biologists to provide specific feedback to foresters and other stakeholders based on the scientific and empirical knowledge of these interviewees (Kjucukov et al., 2022). As a result of the questionnaires, the following most relevant points were observed: (i) negative view of current forestry practices due to the creation of homogeneity in the forest environment and elimination of important habitats; (ii) environments of old-growth forests with structural diversity and open forests with the presence of old-growth trees are essential (Kjucukov et al., 2022); and (iii) space and temporal continuity of forest sites are important (Hofmeister et al., 2019).

The Bialowieza Forest is a contested cross-border forest massif in Poland and Belarus (Giergiczny et al., 2023). These authors have pioneered the documentation of how country-specific legacies shape preferences for greater forest protection over timber production. This is aimed at transitioning value chains based on sustainable yield forestry to ecotourism. Giergiczny et al. (2023) used, for both countries, a quantitative ordered logit model based on questionnaires to Polish and Belarusian ecotourism business owners. The cited authors studied the factors of owners' preferences in relation to the different values of the Bialowieza Forest, with qualitative data to identify attitudes toward the expansion of protected areas of this forest.

Giergiczny et al. (2023) obtained the following results: (i) Belarusian ecotourism business owners support increased protection of the reserve area, but Poles do not; (ii) the proportion of foreign guest covariates with support

for increased protection of the area; and (iii) the size of the hotel business and the role of foresters as customers have diminished interest in protecting the area (Giergiczny et al., 2023). According to Giergiczny et al. (2023), support for increasing the reserve area has spread in Poland, but not in Belarus. There are several motivations (e.g., socioeconomic) for conflicts over the conservation of the Polish part of the Bialowieza Forest that involve actors and stakeholders with competing interests. To this end, encouraging multiple value chains and evidence-based collaborative learning are essential components (Giergiczny et al., 2023).

4.2 Normalized difference vegetation index

The remote sensing index Normalized Difference Vegetation Index (NDVI) is a measure of plant reflectivity, which serves to quantify the greenery of vegetation. Based on the spectral irradiance of visible and near-infrared light, the closer to +1 it is, the greener the vegetation (Hermann et al., 2023). NDVI has been used to identify leaf browning events in Europe's temperate forests (Hermann et al., 2023). Recently, forest death in Europe has become more intense/extensive, being greatly influenced by changes in temperature and precipitation, and can be monitored through the greenness of the forest (Buras et al., 2021; Hermann et al., 2023).

Hermann et al. (2023) quantitatively studied the 3-year meteorological history that preceded forest vegetation reduction events in the temperate biome. This research team took a pragmatic approach based on NDVI to identify NDVI events in European forests between June and August 2002–2022 (Hermann et al., 2023). The hottest summer on record in Europe in 2022 negatively affected 37% of temperate forest regions, with forest greenness reduced more extensively than in any other summer between 2002 and 2022 (Hermann et al., 2023). The rainfall deficit was 26 months in the temperate biome with abnormally persistent temperature during this period, with distinct peaks at the end of the last three growing seasons (Hermann et al., 2023).

In recent years, recurring large-scale droughts and heatwaves since 2018 have inflicted increasingly severe damage to greenery. The frequency and extent of this loss have escalated, underscoring the urgency for comprehensive monitoring of weather patterns across multiple seasons. Such monitoring can offer valuable insights into the heightened risk of forest bleaching during subsequent summers (Hermann et al., 2023).

The remarkable occurrence of a record-low NDVI event during the summer of 2022 underscores the critical importance of comprehending the intricate relationship between forests and meteorological phenomena. With climate change exacerbating these interactions, there is a mounting risk of irreversible forest degradation and potential extinction (Hermann et al., 2023).

4.3 The developed old-growth indicator (OGI)

Assessing the conservation value of forests and making informed management decisions hinge on accurately quantifying the extent of old-growth characteristics within them (Meyer et al., 2021). In their study, these researchers introduced and applied the developed old-growth indicator (OGI) to evaluate the presence of old forest structures in potential beech forest sites across Central Europe, areas marked by a prolonged history of forest management.

Meyer et al. (2021) utilized inventories from primitive beech forests in Slovakia, categorized as old growth, along with data from 39 comparison stands across northern Germany's lowlands. These comparisons stands varied in management intensity, age, and tree species composition, serving as the foundation for the set of structural variables used in developing and applying the OGI.

The comparison process involved trios of stands: (i) a European beech forest over 100 years old left to natural development (ND), (ii) an 80–100-year-old Scots pine forest with a mix of deciduous trees in the understory (OP), and (iii) a Scots pine forest younger than 80 years (YP) (Meyer et al., 2021).

The OGI measures the degree of overlap of the 5–95 percentage bands of a given comparison position with primal forests. The OGI reaches values between 0 and 1 and allows considering all thematic groups and variables separately, as well as calculating an aggregate value (Meyer et al., 2021). The ranks defined by these authors were: 0.03–0.26 for YP stands; 0.07–0.30 for OP; 0.13–0.42 for ND stands; and 0.71 and 0.74 for primeval forests.

The OGI is an easy, comprehensive, and reproducible indicator of the maturity of a forest stand, which allows for differentiated aggregate assessments and can implement an established workflow for forest surveys of national parks and rigorous forest reserves in Germany (Meyer et al., 2021). In addition, OGI can be used as an indicator for development over time and assess the effectiveness of different management regimes for conservation and restoration measures (e.g., leaving or creating deadwood) (Meyer et al., 2021).

4.4 The forest condition index (FCI)

The state of the Forest Index (FCI) is the first indicator, at the European level, developed to date to assess the condition of forest ecosystems at this scale, via mapping (Marín et al., 2021). The FCI considers a variety of factors, including the structure and function of the ecosystem and the biodiversity that depends on it. This index will increase the data supply of European ecosystem services (Marín et al., 2021).

For the development and testing of the FCI, it used harmonized, published, and open datasets. In his test, he presented reliable results for assessing the condition of temperate and other forests, in the Carpathian regions, Dinaric Alps, and the Alps, among others (Marín et al., 2021). These results were compared with data derived from the reports for the EU Habitat Directive and explicit dataset on known primary forests in Europe, showing excellent results, and thus validated (Marín et al., 2021). This indicator considers structural, functional, and forest composition aspects that are relevant to the health and vitality of species and habitats hosted by forest ecosystems (Marín et al., 2021). This information spatially on the condition of the forest can be the reference map (1 km resolution) to monitor the state and changes in condition due to exposure to pressures and threats (Marín et al., 2021).

This index shows the map of the forest condition, contributing to the assessment of current and future changes affecting European forests, and supports the monitoring of the proposed targets on this continent. Information on forest conditions, biodiversity, and the extent of forest degradation is vital for prioritizing human protection and financial resources, preventing further degradation, and restoring and rehabilitating degraded forest ecosystems (Marín et al., 2021).

4.5 Remote sensing and digital platforms

Information from the components of European forest biodiversity, which is not directly visible through remote sensing, is highly valuable for testing and integrating information acquired through remote sensing techniques such as airborne laser scanning (Burrascano et al., 2023). Information from the newly digitized platform can be used to integrate and improve ecosystem assessment by adjusting the linkages between the measurement of structural diversity through airborne laser scanning (e.g., habitat-specific variables) and multitaxon biodiversity (e.g., dark diversity and multitaxa).

The study of forest multitaxa aggregates different knowledge (Tinya et al., 2023) and data on the three main components of ecosystems, species composition, structure, and function, which differs from other data-sharing platforms that assess only one aspect of the ecosystem (e.g., vegetation) (Burrascano et al., 2023). These data, collectively, can promote interdisciplinary studies and elucidate the results of different conservation and biodiversity management policies of multiple taxonomic groups guided by structural characteristics of the stand (e.g., diameter distributions of standing trees and deadwood) (Burrascano et al., 2023; Tinya et al., 2023). However, some important taxa have been neglected and should be included, such as soil-dwelling organisms, Diptera and Hymenoptera, amphibians, reptiles, and mammals, among others (Spina et al., 2023; Tinya et al., 2023).

Munteanu et al. (2022) developed a novel and transferable approach to detecting high conservation value forests (HCVFs) based on the integration of information on potential current anthropogenic pressures (e.g., firewood demand, population density, road infrastructure, and terrain), historical spy satellite imagery, and contemporary remote sensing (Landsat) data. This was done because HCVFs are particularly important (e.g., biodiversity and ecosystem services), but they face many threats (Munteanu et al., 2022). Due to the lack of systematic inventories (e.g., parts of Eastern Europe), these forests remain largely unknown and often unprotected (Munteanu et al., 2022). These authors applied this method to the Romanian Carpathians, mapping forest continuity (1955–2019), canopy structural complexity, and anthropogenic pressures. Munteanu et al. (2022) identified 738,000 ha of HCVF, with more than half of this area identified as susceptible to current human pressures and lack of formal protection. By providing a framework for large-scale HCVF monitoring, this type of approach facilitates the integration of HCVF into forest conservation and management (Munteanu et al., 2022).

Heinken et al. (2022) compiled EuForPlant: a new and extensive list of forest vascular plant species in 24 regions spread across 13 European countries, using vegetation databases and expertise. These authors classified the species specifically by region into four categories that reflect the degree of their affinity with forest habitats: (1.1) species from forest interiors; (1.2) species of forest edges and openings; (2.1) species that can be found both in the forest and in open vegetation; and (2.2) species that can be found partially in forests, but mainly in open vegetation. In addition, they created an additional distinguished category "O," which covers typical species of nonforest vegetation.

EuForPlant comprises 1726 species with 1437 species of herbaceous strata, 159 shrubs, 107 trees, 19 lianas, and 4 epiphytic parasites (Heinken et al., 2022). Heinken et al. (2022) noted that in all regions, generalist forest species outperformed specialized forest species. Although the degree of change between forest affinity categories among regions was relatively low, about one-third of forest species changed categories in at least one of the study regions (Heinken et al., 2022). According to these authors, the list can be used in vegetation science and the ecology of global changes related to community dynamics and forest biodiversity (Heinken et al., 2022).

Heinken et al. (2022) conclude that EuForPlant can be connected with other databases such as those of vegetation plots such as EVA (Chytrý et al., 2016) and sPlot (Sabatini et al., 2021), plant traits such as TRY (Kattge et al., 2020) and ClimPlant (Vangansbeke et al., 2021), climatic ones such as CHELSA (Karger et al., 2017) and SoilTemp

(Recallchts et al., 2020). Interconnecting databases presents a powerful tool for extracting insights into various aspects of forest ecology, including patterns of plant diversity, the structural dynamics of understory plant communities, plant dispersal, and colonization, the niche breadth of plant species, distribution sizes, and the vegetation responses to global environmental changes, among other factors (Heinken et al., 2022).

4.6 Aerial RGB imagery and deep learning

Beloiu et al. (2023) conducted an assessment of the efficacy of the Faster R-CNN Convolutional Neural Network algorithm in the realm of end-to-end object detection. By integrating open-source RGB aerial imagery, the system aimed to pinpoint and geolocate tree species within the upper canopy layer of heterogeneous temperate forests.

The study focused on evaluating the identification of four tree species—*P. abies*, *A. alba*, *P. sylvestris*, and *F. sylvatica*—within these diverse forest environments. The findings revealed several key observations: (i) for single-species models, detection accuracy ranged from 0.86 for *P. abies* to 0.59 for *P. sylvestris*, with *A. alba* and *F. sylvatica* falling in between 0.84 and 0.75, respectively. (ii) In multispecies models, *P. sylvestris* achieved a higher accuracy of 0.92, while the performance for other species either remained consistent or exhibited slight decreases. (iii) Furthermore, the study highlighted the influence of site conditions, such as forest stand structure, and lighting conditions on the model's performance.

Beloiu et al. (2023) concluded that the faster R-CNN algorithm effectively maps the locations of four distinct tree species within heterogeneous forests, offering a foundation for future inventories and targeted management strategies to bolster forest resilience. They emphasize that leveraging solely RGB aerial imagery enables cost-effective monitoring of tree species with broad applicability.

This approach addresses several existing challenges, including subtle visual differences between tree species, intricate canopy structures, imbalanced class distributions, model generalization, and annotation variability (Beloiu et al., 2023). It represents a highly automated, economical, and accessible method for streamlining the identification and geolocation of these tree species, applicable wherever RGB image data is available.

Moreover, the authors have curated a database comprising over 10,000 annotated trees, accessible online for further development and utilization. The object detection models of Faster R-CNN offer significant potential for maintaining ecosystem services and enhancing forest resilience amidst the rapid changes wrought by climate change (Beloiu et al., 2023).

4.7 Deadwood quantity and quality

Deadwood is an important structural feature in managed or conservation forests, providing essential resources for the maintenance of various organisms (Oettel et al., 2020) (Fig. 26.6). The volume of deadwood is often used as an indicator of forest biodiversity or naturalness, but a comparison should be made between forest stands of the same forest type (Oettel et al., 2020). The minimum limits for deadwood that ensures the persistence of saproxylic organisms and maintain the overall biodiversity of forest ecosystems are already known. But reaching these limits is still a challenge in forests where nature conservation is not the only objective (e.g., forests managed for timber) (Kapusta et al., 2020). A prerequisite for developing methods that increase deadwood retention without decreasing forest production is to determine the quantity and quality of deadwood in managed forests (Kapusta et al., 2020).

The Natura 2000 forest area studied was largely shaped by forestry practices that do not lead to the improvement of its resources [lying coarse woody debris (CWD)]. The problems are: (i) the amount of CWD is low, slightly passing the typical amounts for forests not covered by any form of protection; (ii) the CWD pieces are small compared to the size of living trees, and do not fully represent the composition of tree species and the types and stages of decomposition; and (iii) more intensively harvested sites have less CWD than sites without such signals, suggesting that CWD accumulated over the years is taken from the forest, during forestry interventions, even low-value ones (Kapusta et al., 2020).

In addition to the abovementioned practices, local communities contribute negatively by removing the wood cut for their needs (e.g., firewood), especially in places that are easier to access (e.g., by cars). Tree species composition, tree richness, age, volume, and deadwood composition, among others, also affect CWD parameters in managed forests (Kapusta et al., 2020; Oettel et al., 2020). The diversity of species and classes of diameter and degree of decomposition of deadwood differ between forest types. The highest diversity of deadwood is observed in beech and spruce forests ($109\,m^3\,ha^{-1}$) and the lowest in larch forests ($23\,m^3\,ha^{-1}$), carbonate pine, and hardwood floodplain forests (Oettel et al., 2020). These data show that after a relatively short period of abandonment, one already has a considerable amount of deadwood similar to old-growth forests (Oettel et al., 2020). In addition, the proportion of deadwood is related to the total biomass of the wood aboveground. Thus, it should also be considered as an indicator of forest biodiversity (Oettel et al., 2020).

To increase CWD in multifunctional forests, the following measures can be adopted: (i) eliminate or modify forest management practices that are not necessary through recent research, such as removing or fragmenting the old

FIG. 26.6 (A) Dead fallen tree trunk and (B) dead tree with mushrooms and lichens. Scientific Reserve Gemenele, Retezat National Park, Romania. *(Photo credits: Dan Cogalniceanu.)*

CWD; (ii) increase positive practices that affect the quantity and quality of CWD (e.g., not harvesting low-value wood); and (iii) set aside small areas to accelerate CWD retention (Kapusta et al., 2020). It is necessary to make the correct selection of withdrawal zones (e.g., hinder access) in which they are far from paved roads and human settlements, to increase the amount of CWD (Kapusta et al., 2020).

4.8 Tree-related microhabitats (TreMs)

Living or dead tree-related microhabitats (TreMs) are structures that provide unique space or substrate for the presence, development, foraging, or shelter of a single species or entire sets of organisms throughout all or part of their life cycles. More than 60 types of TreM have been identified. The taxa are associated with the presence of TreM invertebrates (e.g., Coleoptera), bats, and birds, with many of these species being stenotypies, rare or critically endangered (Spina et al., 2023). TreMs are therefore a key component of the ecosystem for maintaining temperate forest biodiversity (Bütler et al., 2013).

Tree-related microhabitats (TreMs) are a quantitative indicator of the heterogeneity of forest habitats (e.g., *F. sylvatica* and *F. orientalis*) (Mamadashvili et al., 2023). TreMs can serve as a tool to map, monitor, and enhance forest biodiversity in managed forests (Spina et al., 2023). In this technique, as in others, is important to have a standardized protocol (Mamadashvili et al., 2023).

In the present case, a unified diversity and ordering framework based on Hill numbers is applied to elucidate the unobserved TreM types, starting from the rare to dominant TreMs (Mamadashvili et al., 2023).

The composition of the TreM assemblages was basically determined by the species and altitude of *the Fagus* trees, with a focus on dominant TreMs by diameter at breast height (DBH) (Mamadashvili et al., 2023). The coverage of identified TreMs increased with the number of trees assessed and DAP but was lower in old-growth forests than in production forests (Mamadashvili et al., 2023). The diversity of rare and dominant TreM types was lower in production forests than in old-growth forests, increasing with elevation when only dominant TreMs were standardized for sample coverage (Mamadashvili et al., 2023). It is

important to focus conservation efforts on the forests of both *Fagus* species, at different altitudes, covering all TreM assemblages (Mamadashvili et al., 2023).

Asbeck et al. (2022) analyzed the influence of environmental factors on the occurrence of TreMs groups by comparing primary and managed forests. To this end, they collected data on primary forests in the South (Romania) and Western Carpathians (Slovakia) and managed forests in the Black Forest (Germany). Asbeck et al. (2022) modeled the richness and different groups of TreMs per tree, using generalized mixed linear models, with diameter at breast height (DBH), altitude, slope, and aspect as predictors for *A. alba*, *F. sylvatica,* and *P. abies* in primary and managed temperate montane forests. These researchers observed the following facts: (i) congruent results for overall richness and the vast majority of TreM groups, (ii) trees in primary forests showed higher richness of all and specific types of TreMs, and (iii) the main factors favorable to TreMs were DBH and altitude. Asbeck et al. (2022) recommend that forest and nature conservation managers focus on the conservation of remaining primary forests and adopted biodiversity-oriented forest management approaches in the selection of high-quality habitat trees that already provide a high number of TreMs in managed forests, based on comparison with primary forests.

Asbeck et al. (2022) recommend that the constant removal of trees or tree parts that have "defects" (e.g., exposed sap and heartwood) should be decreased in forest management to preserve these important TreMs as sources of resources. In addition, these authors suggest planting large-diameter trees, beech, and other broadleaf species will increase the number of habitats available for the habitation of other forest species in managed forests (Asbeck et al., 2022).

Spînu et al. (2022) compared the occurrence of TreM in live and dead trees to see if dead trees support more and other TreMs than living trees in the Black Forest region. These authors noted that tree diameter and identity (species combination and vitality status) were the preponderant causes for TreM richness and abundance, which were higher in dead *A. alba*. Although *A. alba* and *P. abies* were similar to those provided by large living trees, their TreM composition was different (Spînu et al., 2022). As such, dead trees cannot replace the habitat functions of living trees. Dead trees (e.g., large, decayed protuberances) can complement them to increase the overall diversity of TreM at the stand level (Spînu et al., 2022).

Management should focus on the habitat quality of trees of different identities and functions for the conservation of biodiversity (Fig. 26.7). Both tree habitats and their TreMs were accounted for in numbers rather than according to the functionality and complementarity in the retention approaches applied (Spînu et al., 2022). The trade-off between sufficient quality and quantities of standing deadwood vs. retention of large living trees needs to be considered when disturbances increase large tree mortality (Spînu et al., 2022). The larger these trees and trunks, the greater the abundance and richness of TreM (Spînu et al., 2022).

TreM has been used for upland forests and natural riparian forests such as willow-poplar (Przepióra and Ciach, 2022). The diversity, density, and richness of TreMs depend on the number of tree species, the percentage of *Salix* sp. willows or multitrunked trees, the density of living trees, and the basal area of living or dead trees (Przepióra and Ciach, 2022). In addition, the TreMs value of an individual tree depends on its species, its diameter, the number of trunks, and its life status (living or dead) (Przepióra and Ciach, 2022). Multispecies forests are influenced by natural changes (e.g., water flow) and TreM richness and abundance are critical points. The presence of a mosaic of habitat patches, dependent on the functioning of a large lowland river, combined with the presence of animal species capable of modifying the ecosystem, contribute to the pattern of abundance and diversity of TreM (Przepióra and Ciach, 2022). Due to the strong threat of disappearance, the urgency of protecting or restoring these habitats is highlighted (Przepióra and Ciach, 2022).

4.9 Diversity and other indices

Biodiversity indicators are the most frequently used tool to monitor the status and changes of biodiversity and the effects of management actions. Oettel and Lapin (2021) analyzed 162 studies and they identified 9 groups of biodiversity indicators (BI), corresponding to 32 BIs and linked 7 groups of management indicators (MI) to 44 MI. The main results were: (i) arthropods, birds, and plants are the most frequently used IBs in managed forests in Europe; (ii) BI associated with specific MI—saproxylic and Collembola species with deadwood, bird families, and terrestrial species with links to regeneration (Oettel and Lapin, 2021); (iii) 17 species identified and proposed as important species in future studies as bioindicators; (iv) high structural diversity (e.g., canopy cover and floristic diversity) is associated with an increase in diversity, especially concerning vascular plants, birds, and terrestrial species (Oettel and Lapin, 2021); and (v) adaptation of forest management to biodiversity requires regular active monitoring of MI to assess temporal and spatial changes and BI to assess the effectiveness of (Oettel and Lapin, 2021).

Oettel and Lapin (2021) showed that high structural diversity is strongly correlated with high levels of biodiversity, highlighting vascular plants, birds, and soil-dwelling species. According to these authors, silvicultural management measures can impact aspects of forest

FIG. 26.7 (A) Ranidae; (B) *Mitopus morio;* (C) Salamandrida;, (D) Gryllidae; (E) mushroom; and (F) herbs and trees. Retezat Mountains (Southern Carpathians), Romania. *(Photo credits: Dan Cogalniceanu.)*

structure such as basal area, canopy cover and diversity, deadwood, diversity of tree species, diameter distribution, age and height of trees, and composition of tree species.

Saproxylic diversity was negatively affected due to the reduction in the amount and heterogeneity of deadwood, as a result of European forestry changing the composition of tree species (Rieker et al., 2022). Efficient conservation requires knowledge about the importance of relevant diversity factors among taxa.

Leidinger et al. (2020) compared the forest structure and biodiversity of various taxa between previously managed forest reserves and stands under progressive integrative management in beech and oak forests. Alpha diversity was higher in the reserves for birds and bats (<light availability) and higher in the stands managed for saproxylic and nonsaproxylic plants and beetles (>light availability). Reserves had 17 indicator species (three species on the red list) and managed stands had 34 (one species on the red list) (Leidinger et al., 2020). Progressive integrative management and reserves are complementary in that they benefit different taxa and partially different communities (Leidinger et al., 2020). The reserves were undisturbed mature stands with low volumes of deadwood. New reserves should include early or late successional stands to benefit demanding species in light and deadwood (Leidinger et al., 2020).

Rieker et al. (2022) investigated the relative importance of host vs space for saproxylic diversity across a spatial extent of 600 km in Germany. Additionally, these authors explored the effects of forest stands, host clades, tree species, and interregional variations on saproxylic diversity. Their findings provide valuable insights for spatial and host-tree-related conservation strategies (Rieker et al., 2022).

The study revealed that the beta diversity of fungal sporocarps and beetles was equally influenced by both space and host factors. However, the beta diversity of molecular fungi and bacteria primarily depends on the host (Rieker et al., 2022). Interestingly, beta diversity exhibited lower variation between regions than among individual forest stands. Notably, differences between regions

significantly impacted fungal beetles and sporocarps (Rieker et al., 2022).

Furthermore, the researchers observed that the host tree clade played a crucial role in shaping beetle beta diversity, while the fungal sporocarp beta diversity was influenced by the host clade. Surprisingly, host tree species outweighed host clade effects in determining fungal and molecular bacterial beta diversity (Rieker et al., 2022).

Given these complex interactions, adopting a simplistic space- or host-centered strategy may not be optimal, as different taxa exhibit divergent responses. To effectively maintain both rare and abundant species, promoting a diverse array of deadwood tree species across the national scale is essential in Europe's temperate forests (Rieker et al., 2022).

The reduced plant diversity in the understory was particularly pronounced under ideal conditions for beech trees. The variation in understory species richness correlated with soil stoniness, the C/N ratio of the soil, fine root mass of trees, and their interaction with these variables, as well as canopy openness and structural diversity of the stand (Hrivnák et al., 2022). Notably, competition for soil resources emerged as the most influential factor shaping plant species diversity in managed forests. Paradoxically, promoting optimal growth conditions for beech trees may inadvertently lead to decreased richness of understory plants (Hrivnák et al., 2022).

The dynamics of understory communities in temperate forests are significantly influenced by large herbivores, nitrogen deposition, and forest management. These factors do not operate in isolation but rather form an intricate and intrinsic network of interactions (Koerner et al., 2023; Vild et al., 2024).

Deadwood, with its inherent diversity, harbors remarkably varied communities, often underestimated in empirical studies (Graf et al., 2022). This underestimation stems from two primary factors highlighted by Graf et al. (2022): first, the limited representativeness of species richness due to an absence of correlation between saproxylic species richness and sample size and second, the tendency in species richness assessments to focus solely on obligatory saproxylic species, overlooking facultative saproxylic species.

To address these gaps, Graf et al. (2022) investigated the proportion of obligatory and facultative saproxylic species among Coleoptera, Arachnida, and Heteroptera in a European mountainous forest. Employing rarefaction/extrapolation techniques based on sample coverage estimators, the authors observed significant proportions: obligatory and facultative saproxylic beetles accounted for 28% and 19% of the total beetle species richness, respectively; obligatory saproxylic Heteroptera and Arachnida constituted 4% and 12%, respectively; while facultative saproxylics represented 24% and 69% of the total species richness, respectively.

In summarizing their findings, Graf et al. (2022) noted that deadwood-associated arthropod diversity contributes significantly, accounting for approximately 50%–70% of Coleoptera, Arachnida, and Heteroptera diversity in temperate mountain forests. This revelation exceeds expectations, indicating the profound ecological importance of deadwood habitats.

The long-term response of biotic communities has focused primarily on community composition, functionality, and species richness (Vild et al., 2024). But changes between communities as a whole and habitat types are also important. However, these remain at an underexplored level in terms of biodiversity alteration. Vild et al. (2024) revisited more than 2000 vegetation plots in temperate forests in Central Europe to assess the changes that have occurred over the past five decades. These authors observed that nutrient-poor open-canopy forest habitats were highly reduced in favor of fertile closed-canopy habitats (Vild et al., 2024). These authors concluded that changes in habitat types represent a successive trend of a century with consequences for forest biodiversity. Therefore, open forest habitats, as a matter of urgency, must be restored for plant diversity through more active management implementation (Vild et al., 2024). Repeated community records, at a local scale, based on species richness, are likely to have been burdened with uncontrolled errors when attempting to detect biodiversity trends at larger spatial and temporal scales (Valdez et al., 2023). Attention should be paid to the paradox of biodiversity conservation, or the zero average net change in species diversity (Jandt et al., 2022). Therefore, Vild et al. (2024) took a qualitatively different approach, i.e., independent of species richness.

In parallel, Ette et al. (2023) devised a novel index, the Biodiversity Conservation Index (BCI), integrating ecosystem, species, and genetic diversity metrics to comprehensively assess forest biodiversity. Leveraging data from the Austrian forest inventory and forest typing databases, Ette et al. (2023) employed a machine learning approach, supplemented by ecological modeling in protected areas and unmanaged forests, to develop the BCI. Their findings revealed an average biodiversity classification of 57% for Tyrolean forests, with ecosystem diversity, genetic diversity, and species diversity registering at 49%, 53%, and 71%, respectively. Notably, coniferous forests exhibited a more favorable preservation status compared to deciduous and mixed forests. Moreover, the BCI demonstrated transferability to Central European regions such as Germany and Italy, underscoring its potential utility in guiding regional forest policy and bolstering biodiversity conservation efforts. The BCI is envisioned as a pivotal tool in delineating restoration priorities, enhancing sustainable forest management practices, and optimizing financial

resource allocation toward biodiversity conservation to realize overarching conservation objectives.

4.10 Biodiversity-productivity relationship (BPR)

Regarding the biodiversity-productivity relationship (BPR), Chisholm and Gupta (2023) underscore its potential synergistic role in biodiversity and carbon conservation. However, the understanding of BPR within forests remains comparatively limited due to inherent risks at the global level, compounded by the substantial biodiversity and carbon components therein. Moreover, the majority of studies on BPR in forests are observational in nature, owing to logistical challenges associated with conducting long-term BPR experiments vis-à-vis their counterparts in herbaceous communities (Chisholm and Gupta, 2023).

Chisholm and Gupta (2023) conducted a comprehensive review encompassing experimental and observational studies spanning the last two decades to evaluate forest biodiversity-productivity relationships (BPRs). Their analysis revealed overarching positive support for a forest BPR, indicating a synergistic relationship between biodiversity and carbon conservation to some degree. However, several crucial caveats were identified: (a) higher-yielding forests often comprise monocultures of highly productive species, albeit average productivity may exhibit an increase with biodiversity; (b) positive relative productivity gains (RPGs) can be influenced by a third variable (e.g., stem density) rather than a causal link from biodiversity to productivity; (c) forest productivity reaches its peak with fewer than 10 species; (d) productivity estimations in forests are frequently associated with substantial errors; and (e) the magnitude and signal of the BPR exhibit variability across grains and spatial extents, occasionally exhibiting weakness at scales pertinent to conservation efforts.

In light of these findings, Chisholm and Gupta (2023) advocate for a more refined focus in forest BPR research, emphasizing the accurate estimation of productivity, the exploration of transgressive overyielding phenomena, the perpetuation of long-term observational and experimental studies, the scrutiny of the strength, shape, and signal of BPRs, and the consideration of confounding effects arising from grain and spatial extension. Notably, they underscore the nascent understanding of BPRs within forest ecosystems, suggesting a continued need for investigation and refinement of methodologies to elucidate the intricacies of this relationship.

Chisholm and Gupta (2023) highlight that while some synergy between biodiversity and carbon conservation exists, it may be less robust than indicated by certain forest biodiversity-productivity relationship (BPR) studies. As scientific understanding advances, forest BPR research may eventually justify the cultivation of polycultures comprising more than 12 species, albeit not to the extent observed in natural tropical forests. Similarly, justifying the protection of endangered tree species solely based on their marginal contributions to ecosystem productivity may pose challenges. Consequently, in many forest ecosystems, effective biodiversity and carbon conservation strategies may necessitate complementary policy measures.

In the assessment of biodiversity for use in BPR studies, various metrics such as Shannon's index, effective number of species, species richness, phylogenetic diversity, and functional diversities can be employed (Chisholm and Gupta, 2023). However, the principal challenge in biodiversity assessment within BPR research lies not in a lack of measurement tools, as with productivity, but rather in the clarity of research intentions. Depending on the objectives of BPR research, biodiversity metrics must align with these goals. For conservation-focused applications, biodiversity metrics should prioritize conservation objectives. If the primary aim is carbon conservation with biodiversity cobenefits, metrics such as species richness or phylogenetic diversity may be appropriate. Conversely, if carbon conservation is the primary objective, the choice of biodiversity metric should be guided by its ability to predict productivity effectively (Chisholm and Gupta, 2023).

Until more precise biodiversity-productivity relationship (BPR) results are attained, studies should incorporate a spectrum of biodiversity measurements, encompassing species richness, phylogenetic diversity, functional trait heterogeneity, and species uniformity. Moreover, it would be beneficial for researchers to furnish the raw data underlying these metrics to facilitate the establishment of a comprehensive database. Such a database would serve as a valuable resource for subsequent theoretical investigations, meta-analyses, and conservation applications (Chisholm and Gupta, 2023).

4.11 Genetic mapping

Matthias et al. (2022) conducted a comprehensive examination of the ecological determinants, such as vegetation cover, influencing the genetic integrity of the European wildcat, along with the protective role afforded by designated conservation areas. Leveraging genotypic data obtained from 1217 presumed European wildcats spanning 13 countries across the continent's diverse biomes, the study yielded several key findings. First, the level of genetic integrity surpassed the threshold indicative of feral cat hybridization, standing at approximately 83%, with the designated threshold set at 80%. Second, the Temperate Island biome, notably Scotland, exhibited the lowest level of genetic integrity, recording merely 46%. Importantly, the integrity of forests, including temperate forests, emerged as a significant determinant promoting the genetic integrity of European wildcats.

However, despite the protective measures in place, biomes most susceptible to hybridization demonstrated limited effectiveness in mitigating this threat. Consequently, the formulation of species conservation and management plans necessitates a tailored approach specific to the biome and landscape context, particularly in the Mediterranean and Temperate Island biomes. Furthermore, Matthias et al. (2022) advocate for the adoption of quantitative monitoring methodologies, focusing on the proportion of the genome inherited from parental groups, as a robust indicator of protected areas' efficacy in conserving the genetic inheritance of parental taxa.

The Bialowieza Forest, characterized as one of the rare remnants of primeval forest in Europe, boasts a complex and multiaged structural composition, enriched by the presence of ancient trees that harbor a unique genetic heritage shaped by centuries of evolutionary forces (Wojnicka-Półtorak et al., 2023). Given the invaluable genetic diversity encapsulated within these ancient specimens, it is imperative to undertake a comprehensive genetic inventory and safeguard their genetic resources for posterity. A pioneering study focusing on the genotyping of the oldest (>100 years old) *P. sylvestris* trees within this forest unveiled a remarkable revelation: each of the 98 trees harbored a distinct genotype, thereby facilitating their identification through genetic profiling (Wojnicka-Półtorak et al., 2023). Armed with this foundational knowledge, these ancient trees assume pivotal roles in the conservation of native genetic resources of *P. sylvestris*, serving as genetic benchmarks for comparison with those cultivated in human-managed commercial forests. Given the paramount economic significance of this species in Europe, such genetic standards play a crucial role in guiding conservation efforts and ensuring the preservation of genetic diversity in forest ecosystems (Wojnicka-Półtorak et al., 2023).

5 Global warming

Forests serve as multifunctional ecosystems by sequestering carbon from the atmosphere, enhancing quality of life, and fostering biodiversity. However, selecting the most suitable species for afforestation is crucial, considering that the establishment of a new forest typically spans 50–100 years, underscoring the paramount importance of well-planned and executed initiatives. Amidst the backdrop of global warming, safeguarding extensive and diverse forest landscapes, such as those found in high-altitude regions like the Fagaras Mountains in Romania, assumes critical significance, necessitating the preservation of temporal and spatial structural heterogeneity (Kameniar et al., 2023). The impacts of climate change on tree growth, including biomass production, and alterations in community composition, exert profound effects on forest dynamics and functionality (García-Valdés et al., 2020). As climatic conditions intensify and become more extreme, mitigating the homogenization of forest landscapes emerges as a pressing imperative. Furthermore, expanding protected areas to encompass intricate forest landscapes enables the facilitation of diverse natural processes acting upon various forest vegetation types, thereby bolstering the sustainability of native biodiversity and ecosystem functionality (Kameniar et al., 2023).

The extensive cultivation of rapidly growing, commercially valuable tree species, notably conifers (Fig. 26.8), across Europe over the last 150 years has contributed to a regional warming of approximately 0.12°C (Naudts et al., 2016). This warming phenomenon is primarily attributed to the darker foliage of these species, which absorbs more solar radiation compared to broad-leaved trees such as *Quercus* spp. Consequently, this warming effect is estimated to contribute about 6% to the total global warming attributed to fossil fuel combustion, with similar effects expected in regions where analogous afforestation practices have been adopted. Meteorological records systematically validate localized observations, reflecting broader biome-scale processes.

Notably, structural overshoot phenomena likely and systematically influence low Normalized Difference Vegetation Index (NDVI) events within temperate biomes, leading to warmer temperatures or prolonged growth periods during seasons (Hermann et al., 2023). Moreover, the pervasive warming trend, evident not only in Europe but also globally, has resulted in alterations in leaf coloration, with a significant decrease in greenness (37%) observed in European temperate forests over the study period from 2002 to 2022. This decline is primarily attributed to droughts and intense summer heat (Hermann et al., 2023), signifying forest stress and compromised health, which may portend potential forest mortality.

Furthermore, the exacerbation of this phenomenon in recent years underscores the critical dependence of forest survival not only on prevailing climatic conditions but also on preceding environmental factors, particularly the accumulation of intense and prolonged drought periods occurring 2–3 years before such events. As a consequence, intensified forest stress and mortality are observed not only in temperate forests but also in other forest biomes, emphasizing the urgent need for comprehensive mitigation measures to safeguard global forest ecosystems.

The escalation in air temperature induced by climate change may exert a warming effect on European forest seed banks, thereby favoring the selection of thermophilic communities (Gasperini et al., 2021). Over the long term, this phenomenon could culminate in the emergence of plant species better adapted to heightened light and heat conditions sourced from the seed bank, consequently reshaping vegetation composition. Forests may progressively become enriched in light-demanding generalist species while

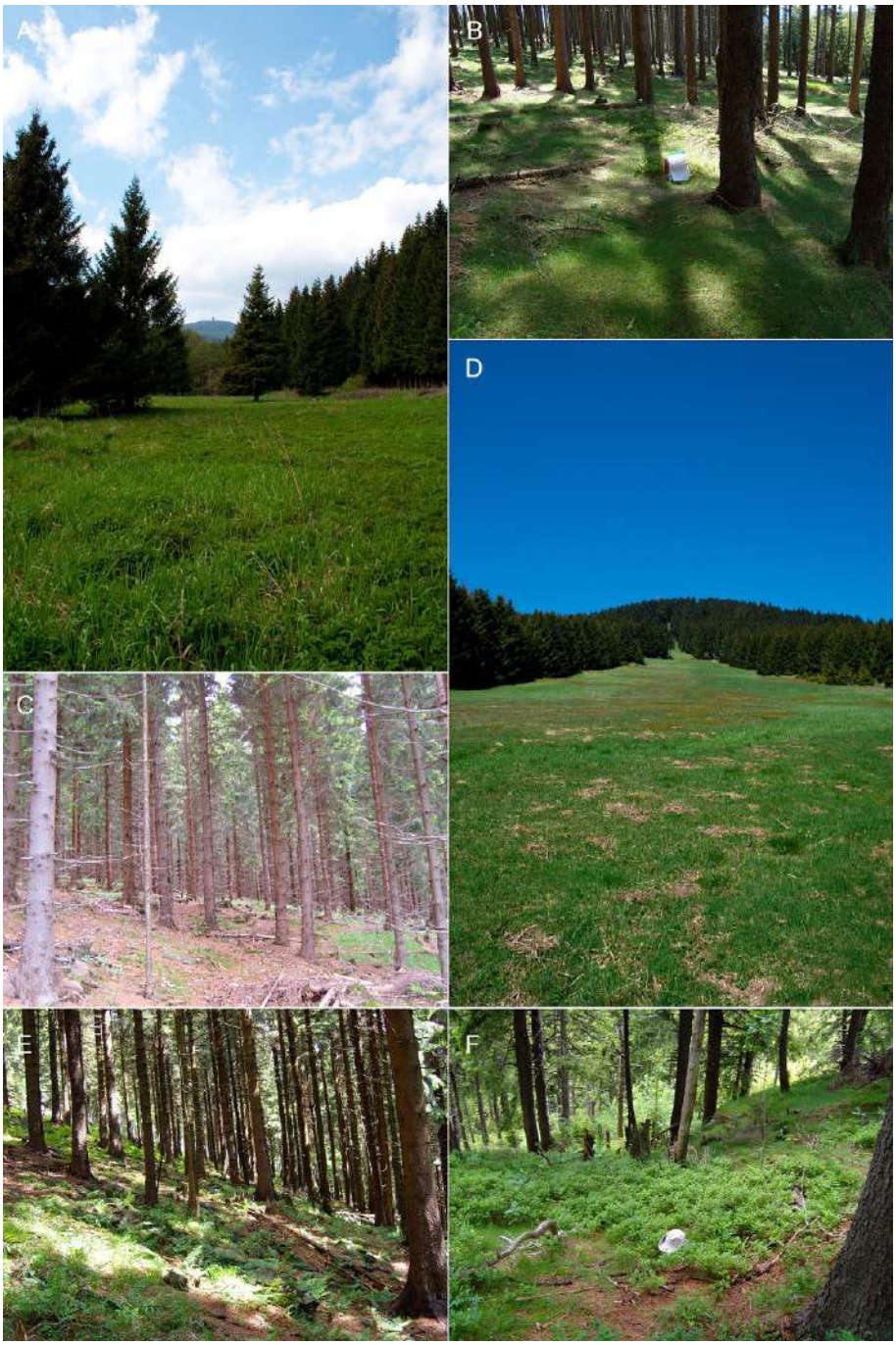

FIG. 26.8 (A) Altitude 580 m, outside spruce forest; (B) altitude 670 m, inside spruce forest; (C) altitude 770 m, inside spruce forest; (D) altitude 770 m, outside spruce forest; (E) altitude 870 m, inside spruce forest; and (F) altitude 970 m, inside spruce forest. Harz Mountains, Braunlage, Germany. *(Photo credits: The author.)*

experiencing a decline in specialist species (Gasperini et al., 2021). Moreover, prevalent natural and cultivated tree species in central Europe, notably *F. sylvatica*, have grappled with heightened occurrences of drought, elevated temperatures, and chronic soil water deficits. Notably, *P. abies* and *P. sylvestris*, which collectively comprise two-thirds of Germany's forest area, exhibit heightened sensitivity to these climatic stressors, with reports of *P. sylvestris* mortality documented in restored areas of the Białowieża Forest (Wojnicka-Półtorak et al., 2023). Although the mechanisms underpinning this phenomenon remain incompletely understood, declining groundwater levels stemming from global warming are posited as a contributing factor (Paluch, 2014).

In response to climate change, forest management strategies may necessitate the introduction of drought-tolerant tree species from temperate regions characterized by more intense summer droughts compared to Central Europe (Bärmann et al., 2023). However, this proposal engenders controversy among ecologists. Moreover, the heightened severity and frequency of wildfires, attributable to climate change, have been documented across various regions worldwide, including the United States, Australia, Amazon, Scandinavia, Spain, and Greece. Climate models project a heightened incidence of fires in the future, necessitating a reevaluation of prevailing forest management policies and conservation strategies to mitigate escalating fire risks (Gutowski et al., 2020). Given the profound impacts of wildfires on forest ecosystems, comprehensive studies are warranted to elucidate their repercussions on local fauna and control communities (Gutowski et al., 2020).

Climate change, compounded by anthropogenic disruptions, facilitates the proliferation of invasive or nonnative alien plant species (IAS) within temperate forests. Natural phenomena such as forest fires, floods, and storms create favorable conditions for the encroachment of exotic plant species into these ecosystems (Langmaier and Lapin, 2020). The proliferation of IAS poses a significant threat to various facets of the environment, including the global economy, local biodiversity, socioeconomic stability, and human health. As such, initiative-taking measures are imperative to mitigate the adverse impacts of IAS on forest ecosystems and associated socio-ecological systems.

6 Conclusions and perspectives

Europe has experienced a greening trend attributed to the proliferation of forest massifs. Various biodiversity techniques and measures have been deployed to conserve and assess the diversity of native forests, encompassing methodologies such as the Normalized Difference Vegetation Index, the development of old-growth indicators, forest condition indices, remote sensing coupled with digital platforms, assessments of tree-related microhabitats, aerial RGB imagery, and applications of deep learning algorithms, among others. These techniques have been refined and have contributed to enhancing the overall diversity of European forests.

Interest has emerged in reintroducing large native mammals and restoring natural processes and ecological complexity, endeavors encapsulated within the framework of rewilding. However, successful implementation necessitates prudent management of population density, with measures such as targeted hunting to mitigate potential impacts on understory ecosystems.

A comprehensive understanding of the ramifications of global warming, including water deficits and heightened wildfire occurrences, holds the potential to enhance predictive capabilities and inform the design of optimal forest management strategies. This may involve preserving seed banks and adapting forest compositions to climate change realities. While the expansion of temperate forests in Europe may have reached its zenith, emphasis is placed on a gradual and strategic transition from existing monocultures, such as conifers, to diversified stands comprising a wider array of species, notably broad-leaved varieties like oak (Fig. 26.9). Such diversification efforts aim to mitigate climate change impacts, as darker-colored coniferous forests tend to absorb more heat, thus contributing to global warming mitigation.

It is imperative, not only in the context of temperate forests in Europe but also globally, to establish limits on the expansion of nonnative forests, as arable land must be prioritized for food production to meet the escalating demand of the present and future, with projections indicating a population of 10 billion people on the planet by 2050. The necessity for new arable land or the rehabilitation of degraded areas is anticipated to equate to the size of two Indias, underscoring the finite nature of available space on our planet.

Efforts toward population demographic control are indispensable, alongside behavioral shifts such as adopting sustainable dietary practices and reducing reliance on globalized supply chains. Intercontinental freight transport, a significant contributor to greenhouse gas emissions, exacerbates dependence on distant products, warranting a transition toward regional self-sufficiency in food production to avert biome destruction in developing continents and safeguard continental food security during times of geopolitical instability.

Reeducation initiatives targeting youth are pivotal to incentivize migration from urban centers to rural areas, fostering agricultural productivity and stemming the abandonment of farms observed in various regions worldwide. Local government investments are essential to support agricultural activities, which are increasingly imperiled by the impacts of global warming. Additionally, revitalizing locally underutilized food resources is crucial for enhancing food security and sustainability.

Implementing these measures can not only safeguard the preservation of rare European “primitive” forests and other ecosystems globally, including tropical forests but also foster greater global stability, mitigating the risk of armed conflicts and promoting harmonious coexistence in a resource-constrained world.

FIG. 26.9 (A) Lichen pine forest in European North (Pomor district forestry, Arkhangelsk forestry, protective forests); (B) Alder forest on the shores of the Łuknajno Lake, near Mikoła, Poland; (C) mixed pine and oak managed forest in the Piska Forest in NE, Poland; and (D) oak and hornbeam forest in the Piska Forest in NE Poland. *(Photo credits: Surina Elena (A) and Monika Mętrak (B—D).)*

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Chapter 27

Temperate forests of North America

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1 Introduction

Found in both hemispheres between 25° and 50° latitude, temperate forests encompass ~10.4 million km^2 (25% of the world's forest habitat) (Tyrrell et al., 2012). In North America, most temperate forests are located in the eastern half of the United States where they originally covered ~2.5 million km^2 (Delcourt and Delcourt, 2000). In most temperate forests, the dominant trees are deciduous; climatic changes in autumn cause a reduction of photoperiod resulting in physiological changes that lead to the cessation of chlorophyll production and the dropping of leaves. Generally, temperate regions are known for seasonal changes in climate (e.g., four distinct seasons) and precipitation is spread throughout the year (Allaby, 2006). North American temperate forests can be generally divided into five distinct regions: (1) deciduous and evergreen forests found throughout the eastern half of the United States and southern Canada, (2) temperate rainforests of the Pacific Northwest that extend from northern California through British Columbia into southern Alaska, (3) mixed hardwood and conifer forest of the intermountain western United States, (4) the temperate Sierras of the southwestern United States and much of northern Mexico, and (5) cottonwood galleries of the Great Plains (Fig. 27.1). However, each temperate forest region contains many smaller ecoregions (see Olson et al., 2001) that contain unique flora and fauna and are immensely important ecologically and economically. Each ecoregion is dominated by a different tree species or community of tree species into which they are categorized.

Temperate forests have provided early human settlements with timber and land for livestock grazing and agriculture for thousands of years. Indeed, humans are still clearing temperate forests for these same resources—timber and land use. The past and current alteration of temperate forests has caused climate and biodiversity emergencies in recent decades including increased atmospheric CO_2 concentrations, increased extremes in temperatures and weather events, and declines in wildlife species (Inkley et al., 2004; Graven et al., 2020; Soto-Navarro et al., 2020). The changing of Earth's climate not only causes negative effects to temperate forests and biodiversity, but also negatively affects human health and well-being via poor air quality, more intense heat waves, increases in infectious diseases and fires, and increased risk of undernutrition (Hoegh-Guldberg et al., 2018; Masson-Delmotte et al., 2022). Both deforestation and forest degradation, which are mostly caused by human activities, account for almost 20% of global greenhouse gas emissions (Cadman et al., 2017). Deforestation and degradation result in habitat loss, which can lead to species extinction and increased contribution to warming that amplifies risk of species extinction (Mackey et al., 2020).

Although in many areas of the world temperate forests are in decline, most North American temperate forests have stabilized or increased in area due to the abandonment of farms and sustainable management of timber (Currie and Bergen, 2008). However, despite this stabilization or expansion, large unbroken tracks of temperate forests are rare and most are highly fragmented, creating a mosaic of forest, human settlements, and agriculture. Today, less than 1% of temperate forests in North America remain unaltered by humans (Reich and Frelich, 2002). Additionally, some portions of temperate forests have been drastically altered compared to their historical state due to invasive species and human development and thus contain a very different plant community than previously existed. As an example, exotic fungal diseases have devastated three north American tree species, the American Chestnut (*Castanea dentata*), butternut (*Juglans cinerea*), and American elm (*Ulmus americana*), which were previously common in their native ranges of eastern North America (Schlarbaum et al., 1998). Additionally, invasive insects such as the hemlock woolly adelgid (*Adelges tsugae*) have greatly decreased populations of hemlock (*Tsuga* spp.), resulting in drastic changes in forest structure in the Appalachians (Orwig et al., 2002). Indeed, human exploitation has also played a role in altering the forest structure. A classic example is found in the southeastern United States in which over 95% of longleaf pine (*Pinus palustris*) has been

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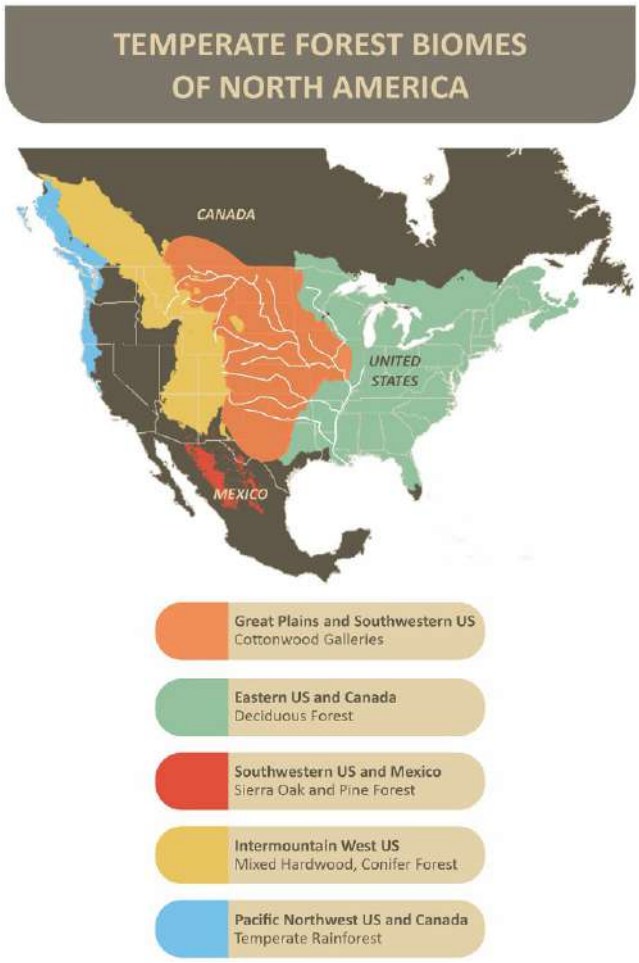

FIG. 27.1 Map depicting locations of the five distinct temperate forest regions: (1) Deciduous forests found throughout the eastern half of the United States and southern Canada, (2) temperate rainforests of the Pacific Northwest, (3) mixed hardwood and conifer forest of the intermountain western United States, (4) the temperate Sierras of the southwestern United States and Mexico, and (5) cottonwood galleries of the Great Plains. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Beth Redlin—USDA.)*

extirpated from its range since European settlement (Oswalt et al., 2012).

Temperate forests are immensely important ecologically, economically, and socially. Other than providing places for human settlements and social activities (e.g., hunting, hiking, camping), temperate forests provide important wildlife habitat, raw materials (e.g., lumber, wood products), act as watersheds, and play a critical role in climate change mitigation (e.g., carbon storage). Additionally, we are just now beginning to understand that many beneficial organisms that provide various ecological services have a dependence on forest habitat. For example, many bees that are important pollinators of crops depend on nearby forests for nesting structures (Ulyshen et al., 2023). Thus, a clear understanding of adequate forest management needs and goals in a changing global environment is a necessity.

2 Geographic location

In this chapter, we define five different temperate forest ecosystems in North America (Fig. 27.1) and describe the common and unique flora and fauna and some of the ecological threats they face.

2.1 Deciduous forests of eastern North America

Covering roughly the eastern half of the United States and southern Canada, temperate forests in this region are primarily defined by a dominating forest structure of deciduous trees. Wide-ranging tree genera are oaks (*Quercus*), pines (*Pinus*), maples (*Acer*), beech (*Fagus*), basswood (*Tilia*), and hickory (*Carya*). However, no tree species is found throughout the entire deciduous forest biome, except for a few early successional species (e.g., willows [*Salix*]) (Braun, 1950).

2.2 Temperate rainforest of the Pacific Northwest

Temperate rainforests are only found in a few places globally but are the dominant forest type within the Pacific Northwest. These forests are dominated by tall old-growth evergreen trees, mostly Sitka spruce (*Picea sitchensis*) and western hemlock (*Tsuga heterophylla*), but other evergreen and some deciduous trees can also be found. Mosses, spike mosses, ferns, and lichen are common along the forest floor and on the trunks of trees.

2.3 Mixed hardwood and conifer forest of the intermountain west

The mixed hardwood and conifer forests are found throughout the Rocky Mountain range from southern Canada into northern Arizona. These forests are dominated by quaking aspen (*Populus tremuloides*) and conifers, including Douglas fir (*Pseudotsuga menziesii*), lodgepole pine (*Pinus contorta*), ponderosa pine (*Pinus ponderosa*), and many others.

2.4 Sierras of the southwestern United States and Mexico

Found primarily in northern Mexico but also in the southwestern United States (Arizona, New Mexico, and Texas), the temperate Sierras are a series of isolated mountain ranges primarily composed of conifers and oaks and are usually bordered by deserts or grasslands.

2.5 Cottonwood galleries of the Great Plains and Southwest

Often overlooked as part of the temperate forest biome, cottonwood (*Populus* spp.) forests or galleries can be found throughout the Great Plains and Southwest along stream and river corridors in these arid and semiarid regions. These forests are restricted to about 1% of the land area within the Great Plains and Southwest (Girard et al., 1989).

3 Biodiversity

Although biodiversity can be measured in a number of ways, it is usually thought of as the assortment of organisms that can be found in one area. Biodiversity is recognized as an essential nonrenewable natural resource (Helm and Hepburn, 2012). Though not as biodiverse as tropical rainforests, temperate forests are rich in biodiversity and contain several vertical layers of vegetation (Fig. 27.2), which in turn drive diverse vertebrate and invertebrate communities. This forest stratification (canopy to forest floor) is immensely important, as it provides a wide array of niches for plants and wildlife such as browse for herbivores and nesting sites for birds. Although temperate forests are categorized by the dominant tree species, numerous endemic plants are often found within the understory. Areas that have high numbers of endemic species are considered biodiversity "hotspots"; North American temperate forests have several examples.

When attempting to understand or compare biodiversity among regions, it is important to consider biodiversity by ecoregions (e.g., oak-pine forests of the southeastern United States) rather than the entire region. Additionally, many microhabitats can exist (even within ecoregions) due to gradients of temperature, moisture, soil type, elevation, etc., which can also be important drivers in determining biodiversity. Temperate forests generally consist of freeze-hardy tree species including pines, oaks, beeches, and maples (Barnes et al., 1997). Here we will discuss the overall biodiversity trends and commonalities for each of the five geographic locations along with the main tree taxa and their importance.

3.1 Deciduous forests of eastern North America

Spanning much of eastern North America, temperate forests in this region make up numerous ecoregions with changing plant communities that are largely driven by latitude and elevation. Tree richness is greatest in the southeastern United States (Barnes et al., 1997) and the tree richness pattern reflects the evolutionary history of broad-leafed trees, the relative newness of continental arctic climates compared to the unglaciated areas of the southeastern United States, and local ecological interactions (Latham and Ricklefs, 1993). Over 110 tree species occupy the eastern United States, 75% of which are deciduous, which is a trait that evolved as an adaptation to cold winters

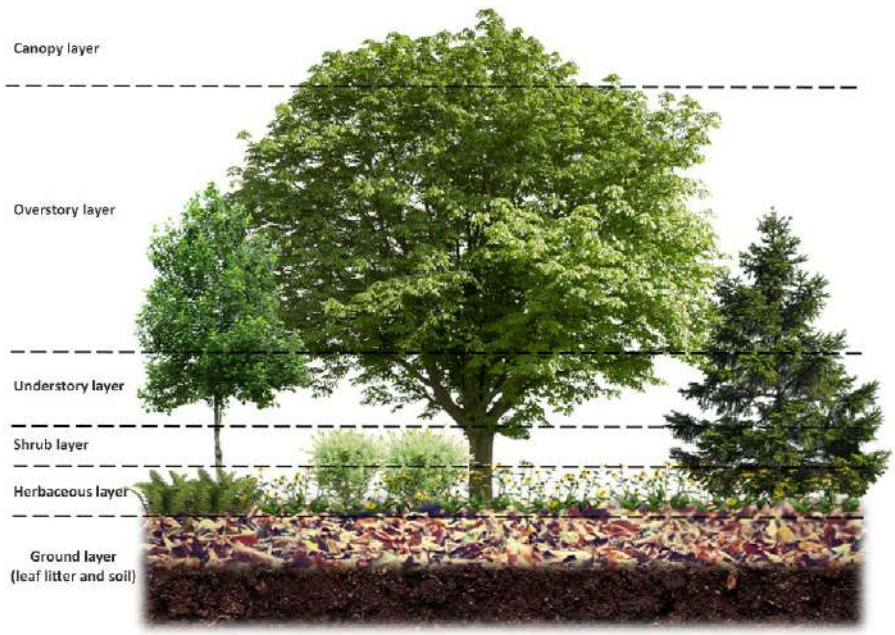

FIG. 27.2 Schematic diagram illustrating the vertical layers of vegetation resulting in numerous niches driving the high biodiversity of the North American temperate forest biome. *(Credit: Alexandra Tsalickis.)*

(Yahner, 1995). Additionally, pines are diverse and are highly adapted to fire and widely distributed on fire-prone sites that are subject to a range of fire conditions (Keeley, 2012). Pines are commonly thought of as drought-tolerant pioneers on nutritionally poor sites (McCune, 1988) and have major economic values as timber, pulp, nuts, resin, and other products. Their ecological importance includes affecting biogeochemical processes, hydrological and fire regimes, and providing food and habitats for animals (Richardson, 2000). Oaks also include fire-resistant species and are the most abundant and widely distributed tree genus in eastern North America (Barnes et al., 1997). They have developed thick bark to protect against fire and if the bole is killed, the root system remains alive, which sends generation after generation of stump sprouts that perpetuate the individual tree and result in oak dominance in the forest (Barnes et al., 1997).

At least 200 species of vertebrates occupy the eastern deciduous forest (Yahner, 1995). These animals have evolved various adaptations that allow them to survive in different weather brought on by seasonal changes. Birds and mammals have higher species numbers in the eastern forests compared to reptiles and amphibians. Most mammal species found here are rodents, insectivores (e.g., shrews and moles), and bats (Kitchings and Walton, 1991). Large mammals common to eastern North American forests are striped skunks (*Mephitis mephitis*), raccoons (*Procyon lotor*), red and gray foxes (*Vulpes vulpes* and *Urocyon cinereoargenteus*), white-tailed deer (*Odocoileus virginianus*), coyotes (*Canis latrans*), and black bears (*Ursus americanus*).

Portions of the eastern deciduous forest host numerous examples of biodiversity hotspots. Hotspots are areas that contain a high number of endemic species and, unfortunately, usually a substantial number of threatened and endangered species. For example, the southern Appalachian Mountains provide refuge for many salamander species found nowhere else in the world (Kitchings and Walton, 1991). The southeastern United States is a global hotspot for aquatic biodiversity containing 40% of the world's freshwater mussels and over 30% of crayfish species (Neves et al., 1997; Richman et al., 2015). Additionally, 79% of North America's freshwater fish species are found in the southeastern United States (Elkins et al., 2019).

3.2 Temperate rainforest of the Pacific Northwest

The temperate forest in the Pacific Northwest is primarily composed of various conifer trees such as Sitka spruce (*P. sitchensis*), western hemlock (*T. heterophylla*), and Douglas fir (*P. menziesii*). Other portions of the temperate rainforest contain fire adapted conifers such as Ponderosa pine, lodgepole pine, and the coast redwood (*Sequoia sempervirens*)—the tallest living trees on Earth. Although conifers are the dominant tree type, there are various deciduous trees [e.g., big-leaf maple (*Acer macrophyllum*), alders (*Alnus*)] that are commonly found along streams and at lower elevations where there is more moisture. A unique feature of the temperate rainforest is the epiphytic bryophytes (mosses and liverworts) that thrive within the understory in areas that receive ample rainfall. These epiphytes are very important ecologically where they act as forest nutrient recyclers, offer food and shelter for numerous invertebrates, and provide nesting material for many birds. Additionally, the main source of commercially harvested moss in North America comes from the temperate rainforest (Peck, 2006).

Unique vertebrates that inhabit the temperate rainforest include black and grizzly bears (*Ursus arctos horribilis*), wolves (*Canis lupus*), mountain goats (*Oreamnos americanus*), black-tailed deer (*Odocoileus hemionus columbianus*), Roosevelt elk (*Cervus canadensis roosevelti*), and endemics such as the mountain beaver (*Aplodontia rufa*) and Douglas squirrels (*Tamiasciurus douglasii*). Notable endangered bird species such as the northern spotted owl (*Strix occidentalis caurina*) and the marbled murrelet (*Brachyramphus marmoratus*) occupy this region as they both require old-growth forests for nesting.

3.3 Mixed hardwood and conifer forest of the intermountain west

The quaking aspen tree (*P. tremuloides*) is a keystone species and major vegetation cover type in the intermountain west. Quaking aspen is the most widely distributed tree species in North America and has tremendous ecological amplitude (Campbell and Bartos, 2001). Aspen groves in the intermountain west are second to riparian forests in supporting the greatest biodiversity (Rogers, 2019). Aspen undergrowth consists of lush and diverse flora which significantly contributes to biodiversity (Rogers, 2019). At a landscape scale, patches of multiaged aspen groves provide other forest landscape benefits such as helping with fire resistance.

At the higher elevations, conifer trees predominate the western United States due to the climatic conditions of the region—arid with reduced precipitation. Coniferous trees can outcompete deciduous trees in these harsher climates, particularly in early growth stages (Yahner, 1995). Common conifers include Douglas fir (*P. menziesii*), subalpine fir (*Abies lasiocarpa*), Engelmann spruce (*Picea engelmannii*), and lodgepole pine (*P. contorta*).

Large mammals commonly found within these forests include mule deer (*Odocoileus hemionus*), mountain lion (*Puma concolor*), elk (*Cervus elaphus*), coyote, bighorn sheep (*Ovis canadensis*), moose (*Alces alces*), and black bears. Numerous birds can be found in these mixed

hardwood and conifer forests but due to the higher elevation and colder temperatures, reptiles and amphibians are not as diverse compared to other temperate forest types.

3.4 Sierras of the southwestern United States and Mexico

The temperate Sierras are dominated by pine-oak forests that grow at elevations between 1000 and 3500 m. Portions of this biome are considered a transition zone between Nearctic and Neotropical regions and much of this region is considered a global biodiversity hotspot (Van Devender et al., 2012). Plant diversity is extremely high with thousands of species found here, many of which are endemic (González-Elizondo et al., 2013). Numerous pine and oak species can be found within the temperate Sierras and the species present are largely driven by elevation, physiography, and local climate. Other common trees and shrubs include madrones (*Arbutus*) and junipers (*Juniperus*).

Wildlife found in the temperate Sierras is vastly different compared to the other North American temperate forests. Although species like white-tailed deer, black bear, mountain lions, and many other mammals can be found in this region, other mammal species are endemic and only found within this type of temperate forests. Some examples include jaguar (*Panthera onca*), Mexican wolf (*Canis lupus baileyi*), and white-nosed coati (*Nasua narica*). Numerous endemic birds can be found within this region such as the thick-billed parrot (*Rhynchopsitta pachyrhyncha*) and eared quetzal (*Euptilotis neoxenus*). Additionally, many of the reptiles and amphibians are also considered endemic to this ecoregion. Historically, the imperial woodpecker (*Campephilus imperialis*), the largest woodpecker in the world, and the Mexican grizzly bear (*U. arctos horribilis*) inhabited this region, but both are presumed extinct.

3.5 Cottonwood galleries of the Great Plains and southwest

The deciduous cottonwood forests are the most extensive and ecologically important forest ecosystems in arid/semiarid regions of the western United States (Lytle and Merritt, 2004). Restricted to stream and river corridors and drainages, cottonwood forests often have an understory composed of smaller trees such as willows and box elder (*Acer negundo*). However, within many riparian areas of the west, cottonwood forests are being replaced with invasive trees and shrubs such as Russian olive (*Elaeagnus angustifolia*), Siberian elms (*Ulmus pumila*), and tamarisk (*Tamarisk* spp.) (Perry et al., 2018).

Despite the relatively small area occupied by these riparian forests, they are known to contain high levels of biodiversity compared to the surrounding landscape (primarily prairie or desert). Many wildlife species are dependent on these forest corridors such as migrating passerine birds which rely heavily on these riparian forests for food, shelter, and breeding areas. Large ungulates such as moose, elk, mule deer, pronghorn (*Antilocapra americana*), and domestic cattle (*Bos taurus*) often utilize these areas for shelter, food, and water. Additionally, reptiles, amphibians, and small mammals are commonly found within these forests.

4 Threats to biodiversity in North America

As global temperatures are warming due to anthropogenic climate change, flora and fauna adapted to their local environmental conditions increasingly face challenges. The resulting increased climate variation interacts with other factors such as pollution, ozone depletion, urbanization, etc., which will amplify the effect on wildlife in North America (Inkley et al., 2004). A large number of scientific publications have indicated that rising temperatures are putting substantial strain on natural and managed systems in North America (IPCC (Intergovernmental Panel on Climate Change), 2022). Particularly vulnerable species will be those that are dispersal-limited as climatic change will occur at a higher rate than most species can expand their range (IPCC (Intergovernmental Panel on Climate Change), 2022). Changes in species ranges have already been observed as a result of changing climatic parameters such as temperature and precipitation (Stendel et al., 2021). Other studies have found northward shifts of bird species in North America (Rushing et al., 2020; Niven et al., 2010). Even when the adaptive capacity of a species allows range shifts in response to environmental disturbance or climatic changes, given the acceleration of climatic shift, as well as other factors such as land use changes and urbanization, species may not be able to shift their ranges quickly enough and may suffer local extirpations (Rose et al., 2023).

The northward shift of species ranges is not unique to wildlife. Trees and other vegetation also struggle to adapt to the warming temperatures and have begun to shift north in North America (Boisvert-Marsh and de Blois, 2021). Decreases in the frequency and intensity of extreme cold events facilitates an overall transformation of temperate ecosystems by poleward-moving tropical organisms of both plants and animals in response to warming temperatures, often referred to as tropicalization (Osland et al., 2021). Additionally, deforestation and degradation result in habitat loss that is a major cause of species extinctions and contributes to warming, which in turn amplifies risk of species extinction (Mackey et al., 2020). Although climate change and decreasing biodiversity are often seen as two separate problems (and often treated as such), they are intricately connected, and consequently an integrated approach needs

to be developed in order to combat these issues. Despite opportunities to preserve biodiversity within temperate forests, scientists need to better elucidate how biodiversity is valuable economically and socially and develop ways to have this value returned to the people managing the land.

5 Conservation areas and their ecological indicators

Forest ecological indicators include organisms that are sensitive to environmental changes and their presence is indicative of forest conditions. Numerous ecological indicators have been utilized for forested habitats. These include many arthropods such as bees, ants, beetles, etc. (Campbell et al., 2018, 2019) but can also include small mammals (Klenner and Sullivan, 2009), birds (Canterbury et al., 2000), and lichens (Thormann, 2006). Thus, the chosen ecological indicator should be best suited to measure the specific aspect of forest health that is desired (e.g., soil health, forest management practices, air pollution, etc.).

Biodiversity is not distributed randomly, and thus conservation areas that should be prioritized for North America's temperate forests should be biodiversity hotspots (Noss et al., 2015). Tropical rainforests are typically the first thought when considering biodiversity hotspots; however, many exist in North America's temperate forests. Although some biodiversity hotspots have already been briefly discussed, below are a few that have received little attention.

6 Biodiversity hotspots in North America

6.1 North American Coastal Plain

The North American Coastal Plain (NACP) is a coastal forested wetland that has a large number of endemic and endangered species (White et al., 2022) and >70% habitat loss (Noss et al., 2015) qualifying it as a biodiversity hotspot according to the classic definition by Myers et al. (2000). The forested swamps and bogs of the NACP provide substantial ecosystem services including habitat provisioning, removal of excess nutrients, attenuating storm surges, and sequestering large amounts of carbon (Engle, 2011). Collectively, the ecosystem services of the NACP are valued at US $1.5 trillion per year globally (Costanza et al., 2014). Unfortunately, between 1996 and 2016, 19,480 km^2 of the coastal wetlands in the NACP were deforested mostly due to agricultural conversion (White et al., 2022). However, despite this large amount of deforestation, the NACP is still spatially larger than other hotspots around the globe (e.g., the Mediterranean Basin of 2.09 million km^2). Species are still being discovered in the NACP, with more than 400 new endemic vascular plants and 6%–7% of the total flora named over the past half-century in the southeastern United States (Noss et al., 2015).

6.2 California Floristic Province

Another biodiversity hotspot in North America is the California Floristic Province (CFP) (Calsbeek et al., 2003; Burge et al., 2016) and it is considered one of the world's top 25 most biologically rich and endangered terrestrial ecosystems (Myers et al., 2000). The CFP is an area with high plant diversity and endemism and exhibits a Mediterranean-type climate that extends to southern Oregon, western Nevada, and northern Baja California, Mexico (Raven and Axelrod, 1978). A total of 5006 species native to the CFP region exist with an estimate of 36.9 endemism (Burge et al., 2016). Compared to other areas of North America with high endemic plant and animal taxa, the CFP is the largest and has more identifiable subspecies than the central and northeastern United States and Canada combined (Calsbeek et al., 2003). Due to climate and land use change, the CFP is experiencing unique pressure (Rose et al., 2023) as temperatures are projected to increase by 2–7°C and land development is projected to more than double through the year 2100 (Pierce et al., 2018; Sleeter et al., 2017).

6.3 Sierra Madre Occidental of Mexico

Part of the Sierra temperate forest, the Sierra Madre Occidental is a mountain range with extensive pine-oak forests that extends northward to southeastern Arizona and south to the Mexican state of Jalisco. This extensive forest contains the greatest species diversity of pine and oak species in the world (Rzedowski, 1993). Within this region, disjunct patches of plant communities (termed "sky islands") exist which harbor high biodiversity and endemism. These sky islands developed due to various climatic changes over time which created a patchy network of montane vegetation, thus driving unique species assemblages (McCormack et al., 2008).

7 Human activities and their impacts

Humans have relied on and played large ecological roles within temperate forests of North America for thousands of years. However, with the colonization by Europeans and the advent of modern machinery, large-scale exploitation of forests became commonplace. Thus, roughly the beginning of the 20th century is when governments realized that conservation measures and practices would need to be developed to reduce forest destruction and humans would need to learn how to sustainably manage these natural resources. Below are some of the negative and positive aspects of humans that have altered temperate forests historically and in modern times.

7.1 Negative impacts of humans

7.1.1 European colonization—The beginning of anthropogenic change

When European settlers first arrived in North America in the 1500s, there was high demand for natural resources to survive. Once pristine temperate forest habitats were exploited for natural resources based on the needs of the first colonizing settlers. Early modern colonialism was unique to other kinds of human-ecosystem relationships due to the rapid and high-intensity environmental modifications (Richards, 2003; Lightfoot et al., 2013). Once European colonization began expanding across eastern North America, forests were replaced with various human settlements and agricultural plantations. Thus, the current temperate forest (especially the eastern deciduous forest) is highly fragmented. Although logging and other exploitations of forests are today highly regulated within the United States and Canada, illegal logging and land clearance for grazing lands continue to be a major threat to the temperate Sierra forests in Mexico (González-Elizondo et al., 2013).

7.1.2 Introduced pathogens

Accidentally introduced pathogens have greatly altered temperate forests. The most destructive pathogens have been various insects and fungal diseases (which often are carried by insects). While many pathogens altered forest ecosystems long ago, such as the American chestnut blight which began in the early 1900s, other pathogens are more recent and the true cost of the pathogen will not be known for some time. An example of a new pathogen affecting temperate forests is laurel wilt disease, which is a fungal disease vectored by a nonnative bark beetle (*Xyleborus glabratus*). This disease was first found in the early 2000s and primarily affects red bay trees (*Persea borbonia*), which is a common tree found in portions of the NACP.

7.1.3 Anthropogenic fire regime changes

In North American forests, fire plays a large role in shaping the ecosystem as fire leads to unique vegetation patterns that are different than would be controlled by climate alone. In eastern North America, pre-European settlement fire regimes consisted of low to mixed severity forest surface burns resulting in oak and pine dominated forests which have declined in size compared to the present (Sommers et al., 2011). With the onset of European settlement in the 1700s–1800s, forests were not only cut and burned, but the fire frequency and severity increased due to agricultural land clearing, logging, and accidental ignition leading to major shifts in forest composition. During the 1920s, fire suppression policies took place which changed fire regimes in the eastern United States (Nowacki and Abrams, 2008). Open lands such as grasslands, savannas, and woodlands transformed into closed canopy forests and fire-dependent plants became replaced by shade-tolerant fire sensitive plants (Nowacki and Abrams, 2008).

While wildfires have decreased in the eastern deciduous forests due to proper forest management practices, the western United States has experienced increases in forest fire activity in recent decades (Abatzoglou and Williams, 2016) due to anthropogenically induced warmer and drier conditions (Hicke et al., 2022). Specific measures of increased fire activity include the area burned (Littell et al., 2009; Williams and Abatzoglou, 2016), the number of large fires, and the fire season length (Westerling et al., 2006). Combined, these factors contribute to widespread forest mortality, carbon emissions, periods of degraded air quality, and substantial fire suppression expenditures (Abatzoglou and Williams, 2016). Some of the significant variables which facilitated the increase in fire-season fuel aridity is the observed warming and drying of the climate (Abatzoglou and Williams, 2016) and decreased precipitation (Hicke et al., 2022), which provided favorable conditions for fire. Evidence showing that climate change has contributed to the recent increase in fire activity includes observed increases in vapor pressure deficit, fire indices, and climatic water deficit over the past several decades (Abatzoglou and Williams, 2016). Climate predictions expect the burned area in western North America to either double or triple by the mid-century (McKenzie et al., 2004; Westerling et al., 2011).

7.2 Positive impacts of humans

7.2.1 Modern forest management

A wealth of knowledge has been accumulated on forest management—much of it driven from our past mistakes. Today, in the United States, the use of proper forest management strategies has led to the sustainable use of forested habitat for various consumable goods (timber products) and recreation. Many forest management techniques are aimed at reducing wildfire and include the use of prescribed fires, understory thinning, and limited herbicide use to reduce forest fuels and invasive plants. Additionally, regulated and licensed hunting has created sustainable game populations and revenues from hunting licenses and ammunition are used to expand and maintain wildlife refuges.

8 Operational and potential monitoring tools to conserve or to recover the biome

There is growing recognition that it is necessary to integrate climate and biodiversity policy agendas at global and national scales (Soto-Navarro et al., 2020). Law et al. (2021) found that strategically increasing the extent of forest protection and forest management (e.g., land

preservation, reduced harvest) would help safeguard climate, biodiversity, and drinking water in the western United States. Additionally, forest protection is crucial to preventing deforestation and forest degradation, reducing greenhouse gas emissions, and maintaining large carbon sinks (Law et al., 2021). Not only would preserving high priority temperate forests in North America increase the amount of protected habitat for animal and plant species, but preservation also would increase landscape connectivity to help maintain biodiversity and ecological functions for climate adaptation (Geldmann et al., 2013).

Historically, forests have often been managed as single entities with treatment (e.g., prescribed burns) applied over large areas. Changes in forest dynamics over time may be missed with this approach; hence, new technological advances have allowed researchers to detect changes at much finer scales and accomplish long-term continuous monitoring. Long-term monitoring can be conducted at the individual plant or site level via in situ sensors and various camera equipment that can measure factors such as tree growth, evapotranspiration, leaf temperature, phenological changes, and soil conditions (Torresan et al., 2021). Additionally, many new or advanced tools can give biodiversity measurements continuously rather than discrete intervals. For example, utilizing advanced digital cameras over wide areas has been successful in detecting animals of interest or conservation concern (Delisle et al., 2021). Other successful technologies utilized for biodiversity measurements include acoustic detectors (Hill et al., 2018) and remote sensing (Pettorelli et al., 2018).

One monitoring tool that is not often utilized or has not garnered a lot of attention by forest managers is the use of various paleoenvironmental archives. Archives are used to extract, reconstruct, and interpret past environmental conditions. Forest managers can use archives to understand past forest changes and help predict how future climatic events may impact forest structure. Archive types include sediment cores, ice cores, speleothems, and tree rings. One relatively novel archive type used to understand past changes in forest structure is bat guano cores. An example is the use of bat guano cores from Alabama in the United States. Bat guano from Fern Cave and Cave Springs Cave, both located in northern Alabama, used pollen and stable isotopes ($\delta^{13}C$, $\delta^{15}N$, $\delta^{2}H$) to understand climate variability and changes in forest structure (Campbell et al., 2017; Tsalickis et al., 2022).

8.1 Case study of paleoenvironmental archives—Bat guano reveals the past temperate forest vegetation change

A bat guano core collected from Cave Springs Cave, Alabama, United States, was ^{14}C dated to ~12,000 cal yr BP (Fig. 27.3A). $\delta^{2}H$ stable isotopes in paleo studies are typically used to determine hydroclimate as $\delta^{2}H$ is sourced from atmospheric and rainfall water. $\delta^{2}H$ values were sampled from each centimeter of the core and interpreted to infer wet versus dry climatic conditions over a 12,000-year period. To determine how the forest composition changed in response to hydroclimatic variability, the $\delta^{2}H$ guano values were compared to a pollen record from the same region that spanned the same age period. The $\delta^{2}H$ values in this study indicated three shifts in moisture throughout the core: (1) a dry period in the early Holocene/late

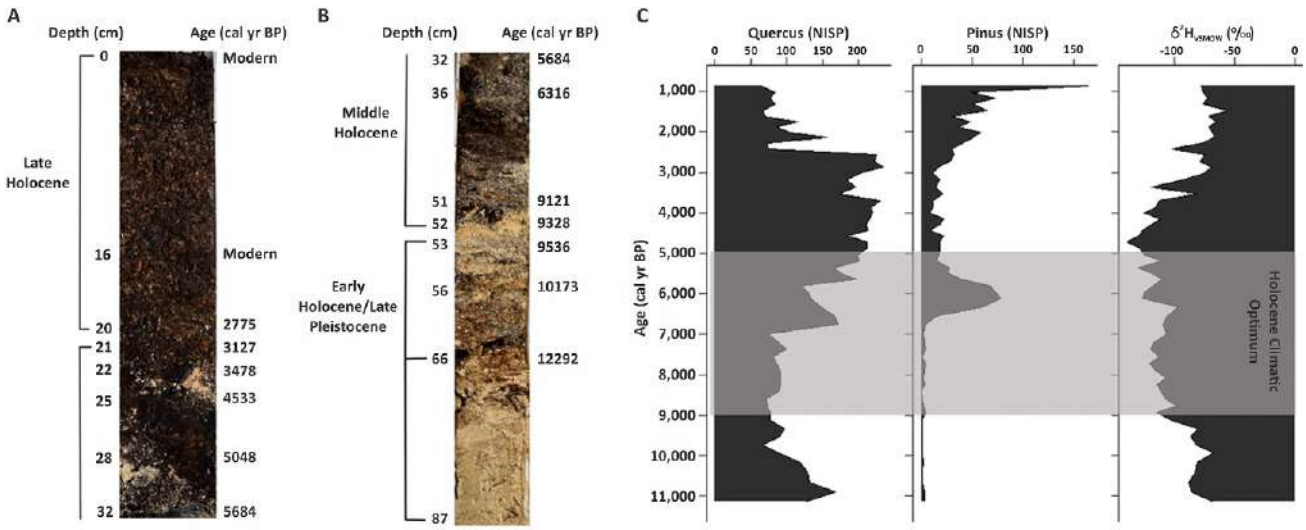

FIG. 27.3 A and B show the top and bottom portions (respectively) of the collected bat guano core from Cave Springs Cave with associated depths and ages (Tsalickis et al., 2022). Graph C indicates the number of individual specimens (NISP) of oak (*Quercus*) and pine (*Pinus*) pollen throughout the Holocene. The oak and pine pollen records were collected from small ponds in Bartow County, Georgia, United States (Watts, 1970). *(Credit: Alexandra Tsalickis.)*

Pleistocene, (2) a wet period during the middle Holocene, and (3) another dry period during the late Holocene. Each of these shifts in moisture coincide with shifts in the dominant tree taxa in the area. The early Holocene/late Pleistocene was oak dominant during a dry period while pine tree species began to increase in the middle Holocene when environmental conditions became wetter. Finally, another shift to oak dominance occurred in the forest when environmental conditions became dry again (Fig. 27.3B). Using bat guano as a paleoenvironmental archive revealed forest composition shifts throughout the Holocene. Understanding how the past hydroclimate altered forest composition will allow forest managers to develop methods and prepare for future hydroclimate changes.

9 Conclusions

North American temperate forests are quite variable in their location as well as their flora and fauna. Human influence also plays a large role in shaping forest structures in many ways including anthropogenic global warming, urbanization, deforestation, and altering fire regimes. Although human-induced climate change will continue to negatively impact temperate forests, new technologies may help lessen the impact of climate change. Additionally, an increased knowledge of the vast importance of temperate forests to the livelihood of humans will lead to a greater understanding of the importance of conservation measures. This chapter has taken a broader view of the North American temperate forest biome, considerations of which have often only included the eastern deciduous and temperate rainforests of the Pacific Northwest. We hope that also including the often overlooked portions of temperate forests such as the Sierras of Mexico and cottonwood galleries of the Great Plains will broaden the conservation strategies in regard to temperate forests.

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Section J

Tropical forests

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Chapter 28

Conservation status of dung beetles (Scarabaeidae: Scarabaeinae) in African tropical forests

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1 Introduction

Tropical forests in Africa play a critically important role in preserving unique species and biodiversity communities, as well as influencing climate change through their vital role in carbon sinking and storage. These forests also have a significant impact on weather patterns across the continent (Abernethy et al., 2016; Cuni-Sanchez et al., 2021; Bennett et al., 2021). African tropical forests are known for their high levels of endemism, making them crucial in preserving biodiversity on a global scale (Abernethy et al., 2016; Cuni-Sanchez et al., 2021; Bennett et al., 2021). Furthermore, these forest ecosystems serve as important water reservoirs, often situated at the headwater of key river systems such as the Congo and the Nile. They effectively regulate the timing and volume of runoff, influencing water availability downstream (Cuni-Sanchez et al., 2021). Notably, African tropical forests encompass both lowland forests and montane forests, primarily concentrated in the Congo Basin and surrounding higher lands (Fig. 28.1). However, there are also outliers existing in a narrow coastal area of West Africa, in the Eastern Arc Mountains of East Africa, and in inselbergs in southern Africa (Lewis et al., 2013; Tranquilli et al., 2014; Willcock et al., 2014; Cuni-Sanchez et al., 2021). Overall, the intricate role of African tropical forests in shaping weather patterns, safeguarding unique species, and providing various ecosystem services highlights their indispensable importance in conservation and sustainable land management efforts across the continent.

2 Biodiversity

The African tropical forest is an ecosystem that supports a diverse range of endemic flora and fauna (Fig. 28.2), comprising only just over 30 protected areas, albeit dense human populations, often resulting from immigration, constituted major threats to this biome in Africa (Struhsaker et al., 2005). It is estimated to be home to around 10,000 plant species, 1000 bird species, 700 fish species, and 400 mammal species, including renowned animals like forest elephants, lowland gorillas, and chimpanzees (Slik et al., 2015; Huntley, 2023). The forest also harbors thousands of insect species, including approximately 116 species of dung beetles (Hanski and Cambefort, 1991). Unfortunately, these forests have been experiencing significant loss, which poses a threat to global insect diversity. Tropical forests are home to the majority of insect species, including dung beetles, and the annual destruction rate of tropical forests stands at 0.5%–1% (FAO, 2001; Pimm et al., 2001; Hanski, 2005).

Dung beetles are one of the dominant invertebrates in tropical forests and are essential for a functioning ecosystem, as they feed primarily on mammal dung, but are also capable of using carrion, rotting fruit, fungi, and decaying plant matter. As a result of manipulating these during the feeding process, dung beetles provide a series of ecosystem functions including but not limited to dung removal, secondary seed dispersal, nutrient cycling, and parasite suppression. Many of these ecological functions provide valuable ecosystem services such as biological pest control, soil fertilization, and reduction of greenhouse (Nichols et al., 2008; Scholtz et al., 2009; Raine and Slade, 2019; Andresen and Urrea-Galeano, 2022). In African tropical forests, dung beetles are classified into four main functional groups based on their feeding behavior: (a) tunnelers or paracoprids, which make vertical tunnels under the dung pile, (b) rollers or telecoprids, which roll a portion of dung horizontally and after a certain distance bury it, (c) dwellers or endocoprids, which do not relocate any portion of the dung pile and live inside it or in the dung-

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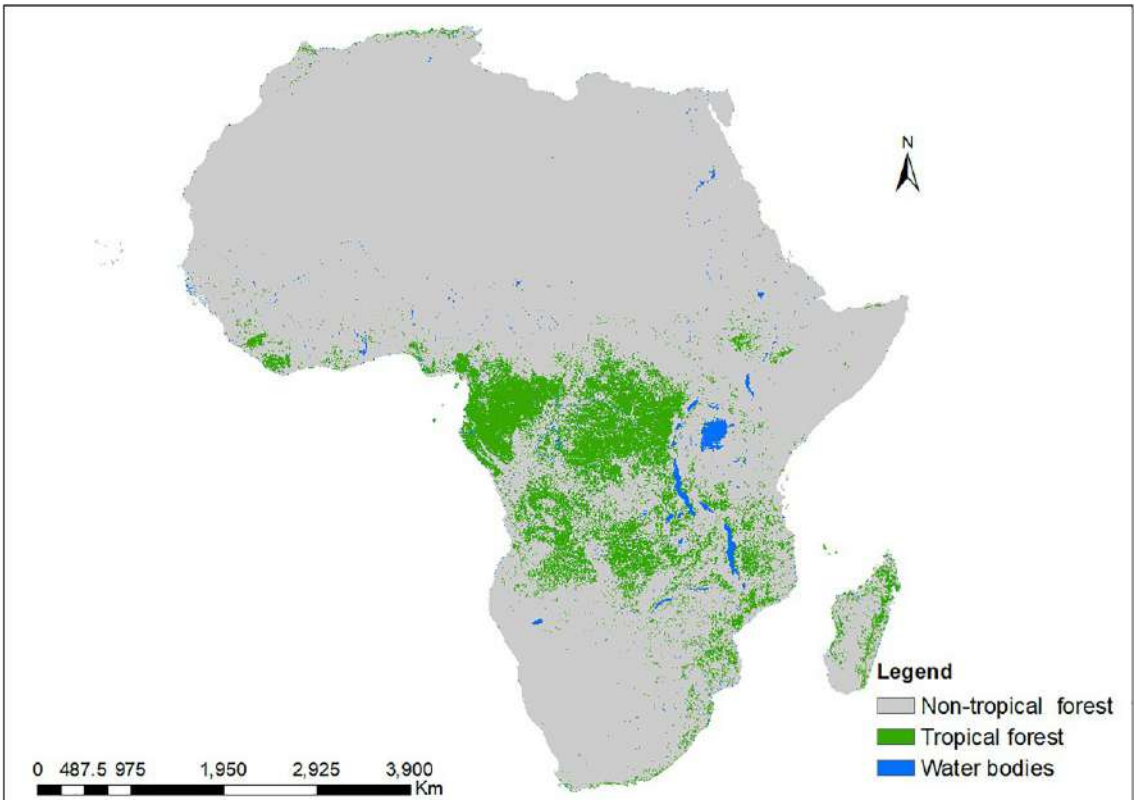

FIG. 28.1 A distribution map representing African tropical forests. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Source: Authors.)*

FIG. 28.2 Forest elephant in Dzanga-Sangha reserve in the Central African Republic. *(Credits: https://discover-afrika.com/le-fascinant-elephant-de-foret/dzanga-sangha8/.)*

soil interface, and (d) kleptoparasites or kleptocoprids, which parasitize or steal nests of other beetles, mainly paracoprids and telecoprids (Bornemissza, 1976; Cambefort and Walter, 1991). The first comprehensive review of dung beetles from African tropical forests was conducted by Cambefort and Walter (1991), including subsequent studies such as Davis and Philips (2005, 2009), Mongyeh et al. (2018), Stanbrook et al. (2021), Kamgaing et al. (2022), Philips et al. (2023), and Daniel et al. (2023a,b). Thus, African dung beetles are particularly interesting because they thrive in areas with a diverse range of mammals and sufficient tree cover. These factors greatly influence the

composition of the dung beetle communities in these forests. There are two main types of dung beetles found in African forests: those that feed on the small droppings of omnivorous mammals, and those that exploit the large droppings of elephants and other large herbivores (Hanski and Cambefort, 1991). While the first type of species can be found in other tropical regions, the second type is unique to Africa.

This biome is home to approximately more than 116 species of dung beetles, some of which are exclusive to forest habitats (Hanski and Cambefort, 1991). In this chapter, we review the latest knowledge of the state of the art of dung beetles from the African tropical forest biome, threats, and conservation status, including study cases of how human disturbance to this biome affects the dung beetle communities.

3 Potential threats to African tropical forests

The tropical forests in Africa, such as those in the Itombwe Mountains in the Democratic Republic of the Congo and West African forests, have been and continue to be impacted by various activities including logging, mining, agricultural conversion, forestry, hunting, urbanization, human population growth, political unrest, and the presence of militias (Gyasi et al., 1995; Estrada and Coates-Estrada, 2002; Fa et al., 2005; Mongyeh et al., 2018; Cuni-Sanchez et al., 2021). These activities result in changes to the forests, including deforestation, fragmentation, and the replacement of forest trees with other vegetation types such as plantations. This in turn affects the distribution of certain mammal species and their dung types, which has a strong influence on the geographical occurrence of dung beetles specialized to those dung types (Davis et al., 2002).

Across a gradient of forest modification, from intact tropical forests to clear-cut areas and isolated forest fragments, there is a gradual decline in dung beetle species richness, abundance, and biomass (Lhoest et al., 2020; Stanbrook et al., 2021). Protected areas tend to have a higher number of large-bodied beetles compared to logged forests, community forests, and plantations (Hanski and Cambefort, 1991; Lhoest et al., 2020; Stanbrook et al., 2021). Studies have shown that the richness and abundance of different types of dung beetles are affected by anthropogenic activities. For example, anthropogenic disturbances like plantations and logging result in a higher abundance of dwellers, while rollers, tunnelers, and kleptoparasites decrease in richness and abundance with increasing anthropogenic impact (Hanski and Cambefort, 1991; Gebert et al., 2020; Stanbrook et al., 2021). These variations in dung beetle communities are influenced by factors such as mammal diversity, light intensity, the presence of understory herbs and shrubs, and underground cover (Lhoest et al., 2020).

4 The ecological implication of dung beetles in African tropical forests

Dung beetles are known to be highly sensitive to changes in temperature and humidity (Roslin and Viljanen, 2011), which change as the physical patterns of a forest change, especially from forested to open habitats, leading to variations in dung beetle communities (Hanski and Cambefort, 1991). However, despite their significant diversity, essential ecological roles, and global endangered status, there is limited research on the impact of primary forest conversion on dung beetle biodiversity and ecological functions (Cardoso and Leather, 2019; Wagner et al., 2021). Dung beetles have been widely recognized as a suitable invertebrate taxon for assessing global patterns associated with forest conversion, given their sensitivity to anthropogenic disturbances and their numerous ecological functions (Bicknell et al., 2014; Fuzessy et al., 2021; López-Bedoya et al., 2021; Nichols et al., 2008; Otavo et al., 2013; Slade et al., 2011; Spector, 2006). These functions include the burial of excrement, nutrient cycling, soil aeration, filtration capacity, secondary seed dispersal, parasite and fly control, and even the mitigation of greenhouse gas effects (Fig. 28.3, Nichols et al., 2008; Slade et al., 2016). According to López Bedoya et al. (2022), the conservation of primary forests is crucial for the preservation of dung beetle diversity and ecological functions. Unfortunately, human activities pose significant threats to forests and their fauna, including dung beetles. Researchers have found negative tendencies in the forest conversion impact on species like *Dichotomius* and *Onthophagus*, while *Copris* and *Sisyphus* have shown neutral or slightly positive correlations. Furthermore, the effects on tunnelers have been observed to be more severe compared to rollers (López Bedoya et al., 2022).

5 Human activities and their impacts

It is widely acknowledged in the scientific literature that human population growth and associated activities have emerged as the primary drivers of global environmental threats, thereby posing significant challenges to biodiversity (Tranquilli et al., 2014; Fuzessy et al., 2021). The adverse impacts of human activities on tropical forests, such as deforestation, forest degradation, and defaunation, are especially noteworthy (Abernethy et al., 2016). These activities have a profound influence on dung beetle communities, as they contribute to habitat loss and the decline in large mammal populations, which are crucial factors determining the structure and composition of dung beetle assemblages (Davis and Philips, 2005, 2009; Nyeko, 2009; Fuzessy et al., 2021). Various key drivers of deforestation in tropical Africa have been identified, including shifting agriculture, firewood production, timber logging, and

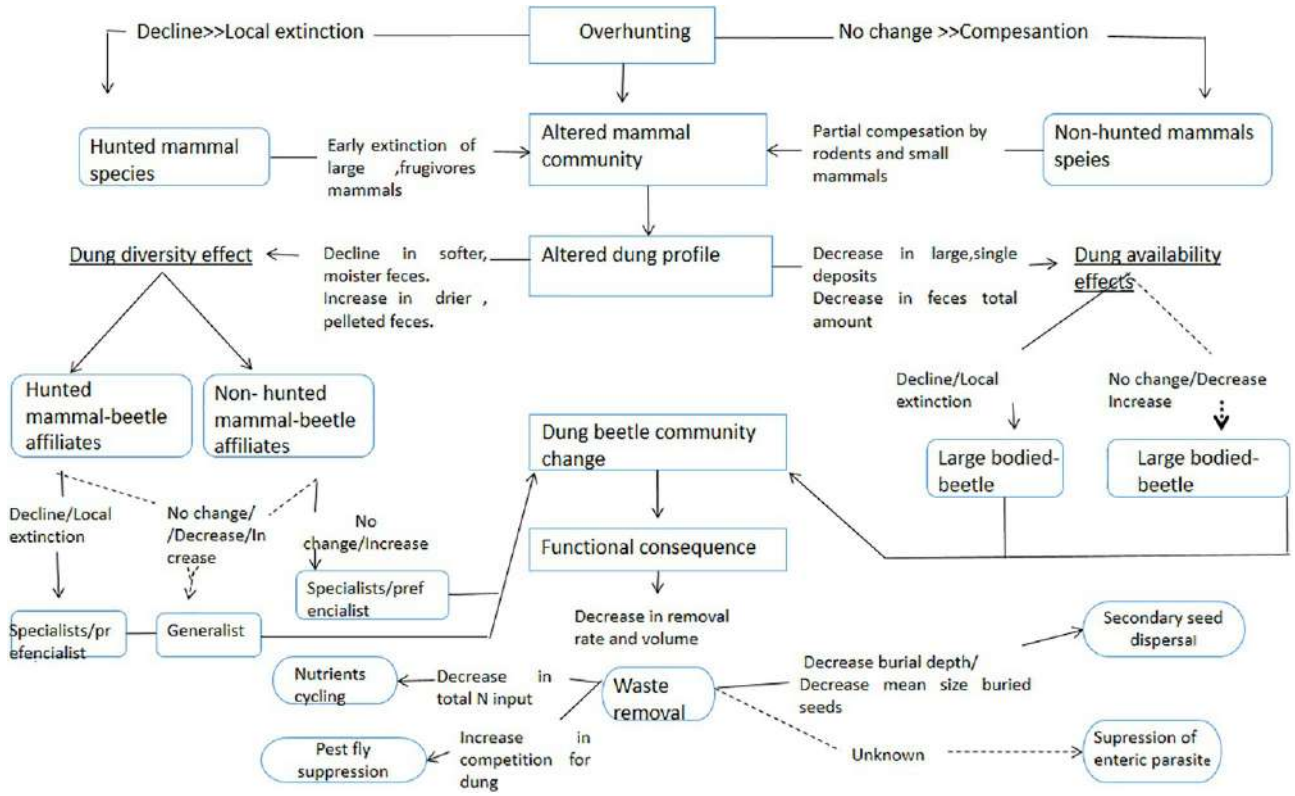

FIG. 28.3 Conceptual diagram showing the possible impact of overhunting mammals in tropical forests on dung beetles and dung beetle-mediated ecological function. *(Credits: Adapted from* Nichols, E., Gardner, T.A., Peres, C.A., Spector, S., 2009. Co-declining mammals and dung beetles: an impending ecological cascade. Oikos 118 (4), 481–487. https://doi.org/10.1111/j.1600-0706.2009.17268.x.*)*

mining (Abernethy et al., 2016; Gillet et al., 2016). These activities have direct and indirect impacts on dung beetles, as forest-dwelling dung beetles are particularly vulnerable to changes in tree cover and the availability of mammalian fauna. Moreover, the widespread destruction of forests poses a significant threat to the overall biodiversity of tropical forests (Hanski, 1999). Different dung beetle tribes respond differently to forest degradation. For instance, Sisyphini and Onthophagini exhibit higher tolerance to forest impoverishment, while Dichotomiini and Oniticellini are restricted to extensive forests with diverse mammalian fauna. Research conducted by Hanski (1999) in the Ivory Coast and Gabon demonstrates that localities with abundant mammalian fauna experience no significant reduction in beetle fauna following the transition from primary to secondary forests. However, there is a decrease in species numbers when the forest structure becomes simplified, thus affecting mammalian fauna (Robinson et al., 1999). This discrepancy in response to disturbance and vegetation quality between forests highlights the importance of intact forest habitats in supporting higher species richness of dung beetles compared to logged forests, secondary forests, agroforests, and tree plantations (Nichols et al., 2007).

6 Agricultural activities

Forest clearance as a result of agriculture (Fig. 28.4) was set back during colonization when various commercial crops were introduced like *Coffea canephora*, commercial timber

FIG. 28.4 Slash and Burn agriculture in the Democratic Republic of the Congo. *(Credits: The Okapi Conservation Project, with a link to their website for further reference: http://www.okapiconservation.org.)*

extraction, and recently slash-and-burn for growing bananas, manioc, and other cash crops (Lebbie, 2001; Stanbrook et al., 2021; Huntley, 2023). This process of creating plantations has caused a significant decline in the specialized rainforest dung beetle fauna, which is primarily attracted to omnivore dung and has been replaced by species that are abundant in savannas (Fuzessy et al., 2021). The reduction of forest canopy in tropical Africa is predominantly propelled by the practice of shifting agriculture, the establishment of cattle ranching operations, the emergence of large farms under ownership of sizable landholders, and the advancement of commercial interests (Zhang et al., 2005; Corlett and Primack, 2008; Lhoest et al., 2020). This phenomenon is particularly prominent in West Africa, with Central and East Africa also experiencing similar trends (Corlett and Primack, 2008). Therefore, land use activities have consistently been recognized as a significant factor contributing to deforestation and forest degradation. These activities increased access to remote areas by hunters and the expansion of farmlands into forested areas to compensate for unproductive land (Tranquilli et al., 2014). In agricultural and plantation settings, there is typically a high presence of a few specific species like *Onthophagus pseudovinctus*, *Tiniocellus spinipes*, *Sisyphus seminulum*, *Onthophagus atronitidus*, *Drepanocerus caelatus*, *Onthophagus lacustris*, and *Trichaphodius gorillae* (Davis and Philips, 2005; Jacobs et al., 2010; Gebert et al., 2020). These species have adapted to the conditions in these areas and rely on a steady supply of excrement from animals and humans (Davis and Philips, 2005; Stanbrook et al., 2021).

7 Loss of wildlife within a standing forest

Dung beetles are highly dependent on mammalian dung such as omnivores (primates, wild pigs) or large nonruminant (monogastric) herbivores (elephants) for their food sources and nesting activities (Cambefort and Walter, 1991). Bushmeat harvesting and hunting are critical threats to the persistence of tropical vertebrates in African forests (Tranquilli et al., 2014; Mongyeh et al., 2018). These activities leave forests intact but lacking nearly all associated mammals in these communities (Andresen and Laurance, 2007). Many animals such as duiker, elephants, porcupines, chimpanzees, and monkeys that are potential sources of dung for attracting dung beetles have been decimated (Mongyeh et al., 2018; Lhoest et al., 2020; Kamgaing et al., 2022). Recent research by Raine and Slade (2019) highlights the impact of selective defaunation, particularly the decline in large-bodied mammals, on the composition of dung beetle communities. This is concerning as many communities in tropical forests heavily rely on wild meat for over 50% of their protein intake (Robinson et al., 1999). Even partial removal of mammalian fauna can have

FIG. 28.5 A depiction showing a pyre of ivory being set ablaze in Kenya. *(Credits: ©Daniel Stiles.)*

repercussions on dung beetle species richness. Forest dung beetles are known to have specific associations with certain types of dung, such as that of omnivores like primates and wild pigs, or large nonruminant herbivores like elephants (Cambefort and Walter, 1991; Kamgaing et al., 2022). Some of the large-bodied species of the dung beetle genus, *Heliocopris*, are specialized in elephant dung (Scholtz et al., 2009). Increasing hunting pressure exacerbates the decline of vertebrate biomass, with notable impacts on preferred game species such as primates and ungulates (Poulsen et al., 2011; Koerner et al., 2017). Notably, Cameroon is identified as a hotspot of hunting-induced defaunation (Fig. 28.5), with estimates suggesting population reductions of 70%–100% in more than 50% of mammal species due to hunting activities (Lhoest et al., 2020). The rise in commercial poaching further exacerbates the negative effects on wildlife populations (Walsh et al., 2003; Bennett, 2011; Nasi et al., 2012). The surge in demand and prices for ivory and other wildlife products in international markets since the 2000s has significantly contributed to the indirect decline of elephant populations in tropical forests (Stiles and Martin, 2003; Wittemyer et al., 2011; Maisels et al., 2013). This in turn affects the abundance of dung beetles that rely on elephant dung for sustenance. Between 2002 and 2011, a staggering 62% of Central Africa's elephants were lost due to poaching (Maisels et al., 2013). This illegal ivory trade has not only eroded the wilderness forests but has also impacted other species, including pangolins.

8 Logging

This often encompasses the clear-cut harvesting of the most valuable tree species (Tranquilli et al., 2014). Additionally, logging leads to the fragmentation of continuous forest areas, which allows hunters and poachers to gain access to these areas (Robinson et al., 1999; Poulsen et al., 2011; Edwards et al., 2017). Central Africa has one of the largest contiguous areas of tropical rainforest globally,

of which >0.6 million km^2 (30%) are under logging concessions and 0.24 million km^2 (12%) are protected (Laporte et al., 2007). The expansion of logging road networks in Central and West Africa (Fig. 28.6) has further facilitated access for poachers and bushmeat traders (Robinson et al., 1999). According to Tranquilli et al. (2014), the result of logging is often the selective harvesting of the most valuable tree species, which directly reduces habitat quality (Fimbel et al., 2001) and adversely affects dung beetles that are sensitive to such environmental deterioration (Bicknell et al., 2014). Commercial logging significantly increases wildlife harvesting from tropical forests. This is achieved by opening up remote forest areas and promoting access for other individuals, thereby altering resource consumption patterns (Robinson et al., 1999). In African tropical forests, the annual harvest of bushmeat may surpass 1 million metric tons, largely influenced by the accessibility of the logged forests and logging camps (Wilkie and Carpenter, 1999). For instance, a single logging camp in the Republic of the Congo reported a harvest of 8251 individual animals, equivalent to 124 tons of wild meat (Wilkie and Carpenter, 1999). The negative impact of logging extends to dung beetle species richness. The reduced species richness and abundance of dung beetles can be attributed to the environmental stress along the roadsides, caused by a large amount of dead wood littering the surface. *Onthophagus fuscatus*, *Onthophagus densepilis*, *Sisyphus* (*Neosisyphus*) *angulicollis*, and *Sisyphus latus* have been encountered within logged forests (Davis and Philips, 2005).

9 Hydroelectric power

Access to electricity is paramount, particularly in urban areas, where the increasing temperatures influenced by climate change have resulted in a higher demand for air-conditioning and refrigeration to maintain suitable living conditions. As a consequence, the domestic electricity demand has surged (Abernethy et al., 2016). In order to meet this growing energy requirement, additional hydroelectric generation is being planned. For instance, the proposed lower Congo River hydroelectric scheme at Inga Falls aims to provide extra energy (Fig. 28.7). However, it is important to note that this ambitious project is anticipated to disrupt the flow and discharge of the Congo River, potentially encompassing the flooding of over 22,000 km^2 of forest (Abernethy et al., 2016). Such an endeavor could directly impact the region's unique wildlife, including the Scarabaeinae fauna, and consequently compromise the ecosystem services they provide. The creation of reservoirs leads to changes in microclimate, including altered temperature and humidity levels. Dung beetles are known to be sensitive to such environmental changes, and alternations in their microclimate can negatively impact their survival and reproductive success. Furthermore, dam constructions disrupt the movement of dung beetles and other forest organisms, thus limiting their ability to find suitable habitat and resources.

10 Mining

Africa contains about 30% of the world's mineral reserve and a wide range of vital minerals, including diamonds, platinum, cobalt, phosphate, chromite, manganese, vanadium, aluminum, coal uranium, and iron ore (Taylor et al., 2009; Edwards et al., 2013). Many regions within tropical forests, such as the central Africa region, are known to possess significant mineral wealth. The western part of this region is particularly rich in large oil reserves (Hansen et al., 2013). All countries in the area boast diverse mineral deposits that form the backbone of their economies through

FIG. 28.6 Logging road inside the forest in Cameroon. *(Credits: InfoCongo.)*

FIG. 28.7 River hydroelectric scheme at Inga Falls (https://www.rechargenews.com/transition/worlds-largest-hydro-dam-could-send-cheap-green-hydrogen-from-congo-to-germany/2-1-871059). *(Credits: MARC JOURDIER/AFP via Getty Images/NTB scanpix.)*

mineral extraction activities and are supplemented with timber revenues. However, it is important to note that transitioning to alternative land uses beyond oil and mineral exploitation presents numerous challenges, including the need for more space and complex management requirements. The increased demand for oil and minerals from foreign investors, particularly from other continents, has exerted mounting pressure on these forests, leading to clearance for mineral extraction. This has been revealed in various studies (Laurance et al., 2006a,b; Zhang and Wilkes, 2010; Janneh and Ping, 2011; Abernethy et al., 2016). For instance, in 2014, a significant proportion of Cameroon's total revenue was derived from oil exploitation, as reported by UNCTAD (2015). Similarly, Abernethy et al. (2016) found that mineral exports heavily influence over 85% of the Democratic Republic of the Congo's total revenue. Unfortunately, these processes have also contributed to a cumulative rate of deforestation, as highlighted by Hansen et al. (2013). The destruction of natural habitats for dung beetles through road construction, infrastructure, and mining pits disturbs the forest floor and stable soil conditions upon which beetles rely for burrowing and nesting.

11 Population growth

Rapidly growing human populations (2–3 increase annually) throughout most of Africa bring increasing demands on forests and their associated natural resources (Struhsaker et al., 2005). The loss of wildlife and rapid deforestation in countries like Equatorial Guinea, the Democratic Republic of the Congo, and Cameroon are closely linked to the high-density and expanding rural populations in these areas (Struhsaker et al., 2005; Laurance et al., 2014; Abernethy et al., 2016). As the rural population continues to grow, there is an increasing demand for forest resources like firewood and charcoal (Abernethy et al., 2016). The demand for land for agriculture increases, leading to deforestation for logging and infrastructure; in addition to that, the use of pesticide and other chemicals in agriculture can contaminate the environment and directly harm dung beetles through disrupting their behavior and reproductive abilities. Moreover, human population growth and associated activities contribute to climate change, which has several indirect effects on dung beetles. Changes in temperature and rainfall patterns can alter the availability of suitable microhabitats and affect the timing of dung beetles' life cycle events such as reproduction and emergence (Scholtz et al., 2009; Hanski and Cambefort, 1991; Latha, 2019).

12 Infrastructure

According to Abernethy et al. (2016), one of the key ways to tackle poverty in African tropical countries is by improving the transport network infrastructure, which promotes the exchange of agricultural products. However, the construction of roads and railways in untouched areas contributes to the rapid deterioration of the forest ecosystem. In fact, Abernethy et al. (2016) suggest that the development of transport infrastructure will play a larger role in driving deforestation in the Congo basin over the next 15 years compared to any other factor. This could indirectly affect the dung beetle fauna. Roads passing through forests may lead to more vehicle collision with dung beetles causing local declines in populations specifically for flightless species, thus decreasing genetic diversity (Hayward et al., 2010).

13 Operational and potential monitoring tools to recover the biome

Dung beetles play a critical role as indicator taxa for biodiversity inventory and monitoring in the ecosystem (Davis et al., 2001). Therefore, it is crucial to closely monitor the African tropical forest biome, which exhibits a higher abundance and species diversity of dung beetles, to ensure their continued existence. This can be achieved through a

combination of traditional field surveys, remote sensing technologies, and molecular and genetic techniques, which collectively provide valuable insights into population dynamics, habitat preferences, and species distribution patterns. Active engagement of local communities, researchers, and conservation organizations is essential to obtain comprehensive data on African dung beetles and coordinate long-term monitoring initiatives. Habitat loss poses a significant threat to the survival of dung beetles, as it can interrupt gene flow among separated subpopulations due to large areas of dividing matrices that prevent the exchange of genes among migrants (Scholtz et al., 2009). Inbreeding resulting from habitat modification, such as road construction, logging, and hydroelectric power, can lead to reduced survivorship and eventual extinction.

Deforestation and agriculture, particularly those associated with cattle raising, can disrupt dung beetle specialists' reliance on wild animal dung, as they shift to utilizing cattle dung with differing texture, size, and fiber content (Wirta et al., 2008). Furthermore, the contrasting environmental conditions, such as humidity and temperatures, between open grazing habitats and originally closed forest habitats make radical shifts in resource and habitat use highly unlikely. This highlights the need for molecular and genetic techniques to gain insights into the adaptability of dung beetles under such conditions. The findings obtained could be applied to other tropical forest ecosystems.

Additionally, initiatives aimed at raising public awareness about the significance of dung beetles and their habitats further will reinforce the importance of monitoring efforts. It is also imperative to promote research and education initiatives that deepen our understanding of African dung beetle ecology. By enhancing knowledge in this field, we can better identify and address potential threats to these beetles and their habitats, thus bolstering conservation efforts.

14 Conclusion

Conserving primary forests and their associated fauna is a matter of utmost importance as these ecosystems play a crucial role in providing essential ecosystem services. Human activities have significantly impacted dung beetle populations, altering their abundance, diversity, and community composition. These changes have detrimental consequences for key ecological processes, as highlighted by Scholtz et al. (2009). A decline in dung beetles, particularly large-bodied beetles, and specialized feeders directly affects ecosystem function by reducing the volume and rate of dung removal and secondary seed dispersal. This in turn has an impact on plant communities, as emphasized by Scholtz et al. (2009). Bioturbation and nutrient recycling, essential for maintaining the primary productivity of the entire system, are diminished. Moreover, the accumulation of dung leads to increased populations of parasite and pest flies, potentially triggering parasitic diseases (Doube, 2018). The consequences of a decrease in dung beetle diversity and abundance extend beyond ecological processes. Predators that rely on dung beetles as a food source, such as golden jackals, baboons, monitor lizards, guinea fowl, owls, robber flies, wasps, and reduviid bugs, are also negatively impacted (Scholtz et al., 2009; Temu et al., 2018).

Given the critical role of dung beetles in ecosystem maintenance, it is imperative to prioritize their conservation by acknowledging their status as species of higher conservation value. Continuous loss of dung beetles could potentially lead to ecosystem collapse. Consequently, research efforts should focus on understanding the genetic and species-specific responses to anthropogenic disturbances, instead of solely relying on broader community, ecosystem, or landscape-based management policies (Scholtz et al., 2009). Additionally, it is essential to implement secondary forest plantation, forest restoration programs, and the establishment of forest nature reserves to restore dung beetle populations caused by deforestation and defaunation. These efforts can contribute to enhancing ecosystem resilience and attracting various animal species to inhabit these habitats. To ensure effective conservation, it is also crucial to address illegal hunting and prevent further threats to animal populations from decimation. Key strategies include enhancing law enforcement and antipoaching operations (Benitez-Lopez et al., 2017), better control of access in logging concessions and protected areas (Van Vliet and Nasi, 2008), regulating hunting of the most sensitive species (Nasi and Van Vliet, 2011; Van Vliet and Nasi, 2008), and creating protected areas, as an effective strategy, for combating deforestation in the African tropical forest (Andam et al., 2008; Gaveau et al., 2009; Naughton-Treves et al., 2005), as demonstrated in countries like Cameroon, the Central African Republic, the Democratic Republic of the Congo, Ghana, Mozambique, Tanzania and Equatorial Guinea (Bowker et al., 2016).

Therefore, the conservation of primary tropical forests and the diverse array of species they support, including dung beetles, is of utmost importance. Appropriate measures must be taken to protect and restore dung beetle populations for the sake of ecological integrity, ecosystem function, and the overall resilience of our planet's ecosystems.

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Chapter 29

The use of phytotelmata by amphibians in the Amazon rainforest: A review and case study

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1 Introduction

The Amazon is the largest tropical rainforest on earth, covering over 5.5 million km^2 of South America, and occupies a portion of several countries: Brazil, Bolivia, Colombia, Ecuador, Guyana, French Guiana, Peru, Suriname and Venezuela (WWF, 2023). The Amazon contains half of the planet's remaining tropical rainforests (Staal et al., 2020) and covers about 40% of the South American continent (D'Amato et al., 2017).

The Amazon biome is composed of a mosaic of ecosystems and vegetation types including rainforests, seasonal forests, deciduous forests, flooded forests and savannas (Magalhães et al., 2022). The Amazon rainforest contains 25% of the planet's biodiversity (Ruiz-Vásquez et al., 2020) and its intrinsic value is key to preserving Amazonian ecosystems with all their environmental functions (Fearnside, 2021).

The Amazon's biodiversity is of global importance, as it plays an important role in the world's water and carbon cycles (Marengo, 2007). The primary cause of decline in biodiversity and ecosystem function is the destruction and degradation of natural ecosystems (Pereira et al., 2010). Approximately 38% of the Amazon rainforest is currently degraded by fire, edge effects, wood logging, and/or extreme drought (Lapola et al., 2023). Between 1985 and 2022, the Brazilian Amazon rainforest lost 50.3 million hectares (MapBiomas, 2023). The loss of native vegetation in Brazil has accelerated in the last 10 years (2013–2022), a period that coincides with the new Brazilian Forest Code, approved by National Congress in 2012 (MapBiomas, 2023).

The new Brazilian Forest Code, enacted to address land use and environmental conservation, has had mixed effects on deforestation, biodiversity, and greenhouse gas emissions (Freitas et al., 2018). These effects have increased the potential for biodiversity loss, especially for amphibians (Toledo et al., 2010), and land-use changes such as agriculture, cattle-raising, deforestation, and selective logging (Cordier et al., 2021) have negatively impacted this group.

The structural complexity of Amazonian forests is a key factor in supporting biodiversity. Multiple canopy layers are created by tall trees that reach heights of over 40 m, which support a wide variety of organisms (Lindenmayer and Laurence, 2017). The Brazilian Forest Code is a threat to this forest structure due to land-use changes that allow logging near riparian forests and Areas of Permanent Preservation (APP) (Toledo et al., 2010).

Tall trees with a diameter at breast height (DBH) greater than 50 cm and of commercial value are cut down by the timber industry (BRASIL, 2006; Noronha et al., 2021). Many of these trees have cavities that accumulate water, support different plant species, and serve as habitat for many species of invertebrates (Zotz and Traunspurger, 2016) and vertebrates such as amphibians (Noronha et al., 2021).

One feature of Amazonian biodiversity is the presence of phytotelmata, which are landscape elements that contribute to the maintenance of local biodiversity (Rangel et al., 2017). Phytotelmata are defined as water bodies

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contained in a part of a terrestrial plant such as bromeliad leaf axils, Brazil nut capsules, bamboo and tree holes (Rödel et al., 2004). They play a crucial role in the life cycle, distribution (Schiesari et al., 2003; Noronha et al., 2021), development and survival of many organisms (e.g., Schiesari et al., 2003; Rangel et al., 2017).

These aquatic environments offer several benefits to organisms; due to their small size and limited access, they provide protection from large predators (Fouilloux et al., 2021); they reduce interspecific competition (Summers and Tumulty, 2014); and they maintain a relatively stable environment that is less susceptible to chemical fluctuations (Fouilloux et al., 2021).

Many species (272) of amphibians use phytotelmata for survival (Lehtinen, 2022) and have evolved to deposit their eggs and tadpoles in these small water bodies within plant structures (Fouilloux et al., 2021, 2022; Burrow and Maerz, 2022). Anurans have evolved to use different types of phytotelmata depending on their specific physiological requirements during different stages of their life cycle (Lantyer-Silva et al., 2018; Noronha et al., 2021). For example, some species prefer bromeliads with large leaf axils, where they deposit eggs directly into the water-filled cavity. Others may choose tree holes that accumulate rainwater or fallen leaves, which provide organic matter as food resources for developing tadpoles.

Due to the importance of phytotelmata in the maintenance and conservation of various invertebrate and vertebrate species, we provide an overview of research on the use of phytotelmata by anurans in the Amazon. We also present a case study that highlights the importance of phytotelmata in the conservation of several vertebrate species, especially amphibians. In addition, we assess the conservation status of species that use phytotelmata, and the use of tools such as camera traps and artificial phytotelmata to study the behavior and reproduction of these species, respectively.

2 Geographic location

2.1 Phytotelm-breeding anurans of Amazonia: Data survey

We obtained occurrence records of Amazonian anurans breeding in phytotelmata from the online database *Phytotelm Breeding Frogs of the World* (Lehtinen, 2022), available at https://sites.google.com/site/phytotelmbreedingfrogsworld/home. From this database, we selected and downloaded data of anuran families occurring in South America. In each case, we selected the anuran species occurring in Amazonian countries based on geographic distribution data available in the online database *Amphibian Species of the World: an Online Reference. Version 6.2* (Frost, 2023), available at https://amphibiansoftheworld.amnh.org/index.php. We also performed a bibliographic search using Google Scholar to find scientific publications on the subject to complement the data obtained from the first mentioned database. After compiling the data, we created a map (Fig. 29.1) to determine whether the sites were located in areas impacted by anthropogenic activities.

3 Biodiversity

Eighty anuran species belonging to six families (Aromobatidae, Bufonidae, Dendrobatidae, Hemiphractidae, Hylidae, and Microhylidae) have been recorded using different phytotelmata (Fig. 29.2) as breeding sites in the Amazon morphoclimatic domain (Fig. 29.1; Table 29.1). The family with the largest number of species using phytotelmata for breeding is Dendrobatidae with 61.25% (49 species), followed by Hylidae with 23.75% (19 species), Aromobatidae with 7.5% (six species), and Bufonidae, Hemiphractidae, and Microhylidae with 2.5% each (two species each).

Due to the high instability of these environments, assessments of the degree of threat to the species were made following the IUCN Red List categories (IUCN, 2022), available at https://www.iucnredlist.org/. Forty species were classified as Least Concern (LC), three as Not Evaluated (NE), and seven as Data Deficient (DD). However, 11 species were considered as Endangered (EN), 10 as Vulnerable (VU), seven as Critically Endangered (CR), and two as Near Threatened (NT) (Table 29.1). All of the threatened species are located on the edge of the Amazon near the Andes mountain range (e.g., Ecuador, Peru) or in the extreme north of the biome (e.g., Guyana, Venezuela). The Brazilian Amazon has no threatened species recorded in either the IUCN or ICMBio lists (Table 29.1).

4 Distribution and use of phytotelmata by *Trachycephalus cunauaru* in the Southern Amazon: A case study

In this study, we evaluated the distribution and use of phytotelmata by *T. cunauaru*. Plant species with phytotelmata were identified, and their diameter at breast height (DBH) measured, to help assess the possible risks associated with population decline. We installed artificial breeding sites and evaluated the use of the species for reproduction using a digital camera trap.

The study was conducted at Fazenda São Nicolau (9° 51′S, 58°14′W), located in the municipality of Cotriguaçu, Mato Grosso, Brazil, in the southern Amazon (see Fig. 29.3). Data were collected using a standardized sampling module (RAPELD) of the Biodiversity Research Program - PPBio (Magnusson et al., 2005). The module covers an area of 5 km^2 and consists of two parallel 5 km trails connected by six perpendicular 1 km trails. The local

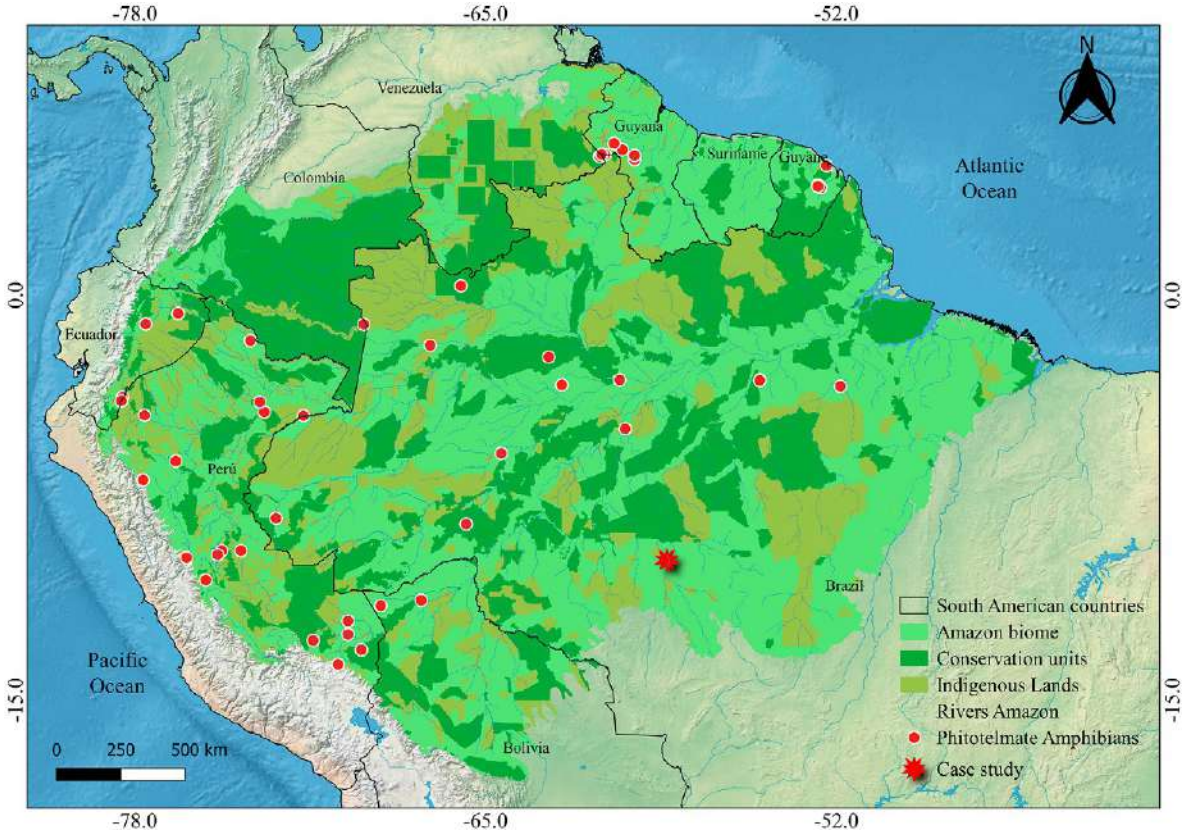

FIG. 29.1 Distribution of studies on anurans that use phytotelmata for reproduction in the Amazon biome. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: The authors.)*

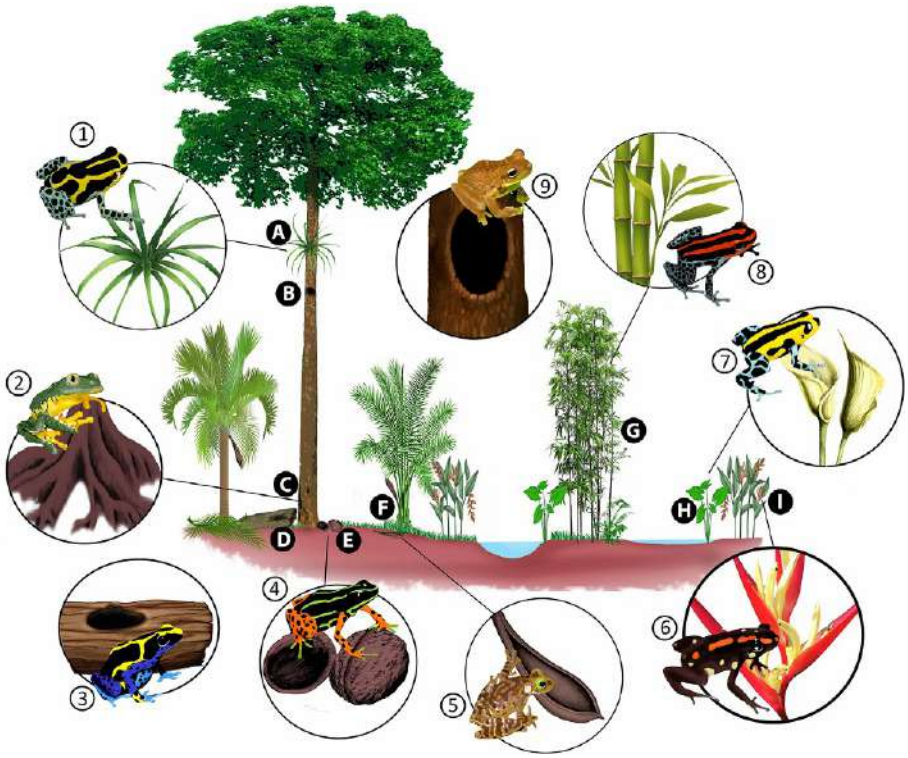

FIG. 29.2 Phytotelmata used for reproduction by various anurans species. Bromeliads (A), Tree holes (B), tree buttresses (C), cavities in fallen trees (D), Brazil nut capsules (E), palm bract (F), bamboo stalk (G), Araceae bract (H), and Heliconia bract (I). Species of anurans: *Ranitomeya variabilis* (1), *Cruziohyla craspedopus* (2), *Dendrobates tinctorius* (3), *Adelphobates quinquevittatus* (4), *Osteocephalus oophagus* (5), *Excidobates captivus* (6), *Ranitomeya sirensis* (7), *Ranitomeya ventrimaculata* (8), and *Trachycephalus cunauaru* (9). *(Photo credits: The authors.)*

TABLE 29.1 Checklist of Amazonian anurans breeding in phytotelmata adapted from Lehtinen's electronic database (2022). Columns include the anuran species, the countries of occurrence, the microhabitat used as breeding site, and the conservation status. Country abbreviations include Bolivia (BOL), Brazil (BRA), Colombia (COL), Ecuador (ECU), French Guiana (GUF), Guyana (GUY), Peru (PER), Suriname (SUR), and Venezuela (VEN). IUCN Red List and ICMBio categories: Data Deficient (DD), Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR), Not Evaluated (NE); CITES Appendix II (lists species that are not necessarily threatened with extinction now, but may become so if trade is not strictly controlled).

Species	Geographic occurrence in Amazon countries	Larval habitat	Conservation status		
			IUCN	CITES	ICMBio
Aromobatidae					
Allobates bromelicola	VEN	Bromeliads	VU	–	–
Allobates femoralis	BOL, BRA, COL, ECU, GUF, GUY, PER, SUR, VEN	Tree holes, tree buttresses, Brazil nut capsules, fallen palm leaves	LC	II	LC
Allobates insperatus	COL, ECU, PER	Palm bracts on forest floor	LC	–	–
Allobates sumtuosus	BRA, GUY, PER, SUR	Palm leaves of forest floor, pools in fallen logs	DD	–	LC
Anomaloglossus beebei	GUY	Giant bromeliads (*Brocchinia micranta*)	EN	–	–
Anomaloglossus roraima	GUY, VEN	Large terrestrial bromeliad (*Brocchinia tatei*)	EN	–	–
Bufonidae					
Rhinella castaneotica	BRA, COL, GUF, PER	Brazil nut capsules, tree holes	LC	–	LC
Rhinella magnussoni	BRA	Brazil nut capsules, tree buttresses	LC	–	LC
Dendrobatidae					
Adelphobates castaneoticus	BRA	Brazil nuts, bracts of palms	LC	II	LC
Adelphobates quinquevittatus	BOL, BRA, PER	Brazil nuts, cavities in fallen logs, bromeliads	LC	II	LC
Ameerega hahneli	BOL, BRA, COL, ECU, GUF, GUY, PER, SUR, VEN	Tree holes, streams	LC	II	LC
Ameerega trivittata	BOL, BRA, COL, GUY, PER, SUR, VEN	Palm bracts, streams	LC	II	LC
Andinobates abditus	ECU	Bromeliads	CR	II	–
Andinobates altobueyensis	COL	Bromeliads	DD	II	–
Andinobates bombetes	COL	Bromeliads (*Mezobromelia capituligera* and *Tillandsia fendleri*)	VU	II	–
Andinobates cassidyhornae	COL	Probably bromeliads	VU	II	–
Andinobates daleswansoni	COL	Bromeliads	EN	II	–
Andinobates dorisswansonae	COL	Bromeliads (*Tillandsia* sp. and *Aechmea* sp.)	VU	II	–
Andinobates fulguritus	COL	Bromeliads and *Heliconia* bracts	LC	II	–
Andinobates minutus	COL	Bromeliads, other phytotelmata	LC	II	–
Andinobates opisthomelas	COL	Bromeliads, fallen palm bracts	VU	II	–
Andinobates tolimense	COL	Bromeliads, tree holes	VU	II	–
Andinobates victimatus	COL	Bromeliads (*Gusmania musaica*)	EN	II	–

TABLE 29.1 Checklist of Amazonian anurans breeding in phytotelmata adapted from Lehtinen's electronic database (2022). Columns include the anuran species, the countries of occurrence, the microhabitat used as breeding site, and the conservation status. Country abbreviations include Bolivia (BOL), Brazil (BRA), Colombia (COL), Ecuador (ECU), French Guiana (GUF), Guyana (GUY), Peru (PER), Suriname (SUR), and Venezuela (VEN). IUCN Red List and ICMBio categories: Data Deficient (DD), Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR), Not Evaluated (NE); CITES Appendix II (lists species that are not necessarily threatened with extinction now, but may become so if trade is not strictly controlled)—cont'd

Species	Geographic occurrence in Amazon countries	Larval habitat	Conservation status		
			IUCN	CITES	ICMBio
Andinobates viridis	COL	Bromeliads	CR	II	–
Andinobates virolinensis	COL	Bromeliads	VU	II	–
"Colostethus" ruthveni	COL	Eggs initially laid in bromeliads but then moved to streams	NT	–	–
Dendrobates auratus	COL	Tree holes, bromeliads, stagnant pools	LC	II	–
Dendrobates leucomelas	BRA, COL, GUY, VEN	Tree holes and trunks, bromeliads	LC	II	LC
Dendrobates tinctorius	BRA, GUF, GUY, SUR	Tree holes, cavities on fallen trees, palm bracts	LC	II	LC
Dendrobates truncatus	COL	Tree holes, pools in tree buttresses, forest pools	LC	II	–
Excidobates captivus	ECU, PER	Heliconia bracts	VU	II	–
Excidobates condor	ECU	Bromeliads (*Guzmania* sp., *Tillandsia* sp., *Aechmea* sp.)	EN	II	–
Excidobates mysteriosus	PER	Bromeliads (*Aechmea* sp.)	EN	II	–
Hyloxalus azureiventris	PER	Initially bromeliads or leaf litter, then transported to streams	EN	II	–
Hyloxalus chlorocraspedus	BRA, PER	Depressions in fallen tree trunks	DD	II	LC
Minyobates steyermarki	VEN	Terrestrial bromeliads	CR	II	–
Oophaga histrionica	COL	Bromeliads	CR	II	–
Oophaga lehmanni	COL	Bromeliads, tree holes, bamboo stalks	CR	II	–
Oophaga occultator	COL	Unknown	CR	II	–
Oophaga sylvatica	COL, ECU	Bromeliads	NT	II	–
Paruwrobates andinus	COL	Epiphytic bromeliads	CR	II	–
Phyllobates aurotaenia	COL	Streamside puddles and fallen palm bracts	LC	II	–
Ranitomeya amazonica	BRA, COL, PER	Bromeliads (*Pitcairnia*, *Aechmea*, and *Catopsis*)	DD	II	LC
Ranitomeya benedicta	PER	Bromeliads (*Aechmea*)	VU	II	–
Ranitomeya defleri	BRA, COL	Bromeliads	LC	II	NE
Ranitomeya fantastica	PER	Bromeliads (*Aechmea* and *Guzmania*), tree holes	VU	II	–
Ranitomeya flavovittata	PER	Bromeliads (*Guzmania*), tree holes	LC	II	–
Ranitomeya imitator	PER	*Heliconia* bracts, Diffenbachia, Xanthosoma leaf axils	LC	II	–
Ranitomeya reticulata	ECU, PER	Bromeliads, tree holes	LC	II	–

Continued

TABLE 29.1 Checklist of Amazonian anurans breeding in phytotelmata adapted from Lehtinen's electronic database (2022). Columns include the anuran species, the countries of occurrence, the microhabitat used as breeding site, and the conservation status. Country abbreviations include Bolivia (BOL), Brazil (BRA), Colombia (COL), Ecuador (ECU), French Guiana (GUF), Guyana (GUY), Peru (PER), Suriname (SUR), and Venezuela (VEN). IUCN Red List and ICMBio categories: Data Deficient (DD), Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR), Not Evaluated (NE); CITES Appendix II (lists species that are not necessarily threatened with extinction now, but may become so if trade is not strictly controlled)—cont'd

Species	Geographic occurrence in Amazon countries	Larval habitat	Conservation status		
			IUCN	CITES	ICMBio
Ranitomeya sirensis	BOL, BRA, PER	Primarily bamboo but also *Heliconia* and *Xanthosoma* axils, palm flower husks	LC	II	
Ranitomeya summersi	PER	Tree holes, *Diffenbachia* and *Xanthosoma* axils	EN	II	LC
Ranitomeya toraro	BRA, COL	Bromeliads (*Aechmea*) and *Phenakospermum* axils	NE	II	LC
Ranitomeya uakarii	BRA, COL, GUY, PER	Bromeliads (*Guzmania*)	LC	II	LC
Ranitomeya vanzolinii	BRA, PER	Bromeliads, *Heliconia* bracts, tree holes	LC	II	LC
Ranitomeya variabilis	BRA, COL, ECU, GUF, PER	Bromeliads (*Aechmea, Guzmania*), tree holes	DD	II	LC
Ranitomeya ventrimaculata	BRA, COL, ECU, PER	Bromeliads (*Guzmania*), *Heliconia* leaf axils, bamboo, tree holes	LC	II	NE
Ranitomeya yavaricola	PER	Small arboreal bromeliads	DD	II	–
Hemiphractidae					
Flectonotus fitzgeraldi	VEN	Bromeliads, aroids	LC	–	–
Flectonotus pygmaeus	COL, VEN	Bromeliads and *Heliconia* axils	LC	–	–
Hylidae					
Cruziohyla calcarifer	COL, ECU,	Depressions on fallen trees, tree buttresses, terrestrial pools (?)	LC	–	–
Cruziohyla craspedopus	BOL, BRA, COL, ECU, PER	Tree holes and buttresses	LC	–	LC
Ecnomiohyla miliaria	COL	Tree holes	LC	–	–
Ecnomiohyla phantasmagoria	COL, ECU	Tree holes?	DD	–	–
Myersiohyla liliae	GUY	Bromeliads	EN	–	–
Nyctimantis rugiceps	COL, ECU, PER	Bamboo stumps and tree holes	LC	–	–
Osteocephalus castaneicola	BOL, BRA, PER	Brazil nut capsules, palm bracts, bamboo	LC	–	LC
Osteocephalus deridens	BRA, COL, ECU, PER	Bromeliads (*Aechmea*)	LC	–	NE
Osteocephalus fuscifacies	ECU, PER	Bromeliads (*Aechmea*)	LC	–	NE
Osteocephalus oophagus	BRA, COL, GUF, GUY, SUR	Bromeliads, palm bracts, tree holes, epiphytes	LC	–	LC
Osteocephalus planiceps	COL, ECU, PER	Bromeliads (*Aechmea*) and other epiphytes, leaf axils of *Mauritia flexuosa*	LC	–	NE
Osteocephalus vilarsi	BRA, COL, VEN	Temporary puddles, tree buttresses	NE	–	LC

TABLE 29.1 Checklist of Amazonian anurans breeding in phytotelmata adapted from Lehtinen's electronic database (2022). Columns include the anuran species, the countries of occurrence, the microhabitat used as breeding site, and the conservation status. Country abbreviations include Bolivia (BOL), Brazil (BRA), Colombia (COL), Ecuador (ECU), French Guiana (GUF), Guyana (GUY), Peru (PER), Suriname (SUR), and Venezuela (VEN). IUCN Red List and ICMBio categories: Data Deficient (DD), Least Concern (LC), Near Threatened (NT), Vulnerable (VU), Endangered (EN), Critically Endangered (CR), Not Evaluated (NE); CITES Appendix II (lists species that are not necessarily threatened with extinction now, but may become so if trade is not strictly controlled)—cont'd

Species	Geographic occurrence in Amazon countries	Larval habitat	Conservation status		
			IUCN	CITES	ICMBio
Phytotriades auratus	VEN	Bromeliads (*Glomeropitcairnia*)	EN	–	–
Tepuihyla exophthalma	GUY, VEN	Bromeliads (*Brocchinia*)	LC	–	–
Tepuihyla shushupe	BRA, COL, PER	Probably tree holes	LC	–	NE
Tepuihyla tuberculosa	BRA, ECU, PER	Probably tree holes	LC	–	LC
Trachycephalus cunauaru	BOL, BRA, ECU, PER	Tree holes	NE	–	LC
Trachycephalus hadroceps	BRA, GUF, GUY, SUR	Tree holes	LC	–	LC
Trachycephalus resinifictrix	BRA, GUF, GUY, SUR	Tree holes	LC	–	LC
Microhylidae					
Chiasmocleis antenori	BRA, ECU, PER	Bromeliads (*Guzmania weberbaueri*)	LC	–	LC
Ctenophryne carpish	PER	Bromeliads	EN	–	–

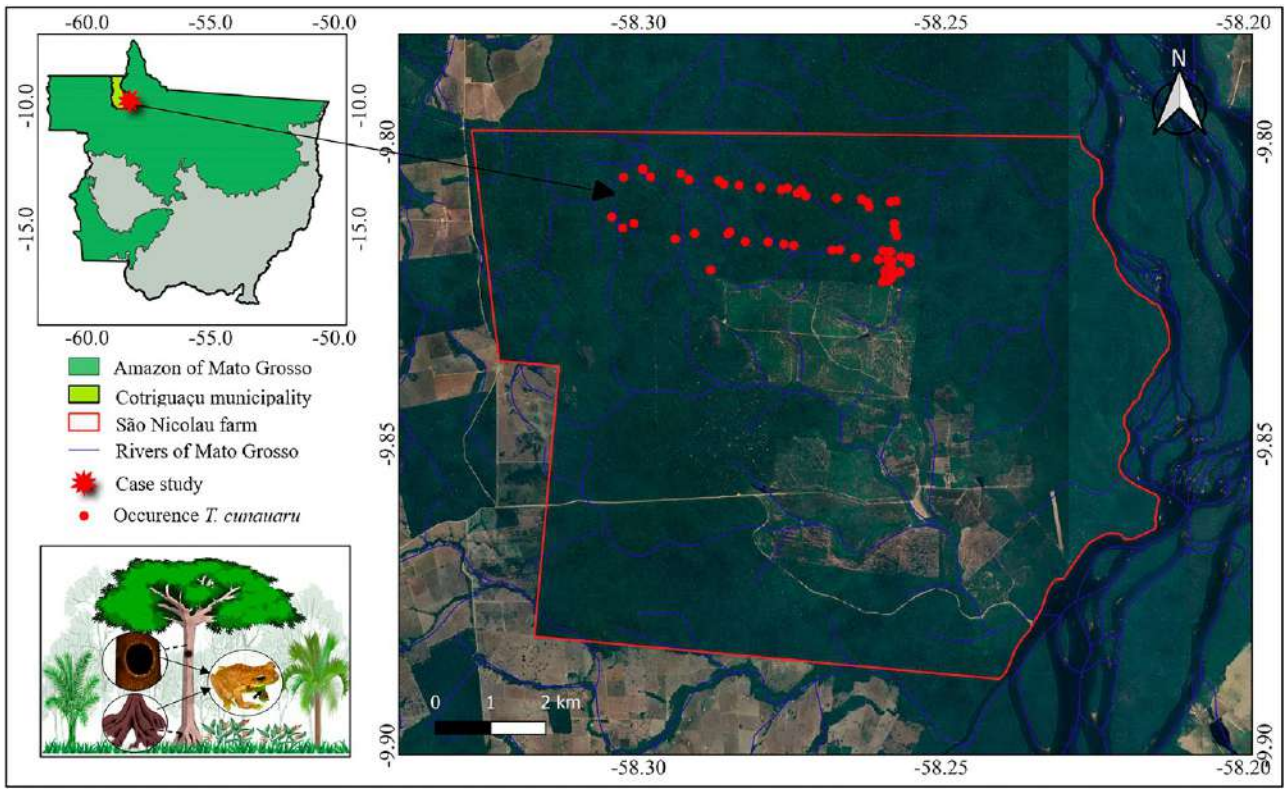

FIG. 29.3 Distribution of *Trachycephalus cunauaru* in the sampling module of the Biodiversity Research Program installed at Fazenda São Nicolau, Mato Grosso, Brazil. Twelve kilometers were covered monitoring the species *T. cunauaru* during three years (2014–2016). Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Photo credits: The authors.)*

vegetation is characterized as open and dense tropical forest (Veloso et al., 1991), and the climate is classified as humid tropical (Am, according to the Köppen classification). The average annual temperature and annual precipitation are 25°C and 2000 mm, respectively (Camargo et al., 2010). The region experiences two well-defined seasons: a dry season from April to September and a wet season from October to March. The region suffers from a high rate of deforestation and is located within the "Arc of Deforestation" (Noronha and Rodrigues, 2018). This deforestation is often driven by the expansion of agriculture, livestock, mining, and the implementation of selective logging projects (Levy et al., 2018; Hänggli et al., 2023).

Identification and monitoring of phytotelmata were conducted during the wet seasons of 2014, 2015, and 2016, at night, between 6 pm and 4 am. Two researchers walked two 5 km trails, one 1 km trail, and one 700 m trail that provides access to the module. The defined sampling area extended 200 m on either side of the trails, resulting in a total search area of 5.08 km^2. All trees with phytotelmata and vocalizing males were georeferenced. This search method was feasible due to the highly distinctive vocalization of *T. cunauaru*, which can be detected at distances of up to 1000 m (Gordo et al., 2013).

We identified 73 trees with natural breeding sites (phytotelmata: tree holes), resulting in a density of 14.1 sites per km^2, and an average separation distance of 188 m. The smallest diameter at breast height (DBH) measured was 13 cm, and the largest was 138.2 cm, with an average of 35.9 ± 20.5 cm. The height that phytotelmata ocurred in which the males vocalized varied from 30 cm to 30 m. The 73 trees identified with phytotelmata (tree holes) belong to 22 different species. Among these species, the most abundant were *Protium altissimum* (26 individuals), *Naucleopsis caloneur*a (7), *Apeiba tibourbou* (5), *Rinoreocarpus ulei* (5), *Inga thibaudiana* (4), and *Virola elongata* (3). Fifteen percent of these trees with phytotelmata are marked for selective logging, approved by environmental agencies. As a result, trees with a diameter at breast height (DBH) greater than 50 cm and located at least 10 m away from small streams (designated as permanent protection areas) may be felled, affecting the distribution of *T. cunauaru* in the area.

Twenty-two natural phytotelmata were monitored monthly due to their height and 15 artificial breeding sites were installed to evaluate their use by *T. cunauaru* (Fig. 29.4). Tadpoles were found in five of the 15 artificial sites, these being in trees with a larger diameter (50 cm DBH and 20 m high). Males started calling at around 7 pm and finished around 3 am. Using images from digital camera traps installed near the artificial breeding sites, we observed that one solitary male used two adjacent artificial breeding sites that were approximately 10 m apart. In addition, nine observations of amplexus were recorded at the artificial sites, demonstrating their importance in the conservation of the species. We also documented the presence of three snake species within artificial breeding sites: *Leptodeira annulata*, *Philodryas olfersii*, and *Chironius multiventris*, highlighting their importance in the conservation of other groups. Other anuran species (*Osteocephalus oophagus, Cruziohyla craspedopus*) that use phytotelmata are present in the area, but were not found in the natural or artificial breeding sites monitored in this study. Additionally, several species of birds and many invertebrates, such as beetles and mosquito larvae, have been observed in these tree holes and phytotelmata in the area.

5 Human activities and their impact on phytotelmata and anurans

Many factors such as habitat loss and degradation, chemical contaminants, climate change, roads, human exploitation, and invasive species, have been described as negatively impacting phytotelmata and amphibians (Cooper et al., 2014; Luedtke et al., 2023).

FIG. 29.4 Use of natural (A) and artificial (B) breeding sites by *Trachycephalus cunauaru* in the pristine forest of the São Nicolau Farm. *(Photo credits: The authors.)*

Habitat loss and degradation are a global concern, primarily driven by human activities, resulting in extensive loss of forest. These habitat changes affect forest age and structure, which are critical for maintaining phytotelmata and the diversity of associated organisms, including amphibians. Mature and older forests have more favorable conditions for phytotelmata formation, such as humidity, shade, tree cavities, and pitcher plants (Teixeira et al., 2018). These trees have larger diameters and a higher number of microhabitats, including phytotelmata (Vuidot et al., 2011; Noronha et al., 2021; Basham et al., 2022). They are vulnerable to logging and several studies have shown that ecological communities are strongly affected by forest management (Petermann et al., 2020). According to the standard regulations for logging in natural forests in the Amazon, in the absence of technical studies that define the minimum diameter to cut for each species, trees with a DBH equal to or greater than 50 cm can be felled (BRASIL, 2009). In our case study, we observed that 15% of the trees used as reproductive sites would be felled, affecting both the phytotelmata and *T. cunauaru* populations. Therefore, sustainable management may affect hundreds of animal species that use phytotelmata as a resource (Teixeira et al., 2018; Noronha et al., 2021), especially those with reproductive habitats as specific as *T. cunauaru*.

Chemical contaminants from pesticides, fertilizers, and other chemicals can have negative effects on amphibian survival and performance (Egea-Serrano et al., 2012) and have been linked to declines in various amphibian populations (Davidson et al., 2001). However, little is known about the effects of chemical contaminants such as pesticides on amphibians that use phytotelmata. The use of the pesticide malathion, which is used to control mosquito populations, has contaminated the water in the phytotelmata (bromeliads) in restinga areas, affecting the size and larval activity of *Phyllodytes luteolus* (Egea-Serrano and Solé, 2017). Furthermore, in agricultural areas, the use of aircraft for pesticide and fertilizer application is common and can reach phytotelmata at the crop-forest interface.

Invasive species can devastate populations of phytotelmatans, such as bromeliads (Cooper et al., 2014). The bromeliad-eating weevil (*Metamasius callizona*), native to Mexico, Guatemala, and Belize, arrived in Florida through shipments of ornamental bromeliads from a grower in Mexico and caused a 97% reduction in the *T. utriculata* population in the Enchanted Forest Sanctuary in central Florida. The loss of phytotelmatas results in habitat loss, a decrease in biodiversity, and alterations in the cycling and availability of water and nutrients.

Linear infrastructure such as roads can lead to forest disturbance, which in turn negatively affects anurans that rely on bromeliad microhabitats. Linear deforestation, exemplified by the presence of roads, causes changes at forest edges that affect forest structure, microclimate, and dynamics. These changes often extend up to 200 m into the adjacent forest, with some detectable effects extending as far as 500 m from the edge (Laurance et al., 2009). Areas near these roads experience greater diurnal variations in light, wind, temperature, and humidity, and are typically drier and warmer than the forest interior (Pohlman et al., 2009). As a result of these effects, bromeliads in forests adjacent to roads exhibit reduced occupancy and lower anuran abundance compared to those in undisturbed forests located far from roads (McCracken and Forstner, 2014).

Many species of phytotelmata, such as epiphytic bromeliads (Cach-Pérez et al., 2014; Males et al., 2023), and amphibians (Luedtke et al., 2023), are among the most threatened groups of plants and animals due to global climate change because of their high sensitivity to environmental variation. Several species of phytotelmata and amphibians show negative responses to factors such as increases in air and water temperature (Benard, 2015), decreases in hydrological regimes and precipitation (Gervasi and Foufopoulos, 2008), the duration of extreme weather events such as intense wildfires and floods (Walls et al., 2013), and the frequency, intensity, duration, and regularity of cyclical climate patterns (e.g., El Niño, La Niña) (Carey and Alexander, 2003). These events may increase the risk of desiccation for phytotelmata due to increased evapotranspiration (Schlippe Justicia et al., 2023), thereby reducing breeding habitat for many amphibian species. Climate change is also likely to have a negative impact on amphibian breeding in phytotelmata by increasing the risk of disease. Amphibians are vulnerable to various diseases, including chytridiomycosis, which is caused by the fungus *Batrachochytrium dendrobatidis* (Serrano et al., 2022). Chytridiomycosis is a major cause of amphibian declines worldwide (Zipkin and DiRenzo, 2022). Climate change is causing the dispersion of this fungus to new areas, increasing mortality in both larval and adult amphibians. These factors can affect amphibians either directly, indirectly, or in synergy with other stressors, such as predators, competitors, introduced species, and UV-B radiation (Kiesecker et al., 2001).

6 Conservation of amphibian species associated with phytotelmata

Tropical rainforests host the world's greatest biodiversity (Pillay et al., 2022), with amphibians being one of the most diverse groups (Latombe et al., 2021). Many amphibian species use phytotelmata in both terrestrial and arboreal habitats for breeding (Noronha et al., 2021). However, tropical forest canopies are among the most species-rich terrestrial habitats on Earth and remain relatively unexplored (McCracken and Forstner, 2014). Although many studies of phytotelmata use have been conducted since the 1980s, studies of amphibians using canopy phytotelmata in the

Amazon are more recent and have focused on species description (e.g., Guayasamin et al., 2006; Gordo et al., 2013; Ferrão et al., 2019) and aspects of reproductive ecology (e.g., Block et al., 2003; Schiesari et al., 2003; McCracken et al., 2009).

Of the 80 currently recorded species that use phytotelmata in the Amazon, 30 species (37.5%) are classified as threatened to some degree by IUCN Red List categories (see Table 29.1). Eleven species are classified as Endangered (EN), 10 as Vulnerable (VU), seven as Critically Endangered (CR), and two as Near Threatened (NT). In addition, more than 10 of the species occur outside of protected areas or indigenous territories. This finding highlights the need to advance ecological research, including studies of behavior, reproduction, and population dynamics, especially for threatened species located in regions undergoing agribusiness expansion, extensive selective logging, and linear infrastructure projects. The urgency is driven by the increased risk of extinction for certain species.

Studies should also be conducted on the distribution of phytotelmata, especially those located in canopies that can be used by amphibians. This is necessary due to the threats they face, including deforestation, climate change, pollution, and their collection for ornamental and gardening use (Zucchi et al., 2019). The biodiversity of this habitat is well documented for insects, but poorly understood for vertebrates.

7 Strategies and potential monitoring tools for conservation

Many amphibian species that use phytotelmata in the Amazon are threatened. Moreover, there is a lack of research on population size, current distribution, and reproductive biology, which are key factors in the conservation efforts for many of these species. Several captive studies are providing information on the biology of amphibians that use phytotelmata using the captive breeding protocol of Klein et al. (2020). This data can help to increase the knowledge of the reproductive biology of these species and may be useful for ex situ breeding programs for conservation purposes (Regnet et al., 2023).

Captive breeding information has been compiled for several *Ranitomeya* species, including *R. amazonica*, *Ranitomeya benedicta*, *R. imitator*, *R. reticulata*, *Ranitomeya sirensis*, and *Ranitomeya vanzolinii* (Klein et al., 2020), as well as *R. variabilis* (Regnet et al., 2023). This research has provided a baseline for further studies on captive breeding and its implications for conservation breeding programs, as highlighted by Klein et al. (2020). A study of the reproductive behavior of *Scinax alcatraz*, a species with a limited distribution found only on the small island "Ilha dos Alcatrazes," has shown that this species reproduces readily in captivity (Lisboa et al., 2021). In the future, if the population of this species experiences a drastic decline due to various factors affecting amphibian populations, it could be considered for inclusion in an ex situ conservation program for potential reintroduction. This strategy could prove valuable for the conservation of endangered amphibians in Brazil.

The use of artificial breeding sites has been effective for the reproduction of several species that depend on phytotelmata in the forest canopy, such as tree holes, as demonstrated by Noronha et al. (2021). This approach can play a key role in the conservation of local species. Moreover, the use of digital camera traps and the installation of artificial breeding sites (see Noronha et al., 2021) are effective alternatives for obtaining data on behavior and ecology while allowing for long-term species monitoring.

The conservation of phytotelmata-using amphibians cannot be achieved without the conservation of their phytotelmata hosts, due to their relationship (Sabagh et al., 2017). To conserve phytotelmata, conservation strategies can be implemented through habitat restoration programs, avoiding clandestine collection, and creating biodiversity-focused land use policies. This urgent action is crucial to protect these microhabitats due to the high rate of endemism of the aquatic fauna found in phytotelmata, including invertebrates and amphibians.

8 Conclusions

Phytotelmata are essential microenvironments in tropical ecosystems, providing refuge and breeding sites for invertebrate and vertebrate species. The fundamental role of phytotelmata in amphibian reproduction highlights their importance for biodiversity conservation, making their conservation and study essential for understanding the distribution of amphibians that use these environments and also for their conservation. The adaptation and preference of phytotelm breeding species to specific plant species demonstrates their ecological importance in the amphibian community.

Eighty species of anurans use phytotelmata in the Amazon for reproduction, with 30 species (37.5%) classified as threatened to some degree according to the IUCN Red List categories, but 18 (11 species endangered and seven critically endangered) are in the worst threats categories. This scenario underscores the need to advance ecological research, including studies of behavior, reproduction, and population dynamics, especially for threatened species located in regions undergoing agribusiness expansion, extensive selective logging, and linear infrastructure projects.

The use of natural breeding sites by *T. cunauaru* is associated with larger diameter trees subject to selective logging, and artificial breeding sites have been shown to be effective for reproduction of this species. Urgent action is needed to mitigate threats and conserve amphibians and associated phytotelmata. Research efforts, such as the use of artificial breeding sites and camera traps, should

continue to improve our understanding of the behavior of these species and facilitate long-term monitoring. Ultimately, the conservation of phytotelmata and their amphibian inhabitants requires integrated conservation strategies, including habitat restoration, anti-poaching measures, and biodiversity-focused land-use policies.

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Chapter 30

Asia's ecological tapestry: Navigating conservation challenges in the era of global warming

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1 Introduction

Ecosystem services and biodiversity frequently cross geopolitical borders (Erg et al., 2012; López-Hoffman et al., 2010). Numerous concerns related to conserving and regulating biodiversity, such as wildlife trafficking, forest fires, animal migration, watershed restoration, building corridors and connectivity, and transboundary protected areas, are intrinsically transnational (Liu et al., 2007). The landscape level is where many ecological, evolutionary, and social processes become visible; hence, managing biodiversity necessitates activities over wider landscapes (Baldwin et al., 2018). The driving force of both environmentally friendly growth and human security is the intrinsic worth that biodiversity offers. Primary production, crop pollination, disease control, reducing emissions of greenhouse gases and climate change, biogeochemical processes such as nutrient cycling, air and water purification, the provision of food and genetic resources, and advantages to education, culture, and spirituality are just a few of these services (Millennium Ecosystem Assessment, 2005; Ali, 2017). Unbelievably, biodiversity is being threatened by human-caused global changes in land use and temperature from the poles to the tropics, as well as on land and in the seas (Watson et al., 2019).

The most pressing issues facing biodiversity are the comprehensive study of biodiversity to conserve it, considering anthropogenic influences, environmental changes, and evolutionary and ecological processes (Cracraft, 1995; Urban et al., 2016).

The tropical woods found in Southern China and Southeast Asia, together referred to as Southeast Asia, are well known for their abundant and distinctive indigenous flora, many of which can be traced back to the end of the Mesozoic Period, some 65 million years ago (Corlett, 2007). The region has been considered a long-term refuge for extinct plant species due to its constant geological circumstances since the Late Cretaceous epoch. It also features one of the greatest percentages of endemism on Earth (Sodhi et al., 2004) and at least four of the 25 major hotspots for biodiversity worldwide (Tang et al., 2018). However, due to changes in land use, urban population expansion, and human-induced climate change, the tropical forests of Southeast Asia are currently among the most vulnerable land-based ecosystems on the planet. This is because, according to Estoque et al. (2019) and Sodhi et al. (2004), they are losing forest cover at some of the fastest rates since the 20th century. Considering that the conversion of natural ecosystems to plantations or subsistence farms results in an alarming decrease in biodiversity, agricultural expansion is thought to be one of the main drivers of the loss of tropical forests in Southeast Asia (Wilcove et al., 2013; Zeng et al., 2018). Central Asia is a vast region that comprises the center of the Eurasian continent, extending from the Volga River and the Caspian Sea in its western portion to Northwest China in the east at midlatitudes. The five countries of Kazakhstan, Kyrgyzstan, Tajikistan, Turkmenistan, and Uzbekistan comprise an area of around $4 \times 106\,km^2$ (Lioubimtseva et al., 2005).

Extreme weather fluctuations, including high-temperature and heat-wave events, are forecasted to intensify and prolong due to climate change (Abasi et al., 2023). The climate of Central Asia is temperately continental and dry. Almost three-quarters of Central Asia is made up of dry and semiarid regions, where hills and plains are the dominant geographical features (Chen et al., 2008; Yu et al., 2019). Central Asia is dominated by flora types found in temperate and semidesert regions. There are now river valley forests, sandstone desert plants, gravel desert plants, and sporadic bushes (Khojimatov et al., 2023). This region is made up of numerous subregions, a variety of altitudinal belts, and some of the tallest mountain ranges in the world, including

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the Pamir and the Tian Shan. The desert transitions to mountain shrubland across the altitudinal gradient, followed by subalpine, alpine meadows, dark coniferous woodland, deciduous broad-leaved forest, and alpine cushion vegetation.

2 Impact of global warming on Asia's biodiversity

Climate change contributes significantly to the current biodiversity loss and accelerated species extinction, as numerous studies have confirmed. Changes in the global climate have been linked to increases in temperature and carbon dioxide (CO_2) concentrations (Millennium Ecosystem Assessment, 2005; Brook et al., 2008; Guo et al., 2017). Numerous significant changes in the global hydrologic cycle (i.e., precipitation, condensation, and evaporation), in addition to a boost in the severity and frequency of extreme weather phenomena involving floods, cyclones, and severe droughts, are triggered by these consequences, and they will have a detrimental influence on global to regional biodiversity (Rinawati et al., 2013; Adler et al., 2009). Tol (2009) states that, in addition to possessing an important effect on biodiversity loss caused by variations in weather patterns and temperatures, climate change may also have an impact on land use, nutrients, and acidifying the oceans, as well as the arrival of invasive alien species into previously uninvited habitats.

Thomas et al. (2004) have determined that temperature and precipitation can be used as proxies for climatic variation to examine how biodiversity loss is affected by climate change globally. These changes are evident in individual organisms, species distribution, populations, as well as the composition and functioning of ecosystems: all impacted by the effects of climate change, which include precipitation and rising temperatures (Gren et al., 2016; Ali et al., 2014). Similarly, Braimoh et al. (2010) further take precipitation and temperature into account when researching the conservation status of the world's waterbird populations. Research indicates that temperature and precipitation have a positive correlation with biodiversity loss. The loss of biodiversity is exacerbated by rising temperatures and increased precipitation. Amano et al. (2018) also discovered that plant species in Finland have been impacted by climate change. All functional groups' temperature-sensitive species, such as bryophytes and lichens, dwarf shrubs, herbs, and grasses, move northward seeking colder climates. Villén-Peréz et al. (2020) noted rising wildfire frequency, tropical storm intensity, deforestation, and severe flooding.

3 Conservation challenges

Reducing the rate at which forests are being converted worldwide is regarded as a low-cost way to mitigate the effects of climate change (Nepstad et al., 2009; Stern, 2007). With conversion, Asia has more economic competitiveness. Comprehending the extent of the disparity between the costs of conservation and the benefits of conversion in this area is vital for a number of reasons:

(1) Four global biodiversity hotspots, with numerous endemic species spanning several taxa, are located in Southeast Asia (Stern, 2007).
(2) With estimates of their biomass stock reaching 225 metric tonnes of carbon per hectare, Southeast Asia's lowland rainforests are huge repositories of soil and vegetative carbon ($tCha^{-1}$) (Myers et al., 2000).
(3) The rate of deforestation in this region is the highest in the tropics right now (Sodhi et al., 2010). Furthermore, this rate is rising. Over 285,000 km^2 remain as the lowland dipterocarp forests in Western insular Southeast Asia, compared to an initial estimate of over 900,000 km^2.
(4) Because poverty is pervasive (Change, 2006), economics is especially important when deciding how to use land.
(5) Southeast Asia produces in excess of 80% of the global palm oil (Berry et al., 2010) and is home to the greatest log producers globally for veneer, plywood, and lumber (Sodhi et al., 2004).

4 Climatic effect/global warming

The diversified and rich ecological tapestry of Asia includes a wide range of ecosystems, including high mountain ranges, tropical rainforests, deserts, grasslands, and coastal regions. That natural diversity is, however, coming under increased threat from climate change and global warming. The following information relates to Asia-specific climate effects and challenges associated with global warming:

4.1 Glacial loss and river flow

Over the past century, most of the Himalayan glaciers have receded. In this region, the summer monsoon provides 80% of the glaciers' moisture. Because of their heavy reliance on snow and glacial meltwater, according to Immerzeel et al. (2010), the Indus and Brahmaputra basins are especially vulnerable to the effects of climate change on snowmelt and glaciers. It is expected that the late spring and summer flows of the Indus and Brahmaputra will decline significantly over the next few decades, with a shift toward boosting winter and spring runoff well before a 2°C increase in temperature. In a 4°C warming scenario, these changes are expected to become dramatic (Diffenbaugh et al., 2013). The Ganges is less reliant on meltwater because of its substantial yearly downstream precipitation during the monsoon season (Immerzeel et al., 2010). van

Vliet et al. (2013) analyze the variations among the river basins in some depth.

Significant variations in downstream flow extremes may result from a shifting snowmelt regime, loss of glacier ice, and variations in precipitation. For instance, in the future, the Brahmaputra might see extremely low flow levels less regularly (Gain et al., 2011). But there could be a significant spike in peak flow, which is linked to the risk of flooding (Ghosh and Dutta, 2012). When coupled with the anticipated rise in sea level, this could have damaging paraphernalia on Bangladesh and other low-lying areas in the vicinity (Gain et al., 2011).

4.2 Monsoon variability

There has been a drop in the previous few decades, according to recent studies of entire rainfall amounts throughout the rainy season of Monsoon (Bollasina et al., 2011; Turner and Annamalai, 2012; Wang et al., 2012). Consistent signs can be observed in the drop that may be caused, influenced, at least in part, by anthropogenic aerosols like black carbon, even though it does not match the expected consequences of global warming (Bollasina et al., 2011; Turner and Annamalai, 2012). Furthermore, while most research concurs that there is a decline, the extent and importance of this decline depend on the dataset and subregion in which the analysis is conducted. The primary sources of freshwater for human societies are precipitation and river runoff; therefore, fluctuations in monsoon variability are expected to pose serious problems. Particularly challenging are projections of the size and spatial distribution of the heaviest extremes of monsoon rainfall (Turner and Annamalai, 2012).

4.3 Sea level rise

The world's sea level rise is not uniform; it is not expected that the rise will be at the same rate in the future with forecasts of local sea level rise in South Asia showing a much larger increase than in higher latitudes (Perrette et al., 2013). Utilizing the data of various studies, including IPCC AR5 (World Bank Publications, 2014), South Asia's regional sea level increase by the end of the 21st century will be roughly 0.65 m (0.4–1.2 m) in a 4°C world and (0.2–0.7 m) in a 2°C world (compared to 1986–2005). This is 5%–10% greater than the global norm of sea level rise. The sea level rise projections for a 2°C and 4°C future (corresponding to RCP 2.6 and RCP 8.5, respectively) do not start to diverge significantly until the second half of the 21st century, with each projection being around 10 years ahead of the corresponding global mean development.

After 2100, the difference in sea level rise between a world warmed by 2°C and one by 4°C will become more pronounced, which will affect how we adapt to these changes. One of the clearest signs of this difference is the speed of sea level rise, which accelerates much more quickly in high-emission scenarios than in low-emission ones.

It shows end-of-century rates in a 4°C world of 13 mm (about 0.51 in)/year [7–24 mm (about 0.94 in)/year] and in a 2°C world of 4 mm (about 0.16 in)/year [1–7 mm (about 0.28 in)/year]. It should be noted that the effects of local land subsidence brought on by human or natural influences are not included in the projections presented here; these factors must be taken into consideration when estimating the overall impact of sea level rise at local and regional levels.

4.4 Water security

Concerns over future water security are growing in the face of changing climate. Water scarcity is dependent on a complex interplay between population expansion, rising levels of economic and agricultural activity, rising levels of total precipitation, the eventual loss of snow cover and water provided by glaciers, as well as regional variances and shifts in seasonality throughout South Asia. It is challenging to conduct a thorough assessment since different metrics are used in different research to evaluate the threats to water security. Numerous studies have revealed that South Asia is already severely water stressed (Fung et al., 2011; Vörösmarty et al., 2010). Its extremely low per capita water storage capacity makes it more susceptible to variations in monsoon patterns and water flow fluctuations (Ministry of Environment and Forests, Government of India, 2012; Rakhmatullaev et al., 2018).

4.5 Loss of biodiversity

Southeast Asia's biodiversity has been in danger over the past few decades, and by 2100 it is predicted to have decreased by 13%–85% (Sodhi et al., 2010). The main concerns are hunting pressure (Hughes, 2017), habitat loss and degradation from extensive logging, and agricultural development (Imai et al., 2018; Wilcove et al., 2013). There has been a 50% decline in natural forests in lowland areas over the last 20 years (Namkhan et al., 2021), and there has been a corresponding increase in agricultural expansion in montane areas (Feng et al., 2021; Zeng et al., 2018).

The goals of establishing protected areas (PAs) are to halt desertification and deforestation, preserve biodiversity (Deguignet et al., 2014), preserve environments, and sustain animal and plant populations (Geldmann et al., 2013). As a result, PAs have higher global species richness than areas outside of them (Coetzee et al., 2014). However, a number of studies have shown that PAs are biased in favor of vested economic interests and do not necessarily target places with high concentrations of biodiversity (Venter et al., 2018), leaving the ecosystems vulnerable (Tantipisanuh et al., 2016).

The long-term stability of ecosystems, natural habitats, and biodiversity is further threatened globally by climate change (Malhi et al., 2020). Significant temperatures, increased frequency and intensity of rainfalls, or substantial droughts are known to have a direct impact on species distribution, communities, and ecosystem functioning (Peh, 2007; Root et al., 2003; Sintayehu, 2018), with indirect consequences on animal food sources (Van der Putten et al., 2010). However, because Southeast Asia's geographical makeup makes long-term climate change estimates questionable, the impact of the region's changing climate on natural habitats has not been quantified.

There are numerous ways to show the effect of the changing climate on Southeast Asia's biodiversity. For instance, the diversity of sun bear food trees in lowland forests in Indonesian Borneo was reduced by 44% as a result of the effects of fire and drought during the severe El Nino event (1997–98) (Fredriksson et al., 2007). Furthermore, an examination of the elevational distribution of frequently occurring birds in Southeast Asia over 28 years revealed that the species was moving upward in reaction to climate change (Peh, 2007).

5 Challenges and opportunities for Asia in the agricultural area with the climate change

Asia's agricultural zone is confronted with many challenges in addition to possibilities. One of the major challenges is constrained accessibility to water resources. Countries like Pakistan, India, Afghanistan, and some Middle Eastern countries like Qatar encounter restrained annual precipitation in some areas, contributing to an absence of potable water assets for agricultural functions while some parts of these countries are facing extreme downpours (Abbas et al., 2023). Furthermore, the difficulty of water shortage is exacerbated through the accelerated evapotranspiration due to increased temperatures. Additionally, the depletion of groundwater sources exacerbates the problem of water scarcity. The confluence of these elements, coupled with the lifestyles of arid and semiarid regions, results in a considerable reduction in agricultural productivity and a related condition of food scarcity of a national scale (Abbas et al., 2023).

Despite these constraints, the rural industry in Asia has potential for growth and progress. One domain that shows particular promise is the development of smart greenhouses or controlled agriculture, which can streamline useful resource allocation and help precision agriculture (Karanisa et al., 2021). According to Karanisa et al. (2021), integrating advanced technology into smart greenhouses permits streamlined water and power resource control, boosting agricultural output and decreasing the ecological impact on biodiversity.

Furthermore, Asian economies have the potential to expand rural enterprises. Most of these economies have made significant contributions to the development of sustainable agricultural practices through the utilization of renewable resources. The goals can be achieved via using sustainable technologies for irrigation and deploying comprehensive water management strategies (Ibrahim and Shirazi, 2021).

In summary, the agricultural zones in Asia face challenging conditions related to limited water resources, population boom, and inadequate agrarian productivity. However, capacity for improvement does exist through the adoption of smart greenhouse technologies and the use of energy-efficient tools and strategies. By addressing and resolving these challenging conditions and capitalizing on the opportunities already available, Asian economies can enhance their agricultural enterprise and ensure food security for their residents in the wake of the changing climate.

6 Mitigation strategies

Mitigation strategies for conservation challenges in Asia's biodiversity encompass a range of approaches aimed at preserving and restoring ecological balance. These strategies often involve a combination of policy interventions, community engagement, scientific research, and technological innovation.

Published records also suggest that the increase adoption of ecologically friendly technologies can lead to reduced GHG emissions (Agrawal and Gibson, 1999). Therefore, investing in green technology could significantly pay toward the reduction of Green House Gas emissions.

Furthermore, promoting sustainable land use practices is essential for mitigating habitat loss and fragmentation. Agroforestry, for example, integrates trees into agricultural landscapes, enhancing biodiversity while providing economic benefits to local communities (Perfecto and Vandermeer, 2010).

Local communities' involvement in conservation efforts is extremely crucial. Conservation efforts where indigenous peoples are involved as stockholders are usually very successful in the long run (Agrawal and Gibson, 1999).

Additionally, leveraging advancements in technology can enhance conservation outcomes. Geographical Information System (GIS) and Remote sensing (RS) techniques, such as the use of maps and satellite imagery of drones and airborne shuttles, can enable investigators to observe variations in land cover/land use and detect banned activities like poaching and deforestation (Field et al., 2018).

The climate change adaptation and reduction plans have been developed and set into action simultaneously. However, given the ongoing and fast-track changes in climate, there is an urgent need to integrate both adaptation and mitigation approaches within policy and practice frameworks. This integration is essential to effectively address the multifaceted impacts of climate change and enhance the resilience of

socio-ecological systems. In summary, effective mitigation of conservation challenges in Asia's biodiversity requires a multifaceted approach that combines protected area management, sustainable land use practices, community engagement, and technological innovation.

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Chapter 31

Atlantic Forests: Phytogeography and conservation

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1 Introduction

The Atlantic Forest domain is recognized as the hotspot with the highest biological diversity in the world (Myers et al., 2000), housing a high diversity of endemic animal and plant species (Brown et al., 2020; Myers et al., 2000). This domain plays an important role in providing ecosystem services at local, regional, and global scale, such as climate regulation, carbon storage and dynamics, as well as nontimber forest products (Pires et al., 2021). The Atlantic domain extends along the coast from the state of *Rio Grande do Sul* to *Rio Grande do Norte* (in addition to the high-altitude forests of Ceará), covering approximately 28° latitude in Brazilian territory. Originally, this domain occupied approximately 131 million hectares, covering 18 Brazilian states in the most diverse landscapes, from the coast to the interior of the continent under the influence of the main orogens on the Brazilian coast. However, the Atlantic Forest has been heavily impacted by land-use changes along history, resulting in a significant reduction in its original coverage. Only 24% of its original coverage still remains standing, with approximately half of this remaining composed of old-growth and well-preserved forests (Ribeiro et al., 2009; Fundação SOS Mata Atlântica & INPE, 2022).

The Atlantic Forest domain is home to more than 20,000 plant species, 1400 vertebrates, thousands of invertebrates, and all groups with high endemism being considered the hottest of the world's hotspots (Laurance, 2009), and thus is a high priority for conservation because it shelters more than half of the biodiversity in an area equivalent to 1% of the planet (Myers et al., 2000). However, the long historical anthropogenic disturbance since the colonial period caused major extinctions and is a great threat to the remaining species. Currently, it is considered that only 13% of the original coverage are old-growth and well-preserved forests with more than 80% of the remaining fragments being considered small (<50 ha) (Fundação SOS Mata Atlântica & INPE, 2022). Furthermore, only 1% of its protected coverage is represented by original forest (Ribeiro et al., 2009).

The high environmental heterogeneity provides both higher species diversity and ecosystems, and determines multiplos plant formations. The factors that determine this differentiation on a domain scale are diverse, rooted in a context of geological diversification of 600 million years to shape the climate of Eastern Brazil, going up to a local scale, where soils and biotic interactions will determine the communities' organization. Therefore, understanding these levels of organization of plant communities and ecosystems has been fundamental in defining conservation and legislation strategies, which Brazilian legislation imposes specific regulations (Federal Law 11,428/2006).

This chapter presents a context of the natural history of this biome in a chronological perspective of the past, present, and future highlighting its importance in providing ecosystem services for human well-being. Thus, we will approach the Atlantic Forest on a higher hierarchical scale, similar to the domain concept presented by Ab'Saber (2003), as being coherent landscape schemes and subjected to a certain macroclimate with an associated biotic community (fauna and flora). Thus, the Atlantic Forest domain comprises different associated plant formations, but each one is intrinsically associated with biotic and abiotic factors that will determine its distribution and organization in space and time.

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2 Geographic location

2.1 The formation of the physical space of the Atlantic Forest

The natural history of the Atlantic Forest begins in the pre-Cambrian period, approximately 600 million years ago, before the Atlantic Ocean formation, when the South American and African continents formed Gondwana. Successive collisions between the two continents was important for the geological diversification that would shape the entire Brazilian surface. This series of collisions is described as the Brazilian Orogenes Cycle (Almeida, 1967; Delgado et al., 2003). During the Brazilian Cycle, the compression of the *São Francisco* Craton between the Amazon and Congo Cratons resulted in the great bending of the mobile belts, with emphasis on the Atlantic Mobile Belt, which suffered greater deformation, resulting in more rugged and elevated relief. Its spatial distribution matches the original coverage of the Atlantic Forest biome (Fig. 31.1). This energetic event resulted in great metamorphism of regional sedimentary basins and bendings, which resulted in the formation of the *Serra do Espinhaço* in eastern Brazil. This large mountain range, eroded over 500 million years, still remains an important orographic barrier, due to the high resistance to weathering provided by its quartzite origin. In the chronological sequence, the continuum of events derived from the separation of the continents enabled the formation of other orogens that would make up the series of mountain chains in eastern Brazil. Thus, these mountain chains are fundamental to establishing the environmental conditions of the Atlantic Forest domain, such as the *Serra da Mantiqueira* in the south of Minas Gerais, *Serra do Mar* and *Serra Geral*, extending from *Rio Grande do Sul* to *Espírito Santo*, and the Borborema plateau in northeastern Brazil, this being the most recent and outside the area of influence of the Atlantic Mobile Belt, which involves the states of *Alagoas*, *Paraíba*, *Pernambuco*, and *Rio Grande do Norte* (Oliveira and Medeiros, 2012).

From the western edge and heading toward the coast, the Atlantic Mobile Belt appears geologically diverse (de Paula Silva et al., 2021). Overall, the western edge is represented by the quartzites of the *Serra do Espinhaço* escarpment and the eastern edge is represented by seas of hills ("mar de morros"), a sequence of homogeneous hills derived from gneiss rocks. In the portion closest to the coast, outside the Atlantic Mobile Belt, there is the presence of Tertiary sediments, such as sand, clay, limestone, and other debris, which accumulated over millions of years under different environmental conditions. This geological diversity also provides an expressive diversity of landscapes, with scarps, ridges, and inselbergs at high elevations of resistant rocks (quartzites and granites), colluvial slopes, seas of hills, depressions, and coastal plateaus (Schaefer, 2013; Fontana et al., 2023). Consequently, pedogenesis acting under these different conditions allows the formation of a wide range of soils shaped mainly, but not exclusively, by climatic factors.

Due to pronounced orographic barriers, climatic conditions over its domain coverage region become quite diverse (Peel et al., 2007; Alvares et al., 2013). In the coastal portion, a tropical climate type without a dry season is established, where the driest month receives more than 60 mm (Af—according to Köppen classification) or even more intense rainfall, driven by the humid monsoon currents promoted by the orogens (Am). Toward the mountains, in the lower altitude portions, a seasonal tropical climate with a dry winter (Aw) is established. In the higher altitude portions, mainly between latitudes 20° and 30°S, the climatic condition is considered subtropical, and may present dry winters and hot (Cwb) or mild (Cwa) summers even in the tropical portion and, to the tropic south, does not have a dry season, but hot (Cfb) or mild (Cfa) summers. In the low portions west of the orographic barriers, the contact of humid currents reduces significantly, to the point of establishing the driest climate in the biome, with more than 70% of precipitation concentrated in the summer months (BSh), as in the basin of middle and lower *São Francisco* River.

At the top of the orogens, under super-humid conditions promoted by orographic rains, there are diverse environmental conditions that determine the establishment of grasslands to forest physiognomies. In mountains, ridges, and peaks resistant to weathering, pedogenesis becomes slow, resulting in the formation of shallow soils, poor in nutrients, with a high aluminum content, but with high organic matter concentrations (Leptosol, Histosol, Cambisols, Humic or Histic, and Podzol) (Schaefer, 2013; Schaefer et al., 2016). These conditions favor the occurrence of rupestrian grassland vegetation to shrubs, and there may be the occurrence of low-height rainforests in pockets of soil (dwarf forest). In the context of the Atlantic domain, the grassland on granites/gneisses, quartzites or even iron formations, are part of the vegetation complex presented in the domain.

Still under the humid climate influence, but in the lower altitude portions of the orogens, the chemical and biological processes intensity provides greater decomposition and formation of humic acids and intensifies the deepening of soils resulting from the decomposition of the matrix rock. Consequently, there is greater deepening of the weathering mantle, thicker saprolites, and more leached soils (Bigarella et al., 1994; Ab'Saber, 2003; Schaefer, 2013). This bioclimatically active condition on crystalline rocks (such as gneiss and granite of the *Serras da Mantiqueira* and *Serra do Mar*) provides high groundwater recharge and a greater density of the perennial drainage network (Schaefer, 2013). This context tends to produce reliefs with more rounded shapes due to the carving of slopes with more friable soil, a process of relief dissection that culminates in

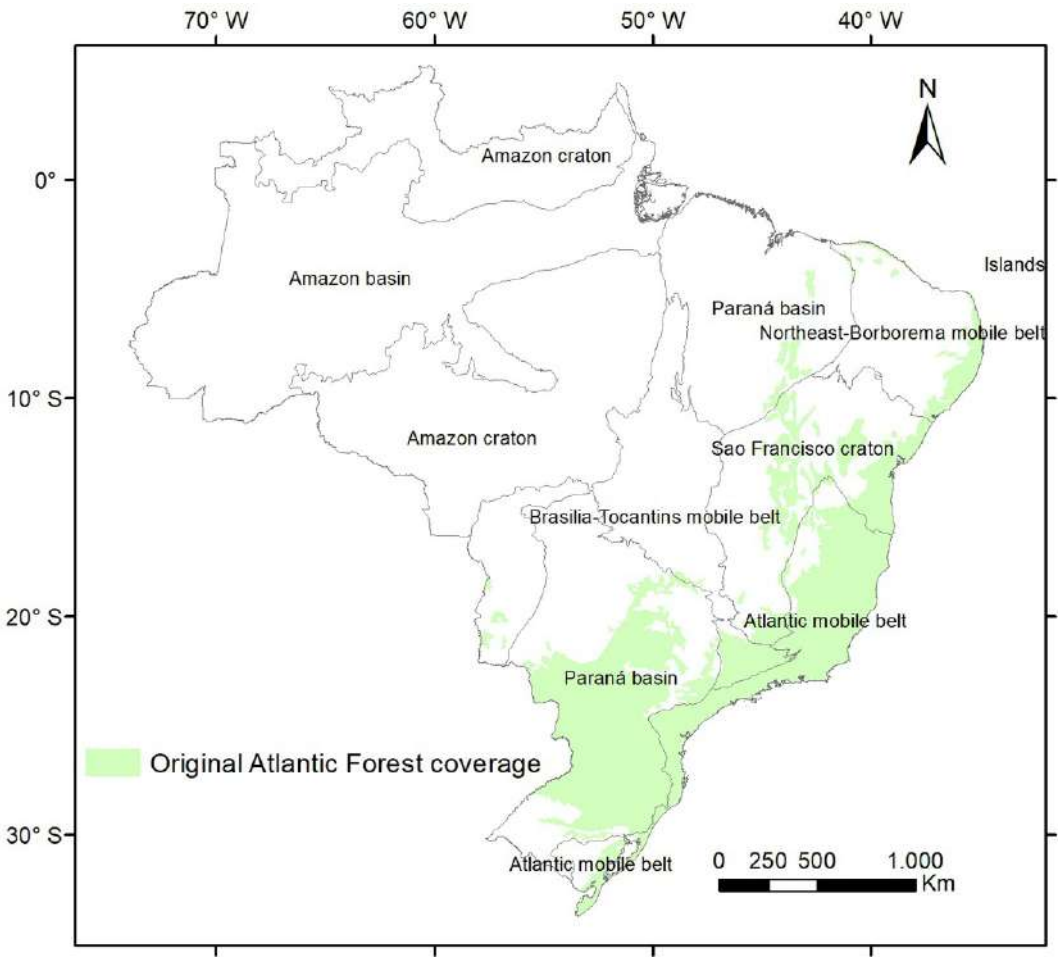

FIG. 31.1 Original coverage of the Atlantic domain over the structural provinces of Brazilian territory. *(Credit: Geodatabase from Labgeo, UFV.)*

seas of hills (Ab'Saber, 2003). These are forms with convex slopes, with very deep, leached, and dystrophic soils, hydromorphic fluvial terraces, and a perennial river system (Fontana et al., 2023). These are favorable environmental conditions for the extensive seasonal forest development, such as the *Zona da Mata Mineira*.

Similarly, in lower altitude areas in the moisture shadows or lee side of orogens, there is not the same water supply, causing the climate to be established with greater seasonality. In the seas of hills at lower altitude (less than 500 m above sea level), the more seasonal climate allow for eutrophic and more clayey soils, due to less leaching, such as in the middle *Paraíba do Sul* basin in *Minas Gerais* and *Rio de Janeiro* or *Mucuri* River basin (Fontana et al., 2023). In the western portion of the *Espinhaço* Range, the drier climate intensifies physical processes, with greater mechanical disintegration than chemical decomposition of the matrix rock, and greater surface erosion and soil removal due to torrential rains typical of seasonal environments. Consequently, the soils are shallower, rich in nutrients due to greater contact with the matrix rock and low leaching, with stony surface layers or even exposure of the matrix rock. Due to high erosion rates, this climate type promotes the flattening of the landscape, with erosion of slopes and clogging of base levels (Bigarella et al., 1994; Ab'Saber, 2003). These favorable environmental conditions promote more open plant physiognomies and deciduous forests, such as in the Middle *São Francisco* River basin in *Minas Gerais* and *Bahia*.

The portion closest to the coast, represented by predominantly flat landscapes, the accumulation of sediments derived from river transport and million-year erosion of orogens (most prominent between the Pliocene and Pleistocene) promotes the formation of the coastal tablelands (Barreiras Group), with soils clayey, deep, and poor in iron (predominance of Argisol). This sedimentary cover is present on a large part of the Brazilian coast, from Rio de Janeiro to Amapá State, and can range from 10 to 160 km. Due to this large extension over the coast, the tableland region presents different climate types, ranging from a semiarid climate (BSh) to a humid tropical climate without a dry season (Af) (Corrêa et al., 2023). In places

where there is climatic seasonality, their soils may present characteristic hardness in the subsurface, due to the accumulation of kaolinite clay and exchangeable aluminum (Santos et al., 2018; Corrêa et al., 2023). These environments were originally represented by large forests (predominantly ombrophilous/rain forests).

The river drainages of the coastal plateaus are narrow, with little or no terrace formation close to the coast. In this context, alluvial vegetation can be affected by tides up to 50 km away from the coast, forming saline swampy environments and favoring the establishment of mangrove vegetation (Corrêa et al., 2023). In small depressions on the coastal tablelands, where there is a predominance of sandy material from local pedogenesis, they form Podzols with high acidity and are nutrient poor. These sandy islands in the middle of the coastal tablelands are obstacles to culture or even the establishment of forest, enabling the development of *Muçununga*, a low-biomass native vegetation that presents a grassland to savannah appearance (Saporetti-Junior et al., 2012).

The coastal sandy areas have a marine origin, and form a complex ecosystem on the fixed dunes, called *Restingas* (Magnago et al., 2010). Sand deposition in this environment is recent (Quaternary) and occurs in ridges with different altitudes (4–8 m), in response to tidal sand deposition at different moments of oceanic transgression and regression (Vidal-Torrado et al., 2023). The *Restinga* soils are markedly sandy, acidic, and poor in nutrients, with the formation of the Arenosol and Podzol classes in the higher portions of the sandy ridges and Gleysols, Histosol, or Fluvisols in the lower portions that are floodable due to the elevation of the water table (Magnago et al., 2010; Vidal-Torrado et al., 2023). In this way, the vegetation that is established captures its nutrients from the rapid cycling in the surface layers and from the saline spray.

The geological diversity and variability of landscapes briefly described here prevails along a few kilometers perpendicular to the coastline throughout the entire latitudinal strip of the Atlantic Forest domain (Fig. 31.1). Consequently, this environmental heterogeneity promotes higher species diversity, plant communities, and physiognomies.

2.2 Patterns of diversity in the Atlantic Forest

The Atlantic Forest sensu lato (or sensu *latissimo*, with the inclusion of deciduous forests—Oliveira-Filho et al., 2006) exhibits a great diversity of forest formations. This diversity is recognized by the protection law (Atlantic Forest Federal Law n° 11,428/2006), which includes not only the forest formations of dense rainforest, mixed rainforest, open rainforest, semideciduous seasonal forest, deciduous seasonal forest, but also mangroves, restingas, high-altitude grassland, swamps, and forest enclaves in the northeast as an integral part of the Atlantic Forest (Fig. 31.2). Among the factors that most determine these formations and species distribution is altitude, since forest formations can be subcategorized into alluvial, lowland, montane, and submontane (IBGE, 2012), and continentality by the distance from the coast (Oliveira-Filho and Fontes, 2000; Oliveira-Filho et al., 2006).

The environmental condition of each formation provides specific biota, with higher species diversity and endemism, which is extremely important for its conservation. To date, the Atlantic Forest is the domain with the most species of vascular plants described, with 16,646 species organized into 225 botanical families (Flora e Funga do Brasil, 2024). Among these species, the National List of Endangered Species (Ordinance MMA No. 148/2022, whose threat category is determined by CNCFlora, 2024) includes approximately 12% of the total plant species in the Atlantic Forest (2062 species according to the data organizer Plantminer, 2024; with 1864 species endemic to the Atlantic Forest), which are threatened to some degree due to habitat loss or excessive and unsustainable exploitation.

The rainforest (or dense ombrophilous forest), also considered the Atlantic Forest sensu stricto (Oliveira-Filho and Fontes, 2000), comprises the coastal rainforests (both on the tablelands and in the restingas) and the high-altitude or montane forests on the humid faces of the main Brazilian orogens, and may occur in mountainous regions of the Northeast Caatinga matrix (Silvera et al., 2020). The rainforest presents high thermal variation, an average of 18.5°C in winter and 23.5°C in summer, and annual precipitation greater than 1420 mm (Arruda et al., 2017), being lower in the forests of the coastal tablelands and higher on the mountains that promote orographic rain. Given the great altitudinal variation, this has been considered the main factor determining diversity, species distribution, and biomass stock (Joly et al., 1999; Alves et al., 2010). This formation has the greatest species diversity among other Atlantic Forest formations, with the occurrence of 9713 species of flora recorded (Flora e Funga do Brasil, 2024). Overall, 14% (1350 species) present some degree of threat, of which 71% (965 species) are endemic to this formation. The greatest species similarity in this forest occurs between the semideciduous forests of the Atlantic domain (Oliveira-Filho and Fontes, 2000).

The mixed ombrophylous forest is characterized by a high density of gymnosperm species, such as *Podocarpus* spp., but mainly *Araucaria angustiflolia*. These forests occur in soils derived from volcanic rocks, rich in Fe and very acidic, from the basaltic plateaus of the Paraná sedimentary basin. Overall, the soils present with a high sum of bases, but with high aluminum saturation (Arruda et al., 2017; Dalmolin et al., 2023). These forests represent the wettest part of the Atlantic biome's forest gradient. Due to the high altitude (average of more than 800 m above sea

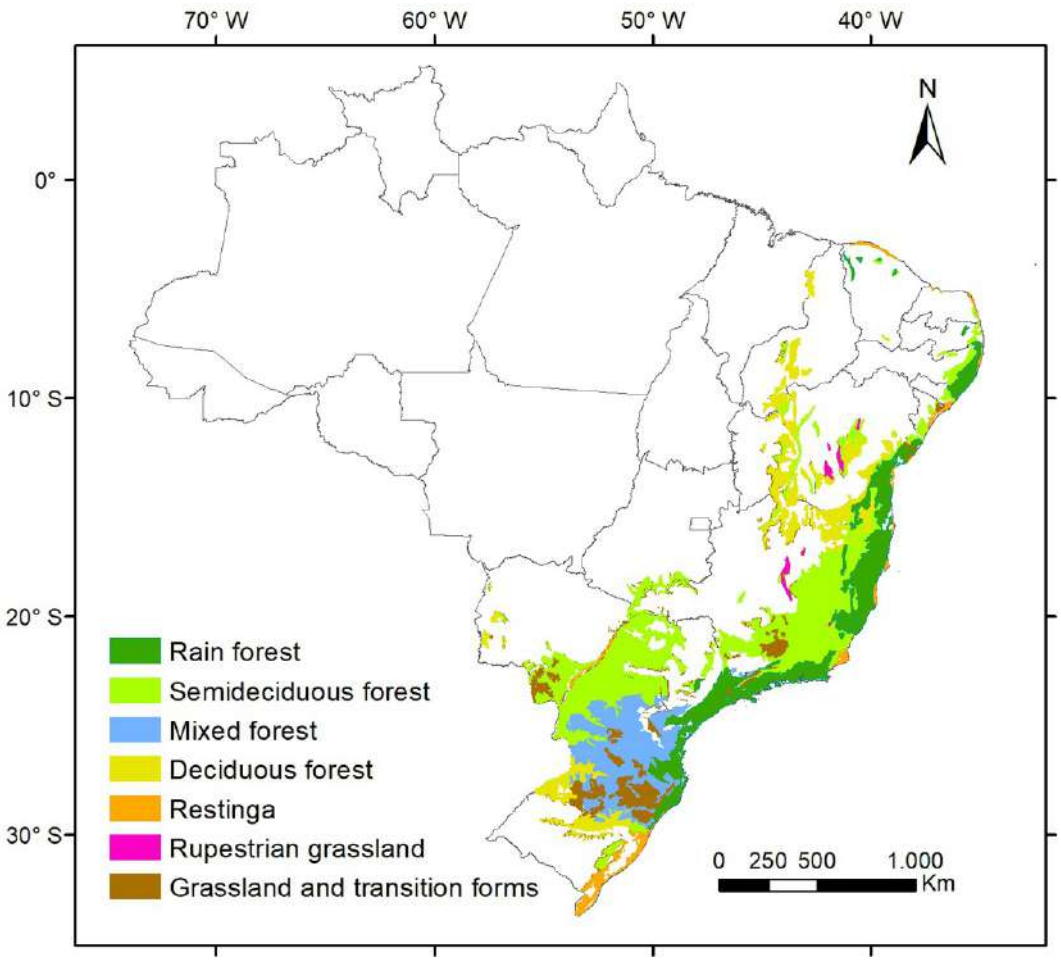

FIG. 31.2 Distribution of plant formations in the Atlantic domain considered by the Law Federal n° 11,428/2006. Map lines delineate the units of federation. *(Credit: The authors.)*

level), it has the highest precipitation (average of 1650 mm/year), and lower temperatures, both in winter (13.7°C) and in summer (21°C) (Arruda et al., 2017). Due to low temperatures, with less activity of decomposers in the soil, and high aluminum levels that complexes organic matter in the soil, the Araucaria plateaus, together with the mountain system of southeastern Brazil, are indicated as the areas with the greatest belowground carbon stock (Gomes et al., 2019). In this formation, 1621 species of vascular plants are recorded, 86 of which are considered threatened to some degree, with 18 species endemic.

Semideciduous Seasonal Forests occur in the interior portion, generally associated with seas of hills, with average altitudes close to 500 m above sea level, or on mountains contrary to the humid Atlantic winds, subjected to a seasonal climate. It also occurs as riverine forests that penetrate the *Cerrado* domain, enabling floristic contact with the Amazon (Oliveira-Filho and Fontes, 2000). The dry season in these environments allows partial deciduousness in forests, causing 20%–50% of individuals in the canopy to lose their leaves to reduce water loss through transpiration (IBGE, 2012). Average temperatures are 24°C in summer and 19°C in winter, with an annual precipitation of 1340 mm (Arruda et al., 2017). Floristic variation is expected at different scales within this formation, with changes in the dominance of species and biomass among regions (Oliveira-Filho and Fontes, 2000), or even at a landscape scale, when edaphic variations between top and bottom determine changes in the diversity and biomass stock (Ferreira-Júnior et al., 2007). In this formation, 3477 plant species are described for the semideciduous forest, with 207 species classified in some threat category, of which 53 are endemic.

The Deciduous Seasonal Forest is the most inland portion of the Atlantic Forest gradient (Oliveira-Filho et al., 2006). This forest has high deciduity during the dry period, when more than 50% of its individuals lose their leaves (IBGE, 2012). These forests have lower precipitation (average close to 1000 mm/year), and higher temperatures with averages of 25°C in summer and 22°C in winter

(Arruda et al., 2017). These forests have the richest soils with higher available nutrients, since they show low levels of exchangeable Al. In the states of *Minas Gerais* and *Bahia*, occur predominantly on limestone soils from the Bambuí Group, when it has high fertility and good drainage (Arruda et al., 2015). This vegetation differs mainly in relation to the portion of the landscape, and can occur in lowland soils or on limestone slopes, presenting different patterns of richness and biomass (Scariot and Sevilha, 2005; Arruda et al., 2015). The flora is numerically inferior when compared to other wetter forest formations, with 840 registered plant species, and 24 species considered to be at some level of threat. However, deciduous forests exhibit high beta diversity (DRYFLOR, 2016), presenting a high differentiation degree or low floristic sharing even between spatially close communities (Arruda et al., 2020).

In relation to the *Campos de Altitude* (highland grassland), its differentiation from the *Campos Rupestre* (rupestrian grassland) has been widely discussed. Vasconcelos (2011) provides an extensive review of these concepts, which propose differentiation in relation to altitude and geological origin (*Campo de Altitude* being granitic/gneissic >1500 m a.s.l., or *Campos Rupestres* being quartzite of 800–2000 m a.s.l—Alves and Kolbek, 2010). Both generally occur associated with rocky outcrops or even on shallow soils (Leptosol or Cambisol and pockets of organic soils—Histosol or Podzol), with significant isolation promoted by the altitudinal difference in relation to the vegetation matrix (Cerrado, Caatinga, or Atlantic Forest). It comprises physiognomy with a herbaceous stratum and shrub dominance, with rare occurrences of trees, except in a portion with greater soil accumulation, forming a true rupestrian grassland complex (Schaefer et al., 2016).

The higher environmental variability that exists along an altitude gradient of the *Campos Rupestres* promotes effective floristic variability, resulting in high species richness on a local scale (Mota et al., 2018; Campos et al., 2018). The ancient character of its environmental context (with less pedogenetic evolution and greater climatic stability, predominantly colder due to altitude) and its degree of isolation enable the maintenance of high species richness and endemism, with taxa that refer to ancestral lineages (Silveira et al., 2016). Comparatively, the genera present in these *Campos Rupestres* of different geological origins do not show significant separations, although the proportion of endemic species is significantly higher in quartzite grasslands (Alves and Kolbek, 2010). Overall, 1755 species of vascular plants are recorded for the C*ampos Rupestres*, compared to 2298 in the *Campos de Altitude*. In addition to the high endemism level, the vegetation on rocky outcrops presents a high level of threat, with 488 threatened species. In the context of the Atlantic domain, *Campos de Altitude* on granites and gneisses, the *Campos Rupestres* on quartzites or even the C*ampos Rupestres ferruginosos* associated with iron formations are part of the domain, synonymized as rupestrian grasslands (Schaefer et al., 2016).

In an edaphic context of purely sandy, acidic soils and low nutrient content, the *Restingas* appear to be a highly diverse complex of plant formations. Regarding the sandy ridges and the water table dynamics, different environments are classified, which can be permanently flooded, seasonally flooded, or permanently dry, with physiognomic variations from fields to forests (Pereira, 2003). Therefore, the effect of the water table tends to be more decisive in the establishment of plant communities than other soil factors (Magnago et al., 2013). Considering the entire complex of *Restinga* formations, 2646 plant species are described. High diversity translates into low floristic similarity, or high beta diversity (Magnago et al., 2011). However, as *Restingas* occur in the coastal strip, the place with the greatest real estate pressure in the country, presents a high level of threat. Of the total species described for *Restingas*, 212 are under some level of threat and 94 are endemic.

The environmental condition of each formation provides specific biota, not only into great diversity, but also into a high endemism. This characteristic confers extreme importance for the conservation of this vegetation complex that is the Atlantic Forest.

3 Human activities and their impacts

3.1 Habitat loss

The cover change use in Brazil has reflected several economic cycles over the last five centuries. Initially, the exploitation of Pau-Brasil (*Paubrasilia echinata*) eliminated around 600,000 ha of coastal forests (Cabral and Cesco, 2008). Following the extractive cycle, new cycles of land-use change and forest loss, such as sugar cane, gold, coffee, and livestock, as well as urbanization and industrialization occurred. The intense forest cover losses without regulation took place until the third decade of the 20th century, when the first Brazilian forestry code was introduced, which established the obligation to have forest reserve areas on properties. However, low enforcement until the end of the 20th century resulted in a high forest fragmentation, which today remains predominantly characterized by small fragments in the steepest portions of the landscape and in soils of lower fertility. Only after Federal Law No. 11,428/2006, the remaining primary or secondary forests in the medium and advanced successional stages became protected from logging, imposing compensation standards for public utility projects and environmental crime for offenders.

Currently, the coverage of primary and secondary fragments with an area greater than 0.5 ha (MapBiomas Project,

2023) comprises 27% of the forest cover in the biome (Fig. 31.3). Minas Gerais was the state with the largest original forest coverage, although 29 million hectares, or 73% of the forest area, was converted to another land use (Fig. 31.4A). Among the biome's forest formations, the greatest losses were observed for semideciduous forests (Fig. 31.4B). These forests had 80% of their coverage reduced throughout Brazilian territory, with the greatest losses in the sea of hills of Minas Gerais. Among the main activities that contribute to land-use change in these environments are livestock farming and coffee cultivation in the higher altitude areas, with semideciduous forests being most affected by these activities. Among the regions of Minas Gerais that maintain the greatest forest expression, Jequitinhonha and the North of Minas Gerais are highlighted, with a predominance of deciduous forests (Fig. 31.4C) although they are also notable for the greatest losses of vegetation cover in the state since the beginning of this century (Pereira et al., 2024). These forests are still being suppressed at high rates in the northern region, annually losing 1.2% of the area, with deforestation surpassing natural regeneration (Dupin et al., 2018). These losses are the result of the expanding livestock and eucalyptus plantation activities in the most seasonal areas of the Atlantic Forest domain. The Espírito Santo, Rio de Janeiro State, the coastal portion of São Paulo, and Bahia states concentrate the greatest losses of dense montane and tableland rainforests. These areas were the target of the first Brazilian economic cycles, while the Paraná and Santa Catarina States concentrate the losses of mixed ombrophilous forests due to agricultural expansions and the demands for coal from the industrialization process at the beginning of the 20th century.

Federal Law No. 11,428/2006 provides the suppression of Atlantic Forest formations for mining purposes, provided that compensation occurs in an ecologically equivalent area within the same region. However, the intensification of the suppression of ferruginous *Campos Rupestres*, combined with

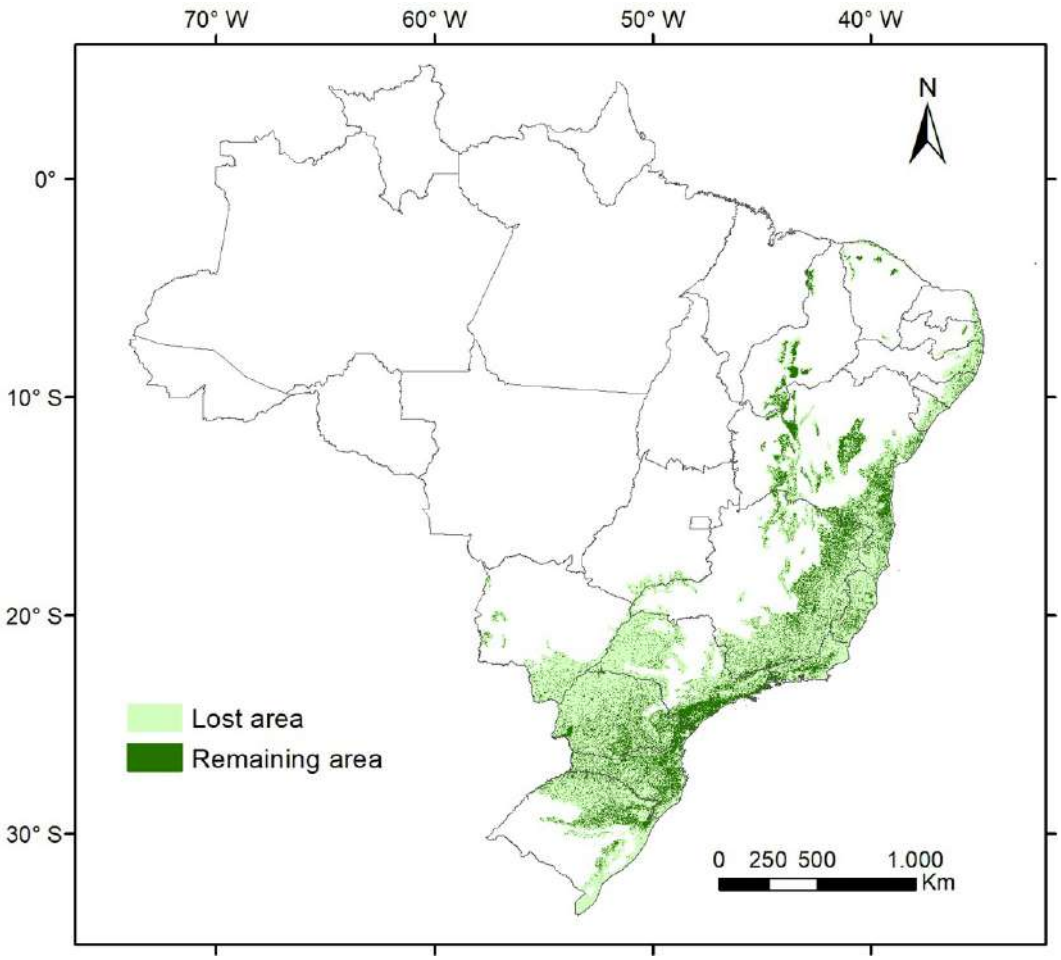

FIG. 31.3 Distribution of forest remnants over the original coverage of the Atlantic domain. Map lines delineate the units of federation. *(Credit: Geodatabase from the MapBiomas Project 2023. Collection 8 of the Annual Land Cover and Land Use Maps of Brazil (1985–2022). MapBiomas Data, V1. https://doi.org/10.58053/MapBiomas/VJIJCL.)*

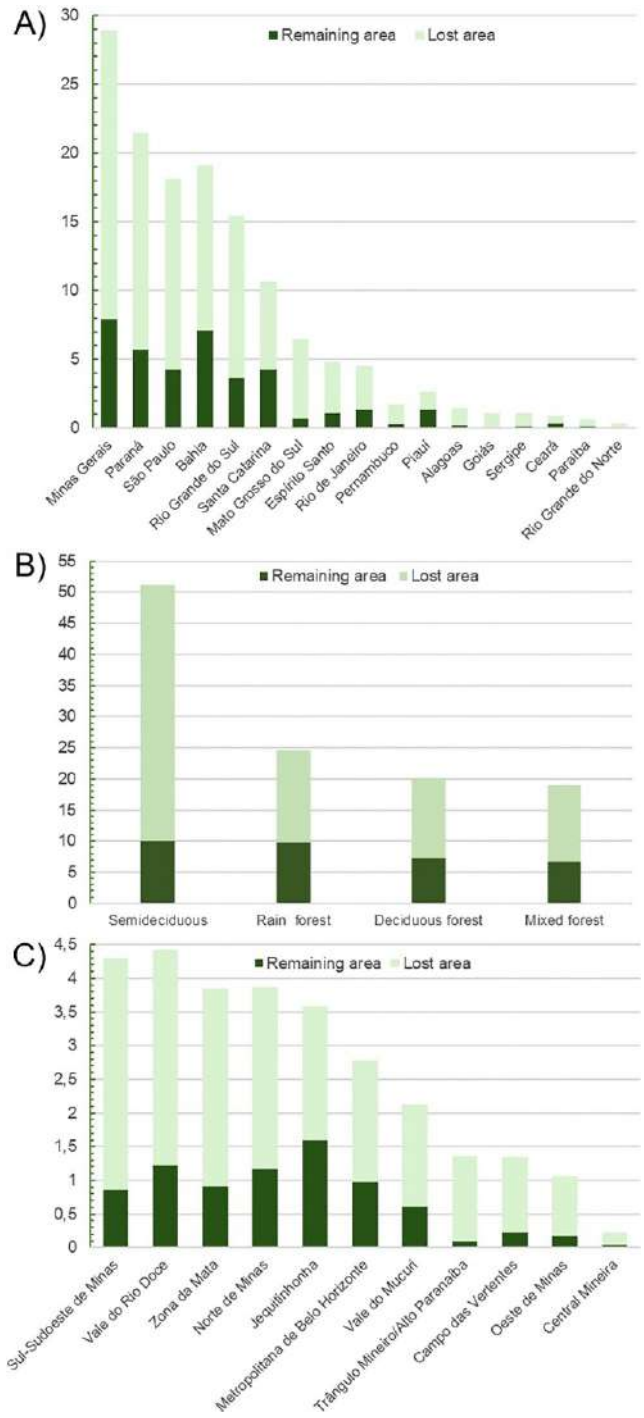

FIG. 31.4 (A) Relationship of original area and forest remnants of the Atlantic Forest among the states of Brazil; (B) among forest formations in Brazil; and (C) among the regions of the Minas Gerais state. *(Credit: Database from the MapBiomas Project 2023. Collection 8 of the Annual Land Cover and Land Use Maps of Brazil (1985–2022). MapBiomas Data, V1. https://doi.org/10.58053/MapBiomas/VJIJCL.)*

the significant expansion of mining concession areas, has led to the redirection of compensations to quartzitic *Campos Rupestres*. Although these fields exhibit similar physiognomy, their geological context allows for a distinct species assembly (Massante et al., 2023). The same happens with forest formations. Evidently, this type of compensation is not effective, although its prohibition is not duly explicit in law.

In addition to habitat loss through deforestation and land-use change, major losses are expected from environmental suitability changes promoted by climate change (Arruda et al., 2024). The estimate for the next 50 years is that half of the current coverage of the Deciduous Forest in Brazil will be subject to collapse, due to the establishment of an environmental suitability similar to the caatinga (with increased seasonality of rainfall, temperature, and lower annual precipitation), with a significant impact on the northern region of Minas Gerais and Bahia. Significant losses are also expected for the remnants of dense rainforest on the northeastern coastal tablelands (Arruda et al., 2024).

3.2 Functional changes and losses of ecosystem services

The land-use changes in Atlantic Forest induce biodiversity loss, consequently affecting multiple ecosystem functions and services (Tabarelli et al., 2005). Furthermore, climate change results in increases in global mean temperature, as well as changes in the frequency and severity of extreme droughts (Fauset et al., 2012). Thus, several regions of South America have experienced a decrease in total annual precipitation, making tropical forests increasingly drier due to the greater intensity of periods of low precipitation (Arruda et al., 2024). These periods of prolonged drought have affected the structure, dynamics, and diversity of tree communities (Fauset et al., 2012; Esquivel-Muelbert et al., 2019; Aguirre-Gutiérrez et al., 2019), increasing the loss of biodiversity and ecosystem services (Poorter et al., 2017; Walker et al., 2020). In this scenario, the forests are adjusting the composition of functional groups through characteristics based on the plant's life history, such as phenology, replacing evergreen species by deciduous species and promoting profound changes in the ecosystem functioning (Aguirre-Gutiérrez et al., 2019).

In relation to nonforest formations, the high concentration of organic carbon in the mountainous soils of rupestrian grasslands makes these environments important carbon sinks (Gomes et al., 2019). However, even their Histosols are highly threatened with extinction, as climate change can cause decomposition of stored organic carbon and its emission into the atmosphere (Benites et al., 2007). The large contribution of organic carbon to the soils of natural ecosystems and the risks arising from its release due to climate change have been largely neglected (Huang et al., 2024).

Human vulnerability and ecosystem vulnerability are interdependent (IPCC, 2022). Therefore, initiatives that aim to understand the current and future effects of climate

change on biodiversity, as well as the underlying processes driving ecosystem changes, are extremely relevant and necessary. Therefore, to maintain climate security, measures to conserve biodiversity and ecosystem services on which humankind depends as a society must involve nature-based solutions to maintain cobenefits between biodiversity and ecosystem services. As forests uptake higher carbon emissions and help regulate the planet's temperature, the Atlantic Forest has an important role in these discussions and mitigation actions.

However, the climatic changes mitigation actions to reduce the loss of biodiversity and carbon stock of tropical forests is a greatest challenge (Matos et al., 2020). This highlights the importance of focusing more efforts on projects that bring cobenefits between carbon conservation, in order to reduce CO_2 emissions and biodiversity conservation simultaneously (Osuri et al., 2020; Matos et al., 2020). These cobenefits between biodiversity and carbon can occur in priority areas where carbon storage can be added to high biodiversity (Osuri et al., 2020). In this context, the Atlantic Forest, considered a hotspot should be a region of high priority for biodiversity and carbon conservation (Magnago et al., 2015). Therefore, it is essential to define a conservation and management plan for the forest remnants, that despite being fragmented and strongly affected by anthropogenic disturbances and climate change, the Atlantic Forest still has a high biodiversity, providing various goods and services to the population. It is worth highlighting that ecosystem services such as pollination, carbon storage, and water supply of river basins, are maintained by the Atlantic Forest for around 72% of the Brazilian population, with more than 145 million people in 3429 municipalities, equivalent to 61% of those existing in Brazil (Hirota, 2022). Therefore, it is necessary to invest more in its preservation, recovery, and restoration due to its environmental and socioeconomic importance. Furthermore, carbon emission is increasing the growth and mortality rates of trees; thus, tropical forests are changing from carbon sinks to sources. Therefore, it is essential to understand how climate affects forest dynamics using potential monitoring tools, in order to predict and adjust the conservation and management methods.

4 Potential monitoring tools for conservation and management

Threats to the Atlantic Forest biome persist due to continuous land-use change (habitat fragmentation), biological invasions, and climate change. Under this challenging scenario, the effective monitoring tool development and implementation to support conservation and management is essential. These tools are important for assessing and monitoring the current conservation status, identifying dynamics and patterns of vulnerable areas over time, which allow a rapid response to illegal deforestation activities (Calders et al., 2023; Dalton et al., 2023; Song et al., 2023). Thus, one of the most effective monitoring methods is through the establishment of permanent plots and carrying out continuous forest inventories for a long time (Phillips, 2023). This method provides information about changes in the stand age communities attributes, such as structure, diversity, composition, estimating forest growth and production, understanding ecological processes, and quantifying the disturbances impacts over time (de Souza and Soares, 2013). The use of permanent plots has been widely adopted in the Atlantic Forest, as evidenced in the Minas Gerais Forest Inventory (Oliveira-Filho and Scolforo, 2008) and the National Forest Inventory (SFB, 2024a).

The Minas Gerais Forest Inventory was designed based on a detailed mapping of the native flora and reforestation in the state. Subsequently, qualitative and quantitative inventories were carried out to identify species diversity and estimate the volume and carbon stock in these forests (Oliveira-Filho and Scolforo, 2008). The National Forest Inventory is an initiative established by the new Forest Code and coordinated by the Brazilian Forest Service that aims to produce comprehensive information about forests in all biomes in the country. This project involves carrying out a socio-environmental survey, landscape analysis, and collecting biophysical data from forests, including information on soil, necromass, litter, land-use classes, and vegetation (SFB, 2024a). The information obtained through these projects is essential for guiding public policies and supporting forest planning and management, contributing significantly to the monitoring and sustainable management of the Atlantic Forest.

Although the method based on field data collection is effective, the use of new technologies associated with remote sensing is essential to increase the accuracy and scope (spatial and temporal) of environmental monitoring and biome conservation activities (Kattenborn et al., 2021; Radočaj et al., 2020). Remote sensing obtains information through sensors, often embedded in aircraft or satellites, which record different types of electromagnetic radiation that are reflected or emitted by the Earth's surface (Lechner et al., 2020). Multi- and hyperspectral optical sensors, for example, have the capacity to provide information such as: vegetation cover, land use, water quality, fire hotspots, and quantitative (biomass and carbon) and qualitative (detection of species of interest) vegetation parameters (Florath and Keller, 2022; Lehmann et al., 2023; Miyoshi et al., 2020; Pereira et al., 2024). The incorporation of these sensors in missions such as Landsat, launched by the National Aeronautics and Space Administration (NASA) and United States Geological Survey (USGS), Sentinel, developed by the European Space Agency (ESA), and CBERS, the result of a partnership between Brazil and China, has been fundamental in

ensuring continuous monitoring of changes in the Earth's surface over time (EMBRAPA, 2018; Radočaj et al., 2020).

The use of unmanned aerial vehicles (UAVs) and technologies such as light detection and ranging (LiDAR) also represents significant advances in the field of remote sensing and environmental monitoring (Coops et al., 2021; Guimarães et al., 2020). The UAVs (drones) are used to collect high-resolution geospatial data using specialized cameras and sensors, allowing the capture of detailed images that are subsequently processed to produce topographic maps, digital elevation models, and high-quality orthophotos (Guimarães et al., 2020; Koh and Wich, 2012). Furthermore, the UAVs has stood out as an effective tool in forest restoration, allowing the assessment of vegetation recovery and seed distribution in extensive and difficult-to-access areas (Albuquerque et al., 2021; Robinson et al., 2022). LiDAR is a technology that uses laser pulses to measure the distance between the sensor and the surface. When integrated UAVs and LiDAR make it possible to obtain three-dimensional data, detailing the topography and vegetation profile provides quantitative forest stand parameters such as biomass and carbon (Camarretta et al., 2020; Mehendale and Neoge, 2020). Furthermore, LiDAR is also used in large-scale monitoring, such as in NASA's GEDI (Global Ecosystem Dynamics Investigation) project, which seeks to investigate the effects of climate change and land use on the forest structure and growth dynamics (Dubayah et al., 2020; Duncanson et al., 2022).

All these technologies, combined with innovative storage and processing data, have been applied by initiatives such as MapBiomas (MapBiomas Project, 2024) and Brazilian Forest Service (SFB, 2024c) to promote conservation and recovery of the Atlantic Forest. MapBiomas is a collaborative initiative that brings together nongovernmental organizations, universities, and technology startups for the transformations of the Brazilian territory through annual land cover and use mapping. The project also carries out monthly monitoring of the water surface and fire validating and preparing detailed reports for each deforestation event. These activities are conducted with the support of distributed and automated data processing in partnership with Google Earth Engine (MapBiomas Project, 2024). The Brazilian Forest Service has strategically used remote sensing to improve the planning, execution, and analysis of the National Forest Inventory in different periods and scales. This initiative uses the Collect Earth Online (CEO) platform, developed by FAO in collaboration with NASA and Google. CEO is a crowd-sourcing web technology designed specifically for earth science analysis, allowing users to collect baseline data using high-resolution satellite imagery and perform big data analysis through Google Earth Engine (SFB, 2024b).

The use of these technological innovations for monitoring the biome presents the advantages of ease of handling large volumes of data without the need to download files, processing using cloud computing capacity and developing algorithms using artificial intelligence based on machine learning techniques, such as support vector machine and deep learning (Kattenborn et al., 2021; Singh et al., 2022). These approaches, combined with traditional method, represent complementary monitoring tools that offer a holistic vision for the conservation and management of the Atlantic Forest, promoting the protection of natural resources and maintenance of biodiversity and ecosystems services.

5 Conclusions and perspectives

The past, present, and future of the Atlantic Forest illustrate a complex interplay between historical exploitation, ongoing conservation efforts, and future challenges under climate change scenarios. Specifically, the vegetation complex of the Atlantic Forest plays a multifaceted role in the world, serving as a biodiversity hotspot, carbon sink, and water and climate regulator, among other important ecosystem services for human well-being. The preservation of primary vegetation, maintenance of secondary vegetations, and promotion of restoration initiatives contribute to global climate change mitigation efforts. These actions require a multifaceted approach involving government policies, conservation organizations, local communities, and local stakeholders. Some highlighted actions involve:

- Establishing and effectively managing protected areas are crucial steps in conserving the biodiversity of highly fragmented ecosystems. This encompasses conservation units, legal reserves, and other forms of protected areas that offer habitat protection and facilitate wildlife corridors.
- Investing in ecosystem restoration projects is crucial for rehabilitating degraded areas and linking fragmented habitats. Given the complexity of environmental conditions and large-scale ecosystem degradation across the Atlantic Forest, both passive restoration through the growth of secondary forests after disturbances and active restoration involving management techniques can be effective and complementary methods to preserve the cobenefits of carbon and biodiversity, as well as multiple ecosystem services.
- Implementing land-use planning measures that prioritize conservation and sustainable development can help mitigate deforestation and habitat fragmentation. Zoning regulations, land-use planning, and environmental impact assessments are essential tools for conservation. Implementing monitoring systems, using remote sensing technologies, and engaging local communities in surveillance efforts can help detect and prevent illegal activities.

- It is also essential that ventures requiring suppression be conditioned to compensate for the same vegetation formation that will be suppressed. This obviousness tends not to occur when suppression occurs in regions of great economic importance, such as the *Quadrilátero Ferrífero,* or with intense real-estate speculation.
- Integrating conservation objectives into sustainable development initiatives, such as agroforestry, sustainable agriculture, and eco-friendly tourism, can help alleviate pressure on natural resources while supporting local economies and livelihoods. The agroforestry system model can promote the recovery of degraded permanent preservation areas through planting and agroforestry management techniques, as well as its potential for expansion to other rural properties in the micro basin. Thus, this sustainable management has positive impacts on two important environmental services, water and carbon stock.

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Chapter 32

Biodiversity, function, and change of the tropical rainforests of Borneo

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1 Introduction

Borneo is the world's third-largest island. Located in the humid regions of tropical Southeast Asia, the island is politically divided among three countries: the sultanate Brunei Darussalam, the Indonesian provinces of North, East, South, Central, and West Kalimantan, and the states of Malaysian Sabah and Sarawak. Over 23 million people currently live in Borneo (census data 2020: 23,720,000 people; Worldpop 2020 dataset: constrained: 26,613,810 people), the majority concentrated along the coast in urbanized areas (Fig. 32.1). These are interconnected by roads that stretch across >4000 km with a noticeable gap between Malinau in Indonesian North Kalimantan and Tawau in Malaysian Sabah. A large group of indigenous peoples has inhabited the island of Borneo over centuries, as Borneo is thought to have been continuously inhabited by people since the Paleolithic Age (Barker et al., 2017). Previous estimates describe approximately 450 ethnolinguistic Dayak groups (sensu heterogeneous interior peoples), that differ in resource use and culture, lifestyle, in group contacts and trade relationships with outsiders; many are believed to be farmers practicing shifting rice cultivation alongside forest resource use through hunting and gathering, but there are also nomadic and seminomadic peoples such as the Penan and Bukitans (Jessup and Vayda, 1988; Minority Rights Group, 2018).

Mega-diverse Southeast Asia has a complex tectonic and evolutionary history (Hall, 2009) and fragmented geography, comprising mainland and archipelagos, which is reflected in the distribution of extant biodiversity and high degree of endemism at local and regional scales (De Bruyn et al., 2014). Borneo, a word derived from a Sanskrit word for water, is located in the Sundaland biodiversity hotspot, one of 36 internationally recognized global biodiversity hotspots, 4 of which are located in Southeast Asia, which together cover 2.4% of the land surface area (Hrdina and Romportl, 2017) and encompass Earth's most biologically diverse and most threatened terrestrial areas. Borneo is estimated to harbor about 4% of the world's plant species, and 3.9%, 6.3%, 2.1%, and 2% of the world's known mammal, bird, amphibian, and reptile species (Das, 2006; Haas et al., 2024; Phillipps and Phillipps, 2018). The island features several distinct ecosystems that are described in more detail in Section 4 and include several forest types and the Kinabalu Montane Alpine Meadows (Fig. 32.1A). Underresearched habitat types in terms of biodiversity values include seminatural habitats such as urban and suburban vegetation (e.g., gardens, parks, and housing compounds), but also settlements and agricultural fields, and the vast expanses of oil palms.

Borneo is believed to be the most diverse part of the Sundaland biodiversity hotspot in terms of plant biodiversity (Raes et al., 2009). About one-third of Borneo's estimated 15,000 vascular plant species (of which 3000 are tree species) (MacKinnon, 1996) are endemic to the island. Neo et al. (2020) describe 65 genera as endemic to Borneo, from 25 families, comprising 164 known species. Botanical richness is predicted to be high in the north and northwestern parts of the island, including areas west of the northern Crocker Mountains range and the area between the Sembakung-Sesajap delta and lowland forests of northern East Kalimantan, with additional patches featuring high plant richness in the area near Kuching and the eastern mountain peaks of the central mountain chain located in central East Kalimantan (Raes et al., 2009). The lowland humid forests are particularly species-rich, with more than 10,000 species, which would represent the largest number of plant species for any ecoregion, globally (Kier et al., 2005). The lowland rainforests are usually dominated by species in the Dipterocarpaceae family (Whitmore, 1984), with trees that can reach more than 50 m in height (Ghazoul, 2016) and that dominate the international tropical timber market contributing significantly to the economy of many Southeast Asian countries (van Gardingen et al., 2003).

Terrestrial Biomes. https://doi.org/10.1016/B978-0-443-36569-0.00032-X
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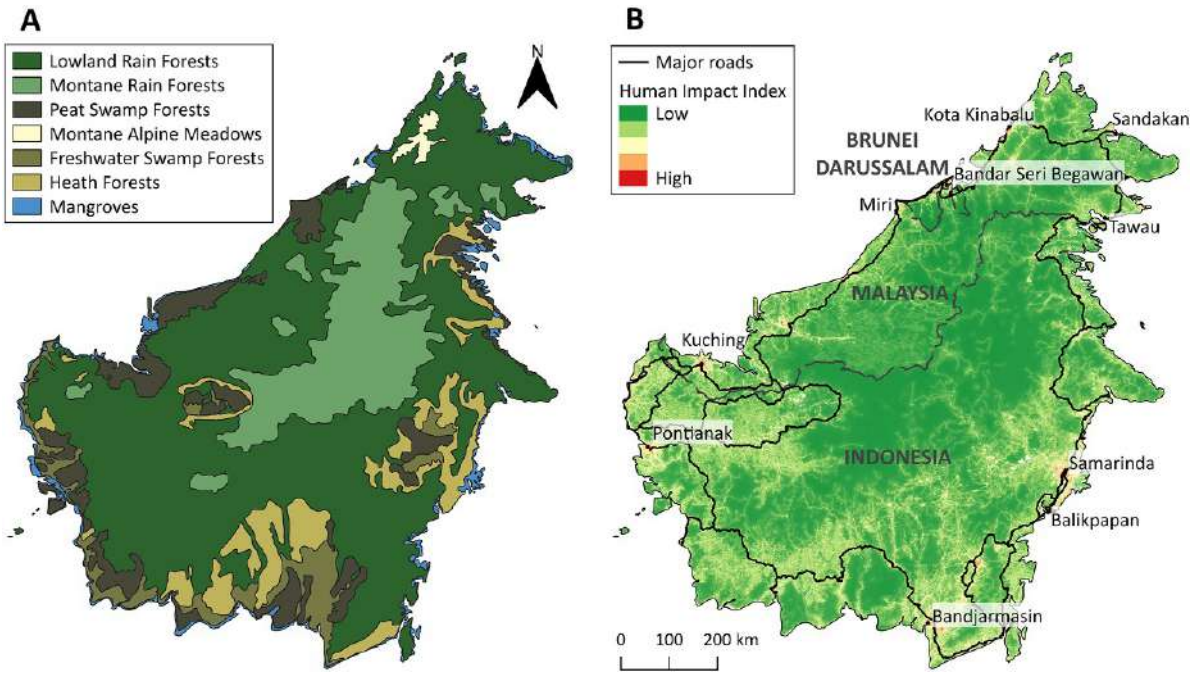

FIG. 32.1 Ecoregions and human footprints in Borneo. (A) The ecoregions found on Borneo. (B) Borneo's populated places, major road network, and the human impact index across the island. Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credits: (A) Data from https://www.oneearth.org/. (B) Data from Sanderson, E.W., Jaiteh, M., Levy, M.A., Redford, A.V.W., Woolmer, G., 2022. The Human Footprint and the Last of the Wild: the human footprint is a global map of human influence on the land surface, which suggests that human beings are stewards of nature, whether we like it or not. Bioscience 55, 891–904. https://doi.org/10.1641/0006-3568(2002)052[0891:THFATL]2.0.CO;2, https://wcshumanfootprint.org/data-access.)*

Birds of Borneo (Myers, 2016) described more than 650 species; of these species, 420 are listed as resident and 52 are endemic species. Among Borneo's bird fauna are the culturally symbolic and ecologically important hornbills (Bucerotidae), which feature as symbols and motifs among the many indigenous cultures of Borneo to establish status, symbolize bravery, and provide spiritual protection (King, 1985). Ecologically, hornbills are also important seed dispersers for fig (*Ficus*) species (Harrison, 2005). The Philipps Field Guide to the Mammals of Borneo recognizes 247 mammal species, of which 63 are listed as endemic to Borneo (Meijaard, 2016), highlighted in Section 5. World Wildlife Fund (WWF) lists more than 150 reptile and amphibian species for Borneo and emphasizes the concentration of amphibian species in Gunung Mulu National Park (Sarawak), harboring Wallace's flying frog (*Rhacophorus nigropalmatus*). Newly discovered species include the sticky frog (*Kalophrynus eok*, described in 2003) found in the Kelabit Highlands of Sarawak, and the Bornean flat-headed frog (*Barbourula kalimantanensis*) (first described in 1978), the only known frog without lungs, that has so far only been found in locations in fast-flowing streams in the Kapuas River basin (Bickford et al., 2008). Among reptile species that can be found on the island are 105 species of lizards inhabiting the rainforests, including the widespread Borneo bloodsucker (*Bronchocela cristatella*), a bright green arboreal species that is also found on neighboring islands, and the Five-lined Flying Dragon (*Drago quinquefasciatus*). Much less is known about the invertebrates of Borneo. A recently discovered snail species in the genus *Craspedotropis*, collected on a field course to Kuala Belalong Field Studies Centre in Brunei Darussalam, was named *Craspedotropis gretathunbergae* n. sp., in honor of the climate change activist Greta Thunberg (Schilthuizen et al., 2020). The Heart of Borneo, an area of 220,000 km^2, is particularly important for its biodiversity, protected by rainforests that the three countries consider irreplaceable (Suis et al., 2023). According to the WWF (n.d.), 29 new invertebrate species were discovered in this region between 2007 and 2009. Among these is Chan's megastick (*Phobaeticus chani*), first described in 2008 as the longest insect in the world (a record now broken by another species), with the body measuring 35.7 cm (Hennemann and Conle, 2008 in *Nature*, 2008).

As of 2009, 15 years ago, 70.6% of the Sundaland biodiversity hotspot was mapped as "a state that can no longer be considered natural environments" (Verma et al., 2020). Between 1993 and 2009, the human footprint (i.e., a map of cumulative human pressures, see Fig. 32.1B) increased both within and outside of protected areas across Borneo, a trend that has not improved substantially since. Key mechanisms underlying this trend are widespread and profound declines

in forest amount and quality and increases in cropland areas and human population densities in the distributional range of wildlife taxa. Between 1973 and 2015, large tracts of forests, in particular at elevations below 500 m above sea level, were exposed to selective timber cutting, illegal logging, drought, and fire as plantation industries expanded and millions of hectares of old-growth forests were converted to degraded scrubland (Gaveau et al., 2016). In that regard, Borneo is similar to other biodiversity hotspots in the world.

The loss in biodiversity and other functions associated with Borneo's forests has implications for ecosystem services, globally and locally. The conversion of forests to other land uses such as oil palm, has been shown to significantly decrease the amount of carbon stored in soil by up to 42% (Rahman et al., 2018). Forest degradation and conversion was estimated to lead to a decline in tree aboveground carbon stocks from around 175 t/ha in primary forests to around 61 t/ha in twice logged and 47 t/ha in salvage-logged forest stands, and around 24 t/ha for oil palm stands (Pfeifer et al., 2016). Increasingly, open forest canopies lead to increases in air and ground surface temperatures and declines in humidity (Hardwick et al., 2015) reducing the microclimatic buffering effect that forests can offer (Blonder et al., 2018) to understory vegetation and wildlife (Boyle et al., 2020). It also increases the susceptibility of forests to other drivers, especially droughts and fire (discussed in Section 7). Loss of forests and their resources impacts indigenous people, who have been using forests for centuries. Many species and habitats valued by local people have global significance (Sheil et al., 2006), with considerable variability in forest dependencies and values that communities associate with biodiversity. The latter depends on the geographic location of the community households, including whether they live upstream or downstream or close or far away from access to markets (Campbell et al., 2003). Local communities' use forest resources in various ways, from peatland forest timber trees (e.g., *Shorea* spp., *Gonystylus bancanus*) to trees used for rubber (*Dyera polyphylla*) and medicinal purposes (e.g., *Dracaena cantleyi* leaves used for the self-medication of joint pains) (Husson et al., 2018). Animal species have been hunted for food (Struebig et al., 2007) or harvested for ornamental purposes (e.g., *Argusianus argus*), and the pet trade (e.g., green leafbirds) (Husson et al., 2018).

Borneo has a rich and complex history, for example trading its natural resources like camphor, birds' nests, and rhinoceros horn with China and India since the 6th century AD (Broek, 1962). There are numerous publications on the subject of Borneo's plants, animals, and people during Borneo's colonization by the British and Dutch, which began in the 1800s (e.g., Büttikofer, 1897; Hose and McDougall, 1912; Harrisson, 1949). Perhaps the most notable is Alfred Russel Wallace's paper on "the Sarawak Law," which is an important document in the original development of evolutionary theory (Wallace, 1855, 1890; van Wyhe, 2016). Wallace lived in Sarawak between 1854 and 1856 at the invitation of Sir James Brooke ("the White Rajah of Sarawak") during his travels around the Malay Archipelago collecting scientific specimens of local species, with the assistance of as many as 1200 local people (Wallace, 1890; Beccaloni, 2018; van Wyhe, 2018). The legacy of colonial rule is reflected in the current boundaries of Bornean territories (the Malaysian and Indonesian states broadly corresponding to the previous British and Dutch colonies, respectively), and the names of places and species around Borneo. These often bear the names of colonial officials including Crocker Range, Rajah Brooke's birdwing (*Trogonoptera brookiana*, named by Wallace), Hose's civet (*Diplogale hosei*), and Raffles' pitcher plant (*Nepenthes rafflesiana*), named after Sir Thomas Stamford Bingley Raffles, a British colonial official who served as the governor of the East Indies. In more recent history, there continues to be heavy involvement of Western institutions and researchers in establishing and conducting research in Borneo (see Section 6 of this chapter), and the authors of most of the literature cited here are based at such institutions, with the authors of this chapter not being an exception. Increasing awareness for the need to found scientific collaborations on greater mutual respect and to integrate findings with equity and justice considerations for diverse knowledge generation is as much needed in Borneo as in many other parts of the developing tropics. As succinctly put by the Spiritual leader of Long Isun "*Dayaks can't be separated from the forest, our lives are spent in the forest. Without her we lose our identity*" (Iney Yeq, cited in MacInnes, 2020).

In this chapter on Borneo, we will focus on the better researched taxonomic group of mammals to highlight some of Borneo's unique biodiversity and their role in supporting ecosystem functioning and services, locally and globally. We will also discuss several key mechanisms underlying today's biodiversity distribution, including threats to species and habitats, ongoing research to map its biodiversity and associated challenges, and conservation strategies that need to be implemented to address key threats.

2 Geographic location

Borneo (755,000 km^2; Britannica.com) is one of the Greater Sunda Islands, located north of Java, west of Sulawesi, and southwest of the Philippines. From northeast to southwest, the island stretches to approximately 1336 km, and from east to west it stretches about 960 km. Key natural ecosystems featured on the island include lowland and mountain rainforest, heath forests, peat swamp forests, freshwater swamp forests, mangroves, and montane alpine meadows (Fig. 32.1A). They have been shaped by

biophysical constraints and anthropogenic disturbance that have included selective as well as large-scale logging, followed by subsequent forest recovery or land-use conversion. Human interactions with rainforests in Borneo have been dated back to 50,000 years ago (Barker et al., 2017). Oil palms now cover huge swaths of Malaysian and Indonesian Borneo.

The densely vegetated heath forests can be found on nutrient poor, acidic, well-drained sandy soils, and contain a thin layer of surface peat (Brunig, 1974; cited in Ikbal et al., 2023). Coastal heath forests stretch along the coastline on raised beaches; lowland heath forests (<1000 m asl) are found inland on sandstone plateaus and hillsides; and lower montane heath forests occur on sandstone ridges (from 800 to 1800 m asl) (Ikbal et al., 2023). *Kerangas* is a term used to refer to dryland heath forests growing in lowlands and is derived from the language of the local Iban translating as ("land where rice will not grow"). Waterlogged lowland heath forests are also referred to as *kerapah* forests (Whitmore, 1984) and are one of the rarest and most unique forest formations in Borneo, confined to peatlands in Sarawak and Brunei (Osaki and Tsuji, 2016; cited in Ikbal et al., 2023). Nafiah et al. (2022) describe heath forests as characterized by two main tropical conifer species, *Agathis borneensis* and *Gymnostoma nobile* and a dominance of species from the Myrtaceae family. The single layer canopy of heath forests can reach between 18 and 50 m high, the forest ground is well-illuminated and epiphytes are abundant, introducing structural complexity into the forest stands (MacKinnon, 1996). Heath forests can store large amounts of carbon below ground and often harbor a high number of endemic species.

Peat swamp forests are another distinctive lowland forest type in Borneo, supporting specialized flora and fauna and existing on peat deposits that, in places, can reach up to 10 m depth in a highly acidic waterlogged environment (Page et al., 2006). Undisturbed peatland forests contribute a significant carbon deposit in the region and play a key role in regulating water flows. Peatland forests have experienced extensive logging over the 1990s, leaving them susceptible to fire and catastrophic damage from burning during extreme weather such as the 1997–98 El Nino (Page et al., 2009). Peatland deforestation, drainage, and conversion to agriculture have lowered water tables and massively increased carbon emissions (Gaveau et al., 2014). In 2015, satellite-derived land cover maps estimated that pristine and degraded peatland forests covered ca. 27% of the peatlands in Sarawak, 25% in Sabah, 72% in Brunei, and 42% in Kalimantan (Miettinen et al., 2016). The freshwater swamp forests, which stretch along the coast of Indonesian Kalimantan (Fig. 32.1) along rivers that meander through floodplains, form a unique ecoregion. They are periodically flooded, less acidic, and more aerobic and feature denser, more diverse vegetation than neighboring peat forests. The trees are taller, exceeding 35 m in the canopy, and several tree and animal species are shared with surrounding lowland rainforests, including the Borneo orangutan (*Pongo pygmaeus*) and rare cats, including clouded leopard (*Neofelis diardi*) and flat-headed cat (*Felis planiceps*).

Like other lowland rainforests in the Sunda region, Borneo's lowland rainforests harbor the majority of tree species; they are usually dominated by species in the Dipterocarpaceae family (Whitmore, 1984), constituting more than 20% of communities across sites (Slik et al., 2003). They are tall, commercial timber trees that constitute the upper canopy and emergent tree species that can reach more than 70 m height, overtopping a dense vegetation understory of less than or equal to 30 m (Milodowski et al., 2021). Euphorbiaceae are the second-most abundant tree family, comprising understory and low to upper tree canopy species (Slik et al., 2003). Oak, beech, and chestnut tree species gradually replace dipterocarps with increasing elevation from approximately 1200 m (Kitayama, 1992). Borneo's central mountain range receives more than 3000 mm annual rainfall and features extensive mountain forests that also derive moisture from the clouds. Over 150 mammal species inhabit this mountain forest ecoregion and include flagship species such as the highly threatened Bornean orangutan and the Bornean subspecies of the Sumatran rhinoceros (*Dicerorhinus sumatrensis harrissoni*). Under growing threats from logging and fire, protected mountain forests can be found in Kalimantan, in the Kayan Mentarang National Park and Betung Kerihun National Park (One Earth and Wikramanayake, n.d.) and Kinabalu Park World Heritage site in Sabah. The forest belt reaches to approximately 1500 m above sea level before transitioning into Ericaceae-dominated forests.

Kinabalu Montane Alpine Meadows is a small ecoregion and highest and sole alpine ecoregion in Southeast Asia. The subalpine zone begins at around 3000–3400 m above sea level and is covered by low-height vegetation comprising shrubs and stunted trees. Patches of meadow can be found in waterlogged shallow depressions and feature herbs that include buttercups, eyebrights, and gentians. Mangrove forest patches, while less species rich in trees (with 12 true species found in forests sampled in northern Borneo), are scattered along the coastline providing a protective barrier for human infrastructure from waves and storms such as tsunamis (Wah et al., 2011).

3 Mammal biodiversity

Borneo is one of the most biodiverse regions on Earth when considering mammal species (Fig. 32.2 examples). Many of these species are endemic, meaning they are found exclusively on the island of Borneo. These endemics include iconic species such as the Bornean orangutan, the Proboscis

FIG. 32.2 Camera trap images derived from forests in Brunei and Sarawak. From top left to bottom right: Sun bear (*Helarctos malayanus*), southern red muntjac (*Muntiacus muntjak*), Sunda pangolin (*Manis javanica*), long-tailed macaque (*Macaca fascicularis*), leucistic lesser mousedeer (*Tragulus kanchil*), and Sunda clouded leopard (*Neofelis diardi*). These camera traps were deployed between February and July 2022 in a mixture of protected and unprotected areas across Temburong, Brunei Darussalam, and in the Belaga District of Sarawak, Murum area. *(Photo Credit: The authors.)*

monkey (*Nasalis larvatus*), and the Borneo (often referred to as "Pygmy") elephant (*Elephas maximus borneensis*). Borneo's high level of diversity and endemism are hypothesized to be caused by historical patterns of climate change. Changing climates would have raised and lowered global sea levels and thus created periods of connection and isolation for Borneo with mainland Asia by the Sunda Shelf (Earl of Cranbrook, 2010). Today, new species of mammals are rarely discovered unless hybridization occurs between two extant species. An unusual discovery was announced in 2005 as the Swiss World Wildlife Fund documented the detection of a "new and possibly carnivorous" mammal species in Kayan Mentarang, in central Borneo (WWF, 2005). However, this was later dismissed as a rarely seen species of flying squirrel (*Aeromys thomasi*) (Meijaard et al., 2006a). The discovery of this novel mammal species in recent history shows just how biodiverse Borneo is, continuing to boast a rich and novel biota despite extensive forest loss across the island (Quek, 2009).

Patterns of mammal diversity vary widely at the species level across the island of Borneo, although, broadly patterns of biodiversity are shaped by a combination of elevation, vegetation, climate, and people. Synthesis studies, such as Chiaverini et al. (2022), have demonstrated that elevation, tree cover density, slope, and lowland open areas are central drivers of mammal species detections by camera traps, particularly at medium-broad scales (2–10 km scales). At the individual species level, clouded leopards of Borneo have been shown to select higher elevation habitats, while jointly exhibiting negative correlation with highly disturbed and human-dominated habitats such as large-scale plantations or recently logged forest (Macdonald et al., 2018). Wildlife patterns are also influenced by periodic pulses in fruit and seed availability, known as supra-annual mast fruiting events, where trees synchronize flowering and fruiting across families followed by years of reproductive dormancy. These events, often led by the Dipterocarpaceae family, are believed to be influenced by the El Niño-Southern Oscillation (ENSO) cycle, resulting in altered climatic conditions that impact forest productivity (Janzen, 1974; Satake et al., 2019). As a result of these fruiting patterns, wildlife may need to extend their foraging range to find fruiting trees, seek alternative foods during intramasting periods, or concentrate foraging efforts on areas where fruit-producing trees exhibit more consistent patterns of productivity (Cosby et al., 2023).

More recently, mammal diversity has been altered by anthropogenic disturbances such as logging and hunting (Cheyne et al., 2016). Many of the endemic species of Borneo are faced with the risk of extinction. One of the most iconic is the orangutan, meaning "the human of the forest" in Malay language, and shares over 97% of our DNA. The large, charismatic primate species classified within the genus *Pongo*, which is the only genus of great apes native to Asia, is one of the most iconic of the mammal species that can be found in Borneo. Nesting in the canopies of trees, orangutans have become an international emblem for wildlife conservation. The Bornean orangutan, alongside its Sumatran counterparts, is classified as Critically Endangered by the IUCN. Its 2016 population estimate approximated 104,700 individuals in the wild. Projections of population declines based on proxies of habitat loss and hunting estimate a population decline of 86% between 1950 and 2025 (Ancrenaz et al., 2023). Main threats include habitat loss and habitat fragmentation due to agricultural expansion and infrastructure development, illegal wildlife

trade, hunting, and climate-induced habitat alteration (Chin, 2002; Goossens et al., 2005; Wich et al., 2008; Meijaard et al., 2011).

4 Conservation areas and their ecological indicators

Conservation prioritization relies on the use of agreed criteria and values (e.g., instrumental versus intrinsic values) that may differ between stakeholders and from space to space. As Wilson et al. (2011) outline, "*we need to set priorities for conservation because we cannot do everything, everywhere, at the same time*." There are many criteria that could be used to guide prioritization for conservation interventions and in practice, approaches aim to combine criteria to cover one or more of the political, social, ethical, and ecological dimensions of conservation (Miller et al., 2006). Commonly used ecological criteria may focus on a species estimated to be at high risk from extinction (e.g., IUCN threatened species), the number of endemic species (Myers et al., 2000), or species number itself (Fleishman et al., 2006). To this end, several studies have focused on using plants and mammals as indicator species for determining priority areas for conservation on Borneo (Drummond et al., 2010; Ng et al., 2021; among others).

Borneo's lowland rainforests harbor the largest number of plant species for any given ecoregion in the world (Kier et al., 2005). Recent analyses, using species distribution models combined with point occurrence data, identified lowland forests of southwestern and central Sarawak, adjacent parts of central Borneo around the Mueller Mountains, and other lowland areas in northeast Sabah and North Kalimantan as centers of plant genera endemism (Neo et al., 2021). Mammals like birds overall are well understood in terms of habitat use, geographic distribution, and extinction risks, with the latter depending to a considerable extent on species traits (Chichorro et al., 2019). Many mammal species, and in particular those already listed as IUCN threatened, are vulnerable to habitat loss and degradation and overharvesting (Schipper et al., 2008). In the tropics, terrestrial mammals are exposed to extensive deforestation and forest fragmentation, while medium- to large-sized mammals are disproportionately targeted for hunting for food, bushmeat trade, and medicinal as well as ornamental purposes (Tomiya, 2013; Bowyer et al., 2019). Terrestrial, large insectivorous squirrels seem very sensitive to logging, and the degree of frugivory in primate species negatively correlates with their ability to persist in logged forests.

Cardillo et al. (2006) predicted latent extinction risk to be particularly high for mammals in Borneo, identifying it as a region where current risk is low but future risk is high. This is presumably because of the many narrowly distributed endemisms on islands and the relatively high importance of small geographic range size as the predictor of extinction risk. The interaction between habitat degradation and hunting has also caused considerable defaunation of mammal species across Borneo (Harrison et al., 2016), creating the "empty forests syndrome" found across the wider tropics (Benítez-López et al., 2017, 2019). Mammals such as the bearded pig (*Sus barbatus*, Fig. 32.3) can be socio-ecological indicator species (Kurz et al., 2023). The species plays an important role in indigenous ceremonial practices and celebrations and its population size is sensitive to hunting practices (e.g., Kadazandusun-Muruts in Sabaha and Iban in Sarawak) but also forest degradation and loss as it requires forests for activities such as wallowing, nest building, and mu-scraping (Love et al., 2017).

FIG. 32.3 Camera trap image derived from forests in Sarawak showing a large male bearded pig (*Sus barbatus*). The camera trap was deployed between March and June 2022 in the Belaga District of Sarawak, Murum area. Bearded pigs can be socio-ecological indicator species and are the most popular hunting target for non-Muslim hunters across Borneo. *(Photo Credit: The authors.)*

Recent camera-trapping studies implemented in Sarawakian Borneo have shown that medium-to-large- bodied mammalian disperser species are more likely to use and be supported by forest than by oil palm plantation. However, when looking across all taxa detected by the camera traps in this study, the levels of species richness and abundance were consistent across the two land-use types. These findings suggest species-specific responses to land-use type, and demonstrate that certain generalist and predator species (e.g., Bornean leopard cat, *Prionailurus bengalensis*) are able to persist widely across oil palm landscapes (Puttick et al., not yet published).

4.1 Protected areas

Protected areas are a widely used in situ conservation tool for biodiversity conservation. They constitute a key component of global biodiversity targets, exemplified in Target 3 of the 2022 Kunming-Montreal Global Biodiversity Framework, adopted during the 15th meeting of the Conference of the Parties (COP15; CBD, 2022). As of 2024, Protected Areas cover $>100\,km^2$ ($<14\%$) of terrestrial

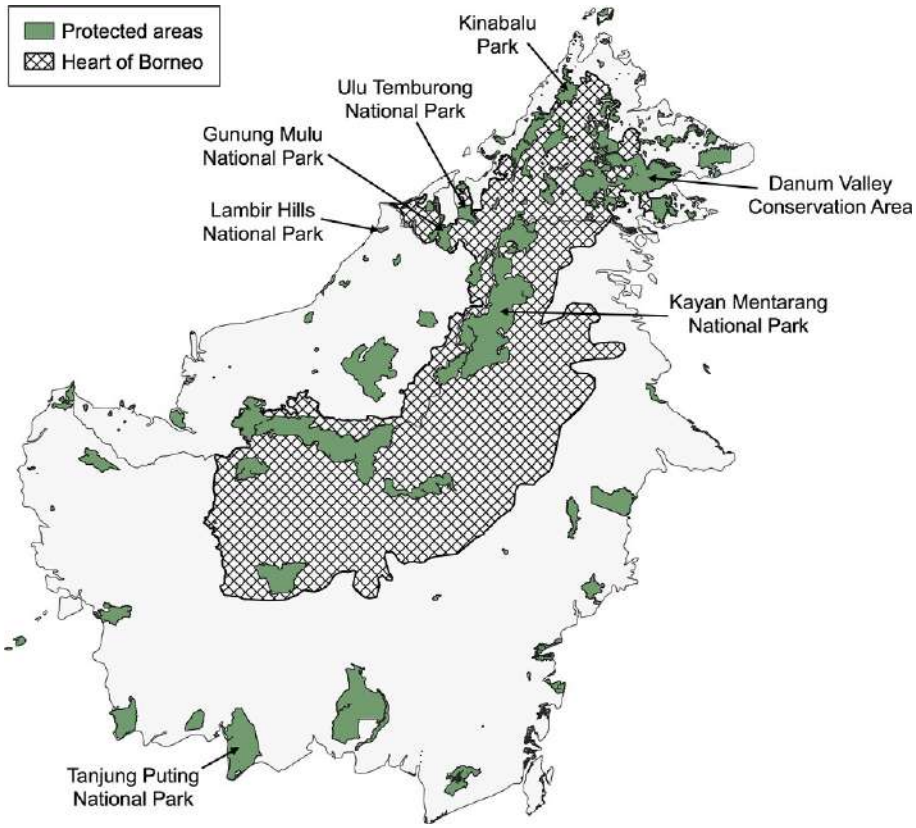

FIG. 32.4 The distribution of terrestrial protected areas across Borneo, and the extent of Heart of Borneo. Protected areas shown encompass different types including national parks, forest reserves, recreation parks, and state parks. Protected area data from UNEP-WCMC and IUCN (2024). Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: Heart of Borneo data from Sloan, S., Campbell M.J., Mohammed, A., Lechner, A.M., Engert, J., Laurance, W.F., 2019. Data From: Trans-National Conservation and Infrastructure Development in the Heart of Borneo [Dataset]. Dryad. doi:https://doi.org/10.5061/dryad.s4m5q53.)*

Borneo (Fig. 32.4). Using the latest downloadable version of Protected Areas for Brunei Darussalam, Malaysian Borneo, and Indonesian Borneo (status 01/03/2024) together with self-reported areas (in km^2) for those protected areas and combining it with area estimates for each of the three regions suggests that 1.536 km^2 (27%), 36.423 km^2 (19%), and 64.202 km^2 (12%) are protected (UNEP-WCMC and IUCN, 2024). Examples for better known Protected Areas (see also Figs. 32.4 and 32.5) include Kinabalu Park (842 km^2) and Danum Valley (439 km^2, IUCN 1a category) in Sabah, Gunung Mulu National Park (527 km^2) and Lambir Hills National Park (64.5 km^2, IUCN 2 category) in Sarawak, Tanjung Puting National Park (4144 km^2, IUCN 2 category and RAMSAR site) and Gunung Palung National Park (1.089 km^2, IUCN 2 category) in Kalimantan, and Ulu Temburong National Park (499 km^2, IUCN 2 category) in Brunei Darussalam (Phillipps and Phillipps, 2018).

Many of these protected areas encompass Borneo's remaining primary rainforest, while others have been previously logged but still represent high-value habitats for wildlife, e.g., Deramakot Forest Reserve, Tabin Wildlife Reserve, and the area surrounding Danum Valley's (Fig. 32.5) core area (Phillipps and Phillipps, 2018). Lambir Hills is a small national park representing an important remaining patch of pristine coastal forest in Sarawak, composed largely of mixed dipterocarp forest with small patches of *kerangas*. The Kabili-Sepilok Forest Reserve is an important location for wildlife and forest research in Sabah, featuring 6000 ha pristine forest, herbarium, and orangutan and sun bear rehabilitation centers. Tanjung Puting National Park is composed mainly of peat-swamp and *kerangas* forests, with large populations of mammals including 4000 wild orangutans. Gunung Palung National Park (Fig. 32.5) features an extinct volcano, contains seven forest types, and harbors a large population of orangutans and an orangutan conservation program, which has been running since 1992.

A 220,000-km^2 forested region is also protected under the Heart of Borneo conservation agreement, which encapsulates some of the protected areas described above. This voluntary transboundary agreement was made between the governments of Brunei Darussalam, Indonesia, and Malaysia in 2007 in order to effectively manage and

Danum Valley Conservation Area, Sabah

A 438-square-kilometer reserve of ancient rainforest, containing trees greater than 300 years old, and the tallest known tropical tree (Douglas 2022, Shenkin et al. 2019, Phillips and Phillips 2020). This is one of the least disturbed remaining tropical forests and is surrounded by a wider landscape of protected logged forest.

The Danum Valley Field Centre investigates differences between primary and logged forests (Douglas 2022). The centre joined the ForestGEO Global Earth Observatory network in 2009, providing a baseline for ongoing studies of forest regeneration, carbon dynamics and biodiversity (https://forestgeo.si.edu/sites/asia/danum-valley).

Gunung Palung National Park, West Kalimantan

Dubbed by some as 'Kalimantan's Eden', this park encompasses two mountains: Palung and Panti. Today it covers various distinct forest types, including peat swamp and freshwater swamp forests, heath forests and lowland and mountain rainforests. Home to gibbons, proboscis monkeys, flying squirrels and bushy – created hornbills, it is most famous for its high concentration of orangutans.

Research in the park centres on orangutans. The Mount Palung Orangutan Project was established in 1994 for classic population ecology studies as well as conservation. But the status of the research station with regards to access to facilities is unclear.

Ulu Temburong National Park, Brunei Darussalam

Known as the "Green Jewel of Brunei", this protected area encompasses primary lowland dipterocarp forest and is characterised by a rugged topography with steep slopes. It was previously a NP within the Batu Apoi Forest Reserve.

The Kuala Belalong Field Studies Centre was established in the early 1990s through a collaborative project between Universiti Brunei Darussalam and the Royal Geographic Society. The site's facilities (from accommodation to educational facilities) can only be accessed by longboats and joined the ForestGeo network in 2009 (https://www.forestgeo.si.edu/sites/asia/kuala-belalong).

FIG. 32.5 Three example field study centers of Borneo. Table 32.1 expands on these examples. *(Photo Credit: The authors.)*

conserve Borneo's forest resources and maintain its natural heritage, balancing conservation with sustainable economic productivity (Governments of Brunei Darussalam, Indonesia, and Malaysia, 2007; WWF Borneo Programme, 2018). The Heart of Borneo forests are important for ecosystem connectivity, preserving biodiversity, and providing ecosystem services such as acting as a water catchment and providing resilience to climate change (WWF Borneo Programme, 2018). Kayan Mentarang National Park (IUCN 2 category, 12,808 km^2) and its surrounding, in particular, are also central to the history and rights of the 27,000 indigenous Dayaks, who live inside or in close proximity to the park. This is officially recognized through the Alliance of the Indigenous People of Kayan Mentarang National Park (FoMMA, MacInnes, 2020).

4.2 Field studies centers and long-term studies

Field study centers play an important role in environmental education, awareness raising, and knowledge generation (Cooper, 1991). Tropical research centers and field stations have generated long-term data on rainforest ecology providing insights into patterns of biodiversity and underlying mechanisms, forest regeneration following disturbance, climate change responses of forest ecosystem processes, and potential for sustainable harvesting (Condit, 1995). In Borneo, at least 10 research and field centers (Table 32.1) have been established during the past decades to support long-term, field-based ecological research.

The Danum Valley Field Center (Table 32.1, Fig. 32.5) is one of the most prolific rainforest research centers in Southeast Asia (Phillipps and Phillipps, 2018). Long-term, plot-based fieldwork has been undertaken at Danum since 1985 to investigate tropical rainforest functioning and tree community dynamics (Douglas, 2022; Newbery et al., 1992, 1999; Newbery and Lingenfelder, 2017, 2020). In 2019, the Danum Valley featured in the *National Geographic* for providing a refuge to threatened yellow meranti trees (*Shorea faguetiana*), known in the world as the highest tropical trees. The tallest individual was climbed and measured as 100.8 m by Unding Jami, a rainforest research assistant (Shenkin et al., 2019). Also in Sabah, the Maliau Basin Studies Centre serves as a gateway to study 588 km^2 of pristine forest in the Maliau Basin Conservation Area, adjacent to Danum Valley. Several new species have been discovered in Maliau Basin over the years (Suetsugu et al., 2017, 2018; Tsukaya et al., 2017; Schilthuizen et al., 2017; Freitag et al., 2018; Johanson et al., 2022).

5 Human activities and their impacts

Like all tropical forested regions, Borneo is highly threatened by human activity and one of the most heavily and rapidly deforested regions in the world (Curran et al., 2004; Gaveau et al., 2016). Since the early 1970s, widespread deforestation, primarily driven by logging and

TABLE 32.1 Examples of better-known field study centers in Borneo.

Field center	Region	Description
Danum Valley Field Centre	Sabah	Established in Sabah in the mid-1980s and supported by a partnership between Yayasan Sabah, local universities, and the Royal Society (Phillipps and Phillipps, 2018). *Research*: optical rainforest functioning and tree community dynamics, rainforest recovery, biodiversity, carbon dynamics (Douglas, 2022).
Maliau Basin Studies Center	Sabah	The site was opened in the Maliau Basin Conservation Area in 2011. *Research*: pristine forest baseline data for comparison with logged and fragmented forest patches in the Stability of Altered Ecosystems (https://www.safeproject.net/, SAFE) Project, a collaboration among the Southeast Asia Rainforest Research Partnership (SEARRP), Imperial College London, and local partners. The large-scale experiment investigates ecological changes occurring due to landscape modification (logging, fragmentation, and conversion to oil palm).
Danau Girang Field Centre	Sabah	Biodiversity conservation field center located on the Kinabatangan river, Sabah, and comanaged by Sabah Wildlife Department and Cardiff University. Recent peer-reviewed publications include research on elephants, proboscis monkeys, and pangolins (Sitam et al., 2023; Davidar et al., 2023; Suzuki-Hashido et al., 2023). https://www.danaugirang.com.my/
Lambir Hills Forest Dynamics Plot	Sarawak	In all, 52 ha plot established within Lambir Hills NP in 1991 by the Sarawak Forest Department, Forest Global Earth Observatory, and Plant Ecology Laboratory of Osaka City University, Japan. *Research*: tropical forest ecology in global context; it has very high tree species diversity with 1200 species identified in the plot (Lee et al., 2002). https://sarawakforestry.com/lambir-hills-national-park/
Kuala Belalong Field Studies Centre	Brunei	Brunei's sole biodiversity focused research center. Its foundation in 1990 was marked by a 14-month-long expedition in 1991–92 by Universiti Brunei Darussalam and the Royal Geographic Society, to collect baseline information for multiple taxa in the surrounding area (Earl of Cranbrook, 1993). https://iber.ubd.edu.bn/kbfsc/
Cabang Panti, Gunung Palung	Kalimantan	Founded in 1983 by Dr. Leighton in collaboration with Dr. Daernedi of the Indonesian Institute of Sciences and rebuilt in 2020 (Rahman et al., 2021a,b). *Research*: orangutans, red langurs, and gibbons, dipterocarp masting, figs (Phillipps and Phillipps, 2018). Long-running projects: Gunung Palung Orangutan Conservation Programme and the One Forest Project. *Example publications*: Knott et al. (2021) and Marshall et al. (2014). https://savegporangutans.org/
Rekut Camp, Barito Ulu	Kalimantan	Established in 1989 as a collaboration between the University of Cambridge and the Indonesian Ministry of Forestry (Brearley, 2006; Hendriks, 2023). *Research*: local hybrid gibbon population (*Hylobates muelleri* × *agilis*), and a long-term phenological study of monthly observations (McConkey and Chivers, 2007; Brearley et al., 2007).
Bukit Soeharto	Kalimantan	Bukit Soeharto Research and Education Forest (BSREF) established by Mulawarman University 1988 (Forestry and Forest Products Research Institute, 2010). *Research:* Rehabilitation of burnt-over forest (Sutisna, 2001) and soil turnover and organic matter storage (Fujii et al., 2020).
Sebangau, Sebangau NP	Kalimantan	A 50 km^2 plot within the 6300-m^2 peat swamp forest is managed by the Centre for International Cooperation in Sustainable Management of Tropical Peatlands, based at the local University of Palangka Raya (Phillipps and Phillipps, 2018; Borneo Nature Foundation, 2021). *Research*: Orangutan Tropical Peatland Project (Ou Trop) at the Setia Alam Research Station (one of Borneo's most active field centers and natural laboratories). *Example publications*: Cattau et al. (2014), Cattau et al. (2016), and Ehlers Smith et al. (2013). https://www.tnsebangau.com/
Camp Leakey, Tanjung Puting	Kalimantan	Established in 1971 by Dr. Biruté Galdikas and Rod Brindamour to study Tanjung Puting's large population of orangutans (Orangutan Foundation International, n.d.). It is run by Orangutan Foundation International (OFI), which has conducted one of the longest continuous studies on any wild animal population, accumulating over 150,000 h of observations of wild orangutans here. Over the years the research camp has supported students and scientists from Indonesia (Universitas Nasional & Universitas Gajamada) and North America. OFI also runs the Pondok Ambung Research Station in the same national park. *Example publications*: Banes et al. (2016), Banes and Galdikas (2016), and Galdikas and Ashbury (2012). https://www.orangutan.org.uk/

clearance for conversion to oil palm plantations, has drastically reduced the island's primary forest cover, with forest area across Borneo declining by 30.2% between 1973 and 2010 (Gaveau et al., 2014). Logging activities lead to the creation of new roads, opening up forests to further exploitation in the form of hunting and other resource extraction. In the present day, larger scale infrastructure projects create further significant threats to biodiversity (Alamgir et al., 2019; Van de Vuurst and Escobar, 2020). Deforestation, coupled with climate change, also increases the frequency and severity of forest fires, particularly in Borneo's tropical peat forests (Posa et al., 2011). The interactions between these threats create serious negative impacts on Borneo's rich and unique biodiversity, which are discussed in more detail below.

5.1 Oil palms

As a monoculture, palm oil supports significantly lower species diversity and abundance than primary and secondary tropical forests, across taxa including arthropods, plants, bats, birds, amphibians, and mammals (Savilaakso et al., 2014; Meijaard et al., 2020; Yue et al., 2015). Species composition also differs dramatically between forests and oil palm, and invasive species risks pose an additional threat to biodiversity (Savilaakso et al., 2014; Yue et al., 2015) (Fig. 32.6).

A few mammal species, such as leopard cats, civets, and porcupines, and even large charismatic species like orangutans and elephants, have been recorded in palm oil plantations, particularly in heterogeneous landscapes with access to large, well-connected forest patches (Yue et al., 2015; Wearn et al., 2016; Ancrenaz et al., 2014; Meijaard et al., 2020). Deforestation for oil palm also reduces ecosystem functioning, e.g., disrupting the nitrogen cycle (Dislich et al., 2016; Hamilton et al., 2016; Rahman et al., 2021a,b).

Hectares of tropical forest are clear-felled for conversion to plantations of oil palm (*Elaeis guineensis*) to produce vegetable oil (Ananthalakshmi, 2019). Palm oil makes up 40% of oil traded globally (Murphy et al., 2021), and is widely used to manufacture biofuel, food, and beauty products. Despite its myriad negative environmental impacts, oil palm is a very high yielding and land-efficient crop. Indonesia and Malaysia are the top two exporters of oil palm worldwide (OEC, 2021), and 90% of global palm oil is produced in Borneo, Sumatra, and the Malay Peninsula (Meijaard et al., 2020). In Malaysian Borneo, in particular, new oil palm plantations are estimated to be responsible for 50% of the deforestation occurring between 1972 and 2015 (Meijaard et al., 2020). By comparison, direct conversion to oil palm has accounted for 11%–13% of deforestation in Kalimantan in the same period (Gaveau et al., 2016). Palm oil is not yet a significant threat in Brunei Darussalam, where its presence is negligible.

FIG. 32.6 Photograph showing a 4×4 vehicle within an oil palm plantation in Sarawak, Borneo. Palm fruit branches are pictured next to the vehicle on the small access road. *(Photo Credit: The authors.)*

5.2 Logging

An estimated minimum of 266,257 km^2 of forest cover was logged between 1973 and 2010 (Gaveau et al., 2014). The two main sources of timber from Borneo are natural timber concessions (Table 32.2) and industrial plantations. Timber is produced in monoculture plantations, predominantly of *Acacia* and *Eucalyptus*, which produce pulp for paper production (Gaveau et al., 2016). These plantations have rapidly expanded since the 1970s. Plantation area is likely to increase as Sabah, Sarawak, and Brunei all plan to expand their timber estate (Timber Trade Portal, 2024; Forestry Department Ministry of Primary Resources and Tourism Brunei Darussalam, 2024).

The creation of disturbed secondary forest through selective logging, wherein trees are selectively extracted for their resource value, negatively impacts biodiversity in a number of ways. Mammal species diversity and population density are often reduced, and other effects have been observed, e.g., changes in behavior and body size in birds and movement trajectories in mammals (Wells et al., 2006; Hearn et al., 2019; Mathai et al., 2017; Cheyne et al., 2016; Cosset et al., 2021). Responses to degradation

TABLE 32.2 Estimated area of natural timber concessions or forests allocated for timber supply, in different regions of Borneo.

Region	Area (ha) (year of estimate)	Percentage area of region (%)
Sabah	1,655,483 (2020)[a]	22.4
Sarawak	6,542,852 (2015)[b]	53
Kalimantan	10,594,500 (2013)[c]	14.3
Brunei	218,650 (2010)[d]	68

[a] *Sabah Forestry Department (2020).*
[b] *Global Forest Watch (2015).*
[c] *Gaveau et al. (2013).*
[d] *Forestry Department Food and Agriculture Organization of the United Nations (2010).*

are often species-specific, and habitat alterations can benefit some species while causing a detriment to others, creating changes in community composition (Costantini et al., 2016; Brodie et al., 2015; Mathai et al., 2017; Heon et al., 2019; Edwards et al., 2017). Long-term alterations to forest structure including changes in tree community composition and canopy degradation have also been documented (Milodowski et al., 2021; Hayward et al., 2021).

Compared to industrial plantations, natural timber concessions can hold significant biodiversity value and so do degraded secondary forests, which can support diverse mammal communities and populations of endangered species (Buřivalová et al., 2020; Gaveau et al., 2013; Brodie et al., 2015; Mathai et al., 2017). And, while biodiversity and functional diversity are lower than or comparable to primary forests, they are higher than in monoculture plantations (Yoh et al., 2023; Edwards et al., 2011, 2014; Kerfahi et al., 2014; Konopik et al., 2015; Lee-Cruz et al., 2013; Tripathi et al., 2016). Trends in logging appear to show expansion has slowed, as the amount of remaining accessible unlogged forests diminishes and logging operations must progress into Borneo's upland interior (Gaveau et al., 2014). Despite this, selective logging still threatens Borneo's forests, and logged forests are more susceptible to the other human impacts we outline here (Gaveau et al., 2013; Buřivalová et al., 2020; Bicknell et al., 2015).

5.3 Roads and infrastructure expansion

Between 1973 and 2010, 271,819 km of logging roads were constructed on Borneo (Gaveau et al., 2014). These logging roads have fragmented habitats and created edge effects increasing access for hunting and illegal logging (Clements et al., 2014). They have changed vegetation structure, altered species community composition, and facilitated the spread of invasive species (Edwards et al., 2017; Padmanaba and Sheil, 2014). Bornean logging roads are also linked to increased deforestation; in Kalimantan forest, loss was twice as high in areas where logging roads were present (Bicknell et al., 2015) (Fig. 32.7).

The Pan-Borneo Highway (also known as the Trans-Borneo Highway or Trans-Kalimantan Highway) is a road

FIG. 32.7 A former logging road, now used for access across the estates of an oil palm plantation in the Belaga district of Sarawak (left, taken 29/03/2022) and a view of the Sultan Haji Omar Ali Saifuddien (SOAS) Bridge in Brunei Darussalam (right, taken 06/06/2022). The trees on either side of the road pictured in the right-hand image are part of Labu Forest Reserve, which the SOAS bridge cuts through, and the hill visible on the horizon is Bukit Patoi, which exists in a protected area close to where the bridge terminates. *(Photo Credit: The authors.)*

network, which connects all four Bornean states (Fig. 32.1). Major projects like the Kalimantan Economic Corridor (KEC) and Sabah Development Corridor (SDC) propose upgrading or expanding this network to improve connectivity between urban and rural areas, and facilitate trade (Dezan Shira and Associates et al., 2011; Sabah Economic Development and Investment Authority, 2024). In Kalimantan alone, infrastructure projects reportedly plan to upgrade over 3300 km and construct over 1900 km of roads (Alamgir et al., 2019). Socioeconomic benefits to this infrastructure development could lead to significant economic development in these regions. More infrastructure projects are materializing across the island, including the completion of the Sultan Haji Omar Ali Saifuddien Bridge in Brunei in 2020, the longest bridge in Southeast Asia. Planned infrastructure development projects, such as those proposed as part of the KEC and SDC, have the potential to create major landscape fragmentation across Borneo, drastically reducing connectivity between protected areas, particularly in the Heart of Borneo region (Alamgir et al., 2019; Sloan et al., 2019), as well as directly impacting species movement, particularly on wide-ranging iconic species such as elephants and orangutans.

Sustainability often features in proposals for large-scale infrastructure projects in Borneo, but questions remain on their effectiveness for meeting green growth targets. One recent example is the construction of the new Indonesian capital Nusantara in East Kalimantan, the largest proposed infrastructure project on Borneo whose initial construction is expected to be finished in 2024, mostly on what is currently an expiring logging concession, with full completion projected for 2045 (Reuters, 2023; Gokkon, 2023a). The city is proposed to be "smart, green, and clean," designed to incorporate forests, minimize environmental impacts, and aim for net-zero carbon emissions (Adri, 2019; Nugraha, 2022). Yet, concerns are growing about the new city causing environmental degradation, threats to nearby ecosystems and endemic mammal species like the sun bear and proboscis monkey (Van de Vuurst and Escobar, 2020; Spencer et al., 2023; Atmoko et al., 2024) and negative impacts for indigenous communities (Teo et al., 2020; Gokkon, 2023b).

5.4 Hunting and wildlife trade

Humans have been hunting in Borneo's forests for 50,000 years (Barton et al., 2012; Barker et al., 2017; Bennett et al., 2000). Hunting is entwined with the cultures of many groups on Borneo (Harrison et al., 2016), and subsistence hunting is the most common form of hunting, as it is worldwide (Bennett et al., 2000; Harrison et al., 2016; Ripple et al., 2016). For example, in Sarawak, 23,500 tonnes of wild meat are eaten annually. Hunting is mainly opportunistic, with regularly hunted species including muntjac (*Atherodes* spp.), mousedeer (*Tragulus* spp.), sambar deer (*Rusa unicolor*), and primates (Cheyne et al., 2016; Yi and Mohd-Azlan, 2020; Bersacola et al., 2019). The most popular and culturally important game species among non-Muslim hunters is the bearded pig (Bennett and Robinson, 2000; Kurz et al., 2021; Cheyne et al., 2016; Barker et al., 2017). Other species hunted due to their importance in indigenous cultures include clouded leopards and hornbills, whose pelts and feathers are worn as part of traditional clothing (Bennett and Robinson, 2000; Harrison et al., 2016). Methods for hunting include the use of guns, spears, snares, and hunting with dogs (Bennett and Robinson, 2000; Cheyne et al., 2016; Harrison et al., 2016). Shotguns are the most popular method, except in Brunei Darussalam where shotgun possession is banned (Bennett and Robinson, 2000; Cheyne et al., 2016; Foreign, Commonwealth and Development Office, 2023). Hunting practices vary significantly between cultural groups, which have their own traditions and taboos, and about which there are many knowledge gaps (Bennett et al., 2000).

There is evidence that hunting drove the extinction of megafauna like the giant Asian pangolin (*Manis palaeojavanica*), and in the modern day it contributes to population declines in orangutans, banteng (*Bos javanicus*), and pygmy elephants (Earl of Cranbrook, 2010; Gardner et al., 2021). In Malaysian Borneo, sites where hunting was present were shown to have 31% lower mammal species richness (Brodie et al., 2015). Hunting also usually cooccurs with logging and has been described as the primary threat to wild species in logging concessions (Meijaard et al., 2006b; Brodie et al., 2015). Overall, hunting has become unsustainable across tropical Asia, with most species targeted for hunting suffering regional declines within the last 50 years, partly driven by increased hunting for markets (Ripple et al., 2016). However, commercial hunting is thought to be largely uncommon across Borneo (Harrison et al., 2016). Exceptions possibly include the Sunda pangolin and Malayan sun bear, both targeted due to use of their body parts in traditional Chinese medicine conferring on them high value (Harrison et al., 2016). Hose's langur (Fig. 32.8) and the helmeted hornbill (*Rhinoplax vigil*) have also declined due to hunting for the illegal wildlife trade (Nijman, 2005; BirdLife International, 2020; Harrison et al., 2016).

That said, there is some evidence of a decline in subsistence hunting in Borneo. Among the Iban in Temburong, Brunei Darussalam, hunters have become a minority and hunting is becoming less prevalent due to loss of traditional ecological knowledge (Franco and Minggu, 2019). Similarly, the Penan have largely given up traditional hunting practices after settling in Brunei and Malaysian Borneo (Brosius, 1991; Allred et al., 2022).

FIG. 32.8 A camera trap image of Hose's langur (*Presbytis hosei*) captured in Temburong, Brunei Darussalam on 26/03/2022. This species is classified as Vulnerable on the IUCN Red List due to hunting. *(Photo Credit: The authors.)*

5.5 Climate change and fires

The effects of climate change interact synergistically with the threats outlined above. Climate change interacting synergistically with land-use change may lead to habitat losses of over 30% for 11%–36% of Bornean mammal species, and up to 74% for the Bornean orangutan (Struebig et al., 2015). Some species may have to migrate further into Borneo's mountainous interior to escape the effects of warming, as suitable ecological conditions are predicted to move upslope (Struebig et al., 2015). El Niño events are also forecasted to become more frequent and severe, increasing the length and frequency of droughts, alongside hotter and drier conditions promoting an increase in fire risks across Borneo (Wooster et al., 2012). Although wildfires have occurred throughout Borneo's history, they do not appear to have significantly impacted the vegetation of its ecosystems prior to the colonial era (Cole et al., 2019). This has been changing, however. In addition to increasing tree mortality and reducing vegetation cover and species richness in forests, fires release massive quantities of CO_2 into the atmosphere (Yeager et al., 2003). Continuing deforestation could quadruple fire events on Borneo by 2050 (Trancoso et al., 2022).

Fires are highly correlated with land-cover changes and deforestation, which drive reductions in canopy closure, increases in local daily temperatures and temperature extremes, and declines in humidity (Langner et al., 2007; Siegert et al., 2001). Peat-swamp forests are more susceptible to fires than other forest types, exacerbated by draining and conversion to agriculture (Langner et al., 2007; Posa et al., 2011; Taufik et al., 2019). Previously burned or disturbed forests are more likely to burn again, and are more susceptible to further damage by fire (Siegert et al., 2001; Posa et al., 2011). In 1997, it was estimated that Indonesia's burning peat and vegetation released an amount of carbon equivalent to 13%–40% of the mean annual global carbon emissions from fossil fuels (Shiraishi et al., 2023; Page et al., 2002).

6 Operational and potential monitoring tools to conserve or to recover the biome

Remote sensing is a valuable tool that is increasingly used by scientists to understand biodiversity patterns in space and time and by practitioners to inform and adapt conservation planning and management. Sensors have been used on the ground, from the air, and from space, measuring the extent and structure of habitats and deriving biodiversity indicators (e.g., using camera traps and acoustic recorders) for use in conservation management and policy planning.

6.1 Measuring and mapping habitats and ecosystem structure

Structural complexity of vegetation stands is considered an essential biodiversity variable that can be measured using remote sensing platforms from the ground to satellites (Camarretta et al., 2020). Ground-based sensors used widely by forest practitioners include laser-based distance measurement tools, which can be used to more accurately measure tree heights. More recently, terrestrial laser scanners and thermal cameras have revolutionized scientific approaches to provide plot- and point-based measurements of vegetation density and habitat structure and interlinkages with microclimates (Eitel et al., 2016; Pfeifer et al., 2019). In Borneo, Pfeifer et al. (2016) used hemispherical canopy photography to show how canopy leaf area, which indicates ecosystem functions (Pfeifer et al., 2019) and availability of microclimate niches (Boyle et al., 2020; Hardwick et al., 2015), declines from undisturbed to heavily disturbed forest stands and oil palm plantations, and demonstrated the potential for satellite sensor data to map canopy closure across the wider landscape. Jucker et al. (2018) combined data from 173 field plots with estimates of canopy height and canopy cover derived from airborne laser scanning data to map aboveground carbon stocks across northern Borneo, important data for initiatives such as the Roundtable on Sustainable Palm Oil (Parish et al., 2021). Thermal sensors and cameras attached to drones have been used to identify and map orangutan nests (Milne et al., 2021; Burke et al., 2019), providing crucial insights into animal distributions and habitat needs. Additionally, Waite et al. (2022) used drone-based photography to quantify liana load of rainforest trees and underlying drivers, while Jucker et al. (2018) utilized high-resolution airborne laser scanning to generate high-resolution microclimate surfaces over a large landscape in Sabah. Other studies used satellite sensor data to map

large-scale land cover changes including deforestation (Fawzi et al., 2018) and conversion of land cover to large-scale industrial plantations (Gaveau et al., 2018).

6.2 Acoustic monitoring

Scientists have long used audio recorders to collect acoustic data in the field, and the development of autonomous recorders has expanded the possibilities for this kind of research (Sugai et al., 2018; Browning et al., 2017). Once deployed, these recorders independently record auditory data on a predetermined schedule, in a process called passive acoustic monitoring (Browning et al., 2017). These recorders are relatively cheap, easy to deploy, and enable researchers to efficiently collect large quantities of data across a wide range of taxa, using broad spatial and temporal sampling schemes (Sugai et al., 2018; Browning et al., 2017; Gibb et al., 2018). The use of passive acoustic monitoring is becoming increasingly popular worldwide, and it has great potential for use in Borneo (Sugai et al., 2018; Browning et al., 2017).

Examples of acoustic monitoring methods, which have already been applied across Borneo, include studies of Irrawaddy dolphin behavior (*Orcaella brevirostris*), orangutan communication, and gibbon density estimation (Sugimatsu et al., 2015a,b; Askew and Morrogh-Bernard, 2016; Erb et al., 2023; Gilhooly et al., 2015). Passive acoustic monitoring has also been used to investigate landscape-scale questions, such as the value of riparian reserves and impacts of logging on bat communities, and the impacts of land-use intensity on bird and insect communities (Mullin et al., 2020; Yoh et al., 2023; Buřivalová et al., 2022). Additionally, acoustic methods have been used to identify cryptic species, and therefore could aid us in understanding Borneo's previously undiscovered biodiversity (Matsui et al., 2014; Tan et al., 2019, 2020). Furthermore, soundscapes can be used as proxies for alpha and beta species diversity when comparing different sites (e.g., logged and unlogged forests), and could be applied to monitor landscape changes and assess habitat quality (Buřivalová et al., 2019; Monacchi and Farina, 2019).

The development of automated classification methods is also improving the ease of identifying recorded taxa and analyzing acoustic data (Sugai et al., 2018; Browning et al., 2017; Sugimatsu et al., 2015a,b; Buřivalová et al., 2022; Yoh et al., 2022). The SAFE Acoustics project, an autonomous acoustic monitoring network deployed in Sabah, which uploads audio in real-time from the field, and aims to use machine learning to fully automate biodiversity monitoring, perhaps best illustrates the exciting and ambitious possibilities for the future of ecology and conservation research in Borneo (Sethi et al., 2020).

6.3 Camera trapping

Camera trapping has become the method of choice for the detection and study of Borneo's rich mammal biodiversity. Commonly used for species inventories, particularly in protected areas, the use of camera traps as conservation and research tools has increased since its early uptake in the research community in the 1990s (Kucera and Barrett, 2011). In recent decades, the increasing accessibility and affordability of the equipment, as well as development of data analysis methods, has led to the popularity of camera traps as a means of monitoring many wildlife species and populations, and they are particularly well suited to the study of the cryptic tropical rainforest mammal species. Camera traps have, for example, produced interesting records of individuals with rare phenotypes, e.g., this record of a leucistic lesser mouse deer (*Tragulus kanchil*) recorded in Brunei Darussalam (Mannion et al., 2023; Fig. 32.2). The method lends itself well to creating species inventories or checklists for defined areas, for instance, protected areas or field study sites. Recently, camera traps were used to detect 43 mammal species in Sabah's forest reserves that fall within the Heart of Borneo (Suis et al., 2023). They were used to identify mammal responses to anthropogenic pressures comparing protected wilderness areas (including in Ulu Temburong) with disturbed forests in Brunei Darussalam (Braunholtz, 2022); Mannion and Khalid, not yet published) and to detect mammal use of forest-oil palm landscapes in Sarawak (Puttick, not yet published). Previous camera trap studies across Borneo's protected areas and forest reserves have documented the presence or absence of medium-large mammal species, giving greater insight into habitat use and species distributions. Wearn et al. (2019) successfully used camera traps in Sabah to show that while mammal abundance, corrected for detection probability, does not decline with forest degradation, it does with forest loss at the landscape scale. Nevertheless, the data also confirmed that land-use change has been altering the drivers of community assembly (Wearn et al., 2019).

Camera traps have been key to research on mammal species distributions and their habitat use across Borneo. Many species that were formerly poorly known or thought to be highly restricted in their distribution have been more widely detected as camera trap studies are undertaken. The endemic Hose's civet (*D. hosei*) is one such species, currently classed as vulnerable (Mathai et al., 2015). The majority (69%) of records of the species have come from camera trap studies published since 2011 (Mathai et al., 2019). The first record of the species in Kalimantan came from a camera trap study in 2012, at a site over 500 km from all previous records (Samejima and Semiadi, 2012). As our knowledge of species distributions and use of habitats

improves, so too does our ability to make predictions of their persistence based on future land use and climate change scenarios and plan for their conservation. Camera traps are also increasingly being deployed in the rainforest canopies including in Borneo providing more insights into how mammals use this elusive habitat (Haysom et al., 2021).

Tools to manage the massive amount of information captured by camera traps are advancing, particularly the use of machine learning to classify species identities (Tabak et al., 2019). Development of wireless camera trap models allows for images to be retrieved without physically visiting the cameras, which is particularly useful for research and monitoring in more remote locations. As of writing ZSL's Instant Detect system (https://instantdetect.co.uk/) is still in development, but this and other real-time remote monitoring systems could be key to future efforts to conserve or recover Borneo's habitats and wildlife.

7 Restoration

Restoration is promoted as a valuable operational tool to restore forest habitats throughout Borneo. Forest restoration includes a variety of activities with the overall aim of recovering ecosystems that have faced degradation (Lamb et al., 2005). Several international pledges, frameworks, methodologies, and policies exist to promote and carry out restoration. These include the United Nations Decade on Ecosystem Restoration (United Nations, 2020), The Bonn Challenge (2011), REDD+ (UNFCCC, 2024), Forest Landscape Restoration or FLR (IUCN, 2023), and ROAM (IUCN and WRI, 2014). Governments are key stakeholders driving restoration efforts, for example, the Sarawak Forestry Corporation (SFC) acts as the statutory body of the Sarawak Government to manage Sarawak's protected areas and carries out restoration of both marine and terrestrial environments (Gumal et al., 2022). SFC has so far carried out several restoration projects, which have gathered international attention, such as the Sarawak Reef Ball Project where artificial reef modules were deployed along the Sarawak coastline to create new, sustainable aquatic habitat for the restoration of coastal ecosystems (The Reef Ball Foundation, 2024; https://www.reefball.org/projects/malaysia/sarawak.htm). There are also many effective, community-led organizations carrying out restoration across Borneo. In West Kalimantan, Indonesian Borneo, groups such as Alam Sehat Lestari (ASRI) work alongside local communities of the Gunung Palung National Park to carry out forest restoration and reforestation. Their activities include deforestation monitoring to track environmental damage, planting over 220,000 trees to accelerate natural regeneration, and collaborating with community representatives to collect tree seeds for nurseries (ASRI, 2007).

Several challenges are faced by restoration efforts. Firstly, restoration interventions can be costly and resource intensive, and therefore funding restoration initiatives can be a significant barrier to implementation. Some researchers argue that approaches such as natural capital can be useful to drive investment into restoration (Farrell et al., 2022). However, there is as much debate on the moral positioning of valuing the restoration of ecosystems within an economic framework, as there is with negotiating the practicalities of valuing ecosystem goods and services (Holl and Howarth, 2000). Secondly, restoration targets and outcomes often vary between stakeholders and across spatial and temporal scales, and, therefore, the restoration process may necessitate complex trade-offs (Budiharta et al., 2018). This is particularly important for the socioeconomic facets of restoration, highlighted by Di Sacco et al. (2021) in their "10 golden rules for reforestation." In a recent collection of research papers on essential science advances needed for scaling effective restoration, Marshall et al. (2022) discuss challenges related to governance, value and supply chains for sustainable produce of forest restoration, identification of priority areas, and knowledge gaps on restoration success drivers. Recent reviews have highlighted that while a plethora of ecological indicators exist for monitoring ecosystems, only a small set of these indicators are used in practice; the most favored indicators include numbers of trees planted and area of land restored (Mansourian and Vallauri, 2022). Socioeconomic indicators of restoration outcomes are routinely excluded from monitoring programs (Mansourian and Stephenson, 2023). This can result in poor or inappropriate restoration outcomes for indigenous peoples and local communities.

Tools and methods on how to better incorporate social processes in restoration science and policy, which is critical to restoration equity and effectiveness, are becoming increasingly available (Löfqvist et al., 2023). Key actions relevant at landscape scales and discussed in Löfqvist et al. (2023) include: (1) placing equity at the center of restoration practice and working closely with local communities, (2) incorporating equity and justice concerns in the study, design, and implementation of restoration interventions, (3) elevating case study knowledge alongside global approaches in global agenda-setting dialogs, and (4) improving scientific communication around opportunities, constraints, barriers, and trade-offs. Monitoring to determine the success or failure of restoration interventions remains poorly fulfilled for many projects.

8 Conclusions and perspectives

Landscapes across Borneo are continuing to change due to past, present, and future human induced land-use change through expanding agricultural frontiers, resource extraction, and infrastructural development. The consequences of this change for the climate, biodiversity,

and society are multifaceted, and often drivers of land-use change and the impacts of land-use change operate synergistically to cause further degradation. Further research surrounding the impacts of human-induced changes in Borneo should be sensitive to the needs of local communities and stakeholders, particularly when considering the rich indigenous history that Borneo has and how the conservation and restoration of forest ecosystems, in particular, impacts indigenous groups. Key drivers of change must be considered in their local context, for example, managing hunting pressures may be the most suitable and locally appropriate method of conservation for certain landscapes, whereas providing support for species to adapt to climate change may be more suitable in others.

Additionally, when considering future anthropogenic pressures, decision-makers of Borneo must strike a balance between economic development that is sustainable and equitable for the members of their societies, while retaining space for Borneo's amazing biodiversity. This could include, but is not limited to, strategically planning infrastructural development to maximize connectivity between centers of economy but minimize cutting and fragmentation of old-growth forests. While minimizing fragmentation through the strategic planning of future developments is important, stakeholders must also focus on the restoration of degraded landscapes, and increasingly so former agricultural production areas, using the best available restoration science. Researchers can contribute to this by producing science that actively and intentionally fills knowledge gaps in conservation and restoration practice. This is particularly important given the recently outlined "30-by-30" targets of the Kunming Montreal Global Biodiversity Framework targets, in which 30% of land and sea must be protected for nature by 2030. While there are many alternative futures for Borneo (Runting et al., 2015), we hope for a future in which biodiversity conservation and socioeconomic outcomes can be jointly achieved.

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Chapter 33

Caatinga

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1 Introduction

The Caatinga, recognized as the largest nucleus of Seasonally Dry Tropical Forest (SDTF; sensu Pennington et al., 2009), is predominantly situated in the northeastern region of Brazil. Covering a vast expanse of 912,529 km^2, it constitutes approximately 10.1% of Brazil's land area (Silva et al., 2017), making it the largest contiguous block of dry forests globally (Miles et al., 2006). Its expansive dimensions exceed those of significant African countries like Namibia and European ones like France. The first indigenous inhabitants of the area named it from the Tupi language, meaning *ka'a* [forest] + *tinga* [white] = white forest, reflecting the captivating sight of a "white forest" created by defoliated trees and whitish trunks during the dry season (Prado, 2003; Navarro, 2013). This term captures the concept of a "white forest," reflecting the challenges of this tough season from human and ecological viewpoints. In classic Brazilian literature, Euclides da Cunha summed up this duality with the poignant phrase "*Barbaramente estéreis*; *maravilhosamente exuberantes*"—"Barbarically sterile; wonderfully exuberant" (Cunha, 1902).

The earliest hunter-gatherers may have arrived in the Caatinga between 25,000 and 30,000 years ago, marking the beginning of human influence there. Although the climate was more humid in the past (Angulo et al., 2006; Silva et al., 2023), which fostered denser vegetation, there must have been a limited supply of natural resources at the time (Heredia, 1994). This area was already the backdrop for historical Indigenous stories and customs, which taught people to live within the region's bounds. However, the 16th-century invasion by European explorers disrupted this interaction and resulted in significant changes throughout Brazil's colonization due to territorial expansion and exploration (Andrade, 1998; Kury, 2012). Since then, understanding ancestry has become essential to overcoming the difficult circumstances inherent in the Caatinga biome, such as its unique geological and climatic features. Through direct engagement with the natural environment, the residents have gained a thorough awareness of the potential applications of many plant and animal species as food and medicinal support. It has given them distinct ethnic traits that differentiate them from other indigenous communities in Brazil (Albuquerque et al., 2007, 2017). Despite these obstacles, the Caatinga maintains its diversity in geography and culture, with over 60 indigenous groups contributing significantly to its conservation efforts—most of which live in the São Francisco River basin (Dawson et al., 2021).

It is essential to understand that the meaning and definition of the term "Caatinga" might differ greatly depending on the source before we continue our investigation of this topic. Northeastern Brazil's seasonal dry tropical forest can be described as:

- **(i)** a biogeographic or phytogeographic domain (e.g., Andrade-Lima, 1981), it could be associated with the Hamadryades province of Martius (1858) or the Nordestina subprovince of Rizzini (1963);
- **(ii)** a biogeographical province (e.g., Cabrera and Willink, 1973; Prado, 2003; Morrone, 2006, 2014);
- **(iii)** a vegetation formation (e.g., Veloso, 1964), it could be a synonym for thorny forest (e.g., Beard 1944, 1955), steppe savanna (e.g., IBGE, 2012), or deciduous thorn woodland (e.g., IBGE, 2012);
- **(iv)** a complex of formations comprising various types of vegetation (e.g., Rizzini, 1963, 1997);
- **(v)** a morphoclimatic and phytogeographic domain (e.g., Ab'Sáber, 1974, 2003; Andrade-Lima, 1981; Neves et al., 2015; Moro et al., 2016, 2024);
- **(vi)** a biome, characterized on a global scale (e.g., IBGE, 2004; Silva et al., 2004); and
- **(vii)** an ecoregion (e.g., Olson et al., 2001; Velloso et al., 2002).

It is also crucial to understand that Caatinga is a diverse environment whose composition is not homogeneous (Fig. 33.1) (Araújo et al., 2022). Also, this diversity extends across and beyond its geographical boundaries. For example, enclaves of the Caatinga biome can be found

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outside this region, such as in the dry valley of the Jequitinhonha River in Minas Gerais State (Prado, 1992) and in Cabo Frio, Rio de Janeiro State (Coe and Carvalho, 2013). Conversely, enclaves of other phytogeographic areas can be found within the Caatinga domain, like the Cerrado in the Araripe plateau, the transition zone in Piauí known as "*Campo Maior*," and the Atlantic Forest in the "*brejos*" of Pernambuco, Paraíba, and Ceará (Tabarelli and Santos, 2004). The Caatinga domain has unique vegetational features, such as riparian forests, deciduous forests, and xeric shrublands. It also has many different ecosystems, biogeographic and geological features, making the area complex and varied.

When studying the Caatinga biome, it is critical to consider worldwide biome protection, its conservation status, and the consequences of climate change on its biodiversity and functioning. In this chapter, we examined recent and historical research to help us comprehend the problems, threats, future changes, and challenges of preserving this unique environment.

2 Physical and biogeographical context

The Caatinga biome mainly covers the northeastern portion of South America and occurs only in Brazil, from roughly 3° to 17° South latitude. Among the previously indicated classifications, the primary ones utilized are as a biome within the Brazilian biodiversity system (IBGE, 2004) and as a component of the global biome Seasonally Dry Tropical Forests (SDTF) (Pennington et al., 2009; Silva et al., 2017). Similar to Ab'Saber's (1974) proposal, Queiroz et al. (2017) and Moro et al. (2024) adopted the term "Caatinga Domain" to describe the main region of the Caatinga. They suggest that in addition to the Seasonally Dry Tropical Forest and Woodlands (SDTFWs), which make up the majority of the area, there are also scattered patches of other biomes like Cerrado (Savanna), Tropical Rainforest, and Rupestrian Grasslands. Notably, including woods in the SDTF emphasizes the importance of nonforested landscapes with an open canopy, expanding upon the idea presented by Pennington et al. (2009). The inclusion or removal of evergreen semideciduous woods is another significant difference between these conceptions of Caatinga as a biome or the center of SDTFW (Zanella, 2011).

The question of whether different biomes in the Caatinga region belong to the same unit has been up for debate continuously. In particular, there is debate over whether the Cerrado and Atlantic Forest enclaves belong in the Caatinga region. In the course of this discussion, specifically to humid enclaves, Lima (2021) emphasized the special qualities of these Atlantic Forest enclaves and their relation to the surrounding ecosystem. Since these enclaves are primarily located at higher elevations and enjoy milder climates due to orographic rainfall, supporting a unique biota that is notably distinct from the Caatinga's typical flora and fauna. In opposition, Araújo et al. (2022) argue that the Caatinga region is a dynamic and heterogeneous system shaped by various ecological and evolutionary processes across various spatial and temporal scales. They support the idea that the region is a mosaic of diverse and interconnected biotas despite different landscapes, including rocky fields, savannas, tropical humid forests, and SDTFWs. However, Lima et al. (2023) contend that including the biome enclaves in the same unit as the Caatinga is inappropriate because they are highly fragmented, typically limited to mountain peaks, and do not cover large areas. Similarly, Moro et al. (2024) propose classifying the humid forest enclaves as a distinct biogeographical district. They emphasize that these areas can be considered within the context of either the Atlantic Forest or the Caatinga Domain, depending on the study's objectives. This flexible classification allows for their inclusion in biogeographical and conservation analyses of either biome, avoiding a rigid binary approach. Considering these opinions, it is clear that additional research and viewpoints are required to develop a thorough knowledge and possibly come to a consensus regarding the categorization and components of Caatinga.

In most of its territory, the Caatinga has a tropical semiarid climate with high mean annual temperatures (between 23°C and 27°C) that lead to high potential evapotranspiration rates (usually above 1500 mm/year; Menezes et al., 2012). There are also long periods—at least 5–6 months—during which there is little or no precipitation (Ab'Saber, 1999). Furthermore, lately, reports have indicated that arid regions are beginning to appear in the Caatinga biome (Tomasella et al., 2023). Most of the region has shallow, rocky soils that are not very good at retaining water, making it more susceptible to water stress. Additionally, because the region's annual mean rainfall is almost entirely below 1000 mm, there is a negative water balance, which gives the Caatinga vegetation its xerophilous deciduous character (Menezes et al., 2012). It is significant to remember that the SDTFW concept incorporates semideciduous forest habitats, which loosen the requirements for deciduousness and water stress (Pennington et al., 2006; de Queiroz et al., 2017).

The Caatinga region also experiences significant annual variations in precipitation (Marengo et al., 2017). For instance, in the same location, years with high rainfall (>1500 mm), typical of humid forest conditions, frequently follow years with little rainfall above 400 mm, typical of desert climates. This variance significantly impacts the local fauna, including insect populations (e.g., Vasconcellos et al., 2010; Brito and Zanella, 2022). According to Brito and Zanella (2022), rainy seasons in the xerophilous Caatinga make it possible to maintain social wasp populations in even greater numbers than in the riparian forest zones of the area. According to these findings, abundant resources during rainy seasons promote mating behaviors, nest-building, and

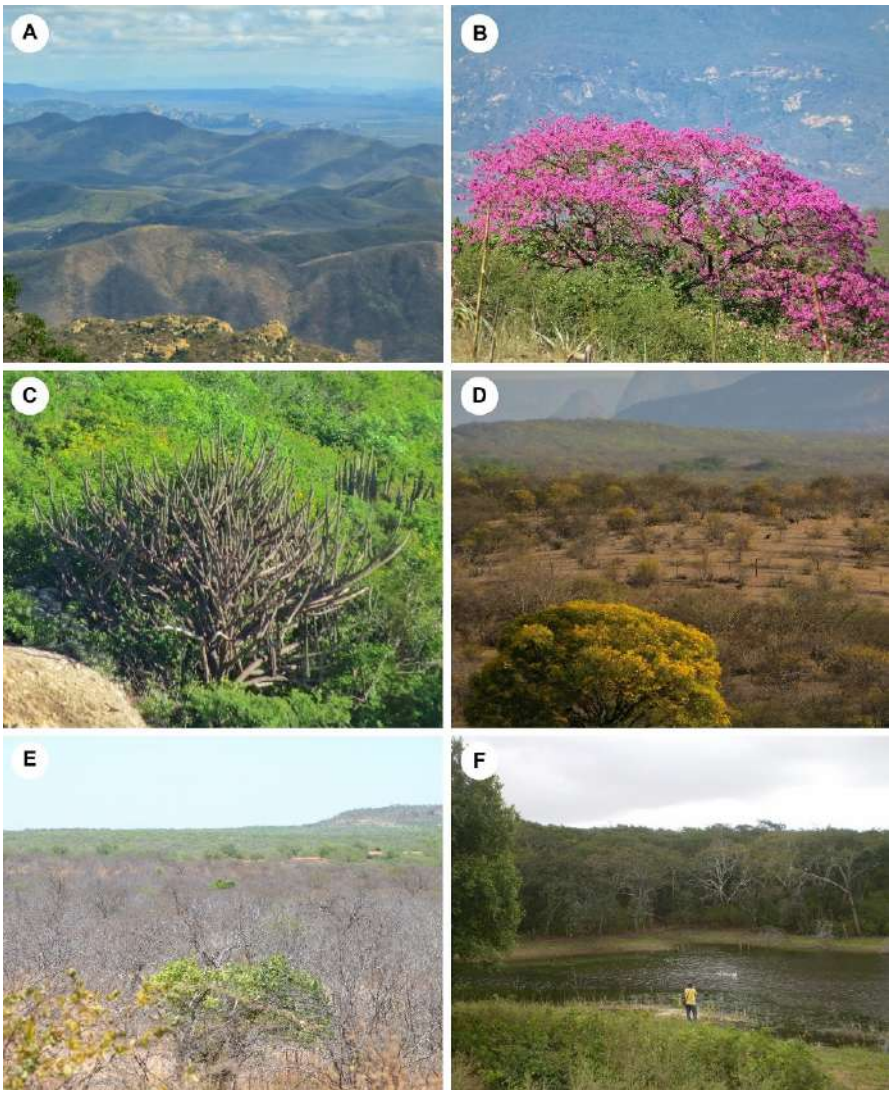

FIG. 33.1 Caatinga biome landscapes. (A) Maturéia, Paraíba, Brazil; (B) Patos, Paraíba, Brazil; (C) Xique-Xique cactus (*Xiquexique gounellei*, Cactaceae), in Carnaúba dos Dantas, Rio Grande do Norte, Brazil; (D) Craibeira tree with *yellow* flowers (*Tabebuia aurea*, Bignoniaceae), in Ouro Branco, Rio Grande do Norte, Brazil; (E) Dry and *green* Caatinga vegetation due to localized rains in Serra Negra do Norte, Rio Grande do Norte, Brazil; and (F) Caatinga at 550 m above the sea level, Desterro, Paraíba, Brazil. *(Photo credits: José Vinícius Alexandre de Medeiros (A–D) and Fernando César Vieira Zanella (E, F).)*

successful reproduction due to the growth of green vegetation during the rainy season, giving species plenty of food and shelter, which helps to ensure their overall survival and local population increase. Furthermore, the diverse soil types in Caatinga play a crucial role in shaping the region's ecosystems; the significance of soil, its physical and chemical properties, and particularly its water retention capacity, sometimes should be regarded as equally or even more crucial than the rainfall patterns in Caatinga (Moro et al., 2016; de Castro Oliveira et al., 2019).

The Caatinga biome has two primary geological formations: the Proterozoic crystalline basement and the Paleozoic and Mesozoic sedimentary basins (Sampaio, 1995). These formations give rise to two distinct habitats within the Caatinga: the Crystalline Caatinga and the Sedimentary Caatinga. The Crystalline Caatinga, covering approximately 75% of the biome, emerged gradually through uplift and erosion until the Tertiary period, resulting in a vast flattened surface with elevations usually ranging from 300 to 600 m (Queiroz, 2006; Queiroz et al., 2017). It covers most of the land in the "*Depressão Sertaneja*" lowlands and comprises old crystalline rocks like granite, gneiss, and schist. These rocks create crystalline terrains, which are stony and shallow but high in nutrients (Sampaio, 1995; Velloso et al., 2002).

In contrast, the Sedimentary Caatinga predominates in the sedimentary basins, featuring deeper soils with optimized water retention but poorer in nutrients than the

crystalline terrains (Velloso et al., 2002; Sampaio, 2010). This composition is prevalent in areas influenced by rivers, streams, and coastal plains, where sediments such as clay, silt, and sand have accumulated over geological time. In addition, there is evidence that the Caatinga has two separate floristic nuclei, one in crystalline areas and another in sedimentary sites. These two areas have very different flower compositions and plant structures, showing different life-form spectra in these environments (Queiroz, 2006; Costa et al., 2015; Moro et al., 2016).

The Caatinga is recognized for its varied landscape, supporting a range of distinct habitats (Fig. 33.2). However, specific environments, such as wet forests, inselbergs, and ecotones, received less attention in research investigations. Wet forests can be found along rivers that intersect the Caatinga biome, atop mountains or slopes, and in riparian and enclave woods. These woods are found on the slopes of plateaus and mountains, primarily in areas of characteristic crystalline Caatinga, where the soils are deeper and hold higher levels of edaphic moisture (Moro et al., 2014). These forests serve as crucial wildlife corridors and significant reservoirs of biodiversity, and they also provide habitat for many endemic species thanks to their rich vegetation and variety of plant species that have adapted to the humid microclimates produced by their proximity to water sources (Lima, 2021; Araújo et al., 2022). The inselbergs are single or clustered rock outcrops scattered across the flat surrounding landscape (Porembski

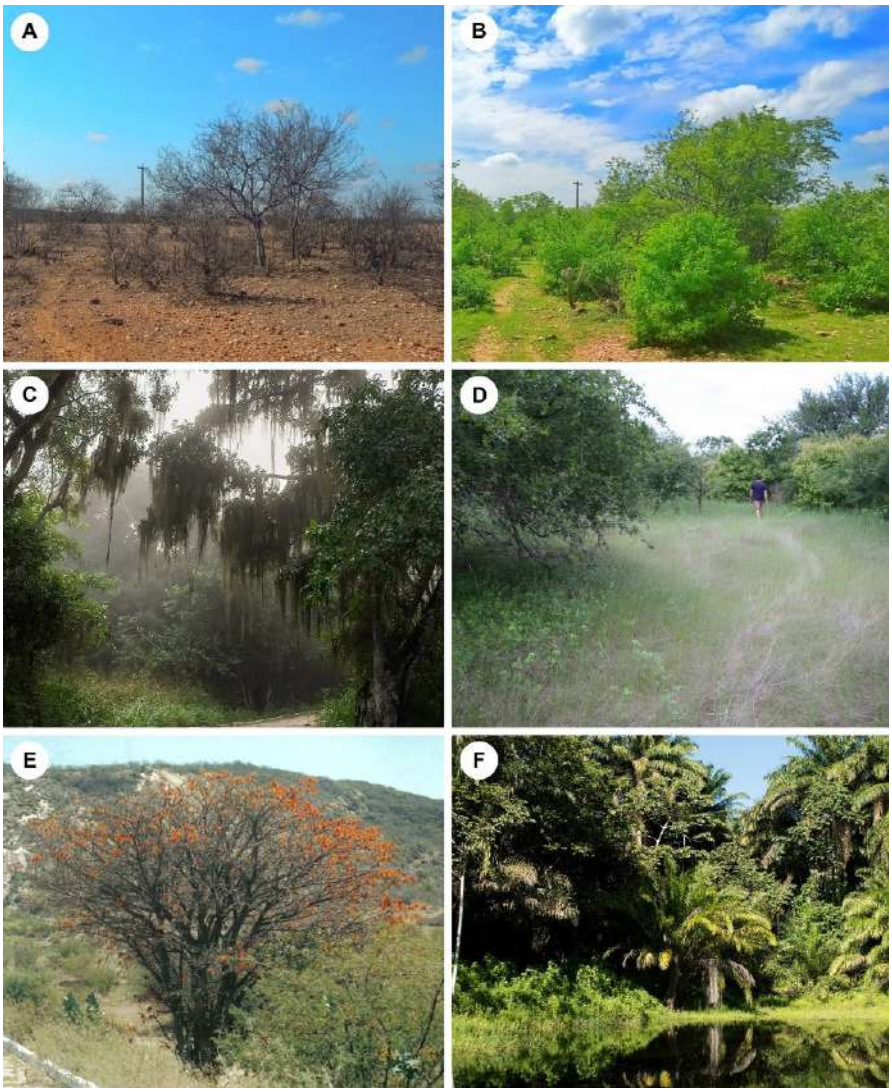

FIG. 33.2 Caatinga physiognomies. Same Caatinga landscape during the (A) dry season and (B) rainy season in Ouro Branco, Rio Grande do Norte, Brazil; (C) a high altitude enclave of semideciduous forest (approximately 1.200 m) in the State Park Pico do Jabre in Matureia, Paraíba, Brazil; (D) a typical Seridó landscape in Jardim do Seridó, Rio Grande do Norte, Brazil; (E) *Erithryna* sp. and the rocky mountain range in Santa Luzia, Paraíba, Brazil; and (F) a ecotonal zone transition between Caatinga and Atlantic Forest in Feira de Santana, Bahia, Brazil. *(Photo credits: José Vinícius Alexandre de Medeiros (A–C), Fernando César Vieira Zanella (D, E), and Emanuelle Luiz da Silva Brito (F).)*

and Barthlott, 2000). These formations can be situated atop mountains or manifest as large exposed rocks that abruptly rise amidst the terrain and exhibit a pattern of alternating shallow soils (Moro et al., 2014, 2016). In certain areas, sediment accumulation and sufficient water availability foster the growth of brushy-arboreal vegetation (Oliveira and Godoy, 2007; Porembski, 2007).

Additionally, the Caatinga has two very large ecotones with the surrounding phytogeographical domains: the "*Agreste*," where the Caatinga has a transition to the Atlantic Forest in the east, and the ecotone to the Cerrado in the west (Queiroz et al., 2017). Caatinga, Cerrado, and Chaco—these three seasonally dry biomes constitute the Diagonal of Open Dry Formations (Werneck, 2011; Zanella, 2011; Luebert, 2021). The STDFW (sensu Prado, 2000; Pennington et al., 2006; Queiroz et al., 2017) idea was based on studying woody plants, so it emphasized how different the biotic composition, biogeographical histories, and ecological factors are in each of these three biomes. However, the use of the term "Diagonal of Open Dry Formations" is still used in biogeographical studies based on open vegetation animals such as vertebrates (e.g., Werneck et al., 2012; Recoder et al., 2014; Lanna et al., 2018; Carmignotto and Astúa, 2022; Oliveira et al., 2024), arthropods (e.g., Zanella, 2011; Kohari et al., 2022) and other taxa (Bonatelli et al., 2022). These different interpretations may be due to the biotic communities in each biome are made up of taxa with different biogeographical histories. Studying groups with contrasting distribution patterns will provide a deeper understanding of the complex history of each biome (Zanella, 2011).

The Caatinga Domain is a very biodiverse area, even when enclaves of other biomes like Cerrado and wet woods are excluded. Fernandes et al. (2020) found that the floristic diversity per area unit was exceptionally high (4.0 species/1000 km^2), nearly two times higher than the Amazon Forest. It is important to emphasize that semideciduous woods in the Caatinga *lato* sensu were considered in this investigation. Also, riparian forests and patches of more humid forests help protect some of the Caatinga region's wildlife (Lima, 2021; Araújo et al., 2022; Lima et al., 2023). Overall, the Caatinga is an important ecosystem, habitat, and geological structure that forms part of a dynamic and intricate system formed by various ecological and evolutionary processes acting across different spatial and temporal scales.

3 Human impacts and threats to biodiversity

The primary threat to the Caatinga ecosystem is the widespread conversion of natural areas into human-modified landscapes (Fig. 33.3) (Quesada et al., 2009; IBGE, 2020; Barros et al., 2021). Human activities have influenced the Caatinga for thousands of years, resulting in profound transformations with important implications for biodiversity conservation and socioecological sustainability (Bueno and Dias, 2015; da Silva et al., 2017). While human occupation of the Caatinga dates back to the Late Pleistocene and Early Holocene, the most notable alterations began with the invasion of European explorers in the 16th century, marking the transition from a hunter-gatherer culture to one based on subsistence and extractivism (Hemming, 1978; Coimbra-Filho and Câmara, 1996; Andrade, 1998; Puntoni, 2002; Tabarelli et al., 2018). Currently, as a response to frequent droughts, low human development indices (HDI), coronelism,[a] and little government assistance, a scenario of extreme poverty has emerged, leading semiarid populations to increasingly depend on slash-and-burn agriculture and livestock (Souza et al., 2019; Dawson et al., 2021).

The Caatinga has been changing and getting worse over time. Old-growth forests have been turned into slash-and-burn farms, and resources from secondary forests that are still there are being taken in large amounts to meet the needs of people who live in densely populated areas (Souza et al., 2019; Barros et al., 2021; Bezerra et al., 2022, 2023). We are referring to tropical landscapes that have been changed by humans through unsustainable actions, like acute and chronic anthropogenic disturbances, which have made the current state of environmental degradation in the Caatinga biome worse (Leal et al., 2005; Rito et al., 2017a, b; Silva et al., 2017; Silva and Barbosa, 2017).

One alarming trend that is primarily due to acute anthropogenic disturbances like slash-and-burn agriculture, which accounts for about 75% of the biome's deforested areas, is the Caatinga biome, where pastures and agricultural practices have replaced nearly half of the area's original vegetation (MapBiomas, 2023). This traditional farming approach used in tropical dry forests has several detrimental effects on biological diversity at different levels of organization in the Caatinga, such as **(1)** shifting the composition and structure of biotic assemblages (e.g., Barros et al., 2021; Correia et al., 2020; Filgueiras et al., 2021; Bezerra et al., 2023, 2022); **(2)** reducing biotic interactions at the local level (Knoechelmann et al., 2020); **(3)** endangering native species and increasing exotic species that alter animal assemblages at the landscape-scale (Alves et al., 2023); and **(4)** impoverishing soil and depleting forest regeneration, leading to the collapse of the nutrient cycle (Moura et al., 2016).

The Caatinga forest, now immersed in anthropogenic landscapes, is also exposed to chronic anthropogenic

a. The hinterland population of Northeast Brazil during the Old Republic (1889–1930) commonly referred to any political leaders as "*coronel*." These leaders held significant power and centralized political authority in their respective regions, who would give favors in exchange for loyalty and vote for them (Carone, 1971).

FIG. 33.3 Human impact in Caatinga. (A) Cattle breeding in Nupeárido Farm, Patos, Paraíba, Brazil; (B) secondary shrubby vegetation, Teixeira, Paraíba, Brazil; (C) Ouro Branco, Rio Grande do Norte, Brazil; and (D) Ouro Branco, Rio Grande do Norte, Brazil: Planalto da Borborema in the background with wind turbines. *(Photo credits: José Vinícius Alexandre de Medeiros.)*

disturbances (sensu Singh, 1998). Fire, gathering firewood, livestock browsing, selective logging, hunting, introducing exotic species, and the regular cycles of cutting down and burning native plants are some of the chronic disturbances that happen on a landscape (Specht et al., 2019; Chaves et al., 2020; Gonçalves et al., 2020, 2021, 2022; Lins et al., 2022). For example, the firewood collection in Caatinga covers about 10.35 million m^3 of native vegetation annually (Riegelhaupt and Pareyn, 2010). These disturbances have slow-moving effects that are not always noticed, but they can have an impact on biological diversity at different levels (Martorell and Peters, 2005; Alvarez-Yépiz et al., 2008), especially in areas where rural people face poverty (Ribeiro et al., 2016; Ribeiro-Neto et al., 2016).

Moreover, additional human disruptions pose a threat and are especially novel to the Caatinga environment. For instance, the recent demand for and expansion of wind energy has increased noticeably in Brazil, especially in the Northeast region, where 90% of the nation's wind energy is generated (ANEEL, 2022). This expansion of wind farms seriously threatens biodiversity conservation, particularly in the Caatinga biome's coastal and high-altitude regions in the Northeast (Fig. 33.3C and D) (Pereira-Neto and Silva, 2012). Unbelievably, many of these wind farms are scheduled to be installed in locations previously determined to be high-priority areas for the conservation of Caatinga species (MMA, 2002; Neri et al., 2019). According to the statistics from the most recent satellite imagery study, deforestation in the Caatinga is on the rise, with a startling 70% increase in just 1 year between 2020 and 2021, primarily focused in a few particular cities (MapBiomas, 2023). This evidence supports the idea that, before conservation policies for the region can be implemented, threats to the ecosystem's conservation are opportunistically increasing.

By combining human impacts (e.g., fires, roads, and land use), it was estimated that 63.3% of the Caatinga had already been converted into anthropogenic ecosystems by 2010 (Silva and Barbosa, 2017). A recent study of this biome shows that, despite favorable climate and soil conditions that could support more forest cover, only about 11% of the original vegetation remains, with merely 4.34% of this being classified as forest cover. The study concludes

that over 90% of the potential forest distribution has been altered, highlighting the significant impact of land-use change on the rapid decrease of forest cover in the Caatinga (Araújo et al., 2023). These detrimental effects may be more significant in the Caatinga biome since, in contrast to other biomes that are prone to fire, such as the Brazilian Cerrado, the flora in the Caatinga does not exhibit fire adaptations (Pivello et al., 2021; Bezerra et al., 2022). Hence, anthropogenic disturbances that remove forest cover and hinder the Caatinga ecosystem's ability to regenerate are expected to severely undermine forest resilience, potentially resulting in desertification instead of the reestablishment of native vegetation after land agricultural use (Vieira et al., 2015).

It is hypothesized that climate change and the anticipated rise in aridity in many STDFs will complete the trajectory of deterioration, from forest to desert, at a regional level (Jansen, 1998; Silva and Souza, 2018). Another important fact is that the Caatinga is the tropical biota most vulnerable to climate change (Magrin et al., 2014). Future scenarios of ecosystem degradation, which will show up in the already clear seasonality, will make it harder for the Caatinga to keep providing essential local and global ecosystem services (Silva et al., 2017). There are already signs of these changes in the Caatinga landscape, like the appearance of arid zones within the Caatinga ecosystem (Antongiovanni et al., 2018; Araújo et al., 2023; Tomasella et al., 2023). In fact, despite potentially presenting some adapted species to extreme weather, 99% of the plant communities in the Caatinga biome are predicted to disappear by 2060, according to the growing evidence of the pervasive effects of climate change on the region (Moura et al., 2023). Climate change is, therefore, far more likely to affect the Caatinga biome, a species-rich but threatened tropical dry forest in Brazil, than was previously thought. To preserve the resilience of the Caatinga, its subsistence, and sustainability, it is imperative to reestablish the connection between vegetation and safeguard areas most vulnerable to global warming (Silva et al., 2017).

The Caatinga biome faces several challenges, including the intersection of human activities with the ecosystem. According to the last estimation, the Caatinga region is home to over 29 million people, or 15% of Brazil's total population (Silva et al., 2017). One of the most vulnerable regions in the world is Caatinga due to its socioecological system's inadequate socioeconomic and green infrastructure, extensive degradation, poverty, and climate change (Tabarelli et al., 2017). Even the remaining nonforested areas are seriously in danger due to human activity accumulated over centuries and millennia (Antongiovanni et al., 2020). As a socioecological system on the verge of collapse, the region's human and wild biota are constantly under pressure. Human communities in the Caatinga biome are essentially reliant on natural resources due to the high density of people living there, which places significant strain on the biome's biodiversity (Miles et al., 2006; Hirota et al., 2011; Silva et al., 2017).

Despite significant disruptions caused by historical occupation processes and irrational utilization, the Caatinga vegetation remains a vital subsistence source for local communities, dependent on collecting essential products to sustain and perpetuate their traditional practices. Various ethnic groups within the region employ diverse approaches in exploiting the biological resources of the Caatinga, influenced by social, cultural, economic, and ecological factors (Albuquerque et al., 2005, 2017). Numerous problems brought about by humans threaten Caatinga's extraordinary biodiversity, and these threats are ever-growing due to human activities that have drastically changed its environment and fragmented its ecosystems, from deforestation and agricultural development to industrial growth and animal rearing. The survival of endemic species and the provision of vital ecosystem services necessary for the existence of both wildlife and human societies are at risk if these challenges are not addressed. Thus, proactive conservation measures and legislative initiatives are essential to reduce human impacts and guarantee the long-term survival of this biologically varied and culturally significant ecosystem.

4 Global warming and its impacts on the Caatinga

The Caatinga, a unique dry tropical forest ecosystem in Brazil, faces significant challenges due to climate change. High temperatures, sporadic and erratic rainfall, and protracted droughts are characteristics of the extreme weather that life in this biome has adapted to survive. Nevertheless, forecasts indicate climate change will intensify these conditions, resulting in heightened aridity and elevated temperatures. According to the Intergovernmental Panel on Climate Change's Fourth Assessment Report (IPCC, 2007), climate scenarios show that increasing aridity poses the biggest threat to Caatinga. Northeastern Brazil is expected to see a 22% decrease in precipitation by the year 2100 (Magrin et al., 2014), and the most severe scenario could see surface air temperatures rise by up to 4°C by the end of this century (Torres et al., 2017). While the exact extent to which the Caatinga can tolerate climate change remains uncertain, there are several factors to consider; in this context, some studies have already made efforts to make predictions of the impact on their ecosystems in scenarios of future changes.

Drought is a recurring phenomenon in the Caatinga biome, characterized by its semiarid climate with irregular and unpredictable rainfall patterns. Predicted global warming is expected to exacerbate these conditions, leading to more frequent and severe droughts in the region (Cunha et al., 2018). Approximately 400,000–500,000 people died during

one of these occurrences, the Great Drought (1877–1879), the most destructive drought in Brazilian history. Most of them perished from starvation and infections brought on by a large-scale exodus of people from the Semiarid Northeast of Brazil in 1877 after the harvest failed (Cunniff, 1970; Marengo et al., 2017). The Semiarid Northeast of Brazil (a geographical aggregation of mixed grasslands, "Caatinga" shrublands, and savanna) has experienced a prolonged drought in recent years. Beginning in 2011, the drought intensified in 2012 and persisted until 2017, impacting around 9 million people and multiple towns; this was one of the worst events in decades. Despite the difficulties caused by the drought, some of its consequences have been lessened by government initiatives to mitigate it, such as lending money to small farmers and delivering water in tank trucks (Brito et al., 2018).

Reductions in rainfall and an increase in aridity are predicted for Semiarid Northeast Brazil due to global and regional climate change (Marengo and Bernasconi, 2015; Vieira et al., 2015). The region is among the most vulnerable to the effects of climate change globally because of this unpredictability of rainfall, exacerbated by land degradation and aridification (Souza and Oyama, 2011; IPCC, 2013, 2014; Marengo et al., 2017). Because of these profound changes, the Caatinga currently includes large regions impacted by desertification (Souza et al., 2015). Climate projections for the Caatinga region suggest that some areas would likely become drier, which could result in desertification (MMA, 2007; Marengo et al., 2017; IPCC, 2019). According to the satellite data, between 1985 and 2020, 9% of Caatinga's land area changed into desertification-prone regions (MapBiomas, 2023). This development suggests that these areas would struggle with depleted soil, making it even more difficult for human habitation and productive endeavors. The Caatinga's considerable desertification has rendered any economic use of these regions impractical (MapBiomas, 2023). From a different perspective, the anticipated aridification due to climate change may promote the emergence of nonwoody plant species with a wide distribution, altering the structural complexity of the vegetation (Moura et al., 2023).

The Caatinga biota have evolved various adaptations, including the capacity to retain water in their stems to survive in harsh environments. During severe droughts like the ones that happened in 2012, tests of water potential in different conditions showed that areas that have been in conservation for more than 60 years have higher soil water availability and leaf water potential. In these areas, species with less wood density store more water and have less annual variation in leaf water potential (Santos et al., 2014). However, Moura et al. (2023) estimate that, in any future temperature change scenario, about 90% of Caatinga plant species—especially narrow-ranged species—will lose appropriate sites, and more than 99% of Caatinga plant assemblages are predicted to lose species by 2060.

Global warming poses a serious threat to the Caatinga biome, which may impact the preservation of biodiversity and the health of ecosystems. Even under conservative projections of future climate change, for example, endemic plants are expected to have their climatic ranges reduced. This will impact species with specialized reproductive strategies, like those that require pollination (Silva et al., 2019). Additionally, it is expected to have an impact on the availability of suitable habitats for medicinal plants within the Caatinga; according to the projected climatic scenario for the year 2090, roughly 70% of the species studied are likely to see reductions in their distribution areas (Silva et al., 2022). Due to climate change, some frog species may experience spatial reconfiguration of their current distributional and diversity patterns, including regional shifts that could reduce local species richness and range size (Medina et al., 2020).

Species may migrate toward higher elevations or more suitable habitats in response to changing climatic conditions, leading to changes in species composition and ecosystem dynamics. For example, as an alternative strategy, certain species of birds can migrate to humid areas during extended drought periods and return to semiarid regions during the rainy season (Silva et al., 2003). However, if suitable habitats become scarce, endemic species with specific habitat requirements may be more vulnerable to range contraction or extinction. For instance, estimates of the possible future range of epiphytic cactus species revealed habitat loss (Simões et al., 2020, 2021) and a spatial contraction of more than 88% of species (Cavalcante et al., 2023). However, as predicted for some fish (Egea-Serrano et al., 2022) and plant species (Rabelo-Costa et al., 2022), other species might become extinct if they lack this adaptability or migratory behavior.

Biological invasions significantly contribute to the global reduction of biodiversity, which is a cause for concern. Dry locations are particularly vulnerable to these losses because of their unique flora and poor ability to recover after destruction. In these settings, invasive species become effective rivals, making native species' struggles worse (Gonçalves-Oliveira et al., 2022). With rising temperatures and changing precipitation patterns predicted for the Caatinga biome, the ecosystem is more vulnerable to the influx and spread of exotic species. These intruders, frequently mistakenly brought in by human activities like commerce and travel, can outcompete native species for resources and upset sensitive ecological balances.

Although there are still uncertainties around climate change projections, current scientific knowledge offers enough proof of negative climate alterations in Caatinga (Torres et al., 2017). Better models are also needed to estimate habitat loss in the Caatinga region at a time when the environment is changing quickly and significantly. When combined with better distribution data, these models make more accurate predictions of the effects of global warming.

These effects could negatively impact the provision of other ecosystem services, carbon sequestration, biodiversity, and the quantity and quality of water resources. Hence, harmonizing human populations, biodiversity, ecosystem services, and climate adaptation through sustainable development is urgently necessary, demanding entirely novel perspectives on the region (Tabarelli et al., 2017).

5 Conservation areas and their representativeness

The Caatinga has historically received little funding for conservation initiatives, making it one of Brazil's most overlooked and little-studied biomes (Silva et al., 2017). With fewer than 8% of the biome designated as Conservation Units and legally protected under Brazil's national nature reserve laws (SNUC law), the Caatinga is currently underrepresented in the country's network of Conservation Areas. Only 1.3% comprises Integral Protection Conservation Areas within this allocation, while 6% falls under Sustainable Use Conservation Areas (Fig. 33.4) (Teixeira et al., 2021; ICMBio, 2024). Besides that, Environmental Protected Areas, among the least restrictive categories, constitute the more representative portion of the Caatinga Protected Areas (PA) system, accounting for nearly 80% of the total area under legal conservation (Teixeira et al., 2021). These factors are a concerning reality, as degradation has led to calls for the enlargement of this network (Specht et al., 2019; Antongiovanni et al., 2020). Moreover, the objective set by the Convention on Biological Diversity's Aichi agreement, aiming to have 17% of the biome designated as protected areas by 2020, remains unmet and distant (Teixeira et al., 2021).

The region has seen a significant increase in deforestation in recent years; according to MMA (2011), 45.62% of the domain was deforested in 2009, and only 50% of the Caatinga preserved is thought to consist of isolated vegetation patches smaller than 5 km^2 (Antongiovanni et al., 2018). In the year 2021, 30.5% of the Caatinga's territory, encompassing over 26 million hectares, was designated as a Biosphere Reserve, and Conservation Units (CUs) cover 9.01% of the biome, totaling 7.8 million hectares (MapBiomas, 2023). Despite its relatively high diversity, merely around 1.30% of the Caatinga is fully legally protected, with an additional 7.96% enjoying weaker legal protection (Leal et al., 2005; Hauff, 2010; de Oliveira et al., 2012; Teixeira et al., 2021). Moreover, four of the 13 primary vegetation types identified

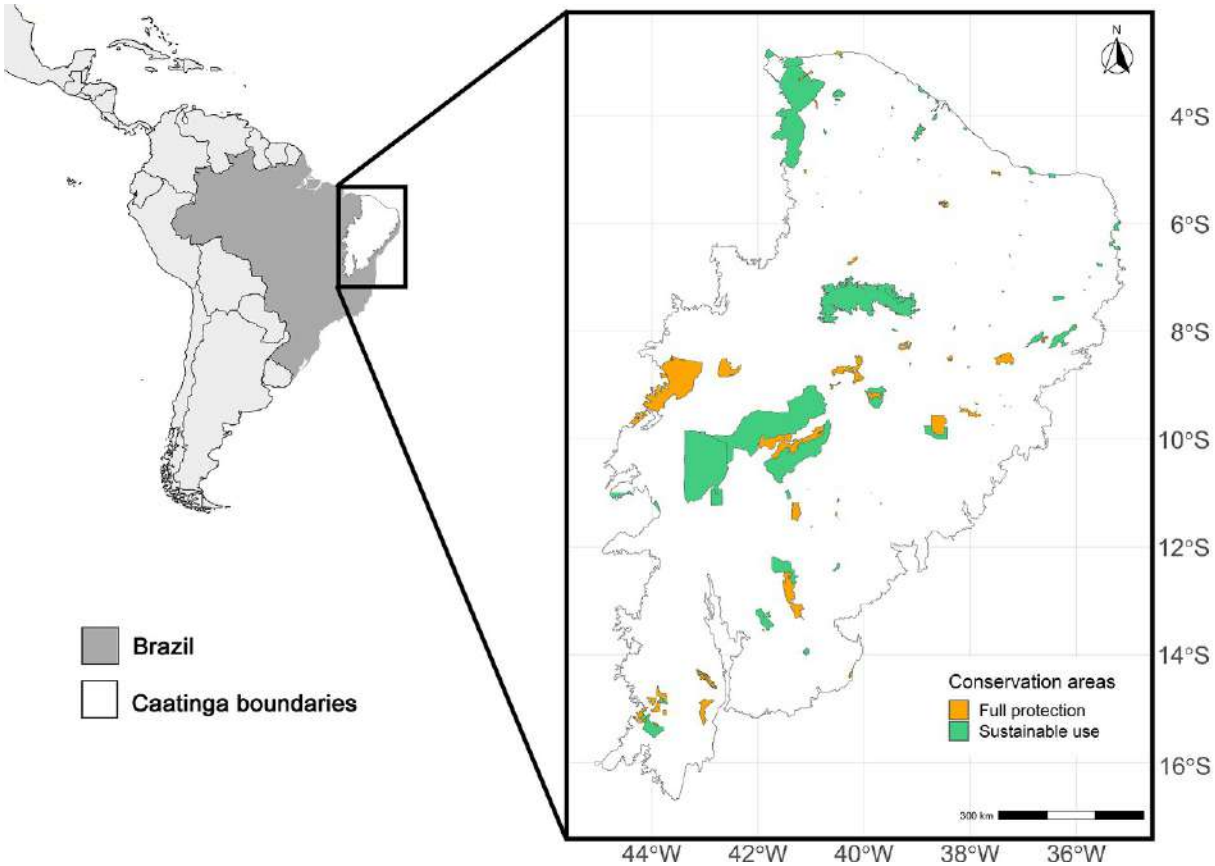

FIG. 33.4 Caatinga boundaries and its conservation areas. *(Source: Elaborated by the authors according to ICMBio – Instituto Chico Mendes de Conservação da Biodiversidade, 2024. Plataforma oficial de dados do Sistema Nacional de Unidades de Conservação da Natureza. Retrieved from: https://cnuc.mma.gov.br.)*

in the Caatinga (sensu Prado, 2003) lack representation within any conservation unit (Tabarelli et al., 2000).

The Brazilian federal government's approach to PAs changed during the 1990s, with a rise in the quantity and size of Sustainable Use PAs. This modification encourages biodiversity conservation while more effectively integrating the nation's environmental concerns with its objective of economic development (Cabral and Brito, 2013). Before 2000, when state investments in PAs were minimal, the majority of large federal PAs were founded. However, state governments have significantly expanded the number and area of PAs in the Caatinga over the last 25 years. Initiatives started by the Ministry of the Environment in 2000 that organized workshops to identify priority conservation areas have supported the establishment of new PAs (Rylands and Brandon, 2005; Gouveia et al., 2010). After a series of participatory systematic conservation planning workshops, 150 priority areas for the conservation of Caatinga biodiversity were identified (Fonseca et al., 2017). Of these, 79 were designated as Full Protection Conservation Units and 54 as Sustainable Use Conservation Units, codified into Brazilian legislation in 2016. This was a significant achievement; however, as of 2023, only 20 new areas had been legally created in these two categories (ICMBio, 2024).

In addition, a dozen investigations were carried out to determine the biome's possible conservation areas and their importance (e.g., Oliveira et al., 2016; Manhães et al., 2018; Fonseca et al., 2017). However, the unique characteristics of the Caatinga, such as its high population density, widespread use of natural resources, and traditional land tenure arrangements, make the establishment of new strict PAs difficult (Gouveia et al., 2010; Salvatierra et al., 2017). Since the 1980s, scientific communities, nongovernmental organizations, and civil society organizations have driven a paradigm shift toward the economic and regional development of the Caatinga biome (de Castro Cardoso, 2007). This transition has influenced public policies with alternative approaches to land management and utilization, the main goal of which is to reconcile economic development with sustainable practices, foster Caatinga conservation, and facilitate society's adaptation to the region's semiarid conditions.

With an increase in the number of conservation units under state management and with more permissive use, such as the Sustainable Use Conservation Units of the Environmental Protection Area (APA) category, the dynamics of creating conservation units in Brazil have changed over time, following a national trend (Teixeira et al., 2021). In addition to failing to reach adequate protected areas for biodiversity conservation in the biome, the network of protected areas in the Caatinga also presents deficiencies in management and budget (Oliveira and Bernard, 2017) and most of them do not cover areas of high endemic species, such as cacti species (Carvalho et al., 2022). Despite these limitations, conservation units have great potential to mitigate the impacts of desertification and degradation in the Caatinga. For example, strictly protected areas in the Caatinga exhibited higher productivity and greater resilience to low precipitation levels (Salvatierra et al., 2017). Another study demonstrated that integrating Caatinga's protected areas system with a designated set of priority areas can effectively preserve suitable climatic conditions for endangered birds amidst climate and land cover changes. On average, the protected areas system of the Caatinga could safeguard climatic stability areas at least 1.7 times more effectively than scenarios without its inclusion (Silva et al., 2020). This underscores the vital role of protected areas in safeguarding ecosystem processes and native vegetation against the adverse impacts of climate change and deforestation.

The Caatinga has generally experienced rapid and abrupt environmental changes directly or indirectly due to human activities (Dias et al., 2019; Araújo et al., 2023). Such environmental degradation and the subsequent loss of habitat may further limit options for conservation, thereby constraining vital conservation areas in future climate scenarios (Vieira et al., 2015, 2021; Gonçalves-Souza et al., 2021). Disrupting landscape structure can lead to changes in species composition, the extinction of endemic species, and disruptions to essential ecological processes needed for ecosystem functioning and biodiversity preservation, leading to significant impacts on protecting habitats and species (Hagan et al., 1996; Castelletti et al., 2003; Püttker et al., 2020).

Moreover, the quest for effective conservation strategies necessitates an interdisciplinary approach; alternative approaches beyond the establishment of Conservation Units (CUs) are essential for effectively safeguarding Caatinga's biodiversity (Dória and Dobrovolski, 2021). Research on conservation in the Caatinga is relatively scarce and not uniformly distributed; however, there is a trend of studies focusing on areas most prone to desertification (Lessa et al., 2019). Given that many species and taxa exclusive to Brazil's unique Caatinga region are not found anywhere else, this region's ongoing deterioration and depletion are of worldwide concern (Queiroz et al., 2017). Conservation strategies tailored to the Caatinga biome should adopt a comprehensive approach to tackle its unique challenges. They should develop strategies integrating scientific research, traditional ecological knowledge, and community engagement in conservation efforts, all reinforced by supporting laws and rules. With only a tiny portion of the Caatinga currently under strict protection, it is imperative to prioritize establishing new conservation areas to maintain ecological processes and mitigate further desertification.

6 Operational and potential monitoring tools to conserve or recover the biome

Compared to what probably existed in the same climate and soil conditions before human disturbance, only 11% of the original vegetation of the Caatinga biome is still there. This

means that nearly 90% of the potential forest distribution has been changed (Araújo et al., 2023), which goes beyond ecological thresholds and requires immediate forest restoration efforts to protect biodiversity and ecosystem services (Manhães et al., 2016). Active forest restoration initiatives are urgent for safeguarding biodiversity and maintaining essential ecosystem services in the Caatinga biome. Moreover, various operational and prospective monitoring measures can be used to conserve and restore the Caatinga efficiently. These consist of activities for knowledge development, policy, and governance frameworks, biodiversity surveys, restoration procedures, conservation efforts, and the incorporation of citizen science programs.

The primary concern lies in the agricultural environment prevailing in Neotropical arid regions; these landscapes, predominantly occupied by cultivated fields and grazing areas, need to implement effective land management practices in settings that enhance rainwater utilization, soil moisture retention, mitigate soil deterioration, increase conservation efforts (and restoration initiatives), and provide a broader array of sustainable agricultural practices (Peterson et al., 2006; Stewart, 2016; Araújo et al., 2021). One proposed strategy to achieve sustainable food productivity while promoting biodiversity conservation is integrating forests with crops and/or cattle farming (Balmford et al., 2012). Maintaining native forests on at least 50% of rural properties is suggested to mitigate water loss from the soil, enhancing productivity, especially during dry years (Araújo et al., 2021). This approach is critical given the projection of more frequent and prolonged droughts due to climate change. Additionally, biotic homogenization is expected in approximately 40% of Caatinga plant communities, particularly in regions with low species diversity (Moura et al., 2023). To address these challenges, creating complex landscapes becomes crucial. High tree coverage in such landscapes helps maintain soil moisture and critical ecological functions (Araújo et al., 2021). Therefore, maintaining agricultural systems within complex ecosystems is proposed as the best strategy for increasing food security during extreme climatic events (Solh and Ginkel, 2014; Araújo et al., 2021).

Biodiversity monitoring is critical, with regular surveys and remote sensing technologies used to measure plant and animal species' quantity, distribution, and dispersion and evaluate the continuous impact in the region. In this context, long-term ecological research is fundamental, with monitoring sites established to assess changes in ecosystem structure and function over long periods, providing an indicator of ecosystem resilience in the face of climate change (Bergallo et al., 2023). These endeavors would be especially labored in the Caatinga, where working conditions are challenging, with daytime temperatures frequently soaring to 40°C and the native flora boasting a striking assortment of thorns, spines, and hooks (Lessa et al., 2019). Though it is the only Brazilian natural area fully contained inside national borders, not enough emphasis has been placed on surveillance. However, the Caatinga biota's contribution to Brazil is remarkably high in biodiversity and has been overestimated (Silva et al., 2004).

Comprehensive monitoring is necessary to assess the progress of habitat restoration activities, and restoration procedures in the Caatinga are also crucial. Priority restoration areas, such as landscape resilience, connectivity, and biodiversity conservation value, should be chosen based on data. The selected restoration areas are intended to guard against habitat loss, fragmentation, and the potential threats posed by climate change to endangered indigenous plant species (Antongiovanni et al., 2022). Beyond merely regaining vegetation, the emphasis is on restoring soil structure and facilitating the successful reintroduction of native species in rehabilitated areas. This involves addressing problems like ongoing human disturbances, agricultural needs, and the possible effects of climate change, all of which highlight the necessity of efficient restoration techniques (Salazar et al., 2021; Antongiovanni et al., 2022). Furthermore, as the Caatinga still has significant portions that are essential to regional connectivity, these restoration efforts need to be customized to the unique circumstances of this seasonally dry tropical forest (Tambosi et al., 2014; Antongiovanni et al., 2018; Salazar et al., 2021).

In the context of the Caatinga biome, policy and governance monitoring involves methodically assessing the efficacy of conservation policies and the governance frameworks established to implement them (Teixeira et al., 2021). It verifies the efficacy of conservation efforts while assessing adherence to laws about logging, land use, and other activities that impact the Caatinga. Policymakers and environmental authorities can resolve enforcement gaps, adjust to new issues, and improve existing policies by regularly performing assessments. The main objectives are ensuring the best possible conservation of Caatinga's distinctive biodiversity, reducing the effects of human activity, and advancing sustainable land management techniques. Therefore, strong public policies are needed to support a new development paradigm (Silva et al., 2017).

Enhancing the understanding of the Caatinga biome requires a multifaceted approach involving both the scientific community and local communities in knowledge generation and conservation efforts. However, the scientific approach to studying the Caatinga presents challenges. It has received the lowest scores in biodiversity survey efforts and knowledge status, resulting in relatively few internationally published papers in conservation science (Lessa et al., 2019). This research gap is concerning because a thorough understanding of the ecological background is necessary for effective and responsive conservation policies. However, compared to humid tropical forests like the Amazon and Atlantic Forest, Caatinga supports fewer research teams and senior researchers (Santos et al., 2011). The long-standing lack of

research infrastructure can partially explain this barrier to knowledge production; prior to the early 21st century, most Brazilian universities were concentrated in large urban centers, mainly in the nation's southern regions, far from the Caatinga region (Pedrosa, 2014).

Consequently, the Caatinga suffers from a significant lack of information about its biodiversity, despite its potential (Fig. 33.5) (Ferreira et al., 2016); previous research has revealed that 40% of the plant species within the Caatinga biome have never been surveyed on-site,

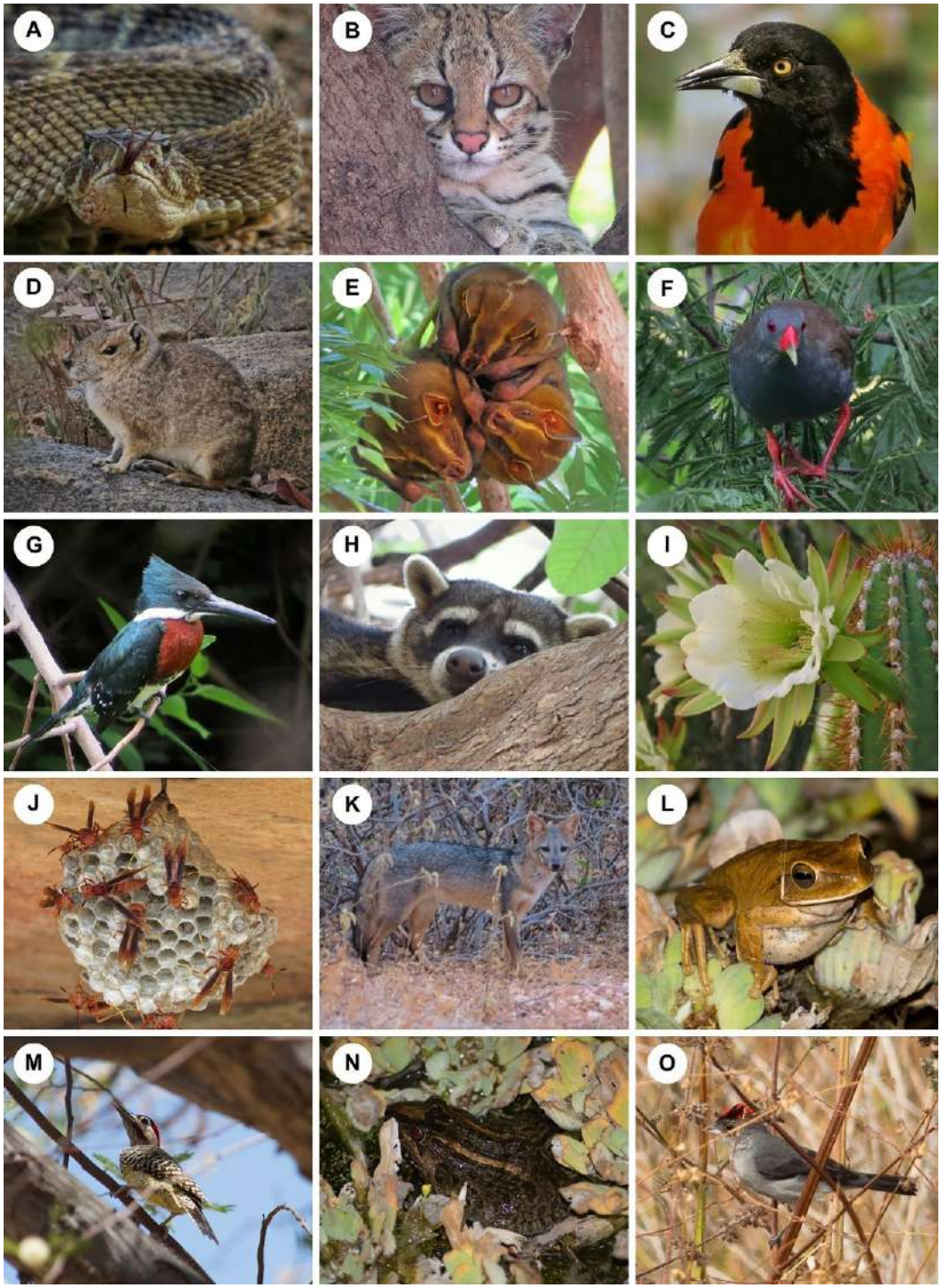

FIG. 33.5 Flora and fauna from Caatinga. (A) *Crotalus durissus*; (B) *Leopardus emiliae*; (C) *Icterus jamacaii*; (D) *Kerodon rupestris*; (E) *Platyrrhinus lineatus*; (F) *Neocrex erythrops*; (G) *Chloroceryle amazona*; (H) *Procyon cancrivorus*; (I) *Cereus jamacaru*; (J) *Polistes canadensis*; and (K) *Cerdocyon thous*; (L) *Hypsiboas raniceps*; (M) *Colaptes melanochloros*; (N) *Leptodactylus macrosternum*; (O) *Coryphospingus pileatus*. *(Photo credits: José Vinícius Alexandre de Medeiros (A–K) and Emanuelle Luiz da Silva Brito (L–O).)*

and most documented species are only found in one location (Moro et al., 2014). Due to these challenges, Caatinga stands out as one of the least investigated areas in Brazil (Santos et al., 2011), with numerous biological groups still needing more comprehensive study (e.g., Leal et al., 2017; Lima et al., 2017; Garda et al., 2018). There are significant limitations in recent species records across most taxonomic groups, with a considerable concentration of records in a few well-sampled groups (e.g., Magnoliopsida). There is also a reflection of transportation infrastructure in the concentration of records, especially road density, human density, and the proximity to universities and protected areas also influences recording efforts (Correia et al., 2019). Interestingly, vegetation cover negatively impacts the knowledge of biotic information, likely due to the influence of accessibility on sampling effort (Tabarelli and Vicente, 2002; Correia et al., 2019; Lessa et al., 2019). Despite this, some notable initiatives are apparent; for example, a recent inventory of Caatinga's flora revealed a 6% increase in registered species (Queiroz et al., 2017; Fernandes et al., 2020).

One particularly effective approach to improve Caatinga's knowledge is community-based monitoring, which provides a cooperative framework in which residents actively participate in observing, recording, and comprehending biodiversity and ecological health changes. Given that local people have unique environmental perspectives, some studies have highlighted how important it is to include them in conservation initiatives (e.g., Pedrosa et al., 2019, 2023). Programs like citizen science further expand knowledge by integrating citizens in data collection, analysis, and interpretation. They democratize the process of creating information and make it more inclusive and representative of various viewpoints. This method not only helps us comprehend the complex ecological dynamics of the Caatinga better, but it also gives local communities the tools they need to take care of their natural legacy. Combining these monitoring tools can provide a comprehensive understanding of the Caatinga biome, guiding adaptive and effective conservation and recovery strategies. It is crucial to tailor monitoring programs to the unique characteristics and challenges of the Caatinga ecosystem, ensuring a holistic approach to its preservation.

7 Conclusions and perspectives

To the people of the Caatinga, their land sounds like poetry. Each part of its landscape serves as a stanza in a beautiful verse, full of passion and resilience. Or, as a novel rich with fervor, each chapter tells tales of adaptation and strength, revealing its profound complexity. This sentiment embodies the essence of the Caatinga, where the human soul intertwines with the landscape, forming links that transcend the harsh realities of its nature. Despite the challenges, life in the Caatinga demonstrates the tenacity of its people, who draw strength from the land and its stories. They transform adversity into opportunity and hardship into inspiration, which finds expression in diverse art forms such as literature, poetry, paintings, music, cinema, and handicrafts, reflecting the indomitable spirit of the Caatinga and compounding its cultural legacy. Yet, beneath its surface lies a profound duality.

Contrary to this essence, human impacts pose the primary threat to this ecosystem; the Caatinga biome faces an array of human-caused threats, including deforestation, habitat fragmentation, and overexploitation of natural resources, all of which endanger biodiversity and ecosystem health. The extensive conversion of natural areas into human-modified landscapes reduces the habitats of numerous plant and animal species and disturbs critical ecosystem processes and services that benefit humans. Furthermore, global warming-related effects like increased aridity and rising temperatures exacerbate the Caatinga's conservation challenges. Conservation activities must prioritize sustainable land-use practices, restoration of degraded area, and the overall conservation of ecosystems. As a result, immediate action is required to establish proactive conservation measures and legislative initiatives that protect this culturally significant ecosystem and its exceptionally rich biodiversity. By focusing on sustainable development and biodiversity protection, we can work toward ensuring the resilience and integrity of the Caatinga biome for future generations.

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Section K

Tundras

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Chapter 34

Asian Arctic tundra: Vast permafrost ecosystems under increasing pressure from climate change and industrial development

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1 Introduction

Moving north, the dense boreal forest is followed by the forest tundra ecotone. Patches of treeless areas are increasing in size until they reach the last trees at the latitudinal treeline, opening up into the tundra (Payette et al., 2001). The tundra biome is characterized by vast landscapes with treeless vegetation with low stature, productivity, and slow growth (Fig. 34.1).

These characteristics are closely tied to the harsh climatic conditions with very low winter temperatures and a short growing season. Low temperatures bring about many other abiotic conditions that limit biological activity and life in the Arctic tundra. Besides the lack of summer warmth, the Arctic tundra biome receives relatively low amounts of precipitation, mainly falling as snow. It is mostly located in the continuous permafrost zone with a shallow unfrozen soil layer, the so-called active layer, during summer time, providing only a shallow rooting zone, limited biological activity, and nutrient availability (Haag, 1974). Nitrogen is the most limiting element, but phosphorus is often also limiting or colimiting (Street et al., 2018). The nutrient limitation is constrained by a slow nitrogen fixation, limited decomposition and mineralization, and fast immobilization by soil microorganisms due to low temperatures (Schimel et al., 1996). Another important growth-limiting factor is light. Due to the generally low light intensity, Arctic plants are rarely light saturated (Chapin, 1987). Furthermore, the Arctic tundra experiences months of almost complete darkness during winter. During the months when the sun is below the horizon for the entire day, the primary production of plants is close to zero, and metabolic processes in animals are reduced (Blix, 2016). The primary production of the Arctic tundra biome is limited to around 50–75 days per year during which plants must produce new leaves and roots (Billings and Mooney, 1968; Bliss, 1966).

The Arctic tundra biome is not only characterized by limitations, but also by abundance. There is usually no shortage of ice, cryptogams, and mosquitoes in this biome. The permafrost shapes the landscapes of the arctic tundra through freeze-thaw cycles, resulting in unique geomorphological landforms such as ice-wedge polygons, or thermokarst lakes, bogs and fens, thaw slumps, and thermal erosion gullies (Jorgenson and Grosse, 2016). Arctic soils are strongly linked to permafrost through cryogenic processes, which form surface soil structures and macromorphological features within the soil (Tarnocai, 2009). Further, the permafrost table acts as a moisture barrier, and hence Arctic soils are generally moist (Tarnocai, 2009). Permafrost dynamics are strongly linked with vegetation. Tundra vegetation, especially mosses, often build a thick insulating layer, reducing permafrost thawing (Blok et al., 2011; Porada et al., 2016). On the other hand, permafrost has also pronounced effects on vegetation by limiting the rooting depth of plants (Blume-Werry et al., 2019). Indirect permafrost effects on vegetation are mediated by soil hydrology, soil biogeochemical processes, and microbial communities, which further influence soil nutrient cycling and availability (Jin et al., 2021). Overall, vegetation type is an important predictor of land surface energy fluxes, regulating water and energy exchange between the atmosphere, the permafrost, and the active layer (Oehri et al., 2022).

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FIG. 34.1 (A) The Arctic tundra is a treeless biome with short-statured vegetation and slow growth. On the way out to the tundra in the Eastern Siberian Arctic, travelers briefly stop to worship the last tree by leaving some food, before continuing into the vast treeless plains (northwest of Chokurdakh, on the way to Kytalyk research station, April 2015). (B) At the northern limit, the Asian Arctic tundra reaches the Arctic Ocean, with waves eroding the permafrost along the coastal areas and major rivers (eastern part of the Yamal Peninsula along the Venuymuyeyakha river). *(Credit: Gabriela Schaepman-Strub, Vitalii Zemlianskii.)*

The permafrost and the active layer also interact with Arctic wildlife. They help determine, for example, soil invertebrate communities (Nielsen and Wall, 2013), affect bird communities and the biomass of mosquitoes by regulating standing water (Corbet and Danks, 1973), and influence animals living in Arctic lakes (Vonk et al., 2015). But, there is not only an influence of permafrost on animals but also interactions in the other direction. Grazing animals and, especially, large herbivores could contribute to establishing a more stable thermal regime by compacting the ground through their weight. This compression reduces the insulating properties of snow cover, allowing for more intense freezing of the underlying permafrost during winter months (Beer et al., 2020; Windirsch et al., 2022).

In the past, the tundra ecosystem was described as rather stable (Dunbar, 1973). However, while being one of the coldest biomes of the Earth, it is disproportionately affected by global warming and other global change drivers (Fig. 34.2).

In the Arctic tundra, biome temperature increase is four times faster than the global average (Rantanen et al., 2022). This enhancement of temperature change is called Arctic amplification and is related to local and global feedbacks,

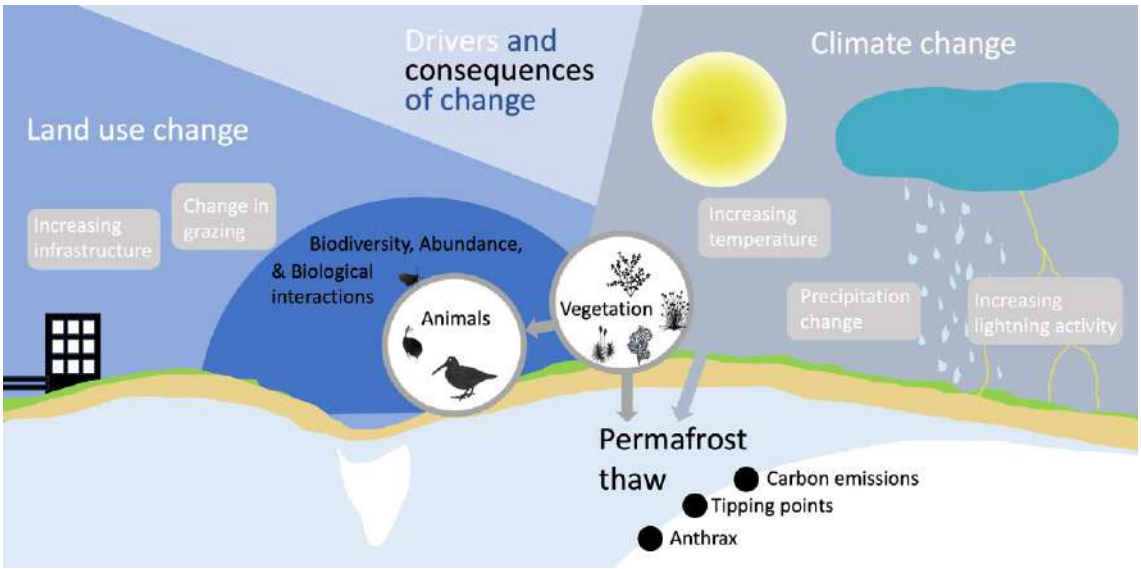

FIG. 34.2 Climate change is currently the strongest driver of large-scale change of Arctic biodiversity and ecosystem functioning. However, land use change is increasing and has strong local impacts on biodiversity. These two global change drivers affect vegetation, animals, and permafrost, triggering further feedbacks, such as carbon emissions to the atmosphere or release of viruses and bacteria that are potentially harmful to people and animals. The Arctic is a highly interconnected system, with important global feedbacks. *(Credit: Ramona Heim.)*

which include temperature and sea ice-related feedbacks and increases in atmospheric and oceanic poleward energy transport (Previdi et al., 2021). Amplified climate warming induces strong temporal and spatial changes to the Arctic tundra biome (Chylek et al., 2022; Niittynen et al., 2020; Rantanen et al., 2022). Besides the rising temperatures, it is also predicted that precipitation will increase significantly (Bintanja and Andry, 2017; McCrystall et al., 2021; Vihma et al., 2016). This is the case due to a higher capacity of the atmosphere to carry water due to increased air temperature (Bintanja, 2018), moisture transport to the poles (Zhang et al., 2013), and evaporation through more open water related to sea ice loss (Bintanja and Selten, 2014). Alongside global warming, Arctic disturbance regimes are undergoing transformations. The occurrence of tundra fires is increasing due to a combination of climate change effects and human activity. Due to the increase in precipitation and convective available potential energy, convective storms and, hence, lightning activity are predicted to increase in Arctic regions (Chen et al., 2021; Veraverbeke et al., 2017). New roads open up territory and thus provide easier access to remote area, which leads to more human ignitions (Foster et al., 2022). Finally, increases in woody biomass and, thus, increased fuel are further fostering fire activity in the Arctic tundra (Hu et al., 2015).

Another significant driver of disturbance in the Asian Arctic is reindeer herding. While the grazing pressure decreased in some regions, it strongly increased, particularly in the Western Siberian Arctic during the last decades (Uboni et al., 2016). Also, infrastructure development in the Russian Arctic related to gas and oil extraction affects the environment strongly, through the construction of pipelines, roads, and railways (Kumpula et al., 2011). Even small disturbances with low intensity can have long-lasting impacts on the Arctic environment as recovery processes are slow because of the short growing season and low temperatures (Rebristaya et al., 1993).

We, therefore, need to reevaluate the stability of this biome, which can only be done by understanding its peculiarities linked to permafrost and its fragility defined by various growth limitations (Armstrong McKay et al., 2022; Bliss et al., 1973; Callaghan et al., 2022). The permafrost in the Arctic tundra is vulnerable to thaw and collapse with strong implications as it is directly or indirectly linked to main ecosystem components and processes of this biome (Bliss et al., 1973). A generally low biodiversity and slow growth make the Arctic tundra especially vulnerable to change, as every impact has long-term effects on the biome (Chapin, 1987; Chapin and Körner, 2013). In addition, this biome is one of the simplest biomes regarding structure, functioning, and biodiversity (Walker et al., 2001; Fauteux et al., 2021). However, it hosts cold-adapted species that are unique to the global biodiversity. And despite its fragility and simplicity, the tundra supports the livelihoods of Arctic residents, especially Indigenous Peoples. Moreover, the tundra serves as a crucial breeding habitat for millions of migratory avian species, forming an intricate ecological nexus that interconnects the Arctic realm with distant regions spanning the entire globe, as far as Antarctica. Finally, boreal permafrost, with a major fraction located in the tundra biome, is estimated to store about 1672 pg carbon, which is roughly twice as much carbon currently contained in the atmosphere (Miner et al., 2022; Tarnocai, 2009). Therefore, boreal permafrost has been defined as a tipping element of the global climate system (Armstrong McKay et al., 2022). Climate warming is already impacting the Arctic tundra biome strongly, leading to permafrost thaw and thus threatening the large amounts of carbon stored in the permafrost (Plaza et al., 2019).

The Arctic tundra biome is impacted by local and global changes and the resulting ecosystem modifications, in turn, have far-reaching local and global feedbacks and consequences that are complex to predict (Wookey et al., 2009). Therefore, it is even more important to prevent or at least mitigate these consequences through conservation efforts. Local decision-making and global actions are of high relevance to mitigate global change effects on Arctic ecosystems (Vincent, 2020). Locally, conservation measures such as protected areas are very important (Reji Chacko et al., 2023), as well as efforts to understand how to stabilize Arctic ecosystems and, thus, permafrost (Callaghan et al., 2022). A global measure is to counteract the rapid progression of climate change (Hornsey and Fielding, 2020; Anderson et al., 2020). While geoengineering, specifically solar radiation management, is currently discussed as a potential solution to "refreeze" the Arctic, impacts of the remaining high carbon dioxide concentration on Arctic biodiversity and community composition, as well as through acidification are not well understood. Further, a recent study has shown that plant physiological forcing might lead to a reduced cooling effect in the Arctic (Müller et al., 2024).

In this chapter, we will shed light on the impacts of global change on biodiversity and ecosystem functioning of the Asian section of the Arctic tundra. First, the geographic location of the Arctic tundra and its Asian section is introduced, followed by an overview of the biodiversity of the flora and fauna and impacts of climate change and human activity. Current protected areas and additional conservation measures for the Asian Arctic tundra are discussed to conclude this chapter.

2 Geographic location

The Arctic tundra has a circumpolar distribution in the Northern hemisphere where it encloses the Arctic Ocean.

The southern boundary of the Arctic tundra biome is given by the subarctic tree line and is very variable in terms of latitude, as visible in the Circumpolar Arctic Vegetation Map (CAVM). In the north, the Arctic Ocean generally forms the boundary of the tundra, while tundra can even occur on Arctic islands. However, many polar island ecosystems belong to the polar deserts, as climate conditions become so harsh that any vascular plants are rarely able to grow anymore. While the tundra biome spans across the Arctic, and is also present in high-altitude regions of the world and in Asia, this chapter focuses on the latitudinal Asian tundra biome, entirely situated in the Russian Arctic and extending from the Ural Mountains in the West to the Bering Sea in the East.

3 Biodiversity

The Arctic tundra is a relatively young ecosystem. Based on the analysis of 32 angiosperm clades, a recent study suggests an initiation and diversification of the Arctic flora since the early Late Miocene (Bliss et al., 1973; Zhang et al., 2023). Dispersal into the Arctic and in situ diversification mainly relate to changes in landscape and climate, and to sea-level fluctuations. Many areas were glaciated or inundated multiple times, while others remained ice-free and served as refugia for terrestrial species, including a diverse fauna that has partially gone extinct. The thawing permafrost is increasingly releasing the remnants of the past, including large herbivores such as wooly mammoths and rhinoceros, but also ancient viruses and bacteria potentially harmful to today's animals and humans (Alempic et al., 2023). Previously glaciated areas are low in plant diversity up to today, while the tundra overall has a lower biodiversity compared to other biomes of the Earth. The Arctic contributes about 3% to the global flora, and about 2% to the global fauna (Callaghan et al., 2004). However, organisms surviving the harsh Arctic conditions are remarkably adapted to survive the extreme cold and highly variable climate. They are joined during the warmer season by migratory animal species that find favorable conditions of a low predator density in tundra. They raise their offspring during the summer, and leave the tundra before conditions become inhospitable again for most species.

Spatial heterogeneity of the tundra landscape is shaped by permafrost dynamics, ranging from large drained thaw lake basins to small-scale, polygonal patterns of few square meters. These processes generate topographic structures affecting wind exposure and snow accumulation, ultimately leading to diverse microclimatic environments, characterized by different air and soil temperature, soil moisture, and active layer thickness. This landscape heterogeneity is interacting with plant communities and their structure, creating ecological niches for animals. Local terrestrial diversity is enhanced in many tundra areas by abundant freshwater bodies and by marine birds along the coast of the Arctic Ocean.

Due to the vast area, remoteness, and permafrost conditions, accessibility to the tundra is very limited and, hence, strongly undersampled compared to other biomes of the Earth. However, the Arctic Council, an intergovernmental forum of the eight Arctic nations and six permanent participants representing Indigenous Peoples of the Arctic, maintains a working group on Conservation of Arctic Flora and Fauna (CAFF). CAFF compiles and synthesizes existing data and knowledge through their Circumpolar Biodiversity Monitoring Program (CBMP) and has performed a comprehensive assessment of Arctic biodiversity (CAFF, 2013). This assessment is regularly updated for the major ecosystems of the Arctic, including the Arctic terrestrial ecosystem (CAFF, 2021). While documenting some trends of changes, the report highlights many knowledge gaps in the Arctic among all organism groups. Despite knowledge gaps, the following sections include a brief description of the diversity of life along biological organization inhabiting the Asian Arctic tundra, from lichens to plants to animals.

3.1 Flora and vegetation of the Asian tundra

The vegetation in the Arctic tundra contains 2218 species across all floristic provinces (CAFF, 2013; Fig. 34.3A) and is very rich in bryophytes (especially mosses and liverworts) and lichens, while vascular plants can contribute less than 50% to the total species richness locally (e.g., Iturrate-Garcia et al., 2016). Still, many studies on Arctic biodiversity focus on vascular plant species, as the identification of lichen and moss species is tedious and requires specific expertise. The following sections describe the biodiversity of vascular plants, bryophytes, and lichens in the Asian Arctic tundra.

The majority of Arctic vascular plant species have a circumpolar distribution (CAFF, 2013). Approximately 1700 vascular plant species were identified for the entire Russian Arctic (Sekretareva, 1999). However, vascular plant species richness and the number of endemic species vary strongly across the floristic regions of the Arctic (Fig. 34.3A). Chukotka has the highest number of species (706 species and subspecies), largely due to the endemism of plants inhabiting the area and paleogeographic features. The Polar Ural—Novaya Zemlya region also has a relatively rich flora (551 species). On the other hand, flora of the West Siberian and Taymyr regions are the least diverse (407 and 445 species and subspecies, respectively). This is linked to their high latitudinal position and the historical formation of regional floras, resulting in the absence of many boreal species common further south. A characteristic feature of the Asian Arctic flora is its depletion as one moves northward. In the South, the northern forest limit serves as a robust ecological boundary, coinciding with

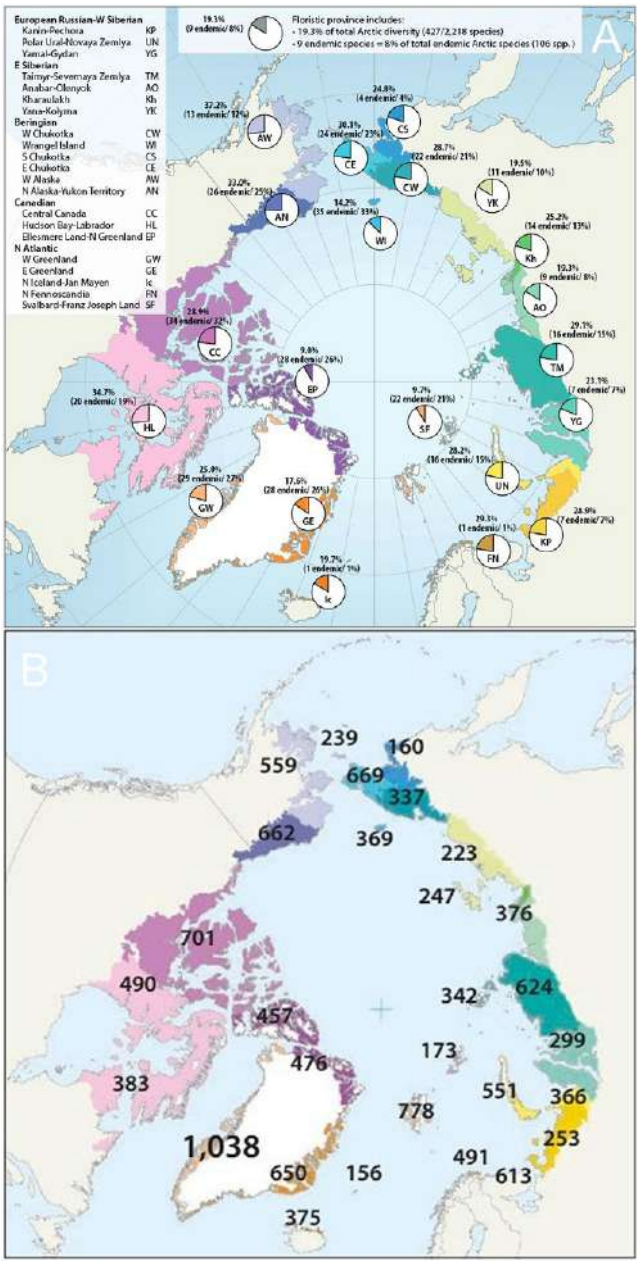

FIG. 34.3 (A) Map of species richness and endemism of vascular plants in the floristic provinces of the Arctic. Species richness of floristic provinces is expressed as a percentage of the total species richness of the Arctic (2218 species), and the endemism of floristic provinces is presented as rounded percentages of the total number of Arctic endemic species (106) (CAFF, 2013). (B) Species richness of lichens in the Arctic floristic provinces. Provinces are depicted in *different colors*. Lichen species richness: North America—1026, Greenland—1136, Europe—1075, and Asia—1178 (CAFF, 2013). Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: CAFF.)*

the isoline of richness at 500 species of vascular plants per 100,000 km^2 (Malyshev, 1975). In the tundra zone, species richness is three to four times lower compared to boreal floras (Matveyeva, 1998). The proportion of Arctic species increases northward, reaching its maximum value in the High Arctic (up to 90%). The leading vascular plant families in the Arctic include Asteraceae, Brassicaceae, Caryophyllaceae, Cyperaceae, Ericaceae, Fabaceae, Juncaceae, Poaceae, Liliaceae, Rosaceae, and Salicaceae (CAFF, 2013). These 10 families account for 60%–70% of the total number of species in different regions. Out of these 10, the Poaceae family is an absolute leader in the number of species in all regions of the Asian Arctic with Asteraceae and Cyperaceae also contributing strongly to vascular plant diversity (Sekretareva, 1999).

The morphological diversity of vascular plants in the tundra is worth noting (Fig. 34.4). Perennial herbs, which form winter buds for a rapid start of vegetation in the short growing season, prevail among life forms. Annuals are almost absent since, in the conditions of a short growing season, they cannot complete their life cycle from seed germination to seed formation. Shrubs and dwarf shrubs are mostly represented by prostrate forms, cushion plants, and other compact life forms that are widespread.

The distribution of the bryophyte flora in the Arctic is similar to that of vascular plants; about 80% of the species are circumpolar (CAFF, 2013). The most species-rich families include Bryaceae, Dicranaceae, Amblystegiaceae, Pottiaceae, Grimmiaceae, Sphagnaceae, Hypnaceae, Mniaceae, Brachytheciaceae, Polytrichaceae, and Splachnaceae, together constituting 70% of the total bryophyte flora. Among the most diverse genera are *Bryum*, *Sphagnum*, *Pohlia*, and *Dicranum*. The liverwort families Scapaniaceae, Jungermanniaceae, Gymnomitriaceae, and Cephaloziellaceae, as well as genera *Scapania* and *Lophozia*, are the most species-rich.

Most Arctic lichen species have a more limited distribution (CAFF, 2013; Fig. 34.3B). Approximately 25% of species occur in the entire Arctic, another 25% in three or four sectors, and about 49% in only one or two sectors. Almost 60% of lichen species occur in the Low and High Arctic. On average, lichen species richness decreases by 15% when moving from south to north, in contrast to vascular plant species richness, which decreases by 95%. According to Matveyeva et al. (2015), there are 321 species of ground lichens recorded in polar deserts, constituting 56% of the total number of species in this group in the entire Arctic (576 species). The leading genera in the ground lichen flora of polar deserts and tundra zone are similar. *Cladonia* leads with 43 species (13% of all species) in polar deserts and 80 species (14%) in the tundra zone, followed by the genus *Peltigera* with 17 (5%) and 37 species (6%) respectively.

Based on the structure and function of plant species, Arctic vegetation has been classified and mapped into 16 dominant zonal vegetation types representing their form and function (Raynolds et al., 2019; Fig. 34.5).

FIG. 34.4 Examples of different life forms of vascular plants in the tundra (A) *Polemonium acutiflorum*, a herbaceous vascular plant (Eastern Yamal). (B) Flowering Arctic bell-heather (*Cassiope tetragona*), an evergreen dwarf shrub (Eastern Yamal). (C) Dwarf birch (*Betula nana*), a deciduous dwarf shrub (Kytalyk National Park, Northeastern Siberia). *(Credit: Vitalii Zemlianskii, Gabriela Schaepman-Strub.)*

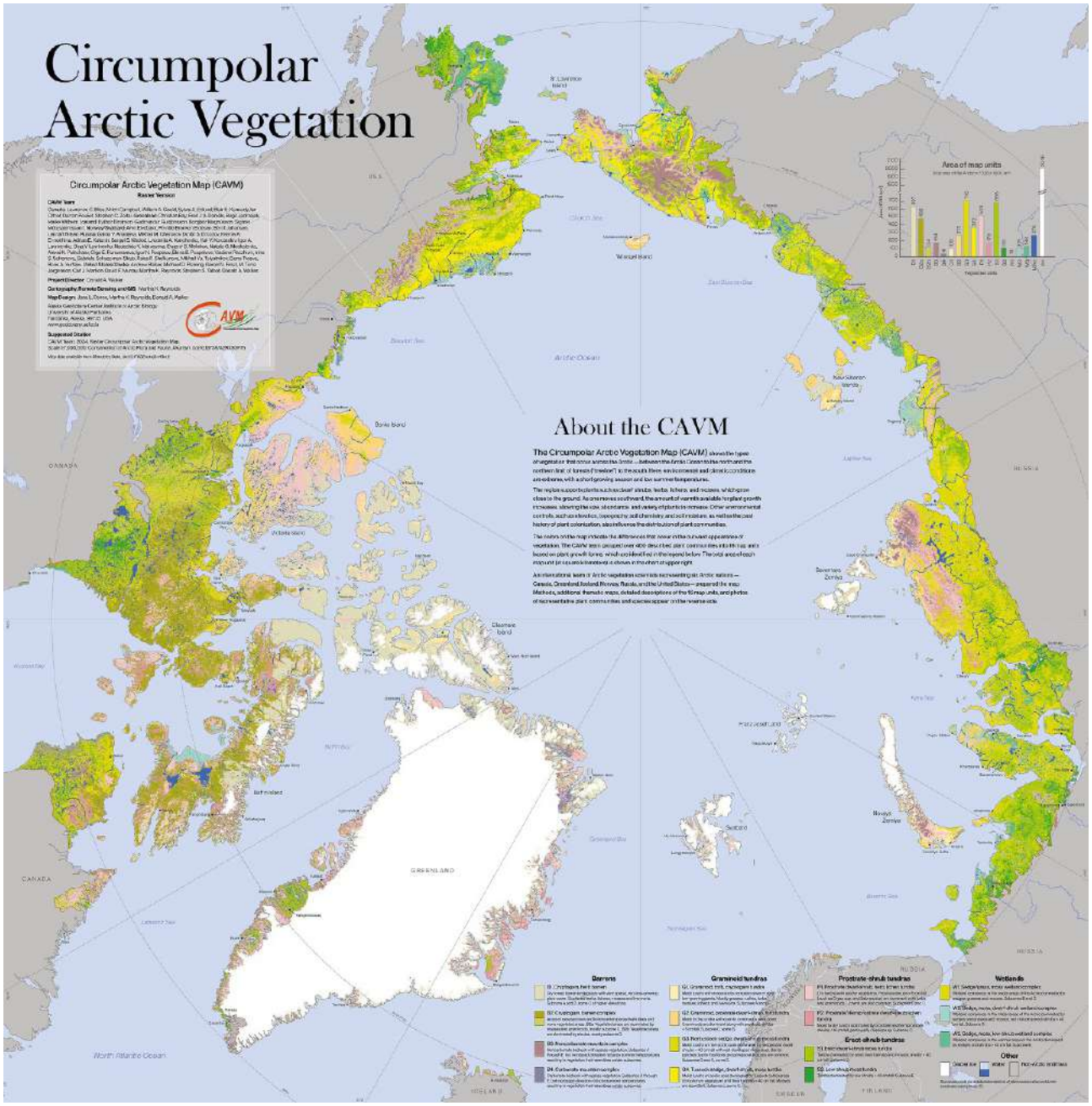

FIG. 34.5 The Circumpolar Arctic Vegetation Map (CAVM) displays the distribution of main vegetation types across the Arctic (CAVM Team, 2024). Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: CAVM team.)*

These vegetation types are named according to their general appearance, being barrens, graminoid tundras, shrub tundras, and wetlands, which are further split along a bioclimatic gradient by numbers (with four generally representing more southern types and one the most northern with lowest biomass) (Fig. 34.6).

3.2 Animals

Similar to plant diversity, also animal diversity is rather low across the animal kingdom in the Arctic. The diversity of arthropod species generally decreases with latitude (Gillespie et al., 2020). Arthropods have a key role in Arctic

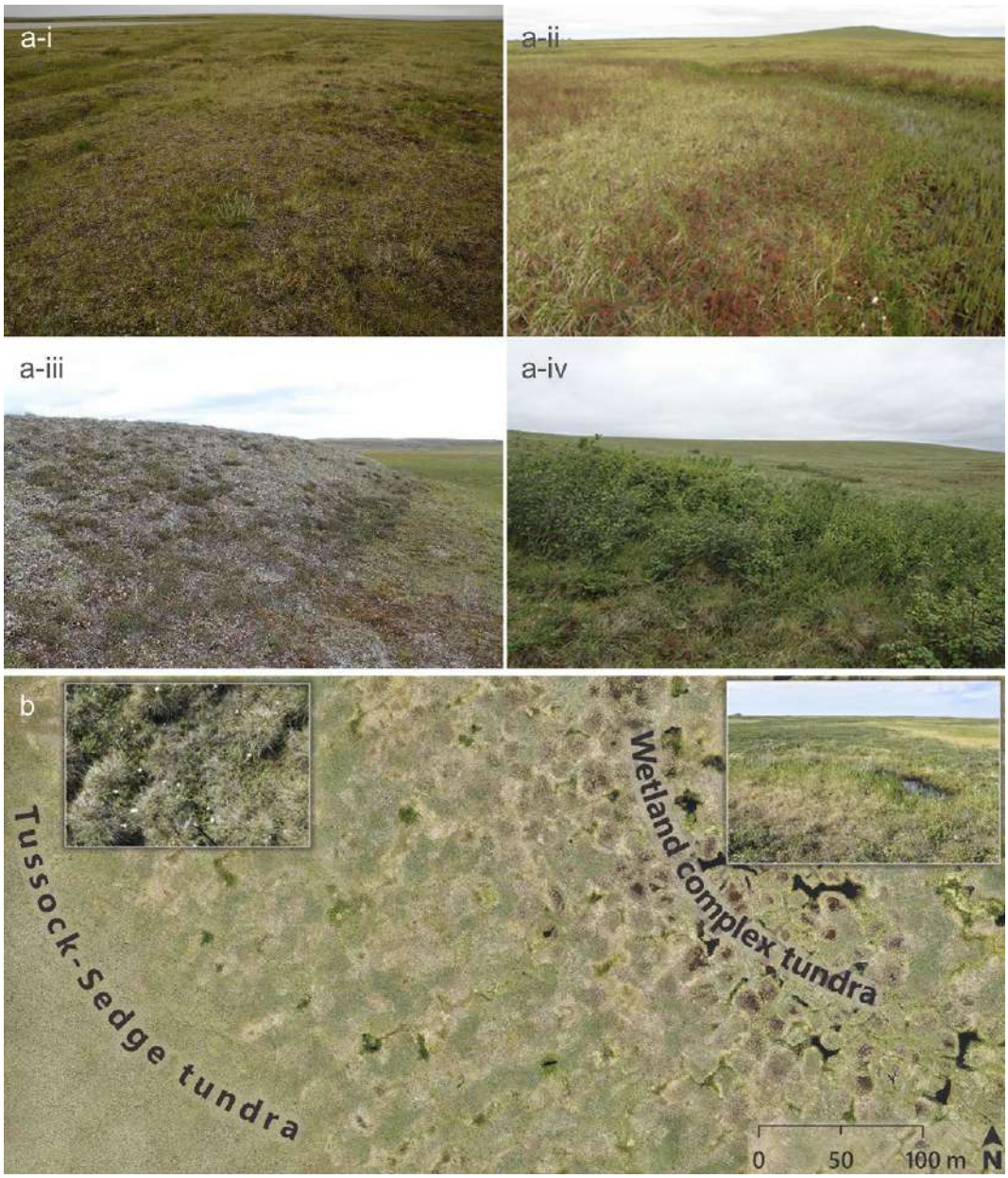

FIG. 34.6 (A) Selected typical tundra vegetation types: (i) Graminoid tundra. Nontussock sedge, dwarf-shrub, moss tundra (G3) (Yamal), (ii) Wetlands. Sedge, moss, dwarf-shrub wetland (W2) (Kytalyk, Sakha), (iii) Prostrate-shrub tundras. Prostrate/hemiprostrate dwarf-shrub tundra (P2) (Yamal), (iv) Erect-shrub tundras. Low-shrub tundra (S2) (Kytalyk, Sakha) (B) Vegetation types are distributed across the landscape at the Kytalyk/Chokurdakh research station in the Kytalyk National Park, Republic of Sakha, Russian Federation. On the Yedoma ridge in the west, tussock-sedge, dwarf-shrub, and moss tundra dominate. Soils of these ridges were formed mostly during the Pleistocene and contain major carbon stocks in the permafrost. In the drained thaw lake bed in the east, the wetland complex tundra is composed of a mosaic of high-centered polygons with drier communities, interspersed with wet-sedge areas and open ponds (sedge, moss, low-shrub wetland complex). The background image is acquired by a camera mounted on an unpiloted aerial vehicle. *(Credit: Vitalii Zemlianskii.)*

tundra ecosystems by connecting different trophic levels (CAFF, 2021). Climate change is strongly and rapidly impacting arthropods that are reacting with variable and contradicting responses (Høye, 2020). In combination with the highly heterogeneous life history in arthropods, it is challenging to predict their long-term responses to the many environmental changes coming with climate warming, such as increasing soil temperature, changes in moisture and surface water seasonality and extent (CAFF, 2021).

Fish species diversity decreases with latitude (Laske et al., 2022). Increasing temperature is predicted to strongly impact Arctic freshwater biodiversity as the organisms are adapted to and limited by the cold environment (Wrona et al., 2013). Research indicates that the productivity and colonization of Arctic freshwater fish will increase with climate change because of the phenotypic plasticity of fish and enhanced colonization of water bodies with a low number of competitors (Campana et al., 2020).

Amphibians and reptiles are rare in the tundra and can be found only at its southern limits, mostly in the forest tundra. Siberian newt (*Salamandrella keyserlingii*), moor frog (*Rana arvalis*), Siberian common frog (*Rana amurensis*), and common frog (*Rana temporaria*) are amphibian species found in the Asian tundra, with most of them reaching the tundra zone only in western Siberia (CAFF, 2013; Arefev et al., 2010). The only reptile species present in the region is the common lizard (*Zootoca vivipara*) with few documented occurrences in the Polar Urals at the northeastern limit of its distribution (Balandin et al., 1997).

There are more than 200 bird species that breed regularly in the Arctic. Most of them are considered as marine birds, while 88 are counted as terrestrial bird species. There are only a few terrestrial bird species like the rock ptarmigan (*Lagopus muta*), the Arctic redpoll (*Acanthis hornemanni*), or the snowy owl (*Bubo scandiacus*) that remain in the Arctic throughout the entire year (CAFF, 2021). Most of the terrestrial bird species that breed in the Arctic migrate to warmer regions during winter. Therefore, the population patterns and trends of most Arctic bird species are not only influenced by changes in the Arctic, but also by changes along their flyways (Lisovski et al., 2021).

Predictions of bird diversity under climate change are complicated, as many species show different trends for different populations and flyways, while unknown for some populations (CAFF, 2021). A metaanalysis on 88 terrestrial bird species in the Arctic showed that more than half of the circumpolar wader taxa shows a declining trend, while nearly half of the waterfowl taxa are increasing (Smith et al., 2020).

Models are suggesting that much of the breeding habitat for migratory birds in the Arctic will be lost because of climate change (Wauchope et al., 2017). Also, extreme events such as droughts and flooding might impact the success of birds breeding in the Arctic, as highlighted in the example of the decrease of nesting habitat of the threatened Siberian crane under flooding (Haverkamp et al., 2022). In addition, anthropogenic factors like industrial activity and harvest also impact the demography of Arctic bird populations (Doyle et al., 2020). Changes in disturbance regimes like increasing fire activity are also affecting the diversity of bird species on a local and landscape scale (Pérez et al., 2018; Heim et al., 2022).

Mammals in the Asian Arctic tundra include large terrestrial animals such as reindeer (*Rangifer tarandus*) and muskoxen (*Ovibos moschatus*). Besides wild reindeer populations, there are reindeers managed as free-ranging livestock. Some reindeer populations are prone to climate and land use change effects, which might be reflected in the declining trends of some populations (CAFF, 2021). The muskox populations generally exhibit a low genetic diversity, which can be related to translocated populations, which were mainly founded from one source that already lost most of its diversity. While there is no clear trend in muskox populations in the American Arctic, in Asia, where muskox have been reintroduced since the 1970s, most populations are increasing in numbers, although the status of some populations is unknown (Cuyler et al., 2020). Muskoxen have survived climate shifts, but because of their low genetic diversity, they are probably vulnerable to infectious diseases and future bottlenecks (CAFF, 2021). Other mammals that are abundant in the terrestrial Arctic are hares and ground squirrels as well as lemmings and voles. Brown lemmings (*Lemmus sibiricus*, *Lemmus paulus*) and collared lemmings (*Dicrostonyx* spp.) are widespread throughout the Arctic and are key components of the food web (CAFF, 2021). Lemming cycles generally have a large amplitude, which makes long-term trend analysis difficult. Ehrich et al. (2020) found that there is no general negative trend to be detected, but an indication of a negative trend for the Russian Arctic. In addition, they showed that low Arctic lemming populations that are sympatric with voles are decreasing (Ehrich et al., 2020). Small mammals that are predators in the terrestrial Arctic are small mustelids and shrews. Other predators are wolverines (*Gulo gulo*), Arctic foxes (*Vulpes lagopus*) as well as wolves (*Canis lupus*) and bears (CAFF, 2021) (Fig. 34.7).

FIG. 34.7 Birds breeding in the Arctic tundra include resident species, such as (A) the snowy owl (*Bubo scandiacus*), and (B) willow ptarmigan (*Lagopus lagopus*), in the photo a male changing from white winter to brownish summer plumage (Kytalyk, early July 2017) and migratory species such as (C) the rough-legged hawk (*Buteo lagopus*, Kytalyk, July 2013), in the photo nestlings covered by most the abundant animal species in the tundra, the mosquito. Only a few mammal species live in the Arctic tundra, such as (D) the polar fox (*Vulpes lagopus*), and (E) muskox (*Ovibos moschatus*; Kytalyk, July 2013) reintroduced to the Russian Arctic. *(Credit: Vitalii Zemlianskii, Gabriela Schaepman-Strub, Michael Schaepman.)*

4 Climate change impacts on tundra biodiversity and ecosystem functioning

Climate change is manifesting in the Arctic through increases in temperature, especially during winter, changes in precipitation, and a lengthening of the growing season through earlier snowmelt in spring. These climatological changes lead to not only slowly changing environmental conditions for plants, animals, and humans living in the Arctic, but also to abrupt large-scale ecosystem disturbances through extreme events, such as rain on snow, flooding, summer droughts, and wildfires. Predicting changes in Arctic biodiversity and ecosystem functions is highly complex, due to interactions between climate,

permafrost, soil nutrient cycling, and biotic interactions that might lead to nonlinear effects and potential tipping points in the system. Also, many processes have high prediction uncertainties such as the impact of thawing permafrost on hydrology (Walvoord and Kurylyk, 2016), and land surface components in global circulation models poorly represent the functions of tundra plants (Thomas et al., 2019). However, many observational and experimental studies provide evidence of climate change impacts on tundra biodiversity and ecosystem functioning. Further, Indigenous Peoples report major ecosystem changes in the Arctic, many impacting their livelihoods (Ksenofontov et al., 2019). In the following, an overview is given on some main effects of climate change on Asian Arctic biodiversity and ecosystem functions as reported from observational and experimental studies.

4.1 Contemporary dynamics of vegetation related to climate change

Climate change can impact biodiversity at very different levels, including genetics, physiology, phenology, distribution, interspecific relationships, ecosystem functions and biome integrity, affecting individual organisms, populations, species, communities, ecosystems, and biomes (Bellard et al., 2012). A main problem in the tundra is the mostly missing standardized, repeated monitoring of biodiversity at time-scales relevant to climate change. Therefore, climate change effects are often tested in experimental setups mimicking selected future change scenarios. Also, space for time approaches are sometimes applied or climate sensitivity analyses performed.

With climate warming, plants are released for increasing time periods from the main growth limiting factor in the Arctic tundra, low temperature. Therefore, the tundra vegetation transits in a general trend toward more dense and taller vegetation often related to the mechanisms of treeline expansion from the south (MacDonald et al., 2007), shrubification (Myers-Smith et al., 2011), or a shift toward more acquisitive functional traits within plant communities, especially under wet conditions (Bjorkman et al., 2018).

Most rivers in the Asian Arctic flow from south to north, draining into the Arctic Ocean, bringing plants from more southern regions. Some of these natural migrants adapt within zonal communities, while others grow in intrazonal habitats. In many areas of the Asian Arctic, a shift in the northern forest boundary is observed. Forest vegetation actively develops at the northern limit of its distribution: in Western Siberia and on the Taimyr Peninsula, seed regeneration and growth of larch seedlings have been noted (Pospelov and Pospelova, 2001; Kirdyanov et al., 2012).

The phenomenon of the "greening of the Arctic" is among the world's most reported large-scale ecological responses to climate change. However, the underlying causes and future dynamics of the Arctic greening and browning trends are complex, spatially variable, and scale-dependent, and in many cases difficult to relate to in situ observational evidence (Myers-Smith et al., 2020). Studies have shown the sensitivity of shrub growth to various temperature indicators, but these relationships are species-dependent and for some species and locations they are inverted (Myers-Smith et al., 2015). Shrubification has mostly been documented in areas with rather moist soils, whereas in dry areas an increase in air temperature might lead to a decrease of shrub growth due to moisture limitation (Myers-Smith et al., 2015). From 1982 to 2012, vegetation productivity trends recorded from satellites were positive and statistically significant across the entire area of the Asian Arctic, except for small areas (Zhiltcova and Anisimov, 2015). Recent research indicates the persistence of this trend (Raynolds et al., 2019). High correlations between vegetation productivity and temperature during the growing season are noted, along with mostly negative correlations with precipitation amounts (Anisimov et al., 2015). In the Low Arctic, vegetation dynamics are influenced by the increased role of shrubs in various types of plant communities, leading to an expansion of areas occupied by productive communities such as willow, alder, and dwarf-pine stands, and meadows (Anisimov et al., 2015; Bonfils et al., 2012; Frost et al., 2013, 2018; Loranty and Goetz, 2012; Myers-Smith et al., 2015). In the High Arctic, the increase in photosynthetic activity is less noticeable (Zhiltcova and Anisimov, 2015).

The shrubification of tundra ecosystems (increase of biomass, coverage, and abundance of shrubs in new territories and within shrub communities) contributes to snow accumulation, higher winter temperatures, microbial soil activity, and, consequently, nutrient accumulation in the soil, stimulating further shrub growth (Dvornikov et al., 2015; Frost et al., 2018). In some cases, the shrubification in tundra ecosystems may have not only direct cause-and-effect connections with ongoing changes in the Arctic climate (temperature and precipitation dynamics) but also indirect links through the increase in the area affected by cryogenic processes (Frost et al., 2013; Lantz et al., 2009, 2010; Racine et al., 2004; Tishkov et al., 2018; Walker et al., 2009). The colonization of disturbed areas by shrub vegetation (including nonclimate-related) initiates cascading feedbacks (through snow retention, accumulation of litter, etc.), affecting the temperature and moisture regimes of permafrost and soil, as well as the carbon budget of ecosystems (Blok et al., 2010; Buckeridge et al., 2010; Kaarlejärvi et al., 2012; Sturm et al., 2005; Tishkov et al., 2018).

While satellite image analyses suggest an increase in productivity in many regions of the Arctic tundra, different plant-community-related mechanisms might be responsible for this trend. For example, a review of change in composition of vegetation based on long-term plot data across the Arctic (however with only a single data point in the Asian Arctic) has reported the majority of plots without significant trend in compositional change (Bjorkman et al., 2020). Forbs, graminoids, and shrubs were slightly more likely to increase their abundance than decrease in the long-term monitoring data. However, experimental warming led to a higher likelihood of lichens and mosses to decrease, while shrubs more likely increased their abundance (Bjorkman et al., 2020). For the Asian Arctic, trends in decreasing moss and lichen cover were directly related to tundra trampling (Tishkov et al., 2018). In some regions of the Asian Arctic, the shift in the role of life forms is mainly driven by fires (Chukotka), excessive reindeer grazing (Yamal and Gydan), shrubification (Polar Urals), but the primary causes are usually associated with changes in geocryological conditions (Frost et al., 2013, 2018; Tishkov et al., 2018). The expansion of areas affected by cryogenic processes, including anthropogenically triggered processes, also leads to the spread of herbaceous vegetation. Habitats with excess moisture and dominance of herbaceous plants are formed as a result of thermokarst processes, frost heave, development of cryogenic landslides, and increased depth of seasonal thawing (Fig. 34.8).

Climate warming is expressed as a steady, but rather slow increase in temperature over long time spans. However, with climate warming also extreme conditions are increasing such as extreme precipitation leading to flooding, or droughts enhancing the risk for wildfires (Fig. 34.9). The trend toward taller and denser vegetation triggered by climate warming might be further enhanced through such extremes and their consequences. Shrub cover

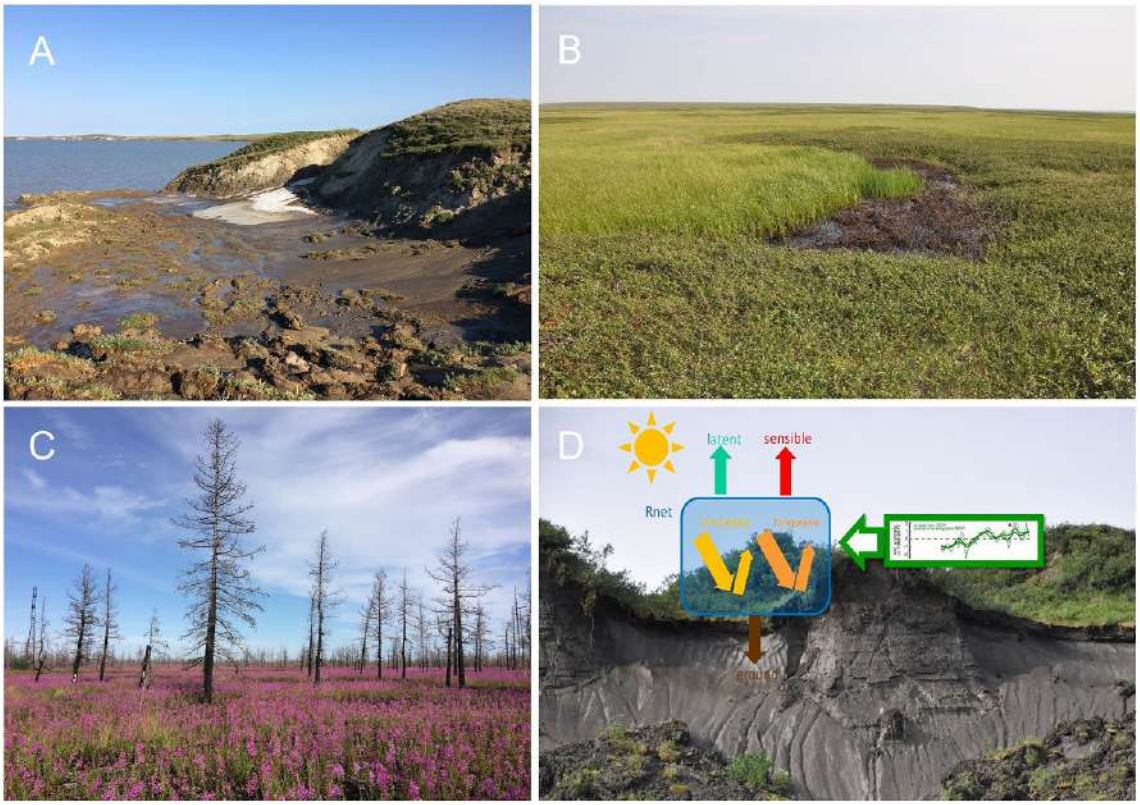

FIG. 34.8 Disturbances in the Arctic tundra lead to vegetation change. (A) Overgrowth of cryogenic landslides by herbaceous species (Gydan, 2017). (B) Degradation of ice-rich permafrost results in vegetation change, visible here by the drowning of dwarf shrubs by subsidence and subsequent replacement by wet sedges (Kytalyk National Park, 2010). (C) Fires can lead to major vegetation changes not only in the short-term, but also over decades. Regrowth of fire weed after a fire has burned at the tree line in the Nadym region. (D) Vegetation has an important function in the Arctic system. Vegetation is a very thin green layer growing on top of permafrost in the Arctic tundra. Vegetation type is affecting the amount of absorbed solar radiation (R_{net}) at the tundra surface, but even more the partitioning of this absorbed energy into evapotranspiration (through latent heat flux), warming of the atmosphere (through sensible heat flux), and warming of the permafrost (through ground heat flux) during summer time. Changing vegetation type is therefore expected to change also this ecosystem function of energy absorption and partitioning in the tundra landscape, and thus affect permafrost degradation and atmospheric warming. For more details see Oehri et al. (2022). *(Credit: Ksenia Ermokhina, Gabriela Schaepman-Strub.)*

FIG. 34.9 Tundra landscapes are expected to be increasingly exposed to extreme conditions under future climate change. (A) The area around Chokurdakh/Kytalyk research station has been exposed to an extreme flooding in 2017, while it experienced extreme droughts and wildfires in 2019 and 2020. (B) The second photo has been taken close to the spot of the 2017 photo and shows the severe drought impact on the vegetation, and skies covered by smoke of large wildfires in 2020. (C) Wildfires burned the East Siberian tundra in many places, as seen here close to Chokurdakh in 2019. *(Credit: Gabriela Schaepman-Strub, Vitalii Zemlianskii.)*

was, for example, shown to be higher in old fire scars compared to unburnt vegetation (Heim et al., 2019). Vegetation modulates the thermal conditions in the soil, and thus also permafrost thaw (Blok et al., 2011; Lawrence and Swenson, 2011; Oehri et al., 2022). If vegetation and permafrost are not able to recover because of a warmer climate and increased disturbance impacts, the changes on the tundra ecosystem may be irreversible (Heijmans et al., 2022). Such changes are of local relevance as thawing permafrost affects the ecosystem and its functioning (Olefeldt et al., 2021) and potentially human and animal health through new outbreaks of zoonotic diseases (Walsh et al., 2018). As there is a very large amount of carbon stored in the Asian Arctic permafrost soils, permafrost thaw is predicted to have global impacts through greenhouse gas emissions (Schuur et al., 2022).

5 Human activities and their impacts

Until recently, the Asian Arctic tundra was an area of relatively low human activity, with traditional forms of land use including reindeer herding, hunting, fishing, and gathering. However, some regions of the Asian Arctic, such as the Yamal-Nenets Autonomous District, have experienced increasing pressure in recent decades. While reindeer herding has intensified, oil and gas extraction have strongly increased. Further, extensive plans for mineral extraction have been published and the planned infrastructure for the operation of the northern sea route will bring additional pressures, especially along the coasts. In addition to climate change, these human activities (Fig. 34.10) affect the permafrost, composition, and functioning of biological plant and animal populations, and local livelihoods through land use change, invasive species, pollution by chemicals, and light. Further, wildfire ignition points and road infrastructure were spatially highly related in the East Siberian boreal forest (Kuklina et al., 2023), an effect also expected in the tundra if road infrastructure is expanding. Finally, local people have reported increased pressure on vegetation and animal populations by technological adaptations (Ksenofontov et al., 2019). For example, snowmobiles in winter time allow for easier movement, reaching wild animals anywhere in the tundra or for recreational activities. Also, motorboats, all-terrain vehicles, tractors, and cars are now employed not only for hunting and fishing, but also recreational activities. Local people have related the decrease of reindeer, birds, and fish to these developments that support unsustainable harvesting (Ksenofontov et al., 2019). On the other hand, some pressures have decreased, for example, the pressure on the Arctic fox due to the decrease of fur trade, or the pressure on trees in the forest tundra due to the reduced use of firewood (Ksenofontov et al., 2019). The following section provides an overview on human activities in the Asian Arctic tundra, starting with the most traditional form of land use, namely reindeer herding, followed by oil, gas, and mineral extraction, and concluding with some emerging human activities and their impact in these remote regions.

5.1 Reindeer husbandry

Reindeer husbandry is a traditional land use practices by Indigenous Peoples of the Asian Arctic. The most widespread practice is herd-based reindeer husbandry, found in Western Siberia, Taymyr, Yakutia, and Far East. Reindeer are kept enclosures in Evenkia and Yakutia, allowing the use of forage in small and hard-to-reach pasture areas not used in herd-based husbandry (Ivanov, 2014). In many areas of the Asian Arctic, including Western Siberia and Yakutia, the population of domestic reindeer is limited by the availability of winter lichen pastures. This aligns with the general consensus among researchers that the quality of winter pastures is usually the decisive factor determining the population size of northern (wild and domestic) reindeer (Bernes et al., 2015). In recent decades, the size of herds in the West Siberian Arctic has significantly increased and has reached or is nearing historical maximum, while in other regions of Russia, there is a

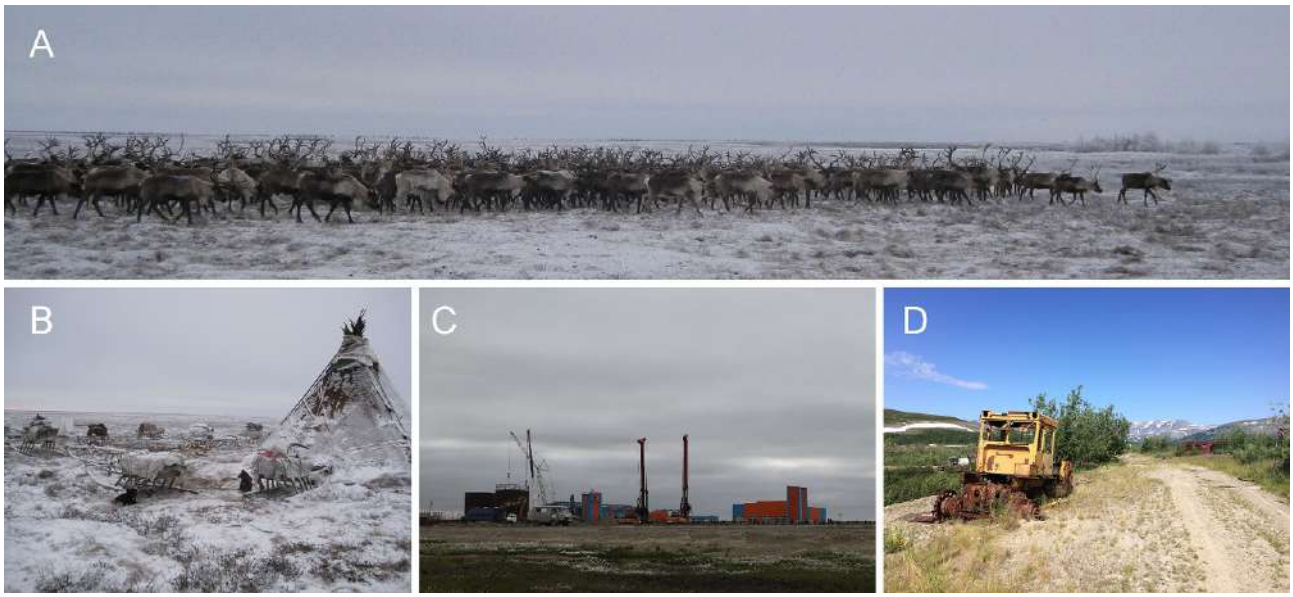

FIG. 34.10 Land use in the Arctic includes traditional practices, such as reindeer herding and industrial activities. (A) Reindeer, (B) traditional huts, called "chum" of Yamal-Nenets, (C) gas infrastructure on Eastern Yamal, and (D) infrastructure ruins, Asian part of Polar Urals, 2018. *(Credit: Ramona Heim, Vitalii Zemlianskii, Ksenia Ermokhina.)*

decline in the number of domestic reindeer (CAFF, 2013; Uboni et al., 2016). At the same time, the area of reindeer pastures is being reduced by the expansion of oil and gas infrastructure (Volkovitskiy and Terekhina, 2023). Specialized studies on the state of pasture vegetation in the Asian Arctic over the past decades have only been conducted in Western Siberia. The results show the exclusion of many forage species from plant communities, a decrease in forage reserves, and often zero reindeer-carrying capacity in tundra and forest-tundra pastures (Ermokhina and Morozova, 2017). Excessive reindeer grazing is considered as the main factor by many researchers, damaging up to 80% of pasture areas (Andreev, 1983; Golovatin et al., 2012). The impact of grazing can induce vegetation succession in tundra ecosystems, typically resulting in a transition from lichen-dominated to moss- or sedge-dominated plant communities. Lichens exhibit a significant overall negative response, also called delichenization (Golovatin et al., 2012; Bernes et al., 2015), while mosses showed no significant reaction to grazing. Vascular plant species richness, in general, shows a significant negative response to grazing, but no correlation has been identified with the density of reindeer herds (Bernes et al., 2015). Also, there is indication of reindeer grazing hindering climate-driven tree and shrub encroachment into the tundra (Bernes et al., 2015). The sharp increase in the reindeer population in the 20th century led to significant vegetation degradation (Isachenko, 1985; Mukhachev, 2014; Podkorytov, 2005). Uneven use of pastures by reindeer herders, large herd sizes, and the lack of a pasture rotation system are the main factors leading to vegetation degradation. The most substantial impact of grazing is observed on watershed surfaces and slopes, frequently resulting in the complete destruction of lichen cover. For example, overgrazing in Yamal has led to the desertification of tundra ecosystems (Golovatin et al., 2012), manifesting in the replacement of typical tundra communities with psammophytes, plants that grow in sandy and often unstable soils. The destruction of vegetation cover contributes to wind erosion and further habitat degradation, leading to the formation of sand drifts and dunes, practically devoid of vegetation and often cover extensive areas (Golovatin et al., 2012).

The impact of reindeer on pasture vegetation is determined by their numbers, duration of presence, season, weather conditions, and the nature of pasture use, including consumption and/or trampling of various plant parts, leading to the death of individual organ or entire plants (Karpov, 1991). However, interactions in the reindeer-pasture system are rarely quantitatively assessed, and there is a lack of data on changes in forage plant stocks and community productivity under grazing, making it challenging to make predictive assessments of the dynamics of reindeer pasture vegetation.

5.2 Extraction of oil, gas, and minerals

Oil, gas, and mineral exploitation have major ecosystem and societal impacts in the tundra through establishment of access roads, new infrastructure built up, but also environmental impacts and risks occurring during or after operation, through chemical, light, and noise pollution.

Oil and gas exploitation infrastructure consumes much larger areas compared to mining infrastructure that is more stationary over time (Akandil et al., 2024). However, pollution from mining can be very diverse and locally and regionally intense. For example, in the Norilsk area, mining for copper and nickel started in the early 1940s and has led to heavy metal contamination, and increased copper and nickel concentrations were found in the soils up to 60 km from the city (Blais et al., 1999). Vegetation has degraded around Norilsk not only through the long-term impacts of chemical pollution of the atmosphere and water, but also through heat emission by buildings and industrial activity resulting in permafrost degradation (Tutubalina and Rees, 2001). Further, Norilsk experienced a large oil spill in 2020, with up to 20,000 tons of diesel fuel polluting the landscape and rivers (Tysiąc et al., 2022). This event led to a transformation of the vegetation of the polluted river floodplains, decreasing the diversity of the plant communities and especially affecting bryophytes and some vascular species that are extremely nonresistant to pollution (Telyatnikov, 2022).

In addition to direct land use change and pollution impacts by the extractive industry, seismic surveys for the exploration of oil and gas resources can also have strong and spatially extensive impacts that affect more area than the extractive infrastructure directly. Heavy vehicles used for mobile camps during seismic surveys caused long-lasting damage, especially in moist vegetation types, leading to degradation of the ice-rich permafrost in the Arctic National Wildlife Refuge in Alaska (Raynolds et al., 2020). In the Russian Arctic tundra, seismic exploration on land seems to still be limited mostly to the vicinities of existing oil and gas fields detected during Soviet times. While a new seismic survey system was tested in Russia in 2020, unpiloted vehicle and plane-based methods are being developed for initial stages of geological exploration, which would reduce the direct impact on the tundra (Investment Portal of the Arctic Zone of the Russian Federation, 2024).

5.3 Emerging human activities

In addition to oil and gas extraction and mining, several other human activities are emerging in the Arctic and might impact the vulnerable tundra ecosystem. These include tourism, development of the Northern Sea Route and military infrastructure in the Russian Arctic, discussed in the following sections.

Currently, the impact of tourism on the Asian Arctic tundra is rather limited due to low transport availability (Kuchumov et al., 2023; Tsvetkov, 2023). Arctic tourism in Russia remains a niche and is expensive with few people taking Arctic cruises, especially after the decline of international tourism since 2022 (Timoshenko, 2020; Tsvetkov, 2023). The existing tourist pressure is unevenly distributed. The most visited Asian Arctic region by tourists is the Yamalo-Nenets Autonomous District as roads and railways connect the region with the rest of Russia, which is not the case in other Asian Arctic tundra areas. However, even there the pressure is very low compared to other Arctic countries or the Murmansk region in the European part of Russia (Golubchikov et al., 2019). This may change in the future with better transport availability related to continued development of infrastructure including the port infrastructure along the Northern Sea Route and reorientation of tourist flows in Russia toward the domestic market (Timoshenko, 2020; Tsvetkov, 2023).

The development of the Northern Sea Route is considered to be an issue of highest importance by the Russian state (Erokhin et al., 2022; Security Council of Russia, 2020). While the first successful navigation dates back to 1914–15 and the route was used for the delivery of Lend-Lease supplies from the United States during World War II, its use has been limited to date due to stable summer sea ice north of the Taymyr Peninsula (Armstrong, 2011). However, the geopolitical importance of the route led the Soviet government to develop infrastructure along the route, which was largely abandoned in the 1990s, but has been partly renovated in recent times (Antrim, 2010; Erokhin et al., 2022). In the Asian sector of the tundra, the port infrastructure of the Northern Sea Route includes the Soviet-built ports of Dudinka (Yenisei river), Dikson (Taymyr Peninsula), Tiksi (Sakha Republic), Pevek (Chukotka) as well as the newly built port of Sabetta, which is used for the export of liquefied natural gas.

In the last decade, the use of the Northern Sea Route has increased, largely surpassing Soviet-era records (Stepanov, 2020). Between 2014 and 2019, the amount of goods transported via the Northern Sea Route rose from 4 to 31.5 million tons (Security Council of Russia, 2020). Intensification of the Northern Sea Route use leads to serious environmental consequences. While the direct impact of ship noise and pollution would primarily affect the marine life of the Arctic Ocean, with potentially devastating consequences for populations of narwhals and other marine mammals (Hauser et al., 2018), the increased accessibility of the Arctic coast would encourage new resource extraction activities, such as export-oriented coal mining, which is already taking place on the Taymyr Peninsula (Li et al., 2021; Peter, 2019). As sovereignty over the Northern Sea Route is a major geopolitical concern for the state, an increased military presence is also expected (Security Council of Russia, 2020; Staun, 2023).

During Soviet times, the Asian Arctic tundra has been covered by a network of USSR military bases. Many military bases were abandoned following Perestroika and the collapse of the Soviet Union. However, the last decade has seen a gradual increase in military activity, with old

bases being reopened and renovated, and new bases being built across the Russian Arctic (Borozna, 2024); Kjellén, 2022). While the current military presence is already at a high level (according to Reuters, Russia maintains at least 10 bases in the Asian Arctic (Gronholt-Pedersen et al., 2022)), the pressure is expected to continue to increase if the geopolitical tensions persists and the use of the North Sea Route intensifies (Sergunin and Gjørv, 2020; Staun, 2023). This would inevitably lead to a number of negative environmental impacts, such as increased carbon emissions, chemical and noise pollution, radioactive contamination, and habitat destruction documented in other parts of the Arctic (Antipov and Tananaev, 2023; Gunnarsdottir, 2024; Kryshev et al., 2022), even affecting the strictly protected areas (UNESCO World Heritage Committee, 2018).

Development of military activities also lead to poorer accessibility for researchers to important parts of the tundra zone. For example, the entire sea coastal area of the Russian Arctic is subjected to a special permission regime as part of a border zone. Apart from the direct impact of military activities in the Arctic, recent geopolitical developments have also contributed to the deterioration of the dialog among stakeholders and rights holders in the Russian Arctic tundra as well as the scientific research, with environmental Nongovernmental organizations, Russian independent researchers, and Indigenous Peoples among the main losers (Borozna, 2024; Tysiachniouk et al., 2023; Zmyvalova, 2023).

Apart from the direct impact of industrial and military infrastructure and pollution on the Arctic tundra ecosystem, there is a strong socio-ecological impact as exemplified in a study on Yamal peninsula with its widespread development of gas and oil infrastructure (Kumpula et al., 2011). A large influx of nonindigenous workers competes for freshwater fish, and off-road vehicle traffic disturbs reindeer pastures, which regenerate with fewer species, thus reducing biodiversity. Biophysical and social impacts could be reduced if information flow between herders and workers was optimized (Kumpula et al., 2011). Factors contributing to social-ecological system resilience in the Nenets and Yamal-Nenets Autonomous regions included environmental factors (for example, ample space and abundance of resources to augment the diet of migratory herds and residents), accepting attitudes toward climate change and industrial development, and consciousness of Indigenous Peoples of their role as responsible stewards of the territories, and governance regimes by herders' own agency, with a shift to smaller, privately managed herds (Forbes, 2013).

5.4 Invasion of species

The introduction of alien species within the Asian Arctic is widespread (Wasowicz et al., 2020), but relatively poorly studied. While there is more comprehensive information on alien species in the northern part of Western Siberia, the nonnative flora of the Eastern and Far Eastern Russian Arctic remains poorly studied. The spread of most alien plants in the Asian Arctic is confined to settlements, transportation routes, and associated infrastructure. The first large settlements emerged in the Asian Arctic in the late 16th century, primarily engaged in forestry, reindeer husbandry, and fishing. The main transportation routes were rivers, limiting the introduction and naturalization of new species. In recent decades, increasing transport connections facilitated the arrival and establishment of alien species.

The low population density of the Asian Arctic, small transport volumes (except for major industrial centers), and harsh climatic conditions constrain the invasion of alien species into ecosystems. On the other hand, the indigenous flora of the Asian Arctic has few species capable of supporting early successional stages, and the restoration of vegetation on disturbed sites is slow, allowing invasive species to successfully establish in anthropogenically altered habitats.

The composition of the invasive flora in settlements of the Asian Arctic largely depends on the environmental conditions, the duration and history of the settlement, specialization, size, and the volume of imported goods (Dorogostaiskaya, 1970; Ishbirdin et al., 1996). Fifty to sixty years after the start of economic development, the proportion of nonnative species in the local flora typically reaches 30%–40%, and in long-settled areas, it can reach 60% (Tishkov, 1996). The main introduction of species seems to occur in the first decade of development, with the influx of introduced species balancing out over time due to the decline of previously introduced species that failed to adapt.

For many invasive species, naturalization occurs relatively easily. For instance, wild chamomile (*Matricaria discoidea*) has widely spread in the Asian Arctic. Many invasive species transition from settlements to seminatural habitats such as meadows, pastures, fallow lands around settlements, abandoned fields and quarries or to natural habitats (Dorogostaiskaya, 1970).

Studies on nonnative flora in the northern part of Western Siberia are more detailed compared to other regions of the Asian Arctic. For instance, Ishbirdin et al. (1996) traced changes in flora and vegetation along the gradient city-settlement-temporary settlement, confirming that the proportion of nonnative species is influenced by the geographical location, duration of settlement existence, settlement size, and characteristics of anthropogenic impacts (presence of cultivated areas, lawn care, areas prone to frequent mechanical disturbances, and building characteristics). A study of the flora around the city of Novy Urengoy, one of the most transformed regions in the Asian Arctic due to oil and gas extraction, revealed an extremely poor flora with only 153 species of vascular plants, 25% of

which are introduced species (Vilchek and Kuznetsov, 1996). Pismarkina (2014; Pismarkina et al., 2019) identified more than 80 alien species in the flora of the southern tundra, forest-tundra, and northern taiga. All findings of nonnative species were within disturbed habitats (roadside verges, riverbanks, weed patches, railway stations, airports, etc.), indicating that alien plants practically did not infiltrate tundra communities (Pismarkina and Bystrushkin, 2020).

There is limited attention given to the study of nonnative flora in the East Siberian part of the Arctic, primarily considering the invasion of alien species within the framework of studying anthropogenic landscapes. For example, approximately 30 alien species (mostly weeds) were discovered in the diamond deposit area of northern Yakutia after 10 years of development in the mid-1960s (Tishkov, 1996).

In the Far Eastern Arctic, the proportion of nonnative species to the total number of species in the regional flora was reported to increase from north to south, except for the Anyuisky region (Kozhevnikov and Kozhevnikova, 2011). In the study of the flora in disturbed habitats in the lower reaches of the Kolyma River (Koroleva and Petrovsky, 1996), only five nonnative species were noted. Overall, Chukotka's flora exhibit a low contribution by nonnative species due to the sparsely populated territory and the absence of major industrial and agricultural centers.

6 Conservation areas and their protection level and potential nature-based solutions

Protected areas in the Arctic contributed 20.77% to the total terrestrial area as of 2021 (CAFF/PAME, 2022), at various levels of protection (Fig. 34.11). However, their distribution regarding future climate conditions is not optimal. The majority of protected areas is located in southern regions that are predicted to have a climate suitable for more southern vegetation types by 2050 already, while threatened vegetation types will be strongly underrepresented in current protected areas (Reji Chacko et al., 2023).

Arctic ecosystems change due to the impacts of global drivers, primarily climate change. Further, Indigenous Peoples rely on natural resources spanning across national borders. Therefore, Arctic terrestrial conservation requires an international strategy that considers the pan-Arctic region as a whole. The Arctic Council's Conservation of Flora and Fauna (CAFF) group is developing strategies "intended to directly conserve species and ecosystems." In the framework of the Arctic Biodiversity Assessment, the Arctic Council has released key messages and recommendations for conservation relating to terrestrial ecosystems, still valid today (CAFF, 2013):

1. Advance the protection of large areas of ecologically important marine, terrestrial, and freshwater habitats, taking into account ecological resilience in a changing climate.
2. Develop guidelines and implement appropriate spatial and temporal measures where necessary to reduce human disturbance to areas critical for sensitive life stages of Arctic species that are outside protected areas, for example, along transportation corridors. Such areas include calving grounds, den sites, feeding grounds, migration routes, and mounting areas. This also means safeguarding important habitats such as wetlands and polynyas.
3. Develop and implement mechanisms that best safeguard Arctic biodiversity under changing environmental conditions, such as loss of sea ice, glaciers, and permafrost.

CAFF is acting through monitoring, assessment and communication with policy makers to conserve biodiversity. Although CAFF's recent activities focused on the Russian Arctic have been severely affected by the geopolitical crisis, the conservation and monitoring framework developed by CAFF remains important for researchers and conservationists to follow. Since the pan-Arctic monitoring programs are described in detail in Chapter 49 on the North American tundra, this chapter focuses on the Russian conservation framework.

As the Asian Arctic tundra is entirely located within the national borders of the Russian Federation, the Russian legislation defines the protection of this section of the pan-Arctic tundra at various levels. The highest level of protection is "*zapovednik*"—a strict natural reserve (corresponds to categories Ia and Ib of IUCN (Krever et al., 2009)) managed in coordination with the federal ministry. No human activity is allowed there, except for scientific research and activities related to its conservation needs. There are five strict natural reserves in the Asian Arctic tundra with a total protected area of about 9,700,000 km^2, with Wrangel Island being the oldest (established in 1976) and Medvezhiye Ostrova ("Bear Islands") being the youngest (2020). While *zapovedniks* protect entire ecosystems, some have specific conservation goals aimed at protecting specific endangered species. For example, Medvezhiye Ostrova was established to protect an important polar bear maternity den location—six islands near the East Siberian Sea as well as the adjacent coastal area on the Sakha mainland. Several others, including Ust-Lensky (Lena-Delta) *zapovedniks*, protect unique habitats such as the mouth of the Lena—the longest river in northern Eurasia. Taymirskiy and Bolshoi Arkticheskiy *zapovedniks* cover vast areas of Arctic tundra in the north of the Krasnoyarsk region—on the Taymyr Peninsula and the Kara Sea islands. Probably the most important protected area in the Asian Arctic tundra is Wrangel Island, a UNESCO World Heritage Site. A Pleistocene refugium, biodiversity hotspot,

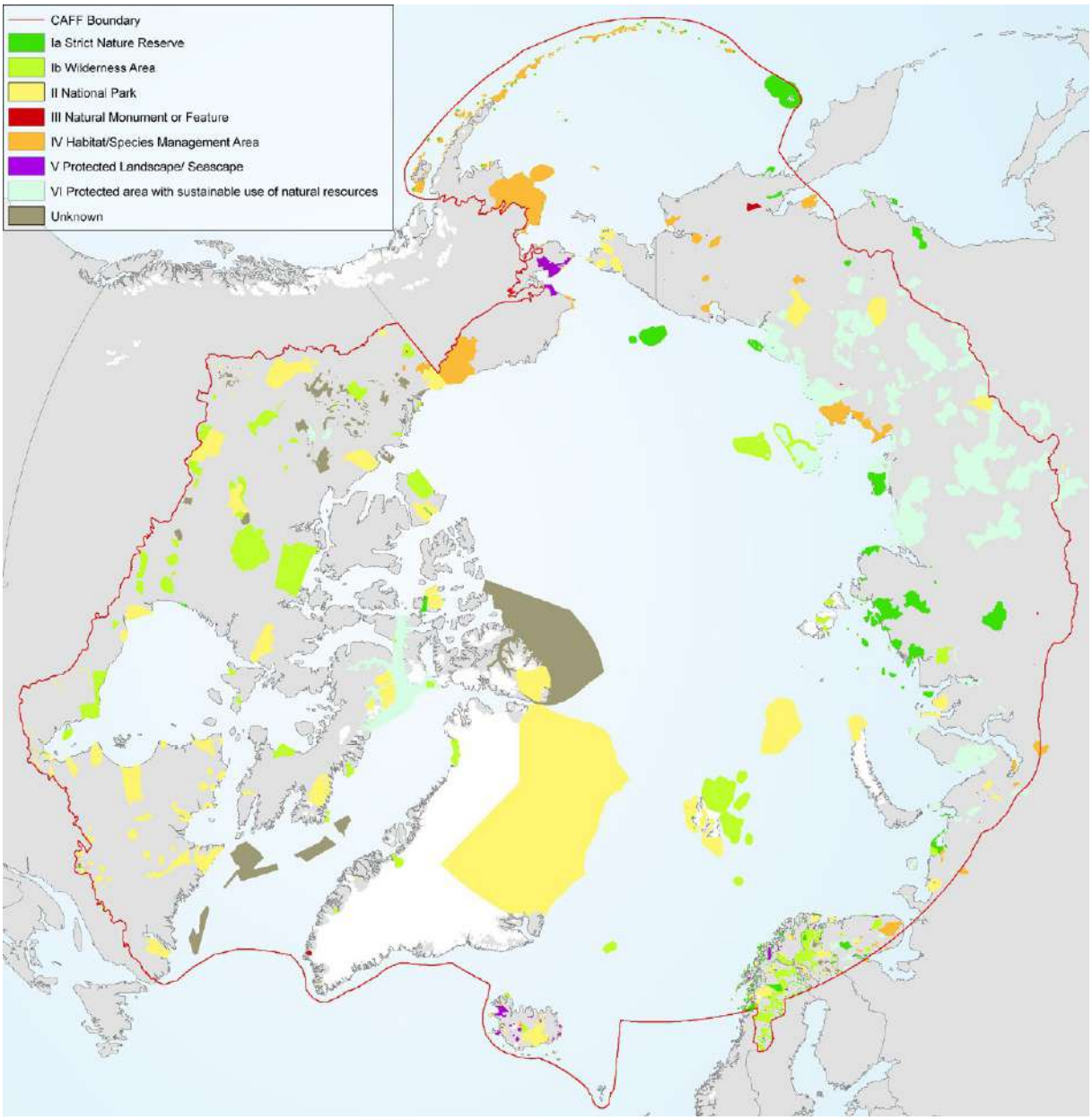

FIG. 34.11 Protected areas in the Arctic classified by their IUCN Management Category, status 2021. A total of 20.77% of Arctic terrestrial areas fall within protected areas. Due to scale, small protected areas might not be visible on the map (CAFF/PAME, 2022). Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Credit: CAFF.)*

and the floristic region with the highest plant endemism in the entire Arctic, Wrangel Island is home to 48% of Arctic endemic plant species (Ickert-Bond et al., 2013; Talbot et al., 1999), with 35 species occurring exclusively on the island. In addition to its floristic diversity, Wrangel Island is also zoologically important for its key role in supporting populations of polar bears (Chinn et al., 2023), muskoxen, Pacific walrus (*Odobenus rosmarus divergens*), and migratory birds, including snow geese (*Anser caerulescens*). The island has become one of the two experimental sites where muskoxen were reintroduced to Eurasia from Canada in the mid-1970s. Since then, the muskox population has grown to approximately 1000 animals (Cuyler et al., 2020).

The next level is a national park ("*natsionalniy park*") (corresponds to II category of IUCN). Unlike *zapovedniks*, national parks allow and even encourage some human activities, such as traditional uses of nature and controlled tourism. National parks have a functional zoning with areas with a strict regime similar to *zapovedniks*, areas of

controlled tourism, and areas open to visitors, as well as protection zones where some human activities are prohibited. One of the three Asian Arctic national parks is Gydanskiy—a former strict national reserve whose status was downgraded in 2019 to encourage traditional land use, which is becoming more difficult in the southern part of the peninsula where mining and industrial activities are expanding. Kytalyk is a new national park located in the northeast of the Republic of Sakha (Yakutia). An area of 16,000 km^2 was originally designated in 1996 as a "resource reserve" by the Republic of Sakha to primarily protect the breeding area of the Siberian white crane (*Leucogeranus leucogeranus*). The protection status has been elevated to the Russian Federation national park level in 2019, with an area of 19,000 km^2 covered (Fig. 34.12). An application to receive UNESCO world heritage status has been submitted in 2021. The area of Kytalyk is one of the most thoroughly studied in the Asian Arctic tundra, as the national park hosts the Kytalyk/Chokurdakh research station established in 2001 (Iturrate-Garcia et al., 2016; Khitun et al., 2020; Rietze et al., 2024). The third national park of the Asian Arctic is Beringia located on the easternmost tip of Eurasia, neighboring Alaska from the west. Research teams of *zapovedniks* and national parks monitor biodiversity in their area.

A lower level of protection is provided by natural parks ("*prirodniye parki*") (IUCN category III–V), *zakazniks* (IUCN categories IV, VI) and natural monuments ("*pamyatniki prirody*") (IUCN category III). These nature reserves allow most human activities with the exception of those that are harmful to specific protected species or landscapes such as hunting or industrial development. They are often overseen by regional authorities, although federal *zakazniks* and natural monuments do exist. These natural reserves are focused on specific landscapes or have a specific species or group of species as the focus of their protection, such as migratory birds or marine animals. For example, the Nizhne-Obskiy State Zakaznik in the Yamal-Nenets region, which is one of Russia's Ramsar sites, aims to protect bird species of the Ob River delta and their habitats. Similarly, the newest *zakaznik* of the region, Tiutei-Yakhinskiy, was founded in 2023 and focuses on the protection of the Atlantic walrus (*Odobenus rosmarus rosmarus*) colony.

Although a complex framework of protected areas in the Asian Arctic covering all bioclimatic subzones of tundra is established, the level of protection they actually provide is often insufficient. Despite its international recognition as a World Heritage site, Wrangel Island has been experiencing increasing military pressure since 2014, endangering the island's ecosystems (UNESCO World Heritage Committee, 2018). The borders of the Bolshoi Arkticheskiy Zapovednik were changed in 2019 to allow coal mining in part of its area. The cases of Wrangel and Bolshoi Arkticheskiy *zapovedniks*, Gydanskiy *zapovednik*/National Park show that the status of protected areas is often reconsidered for economic or political reasons, even for protected areas with the highest protection status.

However, at large scale, climate warming remains the main threat to the conservation of the Asian Arctic tundra. At least five vegetation types will be threatened by climate change by 2050, with protected areas inadequately covering future climate refugia (Reji Chacko et al., 2023). Combined with some evidence of insufficient protection of the most biodiverse regions in Western Siberia (Zemlianskii et al., 2024) and potentially other areas, this highlights the need to expand the existing network of protected areas in the Asian Arctic tundra.

Tundra plants and animals are not only threatened directly by climate warming, but their fate is strongly

FIG. 34.12 Sign at the entry of the Kytalyk National Park, indicating that access is not allowed without permission. *(Credit: Sergey Karsanaev.)*

coupled to degradation of permafrost. It is worthwhile to mention two initiatives that aim to protect permafrost from further thawing. One is the Pleistocene Park, a protected site with an experiment launched in 1989–1997 to restore a megafauna and grassland ecosystem functionally similar to the one of the Pleistocene period (Popov, 2020; Zimov, 2005). The Pleistocene Park was used as a case study to evaluate the feasibility of generating an ecosystem shift that is economically viable in terms of carbon benefits and of sufficient scale to play a significant role in global climate change mitigation (Macias-Fauria et al., 2020). Assuming a megafaunal-driven ecosystem shift, such a nature-based climate solution might create substantial income in carbon markets. However, scaling up such solutions toward a significant effect on the global climate remains challenging due to the large number of animals required over a short period of time (Macias-Fauria et al., 2020). The effectiveness has recently been evaluated and it was shown that the transformation of ecosystems into the tundra-steppe or other similar habitat occurred only slowly, and needs to be supported by anthropogenic efforts. Ungulates supported the existence of shrubs and meadows where fires or clearings occurred, but they hardly expanded the extent of this habitat (Popov, 2020). The second initiative to protect permafrost from thawing is a law "On the protection of permafrost" adopted on May 22, 2018 by the State Assembly of the Republic of Sakha (Law of the Republic of Sakha (Yakutia), 2018). The aim of the law is to strengthen state, municipal, and public supervision over the use and protection of permafrost. The main principles of the law are scientific validity, and a systematic and comprehensive approach to permafrost protection (Burenina et al., 2021). In Yamal, similar regulations came into force on January 1, 2024 (Investment Portal of the Arctic Zone of the Russian Federation, 2024). If effective, these laws will not only reduce the degradation of permafrost, but also support tundra ecosystems critically relying on permafrost.

7 Conclusions and perspectives

Arctic tundra biodiversity is relatively low, but the adaptations of Arctic plants and animals to the harsh climate are unique. Also, Indigenous Peoples well-being critically relies on the plants and animals living in the Arctic, through food subsistence, but also through cultural and relational values. Climate is currently the main driver of biodiversity change in the tundra. And this driver is very strong—Reji Chacko et al. (2023) highlighted that five of eight assessed tundra vegetation types are predicted to be threatened according to IUCN criteria already by 2050. The longer term perspective for tundra ecosystems has been evaluated in previous studies: under a 2°C mean global temperature increase, almost the entire modern tundra zone would become potentially suitable for tree growth (Callaghan et al., 2011). However, forest spread into the tundra might experience a time lag with respect to the suitable climate conditions by a low migration rate of trees, disturbances through insects and wildfires, soil drying, herbivory and human activities (Callaghan et al., 2011). A similar finding resulted from the application of a dynamic vegetation model to predict climate impact on the Siberian tundra treeline migration until 2100 and loss of tundra area: a century-long time lag was found, that later on overcompensated with infilling of stands even under cooling (Kruse and Herzschuh, 2022). Under the most ambitious climate change mitigation strategy (Representative Concentration Pathway (RCP) 2.6), up to 30% of modern tundra might persist in the north, but this tundra area would be separated into two disjunct refugia. However, under the currently more realistic scenarios of RCP 4.5 and RCP 8.5, tundra is predicted to be lost around 2100 already in the Siberian Arctic (Kruse and Herzschuh, 2022). These results clearly show that the main protection potential of the tundra ecosystem lies in immediate actions that halt climate warming through reducing carbon dioxide emissions. As a recent study showed, the cooling potential and permafrost protection by solar radiation geoengineering management is predicted to be severely reduced in the Arctic due to plant physiological forcing under still high carbon dioxide levels in the atmosphere (Müller et al., 2024). Hence, reducing carbon dioxide in the atmosphere is a prerequisite for effectively and sustainably reducing warming in the Arctic, its permafrost, and biodiversity.

While the Arctic is warming, increasing human activity is expected, especially industrial development. Protected areas reduce such additional pressures on Arctic biodiversity and ecosystem functions, such as protecting mosses with their permafrost insulation function from being compressed by vehicles. Hence, further development of protected areas require consideration of multiple aspects that have so far largely been neglected, also in the Asian Arctic. Importantly, current protected areas in the Asian Arctic do not cover most vulnerable vegetation types under warming, already by 2050 (Reji Chacko et al., 2023). Hence, it is advised to evaluate the establishment of new protected areas, with potential refugia identified already (Reji Chacko et al., 2023). This study has limitations as it was not possible to address several important vegetation types, such as wetlands critical for migratory birds, which need to be addressed urgently. Based on existing results demonstrating that current protected areas do not well cover threatened Arctic vegetation types in the future, conservation needs to quickly move beyond relying on current Arctic biodiversity patterns. Instead, existing biodiversity data and process knowledge should be immediately used to improve Earth system models (including a dynamic vegetation component) that represent Arctic plant functional

types, interactions in the ecosystem, disturbances such as permafrost degradation and wildfires, and also take into account carbon benefits. In addition, from the start of the planning of new protected areas, it will be crucial to consider biodiversity and socio-ecological effects outside of protected areas to reduce impacts as much as possible. This requires strategic and validated development toward sustainable development goals, including measurable parameters for the assessment of environmental impacts and socio-ecological aspects. More robust, integrated approaches are needed to critically evaluate social and economic outcomes of regional development as exemplified by a study on Arctic Liquefied Natural Gas Projects in the Arctic Zone of the Russian Federation (Cherepovitsyn and Evseeva, 2021). New methods to select priority investment projects for the development of natural resource extraction in the Russian Arctic are under development, with the aim to consider the environmental and societal impacts and to preserve the ethnic identity and culture of Indigenous Peoples (Novoselov et al., 2017).

The change in biodiversity under climate change and the effectiveness of protection has to be monitored over time. Most plant and animal species are distributed across the Arctic; hence, a coordination of conservation across the Arctic is required, including the assessment of genetic diversity of species and their maintenance. This also requires repeated monitoring over a long time, using standardized protocols across nations. The Arctic Council's Conservation of Arctic Flora and Fauna working group has developed a framework, but has reported that many of the indicators are not monitored regularly for the terrestrial Arctic (CAFF, 2021). As these measurements are hard to achieve with the limited resources available, it might be necessary to reconsider and downscale requirements, but define a set of absolutely critical indicators that can be monitored in the field and combined with remote sensing methods to extrapolate measurements, which require tedious fieldwork.

We might expect increasing industrial activities in the Arctic, outside of the protected areas and the protected areas alone will not deliver all the ecosystem functions needed to maintain permafrost and support the livelihoods of Arctic Indigenous Peoples. Hence, it seems of utmost importance to closely monitor the effect of industrial development on Arctic biodiversity and collaborate with the industry to reduce negative impacts to an absolute minimum. The first steps into this direction have been made with the "Mainstreaming biodiversity in Arctic mining" activity (CAFF, 2021). This activity again showed how important the exchange of experience and coordination between rights- and stakeholders across the Arctic is to achieve sustainable development goals, for the Arctic, its biodiversity, and Indigenous Peoples living and moving across national borders.

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The knowledge represented in this chapter is based on the work of Russian colleagues and their openness to collaborate with other scientists despite adverse political conditions. To maintain the knowledge on biodiversity and promote the conservation and restoration of the Asian Arctic tundra, it is of utmost importance to continue with the scientific assessments and the international cooperation and coordination for monitoring and management of this vulnerable ecosystem that is under immense pressure by climate change and intensifying additional global change drivers. We would like to thank all our Russian and international colleagues for the long-term collaboration and knowledge exchange on Arctic biodiversity and ecosystem functioning—the vastness and remoteness of the Arctic tundra requires coordinated efforts from science to policy for a meaningful and effective conservation.

This chapter is dedicated to Natalia Koroleva, a highly experienced colleague, who has contributed to the assessment of Arctic tundra plant biodiversity in her quiet and diligent way. Sadly, she passed away in spring 2024. We are most grateful for her contributions and miss her as a wonderful person and scientist.

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Chapter 35

The importance of Belarus tundra peat bogs for biodiversity conservation in global warming condition

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1 Introduction

European peat bogs can be viewed as patches of tundra in the temperate zone. They were once typical habitats in Western and Central Europe; but nowadays, as a result of peat extraction, drainage, and farm land management, the degradation of peat bogs has become pronounced (Joosten and Clarke, 2002; Wieder and Vitt, 2006). In Europe, pristine peat bogs are primarily located in Northern and Eastern Europe, as land use and degradation of peatlands in Western Europe have been much more intensive. In Central-Eastern Europe, only five countries, including Belarus, have maintained more than 50% of their peatlands in an almost intact condition (Bragg and Lindsay, 2003; Joosten et al., 2017). Peat bogs are one of the most typical landscape elements of Belarus and cover 314.5 thousand hectares (Zeliankevich et al., 2016).

Moreover, most Belarusian peat bogs are protected as hydrological and landscape reserves. Therefore, they are very suitable model systems for analyzing the effects of environmental changes on plant communities and animal assemblages of European tundra peat bogs.

Peat bogs provide many important ecosystem services, including hosting biodiversity, regulation of climate conditions, and long-term carbon and greenhouse gas sequestration. In relation to climate change, peatlands are generally considered absorbers of carbon dioxide from the atmosphere on the one hand, and a source of methane and, in some cases, nitrous oxide, on the other. CH_4 and N_2O are greenhouse gases with a fundamentally greater global warming potential than that of CO_2. Peat bogs are one of the largest reservoirs of long-term accumulation of atmospheric carbon in Europe. Due to the peat and vegetation, bogs bind many times more carbon per unit area than other terrestrial ecosystems, including forests (Minayeva and Sirin, 2012).

The current biodiversity of European peat bogs has formed from the ecological succession during the Holocene (Spitzer and Danks, 2006; Sushko, 2019a). Many of these peat bogs have been preserved since the early Holocene, about 9000–11,000 years ago, while some of the peat bogs began to form later, in particular by swamping of lakes in the middle Holocene (Geltman, 1982). This is indicated by the thickness and composition of their peat deposit (Pidoplichko, 1961; Geltman, 1982).

Nowadays, specific environmental conditions in peat bogs, such as high water table, acidity, low mineral nutrient availability, and microclimate characteristics, determine their highly specialized flora and fauna, including many cold-adapted subarctic and boreal species (Spitzer and Danks, 2006). Moreover, today, unique peat bog biodiversity is globally impacted through climate warming (Keddy, 2010). Understanding the biodiversity patterns of pristine ecosystems, including peat bogs, and the processes driving them, is of great interest in ecology in terms of current climate change and conservation of unique postglacial biota.

2 Geographic location

The Republic of Belarus is located in Eastern Europe and borders Russia, Ukraine, Lithuania, Latvia, and Poland. The area is 207,600 km^2 (50°00′N, 28°00′E).

The main climate indicators of Belarus are determined by the country's location in temperate latitudes, the absence of mountains, and its relative distance from the Atlantic Ocean. In general, the country's climate is characterized by transitional features from coastal to continental and from northern to forest-steppe zones (Loginov, 1996).

In the direction from northeast to southwest, the average annual air temperature varies from +4.5°C to +7.0°C; the

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average monthly temperature in July rises from +17.0°C to +18.5°C, while in January it ranges from −8.0°C to −4.5°C. The average annual temperature increased by 1 degree over the past 30 years; in addition, temperature fluctuations and temperature extremes increased too.

The growing season lasts, on average, 190 days. High air humidity (about 80%–90% in the cold season, 65%–70% in the spring) is typical for Belarus. The stable snow cover duration ranges from 75 days in the southwest to 125 days in the northeast country. The average long-term snow depth varies in this direction from 15 to 30 cm; in snowy winters, it reaches 50–60 cm (Belova, 2009). The territory of Belarus is located in the conjugation zone of two large botanical-geographical regions—the Eurasian coniferous forest (taiga) and the European broad-leaved forest (Geltman, 1982)—which led to the interpenetration of boreal and nemoral floras and the formation of the vegetation cover structure. These climatic conditions contribute to the preservation of tundra patches in the temperate zone, remnants from the last glaciation retreat.

The area of tundra raised bogs in Belarus varies from several tens to several thousand hectares. The largest raised bog in Belarus "Yelnya" (55,037′N, 27029′E) occupies 18,794 ha. Among other large raised bogs there are "Domzheritskoye" (54,043′N 28023′E)—8995 ha, "Morochno" (51,051′N 26038′E)—5969 ha, "Ostrova Duleby" (53,044′N 29028′E)—5049 ha, "Zaozerye" (53,053′N 29028′E)—4965 ha, "Boloto Mokh" (55,037′N. latitude 28,058′E)—4298 ha (Zeliankevich et al., 2016) (Fig. 35.1).

3 Biodiversity

3.1 Peat bogs as habitats

The main builders of peat bogs are sphagnum mosses (Fig. 35.2). Their high acidity and ability to retain water make sphagnum very important members of the peat bog ecosystem. Excess moisture contributes to the appearance of specialized peat bog plants and acidity prevents the decomposition of their residues, which form peat. Besides the high water level, high variations in temperatures on the moss carpet surface, low oxygen content, poor access to nutrients for plants, and high acidity form specific environmental conditions in peat bogs. Similar environmental conditions characterize the ecosystems of the tundra biome. Such extreme habitat conditions lead to a high competition for space and nutrients among plants of different life forms, such as herbs and shrubs. Most species that are constantly associated with peat bogs have developed specific adaptation strategies embodied in the process of evolution (Rydin and Jeglum, 2006; Minayeva and Sirin, 2012).

Spatial heterogeneity of a different degree is a distinctive feature of peat bogs. They contain different habitats for both terrestrial and aquatic organisms. Large areas only superficially appear uniform because of the ubiquitous sphagnum carpet. In addition, the ancient Belarusian peat bogs contain complexes of hollows and hummocks, small lakes and hummocks, and large lakes, which form specific landscape patterns. Their water is brown and has a pH of 3–4 (Zeliankevich et al., 2016), which creates an environment unfavorable for many plants and animals.

The microtopographic arrangements are relatively stable because the system of hummocks and hollows is maintained over time. The layers of herbs and shrubs provide a range of specific food plants for phytophages. Consequently, local microclimate conditions and ecological niches are relatively constant. The small-scale dynamics of environment conditions indicate a long-term ecological constancy of the peat bog ecosystem, as evidenced not only by the vegetation but also by the fact that the present insect fauna in a bog resembles that of the past (Spitzer and Danks, 2006; Sushko, 2019a).

Peat bogs in Central and Eastern Europe can be considered a habitat island with microclimate conditions not typical of the temperate zone of Europe. This is very important in the context of global climate warming, since peat bogs may become refugia for cold-adapted biota (Rydin and Jeglum, 2006; Spitzer and Danks, 2006).

Many peat bog dwellers, especially high specialized species, have specific habitat requirements, such as wet, acidic, and nutrient-poor sphagnum cover conditions, specific cold-adapted boreal and subarctic food plants. As a result, peat bogs in the southern boreal and temperate zones form habitat islands among the surrounding landscapes. For example, in recent forested and agricultural landscapes of the temperate zone, peat bogs occur as isolated and discrete patches of "edaphic forest tundra" and so are azonal or extra zonal ecosystems relative to the surroundings (Spitzer and Danks, 2006). In Belarus, these islands of habitat are ancient and have persisted since early Holocene periglacial periods, about 9000–11,000 years ago, when conditions were much cooler (Geltman, 1982).

Peat bog surfaces are typically convex as a result of the factors controlling their formation; thus, there are plateaus located at the edges of the border zone (lagg zone), along with a slope and a peak (dome, which can reach heights of 3–7 m, depending on the peat layer thickness). The microhabitats within the peat bog gradient from lagg to dome, differ in their vegetation cover, plant species composition, bog water level, mineral content, and pH of bog water (Fig. 35.3).

Plant community composition is most often used to assess peat bog biodiversity patterns, ecological condition, and habitat quality. These sessile primary producers of peat bogs develop slowly, reproduce at later development stages, produce few offspring, and have limited dispersal potential (Minayeva et al., 2017). Consumers, such as insects, can be more sensitive ecological indicators due to their short life cycles and high reproductive rates, which allow a more rapid reaction to environmental changes. In addition, many

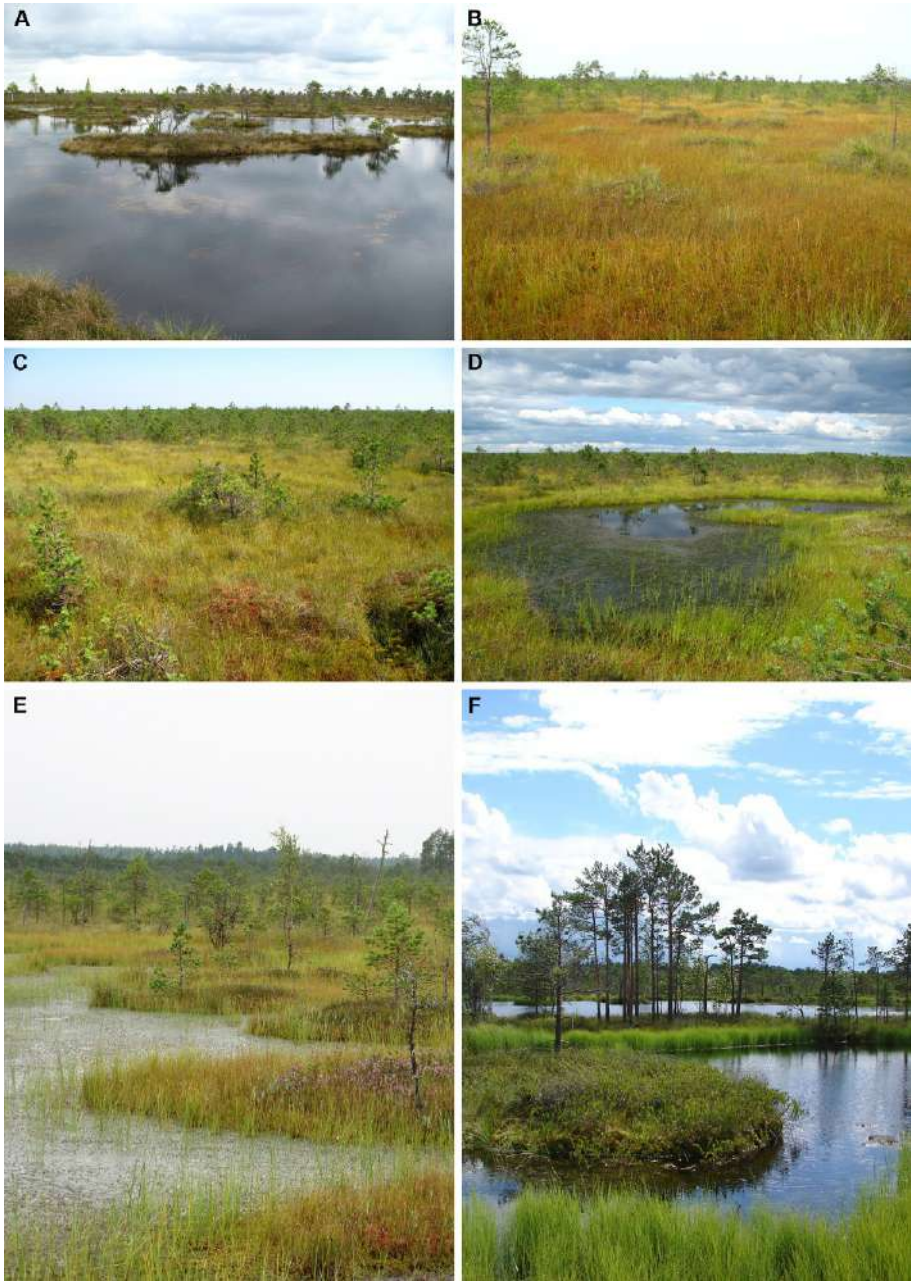

FIG. 35.1 Belarusian peat bogs (A—Yelnya bog; B—Domzheritskoye bog; C—Morochno bog; D—Ostrova Duleby bog; E—Yukhovichskoe bog; F—Moss Bog). *(Photo credits: Natallia Zeliankevich.)*

actively flying species can migrate over long distances, thereby responding to large-scale change by choosing the most suitable environmental conditions (Baguette, 2003; Sushko, 2022). Therefore, to assess peat bog biodiversity patterns, we used plants and insects as a focus group.

3.2 Peat bog plant biodiversity

The peat bog vascular plants flora in Belarus is characterized by a predominance of boreal plants representing 83.7% (72 species), among which arcto-boreal plants constitute 8.1%, boreal 32.6%, and boreal-nemoral 43.0%. The participation of the nemoral elements is low—4.7%, plurizonal—11.6%. The analysis of bryoflora phytogeographic structure indicates its almost exclusively boreal character—the proportion of boreal species is 85.3% (41 species) and that of boreal-nemoral is 4.2% (2 species); only 6.3% (3 species) represent the nemoral element and 2.1% (1 species) each for the arcto-boreal and plurizonal elements.

3.2.1 *Vascular plants*

The peat bog vascular plants included 86 species, representing 51 genera, 30 families, 6 classes, 5 divisions, which

FIG. 35.2 Sphagnum moss is the main builder of peat bog ecosystems. *(Photo credits: Natallia Zeliankevich.)*

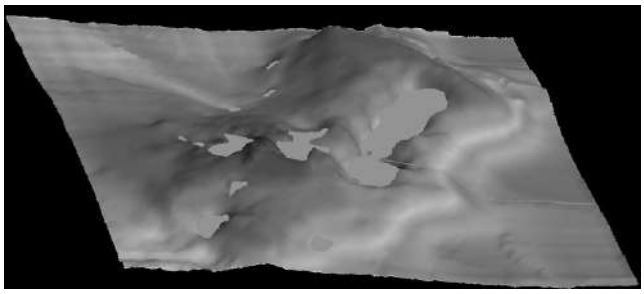

FIG. 35.3 Peat bog "Yelnya" surface relief (3D model). *(Photo credits: Dmitry Grummo.)*

represents 5% of the entire Belarus flora (1669 species). Among them are Equisetophyta (1 species), Lycopodiophyta (1), Polypodiophyta (2), Pinophyta (2), and Magnoliophyta (81), including 52 species of the Magnoliopsida class and 29 species of the Liliopsida.

The most numerous in species are seven families, which makes 61.2% of the species composition. The main role in the peat bogs vascular plant flora is played by the families (Ericaceae, Salicaceae, Betulaceae, Rosaceae, Droseraceae), which are not among the leading ones in the Belarus flora, which demonstrates the specificity of the peat bogs ecotopes. The remaining 23 families are represented by 1–3 species. Large families in the Belarus flora such as Asteraceae, Brassicaceae, Fabaceae, Caryophyllaceae, Ranunculaceae are not included in the raised bogs flora.

Among the genera, *Carex* (13 species) holds a leading position, followed by *Salix* (6 species), *Betula* and *Drosera* (4 species each), *Eriophorum*, *Vaccinium*, *Viola* (3 species each), *Nuphar*, *Oxycoccus*, *Rubus*, *Utricularia*, *Juncus,* and *Calamagrostis* (2 species each). The remaining 38 genera are represented by one species each.

In total, 22 "dedicated" (optimally develop, prefer, or grow only in peat bogs) vascular plant species are noted in the Belarus raised bogs. They belong to 10 families, which is a quarter (25.6%) of the raised bogs vascular plants list, while the randomly encountered species account is 45.5% (40 species). The main families of "dedicated" species are Ericaceae, Droseraceae, and Cyperaceae (Fig. 35.4). The leading genera among the "dedicated" species are *Drosera* (4 species), *Oxycoccus* (2), and *Carex* (2); additionally, there is one species each in the genera *Pinus*, *Betula*, *Andromeda*, *Calluna*, *Chamaedaphne*, *Ledum*, *Vaccinium*, *Empetrum*, *Rubus*, *Utricularia*, *Scheuchzeria*, *Eriophorum*, *Rhynchospora*, and *Nuphar*.

The dominants and edificators of the vegetation cover include 14 species. These are species typical of raised bogs, forming its floristic core: *Eriophorum vaginatum*, *Rhynchospora alba*, *Scheuchzeria palustris*, *Carex limosa*, *Chamaedaphne calyculata*, *Calluna vulgaris*, *Ledum palustre*, *Empetrum nigrum*, *Andromeda polyfolia*, *Oxycoccus palustris*, *Vaccinium uliginosum*, *Pinus sylvestris*, *Betula pubescens*, and *Drosera rotundifolia*.

3.2.2 Bryophytes

From the bryophytes list in peat bogs, 48 species from 23 genera and 17 families have been noted. Bryoflora is represented by six orders of the Bryophyta division: Sphagnales (22 species), Tetraphidales (1), Polytrichales (2), Dicranales (4), Bryales (2), Hypnales (9), and Jungermanniales order (7 species) of the Marchantiophyta division. The leafy mosses dominant family is Sphagnaceae (45.8% of the raised bog bryoflora); the families Hylocomiaceae (8.2%), Dicranaceae (6.2%), Polytrichaceae (4.2%), Aulacomniaceae (4.2%) have the leading positions; among liverworts, Cephaloziaceae (6.2%) and Jungermanniaceae (4.2%) families predominate. Comparative differences in the plant species composition for the raised bogs bryophyte families and for Belarus are not as substantial as for vascular plants. In the main mosses families of raised bogs, there are no species from the epiphytic, epilithic, and psammophytic groups, such as Orthotrichaceae, Pottiaceae, Grimmiaceae families (despite their significant presence in the Belarus bryoflora).

"Dedicated" species are 48% (23 species) of the identified bryophytes in raised bogs; the random species are 29% (14). The main family of "dedicated" species is Sphagnaceae (15 species); it is determined by the edificatory role of sphagnum mosses in the raised bogs vegetation cover. Also, "dedicated" species include the families Cephaloziaceae and Jungermanniaceae (2 species each) of liverworts and Dicranaceae, Polytrichaceae, Aulacomniaceae (1 species each) of bryophytes.

The main edifiers of plant communities are *Sphagnum fallax*, *Sphagnum cuspidatum*, *Sphagnum magellanicum*, *Sphagnum fuscum*, and *Sphagnum rubellum*.

The species richness of plant communities varies significantly: the total number of species (*S*) ranges from 9 to 18;

FIG. 35.4 Most typical plants of peat bogs (A—*Ledum palustre*; B—*Oxycoccus palustris*; C—*Eriophorum vaginatum*; D—*Drosera anglica*; E—*Sphagnum fuscum*; F—*Sphagnum cuspidatum*). *(Photo credits: Natallia Zeliankevich.)*

the Shannon-Weaver diversity index (H') is in the range 0.88–2.20, the Simpson ($1/D$) and Berger-Parker ($1/d$), respectively, are in the range 2.17–7.42 and 1.87–4.97. Several factors influence the dynamics of α-diversity indicators in plant communities, primarily the changes in ecological conditions of the habitats from the lagg to the top of the raised bog, as well as the duration and intensity of fires. A statistically significant negative correlation is established between the Shannon-Weaver index and an increase in water content of habitats ($r=-0.48$; $n=227$; $P=0.95$); and, on the contrary, a positive correlation is noted between an increase in species diversity index (H') of plant communities and in a nitrogen richness ($r=0.17$; $n=227$; $P=0.95$) (Grummo et al., 2010).

3.3 Spatial distribution of plants in the peat bogs

Environmental factors, plant resistance to stagnant moisture, low thermal peat conductivity, nitrogen poverty, substrate low acidity, weak forest cover, or small canopy density determine the predominance of certain plant species in bog plant communities. Variability in the humidity regime, illumination, and mineral content determine the spatial heterogeneity of the raised bogs vegetation cover. Table 35.1 shows the distribution and ecological location of the most typical plant associations in Belarus raised bogs (according to ecological-floristic classification; Braun-Blanquet, 1964).

TABLE 35.1 Distribution and ecological location of the most typical plant associations in Belarus.

Association	Localization	Dominant species	Ecological condition
Vaccinio uliginosi-Pinetum sylvestris	Bog edges or well-drained slopes	*Pinus sylvestris* f. *uliginosa, Ledum palustre, Vaccinium myrtillus, Vaccinium uliginosum; Sphagnum angustifolium, Sphagnum magellanicum*	Bog water level (WL) = 31 ± 9 sm; pH = 3.71 ± 0.01; electrical conductivity (EC) = 92.7 ± 16 μS/sm
Sphagno-Pinetum sylvestris	Bog edges, slopes with forest rings	*Pinus sylvestris* f. *litwinowii; Eriophorum vaginatum; Sphagnum angustifolium, Sphagnum magellanicum*	WL = 13 ± 3 sm; pH = 3.53 ± 0.01; EC = 66.4 ± 11 μS/sm
Ledopalustris-Sphagnetum fusci	High ridges (46 ± 5 sm) and hummocks with oligotrophic conditions	*Chamaedaphne calyculata, Eriophorum vaginatum, Calluna vulgaris, Sphagnum fuscum*	WL = 16 ± 2 sm; pH 3.52 ± 0.05; EC 46.7 ± 11 μS/sm
Sphagnetum magellanici	Hummocks (34 ± 4 sm), along bog edges	*Eriophorum vaginatum; Sphagnum magellanicum, Sphagnum angustifolium*	WL = 14 ± 4 sm (−4 … −35); pH = 3.59 ± 0.04; EC = 58.7 ± 17 μS/sm
Empetronigri-Sphagnetum rubelli	Short ridges and hummocks (22 ± 2 sm)	*Eriophorum vaginatum, Sphagnum rubellum, Sphagnum fuscum*	WL = 11 ± 3 sm; pH = 3.65 ± 0.05; EC = 56.7 ± 12 μS/sm
Sphagnorecurvi-Caricetumlasio carpae	Meso-oligotrophic laggs and open bogs on the slops	*Carex lasiocarpa, Sphagnum recurvum*	WL = 4.40 ± 1.5 sm; pH = 4.5 ± 0.04; EC = 75.2 ± 12 μS/sm
Eriophoro vaginati-Sphagnetum recurvi	Hummocks, ridge-hollow complex, lagg (meso-oligotrophic)	*Eriophorum vaginatum; Sphagnum recurvum, Sphagnum angustifolium*	WL = 5 ± 0.5 sm; pH = 3.71 ± 0.07; EC = 75.4 ± 17 μS/sm
Drosero anglicae-Sphagnetumrubelli	Around lakes on the slopes	*Eriophorum vaginatum, Andromeda polifolia; Sphagnum cuspidatum, Sphagnum rubellum, Sphagnum balticum*	WL = 6 ± 1.3 sm; pH = 3.82 ± 0.04; EC = 56 ± 12 μS/sm
Caricetumlimosae	Hollows	*Carex limosa, Scheuchzeria palustris, Eriophorum vaginatum*	WL = 17 ± 4 sm; pH = 3.61 ± 0.09; EC = 59 ± 16 μS/sm
Rhynchosporetumalbae	Hollows and wet moss carpets	*Rhynchospora alba*	WL = 2 ± 0.5 sm; pH = 3.68 ± 0.04; EC = 49 ± 14 μS/sm

3.4 Peat bog insect biodiversity

In the peat bogs of Belarus, 1384 species of insects (Insecta, Ectognatha) belonging to 13 orders, 32 suborders, 77 superfamilies, 200 families, and more than 738 genera have been recorded (Table 35.2).

Insects of four orders were characterized by the highest species richness: Coleoptera (428 species), Hymenoptera (354 species), Lepidoptera (222 species), and Diptera (more than 135 species). Despite the excessive moisture of raised bogs, the presence of hollows, pools, and lakes, the majority of identified insects are inhabitants of terrestrial habitats (1229 species). There were fewer inhabitants (Fig. 35.5) of the aquatic environment (102 species) and amphibionts (53 species).

3.4.1 Spatial distribution of insects in the peat bogs

The taxonomic composition of insects in peat bogs is characterized by the highest heterogeneity in the vertical aspect. Based on the number of identified taxa, assemblages of the herb-shrub layer (374 species of 9 orders) are distinguished, while in the moss layer (173 species of 8 orders) and woody layer (168 species of 8 orders) the taxonomic diversity is lower. At the same time, the basis of the assemblages was formed by a limited number of taxa: in the moss cover dominated beetles of the families Staphylinidae (58 species) and Carabidae (55 species), as well as ants (13 species), in the herb-shrub layer dominated coleopterans of the families Chrysomelidae (44 species), Curculionidae

TABLE 35.2 Taxonomic composition of insects (Insecta, Ectognatha) of Belarusian peat bogs.

Order	Number of suborders	Number of superfamilies	Number of families	Number of genera	Number of species
Ephemeroptera	–	2	2	2	2
Odonata	2	5	7	18	35
Dictyoptera	1	–	1	1	1
Orthoptera	2	3	3	6	10
Psocoptera	1	–	1	1	2
Hemiptera	4	23	31	121	166
Coleoptera	2	15	48	221	428
Neuroptera	1	–	2	6	13
Hymenoptera[a]	–	–	11	105	354
Mecoptera	–	–	1	1	1
Trichoptera	2	4	6	12	16
Lepidoptera	17	27	51	172	222
Diptera[b]	–	–	36	73	135
Total 13	32	77	200	738	1384

[a]*For the most taxa, except Apidae and Formicidae, literary data were used (Lobodenko, 2000; Shlyakhtenok, 1996, 2007; Tereshkin, 1996).*
[b]*Partially identified to the family.*

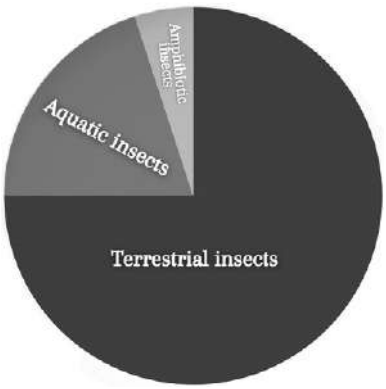

FIG. 35.5 Proportion (%) of insect species (Insecta, Ectognatha) of peat bogs in different environments. *(Photo credits: Gennadi Sushko.)*

(34 species), bugs of the families Pentatomidae (18 species), Miridae (16 species), and Lygaeidae (15 species), cicadas of the families Cicadellidae (31 species) and Cercopidae (6 species), dipterans of the families Chloropidae (9 species), in tree level dominated beetles of the families Curculionidae (21 species) and Chrysomelidae (19 species), bugs of the families Pentatomidae (7 species) and Miridae (5 species), cicadas of the families Cicadellidae (9 species) and Cercopidae (5 species) (Fig. 35.6).

The habitat distribution of insects in seven of the most typical peat bog habitats (plant composition types based on the dominant species in each layer of the vegetation canopy) in accordance with a peat bog gradient was studied: (1) lagg zone (LZ) at the bog margin (*Eriophorum vaginatum-Sphagnum angustifolium*); (2) pine bog (PB) on the slope (*Pinus sylvestris-Eriophorum vaginatum/Ledum palustre-Sphagnum magellanicum*); (3) hollow (HOL) on the slope (*Rhynchospora alba-Sphagnum cuspidatum*); (4) hummock (HUM) on the slope (*Eriophorum vaginatum-Oxycoccus palustris-Andromeda polifolia-Ledum palustre-Sphagnum magellanicum-Sphagnum angustifolium*); (5) open bog space (OBS) on the slope (*Eriophorum vaginatum-Ledum palustre-Chamaedaphne calyculata-Empetrum nigrum-Calluna vulgaris-Oxycoccus palustris-Vaccinium uliginosum-Sphagnum magellanicum*); (6) lakeshore (L) on the slope (*Carex limosa-Sphagnum cuspidatum*); and (7) dome (D) (*Eriophorum vaginatum-Calluna vulgaris-Sphagnum fuscum*).

Moss level. In the sphagnum carpet, 173 species of 5 orders such as Dictyoptera (1 species), Orthoptera (1 species), Hemiptera (14 species), Coleoptera (144 species), Hymenoptera (13 species) have been recorded. The highest number of species in the assemblages of the moss layer in all studied habitats was characterized by insects of two orders, such as Coleoptera and Hymenoptera, while insects of the other orders (Orthoptera, Dictyoptera, and Heteroptera) had low richness. Among them, Staphylinidae (11–34 species) and Carabidae (9–31 species) beetles predominated, as well as ants (3–10 species). In peat bogs, ants build their nests in

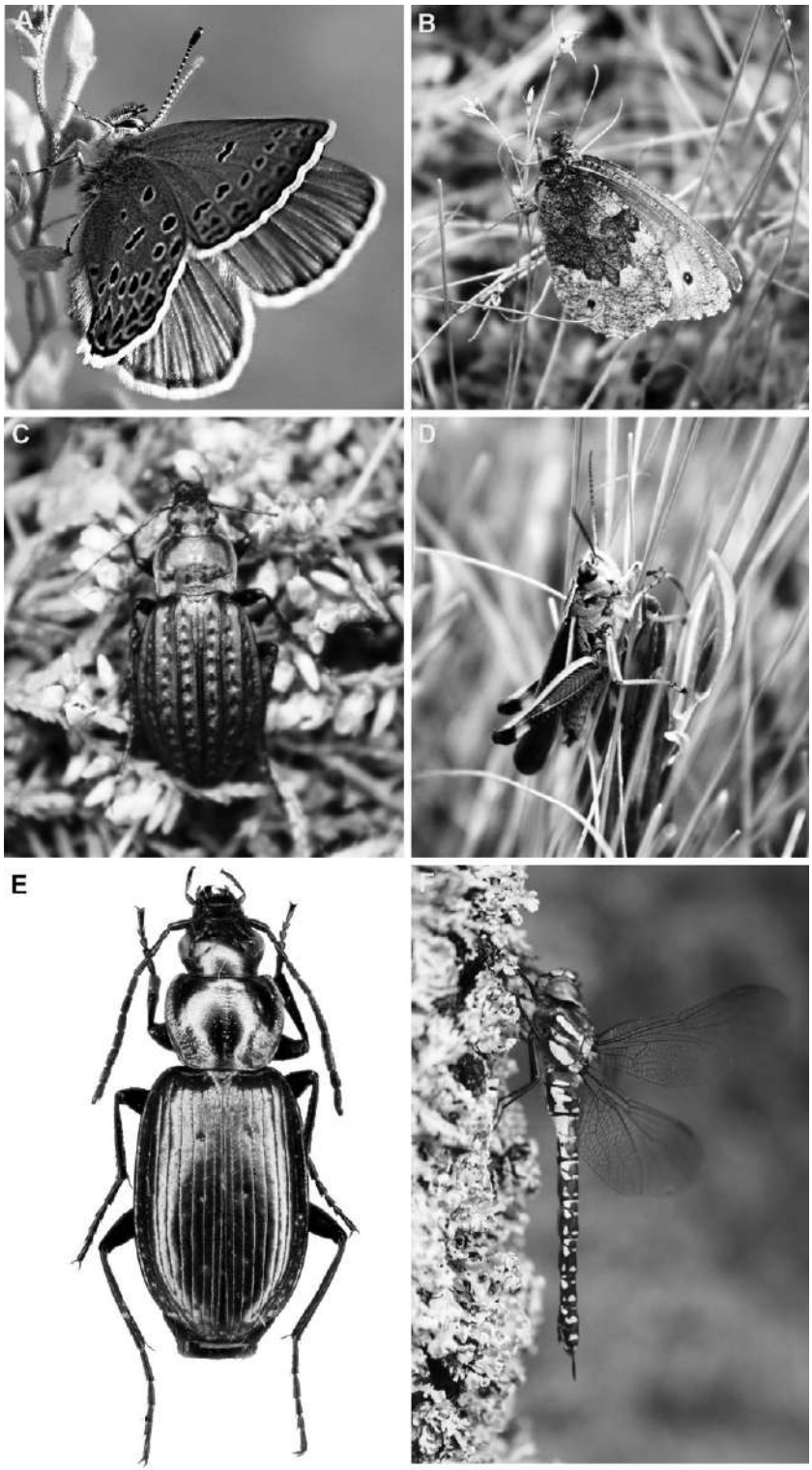

FIG. 35.6 Peat bog insect species (A—*Vacciniina optilete*; B—*Oeneis jutta*; C—*Carabus clathratus*; D—*Mecostethus grossus*; E—*Agonum eriseti*; F—*Aeshna subarctica*). *(Photo credits: Gennadi Sushko (A–E), Alexander Lukashuk (F).)*

hummocks and deadwood. As reported by Blinov (1991), 1 ha of peat bog contains 3626 anthills. However, their species composition was not high. The most abundant were specialized peat bog ants *Formica forsslundi*, *Formica picea*, and *Formica uralensis*, as well as *Myrmica scabrinodis*.

Species richness (ANOVA, $F=3.32$, $P=0.01$) and abundance (Kruskal-Wallis test, Hc = 42.45, $P=0.002$) differed significantly among the assemblages of seven habitats.

The insect species richness ranged from 23 in hollows to 82 in pine bogs. The highest species richness was in the bog slope with relatively average moisture conditions of the moss cover. In the all studied assemblages, the most abundant were 2–3 specialized species, which were the most adapted to the environmental conditions of the peat bogs. In particular, among the beetles such species were *Agonum ericeti* (from 51.69% to 64.44% of all captured individuals) and *Drusila canaliculata* (from 8.99% to 26.12% of all captured individuals) (Sushko, 2019b). In the Heteroptera, *Rhyparochromus pini* dominated. In addition, the beetles *Pterostichus diligens* (from 5.57% to 8.09%) and *Pterostichus rhaeticus* (from 2.10% to 3.66%) were abundant (Table 35.3).

The open bog sites ($H'=1.036$) and pine bogs ($H'=1.025$) showed the highest average Shannon index, whereas the lowest value was recorded from hollows ($H'=0.665$) and the dome ($H'=0.772$). The Pielou evenness index ($J=0.779$) was also lowest in the dome ($J=0.467$).

Herb-shrub layer. In the herb-shrub layer, 374 species of 8 orders such as Dictyoptera (1 species), Orthoptera (9 species), Psocoptera (2 species), Hemiptera (128 species: Sternorrhyncha 7 species, Auchenorrhyncha 46 species, Heteroptera 75 species), Coleoptera (153 species), Neuroptera (10 species), Hymenoptera (7 species), Diptera (64 species) have been recorded. The highest number of species in the assemblages of the herb-shrub layer in all studied habitats was characterized by beetles of the families Chrysomelidae (12–34 species) and Scirtidae (3–5 species), bugs of the families Miridae (5–14 species) and Lygaeidae (3–13 species), cycads of the families Cicadellidae (15–21 species) and Cercopidae (3–6 species), dipterans of the families Empididae (3–5 species) and Sepsidae (3–5 species). Moreover, this layer differs from the moss layer in the higher heterogeneity of microhabitats and ecological resources, as evidenced by the doubling of the number of species. Species richness (ANOVA, $F=6.07$, $P=0.005$) and abundance (Kruskal-Wallis test, Hc = 37.45, $P=0.01$) differed significantly among the assemblages of seven habitats.

The insect species richness ranged from 129 in hollows to 255 in pine bogs. The highest species richness was in the bog slope with a high proportion of shrubs in plant communities. On the other hand, insect assemblages of sites with the presence of only herbs in plant communities, such as hollows and lake shores, were characterized by low species richness. There was also low insect richness in the dome, where heather dominated the herb-shrub layer (Sushko, 2016a,b, 2017; Table 35.4).

The most abundant species in the herb-shrub layer insect assemblages were *Neophilaenus lineatus* (Auchenorryncha), *Stephanitis oberti* (Heteroptera), and *Lochmaea suturalis* (Coleoptera). In some habitats, their abundance exceeded 20.0%. *Mecostethus grossus*, *Metrioptera brachyptera* (Orthoptera), *Cixius similis*, *Neophilaenus lineatus*, and *Lepyronia coleoptrata* (Auchenorryncha), *Stephanitis oberti* (Heteroptera), *Cyphon padi*, *Plateumaris discolor* (Coleoptera) also had high abundance in the herb-shrub layer.

The open bog sites ($H'=1.979$) and pine bogs ($H'=1.930$) showed the highest average Shannon index, whereas the lowest value was recorded from lag zones at the bog margins ($H'=1.509$) and lake shores ($H'=1.511$). The Pielou evenness index ($J=0.779$) was also lowest in the lag zones ($J=0.686$) and in the lake

TABLE 35.3 Overview of the parameters of moss level insect (Insecta, Ectognatha) assemblages of peat bogs.

Parameters	Habitats						
	LZ	*PB*	*HOL*	*HUM*	*OBS*	*L*	*D*
S_{max}	69	82	23	51	81	45	50
S_{min}	37	63	9	34	64	20	33
$S_{mean} \pm SE$	52.6±3.5	68.8±1.7	17.3±2.4	41±2.1	70.1±1.7	33.3±3.1	40.3±1.7
Chao-1	105	112	32	63	98	55	72
Shannon index (H')	1.007	1.025	0.665	0.800	1.036	0.969	0.772
Evenness index (J')	0.579	0.540	0.489	0.481	0.552	0.605	0.467

Habitat symbols: *LZ*, lagg zone; *PB*, pine bog; *HOL*, hollow; *HUM*, hummock; *OBS*, open bog space; *L*, lakeshore; *D*, dome.

TABLE 35.4 Overview of the parameters of herb-shrub level insect (Insecta, Ectognatha) assemblages of peat bogs.

Parameters	Habitats						
	LZ	*PB*	*HOL*	*HUM*	*OBS*	*L*	*D*
S_{max}	210	255	129	215	262	134	167
S_{min}	108	100	37	99	98	39	75
$S_{mean} \pm SE$	140.1 ± 7.2	160.6 ± 14	61.7 ± 9.4	132.1 ± 8	162.7 ± 14	61.7 ± 8	108 ± 8.4
Chao-1	232	279	148	245	282	146	196
Shannon index (H')	1.509	1.930	1.651	1.884	1.979	1.511	1.791
Evenness index (J')	0.686	0.842	0.841	0.855	0.861	0.779	0.851

Habitat symbols: *LZ*, lagg zone; *PB*, pine bog; *HOL*, hollow; *HUM*, hummock; *OBS*, open bog space; *L*, lakeshore; *D*, dome.

shores ($J = 0.779$). At the same time, the diversity of insects in the grass-shrub layer assemblages ($H' = 1.509$–1.930) was higher than in the moss layer insect assemblages ($H' = 0.665$–1.036).

Tree level. The tree level on peat bogs is represented by pine (*Pinus sylvestris*) and birch (*Betula pubescens*). In the tree level layer in general, 168 species of 7 orders such as Psocoptera (2 species), Hemiptera (45 species: Sternorrhyncha 6 species, Auchenorrhyncha 17 species, Heteroptera 21 species), Coleoptera (98 species), Neuroptera (4 species), Hymenoptera (4 species), Diptera (16 species) were recorded (Table 35.5).

In insect assemblages of *Pinus sylvestris* (131 species), higher species richness is recorded compared to assemblages of *Betula pubescens* (99 species). Species richness ($U = 26.07$, $P = 0.04$) and abundance ($U = 47.07$, $P = 0.02$) differed significantly among the assemblages of *Pinus sylvestris* and *Betula pubescens*. The most abundant species on the *Pinus sylvestris* were beetles *Cyphon padi*, *Cyphon pubescens*, *Coccinella hieroglyphica*, *Brachonyx pineti*, *Ampedus balteatus,* and *Hippodamia tredecimpunctata*. Among the most abundant species recorded on *Betula pubescens* were beetles *Lochmaea caprea*, *Deporaus betulae*, *Cyphon padi,* and *Altica aenescens*. Insect assemblages of *Betula pubescens* ($H' = 1.889$) and *Pinus sylvestris* ($H' = 1.699$) showed the similar average Shannon index and high evenness.

TABLE 35.5 Overview of the parameters of tree level insect (Insecta, Ectognatha) assemblages of peat bogs.

Parameters	*Pinus sylvestris*	*Betula pubescens*
S_{max}	99	131
S_{min}	54	80
$S_{mean} \pm SE$	67.5 ± 6.5	97.1 ± 7.4
Chao-1	120	185
Shannon index (H')	1.699	1.889
Evenness index (J')	0.906	0.912

Water bodies. On the peat bogs, there are water bodies such as lakes of different sizes, hollows, and streams. In the water bodies, 145 species of 4 orders such as Ephemeroptera (2 species), Odonata (35 species), Hemiptera (21 species), Coleoptera (87 species), Trichoptera (16 species) were recorded.

Among aquatic and amphibiotic insects, as well as among terrestrial insects, in each taxon we identified several abundant species, while most other species had low abundance. The most abundant species were *Cloeon dipterum*, *Leptophlebia vespertina* (Ephemeroptera), *Leucorrhinia albifrons*, *Leucorrhinia dubia*, *Leucorrhinia rubicunda*, *Sympetrum danae*, *Sympetrum sanguineum,* and *Sympetrum vulgatum* (Odonata), *Cymatia coleoptrata*, *Ilyocoris cimicoides*, *Mesovelia furcata*, *Gerris odontogaster* (Hemiptera), *Acilius sulcatus*, *Enochrus ochropterus*, *Enochrus affinis*, *Ilybius aenescens*, *Anacaena lutescens* (Coleoptera), *Triaenodes bicolor*, *Phryganea bipunctata*, *Agrypnia obsoleta*, *Oligotricha striata*, *Limnephilus elegans* (Trichoptera).

The species diversity of water bodies can be demonstrated by the example of beetle assemblages. The mean Shannon index values significantly differed among the water bodies (Kruskal-Wallis test $H = 12.7$, $P = 0.001$). Diversity measures indicate higher values in the lakes ($H' = 2.969$) and hollows ($H' = 2.736$) compared to the stream ($H' = 1.970$), which had the highest values of evenness ($J' = 0.762$). Evenness varied from $J' = 0.558$ (hollows) to $J' = 0.762$ (streams) (Shatarnova, 2021).

Moths and butterflies. Moths and butterflies are important in the terrestrial ecosystem. They are integral participants in the food chain and pollinators. Among

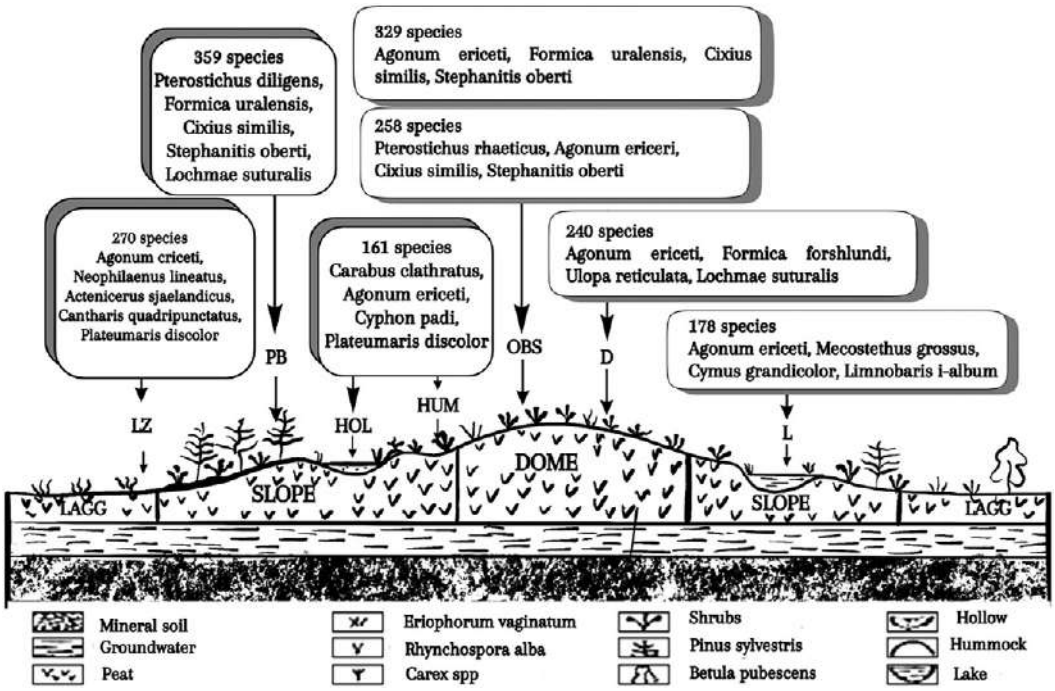

FIG. 35.7 Spatial distribution of the most abundant insect species in peat bog habitats. *(Photo credits: Gennadi Sushko.)*

invertebrates, butterflies are widely used as indicators to assess environmental changes (Sushko, 2022; Sommer et al., 2022). Since adults and larvae of lepidoptera can occupy different ecological niches, we will present the characteristics of these taxa separately.

In the peat bogs, 222 Lepidoptera species (192 species of moths and 30 species of butterflies) were recorded. *Aristotelia ericinella* (Gelechiidae), *Rhagades pruni* (Zygaenidae), *Catoptria margaritella* (Crambidae), *Macaria brunneata*, *Ematurga atomaria*, *Arichanna melanaria*, *Rheumaptera hastata*, *Carsia sororiata* (Geometridae), *Acronycta menyanthidis*, *Hyppa rectilinea*, *Celaena haworthii*, *Orthosia gothica*, *Coranarta cordigera*, *Lycophotia porphyrea*, *Coenophila subrosea* (Noctuidae), *Diacrisia sannio*, *Thumatha senex* (Arctiidae), *Callophris rubi*, *Plebeius argus*, *Vacciniina optilete* (Lycaenidae), *Boloria eunomia*, and *Boloria euphrosyne* (Nymphalidae) were the most common species.

The highest Lepidoptera species richness and diversity were in the border lagg zones. Obviously, many of these species dispersed to peat bogs from other surrounding habitats. The highest butterfly diversity in the border lag zones may be attributed to the edge effect. The highest butterfly abundance was in the pine bogs, which are characterized by higher plant community structural complexity and, as a result, higher butterfly food resources. However, it should be noted that among butterflies with diurnal activity, only two species, *Plebejus argus* and *Boloria euphrosyne*, represented the majority of individuals in the pine bogs (85.99% of all recorded individuals). As a result, the butterfly assemblages of the pine bogs had significantly lower diversity and evenness than those of the other habitats. Overall, comparisons of species richness, abundance, diversity, and composition revealed that different peat bog habitats determine spatial distributions of insects, reflecting spatial heterogeneity in key environmental variables (Fig. 35.7).

4 Conservation areas and their ecological indicator

There are 239.6 thousand hectares of raised bogs that are safe in Belarus within protected areas (Table 35.6). Regimes of protection include:

(1) In the Biosphere Reserve, as well as in the National Parks protected zones, a complete ban on all economic activities, including picking berries, mushrooms, hunting, fishing, etc.
(2) In the National Parks (except for protected zones), Nature Reserves, and in protective forest parts, there are restrictions on certain types of activities (as a rule, logging, drainage, irrigation, road construction, peat and sapropel extraction, chemical products use, etc.). Amateur picking berries, mushrooms, hunting, and fishing allowed.
(3) In the Nature Reserves, limited economic activity is allowed, which does not affect valuable biotopes and rare and protected species populations. The raised

TABLE 35.6 General information about the raised bogs protection in Belarus.

Specially protected natural area status	Total area, ha	% of the raised bogs total area
1. Biosphere reserve	12,814.6	4.1
2. National parks	26,060.0	8.3
3. Republican significance nature reserves	150,214.8	47.7
4. Local significance nature reserves	47,733.7	15.2
5. Natural landmark	2825.0	0.9
Total protected area	**239,648.1**	**76.2**

bogs protection in the Reserves combines protection with environmental education. The most valuable objects are located within the Reserves "Yelnya" (55,037′N 27029′E), "Osveisky" (56,004′N 28003′E), "Kozyansky" (55°25′N 29°22′E), "Korytensky Mokh" (55,028′N 30012′E), "Boloto Mokh" (55,037′N 28058′E), etc.

It is recommended to leave a buffer zone 0.5–1.0 km wide around protected bogs of all categories, where the same regimes should be observed as in the protected area. Bog protection regimes are specified and supplemented in each specific case.

In recent years, the National Ecological Network, which is a system of functionally interconnected protected areas, is greatly developed. It contains the following zones: Cores are areas that ensure the natural ecosystems preservation; Ecological Corridors are areas connecting Cores and ensuring species migration flows, and Protective (or Buffer) Territories allocated to reduce the anthropogenic impact on natural complexes located within the first two zones. Belarus Ecological Network Cores include several large raised bogs with surrounding forests. The raised bogs in Belarus are the natural complexes that perhaps preserved their almost pristine natural appearance for many hundreds of years.

The largest Belarus bog is the "Yelnya" bog (55,037′N 27029′E), the basis of a Nature Reserve in the Vitebsk region. The maximum length from north to south is 20.5 km, from west to east—16.6 km. The total reserve area is 25,301 ha. The maximum peat deposit depth reaches 8.3 m; the average is 3.8 m. On the Reserve territory, there are 36 lakes with more than 1-ha area and 82 lakes with less than 0.5 ha area. The largest peat bog lakes are "Yelnya" (5.4 km^2), "Tchernoje" (0.7 km^2), and "Beloje" (0.5 km^2). The high importance of the Yelnya Reserve for the biological diversity conservation is confirmed by its international conservation status: Important Bird Area (since 1998); Ramsar site (since 2002), and Important Plant Area (since 2005).

In the raised bogs, there are protected plant species included in the Belarus Red Book: *Carex buxbaumii* I (CR), *Betula nana* II (EN), *Rubus chamaemorus* II (EN), *Nuphar pumila* II (EN), *Salix myrtilloides* III (VU), *Carex paupercula* III (VU), *Carex pauciflora* III (VU), *Baeothryon alpinum* III (VU), *Eriophorum gracile* III (VU), *Drosera intermedia* III (VU), *Sphagnum molle* III (VU), *Salix lapponum* IV (NT), *Oxyccocus microcarpus* IV (NT). Among protected insects, there are *Sympecma paedisca* III (VU), *Carabus clathratus* III (VU), *Carabus menetriesi* III (VU), *Carabus nitens* III (VU), *Chlaenius costulatus* (EN), *Bombus muscorum* III (VU), *Bombus schrenckii* III (VU), *Colias palaeno* IV (NT), and *Oeneis jutta* IV (NT).

Among vertebrate animals in Belarus, excluding birds, no species is known to occur only in peat bogs, even though certain species are found mostly on peatlands. Peat bog birds have been well studied in Belarus (Kuzmenko et al., 1985). Among them are rare and protected species such as *Aquila chrysaetos* I (CR), *Pandion haliaetus* II (EN), *Falco peregrinus* I (CR), *Tringa nebularia* III (VU), and others. However, only a few species inhabit exclusively in peat bogs, including *Gavia arctica* II (EN), *Lagopus lagopus* I (CR), *Pluvialis apricaria* III (VU), and *Numenius phaeopus* III (VU), among others.

Bog habitats do not provide a suitable environment for vertebrates necessary to complete their life cycles. Many species that inhabit the bog dome and slope forage in the more nutrient-rich edges of the bogs and in the surrounding landscapes. In contrast, a large number of invertebrate species have been reported almost exclusively from peat bogs, at least in the southern part of the distribution area of these species (Spitzer and Danks, 2006). From this point of view, protected peatlands play an important role in the conservation of specialized insect species that cannot live in other ecosystems. Considering that insects are the most numerous animal taxa, the reduction in the area of peat bogs around the world leads to a large loss of biodiversity.

FIG. 35.8 Peat bog with a drained network of canals. *(Photo credits: Gennadi Sushko.)*

Belarusian peat bogs provide a habitat for many specialized insect species (so-called tyrphobionts). Among them are subarctic and boreal cold-adapted species such as *Aeschna subarctica*, *Somatochlora arctica* (Odonata), *Cixius similis*, *Ophiola russeola*, *Cymatia bonsdorffii* (Hemiptera), *Agonum ericeti*, *Cyphon kongsbergensis*, *Plateumaris discolor*, *Altica longicollis* (Coleoptera), *Formica forsslundi*, *Formica uralensis* (Hymenoptera), *Athrips pruinosella*, *Eupithecia gelidata*, *Carsia sororiata*, *Teia antiquiodes*, *Syngrapha interrogationis*, *Syngrapha microgamma*, *Coranarta cordigera*, *Coenophila subrosea*, *Colias palaeno*, *Vacciniina optilete*, *Clossiana eunomia*, *Boloria aquilonaris*, and *Oeneis jutta* (Lepidoptera).

5 Human activities and their impacts

In the mid-20th century, as in many other European countries, a number of Belarus raised bogs were drained for peat extraction. It significantly reduced the proportion of natural bogs. Today, from the previously raised bogs total area, 16.8% mined out or completely drained. The proportion of partially disturbed by drainage bogs (bogs with some preserved natural parts, or not completely drained) is quite large—30.1%. In the undisturbed state remained 53.1% raised bogs. Some peat bogs with a drained network of canals have not yet not recovered and lost their biosphere functions (Fig. 35.8).

Human influence on raised bog ecosystems is not only due to direct destruction from peat extraction and drainage, but can also occur indirectly when peat bogs continue to function, while their appearance and the peat accumulation rates change. Direct and indirect anthropogenic impact factors are shown in Fig. 35.9.

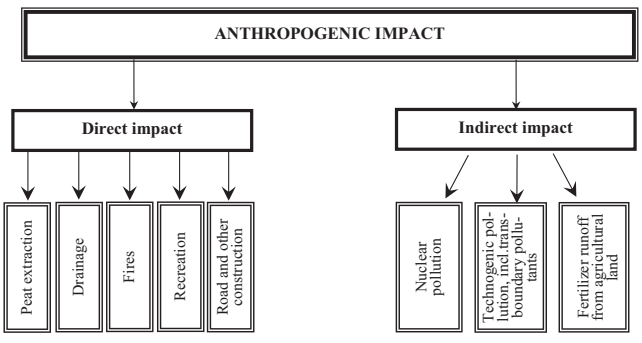

FIG. 35.9 Anthropogenic impact factors. *(Photo credits: Dmitry Grummo, Natallia Zeliankevich.)*

The water regime disruption during the drainage of raised bogs primarily causes changes in ground cover. In ground cover appear species, that prefer a water content decreasing, a mineral substances increasing and an illumination intensity decreasing. The floristic core in undisturbed bog communities is formed by sphagnum mosses, as well as *Andromeda polifolia*, *Oxycoccus palustris*, *Empetrum nigrum*, and *Eriophorum vaginatum*. Depending on the anthropogenic impact degree, the projective cover and biomass of these species, as a rule, decrease until their complete elimination. Thus, the *Oxycoccus palustris* cover within the reclamation site can decrease by 18 times and *Sphagnum magellanicum* cover by 11 times. The indicator and constant species of sphagnum pine forests disappear from the communities' composition under the drainage influence (*Sphagnum fuscum*, *Polytrichum strictum*, and *Drosera rotundifilia*). However, the shrubs cover such as *Vaccinium uliginosum* and *Calluna vulgaris* tends to increase significantly. The vegetation degradation zone runs along reclamation ditches in strips up to 150–200 m wide. The highest species diversity is observed in the zones influenced by weak and moderate drainage canal systems. This is quite consistent with Connell's "intermediate disturbance" theory: the highest diversity is found in the periodically exposed to moderate disturbances communities. Species richness increase occurs, as a result, of the not typical for raised bogs species introduction, which respond positively to changes in the edaphotope properties. A decrease in the anthropogenic pressure influence with distance from drainage systems is accompanied usually by a species diversity decrease.

Another important factor in the raised bogs transformation is fires. Raised bog fires are often associated with drainage. The main patterns of postfire vegetation digression are as follows: (1) species composition impoverishment; (2) tree layer thinning or complete death; (3) regrowth of deciduous tree species (*Betula pendula*, *Betula pubescens*, and *Populus tremula*); (4) coenotic significance (occurrence, abundance, projective cover, biomass) increase of phytocenoses pyrogenic destruction indicators are *Polytrichum strictum* and *Calluna vulgaris*; and (5) the overall communities productivity reduction.

Postfire communities are unstable and always temporary formations. Demutation processes at postpyrogenic sites are accompanied by: (1) deciduous trees oppression and loss, (2) spread of bog pine forms, (3) decrease in the phytocenotic significance of secondary plant communities edificators and dominants, and (4) active growth of sphagnum mosses (*Sphagnum magellanicum*, *Sphagnum angustifolium*, *Sphagnum fuscum*, etc.) and marsh shrubs (*Andromeda polifolia*, *Ledum palustre*, and *Oxycoccus palustris*).

In Belarus, peat bogs are used for berry picking, hunting, fishing, and ecotourism; roads, oil and gas pipelines, and power lines are laid through them. Raised bogs are visited most intensively during the berry-picking season (great bilberries, lingonberries, and especially cranberries). Moderately moist sphagnum cover is quickly restored and is minimally damaged, where medium-density sphagnum mosses show absolute dominance (with large linear moss growth—3–4 cm per year). The most hydrophilic sphagnum mosses are loose and easily destroyed, but due to the large (7–8 cm) sphagnum linear growth, they can restore very quickly (2 years after the anthropogenic load removal). In the least humid (water level 40–60 cm) bog communities (pine phytocenoses, shrub-sphagnum lightly forested ridges), shrubs dominate, which recover rather slowly after trampling. In general, the sphagnum mosses growth is insignificant, which does not contribute to the rapid moss cover restoration.

Belarusian peat bogs support a very high proportion of specialized inhabitants (tyrphobionts and tyrpophiles) (Sushko, 2017, 2019b, 2021, 2022). Such species are cold-adapted and range mainly in northern latitudes (Spitzer and Danks, 2006). High specialization of the peat bog plant and primarily invertebrate animals has increased the risk of extinction in the case of loss of their habitats. Our findings revealed a significant positive response of many herbivorous insect species richness and abundance to the number of host plants and density of plant cover. Changing of these environment variables can have a negative impact on the peat bog specialized insects due to human activities and climate change. Most of the large peatlands in Belarus are protected areas. However, they are also at risk from agricultural activities in neighboring territories. In particular, drainage can contribute to disturbance of hydrological conditions on adjacent peatlands, especially the lagg zones, which are characterized by constant high moisture. As a result, hydrophilic plants, such as *Andromeda poifolia*, *Oxycoccus palustris,* and *Chamaedaphne calyculata*, which are the main food for imagoes of *Coenonympha hero* and the Red Listed species *Oeneis jutta*, may disappear. It is therefore necessary to maintain stable hydrological conditions not only in bogs but also in adjacent farmlands. In addition, there is an urgent need to block drainage ditches to create conditions suitable for peatland vegetation. Another threat to the plants of the peat bog edges should be highlighted. They are strongly affected by the application of agrichemicals on adjacent fields to loss of oligotrophic neighbor peat bog habitats and associated plant communities as well. It should be noted that the lagg zones were characterized by the highest butterfly diversity compared to pine bogs and open bogs. Hence, preserving lagg zones is of primary importance for the conservation of *Coenonympha hero* and *Oeneis jutta*, and for the entire butterfly assemblage as a whole (Sushko, 2022).

On the other hand, our studies revealed that for other specialized peat bog insects, including protected species,

the presence of dwarf shrub-covered habitats, such as sites on slope and on the dome of the bog, is also important, since these habitats provide wider food resources for many insect species than the lagg zones.

Studies of the vegetation composition of peat bogs damaged by human activity, and in particular due to fires, have reported a significant increase in heather proportion in their plant cover and displacement of other dwarf shrub species (Zeliankevich et al., 2016). A decrease in the coverage of shrubs in peat bogs that bloom in May, June, and July, especially wild rosemary, can reduce the amount of food resources for the nectarophilous insects, such as many bumblebees and butterflies, which are active in spring and in the first half of summer, as well as the diversity of food resources for other herbivorous insects is noticeably reduced. In addition, peat bog plant species richness and dwarf shrub cover were important variables affected by the diversity of beetles, bugs, planthoppers, and leafhoppers (Sushko, 2016a,b, 2017, 2018).

6 Operational and potential monitoring tools to conserve or recover the biome

In Belarus, an approach based on the environmental assessment of vegetation cover, remote sensing data, and geoinformatics has been developed for monitoring bogs, including raised bogs (Grummo et al., 2023). The ecological state data of the vegetation cover are used as the bog state indicator. The assessment of vegetation cover state is carried out by establishing the disturbance (preservation) degree. In this case, the various changes in the species and coenotic composition and structure of phytocenoses are taken into account. The disturbance stages are determined by the deviation of the plant communities' composition and structure from their natural state or from the selected state at the certain point.

For bog monitoring, it is promising to use a cartographic method that gives a spatial representation of the degree and scale of vegetation and ecosystems degradation, as a result of the various factors' influence. In the concept of these studies, the basis for analysis and forecast is an up-to-date geobotanical map. Based on the vegetation map and associated databases, using the geographic information system (GIS) capabilities, derivative thematic cartographic products have been created that reflect the current state and forecast of the vegetation biodiversity dynamic in the bog ecosystems (Fig. 35.9).

The data from the online resource "Database of Belarus peatlands" (http://peatlands.by) were also used.

All spatial data operations were performed using specialized software packages (ENVI, ArcGIS, QGIS) as well as the Google Earth Engine cloud computing platform.

6.1 Forecast of the bog ecosystem dynamic due to the climate changes (until 2050)

Note that predicting the climate change consequences for bog's ecosystems is very difficult. Paleoecological data generally indicate a relatively high resistance of bogs to past climate changes. However, modern climate changes may differ significantly from past ones in speed and characteristics. At the same time, paleoecology informs about long time intervals, especially from the human life point of view. It is difficult to assess the consequences of the extreme events increasing frequency and climate instability. If we consider the Holocene small climatic optimum as the closest future climatic scenario, most of the bogs in Eurasia (including Belarus) will not be able to support active growth during climate warming (Fig. 35.10).

The temperature changes direct impact will not significantly affect the bogs structure and condition. In humid conditions, a specific microclimate protects against these influences; in less watered conditions, peat exhibits its thermal insulating properties. The main impact mechanism during climate change is the water regime. The most likely response to climate warming is a decrease in the bog water level. Accordingly, most forecast models are based on the already known reactions of bog ecosystems to the drainage consequences.

The research methodology sequence consisted of 4 stages.

(1) *Derivative satellite, thematic and cartographic products preparation.* A large-scale (M 1: 100,000) modern vegetation map was used as a base for developing forecast. Maps of the vegetation cover dynamics and state, map of the disturbed bogs location served as auxiliary material for the analysis. The studies also

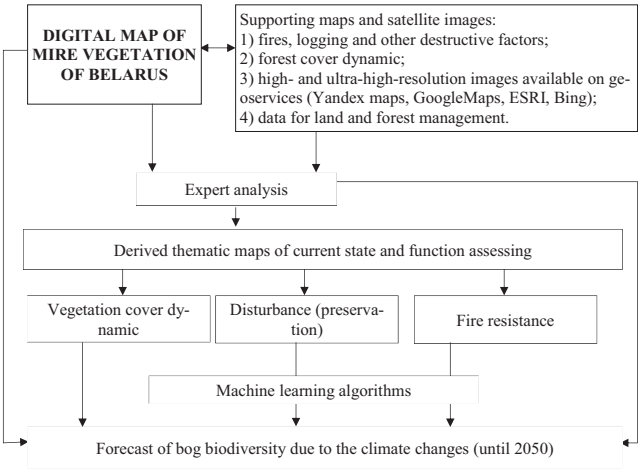

FIG. 35.10 Algorithm for the bog ecosystems state and dynamic studying in Belarus. *(Credit: Dmitry Grummo, Natallia Zeliankevich.)*

used forecast scenarios of climate change for the near future (until 2050) in Belarus.

(2) *Model forecast for the development of bog vegetation biodiversity under climate change*. The dynamics model was based on the modern map of mire vegetation, vegetation successional series under climate change and drainage, existing models and expert assessments of forecasting the ecosystem responses to climate change, and forecasts of climate change in Belarus.

(3) *Combination of analytical satellite, thematic and cartographic products on the GIS platform, drawing up the potential vegetation map (until 2050).*

(4) *Statistical processing using GIS tools, forecasting threats to biological diversity, analysis of the obtained results, drawing findings, and conclusions.*

The main threats to the bog biodiversity associated with climate change include: (a) changes in the spatial distribution bog boundaries (area reduction of boreal-type bogs, floodplain mires) and bog diversity (spatial structure simplification, biotopic diversity reduction, species composition changes, rare species disappearance); (b) bog water level lowering; (c) overgrowing of nonforested ecological bog systems by trees and shrubs; and (d) peat fires.

The predicted changes in biodiversity in the bog ecosystem of Belarus will be as follows:

1. A reduction in the area of bogs most dependent on climate—Nonforested taiga raised bogs, floodplain mires.

   It is predicted that by 2050, under the current climate scenario, the area of the nonforested taiga raised bogs of the FUSCUM type will be reduced by 56.7 thousand hectares (minus 49.2% of the currently occupied area by these bogs). Their distribution will be localized to the northern regions of the country.

   The main dynamics directions of the FUSCUM-type raised bogs will proceed in the following directions: (a) forest bog type formation, (b) main edificator change of the bog vegetation cover (*Sphagnum fuscum* to *Sphagnum divinum*) and, accordingly, replacement of FUSCUM-type bogs with DIVINUM-type, and (c) ridge-hollow and ridge-hollow-lake complexes degradation.

   A reduction in the area of nonforested floodplain mires is also predicted (by 60.7%), primarily due to changes in the rivers' hydrological regime (a sharp reduction in winter water accumulation and spring floods, anomalous summer low water).

2. The biotopic (ecosystem) diversity of bogs is expected to decrease, leading to a simplification of their internal spatial structure due to:
   - Reducing the spread of stable nonforest bogs by 3.5 times.
   - Forest growing in the bogs (+4.3% of the total bog area) and shrubs active growing (+19.8%).
   - Degradation and destruction of open water bodies, swampy water (−0.3%).
   - Natural restoration processes reduction in the degraded bogs with disturbed vegetation (−8.7%).
3. The total bog area affected by climate change is 634.3 thousand hectares or 57.5%.
4. By 2050, it is predicted that stable nonforested bogs share will be about 5% of the total area, and bog forests and shrubs will absolutely dominate phytocenotic and ecosystem bog diversity, making up 80.0% (Fig. 35.11).

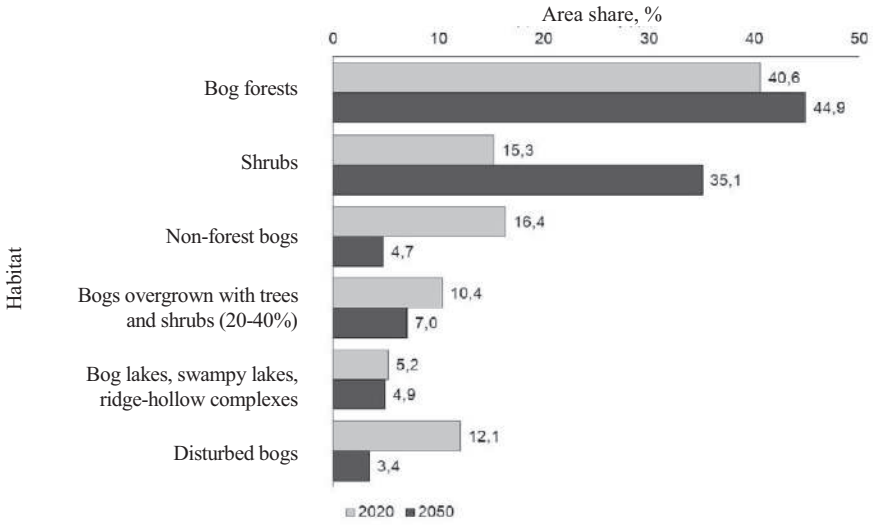

FIG. 35.11 Current (as of 2020) and forecast (as of 2050) structure of the biotopic diversity of Belarus bogs. *(Photo credits: Dmitry Grummo, Natallia Zeliankevich.)*

5. Changes in the bogs flora will be controlled not directly by climate, but through the suitable habitat formation, i.e., the presence or absence of a particular bog type. It is the state of the bogs and their internal spatial structure that will determine the species diversity and largely the associated bog fauna. The following groups of plants will have the greatest direct extinction threat (or significant distribution area reduction):
   - *Empetrum nigrum* group combines oligotrophic species characteristic of high hummocks and ridges: *Empetrum nigrum*, *Oxycoccus microcarpus*, *Rubus chamaemorus*, and *Sphagnum fuscum*.
   - The *Rhynchospora alba* group combines *Rhynchospora alba*, *Scheuchzeria palustris*, *Carex limosa*, *Drosera anglica*, *Sphagnum cuspidatum*, *Sphagnum* balticum, *Sphagnum* majus, Cladopodiella fluians, *Gymnocolea inflata*, etc. The group is typical for highly watered stagnant hollows of oligotrophic bogs.

   Species with limited adaptive capacity, highly specialized, and rare species can disappear as a climate change result and changes in the bog spatial organization. In particular, it is expected:
   - Disappearance and sharp reduction in the protected plant species distribution: *Carex buxbaumii* I (CR), *Betula nana* II (EN), *Rubus chamaemorus* II (EN), *Carex pauciflora* III (VU), *Carex paupercula* III (VU).
   - Endangered (significantly reduced spread): Carex davalliana I (CR), *Nuphar pumila* II (EN), *Carex capillaris* II (EN), *Listera cordata* II (EN), *Baeothryon alpinum* III (VU), Eriophorumgracile III (VU), Salix myrtilloides III (VU), Oxycoccus microcarpus IV (NT), *Lycopodiella inundata* IV (NT), *Drosera anglica* (LC).

     At the same time, climate change may have a positive impact on distribution increasing for a number of species included in the Belarus Red Book: *Coeloglossum viride* II (EN), *Liparis loeselii* II (EN), *Sphagnum mole* III (VU), *Viola uliginosa* IV (NT), *Dactylorhiza fuchsii* (LC), *Utricularia minor* (LC).
6. Bog and wetland fires may become more frequent. The area of bogs with a high fires potential under unfavorable weather and climatic conditions is 221.4 thousand hectares.

7 Conclusions and perspectives

The ecological conditions of European peat bogs are similar to the habitats of the tundra biome. They include many cold-adapted boreal and subarctic plant and animal species. This is especially important for the conservation of the unique postglacial biota in light of recent climate changes. We demonstrated biodiversity features focusing on large taxa such as plants and insects. The presented studies showed that 86 species of vascular plants and 48 species of bryophytes, as well as 1384 species of insects (Ectognatha), were recorded on peat bogs in Belarus. The diversity of these taxa was low and ranged according to the Shannon index from 0.88 to 2.20 for plants and from 0.77 to 1.97 for insects. This diversity is due to the high proportion of several highly specialized species in each taxon. Specialized peat bog species, especially stenotopic ones (tyrphobionts), cannot inhabit other ecosystems, so they require special attention. In Belarus, the area of peat bogs is 314.5 thousand hectares, 53.1% of which have been preserved in an almost intact state. Most of them are protected (239.6 thousand hectares). In addition, their environmental condition is regularly monitored. As a consequence, Belarusian peat bogs are not only refugia for cold-adapted flora and fauna in the temperate zone of Europe, but also they are very suitable model systems for analyzing the effects of environmental changes on plant communities and animal assemblages of pristine European tundra patches in a warming climate time.

Acknowledgments

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Chapter 36

North American tundras: Imperiled landscapes at a continent's latitudinal and altitudinal extremes

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1 Introduction

Tundra ecosystems cap North America's northern extremes ("Arctic tundra") and highest peaks ("alpine tundra"): places known for their harsh winters and ephemeral summers. Tundras are landscapes of short-stature, cold-adapted plants that persist beyond the tree line (e.g., Bjorkman et al., 2018; Bliss, 1971). Despite their relatively low productivity compared to other biomes, the highly seasonal nature of these environments supports rich biodiversity, typified by seas of vibrant wildflowers and vast congregations of migratory animals. Arctic and alpine regions are bellwethers of climate change, and temperature and precipitation regimes are changing at a faster pace than in systems at lower latitudes and elevations (Pepin et al., 2015; Post et al., 2018). It is clear that these areas will undergo great change in the coming century, and imperative that conservation efforts keep pace with, and anticipate, forecastable transformations of tundra ecosystems.

Tundras are characterized by extensive and prolonged periods of snow cover, brief growing seasons, and intense solar radiation. From Greenland and Canada to Mexico (23.4 million km^2; Fig. 36.1), Arctic tundra spans 4.27 million km^2 (Olson et al., 2001), and alpine tundra covers an additional 0.49 million km^2 of mountainous terrain at lower latitudes (Testolin et al., 2020), collectively representing about 20% of the countries' combined land area. About 37% of North American tundra is currently conserved in a formal capacity (UNEP-WCMW, IUCN, 2024).

Although a gamut of underlying physical, biogeochemical, and limiting factors control Arctic and alpine tundras (Bilbrough et al., 2000; Nemergut et al., 2005), the macroecological picture is typically quite similar across tundra systems: few large- and mesopredators, subsisting on a small diversity of birds, ungulates, and small mammals, foraging on a brief eruption of invertebrates and vegetation. Interrelationships among landscape topography, permafrost, snow, and vegetation mediate connections between climate and the animal communities that are found there, so complex seasonal variation is an important facet of tundras (John et al., 2020; Kelsey et al., 2023; Pinzner et al., 2024).

Tundras are productive during the summer and if not for their harsh winters, more animals might remain there year-round as residents. Instead, tundras are a common seasonal destination for migratory animals. Many migratory species breed on tundra landscapes, reinforcing connections between tundras and other biomes of North America—and more broadly, the Americas as a whole. Migratory species transport biomass and nutrients to characteristically nutrient-limited tundra landscapes (Bauer and Hoye, 2014; Giroux et al., 2012), and are nutritionally, economically, and culturally important to people who live there (Henri et al., 2020; Wolfe et al., 2000).

Tundra ecosystems provide a range of services to humans, from aesthetic value to carbon sequestration, forage production for livestock, and water supply (Halofsky et al., 2017; O'Garra, 2017; Runge et al., 2020; Turner et al., 2021). Estimates of carbon storage by permafrost exceed a thousand Pg globally, but this service is threatened by the melting Arctic (Povirk et al., 2000; Schuur and Mack, 2018; Tarnocai et al., 2009). Mountain peaks serve as "water towers" of humanity, providing drinking water to over a billion people globally, but this service is similarly sensitive to ongoing and future climate change (Immerzeel et al., 2020; Viviroli et al., 2007). Across North America, tundras, and the ecosystem services they provide, are threatened by both direct and indirect impacts of humans. In this chapter, we will explore where, how, and why tundra ecosystems of North America face

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change, and discuss ongoing conservation efforts to protect these last relics of Earth's colder past.

2 Geographic location

Tundras can be found across North America, from the Canadian High Arctic to the Trans-Mexican Volcanic Belt (TMVB) (Fig. 36.1). At the northern extreme, tundra is broadly distributed and highly contiguous. As Arctic tundra transitions into alpine tundra further south, patches become smaller and more isolated. The geographic isolation and climatic differences among temperate alpine tundras in North America are central to the variability in plant community diversity found there (Malanson et al., 2015).

Arctic tundra is common at North America's highest latitudes, connected in vast swaths ranging from western Alaska to eastern Canada, and also found in more isolated parcels among the Canadian Arctic Archipelago and Greenland. The lowest-latitude Arctic tundra is found on Sanikiluaq (the Belcher Islands) in the Hudson Bay. Continental low-latitude Arctic tundra sites in North America are generally mountainous and overlap with the highest-latitude sites that might be called alpine tundra, including the Torngat Mountains on the Labrador Peninsula and the Coast Mountains of British Columbia and Alaska.

Alpine floristic zones are scattered across temperate North America, found mostly within the American Cordillera, and extending as far south as the highest peaks of the zacatonal alpino region of central Mexico and Guatemala (Almeida-Lenero et al., 2004; Mastretta-Yanes et al., 2015; Steinmann et al., 2021). The Cordillera includes such prominent peaks as Mt. Whitney in the Sierra Nevada Mountains of California, Mt. Elbert of the Colorado Rockies, and Mt. Robson of the Canadian Rockies, each of which approaches or exceeds 4000 m elevation. In the zacatonal region of central Mexico, the TMVB is capped by Popocatépetl and Citlaltépetl, both of which reach over 5000 m elevation (Almeida-Lenero et al., 1994). Although colloquially known as the "Sierra Nevada," I refer to this range only as the TMVB in this chapter, reserving "the Sierra" and similar monikers for the homonymic mountain range in California.

A few pockets of alpine tundra can be found outside the American Cordillera, mostly in the northeast United States and southeast Canada in the Appalachian Mountains, Canadian Shield, and Arctic Cordillera (Jones and Willey, 2012). Five peaks in the Presidential Range (New Hampshire, USA) including the noted Mount Washington and Mount Adams exceed 1500 m elevation, and all rise above tree line. In the Chic-Choc Mountains (Quebec, Canada), Mont Jacques-Cartier and Mont Albert exceed 1000 m in elevation and also reach the alpine zone. The Torngat Mountains (Quebec, and Newfoundland and Labrador, Canada) lie at the southern extent of the Arctic Cordillera and are the northernmost of the eastern montane tundras, though they might also be considered low-Arctic tundra (see above). Notably, because of the eastern positioning of each of these three ranges, they face different climatic regimes, human threats, and conservation issues than alpine ecosystems found in the west.

3 Biodiversity

3.1 Flora

At their "lowest" reaches (in the sense of both elevation and latitude) tundras are typically abutted by evergreen tree communities, including pines (*Pinus* spp.), junipers (*Juniperus* spp.), and spruces (*Picea* spp.); and also occasionally deciduous trees including birches (*Betula* spp.), aspens (*Populus* spp.), and willows (*Salix* spp.) (Astudillo-Sánchez et al., 2019; Dufour-Tremblay et al., 2012; Jiménez-Noriega et al., 2017; Smithers et al., 2018). Smaller, prostrate species from these genera can also be found as shrubs beyond the tree line (Bret-Harte et al., 2001; Dumais et al., 2014; Ortiz et al., 2002). Krummholz—a community of stunted trees—is characteristic of this transition zone between subalpine forest or boreal forest and tundra (Klasner and Fagre, 2002; Millar et al., 2020; Payette and Delwaide, 1994; Resler and Tomback, 2008).

Graminoids and forbs represent an increasing proportion of the landscape vegetation biomass at higher elevations and latitudes, in meadows and cushion tussock tundra. Species found here are typically short, hardy, and densely growing, able to tolerate strong winds and cold temperatures (Billings, 1974; Billings and Mooney, 1968; Kimball et al., 2021). Cosmopolitan herbaceous tundra plants include Moss Campion (*Silene acaulis*), knotweeds (*Polygonum* spp.), *Phlox* spp., fleabanes (*Erigeron* spp.), sedges (*Carex* spp.), cottongrass (*Eriophorum* spp.), fescue (*Festuca* spp.), and bluegrass (*Poa* spp.) (Cwynar and Ritchie, 1980; DeMarche et al., 2018; Hobbie et al., 1999; Jones and Henry, 2003; McKendrick, 1987). North America has an impressive assortment of herbaceous tundra plants, with distinct biodiversity hotspots ranging from the towering peaks of the TMVB (Villaseñor et al., 2006), to the alpine reaches of the California Floristic Province (Lessing, 2022; Rundel, 2011), and the Arctic National Wildlife Refuge in Alaska and Bathurst Inlet in Canada (Walker et al., 2016). But, tundra plant diversity is imperiled: in the Colorado Rockies, alpine plant species composition has shifted in response to rising summer temperatures over the last 30 years (Oldfather et al., 2023). Further east, snowbank communities in the Presidential Range of New Hampshire are listed as "critically imperiled" by the state (Sperduto and Nichols, 2012).

Ferns and mosses, central to decomposition, community succession, and the hydrologic cycle, can be found across the spectrum of North American tundra landscapes, but

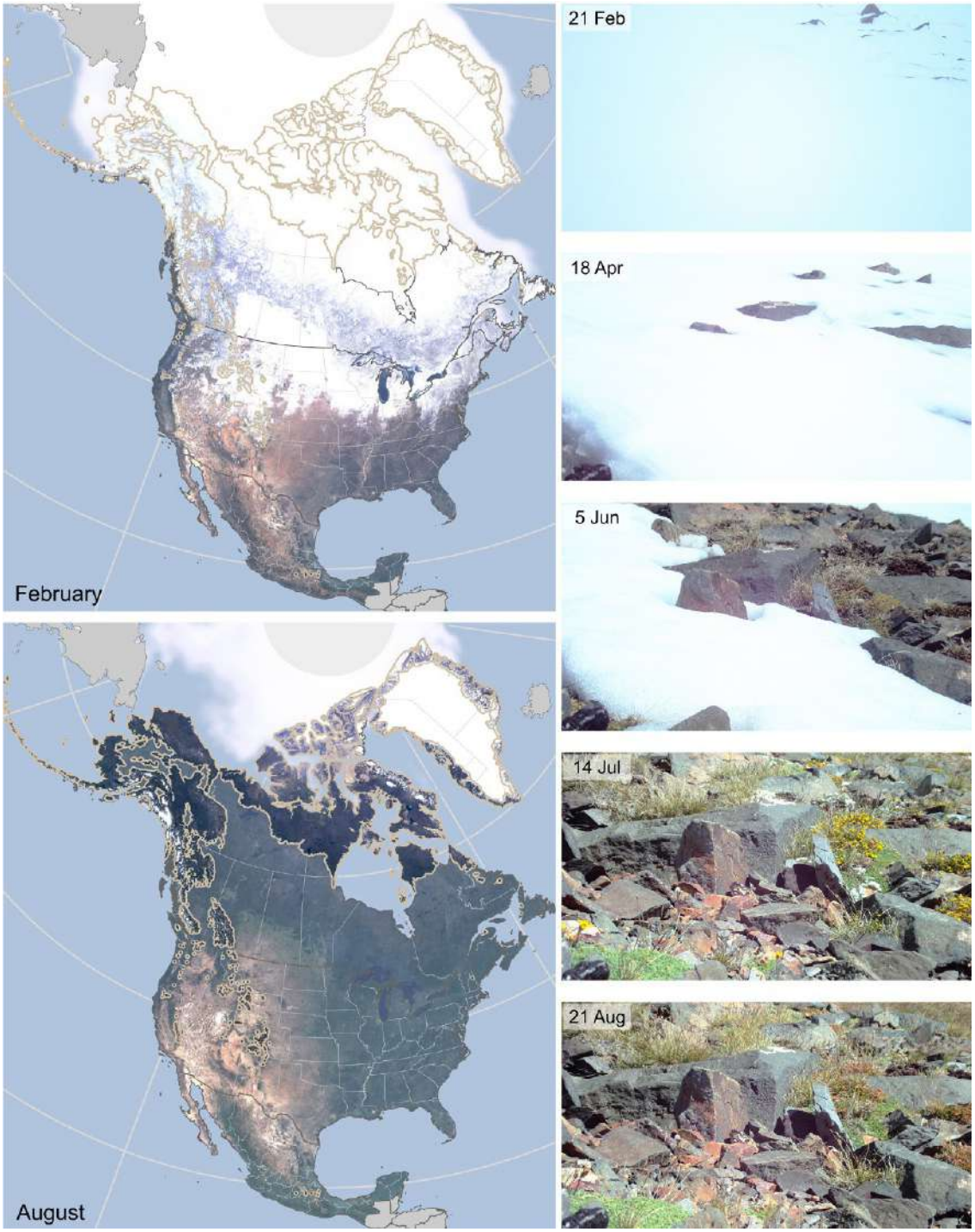

FIG. 36.1 Tundras of North America (left, highlighted in beige) are found at high latitudes and elevations, with decreasing density and contiguity further south. Snow is nearly ubiquitous on the tundra and its neighboring biomes in winter (top), but has generally dissipated by summer and is replaced by lush vegetative growth (bottom). Herbaceous and graminoid-dominated tundra plant communities persist under extensive snow cover and exposure to extreme cold and wind, with a rapid and vibrant spring culminating in a brief summer growing season (right). RGB imagery was compiled from Landsat-8 (2013−23) using a median image reducer in Google Earth Engine (Gorelick et al., 2017). Arctic sea ice index (shown in white) data from NSIDC (Fetterer et al., 2017). Time-lapse imagery compiled from the stationary camera on Mt. Gibbs, California in 2019 (John and Post, 2021). Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Graphic credits: The author.)*

with increasingly obvious abundance in comparatively barren snowbeds and fell-fields (Bueno et al., 2016; Hernández-Cárdenas et al., 2019; Leavitt et al., 2021; Roland et al., 2017; Villaseñor et al., 2006). Lichens are widespread yet declining across the region; they are an important food resource for herbivores during winter and loss of lichen may have important cascading conservation significance (Joly et al., 2010; Joly et al., 2009; Rickbeil et al., 2017).

3.2 Fauna

A highly seasonal invertebrate community undergoes population cycles linked to rising temperatures and access to resources during the North American tundra's fleeting spring. A by-product of their ubiquity and the sparse vegetation, wolf spiders are easy to find on tundra landscapes (Culler et al., 2021; Koponen, 1987; Schmoller, 1970). Mosquitos are also abundant, especially in moist Arctic tundra, and stand to emerge earlier, grow larger, and live longer due to climate change (Koltz and Culler, 2021). Although ants are a nearly global taxon and persist in both the Arctic and alpine tundras of North America, they are not found in Greenland (Böcher, 2015; Francoeur, 1997; Sanders, 2002).

The avian community of North American tundras is generally migratory and may travel great distances to arrive on summer grounds. For example, some Arctic-breeding phalaropes migrate to waters off the coast of South America during boreal winter, although recent reports suggest that populations of these and other long-distance migratory Arctic birds may be declining (Environment and Climate Change Canada, 2023; Saalfeld et al., 2024; Smith et al., 2020). Altitudinal migration is also common among North American birds, particularly at more temperate latitudes where large elevational relief is available and nearby low-elevation habitat is tolerable during winter (Boyle, 2017; Boyle and Martin, 2015). Seasonal tundra visitors range from small insectivores and frugivores to algivorous waterfowl, to golden and white-tailed eagles. Migratory birds can impose considerable impacts on their seasonal tundra homes: for example, breeding lesser snow geese (*Anser caerulescens* syn. *Chen caerulescens*) accelerate evaporation of ephemeral tundra ponds, restricting the temporal window of suitable habitat for pond-dwelling mosquito communities (Park, 2017).

Small mammals are common in tundra systems and include pika, marmots, hares, ground squirrels, chipmunks, and deer mice (Moritz et al., 2008; Reichel, 1986). Perhaps most famous among the small tundra mammals are the collared lemmings (*Dicrostonyx* spp.), keystone herbivores known for their cyclic population dynamics (Korpimäki and Krebs, 1996; Krebs, 2011). Small mammal communities of North America's Arctic and alpine tundras are facing shrinking distributions and shifting toward novel community assemblages (Baltensperger and Huettmann, 2015). In some cases, more temperate mammals are moving north into novel Arctic territories. For example, beavers have recently colonized novel areas of both the Alaskan and Canadian Arctic (Jung et al., 2016; Tape et al., 2018).

Ungulates of North America are generally migratory and traverse latitude and elevation alike during seasonal movements between ranges. Caribou (*Rangifer tarandus*), mountain sheep (*Ovis canadensis* and *Ovis dalli*), and mountain goats (*Oreamnos americanus*) all undertake seasonal movements to maximize access to resources and avoid predation or unfavorable climatic conditions (Enns et al., 2023; Le Corre et al., 2017; Rice, 2008). Notably, migratory ungulates can be tundra transients (for example, some bighorn sheep migrate between low-elevation winter habitat in the desert to high-elevation alpine habitat in the summer), and others are permanent tundra residents (for example, some caribou are migratory but all seasonal ranges are on the tundra). Because different factors can control landscape phenology across the annual range of migratory populations (John et al., 2020), disparate shifts in climatic variables like temperature and precipitation may disrupt historic patterns of these animals' migrations (John and Post, 2022b). Like migratory birds, large herbivores have a considerable impact on the systems in which they live, reinforcing their importance to landscape-scale conservation (Post et al., 2023, 2024).

Small- and mesopredators of North American tundras include coyotes, foxes, bobcats, martens, fishers, and wolverines. Arctic foxes (*Vulpes lagopus*) are a charismatic icon of the North, and could be considered ecosystem engineers because of the fertilizing effect they have on vegetation near their dens (Gharajehdaghipour et al., 2016; Johnson-Bice et al., 2023). However, northward expansion by red foxes (*Vulpes vulpes*) threatens comparatively small Arctic foxes, which may be additionally susceptible to increasing pathogen loads under climate change (Davidson et al., 2011). Many tundra mesopredators face overall range contractions. For example, wolverines occupy just 40% of their historic range (Heim et al., 2017). Their distribution is impacted by both direct anthropogenic effects of landscape change and indirect effects mediated by disrupted snowpack.

There is a small density of large predators in North American tundra ecosystems, but bears, wolves, and even mountain lions are found across broad ranges, often undergoing seasonal movements as their prey base moves and diet preferences shift over the course of the year (Deacy et al., 2016; Dertien et al., 2019; Michelot et al., 2024). Between alpine and Arctic regions, black, brown, and polar bears (*Ursus americanus*, *Ursus arctos*, and *Ursus maritimus*, respectively) can all be found on tundras depending on the time of year. Alpine areas are important foraging,

breeding, and denning habitats for black and brown bears (Aune, 1994; Crupi et al., 2020; Schwartz et al., 1987). Bear distributions are now a fraction of their historic extent, and stand to move or shrink further under future climate change (Barrueto et al., 2023; Kurth et al., 2024). Because bears are opportunistic feeders and commonly use trash dumps as foraging grounds, they are especially prone to human-wildlife conflicts (Peirce and Daele, 2006; Sincerbox and DiGangi, 2018). Most of Earth's subpopulations of polar bears are found in North America (Von Duyke et al., 2017) and although they are pagophilic and formally considered marine mammals, they do use tundra landscapes (Durner et al., 2009). Recently, polar bears have been spending more time on land with diminishing sea ice extent and quality, further increasing the likelihood of human-bear and bear-tundra interactions (Gormezano and Rockwell, 2013a, b; Stimmelmayr et al., 2023).

3.3 Toward the south

At the southern extent of North America's alpine pockets, biodiversity in the TMVB is shaped by a long history of interactions between climate, glaciation, and volcanism (Mastretta-Yanes et al., 2015), and the area is therefore quite unique from its northern alpine counterparts. Plant communities in the highest stretches of active volcanos have been controlled by a disturbance regime of eruptions (Barba-Escoto et al., 2019), but future range contractions of alpine plants are likely under expected warming (Ramírez-Amezcua et al., 2016). Some plants could be further indirectly affected by climate change via long-term effects on ephemeral pollinators, such as the broad-tailed and blue-throated hummingbirds of central Mexico—potentially the sole pollinators of an endemic alpine paintbrush *Castilleja tolucensis* (Arredondo-Amezcua et al., 2018). Thus, future alpine vegetation change in the TMVB is likely to be complex. Conservation of this region is critical for protecting endemic and endangered species in the area: alpine grasslands of Popocatépetl are home to the endemic volcano mouse but are threatened by cattle grazing, as well as fire and forestry management practices (Velázquez et al., 2001). Also found on Popocatépetl is the endangered volcano rabbit (*Romerolagus diazi*), the second smallest rabbit in the world, endemic to just a few of the central Mexican volcanos. Volcano rabbits are similarly threatened by agricultural expansion, although their distribution is not restricted to alpine areas (Osuna et al., 2022).

4 Conservation areas and their ecological indicators

For the analyses that follow in this chapter, I include tundras that are protected under a variety of governances with biodiversity conservation as a leading directive, such as national parks, designated wilderness, protected wetlands, and Indigenous peoples' and community-conserved areas (UNEP-WCMW, IUCN, 2024). The largest single protected jurisdiction among these is Kalaallit Nunaanni nuna eqqissisimatitaq (Grønlands National Park, or Northeast Greenland National Park), with much of its protected space currently ice-covered. Other notable systems of tundra conservation in North America (Table 36.1) include especially large national and provincial parks (which are more common in the Arctic), and contiguous groups of parks that preserve broader alpine ecosystems (which are more often managed under numerous small administrative areas). Excluded from this treatment are publicly owned lands that were not established with conservation as a leading priority, such as the National Petroleum Reserve administered by the Bureau of Land Management in Alaska. However, it should be noted that conservation research coming out of these systems is incredibly valuable for understanding the range of ecological outcomes that stem from a broad spectrum of human activities arising across landscapes managed under multiple jurisdictions (Carter et al., 2020; Wilson et al., 2013).

Managers of protected areas track specific indicators through time and compare them with baseline data to evaluate the state of ecosystems (Kenney et al., 2016). For example, Kootenay National Park in the Canadian Rockies monitors introduced plant species, pika and mountain goat populations, and songbird community diversity to draw broader conclusions about the health of the alpine zone in the park (Parks Canada Agency, 2022a). In addition to its plants and wildlife, Sirmilik National Park in the Canadian High Arctic monitors the quality of its glaciers and permanent ice fields (Parks Canada Agency, 2022b). In general, ecological indicators for protected tundras of North America span both abiotic and biotic components of the ecosystem and provide insights not just on current ecosystem health, but also on how tundra systems might be expected to change in the coming decades.

4.1 Abiotic indicators

Many indicators of ecosystem change across North America's tundras are abiotic (Box et al., 2019), including glacial mass balance and extent (Larsen et al., 2015, 2020; Wolken et al., 2020); sea ice extent and thickness (Meier et al., 2014; Mudryk et al., 2018; Perovich et al., 2020); permafrost continuity and temperature and active layer depth (Biskaborn et al., 2019; Smith et al., 2022); and snowpack depth and duration (Mudryk et al., 2020). Snow is an essentially unifying feature of tundras, and is a major limiting factor for plant phenology, diversity, and productivity in tundra systems (Björk and Molau, 2007;Winkler et al., 2018; Wipf and Rixen, 2010).

TABLE 36.1 A list of the largest standalone protected areas, and several contiguous systems of parks and management areas that preserve tundras of North America.

Country	Protected area	Park list	IUCN category	Area (km^2)	Proportion tundra	Latitude (°N)
Greenland	Kalaallit Nunaanni nuna eqqissisimatitaq	Northeast Greenland National Park	II	977,788	0.24	77
United States	Arctic National Wildlife Refuge	Arctic National Wildlife Refuge	IV	80,241	0.86	69
United States	Yukon Delta National Wildlife Refuge	Yukon Delta National Wildlife Refuge	IV	84,344	0.83	61
Canada	Ahiak Migratory Bird Sanctuary	Ahiak Migratory Bird Sanctuary	Ib	62,928	0.99	67
United States	Gates of the Arctic National Preserve	Gates of the Arctic National Preserve	V	62,332	0.93	68
United States	Wrangell-St. Elias National Park	Wrangell-St. Elias National Park	II	49,532	0.97	61
United States	Noatak National Preserve	Noatak National Preserve	V	49,791	0.99	68
Canada	Canadian Rocky Mountain Parks World Heritage Site	Banff National Park, Jasper National Park, Kootenay National Park, Yoho National Park, Hamber Provincial Park, Mount Assiniboine Provincial Park, Mount Robson Provinaial Park	II	23,581	0.95	52
Canada	Kluane National Park Reserve	Kluane National Park Reserve	II	22,061	0.97	61
Canada	Sirmilik National Park	Sirmilik National Park	II	22,176	1.00	73
United States	Sierra Nevada Mountains	Yosemite National Park, Yosemite National Park World Heritage Site, Ansel Adams Wilderness Area, Golden Trout Wilderness, John Muir Wilderness, Sequoia-Kings Canyon Wilderness Area, Yosemite Wilderness Area, Harvey Monroe Hall Research Natural Area, Kern Wild and Scenic River, Carson-Iceberg Wilderness, Emigrant Wilderness, Hoover Wilderness, Kings Canyon National Park, Hoover State Wildlife Management Area, Round Valley Reservoir, Round Valley Conservation Easement, John Krebs Wilderness Area, Carson-Iceberg Wilderness Study Area, Hoover Forest Preserve,	Ia, Ib, II, V, NA/ other	13,351	0.54	37

TABLE 36.1 A list of the largest standalone protected areas, and several contiguous systems of parks and management areas that preserve tundras of North America—cont'd

Country	Protected area	Park list	IUCN category	Area (km^2)	Proportion tundra	Latitude (°N)
		Independence Creek Easement, Owens River Headwaters Wilderness, Owens River Headwaters Wild and Scenic River, Independence Creek Wilderness Study Area, Sequoia National Park, Mono Lake Tufa State Reserve, Hoover Wildlife Area, Round Valley Wildlife Area				
United States	Absaroka Range	Line Creek Plateau Research Natural Area, Absaroka-Beartooth Wilderness, Bridger Wilderness, Fitzpatrick Wilderness, North Absaroka Wilderness, Teton Wilderness, Washakie Wilderness, Whiskey Basin Wildlife Habitat Management Area, Owl Creek Conservation Easement, Spence & Moriarity Wildlife Habitat Management Area, Owl Creek Ecological Reserve, High Lakes Wilderness Study Area, Meeteetsee Rim Easement	Ia, Ib, V, NA/other	13,288	0.59	44
United States	Mummy Range	Rocky Mountain National Park, Cache La Poudre, Colorado Wild and Scenic River, Comanche Peak Wilderness, Indian Peaks Wilderness Area, Indian Peaks State Wildlife Area, Neota Wilderness, Never Summer Wilderness, Rawah Wilderness, Bowen Gulch Protection Area, Rocky Mountain National Park Wilderness Area, Hell Canyon Research Natural Area, Pennock Creek Research Natural Area, Bowen Gulch Research Natural Area	Ia, Ib, II, V	2235	0.66	40
Mexico	Trans-Mexican Volcanic Belt	La Montaña Malinche o Matlalcuéyatl National Park, Iztaccíhuatl-Popocatépetl National Park, Pico de Orizaba National Park, Cofre de Perote o Nauhcampatépetl National Park, Santuario del Agua y Forestal Manantial El Salto de Atlautla—Ecatzingo State Park, Santuario del Agua y Forestal Manantiales Cascada Diamantes State Park, Sistema Tetzcotzingo State Reserve, Los Volcanes UNESCO-MAB Biosphere Reserve, Nevado de Toluca Flora and Fauna Protection Area	II, VI, NA/other	1955	0.54	19

Continued

TABLE 36.1 A list of the largest standalone protected areas, and several contiguous systems of parks and management areas that preserve tundras of North America—cont'd

Country	Protected area	Park list	IUCN category	Area (km^2)	Proportion tundra	Latitude (°N)
Canada	Chic-Choc Mountains	Gaspésie National Park, Caribou population de la Gaspésie Habitat of a Threatened or Vulnerable Wildlife Species, Serpentine-du-Mont-Albert Habitat of a Threatened or Vulnerable Plant Species, Chic-Chocs Territorial Reserve, Corridor Mont-Saint-Pierre Territorial Reserve	Ia, II, VI	1061	0.54	49
United States	Presidential Range	Great Gulf Wilderness, Presidential Range-Dry River Wilderness, Mount Washington State Park	Ib, III	135	0.39	44

Data from the World Database of Protected Areas (UNEP-WCMW, IUCN, 2024). Proportion tundra is calculated using an area of overlap with Arctic tundra and alpine tundra, defined as areas above the tree line (Olson et al., 2001; Testolin et al., 2020). Latitude was summarized using the latitude of the protected area centroid.

North American tundras face a variable and changing snow regime (Brown et al., 2010; Mote et al., 2005; Pederson et al., 2011). An analysis of MOD10A1 Normalized Snow Difference Index (NDSI) time series at several large protected Arctic and alpine tundra sites (Fig. 36.2) reveals that the end of the snowy season has advanced by 1.33 days/decade between 2003 and 2023 in protected Arctic ecosystems and by 1.92 days/decade in protected alpine ecosystems. Although neither of these trends was statistically significant, interannual variability in snowmelt timing was considerable, especially for temperate alpine tundras. Snowmelt was generally earlier at lower latitudes, and protected areas with the earliest snowmelt timing had the longest snow-free season. Notably, even the coarse-scale measure of snowmelt timing employed here correlates strongly with in situ outcomes, such as the annual snow-limited opening of Tioga Pass on the east site of Yosemite National Park (and see also Jenkins et al., 2023a).

Yosemite and its neighboring protected areas spanning the Sierra Nevada Mountains are home to Sierra Nevada bighorn sheep (*O. canadensis sierrae*), an endangered subspecies of bighorn sheep endemic to California and an iconic ungulate of the High Sierra (US Fish and Wildlife Service, 2007). Bighorn sheep specialize in rugged, open landscapes and gain most of their fat for the year on grasses, forbs, and shrubs when plants are productive during the summer (Denryter et al., 2022; Smiley et al., 2022). Because the vast majority of the Sierra's total annual snowpack is accumulated over just a few large precipitation events, year-to-year variability in snow volume is much greater than in other areas of the United States (Dettinger et al., 2011). The timing of forage availability in the Sierra is strongly associated with snowmelt timing, especially in the alpine zone where snow accumulation is most reliable from year to year and snowpack remains the latest into the year (John et al., 2023). Elevational lapses in the timing of snowmelt related to reliable climatic patterns across topography drive concomitant delays in plant growth (Gagliano et al., 2023; John et al., 2023), providing an uphill gradient along which the mobile consumers can maximize forage intake (Albon and Langvatn, 1992). As a result, the migration timing of bighorn sheep is closely tied to snowpack and plant growth dynamics across the Sierra (John, 2022; John et al., 2024). Long-term change in the Sierra's precipitation regime is likely to be echoed in the dynamics of plants and herbivores that are reliant on snow's insulating, moisture provisioning, or forage scheduling properties.

4.2 Biotic indicators

Indicators of biotic change in North American tundras can be found at the level of the individual, the ecosystem, and everywhere in between. Tundras are characteristically highly seasonal and therefore the timing of life history events—and synchrony in the occurrence of interrelated seasonal life history traits among species—is critical to the ecology of these environments (Post, 2019). Earlier snowmelt, warmer spring temperatures, or modified soil biogeochemistry could alleviate factors that otherwise limit early-season plant growth (Inouye, 2020; Wheeler et al., 2015). Yet, species' ability to temporally respond to ongoing and projected change may be nearing its limits (Iler et al., 2013), so although phenology is a useful indicator of biotic response to climate change, it may not reflect

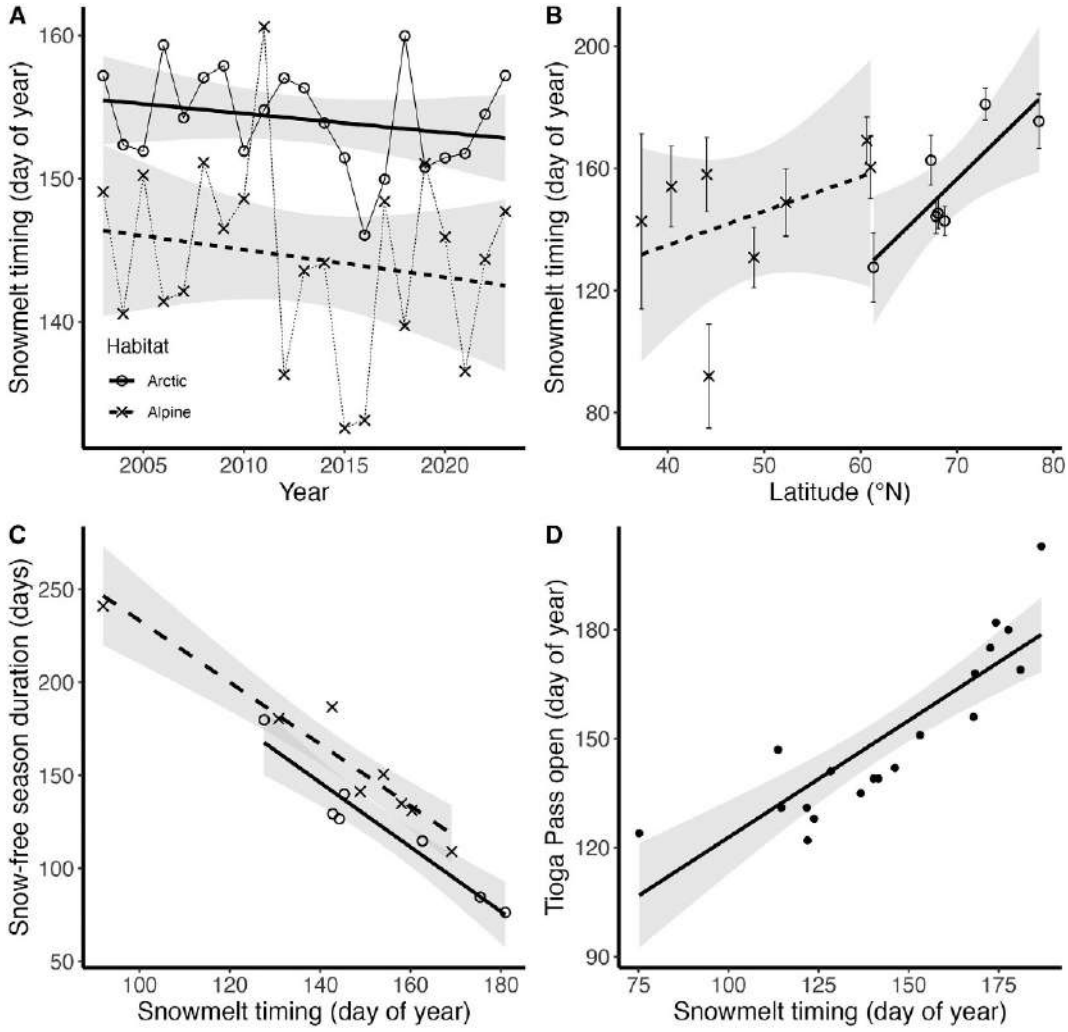

FIG. 36.2 Recent trends in snowmelt timing across large protected swaths of North American tundras. Daily mean MOD10A1 NDSI was summarized for each region described in Table 1.1 except the TMVB using Google Earth Engine (Gorelick et al., 2017). Snowpack phenology was estimated for each protected area in each year using a double-logistic function with parameters for the midpoint of snowmelt, the rate of snowmelt, the midpoint of snow accumulation, and the rate of snow accumulation (Beck et al., 2006; John et al., 2023). Double logistic curves were fit using the R package DEoptim (Mullen et al., 2011; R Core Team, 2019). The duration of the snow-free season was indexed using the time difference between the midpoint of the snow accumulation and the midpoint of the snowmelt. Arctic and alpine sites are differentiated using O's and X's, respectively (A–C). Over the course of the two-decade time series (2003−23), the mean date of snowmelt for large protected Arctic and alpine tundras in North America was variable but generally advancing (A). Summarized across years, snowmelt timing was generally later in Arctic sites than in alpine sites and was delayed at higher latitudes (B). The duration of the snow-free season was generally greater in sites with earlier snowmelt timing (C) but was also positively related to the timing of the snow accumulation season (not shown). The opening date of Tioga Pass (the east entrance of Yosemite National Park in California; data from https://www.nps.gov/yose/planyourvisit/tiogaopen.htm) is highly correlated with the mean timing of snowmelt across the Sierra Nevada alpine region (D). *(Graphic credits: The author.)*

the full extent to which a local environment is changing (Wadgymar et al., 2018).

Woody encroachment is a community-level transition away from open tundra and an indicator of complex, landscape-scale ecological response to climate change (Chapin III et al., 1995; Formica et al., 2014; Terskaia et al., 2020). Shrubs and trees, and consequentially woody encroachment, have cascading impacts on ecosystem processes including soil temperature regulation, nutrient cycling, and snow accumulation (Aguirre et al., 2021; McLaren et al., 2017; Mekonnen et al., 2021; Pomeroy et al., 2006; Sturm et al., 2001). Novel woody communities develop along open tundra peripheries as shrub- and tree lines advance over latitudinal and/or topographic gradients, potentially affecting Arctic and alpine tundras alike (Myers-Smith et al., 2011). Across the northern hemisphere, the tree line has moved uphill at a rate of over 3.5 m per decade, with the fastest advance in subarctic alpine systems (Lu et al., 2021). Thus, shifting patterns of vegetation communities are useful for tracking ecological response to climate change, but also of

ubiquitous concern across North America's tundra regions (Capers et al., 2013; Kellner et al., 2023; Lantz et al., 2010; Tremblay et al., 2012).

Wildlife health is of considerable interest to managers of protected areas, and consistent evaluations of demographics, movements, and body condition reveal how populations respond to change and inform future management decisions. The physical condition of individual animals is a common indicator of wildlife health. For instance, body fat (Rode et al., 2024; Stephenson et al., 2020), horn size (Karabatsos et al., 2023), disease (Clifford et al., 2009; Malmberg et al., 2024), and parasite load (Aguilar et al., 2023; Kutz et al., 2013) are all useful for assessing ecological and evolutionary impacts of anthropogenic stressors on large ungulates. At the same time, many of these same indicators are also useful for developing further management actions that might, for example, limit exposure of wildlife to epizootic disease from domestic livestock (Cassirer et al., 2018; Tessaro, 1986).

Unfortunately, collecting physical data on plants and animals requires expensive and time-consuming field campaigns, made especially difficult by the often rugged and inaccessible tundra terrain. Thus, coarser-scale and sometimes remote measurements are useful for examining outcomes of climate change and human impacts that may be difficult to collect at the individual level. For example, regular censusing enables inference about wildlife population vital rates without requiring direct contact with animals (Johnson et al., 2010; Post and Forchhammer, 2008). An array of remote sensing measurements provide insights into vegetative change at scales spanning meters to hundreds of kilometers: products such as the Normalized Difference Vegetation Index (NDVI), Enhanced Vegetation Index (EVI), and Solar-Induced Chlorophyll Fluorescence (SIF) are often used as proxies of vegetation biomass or productivity (Box et al., 2019; Jespersen et al., 2023; Luus et al., 2017).

5 Human activities and their impacts

A major conservation challenge spanning the North American Arctic is the management of declining caribou populations. Caribou (or reindeer, *R. tarandus*) are a cultural keystone species of great importance to First Nations and Indigenous peoples across North America (Borish et al., 2021). They could also be considered an umbrella species, as their conservation has broader significance for other cooccurring species. Caribou—and tundra species more broadly—are threatened by both direct and indirect human impacts: direct effects (such as the development of roads and mining projects) cause degradation of habitat quality or outright loss of habitat; and indirect effects (such as modified primary productivity dynamics in response to anthropogenic climate change) impact reliability and quality of resources, predation pressure, and disease and insect harassment.

5.1 Direct effects

Development poses one of the most immediate and direct threats to caribou (Festa-Bianchet et al., 2011). Development projects have a strong negative effect on caribou site occupancy, and animals typically avoid roads, mines, and other activities associated with natural resource extraction (Boulanger et al., 2021; Johnson et al., 2005; Wolfe et al., 2000). Notably, caribou avoid crossing mining roads when they encounter traffic (Severson et al., 2023; Smith and Johnson, 2023), and even management of seasonal road activity only partially mitigates the effect of mining roads on caribou migration (Boulanger et al., 2023). If a development project overlaps with parturition and/or calf-rearing habitat, access to important calving resources may be limited by a site's sudden undesirability (Johnson et al., 2020). Critically, the Arctic is becoming increasingly accessible, and given the region's rich natural resources, it is almost certain that proposals for development projects will rise as future conditions lend themselves economically favorable to reaching and extracting resources in the North (Martínez-Alier, 2023).

One such road development proposal seeks to cross federal, state, and some privately owned tundra in the southern foothills of the Brooks Range in central Alaska (Fig. 36.3A and B). This project would support gold, copper, and rare earth metal extraction from the Ambler Mining District. Each of the three proposed routes cuts through continuous mountain permafrost for almost the entirety of the road's 211 miles (approx 340 km; United States Bureau of Land Management, 2020). Regardless of the final route selected, the project is expected to lead to the loss of over 4000 acres of Western Arctic Caribou Herd habitat (United States Bureau of Land Management, 2023). Although the road itself is promised to have a narrow footprint, its development will support other mining activities in the District, about half of which lies within the Gates of the Arctic National Park and Kobuk Valley National Park. The new road is expected to exacerbate the melting of permafrost, leading to thaw slumping and necessitating future construction work as inevitable road deterioration occurs. Notably, a previously developed road stretching 80 km that connects the Red Dog Mine to a Chuchki Seaport has already been documented to impact migration by the same caribou population (Wilson et al., 2016) as well as tundra vegetation diversity in the area (Neitlich et al., 2022). After native groups were systematically excluded from initial Ambler Road planning and consultation (O'Neil, 2023), six tribal councils and a conference of chiefs came together to sue the US Bureau of Land Management under the National Environmental Policy Act arguing that the Ambler Road project would harm the way of life of local people and the local environment (Alatna Village Council et al. vs. Padgett et al., 2020). In October 2023, the United States Bureau of

Land Management issued a draft Supplemental Environmental Impact Statement for the Ambler Access Project, acknowledging that the proposed road poses a danger to local social-ecological systems (United States Bureau of Land Management, 2023). The road's exact future, and its eventual impacts on the local ecosystem, remain to be seen.

Another road development project that has already been approved and is under construction lies 4000 km to the east, near the Kangerlussuaq Fjord of Qeqqata, Greenland (Fig. 36.3C and D). This 170 km road will connect the inland village of Kangerlussuaq with the coastal port city of Sisimiut, promising to bolster local economies through tourism and export opportunities (Oxford Global Projects, 2021; Schober, 2019). It follows a similar but not overlapping path to the Arctic Circle Trail, a popular hiking route that connects the two communities. Caribou of this region spend their winters near the coast before migrating east during spring, then giving birth and rearing young near the Inland Ice in summer (Thing, 1984). Because of local geographic barriers restricting opportunities for alternative migration routes, the early-stage ATV path and the eventual completed road will cut directly through the historic migratory corridor of the caribou herd there. Outcomes of the road for caribou habitat quality and migration patterns are yet to be determined, but warrant close study as construction moves along.

5.2 Indirect effects

Indirect impacts of human activities on tundra communities are mediated through cascading effects of development and via anthropogenic climate change. The Arctic is warming far faster than the global average (Hansen et al., 2010), and alpine areas are similarly outpacing nearby lowlands in their rate of warming (Hock et al., 2019; Pepin et al., 2015). Modified temperature and precipitation regimes alter sea ice and permafrost dynamics, glacial mass balance, snow cover, and fire regimes, moderating plant growth, animal metabolism, and ecosystem exchange. While the capricious variability of precipitation makes it difficult to forecast, the future for other factors such as glacial cover is clear: the remaining ice-covered vestiges of the American West are declining. Periglacial species (and particularly glacial foreland obligates) will face dramatic habitat loss over the coming decades (Apple et al., 2022; Giersch et al., 2017; Hayes and Berger, 2023; Stibal et al., 2020).

Sea ice has considerable effects on the earth's climate system, and changes in sea ice extent have cascading importance for Arctic ecosystems (Bhatt et al., 2014; Post et al., 2013). Sea ice connects terrestrial mammal metapopulations, and connects the seasonal ranges of those that migrate between islands (Carmichael et al., 2008; Jenkins et al., 2023b; Poole et al., 2010). Indirectly, sea ice impacts terrestrial wildlife by moderating temperature and precipitation patterns in the Arctic, thereby exerting some control over vegetation dynamics (Dial et al., 2024; Post et al., 2023). Caribou reproductive success is linked to plant phenology associated with long-term patterns of sea ice variation (Kerby and Post, 2013; Post and Forchhammer, 2008). Caribou populations across Alaska, Canada, and Greenland have declined under a backdrop of concurrent sea ice decline (Fauchald et al., 2017; Post et al., 2023).

Shifts in permafrost and snowpack dynamics disrupt the quality of winter shelters for animals that hibernate (Berteaux et al., 2017). Bear dens are frequently dug into snow banks, and changing snowpack dynamics may alleviate limiting factors that prolong bear hibernation (Rickbeil et al., 2020). Recent changes in permafrost and snowpack cycles have similarly led to shifts in phenology and thermoregulation by hibernating ground squirrels (Chmura et al., 2023; Lane et al., 2012). Hoary marmots also depend on snowpack for protection during harsh winters, and their recent population decline in the North Cascades is attributed to harsh conditions during periods of snow drought (Johnston et al., 2021). Given anticipated changes in North America's alpine snowpack over the course of the next century (Siirila-Woodburn et al., 2021), hibernation propensities and schedules will likely shift, potentially complicating other life-history tradeoffs for hibernating mammals (Cordes et al., 2020; Goldberg and Conway, 2021).

Tundra fires are not currently common in North America. Although they are a natural feature of tundra systems (Higuera et al., 2011; Wein, 1976), they are rare—but they are expected to increase in frequency and intensity as the Arctic experiences warmer, drier summers and woody cover increases (Higuera et al., 2008; Hu et al., 2015; Rocha et al., 2012). Fire drives landscape-scale change in tundra ecosystems, disrupting hydrological processes, restructuring plant communities, and interfering with carbon flux (Gaglioti et al., 2021; Jones et al., 2015; Mack et al., 2011; Racine et al., 2004; Yoseph et al., 2023). Fire also impacts plant communities in alpine tundras at lower latitudes (Cansler et al., 2016; Douglas and Ballard, 1971), and may increase at high elevations under projected warming (Alizadeh et al., 2021). Yet, tundra species invest heavily in belowground growth (Iversen et al., 2015), and are quick to resprout following the fire, suggesting that tundra communities may be resilient to fire in some cases (Bret-Harte et al., 2013; Chen et al., 2022; Hewitt et al., 2013). Of course, it remains to be seen how novel climatic and biogeochemical conditions of the future Arctic will interact with fire to sway vegetation species dominance and broader ecosystem change.

6 Monitoring tools to study and conserve tundra landscapes

Indicators of tundra change are tracked from local to continental scales by community groups, governmental agencies,

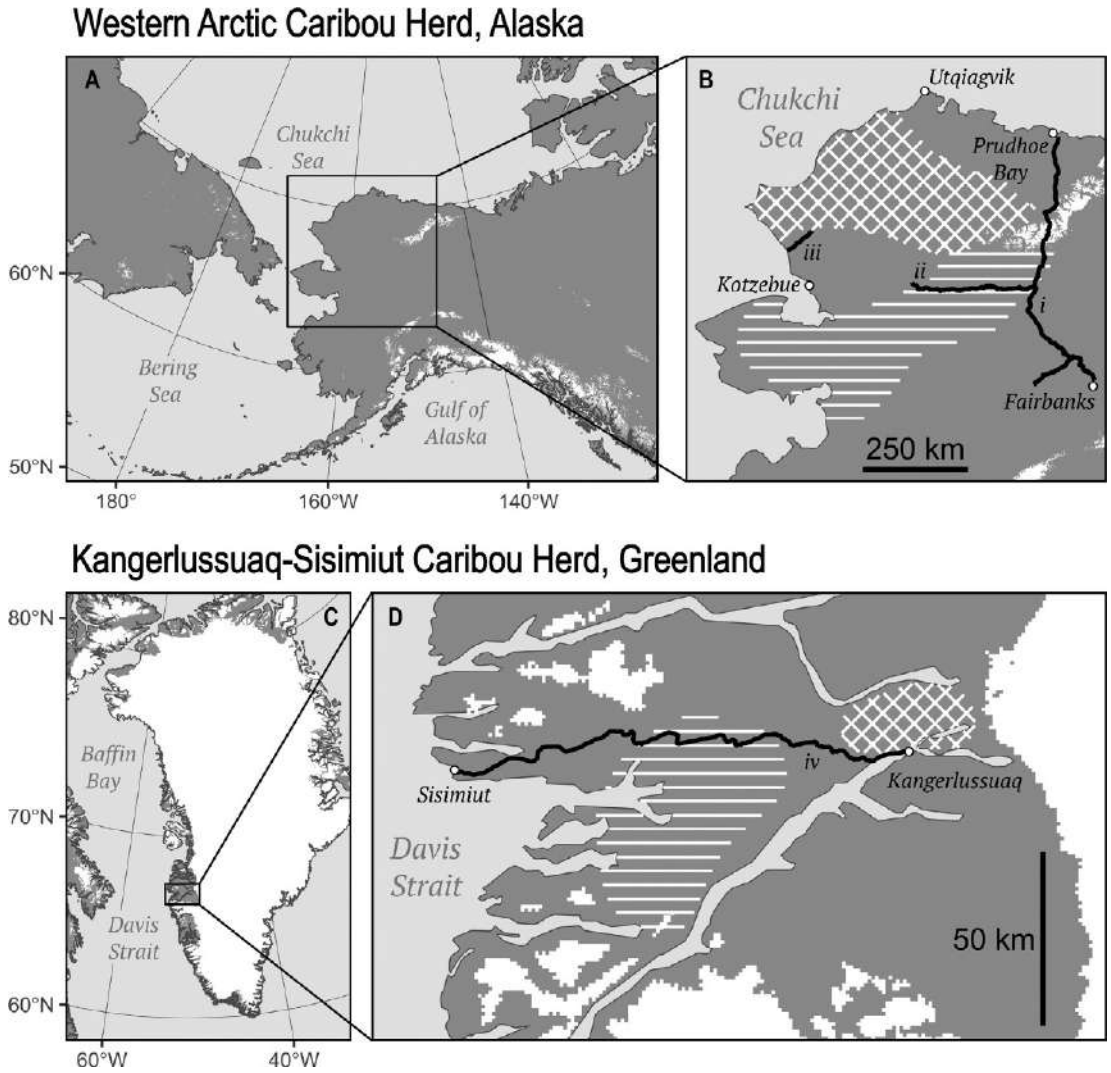

FIG. 36.3 Proposed and ongoing road development projects in the North American Arctic stand to disrupt the migration of barren-ground caribou (*R. tarandus groenlandicus*) between their winter range (horizontal bars) and summer range (cross-hatching). In Alaska (A), the Western Arctic Caribou Herd undergoes a generally south-north migration during spring (B), with many animals spending their summer on the North Slope of the Brooks Range. A proposed access road connecting the Dalton Highway (i) to the Ambler Mining District and (ii) bisects a stretch of the herd's winter range. Previous natural resource development projects have revealed that caribou in this herd are sensitive to road development and activity (e.g., the Red Dog Mine Road; iii). In southwest Greenland (C), the caribou of the Kangerlussuaq-Sisimiut Herd undertake a spring migration from their mountainous coastal winter range toward the Inland Ice (D). A road connecting Sisimiut to Kangerlussuaq is currently under construction (iv), and will effectively bisect the herd's historic migratory corridor. Vector data for country outlines from (Massicotte and South, 2024); Alaska road shapefiles from the AK Department of Transportation and the US Bureau of Land Management Supplemental Environmental Impact Statement for the Ambler Access Project. Approximate route for Sisimiut-Kangerlussuaq road from Greenland's National Museum and Archives (Harmsen and Myrup, 2017). Snow and ice data from NSIDC (US National Ice Center, 2008). Seasonal ranges of caribou from Alaska Center for Conservation Science (https://accscatalog.uaa.alaska.edu/) and (Thing, 1984). Map lines delineate study areas and do not necessarily depict accepted national boundaries. *(Graphic credits: The author.)*

and academic researchers. Recent technological advances have improved efforts to understand how and why tundra landscapes are changing and the extent to which they can be expected to change in the coming decades. With increasing accessibility and open access to remotely sensed data products (Beamish et al., 2020; Gorelick et al., 2017), the picture of tundra responses to climate change grows clearer. In addition to the developing, ongoing, and ever-improving satellite record, in situ sensor networks, field stations, and other long-term monitoring programs reveal patterns of environmental and ecological change in North America's tundra systems.

6.1 Local monitoring efforts and infrastructure

Concerted local monitoring efforts are expanding across the Arctic, and research output on community-based monitoring

is especially well-represented for Inuit and First Nations communities in North America (Kouril et al., 2016). Indigenous and local knowledge holders archive a vast amount of ecological information, valuable for academic research and resource management decisions alike (Danielsen et al., 2021). While local ecological "news" is useful for conservation and scientific research, it is also central to the lives of tundra residents. Thus, forums for sharing information on the status and trends of sea ice, snowpack conditions, and wildlife populations are practical for people who regularly interact with tundra landscapes. One such platform is SIKU (http://www.siku.org), a social network that facilitates information sharing among Inuit communities and emphasizes self-determination in community research and conservation (Heath and Arragutainaq, 2019). Through this and other local monitoring programs, communities can guide local conservation and land management decisions with evidence-based recommendations (Mercer et al., 2023).

Local governments also use concerted local monitoring to track invasive species and disease outbreaks. For example, caribou represented over a quarter of daily food consumed based on a survey of Inuvialuit and Inuit communities in northern Canada (Chiu et al., 2016). Although chronic wasting disease (CWD, an infectious prion that affects gray matter in the brain and spinal cord of cervids) is not yet known in North American caribou populations, several pathways make transmission of the disease to the Arctic possible (Gilch, 2022), and surveillance programs have been put in place by local governments across the continent. Further south, CWD has been found in captive and free-ranging populations of elk and deer in several Rocky Mountain states and provinces (Richards, 2021). Ongoing and future work on the disease's effects on southerly populations will be useful in guiding management decisions both in North America and other continents where CWD appears (Galloway et al., 2021; Monello et al., 2014; Mysterud and Edmunds, 2019).

Academic researchers use local monitoring to draw broader inferences about tundras' response to climate change. Field stations supporting long-term ecological research stand to contextual tundra dynamics in broad patterns of ecosystem change that might not be detected by smaller isolated studies (Gauthier et al., 2024). Long-term ecological research stations can be found across alpine and Arctic tundras of North America, including UCLA's White Mountain Research Center at Barcroft Station in California; Rocky Mountain Biological Laboratory in the Colorado Rockies; Arctic LTER and Toolik Lake Field Station in the Brooks Range, Alaska; The CEN Whapmagoostui-Kuujjuarapik Station in Québec on Cree land; Bylot Island Field Station in Nunavut, Canada; and the Zackenberg Field Station in northeast Greenland. Some stations provide not just logistical support for academic researchers, but also cultural and interpretive programs that facilitate culturally aware research development and broad dissemination of research findings. Although this is far from a universal feature of ecological field stations, it is becoming more common, increasing the local relevance of work being done on-site, and improving connections between visiting scientists and the communities that host them during field campaigns (Tydecks et al., 2016).

6.2 Monitoring at the federal level

National-level ecosystem monitoring efforts are headed up by federal agencies and can be found across the continent (Danielsen et al., 2014; Fancy et al., 2009; Garcia-Alaniz et al., 2017; Vaughan et al., 2001). For example, the United States National Parks Service runs an Inventory and Monitoring program with 32 networks of national parks across the country tracking changes in ecological indicators from rare plant population dynamics to migratory bird movements. Several of these networks, including the Arctic network, the Rocky Mountain network, the Greater Yellowstone network, and the Sierra Nevada network, chronicle the changing ecological conditions of tundra systems. Federal monitoring is typically limited by funding and administrative boundaries; whereas budgetary constraints limit the extent and depth of potential monitoring efforts, national and administrative boundaries limit comparisons of data collected across geography by artificially subdividing the sum total of North America's tundras (Grant and Quinn, 2007; Kark et al., 2015).

6.3 Monitoring at the ecosystem level: Building collaborations that transcend boundaries

Because political boundaries do not reflect contiguous tundra systems—and certainly do not account for their connections to other ecosystems—collaborative efforts that span administrative and international borders are critical for the effective conservation of North American tundras. Interinstitutional, multiagency, and international monitoring are required to contextualize the ecological change that is unfolding across North America. Through networks of automated monitoring stations, far-reaching collaborations with standardized research protocols, and broad-scale open access to data and workflows, patterns of tundra change can be assessed across a more complete scope of global environmental change.

Automated monitoring stations provide in situ measurements across a variety of spatial scales, revealing how on-the-ground change corresponds to broader-scale climatic variability. Local networks of monitoring stations reveal fine-scale patterns of population- and community-level change. For example, networks of time-lapse and motion-sensing cameras have been used to characterize variation in plant growth, snow cover, and animal occurrence across

regions at scales relevant to local ungulate populations in the Sierra Nevada Mountains (CA, USA), Arctic Alaska, and low-Arctic Greenland (John et al., 2023; Kerby, 2015; Leorna and Brinkman, 2024). Continental sensor networks lend insights on tundra change, more broadly, in comparison to other biomes. For example, the Phenocam Network compiles time-lapse imagery from nearly 400 sites across the United States and Canada, providing insights into plant phenological response to changing climatic regimes across biomes of North America (Richardson, 2019; Seyednasrollah et al., 2019). Snow Telemetry Network (SNOTEL) stations are distributed across a similarly impressive spatial extent and collect long-term data on precipitation, snowpack, and air temperature (Serreze et al., 1999). Because SNOTEL stations span tundra and other biomes, they provide an important lens for the changing snowpack situation of North America overall.

International networks dedicated to tundra research and monitoring are critical to understanding how ongoing change in North American tundras compares with broader patterns of tundra change across the globe. Such networks can be found among both alpine and Arctic tundras, with a variety of foci and a range of interdisciplinarity. The Global Observation Research Initiative in Alpine Environments (GLORIA; https://www.gloria.ac.at) uses a standardized monitoring approach to identify global patterns of change in alpine biodiversity (Pauli et al., 2015). Through a combination of far-reaching global scope and consistent resurveying efforts, the GLORIA network reveals how alpine biodiversity is changing across broad climate zones, with a central goal of informing decision-making (Grabherr et al., 2000). Network sites can be found across North America (e.g., Ababneh and Woolfenden, 2010; Swerhun et al., 2009) and contextualize ongoing alpine vegetation change with a global view of alpine climatic change (Pauli and Halloy, 2019). Further north, the Circumpolar Biodiversity Monitoring Program (under the Arctic Council's biodiversity working group, CAFF) is developing frameworks for assessing Arctic tundra community compositional change (Barry et al., 2023). Harnessing recent advancements in low-cost unpiloted aerial systems, the High-Latitude Drone Ecology Network (HiLDEN; https://www.arcticdrones.org) is developing protocols for systematic repeat imaging of the Arctic to untangle fine-scale patterns of tundra vegetation change. HiLDEN synthesizes aerial imagery across a panarctic extent, revealing broad-scale drivers of highly localized patterns of vegetation productivity and functional change (Assmann et al., 2020; Assmann et al., 2019; Myers-Smith et al., 2020; Talucci et al., 2020).

Addressing geographic biases in recent advancements in tundra research will improve the generalizability of research findings. For example, although tundra research has grown considerably over the last several decades, studies of research output reveal a disproportionate representation of the Tibetan Plateau in alpine tundra vegetation work (Verrall and Pickering, 2020), and underrepresentation of research and monitoring in the Canadian High Arctic, Arctic Cordillera, and Greenlandic Arctic tundra ecosystems (Virkkala et al., 2019). Connecting research advancements between alpine and Arctic tundra systems will improve inference about global change and the spectrum of Earth's cold places. For example, the International Tundra Experiment (ITEX) and the Herbivory Network standardize techniques and synthesize findings across both alpine and Arctic tundra (Barrio et al., 2016; Henry et al., 2022). A clearer image of tundra change will crystallize as geographic weak points of North American tundra monitoring are bolstered.

Networks and alliances dedicated to tracking migratory animals are often—and necessarily—international. For example, the Arctic Program for Regional and International Shorebird Monitoring (Arctic PRISM) is a collaboration between United States and Canadian governmental scientists with the shared goal of monitoring shorebird population trends and distributions to inform conservation targets (Bart and Johnston, 2012; Skagen et al., 2003). The CircumArctic Rangifer Monitoring and Assessment Network (CARMA; https://carma.caff.is) is similarly international in its scope and is dedicated to the long-term persistence of another migrant, caribou. CARMA brings together scientists, managers, and conservationists, with standardized field protocols for monitoring caribou individual condition and population vital rates (Russell et al., 2013, 2015). Other collaboration networks are less species-specific, focusing instead on common ecological indicator data, such as GPS collar data aggregated by Movebank (Kays et al., 2022). GPS collars facilitate broad-scale studies of wildlife movements in response to environmental change that might otherwise require around-the-clock observations by field ecologists (Berger et al., 2022; Brushett et al., 2023; Forshee et al., 2022). Paired with measured and modeled remote sensing data, animal tracking projects lend insights into how tundra wildlife movements relate to environmental characteristics, and how these may be modified under foreseeable climate change (Pedersen et al., 2021).

7 Perspectives and outlook

North America's tundras are unique relics of the Pleistocene's cold, open landscapes, and yet they are highly interconnected with warmer biomes made distant by geography and topography. Tundras are the permanent and only home for the hardy communities that are found there year-round, and the summer breeding grounds for a rich and wide-ranging biodiversity that lives seasonally in far-away places. However, these systems and their inhabitants are imperiled: tundras face considerable change at the hands

of humans over the next century. Through direct effects of development and natural resource extraction and indirect effects mediated by climate change, tundras will experience modification in biogeochemistry, shifts in vegetation structure, and degradation of wildlife habitat. Because elements of the future of tundra systems can be anticipated, conservation plans should consider forecastable environmental change. On the other hand, the uncertain nature of some future conditions requires further progress in both climate change ecology and ecoforecasting.

Many wildlife refuges were established with a particular taxa-specific goal in mind (e.g., to protect migratory birds), but bring about more expansive benefits to biodiversity and society as a whole. Local umbrella effects of conservation of tundra-breeding birds, for example, are greatly valuable for year-round tundra obligate species. Considering the broad interconnectedness of tundras to other biomes, a more far-reaching conservation significance is realized. For example, tundra-breeding red phalaropes forage in marine pelagic waters off the coast of Chile during winter (Saalfeld et al., 2024). Conserving breeding ranges of these, and many other Arctic-breeding species has the extended benefit of preserving global-scale systems of biomass and nutrient flux. Birders across North America owe a debt of gratitude to the organizations that conserve tundra spaces, which enrich the lives of those who do not live among tundra landscapes but who do enjoy the symphony of calls and great aggregations emblematic of the migratory season.

7.1 The certain—And uncertain—Future

Tundra systems will change in the coming decades. In some regions, the collection of environmental features that characterize tundra is expected to be completely lost by the end of the century (Rowland et al., 2016). Bioclimatic factors are subject to spatial rearrangement, but the boundaries of most national parks are not. Thus, as species distributions shift in response to climate change, current protected areas may become less effective refugia for species facing extinction (John and Post, 2022a; Reji Chacko et al., 2023). Land managers should prepare for forecastable environmental change where possible, and policymakers should account for anticipated change when establishing new protected areas.

Some aspects of North American tundras' future remain unclear. For example, a series of oil and gas leases have recently been awarded, then rescinded, in the Arctic National Wildlife Refuge (ANWR; Alaska, USA). Whereas industrial representatives cite the economic boon that more drilling would bring to Alaska, conservationists and local stakeholders raise concerns about ecological outcomes (Gwich'in Steering Committee v. Haaland, 2021): proposed drilling sites lie directly within the calving grounds of the Porcupine caribou herd, promising to disrupt movement and reproduction in the culturally valuable wildlife population. The status of these leases currently teeters on the waffling actions of successive presidential administrations, and the final decision on drilling in ANWR is yet to be seen. Ongoing disputes in courts, Congress, and the White House portend broader future disagreements on natural resource policy in the North. As the Arctic becomes increasingly accessible, more and more such clashes are likely to arise in the absence of a cohesive prioritization of tundra conservation across levels and branches of government.

7.2 Research priorities

Effective conservation for the future requires a clear understanding of mechanistic links between ecology and the environment and a confident expectation of future climatic changes. Thus, a need is evident to advance several lines of research. First, although broad-scale forecasts of Arctic and alpine warming are decisive, it is critical to improve forecasts of precipitation change. As climate change may drive local shifts in precipitation duration, form, frequency, intensity, or variability (Ohba, 2021; Persad et al., 2020), knowledge of expected local change could inform anticipatory, on-the-ground conservation action. Ubiquitous across tundra systems is the need to develop a cohesive framework for anticipating change in the concurrence of seasonality among interacting abiotic environmental dynamics. Seasonal variability in temperature, cloud cover, wind, snowfall and snow cover, rain, and the coincidence of all these factors are at the center of cascading ecological outcomes that characterize tundra systems. Incongruous changes among seasonal environmental factors may disrupt interactions that maintain tundra communities.

Second, drivers of spatiotemporal variability in responses of plant growth and plant communities to local climatic change must be determined (Capers et al., 2013; Myers-Smith et al., 2011; Verrall and Pickering, 2020). Nuanced approaches to detecting vegetative growth, microclimatic variability, and biogeochemical cycling can explain complex dynamics of tree line advance and woody encroachment in tundra systems (Myers-Smith et al., 2020; Reid et al., 2022). Interactions between climate and herbivores in controlling tundra vegetation community composition, productivity, and phenology must be explored more widely in order to anticipate cascading outcomes of wildlife conservation and management (Barbero-Palacios et al., 2024; Post et al., 2023).

And third, an improved understanding of the patterns and outcomes of tundra fires is urgently needed, as these disturbance events are projected to increase in the coming years (Foster et al., 2022; Hu et al., 2015). Although tundra fire is not currently common, future increased tundra fires will impact biogeochemical cycling, patterns of tundra vegetation change, and air quality (McCarty et al., 2021). In addition,

fires pose a considerable threat to the people who live in landscapes that burn, and in particular Indigenous people and members of First Nations (e.g., Mottershead et al., 2020). Because fire management practices are not transferrable to the Arctic, it is urgent to fill knowledge gaps on tundra fire initiation, propagation, management, and effects, before these disturbance events—and potential environmental disasters—grow more common (McCarty et al., 2021).

7.3 Building connections

It is critical that research priorities be guided locally within communities and regions of North America. Broad collaborations with global research programs inform questions about global change ecology, but conservation decision-making must also consider more local factors. Identifying local knowledge gaps informs conservation efforts and policy decisions that operate at fine geographic scales. Connections among scientists, land managers, local stakeholders, and conservationists will help optimize decision-making in the context of community needs (Willis, 2015). Local development of research provides a link between academic, conservation, and local knowledge systems (Buschman, 2022; Danielsen et al., 2014), and yet in spite of the general acknowledgment that community involvement in research is "good," only slight increases in local involvement have been seen in Arctic research (Brunet et al., 2014). Clearly, raising Indigenous and local voices in research development and practice is a pathway toward conservation relevance and an underutilized knowledge pool.

Connectivity is critical for conserving mobile species. Thus, it is imperative that local conservation research groups work hand-in-hand with neighboring groups to ensure landscape contiguity, or at least maximize the permeability of barriers to movement and gene flow (Keyghobadi et al., 2005; Sawyer et al., 2013). Although many development projects are not on strictly protected lands, they may present barriers to migrants, and therefore restrict connections among populations and communities. Broad-scale, ecosystem-level campaigns to conserve contiguous habitat, such as the Y2Y Conservation Initiative (https://y2y.net), are useful centers of momentum that generate public interest, concentrate conservation efforts, and ultimately realize conservation goals (Hebblewhite et al., 2022). Yet broader collaborative approaches are warranted for migratory species. Conserving stopover habitat for long-distance migrants, and migratory corridors for short-distance migrants, will be critical for maintaining the wide-ranging geographic requirements of migratory animals (Brown et al., 2017; Kauffman et al., 2021; Skagen, 2006).

Ultimately, combining resources, sharing ideas, and fusing data from the various focused projects discussed above will lead to a more robust view of tundra change and more effective conservation of tundra environments. Increasing collaborations across disciplines will improve both the depth and breadth of our understanding of tundra change by contextualizing variability across fields of study and levels of organization. Improving discourse between stakeholders, researchers, and decision-makers will increase the relevance of research aims and the effectiveness of conservation applications. Tundras are changing, and will continue to do so in the future; the most pragmatic conservation actions will account for foreseeable change and accommodate forecasting uncertainty.

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TERRESTRIAL BIOMES:

Volume 2

Global Biome Conservation and Global Warming Impacts on Ecology and Biodiversity

Edited by Germano Leão Demolin-Leite

Synthesizes information on the threats to and conservation best practices for the world's major terrestrial biomes

Terrestrial Biomes: Global Biome Conservation and Global Warming Impacts on Ecology and Biodiversity explores the effects of anthropogenic activities on Earth's terrestrial biomes, species, and climate. It summarizes operational and potential monitoring tools to conserve or recover terrestrial biomes at a global scale. Written by international experts in ecology and biodiversity conservation, this book identifies the challenges and threats to terrestrial organisms and connects them to real cases of conservation.

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About Editor

Dr. Germano Leão Demolin-Leite is a researcher for the Brazilian National Council for Scientific and Technological Development as well as a full professor in the Institute of Agricultural Sciences at the Federal University of Minas Gerais (ICA/UMFG). He currently teaches courses on animal biology and plant production. Dr. Demolin-Leite's recent research focuses on degraded area recuperation. He has been a coordinator of the Zoology and Entomology Laboratory and Insectarium G.W.G. Morães (*Trichogramma* sp.). He has published more than 230 scholarly journal articles and 6 books. Dr. Demolin-Leite also serves on the review board for numerous journals and periodicals, including Elsevier's *Ecotoxicology and Environmental Safety*.

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