CHAPTER **3**

STATUS, TRENDS AND FUTURE DYNAMICS OF BIODIVERSITY AND ECOSYSTEMS UNDERPINNING NATURE'S CONTRIBUTIONS TO PEOPLE

Coordinating Lead Authors:

Jeannine Cavender-Bares (USA); Mary T. K. Arroyo (Chile)

Lead Authors:

Robin Abell (USA); David Ackerly (USA); Daniel Ackerman (USA); Matias Arim (Uruguay); Jayne Belnap (USA); Francisco Castañeda Moya (Guatemala); Laura Dee (USA); Natalia Estrada-Carmona (Colombia/France); Judith Gobin (Trinidad and Tobago); Forest Isbell (USA); Gunther Köhler (Germany); Marten Koops (Canada); Nathan Kraft (USA); Nicholas Macfarlane (Canada); Cristina Martínez-Garza (Mexico); Jean-Paul Metzger (Brazil); Arturo Mora (Ecuador); Mike Oatham (Trinidad and Tobago); Adriano Paglia (Brazil); Julieta Pedrana (Argentina); Pablo Luis Peri (Argentina); Gervasio Piñeiro (Uruguay); Robert Randall (Canada); Wren Walker Robbins (USA); Judith Weis (USA); Silvia Renate Ziller (Brazil)

Fellow:

Rodolfo Jaffe (Brazil/Venezuela)

Contributing Authors:

Marcelo Aizen (Argentina); Diva Amon (Trinidad and Tobago); Manuel Arroyo-Kalin (UK); Abigail Barker (UK); Keith Barker (USA); Darcy Bradley (USA); Kate Brauman (USA); Eduardo S. Brondizio (Brasil); Jarrett Byrnes (USA); Chris Caldwell (USA); Alejandro Casas (Mexico); Kenneth G. Cassman (USA); Joel Cracraft (USA); Frank Davis (USA); Juan Dupuy (Mexico); Brian Enquist (USA); Beth Fallon (USA); Curtis Flather (USA); Lee Frelich (USA); Susan Galatowitsch (USA); Kelly Garbach (USA); Jeff Grignon (Menominee Nation, USA); Hanno Seebens (Germany); Sharon Jansa (USA); Jon Keeley (USA); Christina Kennedy (USA); Kenneth Kozak (USA); Ulrike Krauss (Saint Lucia); Elena Lazos Chavero (Mexico); Ian McFadden (USA); Mike Melnychuk (USA); Christian Messier (Canada); Ana Isabel Moreno-Calles (Mexico); Mark Nelson (USA); Danilo Mesquita Neves (Brazil); Cathleen Nguyen (USA); Mary O'Connor (Canada); Josep Padullés (Spain); Shyama Pagad (New Zealand); Alain Paquette (Canada); Lindsey Peavey (USA); Jesus Pinto-Ledezma (Bolivia); Diana Ramírez-Mejía (Mexico); María Francisca Flores Saavedra (Chile); Jane Smith (USA); Valeria Souza (Mexico); Ireri Suazo-Ortuño (Mexico); Katharine Suding (USA); Hans ter Steege (The Netherlands/Brazil); Gerardo Suzan (Mexico); Wolke Tobón (Mexico); Ignacio Torres-García (Mexico); Mark van Kleunen (Germany)

Review Editors:

Patricia Balvanera (Mexico), Alexis Cerezo (Guatemala)

This chapter should be cited as:

Cavender-Bares, J., Arroyo, M.T.K., Abell, R., Ackerly, D., Ackerman, D., Arim, M., Belnap, J., Castañeda Moya, F., Dee, L., Estrada-Carmona, N., Gobin, J., Isbell, F., Jaffe, R., Köhler, G., Koops, M., Kraft, N., Mcfarlane, N., Martínez-Garza, C., Metzger, J. P., Mora, A., Oatham, M., Paglia, A., Pedrana, J., Peri, P. L., Piñeiro, G., Randall, R., Robbins, W. W., Weis, J., and Ziller, S. R. Chapter 3: Status, trends and future dynamics of biodiversity and ecosystems underpinning nature's contributions to people. In IPBES (2018): The IPBES regional assessment report on biodiversity and ecosystem services for the Americas. Rice, J., Seixas, C. S., Zaccagnini, M. E., Bedoya-Gaitán, M., and Valderrama, N. (eds.). Secretariat of the Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany, pp. 171-293.

TABLE OF CONTENTS

CHAPTER 3

STATUS, TRENDS AND FUTURE DYNAMICS OF BIODIVERSITY AND ECOSYSTEMS UNDERPINNING NATURE'S CONTRIBUTIONS TO PEOPLE

EXECUTIVE SUMMARY

① **The Americas house a large fraction of the Earth's terrestrial and freshwater biodiversity distributed across 140 degrees of latitude (*well established*).** Around 29 per cent of the world´s seed plants, 35 per cent of mammals, 35 per cent of reptiles, 41 per cent of birds and 51 per cent of amphibians are found in the Americas (*established but incomplete*) {3.2.2.2}, as well as the world´s most diversified freshwater fish fauna of over 5,000 species (*well established*) {3.2.3.1}. The South American subregion is by far the richest subregion for plants and vertebrates (*well established*) {3.2.2.2}. However, the smaller Caribbean and Mesoamerican subregions are very rich for their areas, and North America contains both biodiversity hotspots and unique lineages {3.2.2.2}. The moist tropical lowland forests and tropical high Andean ecosystems contain high biodiversity on a global scale (*well established*) {3.4.1.1, 3.4.1.5}. Numbers of species and total evolutionary distance are generally higher in the tropics, while evolutionary distinctiveness tends to be higher in temperate latitudes {3.2.2.2}. Phylogenetic endemism is important for different taxa in different regions, and geographic patterns of plant functional diversity depend on the trait considered {3.2.2.2}. Biodiversity in all subregions has conservation significance {3.2.2.2} and all biomes provide nature´s contributions to people; the five most important terrestrial biome contributors are: Tropical and subtropical moist forests; Temperate and boreal forests and woodlands; Tropical and subtropical dry forests; Mediterranean forests, woodlands and scrub; Tundra and high elevation habitats (*established but incomplete*) {3.4.1.10}. For aquatic systems, freshwater habitats stand out (*established but incomplete*) {3.4.1.10}.

② **The biodiverse American tropics became a major center of origin for domesticated plants (*well established*) and of traditional agriculture.** Many plants domesticated in Mesoamerica, the Andean region, and the Amazon Basin have become important crops globally (*well established*) {3.3.3, 3.4.1.1, 3.4.1.5}. Traditional agricultural systems harbor high levels of biodiversity and represent a high-quality matrix that allows forest species movements among patches (*established but incomplete*) {3.3.3}. Traditional farming systems have a structural complexity and multifunctionality that benefit people and ecosystems; they allow farmers to maximize harvest security and reap the benefits of multiple use of landscapes with lower environmental and biodiversity impacts (*established but incomplete*) {3.3.3}.

③ **Many terrestrial biomes, or large parts thereof, in the Americas have lost around 50 per cent or more of habitat, leading to losses in biodiversity and ecosystem functions (*well established*). A few biomes, however, are now showing recuperation or are fairly stable (*established but incomplete*).** Close to 50 per cent of the Great Plains grasslands, including over 95 per cent of tallgrass prairie; some 88 per cent of the south atlantic forest; nearly 70 per cent of the South American Río de la Plata grasslands; 82 per cent of mesic broadleaf forest in Mexico; 72 per cent of tropical and dry forest in Mesoamerica; 66 per cent of tropical dry forest in the Caribbean; 50 per cent of the broader South American Mediterranean-climate biome; and 50 per cent of Cerrado has been transformed, mostly ongoing, leading to declines in native species richness and population sizes and nature's contributions to people (*well established*) {3.4.1.1, 3.4.1.2, 3.4.1.4, 3.4.1.6, 3.4.1.7, 3.4.1.10}. Notwithstanding a perceptible trend for conversion of páramo and puna in some parts of the northern Andes, the tundra and high elevation habitat biome is the least transformed {3.4.1.10}. Agriculture and deforestation have led to depletion of soil organic carbon, lowering of carbon stocks and affected the water cycle (*established but incomplete*) {3.2.1.1, 3.2.1.2}. Presently Caribbean forests are expanding (*well established*) {3.2.2.1, 3.4.1.1} and North American forests are stable to slightly increasing (*established but incomplete*) {3.2.2.1}.

4 **Experimental evidence and empirical observations support linkages between biodiversity and ecosystem productivity, stability and resistance to stress (*well established*).** A large number of studies across taxonomic groups and biomes (temperate and tropical forests, grasslands and marine systems) show greater productivity, stability, and stress resistance of ecosystems with higher biodiversity {3.1.2, 3.1.3}, indicating that biodiversity is relevant to sustainability. The majority of studies within the Americas were conducted in North America, but studies in Mesoamerica and South America are consistent with results for North America and global findings.

5 **The transformation of wetlands in the Americas has led to loss of biodiversity (*established but incomplete*) and ecosystem functions (*well established*).** From 1976 to 2008, the Brazilian pantanal experienced a huge loss of floodplains (*well established*) affecting biodiversity (*established but incomplete*) {3.4.1.9}. One-third of the freshwater marshes in the lower Paraná delta were converted between 1999 and 2013 (*well established*) {3.4.1.9}. The vast biologically rich South American Pantanal has been increasingly degraded due to cattle ranching and cropping (*well established*) {3.4.1.9}. Mechanized peat mining in southern temperate peatlands has promoted invasive plant species, increased beaver presence and produced hydrological changes (*well established*) {3.4.1.9}. In recent years, the United States of America lost an average of 5,600 hectares per year of wetland habitat, lowering capacity for water filtration {3.4.1.9}. In the past four decades, invasive species have become an increasing threat to biodiversity in the Florida Everglades and other wetlands (*established but incomplete*) {3.4.1.9}. Some wetlands in Mesoamerica have been contaminated with heavy metals and pesticides (*established but incomplete*) {3.4.1.9}.

6 **Oceans of the Americas contain high biodiversity, can have high numbers of threatened species, and include large numbers of species that are important for human well-being (*established but incomplete*).** Respectively, over 12,000 marine organisms have been found in the Caribbean, 10,000 in the Humboldt Current system, and 9,000 on the Brazilian shelves {3.2.4.1}, numbers that are considered to be conservative. Oceans of the Americas contain three of the seven global threat hotspots for neritic and epipelagic oceanic sharks in coastal waters (*established but incomplete*) {3.4.2}. The highest number of threatened or endangered marine mammal stocks around the globe are found in the Pacific, but some populations have recently begun to recover (*well established*) {3.4.2}. Stock assessments for a number of chondrichthyes in the Americas report declines of 20 to 80 per cent from unfished conditions. In Canada, marine fish populations declined by an average of 52 per cent from 1970 to the mid-1990s and then remained stable (*established but incomplete*) {3.4.2}.

7 **Biodiversity in coastal habitats has experienced major losses in recent decades (*well established*).** Coral reefs in the Caribbean declined in cover by more than 50 per cent by the 1970s, with only 10 per cent remaining by 2003, followed by widespread coral bleaching in 2005 and subsequent mortality from infectious diseases (*established but incomplete*) {3.4.2.1}. Coastal salt marshes and mangroves are rapidly disappearing (*established but incomplete*) {3.4.2.1}. Considerable declines in seagrasses have occurred (*established but incomplete*) {3.4.2.1}.

8 **Urban expansion constitutes both a threat to biodiversity and an opportunity for biodiversity conservation {*established but incomplete*).** Urban areas are now home to 80 per cent of the population of the Americas {3.3.4}. Urban encroachment is associated with declining native species richness and shifts in species composition, yet increased total plant diversity with cultivation of non-native species (*established but incomplete*) {3.3.4}. Remnant vegetation in cities can support significant native biodiversity, such as bees and birds (*well established*). Botanical gardens, major reservoirs of *ex situ* conservation, and important for recreation and environmental education, found mostly in urban areas, are unequally distributed among subregions and biomes (*well established*) {3.3.4}. Green areas that incorporate native biodiversity have the potential to accomplish the dual goals of conservation and human well-being {3.3.4}.

9 **Alien species continue to appear in terrestrial, freshwater, and marine habitats in the Americas, but rates of introduction, where known, differ among subregions (*established but incomplete*).** Terrestrial and marine habitats house outstanding numbers of alien plants and bird species {3.2.2.3, 3.2.4.2}. North America and the Caribbean are the mostly strongly invaded subregions (*established but incomplete*) {3.2.2.3}. Rates of appearance of alien species are currently somewhat lower in North America than in South America (*established but incomplete*) {3.2.2.3}. Marine habitats of the North American subregion are more heavily invaded than other subregions, with the Pacific Ocean more invaded than the Atlantic (established by incomplete) {3.2.4.2}. For freshwater, temperate piscivorous, and carnivorous fish cause negative impacts on the native fish fauna (*established but incomplete*) {3.2.3.2}. In the Americas, several endangered and threatened species have declined as a result of emerging infectious diseases {3.2.6}. Strongly invasive alien species can entail significant economic costs for infrastructure {3.2.3.2}, and significantly lower productivity (*well established*) {3.2.2.3}.

10 Overall, species threat level is high in the Americas, but the underlying causes vary among subregions (*established but incomplete*). Based on 14,000 species assessed that occur in the Americas, close to a quarter of species face extinction risk (*established but incomplete*) {3.5.1}. Aggregate threat risk over the past two decades was highest in South America and the Caribbean (*well established*) {3.5.1}. Since 1989, the number of threatened North American freshwater fishes has increased by 25 per cent, with 7.5 extinct taxa per decade post-1950 {3.2.3.1} (*well established*). In Central America, 42 per cent of close to 500 known amphibian species have been assessed as threatened (*well established*) {3.2.3.1}. The International Union for Conservation of Nature category "Invasive species, other problematic species, genes and diseases" is the main cause for extinction risk in the North American subregion, while the categories "Biological resources use" and "Agriculture and aquaculture" are the most important causes in the Mesoamerican, Caribbean and South American subregions (*established but incomplete*) {3.5.1}.

11 While protection measures in the Americas have increased and diversified over the past 30 years, major differences in protection effort persist between terrestrial and marine ecosystems and among biomes (*well established*). The increase in protection has been notable in South America where 25 per cent of this subregion is now protected. South America, Mesoamerica, and the Caribbean lag behind North America in terms of marine protection (*well established*) {3.5.2}. Twenty percent of all designated key biodiversity areas globally are found in the Americas (well established} {3.5.2}, yet, less than 20 per cent of these are completely covered (*well established*). Certain biomes are still poorly protected (*well established*) {3.5.2}. Temperate grasslands in general and South American Mediterranean forests, woodlands and scrub and drylands are among the least protected biomes {3.5.2}. Tropical and subtropical savanna and grasslands, tropical and subtropical dry forests, and tropical and subtropical coniferous forests are poorly protected {3.5.2}. Indigenous reserves and private initiatives and are increasingly important {3.5.2}.

12 Many Aichi targets are unlikely to be met in some countries (*established but incomplete*). Although the rate of loss of natural habitat has decreased in some biomes, degradation and fragmentation continue {3.4.1.10}, making it unlikely to achieve Aichi target 5. Unsustainable fishing continues {3.4.2} (Aichi target 6). Likewise, many intensive agricultural, silvicultural and aquacultural systems do not follow biodiversity-friendly practices {3.3.5} (Aichi target 7). Alien and invasive alien species are widespread and continue to appear across the Americas {3.2.2.3, 3.2.3.2, 3.2.4.2, 3.4} (Aichi target 9). Coral bleaching continues in response to coastal pollution and global warming {3.4.2.1} (Aichi target 10). Total protected area coverage for the Americas is 14 per cent, with 18 per cent terrestrial and 9 per cent marine, but some biomes remain severely under-protected {3.5.2}. Better biome representation would allow meeting Aichi target 11. Although conservation efforts have improved, overall extinction risk for species has increased in some subregions {3.5.1} (Aichi target 12).

13 Major biodiversity data and knowledge gaps persist across the Americas (*well established*). Basic exploration is incomplete, especially in the richest biodiversity areas. Brazil contributed the largest number of new plant species to the global inventory from 2004 to 2016, and 42 per cent of recently described new mammals species worldwide between 1993 and 2008 came from the Americas (*well established*) {3.6}. In South America experts predict that around 50 per cent of marine biodiversity remains undiscovered (*established but incomplete*) {3.6}. Research on functional diversity and the relationship between biodiversity and ecosystem functions across taxonomic groups is growing but remains scarce in some subregions. Enormous data gaps persist at the biome level in all subregions. Despite its very high biodiversity, South American houses the fewest georeferenced species occurrence records per unit area, while the highest number is in North America, despite much lower richness {3.6}. Major challenges for the future are: scaling up from ecological studies to the biome level, coordinated conservation efforts in biomes that cross country boundaries, making all biodiversity data available online, and the production of standardized biodiversity data useful for policymakers {3.6}.

3.1 BACKGROUND

3.1.1 Setting the stage

The Intergovernmental Science-Policy Platform on Biodiversity and Ecosystem Services (IPBES) recognizes that humans benefit both consciously and unconsciously from ecosystem functions and biodiversity, through the ecosystem services they are coupled with, referred to as nature's contributions to people (NCP).

The biodiversity of the Americas comes from many different marine, freshwater, and terrestrial sources and offers humankind numerous products and services. To protect the enormous potential of this biodiversity to provide NCP, it is critical to understand the geographic distribution of biodiversity as well as how biodiversity, and the ecosystem functions that both depend on and support biodiversity, have been changing over time.

This chapter assesses: (1) our current understanding of the distribution, status and recent trends of ecosystem functions and biodiversity across the Americas; (2) how people interact with biodiversity, highlighting the importance of local and indigenous knowledge; (3) how biodiversity and ecosystem functions vary within and have changed across the units of analysis in each subregion; (4) current understanding of the extent to which biodiversity is imperiled and protected; and (5) major data and knowledge gaps in all of these realms. The chapter focuses on biodiversity and ecosystem function **(Figure 3.1)** in the context of how they contribute to NCP (Chapter 2) and are impacted by drivers of change (Chapter 4).

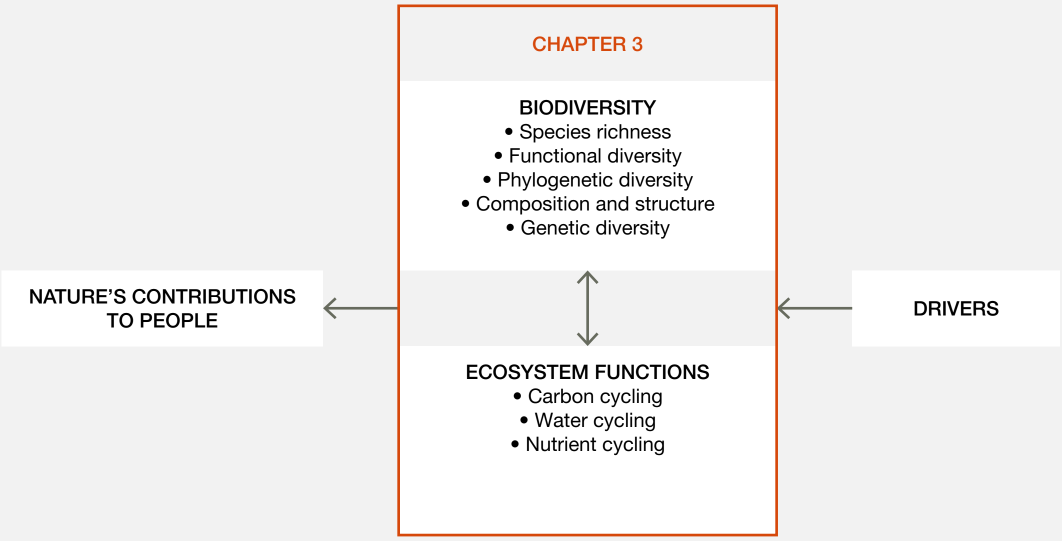
3.1.2 How is biodiversity linked to ecosystem functions and ecosystem services?

Biodiversity loss is known to substantially decrease ecosystem function and stability (Cardinale *et al.,* 2011; O'Connor *et al.,* 2017). Consequently, biodiversity loss and ecosystem degradation diminish the ability of humans to benefit from or establish spiritual relationships with other living beings.

The relationships between biodiversity and ecosystem function have been rigorously investigated in numerous experiments (e.g. Cardinale *et al.,* 2011) and in theoretical (Loreau, 2010; Tilman *et al.,* 1997) and observational studies in a wide range of ecosystems, including grasslands (Grace *et al.,* 2016; Hautier *et al.,* 2014), forests (Gamfeldt *et al.,* 2013; Liang *et al.,* 2016; Paquette & Messier, 2011), drylands (Maestre *et al.,* 2012) and marine systems (Dee *et al.,* 2016; Duffy *et al.,* 2016), many conducted in the Americas. Recent studies have also revealed many potential benefits of increasing plant diversity in managed production systems, including enhancing the production of crops, forage, wood, and fish; stabilizing productivity; enhancing

Figure ③ ① **Within the IPBES conceptual framework, Chapter 3 focuses on the status and trends of biodiversity, which encompasses a range of dimensions, and ecosystem functions, as well as the linkages between them.**

Biodiversity and ecosystem functions underpinning NCP, treated in Chapter 2, are influenced by drivers of change, treated in Chapter 4. See Chapter 1 for the complete IPBES framework. Source: own representation.



pollinators and pollination; suppressing weeds and other pests; and accumulating and retaining soil nutrients and carbon (Balvanera *et al.,* 2006, 2014; Cardinale *et al.,* 2012; Kremen & Miles, 2012; Letourneau *et al.,* 2011; Quijas *et al.,* 2010; Scherer-Lorenzen, 2014).

3.1.3 Conceptual and theoretical linkages between biodiversity and ecosystem functions and services

Biodiversity loss can alter ecosystem function. Here, we focus on relationships between plant diversity and productivity. Theory (Thébault & Loreau, 2003) and experiments (Lefcheck *et al.,* 2015) have shown that these relationships are largely generalizable to other trophic levels. Furthermore, given that rates of primary productivity limit the energy available to animals at all higher trophic levels, effects of changes in biodiversity on productivity have many cascading effects on other pools and fluxes of matter and energy in ecosystems (McNaughton *et al.,* 1989).

Plant species richness increases primary productivity when interspecific competition is reduced relative to intraspecific competition (Loreau, 2004; Vandermeer, 1981). Reduced competition among species for resources can occur in diverse communities because different plant species consume somewhat different resources (e.g. different forms of nitrogen) or consume the same resources at somewhat different times (e.g. phenological niche partitioning) or places (e.g. different rooting zones) (McKane *et al.,* 2002; Tilman *et al.,* 1997). Such resource partitioning likely contributes to both coexistence and positive effects of plant diversity on ecosystem productivity in many ecological communities (Turnbull *et al.,* 2016). Similarly, increased plant species richness can lead to increased ecosystem productivity when there is reduced apparent competition in diverse communities because plant species can avoid natural enemies, such as specialized herbivores or pathogens, that become diluted in diverse communities (Petermann *et al.,* 2008; Turnbull *et al.,* 2016). Strong effects of complementarity between species or groups of species (Brooker *et al.,* 2008), such as between grasses and legumes (Temperton *et al.,* 2007), contribute to the positive effects of plant diversity on ecosystem productivity. Results from the five longest-running grassland biodiversity experiments suggest that these complementarity effects grow stronger over time, while the importance of individual species that are particularly productive become less important for ecosystem productivity (Fargione *et al.,* 2007; Isbell *et al.,* 2009; Marquard *et al.,* 2009; Reich *et al.,* 2012; van Ruijven & Berendse, 2009). Based on abundant empirical evidence, it is now well-established that local complementarity effects often explain positive effects of biodiversity on ecosystem productivity (Cardinale *et al.,* 2011; Loreau & Hector, 2001), especially in long-term studies (Cardinale *et al.,* 2007; Fargione *et al.,* 2007; Isbell *et al.,* 2009; Marquard *et al.,* 2009; Reich *et al.,* 2012; van Ruijven & Berendse, 2009), but the precise mechanisms are not always possible to discern and are the subject of ongoing research.

Biodiversity experiments address limitations of observational studies and have been designed and conducted to tease apart effects of changing numbers of species (richness) from effects of changing identities of species (composition) (c.f., O'Connor *et al.,* 2017). Such experiments have revealed some surprisingly productive species and combinations of species, even when excluding legumes (van Ruijven & Berendse, 2005; Wilsey & Polley, 2004) or mixing species within functional groups (Bullock *et al.,* 2007; Reich *et al.,* 2004). Changes in grassland plant species richness can influence plant productivity as much as changes in species composition (Hector *et al.,* 2011), intensive agricultural management (Weigelt *et al.,* 2009) and many other factors long known to regulate plant productivity (Hooper *et al.,* 2012; Tilman *et al.,* 2012). Similar strengths of biodiversity effects on ecosystem function have been found in terrestrial and aquatic habitats (O'Connor *et al.,* 2017). Meta-analysis reveals that herbivore diversity influences more ecosystem functions than plant diversity (Arias-González *et al.,* 2016; Lefcheck *et al.,* 2015). Additional examples of biodiversity links to ecosystem functions in different biomes and other units of analysis can be found throughout the chapter.

3.2 CONTINENTAL DISTRIBUTION OF ECOSYSTEM FUNCTIONS AND BIODIVERSITY

3.2.1 Status and trends of ecosystem functions linked to biodiversity

3.2.1.1 Carbon cycling and energy fluxes

Status. The carbon cycle is strongly linked to land cover change (section 3.3.2) and energy flux since energy enters and moves through ecosystems in the form of carbon-based molecules. Therefore, the carbon cycle has major implications for ecosystem function and provisioning of ecosystem services. Land use change increases carbon emissions or sequestration depending on the nature of vegetation replacement. Agriculture and deforestation are the main land use changes that have altered carbon fluxes and stocks. Overall, agriculture has reduced carbon inputs to ecosystems through harvest and/or increased carbon output from cultivation; human appropriation of primary production (a measure of the amount of energy captured by humans from ecosystems) is particularly high in agricultural regions of the Americas (Haberl *et al.,* 2007). Agricultural soils lose carbon when monocultures of annual crops are planted without rotations (Ernst & Siri-Prieto, 2009; Franzluebbers, 2005). However, recent trends in double cropping, no-till practices and used cover crops have the potential to at least partially restore soil organic carbon stocks (Franzluebbers, 2005; Poeplau *et al.,* 2015; Rimski-Korsakov *et al.,* 2016).

Forest ecosystems of Americas contain near 250 picograms of carbon (Köhl *et al.,* 2015), with large amounts of biomass carbon stored in South American forests and high soil carbon stocks located in the permafrost boreal regions of Canada (Jackson *et al.,* 2017). Deforestation (section 3.2.2.1 and Chapter 4) has significantly decreased plant biomass stocks (80 to 95%) throughout the Americas (Chapin *et al.,* 2012) and also soil carbon stocks (Villarino *et al.,* 2016) except in moist forests replaced by pastures that may increase soil organic carbon stocks (Eclesia *et al.,* 2012; Guo & Gifford, 2002). Maintaining the integrity of forests in the Americas thus is essential for climate regulation. Croplands today in the Americas contain 20 to 40% less carbon than under native forest, savannas or grasslands (Alvarez, 2005; Guo & Gifford, 2002).

Recent trends. Forest regrowth in some parts of the North American subregion increased between 1990 and 2015 (Keenan *et al.,* 2015), and primary production in plantation forests mostly in South America has sequestered significant atmospheric carbon (Wright *et al.,* 2000). Recent decreases in deforestation rates in Amazonia have favored net atmospheric carbon sequestration (Davidson *et al.,* 2012; Nepstad *et al.,* 2014; Zarin *et al.,* 2016). Afforestation of grasslands has increased carbon uptake (primary production) and biomass carbon stocks (Vassallo *et al.,* 2013), and increased soil organic carbon on dry sites but decreased soil organic carbon contents on humid sites (Berthrong *et al.,* 2012; Eclesia *et al.,* 2012). While recent woody encroachment in the USA and Argentina has increased biomass stock, it may have negative impacts on deep carbon storage (Asner & Archer, 2010; Jackson *et al.,* 2002). Satellite-detected trends in the Normalized Difference Vegetation Index (a proxy of primary production) support observed changes in carbon stocks (Hicke *et al.,* 2002; Paruelo *et al.,* 2004). Finally, oceans around the Americas must be storing significant amounts of carbon, given they represent a significant fraction of the 2 picograms/year global ocean total. The Americas total is not available.

The net impact of land use on climate change is still under debate (Anderson-Teixeira *et al.,* 2012; Houspanossian *et al.,* 2017; Jackson *et al.,* 2008). Meanwhile, it is clear that soil organic carbon loss severely affects soil fertility and plant production and that such losses are associated with nutrient releases and erosion that promote the eutrophication of rivers, lakes and oceans, all affecting human well-being. Several studies show negative impacts of land use changes on water cycling and other ecosystem services (Jackson *et al.,* 2005; Trabucco *et al.,* 2008).

3.2.1.2 Water cycle and regulation

Status. The water cycle is strongly regulated by evapotranspiration, which reduces water available for runoff and groundwater recharge (Brauman *et al.,* 2007). Evapotranspoiration depends on the physical structure of vegetation and characteristics of individual species, particularly rooting depth, which controls plant access to water in water-limited environments (Le Maitre *et al.,* 2015). Woody vegetation generally has higher evapotranspiration than other vegetation, reducing streamflow (Bosch & Hewlett, 1982; Brown, *et al.,* 2005; Sahin & Hall, 1996). Studies supporting this conclusion are largely from temperate regions (Andréassian, 2004), although some research has also been carried out in the tropics (Cashman, 2014; Tomasella *et al.,* 2009). The reduction in woody vegetation is also associated with higher soil infiltration (Farley *et al.,* 2005; Ochoa-Tocachi *et al.,* 2016). Changes in infiltration have been also attributed to the impact of conversion on soils, which are compacted by timber harvesting and cattle grazing (Tomasella, *et al.,* 2009). The hydrologic impact of forest conversion to pasture depends on grazing intensity, with high-density grazing causing more surface flow (Ochoa-Tocachi *et al.,* 2016). These kinds of links with biodiversity are important for water regulation.

Recent trends. Reduced evapotranspiration can lead to reduced rainfall. Measurements and models of climate impacts of deforestation demonstrate a threshold by which complete deforestation of the tropics would substantially reduce rainfall (Lawrence & Vandecar, 2015). More realistic measurements and models of deforestation in the Amazon and non-Amazonian South America, however, show land use change to reduce precipitation only on the order of a few percent (Lawrence & Vandecar, 2015). The impact of changing climate on streamflow is complex, and most large rivers worldwide have not changed measurably at this point. Ten of the 14 large rivers that show increasing discharge are in the Americas. These rivers mostly correspond to places where rainfall has measurably increased (Milliman *et al.,* 2008).

3.2.1.3 Nutrient cycling

Status. Over the past century, land use change, new agricultural practices, and fossil fuel combustion have drastically disrupted nutrient cycles worldwide (Canfield *et al.,* 2010). Latin America showed high biological nitrogen fixation in native ecosystems until the mid-1990s (26.6 teragrams of nitrogen) and maintained fertilization and legume crops at relatively low rates (5.0 and 3.2 teragrams of nitrogen, respectively). In contrast, North America is characterized by relatively low natural fixation (11.9 teragrams of nitrogen), and high fixation by legume crops (6.0 teragrams of nitrogen) and fertilization (18.3 teragrams of nitrogen) (Galloway *et al.,* 2004). While increased nitrogen input into agricultural ecosystems in the Americas has increased food production, it has promoted a four-fold increase in river nitrogen exports and a four- to seven-fold increase in nitrogen emissions to the atmosphere (Galloway *et al.,* 2004) resulting in reduced drinking water and air quality, freshwater eutrophication, biodiversity loss, rain acidification, stratospheric ozone depletion, climate change and coastal ecosystem destruction (dead zones). Severe pollution occurs with the discharge of the Mississippi River into the Gulf of Mexico, of several rivers on the eastern coast of North America and from some rivers in South America associated with agriculture (Diaz & Rosenberg, 2008).

Recent trends. As a result of the green revolution, nitrogen inputs increased in the Americas, particularly in South America over the past two decades (Austin *et al.,* 2006). Soybean crops expanded from 17 to more than 46 million ha between 1990 and 2010 (FAO, 2011). Some 48% of all croplands in southern South America (Brazil, Argentina, Uruguay, Paraguay, and Bolivia) are soybean (FAO, 2011). In addition, both North and South America have become key grain exporters; currently around 8 teragrams of nitrogen are being exported from the Americas, mainly to Europe and Asia, while around 6 teragrams of nitrogen come back as fertilizers, generating an imbalance in the region of 2 teragrams of nitrogen per year as a result of international trade (Galloway *et al.,* 2008). However, the Americas show a better nutrient balance in agricultural systems than other regions of the world (Vitousek *et al.,* 2009). Although technology is available for improved nutrient recycling in cities and farms, it is seldom used in the Americas (Grimm *et al.,* 2015; Snapp at al., 1998). The use of legumes and catch crops (i.e. fast-growing crops that are grown between successive plantings of a main crop) to tighten or close the nitrogen cycle via synchronization of nutrient uptake and mineralization to avoid nutrient loses is a challenging issue for the Americas.

3.2.2 Status and trends of terrestrial biodiversity

3.2.2.1 Land cover status and trends

With better technology and availability of country surveys, we now have fairly reliable estimates of land cover in the Americas, especially for forests (**Figures 3.2.** and **3.3**). More than two-thirds of the Americas is composed of closed to open vegetation, including forests, savannas, and grasslands, as well as mosaics of those vegetation types. About 16% of the region is occupied exclusively by croplands (e.g. corn, soybeans, wheat, sugarcane, and grazing land) and 1% by urban or bare land (Tuanmu & Jetz, 2014).

Forest cover in the Americas represents ca. 40% of the global forest cover, with ca. 842 millions of hectares in South America, 723 millions of hectares in North America and 20 millions of hectares in Central America (Keenan *et al.,* 2015). Following the last update of Global Forest Watch (2017), which differentiates native from planted forests, the Americas have 1.668 millions of hectares of natural forest and ca. 67 millions of hectares of planted areas (e.g. timber, oil palm, rubber). Around 870 millions of hectares of the natural forest cover is considered primary forest (no clear indications of human activity or significant disturbance) and 797 millions of hectares is naturally regenerated native forests with clear indications of human activities (Global Forest Watch, 2017).

Figure ③ ② **Forest gain and loss across the Americas between 2000 to 2012. Source: Modified from Hansen *et al.* (2013).**

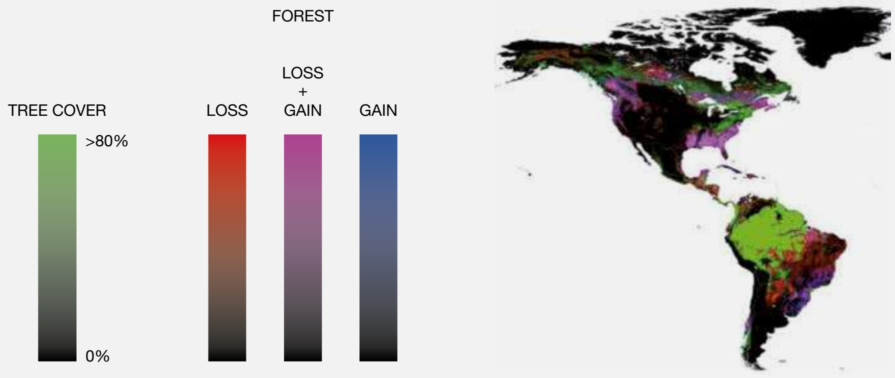
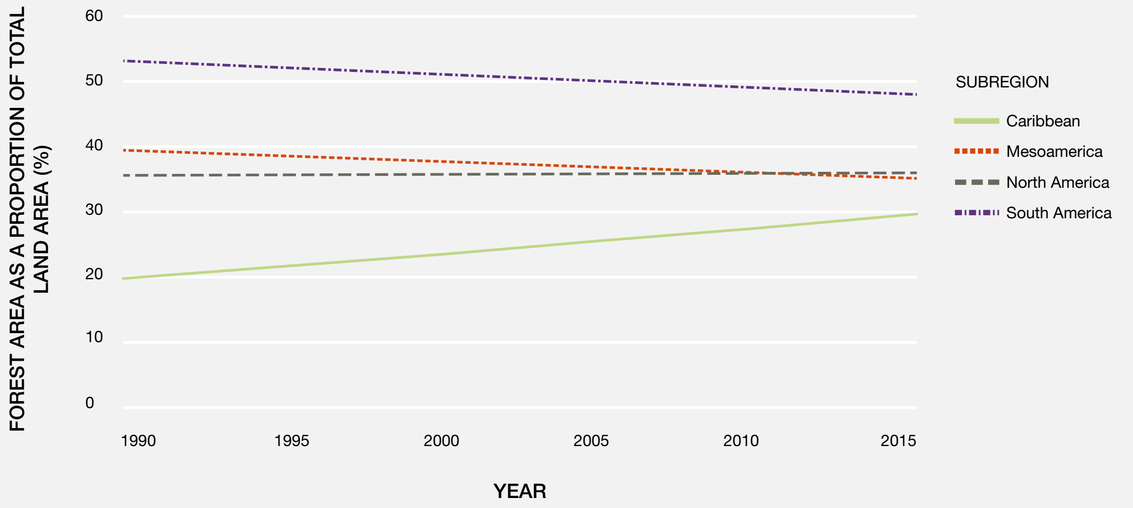


Figure 3 3 **Total forest cover trends by subregions. Indicator data source: FAO (2015). The figure prepared by Task Group on Indicators and Knowledge and Data Technical Support Unit.**



Forest cover has changed throughout the Americas in recent decades **(Figure 3.3)**. It continues to decline in most subregions except in the Caribbean where forest regrowth predominates (see also 3.4.1.1). In North America the overall amount of forest has slightly increased (**Figures 3.2** and **3.3**). Further details on declines can be found for the specific biomes assessed in section 3.4.

Grasslands and shrublands are frequently confounded with agricultural areas or pasturelands at coarse scales and usually represented as a “mosaic of vegetation and cropland” (Arino *et al.,* 2012). This mixed class covers about 12% of the Americas (and includes almost 80% of the croplands) (Arino *et al.,* 2012) distributed predominantly in the USA (Central Great Plains), Canada (e.g. northern grasslands), Chile (Patagonian grasslands), Brazil (campos sulinos) and Argentina (pampas, Patagonian grasslands). Shrublands or savannas represent another 10% of the Americas’ land cover, with extensive coverage in the USA (e.g. Californian chaparral, arid shrublands, Great Plain shrublands) and Brazil (Cerrado). For details of changes in the different biomes of the Americas (section 3.4).

3.2.2.2 Status and patterns of diversity for taxonomic groups

Overall richness patterns. Despite several centuries of exploration, accessible and accurate data for biodiversity across the entire Americas is limited to a very small number of taxonomic groups. Data compiled at the subregional level for such groups confirms that the Americas region (comprising 28% of the world´s land area, including water bodies), holds significant proportions of the world’s biodiversity, as high as 51% for amphibians and 41% for birds **(Table 3.1)**. Species richness is highest for all taxonomic groups in the South American subregion and far higher in South America than in North America **(Table 3.1)**. Mesoamerica and the Caribbean are very rich in relation to their land area. For example, the Caribbean subregion (<1% of the Americas´ land area) is more diverse than North America (51% of the Americas´ area) for reptiles and is not that far behind for plants **(Table 3.1)**. Mesoamerica (6% of the Americas´ land area) has more species in all taxonomic groups — in three out of five cases over twice as many — as the much larger North American subregion. The Americas account for some 33% of plants that have been recorded to be useful to humans globally **(Table 3.1)**. The absolute numbers of useful plants in **Table 3.1** are likely to be conservative, given that comprehensive surveys of useful plants have still to be undertaken in many parts of the Americas.

Continent-level spatial patterns. The development of new biodiversity metrics that go beyond traditional species richness and better spatial data coverage of species over the past 15 years have greatly improved our understanding of how biodiversity is distributed at a finer geographical scale within the Americas. New patterns have emerged that are highly relevant for the valuation of biodiversity across the region. See the glossary for definitions of the biodiversity metrics assessed.

Reflecting the subregion-level survey data **(Table 3.1)**, amphibians, birds, mammals, and plants all show high species richness in tropical South America

Table ③ ① **Species richness for taxonomic groups where data could be compiled for IPBES subregions of the Americas. The percentages under the subregional headings give the amount of land (including water bodies) in relation to the total for the Americas. The percentages for the different taxonomic groups and useful plants in each subregion are calculated in relation to the totals for the Americas.**

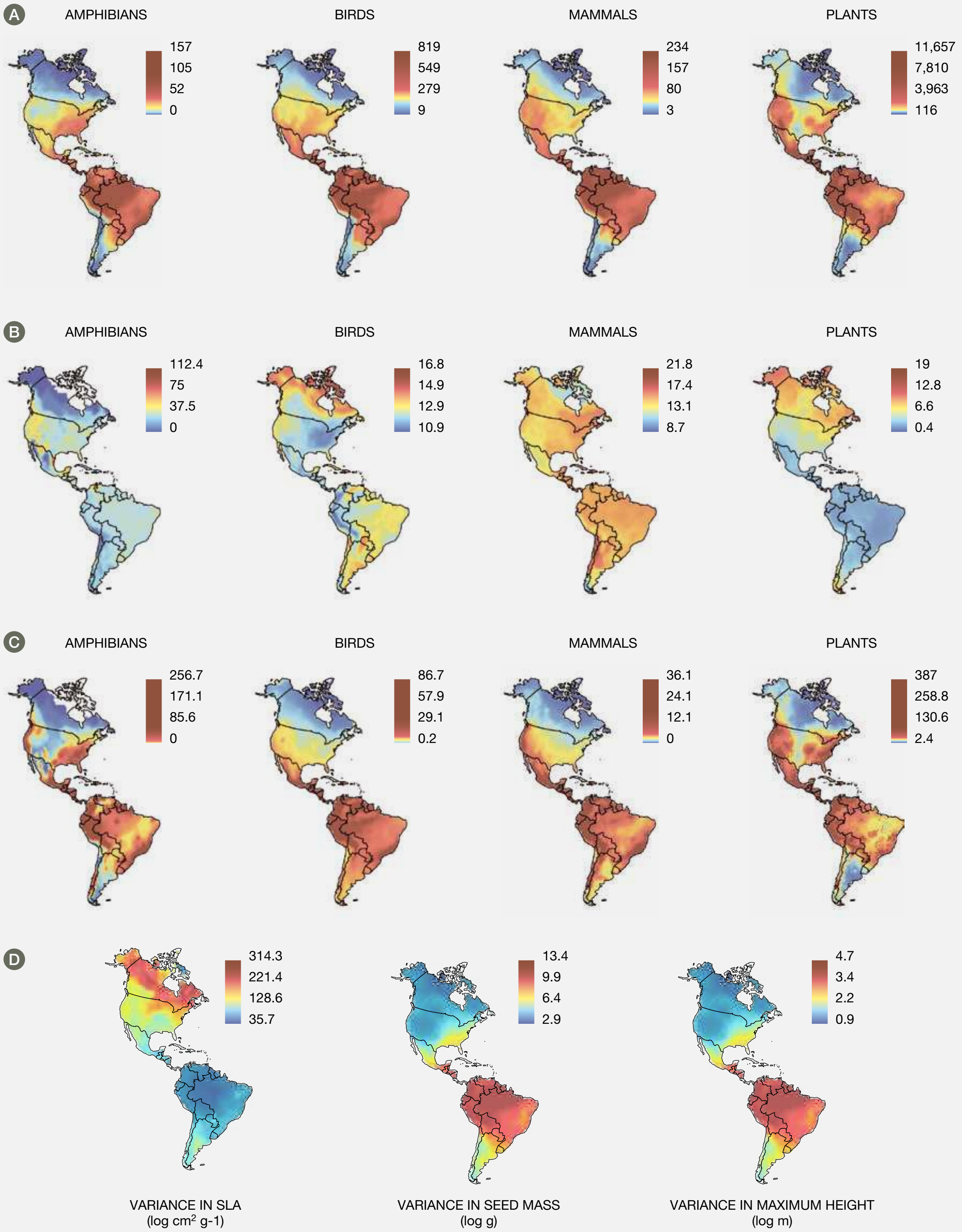
Taxon	Total for Americas	% of world total	North America (51%)	Mesoamerica (6%)	Caribbean (<1%)	South America (42%)
Plants[1,8] (seed plants only)	98,473 (108,320)[2]	29	13,214 (13%)	26,551 (27%)	11,473 (12%)	63,725 (65%)
Useful plants[3] (seed plants only)	10,188	33	4,252 (42%)	4,217 (41%)	2,915 (29%)	5,621 (55%)
Birds[4] – breeding species	4,374	41	649 (15%)	1,191 (27%)	320 (7%)	3205 (73%)
Mammals[5] native – terrestrial	1,963	35	458 (23%)	627 (32%)	185 (9%)	1,266 (64%)
Amphibians[6]	3,928	51	307 (8%)	812 (21%)	234 (6%)	2,809 (72%)
Reptiles[7]	3,652	35	431 (12%)	1231 (34%)	637 (17%)	1,990 (54%)

1. Compiled by the Royal Botanic Gardens, Kew. Seed plants include angiosperms and gymnosperms and both native and non-native species. Data are from the World Checklist of Selected Plant Families (published and unpublished), which is 90% complete. The families Melastomataceae and Asteraceae and the genus *Solanum* are not included at this stage.
2. Estimate if the two missing families and *Solanum* are included. Percentage of the world total in the Americas is based on the estimated total and a world total of seed plants of 370,492 (Lughadha *et al.,* 2016). The subregional totals have not been adjusted and thus are conservative.
3. The useful plant data come from the Royal Botanic Gardens, Kew Useful Plants Data Base. This database is formed from a combination of resources amounting to 31,128 species. The data are for seed plants only and include exotic species, but not commercially-grown crops.
4. Compiled by Chapter 3 from Del Hoyo *et al.,* 1992a, b; Gill and Donsker (2017); Rodewald (2015); Wetmore *et al.* (1957). World total data from Gill and Donsker (2017).
5. North America – Bradley *et al.* (2014); Caribbean – Upham (2017) and IUCN, (2014); Mesoamerica ¬– IUCN, (2014); South America – IUCN, (2014). Total for calculation of world %: IUCN Red List.
6. amphibiaweb.org.
7. reptile-database.org
8. After the Summary for Policy Makers (SPM) for the Americas assessment was completed in early December 2017, in a paper published in Science on 14 December 2017, Ulloa Ulloa *et al.* (2017) reported 124,993 species of vascular plants (seed plants, ferns and fern allies) for the Americas region found in 6227 genera and 355 plant families. The number of species reported corresponds to 33% of the world total.

and Mesoamerica **(Figure 3.4a)**. For mammals and amphibians, the highest richness is found in the Andes, the coastal northwest of South America and the Atlantic coast of Brazil; plants **(Figure 3.4a)** reach their highest richness in Mesoamerica, the Andes and other regions of South America. Avian richness shows peaks in both the lowlands and parts of the Andes. Evidence from ants (Dunn *et al.,* 2009) and soil fungal communities (Tedersoo *et al.,* 2014) suggests that these taxa may also reach their peak diversity in tropical regions, although considerable gaps exist in spatial sampling for these groups. Outside of the tropics, amphibians and plants show moderately high species richness in the southeastern USA **(Figure 3.4. a)** (Buckley & Jetz, 2007), and plants and mammals both reach high or moderately high richness in the western USA.

In contrast to species richness, which is broadly congruent across taxa and reaches its peak in tropical South America, highest evolutionary distinctiveness is found outside of the tropics for all taxa **(Figure 3.4. b)**. For the taxa where information is available, this indicates that the regions where co-occurring species are more distantly related on average tend not to be found in the tropics. Among amphibians, high evolutionary distinctiveness is found in western North America and parts of Mesoamerica. Mammals have high evolutionary distinctiveness throughout the Americas, especially in the Mediterranean region of southwestern South America. Birds and plants both have hotspots of evolutionary distinctiveness at high latitudes, indicating that even in these regions where low numbers of species persist, the species that do occur are drawn from distinct branches across the tree of life. Birds also achieve moderately high

Figure ③ ④ **Terrestrial biodiversity across the Americas in amphibians, birds, mammals and plants, reported as: Ⓐ species richness (SR); Ⓑ evolutionary distinctiveness (ED); Ⓒ phylogenetic endemism (PE); and Ⓓ plant functional diversity (FD). Source: own representation.**



FD was measured as the variance in specific leaf area (log cm^2/g), seed mass (log g) and plant maximum height (log m) in 1 degree latitude and longitude grid cells using BIEN 2 and TRY Data. The red end of the color spectrum indicates greater SR, ED, PE and FD. Vertebrate metrics were calculated in 108x111 km cells; plant richness, ED, PE and FD in 100x100 km cells. A quantile color scale that emphasizes variation in lower values is used for species richness and PE. Species distributions: Birds, BirdLife International & NatureServe (2012); amphibians and mammals, IUCN, (2009); plants, Botanical Information and Ecology Network (BIEN 2) database, Enquist *et al.* (2016); Maitner *et al*. (2017). Phylogenies: Mammals, Fritz *et al*. (2009); birds, Jetz *et al*. (2012); amphibians, Pyron (2014); plants ED, BIEN 3 phylogeny, Maitner *et al*. (2017); plants PE, Zanne *et al*. (2013) (trimmed to genus). R software (R Development Core Team, 2017) and picante package (Kembel *et al*., 2010); Nipperess and Wilson (2017) were used for calculations of the phylogenetic metrics and the R packages raster (Hijmans, 2016) and letsR (Vilela & Villalobos, 2015) were used to create the rasters.

evolutionary distinctiveness in the tropical lowlands of South America. Overall, these trends indicate that subtropical, temperate or boreal regions can be rich in certain dimensions of biodiversity. This, of course, does not mean that the tropics have less overall evolutionary diversity, but rather that tropical species often co-occur with many close relatives, reducing their evolutionary distinctiveness.

In all taxa, high phylogenetic endemism occurs in Mesoamerica and in parts of tropical South America, particularly the coastal northwest and tropical Andes **(Figure 3.4. c)**. Amphibians, mammals, and plants have further hotspots of phylogenetic endemism in the western USA, and amphibians and plants also have high phylogenetic endemism in the southeastern USA. Central and part of southern Chile also stand out for some groups. With some deviations, geographic patterns of phylogenetic endemism in these particular areas of the Americas tend to mirror species richness, signifying overall that they generally house large numbers of evolutionary distinct species and lineages not found elsewhere. Such areas are worthy of special concern in conservation decision-making but are sometimes located where protection measures are still poor (3.5.2).

Variation in functional traits, a measure of functional diversity, can tell us about the diversity of ecological adaptations among a set of organisms and the potential of particular ecosystems to adjust to environmental change. Data are available on three functional traits for plants; specific leaf area, seed mass and plant height. Specific leaf area (the area of a leaf divided by its dry weight) is tightly linked to photosynthetic rates and nutrient content. It is indicative of the life history strategy of the plant along a spectrum ranging from rapid growth and competitive resource capture to slow growth and stress tolerance (Wright *et al.,* 2004). Seed mass is indicative of reproductive and dispersal strategy (Leishman *et al.,* 1995; Moles *et al.,* 2005), and plant height is a critical indicator of life history, indicating growth form and habit (Loehle, 2000). These three traits are important for understanding major axes of variation in plant function and ecological strategy (Westoby, 1998). As with species richness, we tend to see the greatest diversity in seed mass and height of vascular plants in tropical regions of the Americas (Lamanna *et al.,* 2014) **(Figure 3.4. d)**. Nevertheless, temperate regions tend to be enriched in functional diversity for specific leaf area relative to tropical areas **(Figure 3.4. d)**. This might reflect a tendency to retain more diversity in leaf economic strategies under harsher and less equitable climatic conditions (Lamanna *et al.,* 2014; Swenson *et al.,* 2012). Variation in different plant functional traits is maximized in different regions. Likewise, different components of diversity are highest in different regions and these patterns vary by taxonomic group. As a consequence, conservation efforts across regions will be crucial for maintaining both the diversity of ecological strategies we observe in plants and the full spectrum of biodiversity across the tree of life, the basis of many NCP.

3.2.2.3 Patterns and trends in alien and invasive alien species

Status. We define alien species as species that become distributed beyond their native ranges intentionally or unintentionally aided by humans. The introduction and spread of alien species in the Anthropocene has led to greatly heightened levels of dispersal of organisms around the globe. Invasive alien species are alien species that modify ecosystems, causing potential damage to the environment, human health, and consequently, the economy. The distinction between these two categories is not always clear because designating an alien species as an invasive species requires detailed studies and objective and comparable criteria. The economic damage caused by alien invasive species can be severe. For example, globally, invasive insects (some of which carry diseases) are estimated to cost a minimum of $70.0 billion per year, while associated health costs exceed US$6.9 billion (Bradshaw *et al.,* 2016). Control of invasive species requires knowledge of global and local introduction trends and distinguishing harmful alien species from the more benign ones; that said, not all alien species are harmful **(Table 3.2)**.

Comprehensive data on naturalized alien species for the Americas is available for plants and birds. Currently the North American (which includes Greenland and Mexico) and the South American (which includes Mesoamerica south of Mexico and the Caribbean) biogeographic regions are home to 3,513 (39%) and 1,806 (20%) respectively of the world's 9004 plant species that have been introduced from one continent to another (Van Kleunen *et al.,* 2015, and personal communication). Additional intra-continental plant movements beyond their natural ranges within North and South America, bring the total numbers of alien records to 5,958 and 3,117, respectively. North America has been a much larger donor of alien plant species to other continents than has South America; additionally North America, as defined by IPBES, is one of the most heavily invaded areas of the world (Van Kleunen *et al.,* 2015). The Caribbean subregion is also strongly invaded in relation to its land area (see also **Figure 3.5**, where there are many plant species).

Some 3,661 alien bird introductions (first known occurrence of a given species in a given country) were reported across the globe from 1500 to 2000 (Dyer *et al.,* 2017). Relative to other regions the Americas, particularly the North American and Caribbean subregions, support large numbers of alien birds (Dyer *et al.,* 2017). Reports of introduced birds are currently lacking in some tropical areas in northern South America (Dyer *et al.,* 2017).

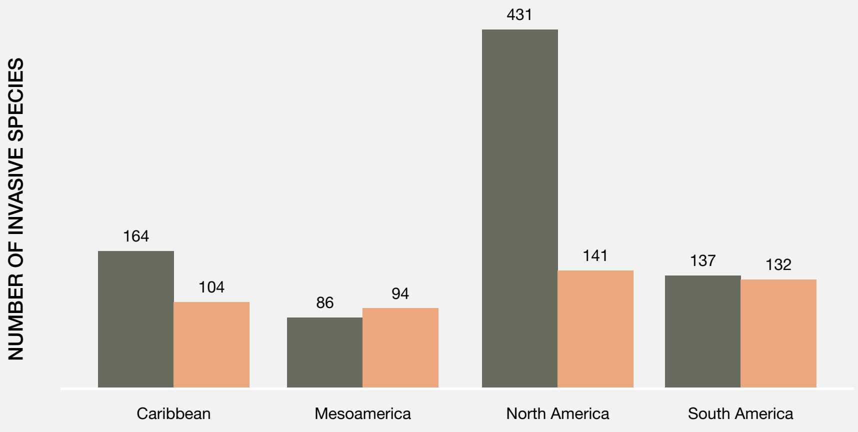
Table ③ ② **Multiple effects of mostly recent terrestrial alien introductions in the Americas. Alien species can have both negative and positive impacts on humans and biodiversity. See Chapter 4 for additional examples. 🔴 = negative impact; 🟢 = positive impact.**

Sources: 1 Morales *et al.* (2013); Aizen *et al.* (2014); 2 Sanguinetti & Singer (2014); 3 Dangles *et al.* (2008); 4 Herms & McCullough (2014); 5 Martyniuk *et al.* (2015); 6 Peña *et al.* (2008); Taylor *et al.* (2016); 7 Zamora Nasca *et al.* (2014); 8 García *et al.* (2015); 9 Baruch & Nozawa (2014); 10 Svriz *et al.* (2013); 11 Pauchard *et al.* (2009); Barros & Pickering (2014); 12 Muñoz & Cavieres (2008); 13 León & Vargas-Ríos (2011); 14 Díaz-Betancourt *et al.* (1999); 15 Rodrigues da Silva & Matos (2006); 16 Choi (2008); 17 Jiménez *et al.* (2014).

	Insects
🔴	European *Bombus terrestris* reduces fitness of native plants and replaces the native bumblebee, *B. dahlbomii*.[1]
🟢	Introduced bees increase fitness in some native orchids.[2]
🔴	Three potato moths reduce crop harvest in the northern Andes.[3]
🔴	The Asian emerald ash borer beetle (*Agrilus planipennis*) has killed millions of ash trees in N. America.[4]
	Plants
🔴	Seed set on the native *Austrocedrus chilensis* is reduced by interference of introduced conifer pollen.[5]
🔴	Encroachment of exotic plantation tree species into native forests reduces habitat area.[6]
🔴	*Ligustrum lucidum* reduces soil water availability in secondary forests.[7]
🔴	*Teline monspessulana* increases fire proneness in native forests.[8]
🔴	Aggressive *Syzygium jambos* interferes with natural regeneration in abandoned coffee plantations.[9]
🟢	*Rubus rubiginosa* acts as a nurse plant for regeneration of native forest trees on drier sites.[10]
🔴	Non-native species on trails homogenize the floras of protected areas, reducing landscape value.[11]
🔴	*Taraxacum officinale* reduces pollinator visits on native species in the high Andes of central Chile.[12]
🔴	*Ulex europaeus* invades páramos, displacing native species and possibly harming water supply.[13]
🟢	Introduced weeds in the Americas include many edible species.[14]
🔴	Post-fire invasion by *Pteridium aquilinum* in the Atlantic rainforest hinders natural forest regeneration.[15]
	Mammals
🔴	North American beaver affects forest hydrology and forest regeneration in Tierra del Fuego.[16]
🔴	American Mink preys on the eggs of water birds and the iconic Magellanic woodpecker.[17]

Figure ③ ⑤ **Invasive alien plant and animal species considered to threaten native biodiversity and ecosystems listed in the Global Invasive Species Database (GISD) that are found in the four subregions of Americas.**

Data include a few marine and freshwater species. Grey bars are species that have been reported somewhere in that subregion as being strongly invasive; orange bars are additional species listed in GISD that occur in the subregion but that are not necessarily invasive there or whose invasive status is unknown. Source: Data from Global Invasive Species Database http://www.iucngisd.org/gisd/. Accessed March, 26 2017.



Although much progress has been made, we currently cannot say how many alien species in the Americas are harmful. Comprehensive risk analyses are lacking in most countries. In general, the number of harmful species is likely to higher than currently visualized because detailed studies are lacking and due to the fact that many potentially strongly invasive species will be still in a lag phase. In Mexico, a comprehensive risk analysis found 41% of 472 species (including aquatic species) analyzed out of a total of 1,683 potentially invasive species to be very high-risk species (Gonzalez Martínez *et al.,* 2017).

Across all taxonomic groups, some 521 species considered to be harmful to biodiversity in Global Invasive Species Database are known to be strongly invasive somewhere in the Americas. North America has far more such species than the other subregions, but for its small land area, the Caribbean is clearly very susceptible to invasion **(Figure 3.5)**. Additional species found in Global Invasive Species Database that are not considered to be invasive at the moment in a particular subregion could eventually become invasive **(Figure 3.5)**. For the World's 100 Worst Invaders found in Global Invasive Species Database, 78% have been recorded to occur in at least one subregion of the Americas. Beyond invasive species that harm biodiversity, many alien species have negative impacts on agriculture and forestry. For example, in Brazil, more than 500 species of alien pathogenic fungi, 100 viruses, 25 nematodes and one protozoan attack crops and reduce crop production an estimated 15% (Pimentel, 2002). Chapter 4 provides more information on the effects of harmful invasive species and on their drivers.

Recent trends. Globally, 37% of all recorded naturalized aliens from a wide spectrum of taxonomic groups were recorded for the first time as recently as 1970¬–2014 (Seebens *et al.,* 2017). This signifies that invasion risk is currently high and with increasing globalization will not cease. For the Americas, rates of appearance for different groups have varied over time, with a tendency for steeper early climbs and an earlier tendency to decline in North America than in South America **(Figure 3.6)**. Insects showed a very rapid rate of increase in South America as of the 1950s.

For birds, half of the naturalized alien introductions worldwide occurred after 1956, in concert with increasing globalization and economic growth. As with plants, early bird introductions came mostly from Europe. However, more recently the Indian subcontinent, Indochina, and sub-Saharan Africa have become important sources of alien birds (Dyer *et al.,* 2017). For the Americas, as of 1983, at the country level, 102 new alien birds were registered for the Caribbean subregion, 8 the Mesoamerican subregion, and 19 for South America. At the individual state (USA) or province (Canada) level, 163 were recorded for the North America subregion - calculated from Supplementary material (Dyer *et al.,* 2017).

Overall, alien introductions and their spread are likely to continue in the Americas (Seebens *et al.,* 2017) opening the door to additional negative effects on biodiversity, forestry and agriculture. Modeling suggests that many established alien species in the Americas do not yet fully occupy their climatic niches (Arriaga *et al.,* 2004; Peña-Gómez *et al.,* 2014) and thus can be expected to expand further, facilitated by disturbance. We are currently in a modern era of assisted dispersal heightened by global travel, tourism, and the introduction of pets and pest-carrying plant parts (see Chapter 4). A dramatic example of how alien species have increased recently in relation to increasing vector availability is seen in the Galápagos Islands **(Box 3.1, Figure 3.7)**.

Knowing which geographic areas are likely to receive more alien species is useful for the development of early-warning systems. According to a recent analysis of current invasion vectors and environmental susceptibility to invasion, the threat of invasion is very unevenly distributed across the Americas (Early *et al.,* 2016) **(Figure 3.8)**. The dominant invasion vectors differ between high-income countries (imports, particularly of plants and pets) and low-income countries (air travel). Climate change, further biome transformation (e.g. 3.4.1.6) and increased fire frequency (e.g. 3.4.1.4) are expected to hasten the spread of invasive alien species once established. Given that strongly invasive alien species, in addition to signifying economic costs, have been the cause of many extinctions (Bellard *et al.,* 2016), alien species are a component of biodiversity that requires attention.

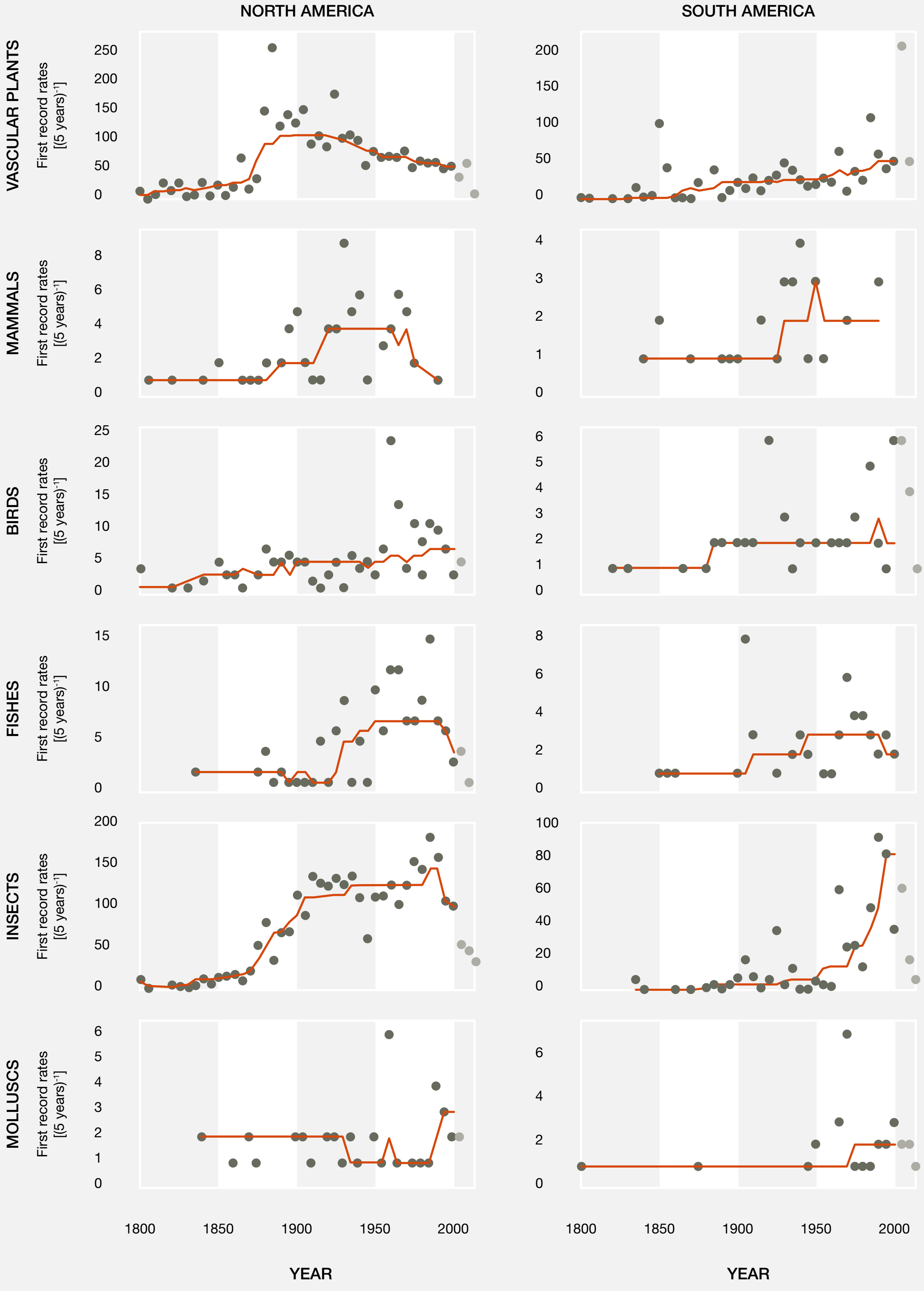
3.2.3 Status and trends of freshwater biodiversity

3.2.3.1 Patterns of diversity for taxonomic groups

Taxonomic groups. The Americas hold the most diversified freshwater fish fauna in the world, with 1,213 species in the North American biogeographical region and 4,035 in the South American biogeographical region for a world total of over 13,600 species (Burkhead, 2012). Other freshwater taxonomic groups of note include crayfishes, with high diversity in the southeastern USA (Crandall & Buhay, 2008); amphibians, with nearly half of all salamander species found in North America; 40% of all water-dependent frog species found in the Neotropical realm (Vences & Köhler, 2008); 11 of the world's 23 crocodilian species (Martin, 2008); the vast majority of the world's temperate freshwater turtle species in North America (Bour, 2008);

Figure 3 6 **Trends in the appearance of alien species in North America and South America from 1800 to 2000.**

Source: Based on data in the global alien species first record database www.dx.doi.org/10.12761/SGN.2016.01.022. Accessed: August 25, 2016.



Box 3 1 Alien species in the Galápagos Islands.

Described by UNESCO (United Nations Educational, Scientific and Cultural Organization) as a "living museum and showcase of evolution", the Galápagos Islands, a World Heritage site, are today a major tourist attraction. Some 1,476 of 1,579 alien terrestrial (and marine) species have become established on the islands (Toral-Granda *et al.,* 2017); 50% of aliens were first reported after the 1990s and just over 50% were introduced through unintentional human assistance. The rate of introduction represents an average of around three species per year. Geographic origins and modes of introduction have diversified over time, reflecting the increase in human influence on the islands. In general, islands are prone to invasions. The mythical oceanic Robinson Crusoe islands are also strongly invaded (Wester, 1991).

Figure 3 7 **Normalised decadal values for the cumulative number of alien species, residents and tourists, in the Galápagos Islands, ecuador. Source: Based on data given in supplementary material in Toral-Granda *et al.* (2017).**

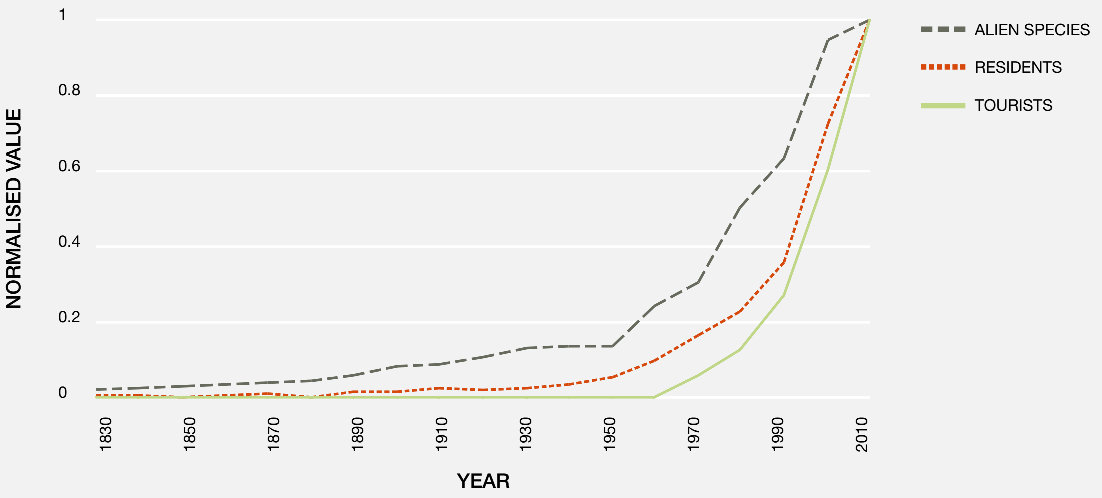
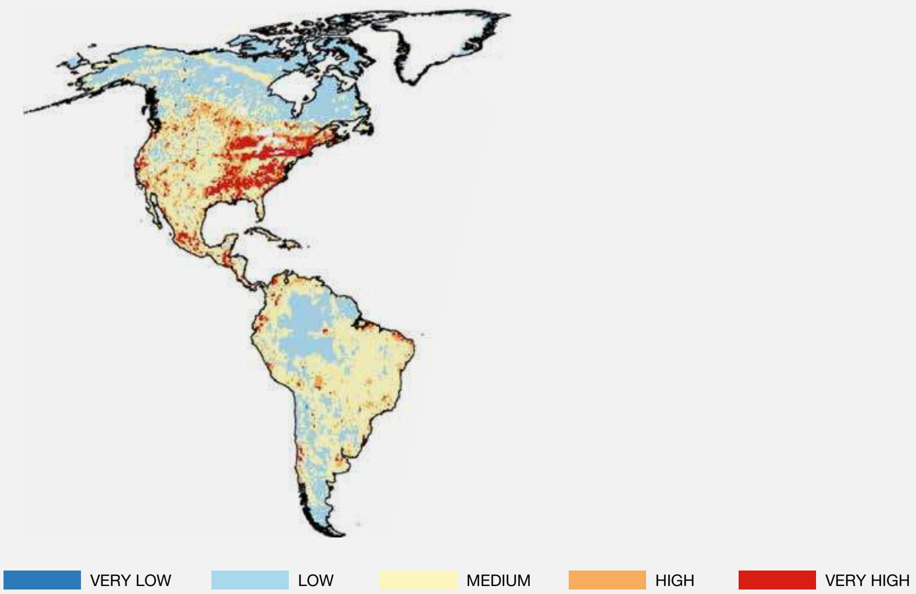


Figure 3 8 **Invasion threat across the Americas in the 21st century. Source: Modified from Early *et al.* (2016).**



the most diverse freshwater bivalve fauna globally also in North America (Bogan, 2008); and an especially diverse assemblage of decapods in Central America (Wehrtmann *et al.,* 2016).

Freshwater species contribute NCP in numerous ways. Freshwater mussels cleanse water (Nobles & Zhang, 2011). Fish regulate nutrients in water (Holmlund & Hammer, 1999). North American Pacific salmon transfer nutrients from marine to freshwater realms when they die en masse after migrating upstream (Flecker *et al.,* 2010). In the Amazon, Orinoco, and parts of Central America, frugivorous fish disperse seeds for floodplain forest trees (Flecker *et al.,* 2010). An estimated 450,000 tons of riverine fish are landed each year in the Amazon, with important implications for the food security of local people (Junk *et al.,* 2007; McIntyre *et al.,* 2016). However, riverine fish catch is estimated to be low in large North American rivers like the Mississippi, where recreational fisheries dominate commercial or artisanal fisheries (McIntyre *et al.,* 2016). Overall, reported inland fish catch in the Americas is low compared to other regions (Bennett & Thorpe, 2008).

Status. Much freshwater biodiversity in the Americas is threatened, derived largely from catchment land use, water use and direct habitat alterations (Vörösmarty *et al.,* 2010) (see Chapter 4 for discussion of drivers). Some 23% for the Nearctic and 22% for the Neotropics of freshwater mammals, amphibians, reptiles, fishes, crabs and crayfish collectively fall here. The well-studied North American biogeographical region freshwater fish fauna in the 20th century had the highest extinction rate worldwide among vertebrates (Burkhead, 2012). Some 72% of freshwater mussels in the USA and Canada were considered imperiled as of the early 1990s (Williams *et al.,* 1993). In Central America, 42% of ca. 500 known amphibian species have been assessed as threatened, with stream-dependent species at particular risk (Whitfield *et al.,* 2016). Regions with low threat are remote areas in northern Canada, Alaska, and the Amazon.

Recent trends. In North America (including Mexico), since 1989, the number of threatened freshwater fishes has increased by 25%; extinctions peaked after 1950 with 7.5 extinct taxa per decade post-1950 (Burkhead, 2012). This level of extinction gives reason for great concern. In the Caribbean native fish species continue to decline and be extirpated with dam building, pollution and overharvesting exerting considerable pressure (Cooney & Kwak, 2010). In the 1970s, a noticeable decline in populations of freshwater turtles in the Amazon was observed (Eisemberg *et al.,* 2016). Amphibian population declines in Mesoamerica and South America have been documented largely beginning in the 1970s–1990s, with the majority in the 1980s (Young *et al.,* 2001). Freshwater mussel extinctions have been documented in the USA from the beginning of the 20th century, with a peak of eight extinctions in the 1920s through the 1940s and seven documented extinctions in the 1970s (Haag, 2009).

3.2.3.2 Patterns and trends in alien and invasive species

Status and recent trends. Data on freshwater alien species is scattered, making it difficult to provide an overall picture for the Americas and its subregions. Where databases are available, numbers of alien species can be high, as seen in over 1000 species in the USA (plants excluded) (Fuller & Neilson, 2015) and 50 species of fishes (including marine species) in Mexico (Mendoza & Koleff, 2014) (see also **Box 3.2**). The impacts of aquatic alien species are multiple and can be severe **(Table 3.3)**. The spread of some aquatic invasive alien species, moreover, has been very rapid, leaving cause for concern.

In North America, alien freshwater species have been arriving for close to two centuries and continue to arrive. Some of the earliest known introductions occurred in the late 1800s when fish were transported from coast to coast (Benson & Boydstun, 1999). Crayfish and other freshwater organisms were moved from the southeastern USA to the western USA to serve as game species or forage for game species. Temperate piscivorous and carnivorous fish species have been reported to cause much harm to native fish fauna, especially in Cuban freshwaters, Lake Atitlán (Guatemala) and Lake Titicaca (Bolivia and Perú) (Revenga & Kura, 2003).

The zebra mussel (*Dreissena polymorpha)*, native to Europe, and the Asian clam *(Corbicula fluminea)* are estimated to cost the $1 billion a year, largely through impacts to infrastructure (Pimentel *et al.*, 2005). Their spread has been recent, with the first established zebra mussel population recorded in the USA in 1988 (Benson, 2012). Some freshwater invasive species in South America have also spread very rapidly. For example, the exotic freshwater water-fouling mussel, *Limnoperna fortunei*, was introduced into Río de la Plata estuary in 1991; from there it spread at a rate of up to 250 km year^{-1} and is now found in freshwater systems in Argentina, Uruguay, Paraguay, Brazil, Bolivia (Darrigran *et al.,* 2012; Darrigran & Ezcurra de Drago, 2000; Oliveira *et al.,* 2015). This mussel, which is similar to invasive Dreissinids in North America, has altered benthic communities and is predicted to expand further. This example shows that insufficient measures to prevent the introduction of invasive aquatic species can have severe consequences. The most invaded freshwater system in the Americas, and a warning to what can happen without adequate control from the beginning, are the Great Lakes of North America **(Box 3.2, Figure 3.9)**. Other examples of freshwater invasions are shown in **Table 3.3**.

Table 3 3 Multiple effects of freshwater invasive species in the Americas. See Chapter 4 for additional examples. 🔴 = negative impact; 🟢 = positive impact.

Sources: 1 Junk (2007); 2 Thompson *et al.* (1987); 3 Brown & Maceina (2002); 4 Perry *et al.* (2001); 5 Pyron *et al.* (2017); 6 Wilson *et al.* (2011); 7 Howard (2016); 8 Leal-Flórez (2008); 9 Montecino *et al.* (2014); 10 Villamagna & Murphy (2010); 11 Bacheler *et al.* (2004).

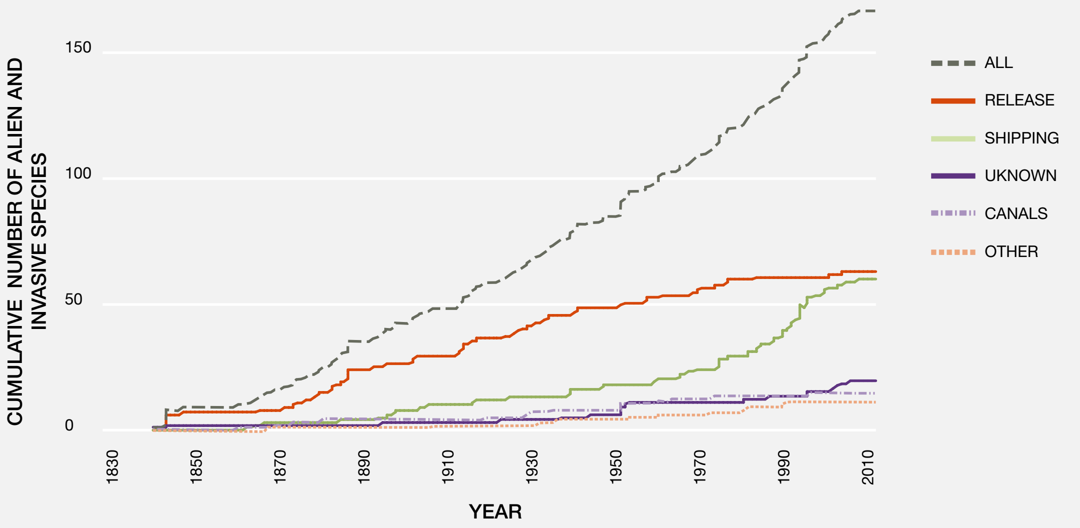
	Invasive species
🔴	Introduction of rainbow trout in Lake Titicaca decreased native fish food supply.[1]
🔴	Purple loosestrife (*Lythrum salicaria*) has reduced the biomass of 44 native plants and dependent endangered wildlife species.[2]
🔴	Infestations of the aquatic weed hydrilla *(Hydrilla verticillata)* have reduced angling up to 85%.[3]
🔴	The rusty crayfish (*Orconectes rusticus*), native to the Ohio River basin, is spreading in the USA and replacing native species.[4]
🔴	Asian carp contributed to modifications in native fish assemblages in the Wabash River, USA, likely by competing with native planktivore / detritivore fishes.[5]
🔴	The cane toad (*Bufo marinus*) has spread to the Caribbean and is killing the threatened endemic Jamaican boa *(Epicrates subflavus)*.[6]
🔴 ?	Hippos introduced into Colombia are now multiplying and may contribute to eutrophication via their waste.[7]
🔴 🟢	Accidental introduction of *Oreochromis niloticus* into Colombia's Santa Marta estuary has provided local fishermen with a source of income during short periods of low salinity, when native fish catches drop. However, this same species has had negative impacts in many other American ecosystems.[8]
🔴	The alien diatom *Didymosphenia geminata* recently expanded in southern Chile and Argentina greatly reducing aesthetic value of lakes and streams.[9]
🔴 🟢	The water hyacinth (*Eichhornia crassipes*), native of lowland tropical America, has become invasive in many countries of the region with mostly negative effects on waterways, but some positive effects on biodiversity.[10]
🔴	In Puerto Rico, there is a significant overlap in diet between the native *Gobiomorus dormitor* and largemouth bass *Micropterus salmoides* introduced from North America.[11]

Box 3 2 The Great Lakes history with invasive species.

The Great Lakes in North America have accumulated an excess of 165 alien species in what is still an ongoing process **(Figure 3.9)**. Some species have had significant negative impacts on aquatic ecosystems (Higgins & Vander Zanden, 2010). Among the most damaging is the sea lamprey *(Petromyzon marinus)*, which appeared in the 1830s and spread throughout the Great Lakes during the 20th century, impacting several fisheries. Zebra mussels and quagga mussels, first detected in the late 1980s, create dense colonies that harm ecosystems, harbors and waterways and clog water intakes in water treatment facilities and power plants.

Figure 3 9 Trends in the accumulation of alien and invasive species in the North American Great Lakes over time.

The upper line of the graph shows total cumulative number; the other lines show the contribution from various vectors. "Release" includes both intentional and unintentional; "other" includes railroads, highways, aquaria, and baitfish. Source: Data compiled from Kelly (2007), Kelly *et al.* (2009) and Ricciardi (2006).



3.2.4 Marine biodiversity

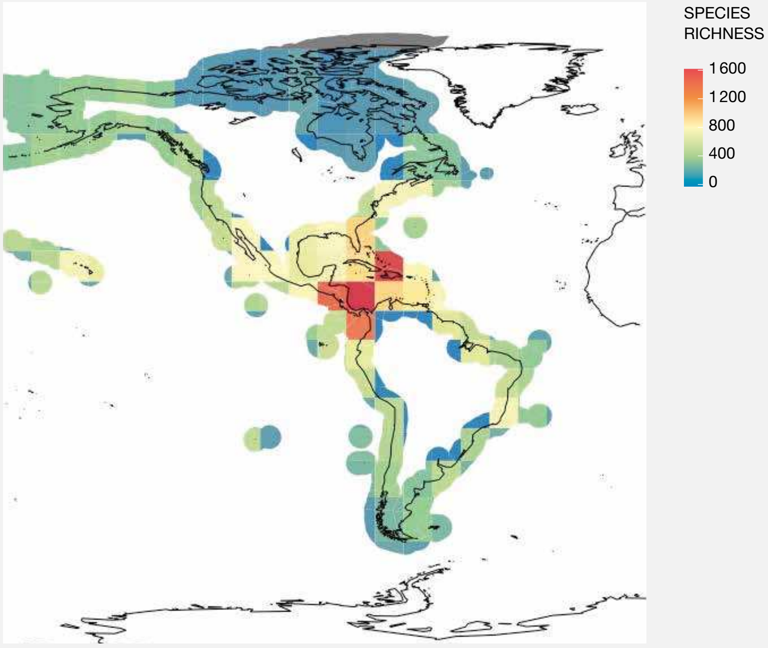
3.2.4.1 Patterns of diversity for taxonomic groups

Status. Marine life in the Americas is found in the Atlantic, Pacific and Arctic oceans, and in the Caribbean Sea. Atlantic and Pacific offshore and deepwater areas (> 200 m) encompass a range of habitats with a wide diversity of species (OBIS, n.d.) **(Figure 3.10)**. Exceptional diversity is being revealed for the oceans. Including all taxonomic groups (except bacteria and phytoplankton), 12,046 marine species have been found in the Caribbean realm (Miloslavich *et al.,* 2010), 10,201 in the Humboldt Current System, 6,714 in the Tropical East Pacific, 9,103 on the Brazilian shelves, 2,743 in the Tropical West Atlantic, and 3,776 on the Patagonian shelf in South America (Miloslavich *et al.,* 2011). These numbers are considered to be conservative (see 3.6). Marine mammals in the Americas include 74 cetacean species, 22 pinnipeds, 3 sirenians, 3 mustelids and the polar bear. Additionally, six of the world's seven sea turtles and more than 400 chondrichthyan species occur in the Americas. The Arctic Ocean, in its waters, ice and seafloor, hosts unique biodiversity of many thousands of species, including mammals, seabirds, fish, invertebrates, and algae (Gradinger *et al.,* 2010) in a rapidly changing environment (see Chapter 4). The Caribbean basin deep-sea species database (OBIS. n.d.) lists 1,530 species from 12 phyla, but much more work is needed (Miloslavich *et al.,* 2010). The Caribbean Sea holds most of the Americas' biodiversity associated with coral reefs.

In many species of coastal fish, mangroves, seagrasses, squids, non-oceanic shark species, and corals, diversity generally peaks near the equator (Tittensor *et al.,* 2010). In contrast, pinniped (seals and sea lions) diversity is highest in polar regions. Cetacean species diversity peaks in the subtropics in both oceans, and is highest on the Atlantic coast of Argentina (Tittensor *et al.,* 2010). Shark species peak in biodiversity between 30 and 40 degrees N and S; southeastern Brazil and the southeastern USA are considered global hotspots of shark biodiversity with high species richness, functional diversity, and endemism (Lucifora *et al.,* 2011). Brazil alone has 31 endemic shark species (Lucifora *et al.,* 2011). Seaweed biodiversity peaks in temperate regions around 35 degrees latitude N and S in the Pacific (Gaines & Lubchenco, 1982), although it is also highly diverse in the Caribbean (Kerswell, 2006). Kelp diversity is greatest in colder parts of both oceans, and algal diversity reaches its nadir in the southeastern Atlantic (Argentina, <100 species) (Kerswell, 2006). The America's host hundreds of thousands – if not millions – of invertebrate species; their biogeographic patterns are still poorly known (Sala &

Figure 3 10 **Species richness across coastal fishes, marine mammals, mangroves, corals, foraminiferans, euphausids, cephalopods, tuna and sharks in the coastal ecoregions of the Americas.**

Source: own representation from supplementary data in Tittensor *et al.* (2010).



Knowlton, 2006). Invertebrate diversity within many distinct taxonomic groups generally (with some exceptions) follows the latitudinal trend of increasing species diversity per area at lower latitudes, as seen in South American crabs on both coasts (Astorga *et al.,* 2003), and fish (Rohde *et al.,* 1993), molluscs (Roy *et al.,* 1998) and foraminifera (Rutherford *et al.,* 1999) in North America. Different biogeographic regions, reflecting major oceanographic features, have distinct invertebrate species assemblages off South America, North America, the Arctic and the Caribbean. This pattern is exemplified by the spatial distribution of the estimated 1,539 species of echinoderms inhabiting Latin America (Pérez-Ruzafa *et al.,* 2013). The Western Atlantic and the coast of South America host an exceptionally high diversity of the world's 2064 ophiuroid echinoderms (335 species) with high rates of endemism (Stöhr *et al.,* 2012).

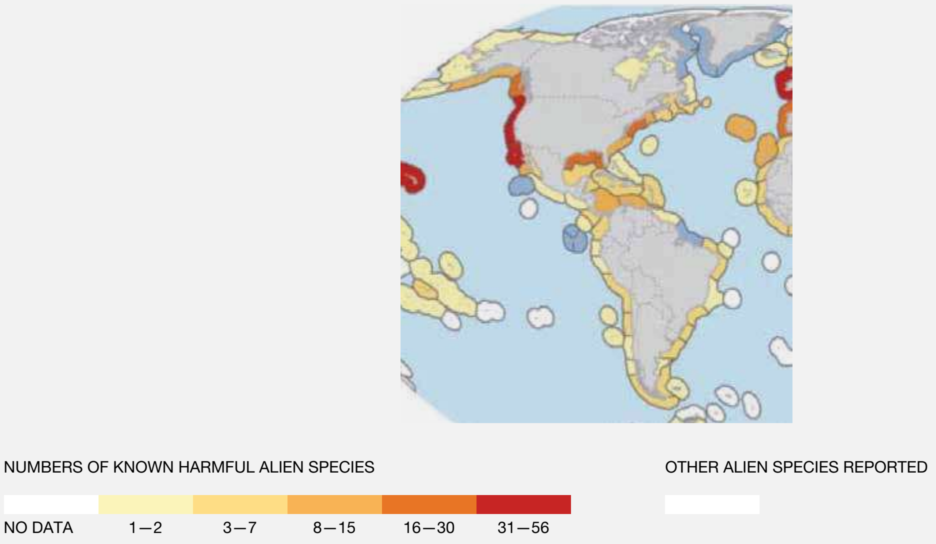
3.2.4.2 Patterns and trends in marine invasive species

Status. Pagad *et al.* (2017) document 2,103 introduced marine species worldwide, of which 305 are considered strongly invasive. According to this source, the North American continent has 388 alien marine species (70 invasives); Mexico, 94 (6 invasives); the Caribbean Sea, 47 (5 invasives); Brazil, 49 (9 invasives); Argentina, 25 (1 invasive) and Greenland, 1 (not invasive).

In general, North American waters are more heavily invaded than those of other subregions **(Figure 3.11)**. However, differences between North and South America are less evident when considering invasive algae (including native invasive species) (Figure 1.A in Seebens *et al.,* 2016). Miloslavich *et al.* (2011) report more alien species at cooler latitudes in South America, but this difference might be influenced by sampling density at other latitudes. Most alien and invasive invertebrate and algal species are found in bays and estuaries, with few occurring on outer coasts (Ruiz *et al.,* 2015). San Francisco Bay, USA, may the most invaded marine region on Earth, with more than half its fish and most of its benthic invertebrates being non-native (Cohen & Carlton, 1998).

As in terrestrial habitats, recent invasions may not be detected for many years. Controlling the introduction of marine species and their impacts, however, is far more difficult than controlling terrestrial and freshwater species given that they are not so obvious. Moreover, introduced marine species can transport many other alien species. For example, the American oyster, *Crassostrea virginica*, was introduced to the Pacific coast to supplement stocks of local species. Oyster drills (*Urosalpinx cinerea*), slipper shells (*Crepidula fornicata* and *C. plana*), polychaetes (*Polydora cornuta*), and cordgrass (*Spartina alterniflora*) may have been introduced with them (Ray, 2005). Worms used for live bait (*Glycera dibranchiata*) are shipped packed in seaweed, which carries many potentially invasive organisms such as snails, crabs, isopods, insects, plants and algae (MD Sea Grant, n.d.). Some species arrive by multiple mechanisms, e.g. the Chinese mitten crab (*Eriocheir sinensis*) may have arrived in ballast water and in live trade as food (Ruiz *et al.,*

Figure 3 11 **Relative invasion levels in marine habitats across the Americas based on the number of species with high ecological impact scores per ecoregion. Source: Modified from Molnar *et al.* (2008).**



2000) or as pets from the aquarium trade (Dee *et al.,* 2014; Smith *et al.,* 2008). Overall strategies to deal with marine invasion require international collaboration.

Recent trends. As is occurring in terrestrial and freshwater systems, the spread of alien species in marine systems of the Americas continues. Marine species once established, can spread very rapidly. The Asian green mussel *Perna viridis*, native to the Indo-Pacific, was first observed in Caribbean waters in 1990 (Agard *et al.,* 1992). Within 10 years, green mussels were found along the coasts of Venezuela, Jamaica and Tampa Bay, Florida (Benson *et al.,* 2001; Buddo *et al.,* 2003; Ingrao *et al.,* 2001; Rylander *et al.,* 1996). Rates of marine introduction seem to be increasing in some places. For example, Cohen and Carlton (1998) estimated that the San Francisco Bay and Delta ecosystem has received about one new invasive species every 36 weeks since 1850: as of 1970, the rate increased to one new species every 24 weeks. A huge number of marine species (280) were recently found to have crossed the Pacific from Japan to the west coast of North America on debris swept to sea by the 2011 tsunami (Carlton *et al.,* 2017), warning that increasing amount of debris in the oceans are a potential source of invasive marine species. Invasions related to human food production are a current concern. Non-native shrimp (Asian tiger shrimp, *Penaeus monodon*), oysters (*Ostrea edulis*) and Atlantic salmon (*Salmo salar*) cultured in marine enclosures, have generated concern over disease and other impacts that might arise from their escape.

3.3 BIODIVERSITY AND PEOPLE

3.3.1 Cultural diversity: How many indigenous groups and languages are represented in the Americas?

Cultural diversity is defined as the spiritual, material, intellectual, and emotional processes and dynamics developed by a social group. It is composed of livelihoods, values, traditions, knowledge, and beliefs centered on nature (Berkes, 2008; Posey, 1999; UNESCO, 2002). Traditional cultural and spiritual values provide the context in which environmental stewardship can be nurtured (Kothari, 2009; Robson & Berkes, 2012).

In the 1980 census, half of the Latin American countries quantified their indigenous populations based on linguistic criteria (CEPAL, 2014; Correa, 2011). As of 2000, 16 out of 19 countries identified their indigenous populations on the basis of self-determination, common origin, territories, and linguistic and cultural dimensions (Bartolomé, 2006; CEPAL, 2014; F. Correa, 2011; International Labour Organization, 1989). Based on these criteria, in 2014, 826 native populations were legally recognized in the Americas (305 in Brazil, 102 in Colombia, 85 in Peru, 78 in Mexico, 39 in Bolivia) and 15 First Nations populations were recognized in Canada and the USA (United Nations Development Programme, 2014). In Latin America in 2010, indigenous peoples numbered about 45 million. Mexico is home to 17 million (15.1% of the total population); Peru, 7 million (24%); Bolivia 6.2 million (62.2%) and Guatemala, 5.8 million (41%) (CEPAL, 2014). In Canada, First Nations population was less than 1 million (2.6% of the total population) and in the USA, 5.1 million (1.7%) (United Nations Development Programme, 2014). In the 2011 census, aboriginal peoples in Canada totaled 1.4 million, or 4.3% of the population. Some 600 First Nations governments or bands with distinctive cultures, languages, art, and music were recognized. In the USA, 566 distinct Native American tribes are recognized by the government as of 2016, including indigenous peoples of Alaska and Hawaii (Federal Register, 2016). In 2010, the US Census Bureau estimated that about 0.8 or 0.9% of the USA population was of native American descent; one-third of that population lives in California, Oklahoma and Arizona (USA quickfacts census, 2012).

Languages underpin ethnobotanical, ethnozoological and ethnoecological knowledge and guides a people´s spirituality and worldview. Indigenous and local knowledge (ILK) is transmitted by language and thus conserving languages is crucial for understanding biodiversity as it relates to human well-being. Over 1,000 indigenous languages are spoken across the Americas. Most of the indigenous American languages in North America are in trouble, dying or already extinct. Other subregions also face language extinction but are somewhat more stable (Chapter 2, Table 2.2).

3.3.2 Cultural and biological diversity: Traditional knowledge and worldviews among the indigenous communities of the Americas

Traditional knowledge. “Traditional knowledge is the ancestral wisdom and the collective and integrated knowledge that indigenous, Afro-descendants, First Nation peoples, and local communities share based in their praxis in the interrelationship human-nature, transmitted from generation to generation” (De la Cruz *et al.,* 2005). Biodiversity has significance to indigenous communities for human nature, culture and spirituality. Traditional knowledge is collective, intergenerational and linked to the right of free determination and worldview (De la Cruz, 2011; Robson & Berkes, 2012). These interrelationships constitute the biocultural heritage of indigenous people that is intimately related to their connection to land and sacred or spiritual places, and influence how people interact with and manage land. A good example is seen in the indigenous Menominee people who inhabit the Great Lakes region **(Box 3.3)**.

Worldview. The "worldview" is the structured group of diverse ideological systems by which a social group understands the universe and the order of systems, knowledge, and interrelationships with nature. (López-Austin, 1990). Recognition of worldview signifies appreciation for a system that has the potential to be less damaging to the environment than many current dominant practices. The worldview is interrelated with territory, nature, religion, politics and the economy (Zolla & Zolla, 2004). Most indigenous populations share principles that derive from their worldview, including the principle of reciprocity, the principle of correspondence between the micro-cosmos and the macro-cosmos and the principle of complementarity, in which the cosmos functions with all of its parts (Zolla & Zolla, 2004).

For the Otomi people, an indigenous group in Mesoamerica, worldview explains the universe; the origin and destiny of humanity; the origin of their territory and mountains as the source of fertility and force; the dialogues between humans and animals to seal protection; the creation of plants, health, and sickness as a unity among body, soul and land; and the circle of time and space (Galinier, 1997; Pérez, 2008). Humans are integrated with land, animals, plants, and mountains. Well-being consists of finding equilibrium among these parts. "To be fine is to dominate our soul (*ro mui*)" (Pérez, 2008). Among the Kichwa people in Ecuador, the Sumak Kawsay ("good living") is based on a communitarian space, continuous dialogue with Mother Nature or Mother Earth (pachamama), the conservation of ecosystems, different ways to produce knowledge by all members, social organization based on the principle of reciprocity and solidarity (minka, ranti-ranti, makikuna, uyanza). For Manuel Castro (ECUARUNARI, Ecuador), Sumak Kawsay implies social equity, justice, and peace (Houtart, 2014). For Eugenia Choque, *suma jakaña* means to achieve food sovereignty, and for Xabier Albó it denotes to "live together well" (Houtart, 2014). This worldview constitutes an alternative for development and a "cosmic ethic" (Gudynas, 2009, 2011; Houtart, 2014). Much can be learned from the worldview of indigenous peoples when it comes to sustainability and biodiversity conservation.

3.3.3 Domestication and use of biodiversity and agroforestry

Domestication. The northeastern USA, Mesoamerica, the Andean region of Peru, Ecuador and Bolivia, and the Amazon basin are widely recognized as primary sites of management and domestication of biological diversity in the Americas (Casas *et al.,* 2007; Chacón *et al.,* 2005; Clement *et al.,* 2010; Galluzzi *et al.,* 2010; Harlan, 1971; Kwak *et al.,* 2009; Parra & Casas, 2016; Perry *et al.,* 2007; Smith, 1994) (see also Chapter 2).

Many plants were domesticated in Mesoamerica (mainly 30 food species, such as maize, beans, tomatoes, cacao, squash, and chili), and the Andean region (potato, quinoa, squash, maize, beans, chili), Brazil, Paraguay (mate, pineapple, some nuts) (Harlan, 1961; Kloppenburg, 1991; Nemogá Soto, 2011) (see also Chapter 2). In the

Box 3 3 The Menominee Nation: an example of indigenous knowledge and practice.

The Menominee Nation is a nation of indigenous people of North America that has existed for thousands of years. Currently situated in Wisconsin (USA), it stewards one of the significant regions of contiguous vestiges of old growth hardwood forest that remain in the Great Lakes Region. The present-day Menominee reservation is only a fraction of the estimated 4.05 million hectares of ancestral lands accessed by the Omaeqnomenewak prior to European contact. Treaties with the USA government between 1817 and 1856 resulted in a large loss of land, down now to approximately 95,313 ha (Omaeqnomenew Masenahekan, 2004). Much of the Menominee forest is old growth due to efforts by early leaders to manage the resource sustainably in a time when land barons were harvesting what they perceived were unlimited supplies of timber. Some 68% of the region was covered by old-growth forests in the late 1800s (Frelich, 1995), but only about 1% of Wisconsin's old-growth forests remain today as a consequence of producing more than 8.26 million cubic meters of timber annually in the late 1800s. Guided by tribal leaders' philosophy for managing forests and processing of forest products, Menominee forested land provides economic benefits not only through sustainable timber harvesting and wood product manufacturing but also through access to culturally important plant and animal species and ecosystems. As a result, the Menominee forest is home to ecosystems not seen in Wisconsin since before the great forest clear-cuts of the 1800s. The current sustainable forest management is a reflection of the worldview of early tribal leaders expressed in the following management goal: *Maintain the diversity of native species and habitats, continue to improve environmental and cultural protection, improve planning efforts, further develop economic opportunities, promote communication, and increase environmental education for the Menominee people, while maximizing the quantity and quality of forest products grown under sustained yield principles* (Menominee Tribal Enterprises, 2012).

northeastern USA, native peoples domesticated perhaps 20 plant species, dogs, and turkeys; in the Mesoamerica subregion nearly 200 plant species, dogs, turkeys, and cochineal were domesticated (Casas *et al.,* 2017; Zarazúa, 2016). In the Andean region of Peru, 182 plant species, dogs, and two species of camelids (llamas and alpacas) were domesticated (Wheeler, 2017), as well as the guinea pig and possibly the duck *Cairina moschata* (Torres-Guevara *et al.,* 2017). In the Amazon, at least 80 species of edible plants have been domesticated (Clement, 2017; Clement *et al.,* 2016). In Mexico, incipient management may include 800 to 1,200 plant species, whereas in Peru nearly 1,800 species are incipiently managed (Casas *et al.,* 2016; Casas *et al.,* 2017; De Jong, 1996; Fraser *et al.,* 2011; Moreno-Calles *et al.,* 2016; Moreno-Calles *et al.,* 2016; Peri *et al.,* 2016; Somarriba *et al.,* 2012; Torres-Guevara *et al.,* 2017).

In addition to agricultural development, local populations manage a high diversity of forests (tropical, dry, temperate, boreal) and ecosystems (coastal, wetland, mountain, plain, desert, aquatic) from which they obtain food, medicine, wood, fuelwood, water, tools, handicrafts, colorants, fodder, ornamental, biological control and instruments. Traditional agricultural systems in the Americas, a result of millennia of cultural and biological evolution, harbor high levels of biodiversity, planned and associated, and represent a high-quality matrix that allows forest species movements among patches (Galluzzi *et al.,* 2010; Larios *et al.,* 2013; Perfecto & Vandermeer, 2008). Traditional farming systems can have a structural complexity and multifunctionality that benefit people and ecosystems and allow farmers to maximize harvest security and reap the benefits of the multiple use of landscapes with low-environmental impacts (Altieri, 2000; Galluzzi *et al.,* 2010). For example, Mayan milpa systems, characterized by open field gaps, reforested plots, and mature closed-canopy forests are recognized for their high agrobiodiversity. In Mayan milpa systems of Greater Petén on the Yucatán Peninsula, around 99 cultigens of native species have been reported as dominant plants on the open multi-crop maize fields, and more than 30 native tree species are managed or protected inside the long-lived perennial reforestation plots and under closed canopies (Ford & Nigh, 2015). Saving such biodiversity should be a priority.

Use of biodiversity. Besides domestication, the biologically-diverse Americas contain a large amount of other biodiversity used by people, including plants, vertebrates, arthropods, fungi, lichens, bacteria, and yeasts. For Mexico, the ethnobotanical data bank at the Universidad Nacional Autónoma de México records close to 7,000 useful plant species out of a total of 24,000 for the country (Casas *et al.,* 2017; Casas *et al.,* 2016). Studies in some regions of Mexico indicate that, on average, nearly 40% of plant species are useful. Such information leads to an estimate of around 10,000 useful plants in Mexico. In Peru, different studies have recorded some 4,400 useful plant species (Torres-Guevara *et al.,* 2017). Mesoamerican peoples are known to use about 7,000 plant species, mainly for medicines; 3,000 animal species (including insects); and 120 fungal species (Caballero & Cortés, 2001; Hernández, 1985; Rojas, 1991).

Agroforestry. In Latin America, an estimated 200 to 357 million ha are under agroforestry (Somarriba *et al.,* 2012). About 12 recognizable types are found, seven in the tropics and five in temperate zones (AFTA, 2017; Jose *et al.,* 2012; Kort *et al.,* 2014; Nair, 1985; Nair *et al.,* 2008; Peri *et al.,* 2016; Somarriba *et al.,* 2012). Agroforestry systems in North America and part of southern South America are of recent origin, while central and northern South American agroforestry systems are bound to highly diverse cultural zones, where societies have preserved their traditional knowledge over thousands years (Casas, Parra *et al.,* 2016; Casas, Parra-Rodinel, Rangel-Landa *et al.,*2017; De Jong, 1996; Fraser *et al.,* 2011; Moreno-Calles, Casas, Rivero-Romero *et al.,* 2016; Moreno-Calles, Casas, Toledo *et al.,* 2016; Somarriba *et al.,* 2012; Torres-Guevara *et al.,* 2017). Ethnoagroforestry management conserves native wild plants, wild and domesticated animals, and the interactions among them (Moreno-Calles *et al.,* 2016; Pell, 1999). Species richness of non-volant mammals and amphibians is similar for agroforestry systems and forests (Chaudhary *et al.,* 2016; Danielsen *et al.,* 2009; García-Morales *et al.,* 2013; Mendenhall *et al.,* 2014; Philpott *et al.,* 2008). However, forest birds, particularly specialist species, and phytophagous bats have declined over time in richness and abundance, respectively, in agroecosystems (Danielsen *et al.,* 2009; García-Morales *et al.,* 2013; Mendenhall *et al.,* 2014; Philpott *et al.,* 2008; Gonçalves *et al.,* 2017).

Agroforestry systems are being lost due to human migration, access to commercial markets, land use change, and the disinterest of government agencies (Montes-Leyva *et al.,* 2017; Van Vliet *et al.,* 2012). The creation of agroforestry systems based on traditional indigenous and local knowledge and novel technological advances promises improvement of ecological interactions, provision of multiple products and ecosystem services (Jose *et al.,* 2012; Moreno-Calles *et al.,* 2016; Moreno-Calles *et al.,* 2016; Peri *et al.,* 2016), and if stimulated, would contribute to biodiversity conservation.

3.3.4 Status and trends of biodiversity in urban anthropogenic systems

Status. Urban areas are home to about 80% of the population in the Americas. Urban land in the North American (excluding Greenland) and Mesoamerican subregions accounts for 5% of the total land (Güneralp

& Seto, 2013). The Caribbean subregion has the highest urban land fraction (16%) and South America the lowest (2%). Currently, the Americas host eight (20%) of the world´s 40 Megacities (population over 10 million): two in the North American subregion, one in the Mesoamerican subregion and five in the South American subregion. There are many other large cities in the Americas that do not qualify as megacities **(Figure 3.12A)**. Urban ecosystems in the Americas are expected to continue to expand and coalesce (Seto *et al.,* 2012). This signifies that urban areas will be the main contact point with nature for an increasingly large proportion of the Americas population. Policies that conserve and enhance urban biodiversity will thus enhance human well-being.

Urban areas in many parts of the Americas are surrounded by high-diversity ecosystems. Major changes in species richness, species composition, and ecosystem functioning have accompanied urbanization (McPhearson *et al.,* 2013; Pauchard & Barbosa, 2013) although cities may be hotspots of plant biodiversity because of human cultivation (Müller *et al.,* 2013). A survey of spontaneous and cultivated flora across seven USA cities found a positive association between species richness and urbanization (Pearse *et al.,* 2018), a pattern that has been observed in other regions (Hope *et al.,* 2003; Walker *et al.,* 2009). However, urbanization can lead to loss of spontaneous species richness and phylogenetic diversity and selects for plants with functional traits that allow them to disperse and reproduce well in the urban environment (Knapp *et al.,* 2012). That is, the urban flora is a non-random sample of plant biodiversity.

Cultivated plant species in North America, and perhaps across the Americas, include a high number of introduced species (Pearse *et al.,* 2018). Such introduced species can escape cultivation (Knapp *et al.,* 2012; Pearse *et al.,* 2018) and interact with native species, changing the floral composition in urban areas and beyond (Shochat *et al.,* 2010). Indeed, the proportion of exotic plants is expanding, and the number of native species is declining in urban areas in the Americas (Reichard & White, 2001; McKinney, 2002; Kowarik, 2008; MacGregor-Fors & Ortega-Álvarez, 2013), while urban floras are tending to homogenize (La Sorte & McKinney, 2007). Consequently, urbanization affects community assembly and leads to more simplified (Aronson *et al.,* 2014; McKinney, 2002; Stranko *et al.,* 2010) and more homogenized ecosystems (Groffman *et al.,* 2014; Hall *et al.,* 2016; La Sorte & McKinney, 2007; McKinney, 2006; Steele *et al.,* 2014).

Some plant and animal species tend to do well in the physical structure of the urban landscape and are able to take advantage of the availability of resources such as human garbage. However, animal species richness tends to decline along urbanization gradients (Aronson *et al.,* 2014; Chace & Walsh, 2006; González-Urrutia, 2009; Groffman *et al.,* 2003; Hamer & McDonnell, 2008; McKinney, 2002, 2008; Moore & Palmer, 2005; Ortega-Álvarez & MacGregor-Fors, 2011; Paul & Meyer, 2001; Stranko *et al.,* 2010; Urban *et al.,* 2006). That said, nonlinear relationships have also been reported for animal species along these gradients (Blair & Launer, 1997; Faggi & Perepelizin, 2006; Germaine & Wakeling, 2001; McIntyre *et al.,* 2001; McKinney, 2008).

Urban environments are associated with a decline in native mammals, with the rare exception of species able to thrive near humans. Carnivorous and large mammals have been progressively excluded from urban areas, while middle-size omnivorous mammals that eat anthropogenic foods tend to persist (McCleery, 2010; Pereira-Garbero *et al.,* 2013). Many small mammals in the Americas are poorly represented in cities except rats and mice (Cavia *et al.,* 2009; Childs & Seegar, 1986; Himsworth *et al.,* 2013). The response of reptile biodiversity to urbanization is poorly understood, although positive trends were reported for turtles and snakes (Barrett & Guyer, 2008). In Arizona, lizard diversity and abundance follows a humped pattern on a residential density gradient (Germaine & Wakeling, 2001).

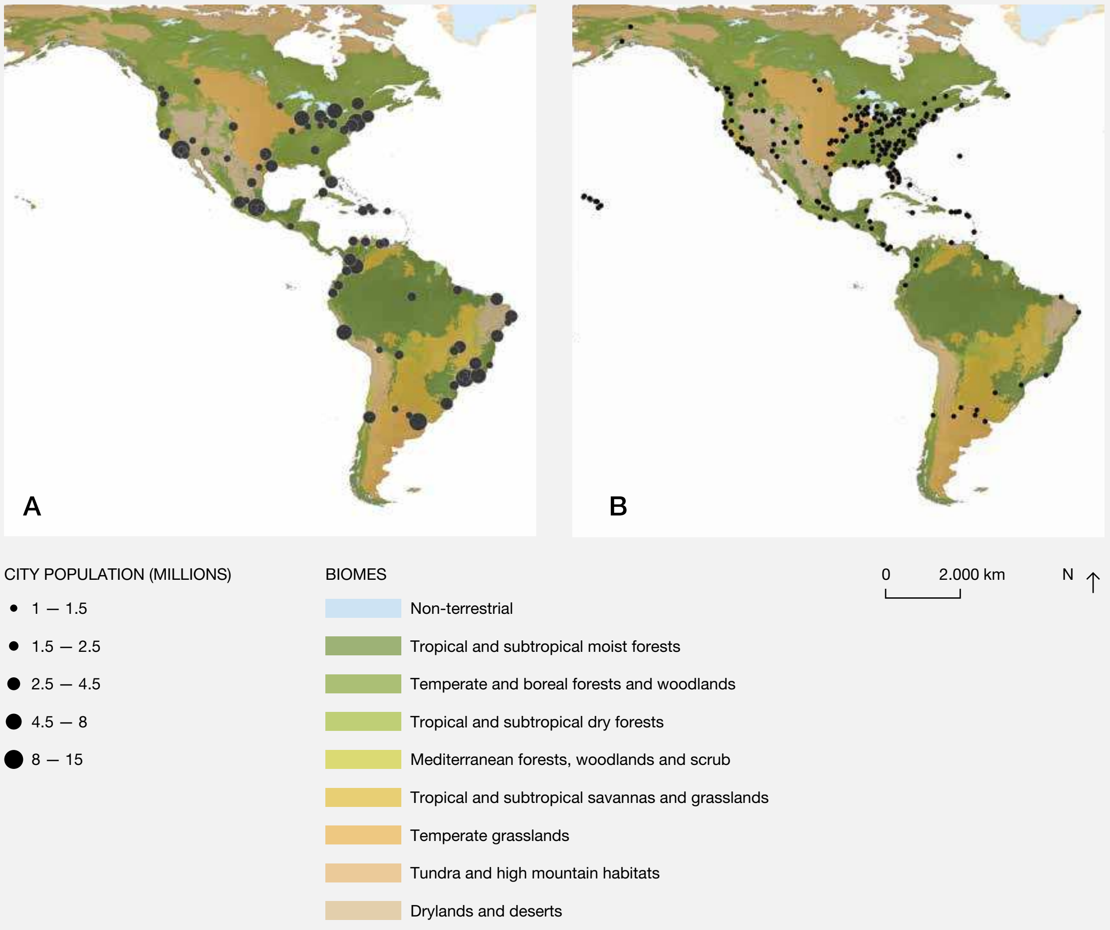
Birds are among the most studied urban animals. Avian diversity and urbanization are negatively correlated, while the total abundance of birds may increase with urbanization (Chace & Walsh, 2006; González-Urrutia, 2009; Ortega-Álvarez & MacGregor-Fors, 2011). As in other taxa, these trends are associated with shifts in functional traits along urbanization gradients (Chace & Walsh, 2006; Leveau, 2013; McKinney, 2002) and species ability to use waste as food (Marateo *et al.,* 2013). Urban bird diversity is enhanced by increases in the number, size, connectivity and habitat heterogeneity of urban parks and vegetation remnants (Beninde *et al.,* 2015; Díaz & Armesto, 2003; Garitano-Zavala & Gismondi, 2003; González-Urrutia, 2009; Juri & Chani, 2009; Manhães & Loures-Ribeiro, 2005; Maragliano *et al.,* 2009; Ortega-Álvarez & MacGregor-Fors, 2011; Perepelizin & Faggi, 2009; Sacco *et al.,* 2013; Villegas & Garitano-Zavala, 2010). Significant raptor diversity has been reported, even in larger cities. For example, more than 20 raptor species were recorded in Buenos Aires, Argentina (Cavicchia & García, 2012). Some 24 species (83% of Chilean raptor species) were observed in the Chilean Metropolitan Region of which 18 occur in the vicinity of Santiago; seven are considered urban or suburban (Jaksic *et al.,* 2001). In Baja California, Mexico, raptor richness was unaffected by the anthropogenic transformation of the habitat (Rodríguez-Estrella *et al.,* 1998). At the same time, non-native avian species have progressively established in urban areas. In Mesoamerica some urban areas now have non-native avian abundances similar to those observed in developed countries at temperate latitudes (González Oreja *et al.,* 2007). In the midwestern USA, raptors such as the peregrine falcon, whose populations plummeted with

Box 3 4 Botanical gardens in the Americas.

Botanical gardens are stores of plant biodiversity that provide *ex-situ* conservation and biodiversity education to urban populations. However, there is a large imbalance in the distribution of botanical gardens across the subregions. Of the 2,728 botanical gardens registered globally with the botanic gardens Conservation International, 765 occur in North America, 127 in Mesoamerica, 46 in the Caribbean, but only 164 in South America **(Figure 3.12. B)**. South America's relatively low number is noteworthy given that it houses higher plant species richness and more megacities than North America. Some very rich biomes, like the South American Mediterranean forests, shrublands and scrub biome, have a very poor representation of certified botanical gardens **(Figure 3.12. B)**.

Figure 3 12 Ⓐ Largest cities in the Americas based on population size shown by biome. Ⓑ Location of accredited botanical gardens from the BGCI garden search database in relation to biomes.

Source: A: http://simplemaps.com/data/world-cities, last updated in 2015. B: Search database www.bgci.org. Accessed August 5, 2017.



pesticide use in the mid-20th century, have been successfully reintroduced in cities where tall buildings provide suitable nesting sites (Tordoff & Redig, 2001).

Arthropods show a range of responses to urbanization (McIntyre, 2000; Müller *et al.,* 2013, Raupp *et al.,* 2010). In the Phoenix area, for example, birds were found to be a dominant force controlling arthropod ecology (Faeth *et al.,* 2005). While some urban gradients involve small changes in richness or abundance of arthropods, community composition may change considerably (McIntyre *et al.,* 2001). In Palo Alto, California, butterfly diversity has progressively declined with increasing urbanization (Blair & Launer, 1997). However, several studies show a positive relationship between urbanization and some bee guilds (e.g. cavity-nesters within urban areas, Potts *et al.,* 2010).

Recent trends. An increase in high-rise buildings has greatly increased population density in many cities of the Americas. Urban ecosystems within these cities have increased in size as the human population has grown (Grimm *et al.,* 2008). This portends large-scale transformations for the provision of water, food, and services (Vörösmarty *et al.,* 2000). Associated transportation systems have created a network of interconnected urban habitats that has grown significantly in extent, density and flow (Kohon, 2011; Rodrigue *et al.,* 2017).

Over the past two decades, the uneven accessibility of urban greenspace has become recognized as an environmental justice issue as awareness of its importance to public health has become recognized (Dai, 2011). Some cities in Latin America have begun to set goals to plan for a minimum of 9 m^2 of green area per inhabitant[1]. Data on green areas for cities in the Americas is scarce and this is an area that needs better attention. The percentage of urban areas dedicated to green areas is highly variable across the Americas **(Figure 3.13)**. Considerable variation, moreover, can occur within individual cities. For example one of the wealthiest suburbs of Santiago, Chile has 56 m^2 per inhabitant, while one of the poorest has only 2.4 m^2 (Reyes & Figueroa, 2010). Generally, the incorporation of green areas of any kind has promoted urban biodiversity (Cameron *et al.,* 2012), although the development of green areas has not been commensurate with the population increase in urban areas. Thus, conserving biodiversity in urban areas should be a priority. The establishment of green areas using native species can simultaneously contribute to biodiversity conservation and human well-being and should be a priority.

The Americas are projected to experience significant increases in urban land extent **(Figure 3.14. a)** (Güneralp & Seto, 2013). Moreover, North America is expected to have more than 50% of its total urban lands within 25 km of protected areas and 90% of its urban lands within 50 km of protected areas by 2030; in contrast, South America is projected to have about 65% within 50 km while Mesoamerica and the Caribbean are projected to have somewhat more **(Figure 3.14. b)** (Güneralp & Seto, 2013). Documented changes in hydrology with urbanization, including alteration of wetlands (Steele *et al.,* 2014), pollution, simplification of freshwater environments and loss of riparian vegetation, will tend to reduce biodiversity among algae, plants, invertebrates and vertebrate communities (Groffman *et al.,* 2003; Moore & Palmer, 2005; Paul & Meyer, 2001; Stranko *et al.,* 2010; Urban *et al.,* 2006). Amphibians are particularly vulnerable to urban development (Hamer & McDonnell, 2008), habitat loss, homogenization and isolation (Birx-Raybuck *et al.,* 2010; Cushman, 2006; da Silva *et al.,* 2011, 2012; Delis *et al.,* 1996; Fahrig, 2003; Fahrig *et al.,* 1995; Sutherland *et al.,* 2010) and changes in hydrodynamics (Barrett *et al.,* 2010; Eskew *et al.,* 2012; Price *et al.,* 2011).

Long-term data on biodiversity in cities of the Americas still tends to be limited and fragmented. In the USA, two

---

[1] http://ipco.gob.mx/images/documentos/estudios/piam_colima_final_2010.pdf

Figure 3 13 **Percentage of urban areas dedicated to green areas in different cities of the Americas. Based on data from World Cities Culture Forum (http://www.worldcitiescultureforum.com/data/of-public-green-space-parks-and-gardens).**

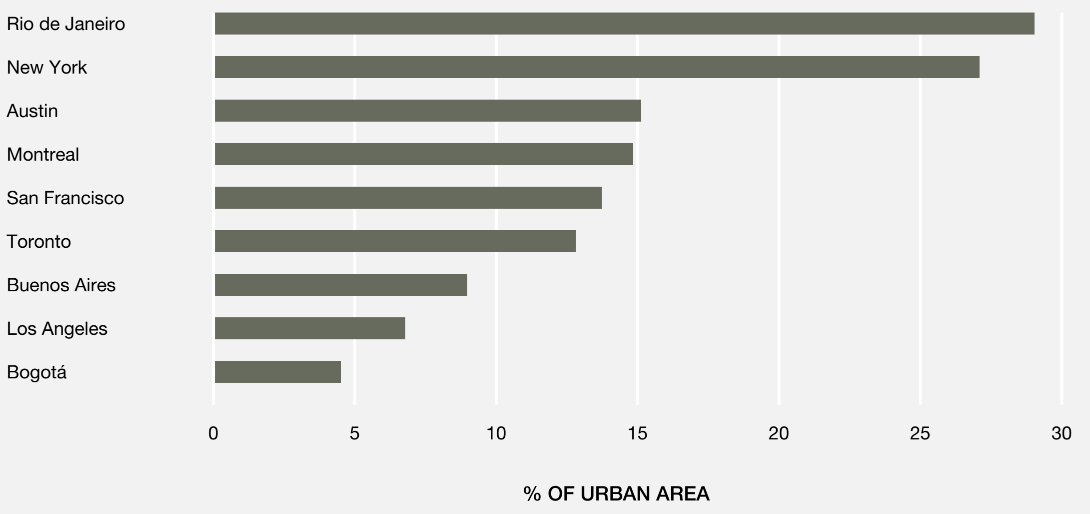
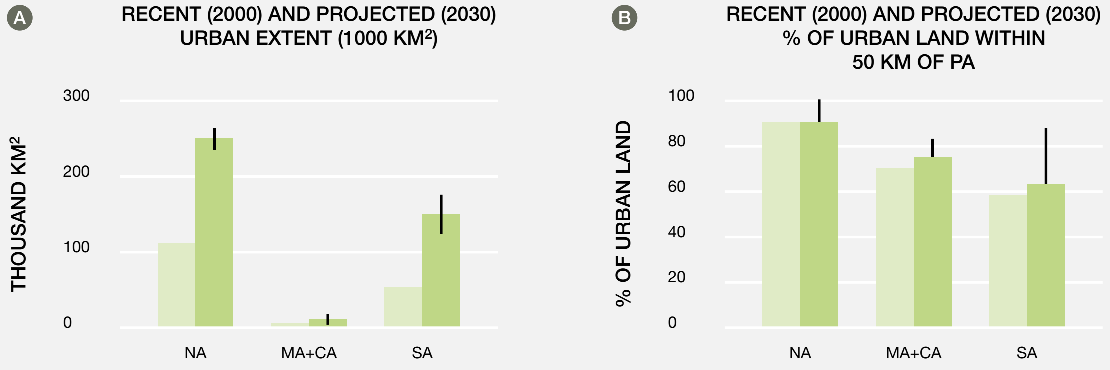


Figure 3 14 Ⓐ **Total urban extent in 2000 (light green) and projected in 2030 (dark green); Ⓑ percentage of total urban land in 2000 (light green) and projected (dark green) in 2030 within 50 km of protected areas (PA) for North America (NA), Mesoamerica, the Caribbean (MA+CA) and South America (SA).**

Source: Modified from figures in main text and supplementary material in Güneralp & Seto (2013).



urban Long-Term Ecological Research sites (Baltimore and Phoenix) have been established to gather social and ecological data (Redman *et al.,* 2004). Such Long-Term Ecological Research sites are valuable for the purpose of comparative international research on urban socio-ecological systems and their links to decision-making. The City Biodiversity Index ("Singapore Index"), which integrates biodiversity data, has been widely used in South East Asia to assess the role of cities in conserving biodiversity. This, or some similar index, could be adapted for use in the Americas.

3.3.5 Status and trends of biodiversity in agricultural, silvicultural and aquacultural anthropogenic systems

European colonization simplified agricultural systems and landscapes across the Americas, reducing crop diversity, marginalizing several native crops and eroding knowledge associated with traditional farming practices (Galluzzi *et al.,* 2010; Galluzzi & López Noriega, 2014; Khoury *et al.,* 2014; Kremen & Miles, 2012; Winograd *et al.,* 1999). As a consequence, large amounts of land in the Americas are today devoted to intensive cropping and forestry (c.f., Beddow *et al.,* 2010). Conversion of land from natural systems to crop production and agriculture has important impacts on habitat for biodiversity and differs by biome (Ramankutty *et al.,* 2010) and type of farming system. For Latin America, expansion of pastures is the main cause of habitat loss and is responsible for more than two-thirds of deforestation in the Amazon region, with agrofuel and fodder (soybean) monocultures also adding pressure to forests (Altieri, 2009; Pacheco *et al.,* 2011; Thornton, 2010). Agricultural intensification changes and diminishes ecological functions (Goijman *et al.,* 2015) and can lead regionally to shifts in species composition (section 3.4 for details of impacts in different terrestrial biomes). Traditional knowledge and systems for the maintenance of crop genotypes have been lost as agriculture has been commercialized. For example, there is evidence of a loss of large numbers of native potato in Cusco (Gutiérrez & Schafleitner, 2007), due to the introduction of commercial strains. This is a vast area of knowledge that was not possible to cover in the present assessment and warrants an assessment on its own merits.

Non-native species are often the base of production systems and can impact ecosystem services needed to support production in the long term. Fishes in aquaculture represent a good example, as nearly all countries culture tilapias, carp and trout, none of which are native to the Americas. Although Brazil contains 20% of the world's fish species, aquaculture is based solely on non-native species – some are native to the country but produced beyond their native ranges (I3N, 2016). The same trend is present in silviculture. Pines (*Pinus* spp.) are widely invasive in the southern hemisphere, with at least 16 species that have spread from planting sites into natural or seminatural vegetation (Richardson *et al.,* 1994), while acacias (*Acacia* spp.) and gums (*Eucalyptus* spp.) are either not planted as much or are less aggressive. These taxa, either in plantations or invasions, have been documented as intensive water users; areas invaded with these trees tend to have low economic value and low productivity (Versveld *et al.,* 1998).

Recent trends. The Americas have led world production of high-demand agricultural products like soybeans, sugarcane, and cattle meat over the past five decades. During this period, the net agricultural production of the region has grown together with its population (Ramankutty *et al.,* 2002). This has led to increases in the conversion of land to agriculture **(Figure 3.15)**. The apportionment of land to agriculture (aggregated within each subregion) shows the greatest increases in Mesoamerica followed by South America, but recent declines in the Caribbean and North America. The total extent of arable and pasture lands in Latin America has increased at an annual rate (1990–2008) of 0.87% for South America (16.4 million ha) and 0.15% for Mesoamerica (828,000 ha). Pasture land grew by 11.3 million ha (0.14% per year) in South America, while in Mesoamerica it declined 2.7 million ha (–0.17% per year) (Pacheco *et al.,* 2011). Conversion of land for agricultural purposes has often come at the expense of forest, woodland, and other vegetation types (section 3.4).

From 1992 to 2010, richness and phylogenetic diversity of crop production and exports from all subregions have been relatively constant. However, South America and Mesoamerica have higher phylogenetic diversity in crop production than does North America, and Mesoamerica has higher crop species richness than both North and South America (Nelson *et al.,* 2016). In contrast, North America has a higher consumption of species richness than other subregions, even while all subregions have similar phylogenetic diversity in crop consumption (Nelson *et al.,* 2016).

Pollinator-friendly agricultural systems can help maximize crop yields by preserving the pollination services offered by wild bees (Garibaldi *et al.,* 2014; Shaver *et al.,* 2015). Pollinator loss has been particularly rapid in tropical regions (Ricketts *et al.,* 2008) as well as in extensive temperate regions that have experienced drastic land use transformations, like the Pampas of South America (Medan *et al.,* 2011) and the USA Midwest and Great Plains (Koh *et al.,* 2016). The high use of pesticides across the Americas (Liu *et al.,* 2015) is an important additive and interactive cause of bee declines (Goulson *et al.,* 2015).

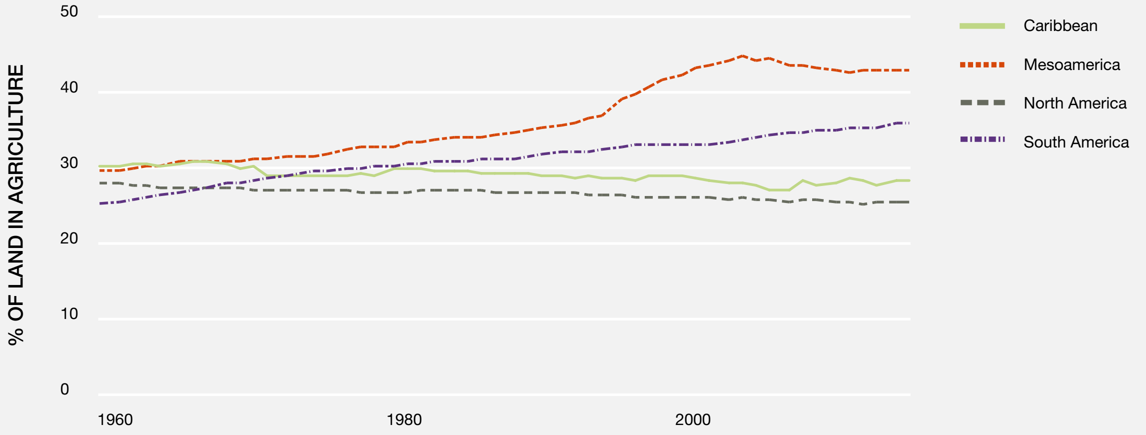
Aquaculture has increased in the Americas. In the USA, aquaculture growth for marine fish and shellfish has been below the world average, rising annually by 4% in volume and 1% in value (Naylor, 2006). The main marine species are Atlantic salmon, shrimp, oysters, and hard clams, which together account for about one-quarter of total USA aquaculture production. In South America, Chile is now the second largest producer of salmon globally after Norway (Buschmann *et al.,* 2006). Excessive use of antibiotics in Chilean salmon farms have resulted in antibiotic resistance (Burridge *et al.,* 2010), and this trend may be widespread.

3.3.6 Emerging diseases and biodiversity

Emerging infectious diseases have become a major concern (Hatcher *et al.,* 2012). Bacteria, viruses, protozoan, fungi, helminths and drug-resistant microbes are commonly reported in emerging infectious diseases outbreaks worldwide affecting a wide taxonomic spectrum (Jones *et al.,* 2008; Pedersen *et al.,* 2007). Multiple mechanisms and causes for emerging infectious diseases have been recognized, including biodiversity loss, land use change,

Figure 3 15 **Changes in the percentage of land in agriculture for each subregion from 1961 to 2014. Greenland is not included in the calculation for North America.**

Percentages of individual countries were multiplied by their area, summed, and divided by the total area of the subregion. Source: World Bank (2017). World Development Indicators (https://data.worldbank.org/indicator/AG.LND.AGRI.K2) Last updated Date: March 23, 2017.



urbanization, climate change, human demographics, international travel and commerce, species invasions, pollution, microbial adaptation, war and famine, poverty, and breakdown of public and animal health measures (Hatcher *et al.,* 2012; Jones *et al.,* 2008; Loh *et al.,* 2015). Individually or synergistically, these causes affect patterns of species distributions and favor invasions of reservoirs, hosts, vectors, and pathogens affecting native species (Keesing *et al.,* 2010; Suzán *et al.,* 2009).

Emerging infectious diseases are reported in marine and terrestrial ecosystems and are responsible for several species and populations extinctions worldwide. Coral reef fragmentation, pollution, and warming have favored toxins and pathogens like *Serratia marcescens* (white pox disease) and Vibrio AK-1 (coral bleaching), producing widespread coral reef mortality (Sutherland *et al.,* 2010; Vega Thurber *et al.,* 2014). Likewise, marine mammals have been threatened by morbilliviruses, poxviruses, and papillomaviruses globally (Harvell, 1999). In terrestrial systems, plant communities have been decimated by emerging infectious diseases such as Dutch elm disease (*Ophiostoma* spp.), chestnut blight (*Cryphonectria parasitica*), and jarrah dieback (*Phytophthora cinnamomi*) that affects hundreds of host plants (Anderson *et al.,* 2004). Several examples of emerging infectious diseases have been reported to affect vertebrates, including *Batrachochytrium dendrobatidis*, a fungal infection producing population and species extinction in amphibians worldwide, and malaria infection in Hawaiian birds (Smith *et al.,* 2009). In the Americas, several endangered and threatened species have declined as a result of emerging infectious diseases such as West Nile virus in native birds (Robinson *et al.,* 2010; Smith *et al.,* 2009), plague in prairie dog colonies (Stapp *et al.,* 2004) and White-nose syndrome in North American bats (Frick *et al.,* 2017). Several infections affect top predators, including canine parvovirus in wild carnivores (Pedersen *et al.,* 2007) and canine distemper, which is associated with extinction in the wild of the black-footed ferret (McCarthy *et al.,* 2007; Thorne & Williams, 1988). Increasing spread of infectious diseases can be expected with globalization, calling for greater vigilance.

3.4 STATUS AND RECENT TRENDS OF BIODIVERSITY BY UNITS OF ANALYSIS

3.4.1 Terrestrial biomes

In this section, snapshots of the status and recent trends in biodiversity for the major terrestrial biomes are examined in each subregion where they occur (see Chapter 1 for official units of analysis map of the assessment). Although coverage is extensive, space limitations prevented assessment of all biomes in each subregion and exhaustive treatments for the biomes that are assessed. Status and recent trends in biodiversity and the relative importance of NCP are synthesized in **Figures 3.24** and **3.25**, respectively. Summary data on species richness for the biomes assessed in each subregion can be found in **Table 3.4**.

3.4.1.1 Tropical and subtropical moist forests

Mesoamerican subregion

Status. Species diversity in the Mesoamerican broad-leaved tropical/subtropical moist broadleaf biomes is high, with low to moderate species endemism (Myers *et al.,* 2000; Ray *et al.,* 2006). In Mexico, moist wet forests and montane cloud forests have the highest diversity of plant species per unit area among vegetation types (Rzedowski, 1991). Tropical lowland broadleaf moist forests house around 17% of the flora of Mexico, while montane mesophyll forests contain around 9% of the flora (see also, **Table 3.4** for numbers) (Challenger & Soberón, 2008). Mesoamerican coniferous forests in general support low to moderate species diversity. Notably, however, Mexican coniferous forests contain very high numbers of pine and oak species **(Table 3.4)**. Species diversity and endemism for amphibians are high in the moist forests of the Mesoamerican highlands (Köhler, 2011; Lamoreux *et al.,* 2015). In Mesoamerican lowland rainforests, the diversity of mammals decreases from eastern Panama to southern Mexico (Voss & Emmons, 1996). The mesic forests of southeastern Mexico have been classified as critically endangered (Hoekstra *et al.,* 2005).

Recent trends. Over the past 50 years, loss of lowland moist forest in Mexico was acute, the yearly deforestation rate reaching 2.6% for 1976-1993 and 1.3% for 1993–2002 (Challenger & Dirzo, 2009); by 2002 primary forest was down to only 17.5% of the original area. Before the late 1980s, forest loss was generally caused by small-scale slash-and-burn agriculture. In the past 25 years, however, large-scale cropping and pastures became the main causes of tropical habitat loss (Gibbs *et al.,* 2010; Laurance, 2010). Montane mesophyll forest (including cloud forest) was reduced from less than 50% to 28% of its original extent over the period 1976 -2003; coniferous forests fared better, with around 50% still remaining (Challenger & Dirzo, 2009).

Removal and fragmentation of moist forest have led to a significant decrease of regional species diversity (Ray *et al.,* 2006). Many amphibian species have experienced severe local and regional declines across the moist forests of the Mesoamerican highlands due to habitat destruction, emerging infectious diseases and other factors (Lamoreux *et al.,* 2015; Stuart *et al.,* 2008). The increased use of pesticides and fertilizers, loss of live fences, and decline of natural habitat fragments within agroecosystems – have also exacerbated biodiversity losses due to habitat reduction (The Nature Conservancy, 2005).

In general, tropical forests seem to be resistant to the impacts of invasive plant species (Denslow & DeWalt, 2008), and compared with habitat loss and fragmentation, exotic invasive species are considered a relatively minor threat to moist forest biodiversity as seen in Mexico (Challenger & Dirzo, 2009; Dirzo & Raven, 2003). Of the 42 exotic species reported by Rejmánek (1996), most are confined to pastures, clearings, or other highly disturbed sites (Foster & Hubbell, 1990; Hammel, 1990). However, there is some evidence that invasive species are increasing (Aguirre-Muñoz & Mendoza, 2009; Espinosa & Vibrans, 2009). The Asian house gecko, *Hemidactylus frenatus*, has been widely introduced in Mesoamerica and is replacing the native leaf-toe gecko, *Phyllodactylus tuberculosus*, especially along the forest edge and in disturbed forests (G. Köhler unpubl. data). It is known to carry the pentastomid parasite, *Raillietiella frenata*, native to Asia, and has been shown to transfer this parasite to *Rhinella marina*, a toad native to Mesoamerica (Kelehear *et al.,* 2015). Several species of Caribbean frogs of the genus *Eleutherodactylus* have been documented as invasive species in Mesoamerican Tropical/Subtropical Moist Broadleaf Forests (Crawford *et al.,* 2011; Köhler, 2008).

Caribbean subregion

Status. The tropical moist forest biome is thought originally to have covered around 81,000 km^2 in the Caribbean (Dinerstein *et al.,* 1995). As of European colonial times and especially before the 1900s (Gould *et al.,* 2012; Lugo *et al.,* 2012), much forest was cleared for agriculture (Fitzpatrick & Keegan, 2007). Dinerstein *et al.* (1995) estimate that 50% of the original wet forest in the Greater Antilles (90% in Jamaica and Hispaniola) and 25% in the Lesser Antilles was removed or degraded. Land too steep or distant from coastal markets was often left untouched and today forms the core of the remaining biodiversity in Caribbean islands. Vegetation at higher altitudes on the islands of the Lesser Antilles was often retained for "attraction of the rains" (Fitzpatrick & Keegan, 2007; Lugo *et al.,* 2012).

In general, endemism is high for plants and vertebrates in the Caribbean subregion, as is plant species richness. The biodiversity data for the subregion **(Table 3.4)** to some extent correlates with Caribbean tropical moist forest extent, given that this biome contains a high proportion of Caribbean terrestrial biodiversity. In the Lesser Antilles, the upland moist forests are more species diverse and host the majority of the endemic plant species due to biogeographic factors and human deforestation of the lowlands (Adams, 1997). On the other hand, montane moist forests in the Dominican Republic appear to have lower rates of species richness and endemism than do dry forests (Cano-Ortiz *et al.,* 2015). Cuban invertebrates seem to show high endemism levels similar to those found in vertebrates (e.g. Alayo, 1974; Alayón García, 1999; Starr, personal communication). Among those assessed, some 316 species of plants and vertebrates in the Caribbean are considered threatened (Anadón-Irizarry *et al.,* 2012; IUCN, 2017).

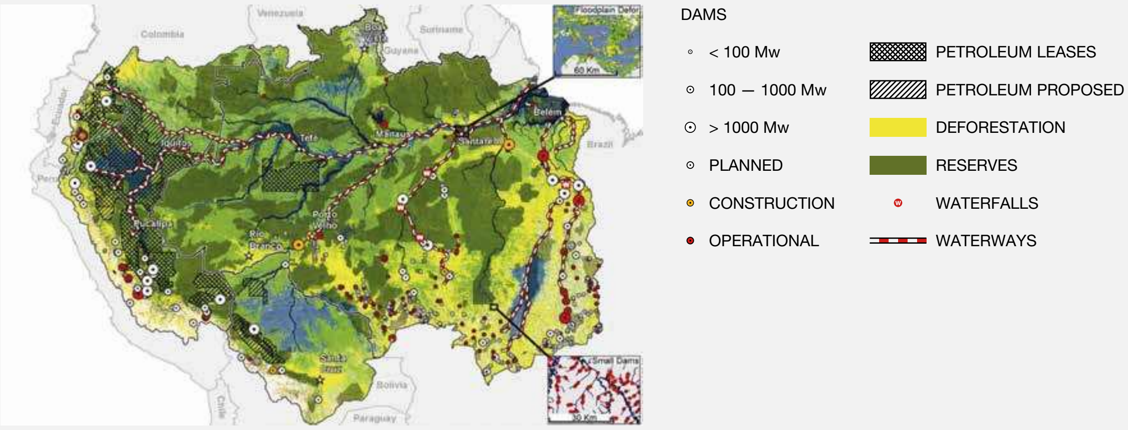
In the Caribbean, terrestrial habitats, including productive areas, are affected by a multitude of invasive alien species — among them, agricultural pests that were introduced with crops. Adverse impacts of invasive species are most severe in the Greater Antilles and Northern Lesser Antilles on islands that have been isolated for the longest time periods and have the greatest degree of human degradation and disturbance (Kairo *et al.,* 2003). However, there is speculation that some invasive exotic plants may act as nurse plants for native species and will decline in importance once native species recover from human disturbance (e.g. *Leucaena* in Puerto Rico) (Lugo *et al.,* 2012). There is also growing acceptance that exotic species have become important components of many island ecosystems (Lugo *et al.,* 2012).

Recent trends. Forests at mid- to high altitudes began to regenerate when agriculture declined after World War I (Gould *et al.,* 2012; Lugo *et al.,* 2012). In Puerto Rico, forest cover increased from approximately 5% to over 30% between 1940 and 1990 (Aide *et al.,* 2000). Tropical moist forest tends to regrow in mountainous areas where agriculture is more likely to be small scale (Asner *et al.,* 2009); 2,550 km^2 of mountain tropical moist forest regenerated between 1984 and 2002 in the Dominican Republic (Grau *et al.,* 2008) and 1,036 km^2 of dry/moist/wet mountain forest regenerated between 1991 and 2000 in Puerto Rico (Parés-Ramos *et al.,* 2008). The area of Caribbean forests in general increased by an average of 0.81% between 1990 and 2010 (FAO, 2011), but in Puerto Rico, the increase has now ceased (Grau *et al.,* 2008). Loss of native predators and herbivores due to introduced predators in post- and pre-Columbian times continues (Kairo *et al.,* 2003).

South American subregion

Status. In South America, this biome is centered on the Amazonian wet forest, Atlantic coastal forest, and Andean tropical montane forest. It is also found on the western side of the northern Andes (at low altitudes) and in lowland Venezuela, Guyana, Suriname, and French Guiana. Amazonian wet forest covers 6.7 million km^2 — half of the planet's remaining tropical forests. Around 17% of Amazonian wet forest has been destroyed (Charity *et al.,* 2016) (see also **Figure 3.16**). Andean tropical montane

Figure ③ ⑯ **The Amazon basin showing basin-wide deforestation (including all areas classified as under human use in both forests and savannah ecosystems), main waterways and river channel network, protected areas, hydroelectric dams, areas available to be leased for oil exploration, and proposed areas for future lease for oil exploration. Source: Castillo *et al.* (2013).**



forest comprises cloud forests (northern Andean forests, Yungas forests and Bolivia-Tucuman forests) and seasonal (wet) forest mostly found above 1,500 m.a.s.l. Atlantic coastal forest once covered around 1.5 million km^2 but today is down to ~12% of its original pre-colonial extent (Ribeiro *et al.,* 2011). Continuous expanses of forest (measured as the proportion of forest more than 1 km from the forest edge) have decreased from 90% (historical) to 75% (today) in Amazonian wet forest and from 90% to less than 9% in Atlantic coastal forest (Haddad *et al.,* 2015). The Amazon, long thought to be a pristine forest, is now recognized as having been subject to long-standing indigenous management and transformation (Roberts *et al.,* 2017). At least 138 crops in 44 plant families, mostly trees or woody species, were cultivated, managed or promoted in Amazonia upon European contact, although some were subsequently lost (Clement, 1999). Human use of biodiversity has been associated with the origin of many new varieties of manioc in both Amazonian wet forest and Atlantic coastal forest (Emperaire & Peroni, 2007).

Exceedingly rich **(Table 3.4)**, Amazonian wet forest is estimated to house one-tenth of all known species of plants and animals (Charity *et al.,* 2016), although these estimates require careful verification. Although opinions differ widely regarding total tree species richness **(Table 3.4)**, it seems that relatively few species account for the bulk of the Amazonian wet forest trees (ter Steege *et al.,* 2013, 2016). Also very rich, Atlantic coastal forest has high endemism (Kier *et al.,* 2009; Mittermeier *et al.,* 2005; Tabarelli *et al.,* 2010). For example, 16–60% of birds, mammals, reptiles, and amphibians in Atlantic coastal forest are endemic (Mittermeier *et al.,* 2005; Tabarelli *et al.,* 2010). Andean tropical montane forest likewise has many range-restricted species (Fjeldså & Rahbek, 2006), high bird species diversity **(Table 3.4)**, high species turnover along altitudinal gradients and high endemism. Epiphytes, which have high water storage, are especially abundant in Andean tropical montane forest (Brown, 1990; Kessler, 2001; Kramer *et al.,* 2005; Krömer *et al.,* 2006; Küper *et al.,* 2004; Roque & León, 2006), as they are in Atlantic coastal forest (2,256 species of hemi-epiphytes, the equivalent of 15% of all vascular plants in these forests) (Freitas *et al.,* 2016).

South American tropical and subtropical moist forests provide important biodiversity-linked NCP. Amazonian wet forest stores 10% of global carbon and places seven trillion tons of water per year into the atmosphere, contributing to the stabilization of local and global climate and nurturing agriculture (Charity *et al.,* 2016). Although globally less relevant than Amazonian wet forest, mature Andean tropical montane forest has higher above-ground biomass than was originally thought (Spracklen & Righelato, 2014). Slope stability, critical in Andean countries, is higher in secondary Andean tropical montane forest than in forest land converted to pastures (Guns & Vanacker, 2013). Pollination provided by wild bees and birds, and animal dispersal are additional biodiversity-linked ecosystem services provided by this biome (see **Box 3.5**). Currently, many orchids in Ecuador are grown commercially (Mites, 2008), and orchid greenhouses are now a major tourist attraction.

Recent trends. Deforestation rates in the Amazon decreased during the past decade but increased again as of 2015 (RAISG, 2015). Habitat loss in Atlantic coastal forest remains high in most regions, attaining annual rates of 0.5% for the whole biome (Teixeira *et al.,* 2009, see also Chapter 4). Between 2000 and 2012, the net loss of Atlantic coastal forest was proportionally lower than for other tropical woody biomes **(Figure 3.19)**, but this is considered to be due mainly to the establishment of exotic tree plantations (Salazar *et al.,* 2015). Andean tropical montane forest was lost in all Andean countries between 2005 and 2010: between 1985 and 2000 Colombia lost close to one million ha of montane forest (Tejedor Garavito *et al.,* 2012).

Deforestation has impacted tree species in Andean tropical montane forest, judging by the 235 species classified as globally threatened according to the International Union for Conservation of Nature (IUCN) Red List of Categories and Criteria (Tejedor Garavito *et al.,* 2014). Upon taking recent deforestation into account, some Andean species representing different taxonomic groups in the IUCN lists were judged as requiring updating in terms of extinction risk (Tracewski *et al.,* 2016), suggesting heightened impacts. Reductions in habitat and biodiversity in Andean tropical montane forest are in part due to down-burning fires set in páramo and puna (e.g. Román-Cuesta *et al.,* 2011). Ongoing deforestation is affecting range sizes (Peralvo *et al.,* 2005; Ocampo-Peñuela & Pimm, 2015), genetic connectivity among populations (Klauke *et al.,* 2016) and stream quality (Iñiguez-Armijos *et al.,* 2014). Moreover, hydrologic connections between the atmosphere and surface waters and their downstream effects have been altered in Andean tropical montane forest - soil moisture can be significantly lower in pasture compared with forest (Ataroff & Rada, 2000).

Forest fragmentation has been associated with long-term losses in species richness and changes in species composition (Haddad *et al.,* 2015; Metzger, 2009; Laurance *et al.,* 2017). In Atlantic coastal forest, old-growth forest patches operate both as irreplaceable habitats for forest-obligate species and as stable source areas (Tabarelli *et al.,* 2010). Fragment size distribution, structural connectivity, matrix quality, remaining forest cover, presence of old-growth forest patches and/or proportion of edge-affected habitats have been identified as key correlates of species richness and abundance in bats, reptiles, birds, canopy/emergent trees, small mammals, mammalian carnivores, butterflies, chironomid insects, and frogs (Tabarelli *et al.,* 2010). Multi-taxa data collected at regional and local scales in the northern Amazon demonstrate reduced species richness with increasing anthropogenic disturbance and considerably more biotic homogenization in arable croplands and cattle pastures than in disturbed, regenerating and primary forest (Solar *et al.,* 2015). Likewise, multi-taxa studies reveal a threshold forest cover that triggers local extinctions (Joly *et al.,* 2014). A survey of a wide range of taxa within a large forest mosaic recorded only about 50% of old-growth forest species richness within patches of tree plantations (*Araucaria, Pinus* and *Eucalyptus*) (Fonseca *et al.,* 2009). Overall, habitat degradation has driven a fraction of Atlantic coastal forest´s unique biodiversity to near extinction (Joly *et al.,* 2014; Tabarelli *et al.,* 2010). Nevertheless, landscape dynamics suggest young secondary forests are beginning to expand in the Amazon, reducing forest isolation and maintaining a significant amount of the original biodiversity (Lira *et al.,* 2012). On the other hand, reduction of traditional practices in Atlantic coastal forest has led to the local loss of cultivar varieties (Peroni & Hanazaki., 2002).

Overharvesting in Amazonian wet forest has caused recent declines in animal populations and basinwide collapse in aquatic species (Antunes *et al.,* 2016). Likewise, many species have proven susceptible to road kill, predation or hunting by humans near roads (Laurance *et al.,* 2009). Hunting of large mammals that disperse seeds of many Neotropical trees can lead to important losses in above-ground biomass (Peres *et al.,* 2016). Defaunation thus has the potential to erode carbon storage, even when only a small proportion of large-seeded trees are extirpated (Bello *et al.,* 2015). The conservation of large frugivorous vertebrates is therefore important to reduce emissions from deforestation and forest degradation.

Box ③ ⑤ Nature´s contributions to people (NCP) of the South American Atlantic coastal forest.

Reflecting the very high NCP contribution of tropical and subtropical moist forest **(Figure 3.25)**, the importance of the Atlantic coastal forest goes beyond its rich and diverse biota. First, Atlantic coastal forest provides water for 125 million people, representing three-quarters of Brazil's population and for electricity production. Additionally, Atlantic coastal forest provides food. The fruits of the Myrtaceae species, palms, legumes, and passion flowers are important components of the diet of traditional and local people, while other species provide raw materials such as fibers and oils. Many traditional populations rely on Atlantic coastal forest vertebrates as a source of protein. This part of the more inclusive tropical and subtropical moist forest biome plays an important role in climate regulation and soil stability. Disrupting this stability signifies increased landslides and floods, with disastrous consequences for human populations. In terms of agriculture-related NCP, Atlantic coastal forest hosts some 60 species of Euglossini bees, known to be long-distance pollinators. Finally, the cultural value of Atlantic coastal forest dates back >8,000 years. Atlantic coastal forest remnants are increasingly important for recreation in urban areas, where they serve as parks or urban forests (Joly *et al.,* 2014).

3.4.1.2 Tropical and subtropical dry forests

Mesoamerican subregion

Status. Tropical and subtropical dry forests are rich in biodiversity, particularly insects, as seen for data for mostly northwestern Costa Rica and Mexico **(Table 3.4)**. The flora of Mexican lowland dry forests shows outstanding endemism (25% at the generic level and 40% the species level) (Challenger & Soberón, 2008). An estimated 72% of this biome, found mostly along the Pacific side of the Mesoamerican subregion, from Panama to western Mexico, is converted (Portillo-Quintero & Sánchez-Azofeifa, 2010). Today Tropical and subtropical dry forests are considered among the most threatened of all terrestrial ecosystems worldwide (Calvo-Alvarado *et al.,* 2013; Janzen, 1988; Frankie *et al.,* 2004). Mexico contains the largest remaining extent in the Mesoamerican subregion (181,461 km^2) (Portillo-Quintero & Sánchez-Azofeifa, 2010).

Tropical and subtropical dry forests have attracted far less attention than tropical moist forests. Not surprisingly, comprehensive information on population trends is less abundant. However, several large mammals have gone locally extinct, including the greater anteater (*Myrmecophaga tridactyla*) from Costa Rica (Janzen, 2002). For the dry forests of Mexico, seven mammals, one reptile, and seven birds have been reported as extinct: twelve plant species have been registered as extinct in states of Mexico dominated by dry forest extinct (Baena & Halfflter, 2008; Flores-Villela & Gerez, 1994). For the Chamela-Cuixmala region of Mexico, at least 40 vertebrate species (fishes not included) are at risk of extinction, representing about 15% of the regional vertebrate diversity (Ceballos *et al.,* 1993).

More open Tropical and subtropical dry forests is more susceptible to invasion than closed moist tropical forest. Invasive species, especially plants, abound. In Chamela, Jalisco, Mexico, 20 exotic species from seven families of plants have been recorded, the grass family *(Poaceae)* being amply represented, along with three exotic animal species, one rodent *(Mus musculus)* and two birds (*Bubulcus ibis* and *Passer domesticus*) (CONABIO, 2016). For Yucatan forests, 90 species of plants from 28 families have been registered as exotic (again, the most species-rich family is Poaceae, followed by legumes) as well as 18 species of animals, including three birds, one rodent and five reptiles (CONABIO, 2016).

Recent trends. Tropical and subtropical dry forests in Mesoamerica have disappeared rapidly over the past 50 years (Bawa *et al.,* 2004; Janzen, 1988). The deforestation rate in Mexico was estimated to be 0.5% per year for the period 1993–2002; by 2002 only 26% of the original cover, by the authors' definition, remained, and only 38% of that is considered to be old-growth forest (Challenger & Dirzo, 2009). Most of this deforestation may be attributed to conversion to pastures and agricultural crops (Masera *et al.,* 1995). However, a major effort to promote natural regeneration of Guanacaste dry forest is ongoing (Calvo-Alvarado *et al.,* 2009) and should serve as a stimulus for other countries in the Mesoamerican subregion for the recuperation of this biome. In the 1970s, the scarlet macaw (*Ara macao*) still occurred in the Guanacaste Conservation Area (Janzen, 2002); reintroduction can be expected in the future as forests regenerate.

The Africanized honeybee (*Apis mellifera*) arrived in the Guanacaste Conservation Area in the early 1980s and now is a low-density member of the local bee fauna (Janzen, 2002). In the 1990s, wild native bee diversity and abundance severely declined throughout Guanacaste Tropical and subtropical dry forests; this is thought to be a possible consequence of reduced flower abundance due to the elimination of pastures and forest not counterbalanced by Tropical and subtropical dry forest restoration (Janzen, 2002). The flammable African pasture grass jaragua (*Hyparrhenia rufa*) has now reached high abundance in Guanacaste, increasing fire frequency with complex impacts on biodiversity (Bonoff & Janzen, 1980; Janzen, 2002; Janzen & Hallwachs, 2016).

Caribbean subregion

Status. Some 92% of the areas suitable for Tropical and subtropical dry forests in the Caribbean are found in Cuba and the Dominican Republic, a total of 124,488 km^2, which is close to 9% of this biome in Latin America overall (Portillo-Quintero & Sánchez-Azofeifa, 2010). Around 66% of Tropical and subtropical dry forests has been converted to nonforest in the Caribbean (66% in Cuba, 78% in Haiti, 58% in the Dominican Republic, 54% in Jamaica and 64% in the Cayman Islands) (Portillo-Quintero & Sánchez-Azofeifa, 2010).

In the insular Caribbean, a typical island pattern of moderate to low species richness **(Table 3.4)** but high species endemism is observed in Tropical and subtropical dry forests (Banda-R *et al.,* 2016). The endemism rate in this biome´s woody plant species is 77.5% in the insular Caribbean (Linares-Palomino *et al.,* 2011). Mirroring the poor conservation state of Caribbean ecosystems, available data show a large proportion of species in Tropical and subtropical dry forests to be vulnerable to extinction or under a greater threat level according to IUCN Red Data List criteria (IUCN, 2017). Terrestrial and freshwater Tropical and subtropical dry forests ecosystems include 51 threatened plant species, 108 threatened reptile species, 16 threatened amphibian species, 35 threatened birds species and four threatened mammal species (IUCN, 2017).

In pre-Columbian times, humans altered habitats using fire and shifting cultivation – especially in Tropical and subtropical dry forests where soils are fertile. Humans also caused the extinction of large mammal species by overhunting or modifying habitat (Fitzpatrick & Keegan, 2007). In European colonial times large areas of this biome were cleared for agriculture in the insular Caribbean, and by the start of the 1900s Tropical and subtropical dry forests on most islands had been largely cleared or degraded (Fitzpatrick & Keegan, 2007; Gould *et al.,* 2012; Lugo *et al.,* 2012).

Recent trends. As mentioned earlier, the Caribbean forest area (both Tropical and subtropical dry forests and moist forests) increased by an average of 0.81% between 1990 and 2010 (FAO, 2011) as agriculture declined on most islands, domestic energy requirements shifted to imported fossil fuels, living standards increased and population levels stabilized or declined and people moved to urban centers from rural areas (Walters & Hansen, 2013). In Puerto Rico, forest cover increased from approximately 5% to over 30% between 1940 and 1990, particularly Tropical and subtropical dry forests (Aide *et al.,* 2000; Ramjohn *et al.,* 2012). However, during the same period urban expansion and tourism lead to declines in Tropical and subtropical dry forests in coastal areas (Gould *et al.,* 2012; Lugo *et al.,* 2012). Notwithstanding, some local declines of the last kind, Caribbean dry forest seems to be on the way to recuperation.

South American subregion

Status. The definition of Tropical and subtropical dry forests in South America lacks consensus (Banda-R *et al.,* 2016; Portillo-Quintero & Sánchez-Azofeifa, 2010; Salazar, *et al.,* 2015). Some authors include the Caatinga and Chaco in tropical and subtropical dry forests while others do not. This makes assessing this biome difficult in South America. The biome scheme adopted by the Americas assessment considers dry Chaco as part of tropical and subtropical savannas and grasslands (3.4.1.6), while Caatinga is considered under drylands (3.4.1.8).

Species diversity in South American Tropical and subtropical dry forests is moderate to high with high species endemism **(Table 3.4)** (Banda-R *et al.,* 2016; Linares-Palomino *et al.,* 2011; Ojeda *et al.* 2003; Pizano & García, 2014; Sandoval & Barquez, 2013). According to one source, between 45-95% of Tropical and subtropical dry forests in the Andean countries has now been converted (Venezuela, 74%; Colombia, 67%; Ecuador, 75%; Peru, 95%; Bolivia, 45%) (Portillo-Quintero & Sánchez-Azofeifa, 2010). The figure for Bolivia is likely to include some Chaco. However, another source for Colombia suggests a greater loss at more than 90% (Gómez *et al.,* 2016; Pizano & García, 2014). Some 58 species of amphibians found in Colombian dry forest have been assessed to be at some level of risk; many mammals likewise are at risk (Pizano & García, 2014).

Recent trends. Reflecting the poorer state of knowledge of tropical and subtropical dry forests compared to moist forests (c.f. 3.4.1.1), little data is available on recent trends in this biome in South America. The biome in Eastern Andean Colombia now shows one of the highest fragmentation levels among all vegetation types (Armenteras *et al.,* 2003). Deforestation rates have descended notably of late in Ecuador (Ministerio del Ambiente, 2014; Sierra, 2013). However, over the period 1990-2008 some 31% of the remaining 4985 km^2 of dry and semi-dry coastal forest was removed (Sierra, 2013). For Venezuela, 88% of 3522 km^2 of Maracaibo Tropical and subtropical dry forests was lost between 1985 and 2010 (Morón Zambrano *et al.,* 2015). These data attest to a general tendency for very high deforestation rates in Tropical and subtropical dry forests in South America (Armenteras & Rodríguez Eraso, 2014) and are of great concern given the high NCP contribution of this biome **(Figure 3.25)**.

3.4.1.3 Temperate and boreal forests and woodlands

North American subregion

Status. Temperate and boreal forests in North America cover most of the eastern USA and Canada and the Pacific Northwest. Boreal forests, which include many coniferous tree species, occur in colder regions, while deciduous hardwood forests occur in both cold and warm temperate regions. Temperate forests occupy ca. 70% of the land area that was forested at the time of European settlement (Flather *et al.,* 1999). Large numbers of plant and animal species depend on these forest habitats. An estimated 90% of the resident or common migrant vertebrate species in the USA (Flather *et al.,* 1999), and likely in Canada, use forest habitats. The number of forest-associated species is highest in the Southeast and in the arid ecoregions of the Southwest (U.S. Forestry Service, 2015).

Several natural forest types and numerous species have been greatly reduced by human activities. For example, longleaf pine, and loblolly and shortleaf pine forests now cover less than 2% of their presettlement ranges (Noss *et al.,* 1995). Less than 1% of North American temperate deciduous forest has not experienced anthropogenic disturbance (Frelich & Reich, 2009). Temperate deciduous forests have a smaller fraction of original primary forest remaining than do boreal or tropical forests, although most of the original species remain present (Frelich, 1995); 94% of forest-associated vascular plants fully occupy their former range (Nelson *et al.,* 2016). Logging, grazing, fire suppression and manipulation of wildlife populations have altered forest composition, structure, and landscape. An estimated 32% of amphibian species and 12–15% of mammals, birds, reptiles, and fish are possibly extinct or at risk of extinction in USA forests (Nelson *et al.,* 2016). In addition, 32–34% of vascular plants and select invertebrates are possibly extinct or at risk of extinction (Nelson *et al.,* 2016) **(Figure 3.17)**.

North American forests sequester large amounts of carbon. In the USA, the highest carbon stock densities (> 80 Mg/ha) are found in the upper Lake States, Pacific Northwest, northern New England and coastal areas of the southeastern USA (Heath *et al.,* 2011). Kurz *et al.* (2013) estimated carbon stock densities above 200 Mg/ha in many managed boreal forests of Canada. However, these values cannot be directly compared because the Canadian estimates included carbon in dead wood and soil. Temperate forests also absorb significant levels of air pollution, including particulate matter, nitrous oxides, sulfur dioxide and ozone, providing benefits to human health (Nowak *et al.,* 2013).

Recent trends. Moderate habitat degradation has occurred over the past 50 years, although forest cover is stable (Fig. 3.2, 3.2) (Hansen *et al.,* 2013), and some sources report that the amount of forest cover has slightly increased (Keenan *et al.,* 2015). Some 92% of the non-federal land in the USA that was in forest land use in 1982, remained as forest in 2007. Of the 12.8 million hectares of forest land that was transformed during this period, most (54%) was converted to developed lands; 22% went into pasture or rangeland, 14% changed to cropland or other another type of rural land, and about 10% went into water areas or federal ownership (USDA, 2007).

The arrival in recent decades of exotic pests and pathogens has caused declines in some of the most highly abundant tree species and genera in North America, including elms and hemlocks (Orwig *et al.,* 2002) and ash and oaks (Juzwik *et al.,* 2011). Tree mortality caused by insects and diseases was reported on nearly 1.82 million hectares in the USA in 2013 (USDA, 2015). Weed *et al.* (2013) identified 27 insects (6 non-indigenous) and 22 diseases (9 non-indigenous) that notably disturb North American forests. In Canada, the mountain pine beetle has killed trees on 20 million ha in British Columbia and Alberta. European earthworms, arriving in plant root balls and introduced for use as fishing bait, have invaded Canada and many parts of the USA and have caused population declines in many native understory herbaceous plant species (Holdsworth *et al.,* 2007; Wiegmann & Waller, 2006). The worms feed on the upper layer of the forest soil, where symbiotic fungi occur, causing fungi as well as the plant species that host them to decline and leading to changes in soil properties, nutrient cycling and ecosystem functions (Frelich *et al.,* 2006; Hendrix *et al.,* 2006; Ewing *et al.,* 2015; Hale *et al.,* 2005; Resner *et al.,* 2015). Oil extraction in the tar sands of Alberta has led to forest losses of 141,000 km^2 (Johnson & Miyanishi, 2008).

High-latitude forests in North America have warmed rapidly since the mid-1900s (Chapin *et al.,* 2005; Allen *et al.,* 2010). From 1902 to 2002 tree ring studies evidence declining growth, with increasing rates of decline since 1942, particularly in critical boreal conifer species (Lloyd & Bunn, 2007). The breeding ranges of some mobile species (e.g. certain bird species), have been expanding northward in association with climate amelioration (USDA, 2007). Current research suggests a northward shift of boreal forests is occurring (yet data is still limited) (Evans & Brown, 2017), with upward altitudinal shifts of tree species in some locations (Beckage *et al.,* 2008).

In the southwestern part of the biome, over the past 30-40 years, forests have come under increasing stress as a result of severe drought. This has seen an increase in tree death, stronger outbreaks of bark beetle and an increase in the area affected by wildfire (Williams *et al.,* 2013) **(Figure 3.18)**, illustrating multiple effects and predicting future changes in forest composition.

South American subregion

Status. South American cool temperate forests are found in Chile and Argentina. Strongly isolated from the nearest closed-canopy forests on the eastern side of South America (Armesto *et al.,* 1998), southern temperate forests are important for carbon sequestration and storage and play a pivotal role in water regulation (Armesto, 2009; Peri *et al.,* 2012). In Chile, where most of southern temperate forest is found, around 78% of the original forest remains (calculated from Luebert & Pliscoff, 2006), thanks to large masses of remote forests in the southern part of the country, much of which is in protected areas. Several forest-dwelling mammals, nevertheless, are threatened (e.g. Darwin´s fox: *Pseudalopex fulvipes; huemul: Hippocamelus bisulcus*), but overall southern temperate forest biodiversity in a far better state than in the Mediterranean-type climate forests to its north.

Plant species (including trees) richness in southern temperate forests is low **(Table 3.4)**. Tree species richness drops off dramatically with latitude, while mean latitudinal range size increases (Arroyo *et al.,* 1996). However, interestingly, these forests have higher woody phylogenetic diversity relative to their species richness than South American forests from lower latitudes (Rezende, 2017). Iconic organisms, including the smallest deer and one of the most long-lived tree species in the world, are important tourist attractions. Geographic isolation has fostered outstanding endemism levels across a wide array of taxa (Arroyo *et al.,* 1996; Stattersfield *et al.,* 1998; Villagrán & Hinojosa, 1997; Vuilleumier, 1985) and include a third of

Figure ③ ⑰ **Trends in the percentage of forest-associated species determined to be possibly extinct or at risk of extinction. Source: Based on Nelson *et al.* (2016), using data from NatureServe (http://www.natureserve.org).**

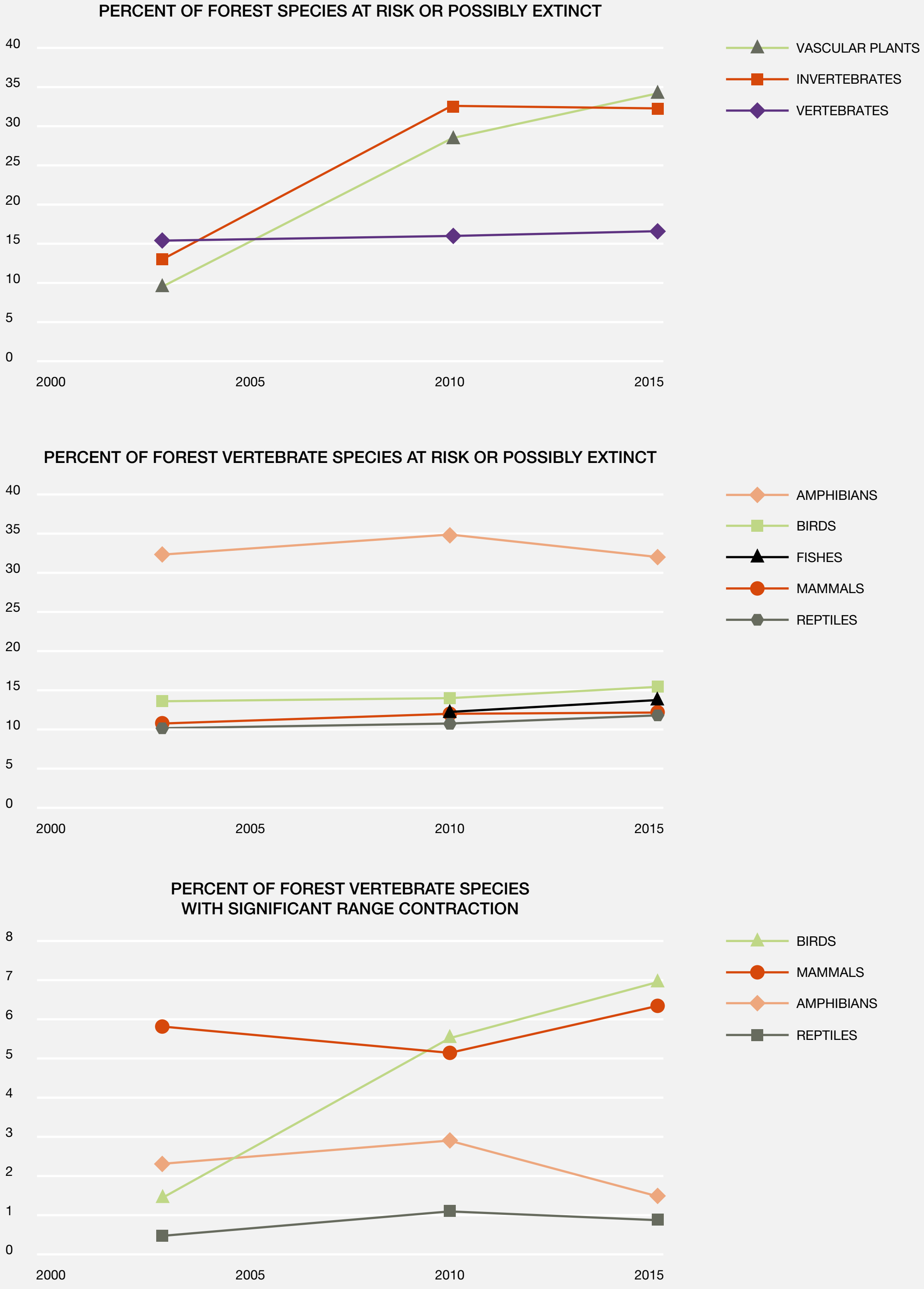
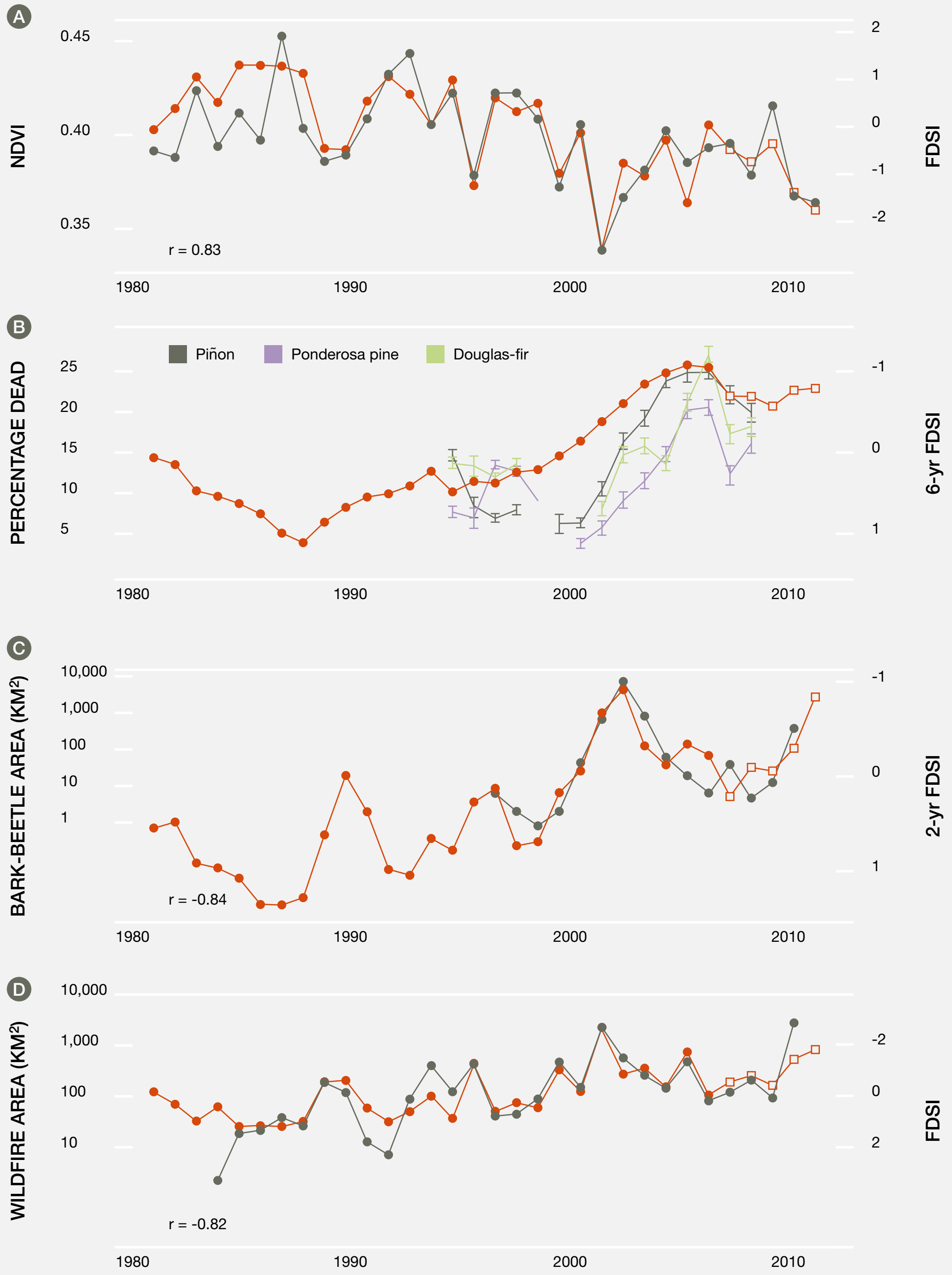


Figure 3 18 **Normalized Difference Vegetation Index (NDVI) Ⓐ, tree mortality Ⓑ, bark beetle outbreak Ⓒ and area affected by fires Ⓓ from 1980–2012 compared with the FDSI (Forest Drought Stress Index, red, right y-axis) for forests in the southwestern USA. Source: Williams *et al.* (2013).**



woody plant genera, two woody families (Arroyo *et al.,* 1996), and almost all trees (Villagrán & Hinojosa, 1997); several endemics are shared with Mediterranean forest. Comprehensive surveys reveal large numbers of edible, medicinal, dye, basketry and ornamental plants and edible fungi in these forests used by indigenous peoples and local people (Tacón *et al.,* 2006). The important ecosystem services supplied by southern temperate forests are enhanced by an especially high level of protection in the far southern part of their distribution (Luebert & Pliscoff, 2006).

Recent trends. Substantial habitat loss has occurred in the northern part of South American temperate forests over the past 40–50 years. The main losses came from deforestation for plantation forestry, farming, and raising of livestock. From the mid to late 90s until around 2013, 138,000 ha of native forest were lost in southern Chile, principally to plantation forestry (70%) (Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas, 2016). From 1985 to 2011 a total gross loss of temperate forest of 30% was reported for the Coast Range in Chile but the net woody cover loss was only 5.1% due to other shrubland and agricultural and pasture land being converted to secondary forest following natural regeneration (Zamorano-Elgueta *et al.,* 2015).

Twelve introduced mammalian herbivores (including three species of deer and beaver) are found in the southernmost forests, leading to altered forest regeneration and increased exotic plant richness in some forest types (Vázquez, 2002). Exotic plants are known to generate significant impacts on biodiversity of understory vascular plants, epigeal beetles and birds in *Nothofagus dombeyi* forest by diminishing species richness, abundance and diversity and generating modifications in assemblage composition (Paritsis & Aizen, 2008). The invasive *Ulex europeaus* has become a serious threat to Chilean agriculture and plantation forestry in some parts of the temperate forest zone (Norambuena *et al.,* 2000). Exotic beavers cut down trees and have altered water regulation, silting levels and landscape values (see also chapter 4). Introduced conifers have now began to seed naturally in steppe vegetation and are associated with declines in plant species richness and cover (Taylor *et al.,* 2016).

Fast-growing exotic plantation trees tend to consume more water than native trees and can be associated with reduced seasonal water provision (Lara *et al.,* 2009). Nevertheless, there have been some recent positive signs of native forest recuperation. Between 1983 to 2007, in a part of the Araucania in Chile, the dominant land cover transitioned from agriculture to native vegetation, with largest increases occurring around residential areas found close to closed stands of native forest (Petitpas *et al.,* 2016). These positive changes are attributed to the growth of tourism and a growing cultural preference for "natural" spaces.

On a longer timescale, in northwestern Patagonia in Argentina, during the last century, forests (mainly *Nothofagus*) expanded to cover almost 50% of the historically burned land, and more than 60% of the shrublands (Gowda *et al.,* 2012). The estimated carbon stock recovery time for severely burned *Nothofagus* forests in Patagonia is 150¬–180 years (Bertolin *et al.,* 2015) indicating a severe ecosystem service loss due to burning. However, regrowth is far from homogeneous in time and space: net forest expansion took place mainly from 1914 to 1973, probably favored by a wetter climatic period, and has shown a marginal retraction since then. Although forest gains remained high during the last 30 years, substantial areas of forests in this area were converted to grasslands and shrublands as a result of recent fires associated with extremely dry springs (Gowda *et al.,* 2012). A major drought in 1998–1999 coincident with a very hot summer led to extensive dieback in a *Nothofagus* species (Suarez *et al.,* 2004). In another dominant *Nothofagus* species, several periodic droughts have triggered forest decline as of the 1940s (Rodríguez-Catón *et al.,* 2016).

Over the past 20-30 years, the biodiversity of southern temperate forests has become widely recognized for its ecotourism and tourism values. For example, the recent scientific finding of outstanding bryophyte diversity in the southern temperate forests, which led to the concepts of "miniature forests" and "tourism with a hand lens" (Rozzi *et al.,* 2008) in the Cape Horn Biosphere Reserve on the southern tip of the continent, has seen a substantial increases in visitors, favoring local human well-being in an area where climate precludes agriculture and plantation forestry.

3.4.1.4 Mediterranean forests, woodlands and scrub

North American subregion

Status. The Mediterranean climate zone in North America encompasses the California Floristic Province, including southwestern Oregon, California west of the Sierra Nevada and a portion of northwestern Baja California, Mexico (Baldwin *et al.,* 2012; see Ackerly *et al.,* 2014, for a stricter definition and mapping of Mediterranean-climate regions based on current climate). The broader Mediterranean forests, woodland and scrub area has a very rich and endemic flora **(Table 3.4)** (Burge *et al.,* 2016), with many evolutionary lineages represented (Baldwin, 2014). High levels of plant endemism are found in ephemeral vernal pools (Keeley & Zedler, 1998) and on serpentine soils (Anacker, 2014). California has more than 300 endangered and threatened species listed by the USA government, the largest for any USA state, and more than 100 others are listed by the state (California Natural Resources Agency, 2015). Hobbs & Mooney (1998) report 49 extinct taxa for seven groups of organisms (including some subspecies) (34 for plants) along with numerous cases of local population extinctions. According to the most recent account, 17 taxa (13 species and four subspecific taxa) of Californian vascular plants are globally extinct (Rejmánek, 2017) with 15 additional species extinct in California but found elsewhere (together 0.53% of the Californian flora); extinctions are associated with small range sizes and lowland habitats.

North American Mediterranean forests, woodland and scrub houses 991 species of alien plants and 109 species of alien vertebrates (including 26 mammals) (Zavaleta *et al.,* 2016).

Some 183 plant species are currently listed as invasive plants capable of damaging the environment and economy by the California Department of Food and Agriculture (California Natural Resources Agency, 2015). Coastal sage is very heavily invaded (Cleland *et al.,* 2016). Brooms and gorse invade woodlands and shrublands and can displace native vegetation when not controlled (California Invasive Plant Council, 2017).

Forests in the Sierra Nevada play a critical role in water supply. Most urban and agricultural water originates in these mountains, and 30% of California's water is stored for a part of the year in the snowpack. Healthy forests reduce flood risks and lead to more predictable water flows.

Recent trends. In the past 50 years, urbanization, exurban development, and agriculture have caused considerable conversion of natural habitat (Brown *et al.,* 2005; Wilson *et al.,* 2016); for example, a fourfold increase in vineyard acreage between 1976 and 2010 removed much oak woodland in coastal counties (Davis *et al.,* 2016). Vegetation fragmentation — possibly exacerbated by changing climate in some cases – and the secondary effects of urbanization such as predation by urban cats on birds have reduced butterfly richness, bird abundances, genetic connectivity and species diversity in some taxa and produced declines in plant species richness in different vegetation types (Benson *et al.,* 2016; Casner *et al.,* 2014; Cooper *et al.,* 2012; Johnson & Karels, 2016). Nevertheless, urban and semi-urban areas can house considerable plant diversity (Schwartz *et al.,* 2006) and support high levels of bee diversity (Frankie *et al.,* 2009) and thus could turn out to be very relevant for conservation.

Mediterranean forests, woodland and scrub has experienced warming (Diffenbaugh *et al.,* 2015). Upward elevational range shifts, consistent with warming, have been reported in small mammals (Moritz *et al.,* 2008), birds (Tingley *et al.,* 2009) and plants (Wolf *et al.,* 2016), as well as earlier butterfly appearance (Forister & Shapiro, 2003) and arrival of migratory birds. Downward elevational shifts have also been reported in birds (Tingley *et al.,* 2009) and plants (Crimmins *et al.,* 2011). For plants, there is disagreement both about the trends and inferred link to climate (Stephenson & Das, 2011). Since the 1920s, tree densities increased and size class distributions have changed in forests across California (Dolanc, *et al.,* 2014; Dolanc *et al.,* 2014; (McIntyre *et al.,* 2015), in part due to changing fire regimes (see below). Reductions in the density of large trees are correlated with increased severity of summer water deficits (McIntyre *et al.,* 2015).

California experienced a severe drought from 2012 to 2016, and even before it ended some calculations estimated that it exceeded in duration and intensity those observed for at least a century and possibly more than 1,000 years (Griffin & Anchukaitis, 2014). By one estimate, the intensity of the drought was increased by up to 27% due to increased temperatures on top of low rainfall (Williams *et al.,* 2015). Widespread tree mortality has been observed, especially in Sierra Nevada conifer forests, with estimates exceeding 100 million dead trees spread over more than 3 million ha of forest (US Forest Service, 2016).

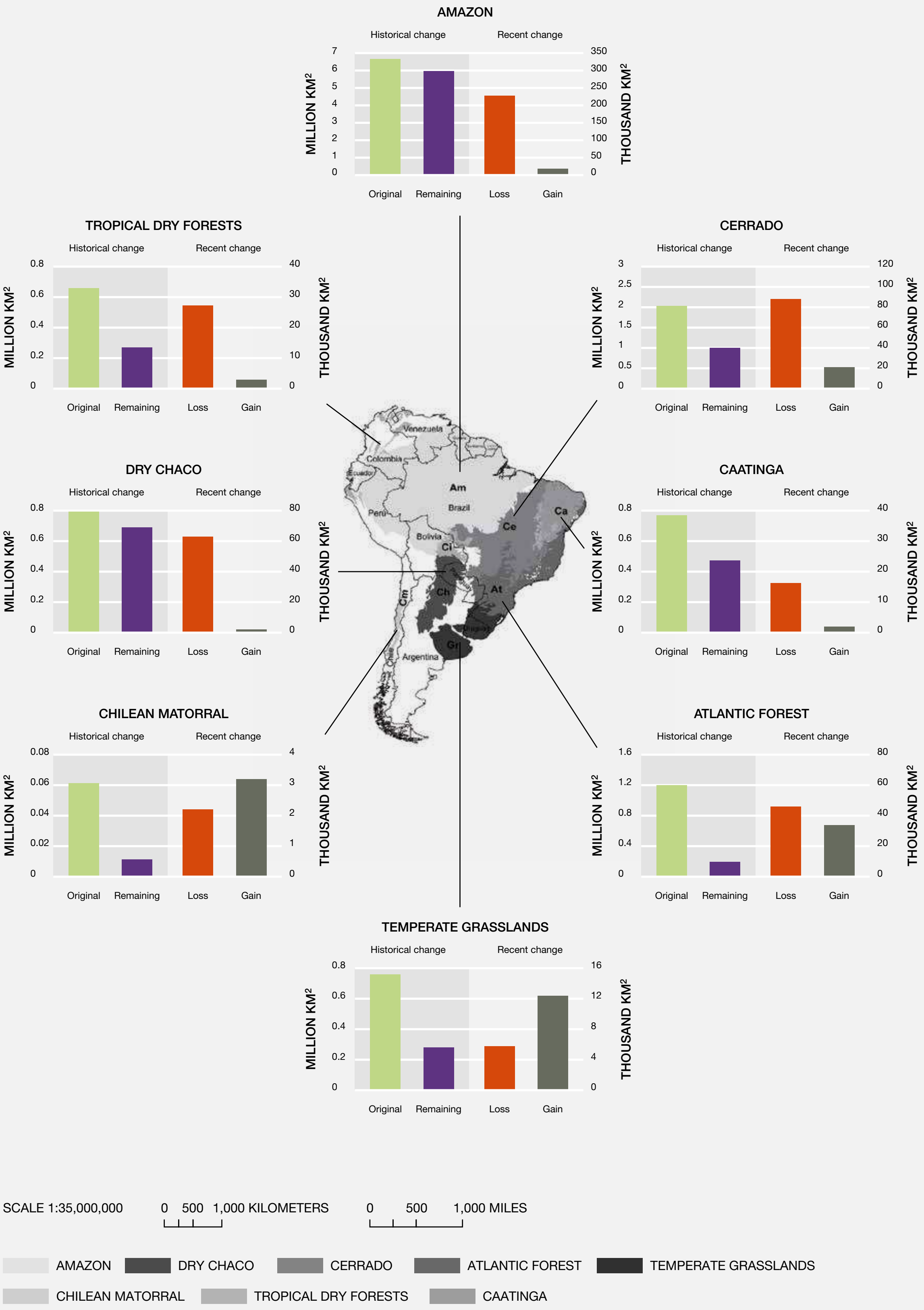
Several recent invasions of pathogens and disease have impacted biodiversity. Virulent pathogens affecting amphibians have been detected in a high proportion of wetlands (Hoverman *et al.,* 2012); chytrid fungus has been attributed to amphibian declines in northern California, especially in high elevation populations of mountain yellow-legged frog (Piovia-Scott *et al.,* 2015; Briggs *et al.,* 2005). Sudden oak death (*Phytophthora ramorum*) arrived in the mid-1990s on horticultural trade plants and has caused extensive oak mortality in moist-climate, coastal woodlands (Zavaleta *et al.,* 2016).

In Sierra Nevada pine forests, fire suppression led to marked increases in overall forest density, especially in small trees (McIntyre *et al.,* 2015). Dense forests contribute to catastrophic wildfires that exceed the range of historical fire variability, such as the 104,000 ha Rim Fire in 2013, the largest fire on record in the Sierra Nevada (Kane *et al.,* 2015). At mid- to high elevations, larger areas are being burned, likely due to past fire suppression, changing fire management policies, and warmer and drier climatic conditions. The length of the fire season increased by over two months from 1970 to 2003, associated with warming trends (Westerling *et al.,* 2006). More frequent fire has led to much type conversion of shrubland to grassland (Zedler, 1995; Halsey & Syphard, 2016).

South American subregion

Status. Part of a Biodiversity Hotspot (Myers *et al.,* 2000), South American Mediterranean forests, woodland and scrub found in central Chile, under a broad definition, is characterized by high endemism, richness and phylogenetic diversity (Arroyo *et al.,* 2002; Rundel *et al.,* 2016; Scherson *et al.,* 2017) **(Table 3.4)**. Around 50% of Mediterranean forests, woodland and scrub has been transformed (Luebert & Pliscoff, 2006) – this percentage is considerably higher under a narrower definition of the biome **(Figure 3.19)**. Many native species are threatened (Ministerio del Medio Ambiente, 2017), although only a small fraction (ca. 3.5%) of all Chilean species have been analyzed (OECD/ECLAC, 2016). Alien species including close to 600 plant species (Fuentes *et al.,* 2015; Jiménez *et al.,* 2008), >100 insect species (Ministerio del Medio Ambiente, 2017), and 30 vertebrate species (Iriarte *et al.,* 2005; Jaksic, 1998) – several of which are considered harmful by stakeholders (COCEI, 2014) – are abundant in disturbed areas, urban areas, and semi-natural grasslands (Arroyo *et al.,* 2000; Contreras *et*

Figure 3 19 **Total change in vegetation type and recent change (2000–2012) in forest cover for several biomes in South America. Source: Modified from Salazar *et al.* (2015).**



al., 2011; Estay, 2016; Figueroa *et al.,* 2011; Gärtner *et al.,* 2015; Jaksic, 1998; Martín-Forés *et al.,* 2015). Plant-animal interactions for pollination and seed dispersal are especially well developed and critical for vegetation regrowth and restoration. Other biodiversity-NCP links include the provision of medicinal plants (Niemeyer, 1995), nectar and pollen sources for honey making (Bridi & Montenegro, 2017), runoff control on steep slopes (Pizarro Tapia *et al.,* 2006) and the aesthetic value of the rural-natural landscape mosaic.

Recent trends. One study suggests Mediterranean forests have recently increased but this is acknowledged as likely due to the inclusion of exotic forests **(Figure 3.19)** (Salazar *et al.,* 2015). National data for approximately between the last decade of the past century and the first of this century for Mediterranean-climate forest (V-VIII Regions) come up with a net loss of 99,451 ha, mainly distributed among conversion to exotic plantation forestry (24%), agriculture (11%), scrub and open vegetation (59%), and urban areas (2%) (Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas, 2016). Exotic plantation forestry accounted for most of the forest loss in the southern part of the biome. Although some passive renovation has been occurring, previously forested areas tend to remain as scrub (Schulz *et al.,* 2010, see also Hernández *et al.,* 2016). Plantation forests have been shown to have a negative effect on annual stream flow in the biome (Iroumé & Palacios, 2013) and loss and fragmentation of native forests have negatively affected many plant and animal species (Braun & Koch, 2016; Bustamante & Castor, 1998; Muñoz-Concha *et al.,* 2015; Saavedra & Simonetti, 2005; Soto-Azat *et al.,* 2013; Vergara *et al.,* 2013; Vergara & Simonetti, 2004) and pollination services to native plants (Valdivia *et al.,* 2006).

Among the new insect invaders (Grez *et al.,* 2010; Ide *et al.,* 2011; Lanfranco & Dungey, 2001; Montalva *et al.,* 2011) and introduced fungal diseases (Durán *et al.,* 2008; Slippers *et al.,* 2009), some are spreading at remarkable rates (e.g. Schmid-Hempel *et al.,* 2014, Grez *et al.,* 2016). *Bombus terrestris*, introduced in the 1990s for crop pollination, moved rapidly into Argentina and is now displacing native bumblebees there (Geslin & Morales, 2015). Many native plant species in Mediterranean forests, woodland and scrub are visited by *B. terrestris* (Montalva *et al.,* 2011), but the impacts of *B. terrestris* on the wider bee fauna of central Chile **(Table 3.4)**, likely to assist crop pollination, are unknown. The escaped introduced frog *Xenopus laevis* has now been found to harbor amphibian pathogens, posing a potential threat to the biome´s highly endemic amphibians (Soto-Azat *et al.,* 2016) and showing that single invasions can have secondary effects.

Between 1994 and 2015, fire affected close to 128,000 ha of closed Mediterranean forest as well as huge areas of exotic plantation forests (based on Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas, 2016). A recent megadrought ushered in a notable increase in fire frequency and extent in Chile (with most fires in the Mediterranean area) **(Figure 3.20)**, culminating in the massive forest fires of the austral summer of 2016 which affected 518,000 ha, including 105,000 ha of native forest and 284,000 ha of exotic forest plantations (CONAF, 2017), mostly in the Mediterranean zone. Although there is still some discussion on the issue, it is generally agreed that unlike North American Mediterranean forests, woodland and scrub, South American Mediterranean forests, woodland and scrub was cut off from natural lightning strike fires as of the Miocene and consequently is not strongly adapted to fire (Rundel *et al.,* 2016). Although many native woody species can resprout after fire, recovery of Mediterranean forest may require 25–30 years and often is never complete (Montenegro *et al.,* 2003), indicating limited resilience. Fire additionally provokes the entrance of invasive species (Contreras *et al.,* 2011; Gómez-González *et al.,* 2011; Gómez-González & Cavieres, 2009; Pauchard *et al.,* 2008) further altering species composition and NCP delivery. Warmer and drier conditions in central Chile also saw a significant decrease in growth rates of *Nothofagus macrocarpa* as of the 1980s (Venegas-González *et al.,* 2018).

Urban expansion in central Chile, often recent, has also contributed to local habitat and species losses (Pauchard *et al.,* 2006; Pavez *et al.,* 2010; Simonetti & Lazo, 1994). However, urban spaces clearly can play an important role in maintaining biodiversity, as seen by the 42 native bee species in a semi-natural botanical garden in Santiago (Montalva *et al.,* 2010).

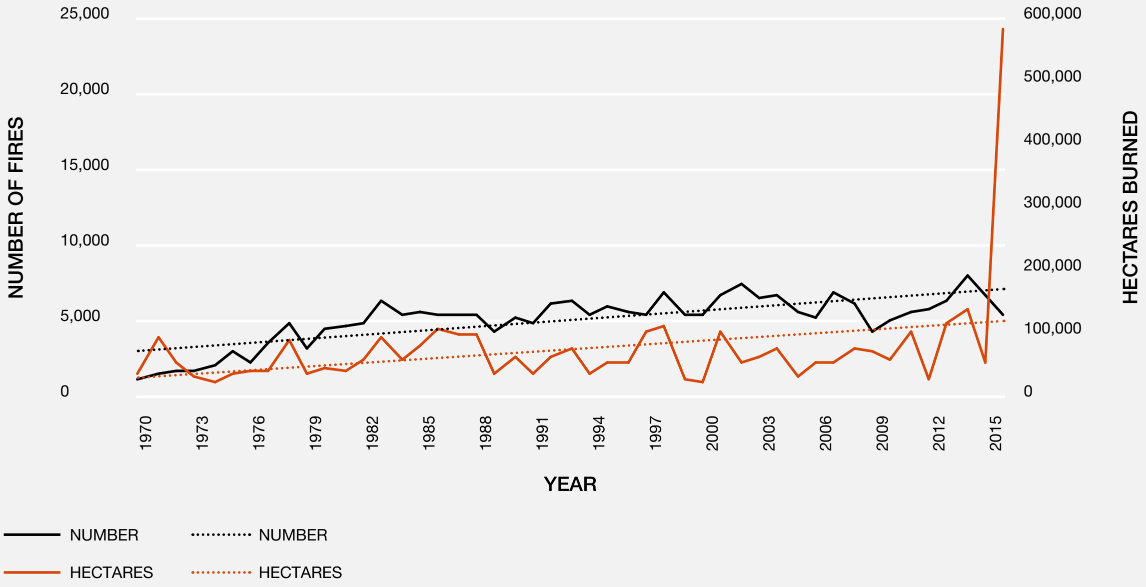
3.4.1.5 Tundra and high mountain habitats

North American subregion

Status. Species richness beyond latitudinal treeline in North American arctic tundra is low in relation to its vast area **(Table 3.4)**, and decreases with increasing latitude (Meltofte, 2013; Walker, 1995). Endemism is rare because many tundra-adapted taxa are distributed across both North American Arctic tundra (including Greenland) and the Eurasian arctic tundra. For example, 80% of vascular plant species in the Arctic are common to both regions, so just 1.1% of North American Arctic tundra vascular plant species are endemic (Callaghan *et al.,* 2004; Elven at al., 2011). The extent and biodiversity of the North American arctic tundra remain largely unchanged compared to pre-European settlement, with localized reductions in extent associated with natural resource extraction and permanently settled villages and cities (Raynolds *et al.,* 2014; Young & Chapin III, 1995). Non-native species in arctic tundra are uncommon and usually associated with human activity (Ackerman & Breen, 2016; Elsner & Jorgenson, 2009; Forbes & Jefferies,

Figure 3 20 **The number of forest fires (grey line) and hectares affected (red line) in Chile between 1970 and 2016.**

Year refers to the austral spring-summer season beginning in the year indicated. Source: Data from http://www.conaf.cl/incendios-forestales/incendios-forestales-en-chile/estadisticas-historicas/.



1999). Carbon storage in North American Arctic tundra soils is high relative to other biomes, due to low rates of organic matter decomposition. Hugelius *et al.* (2013) estimate 25–100 kg C/m^2 across most of North American Arctic tundra. At local scales, stocks of carbon and soil nutrients vary widely based on vegetation community type (Shaver *et al.,* 2014). Across all community types, soil nitrogen is dominated by non-mineral forms, so primary productivity in North American Arctic tundra is often limited by rates of nitrogen mineralization (Chapin & Shaver, 1985; Shaver *et al.,* 2014; Chapin *et al.,* 1995).

Globally, North American Arctic tundra stores carbon in soils frozen year-round called permafrost (Michaelson *et al.,* 1996). Biodiversity alters this ecosystem service through plant traits (Chapin *et al.,* 2000). For example, plants with extensive mat growth forms, like *Sphagnum* spp., insulate permafrost soils from direct sunlight (O'Donnell *et al.,* 2009). Permafrost stores 1,330–1,580 picograms of organic carbon, nearly half of the global organic carbon pool (Schuur *et al.,* 2015). Locally, North American Arctic tundra benefits subsistence hunters, providing game species including caribou (*Rangifer tarandus*) and ptarmigan (*Lagopus* spp.) (Alaska Department of Fish and Game, 2016).

Western North American alpine ecoregions contain diverse ecosystems and over 1,400 plant species (Malanson *et al.,* 2015). Similarity among plant communities throughout mountain ranges of western North America is driven primarily by geographic distance, but also by hydroclimatic variables (Malanson *et al.,* 2015). Endemism is common in western North America (45% of plant species), while exotic species are rare (Malanson *et al.,* 2015). Native biodiversity of the western North America high altitude areas remains largely intact since European colonization. The eastern North America alpine ecoregion (Appalachian Mountains) is understudied and lacks a comprehensive record of biodiversity.

Recent trends. Species richness has not changed significantly in North American Arctic tundra. Some boreal plant species, including trembling aspen (*Populus tremuloides*) and white spruce (*Picea glauca*), have expanded locally into North American Arctic tundra due to infrastructure development (Ackerman & Breen, 2016; Elsner & Jorgenson, 2009). The only reported extinction is the Eskimo curlew (*Numenius borealis*), an over-exploited migratory shorebird (Harris *et al.,* 2012). It is very well established that woody deciduous shrubs native to North American Arctic tundra have become increasingly dominant due to warming (Fraser *et al.,* 2014; Moffat *et al.,* 2016; Myers-Smith *et al.,* 2011, 2015; Naito & Cairns, 2015; Pizano *et al.,* 2014; Sturm *et al.,* 2001; Tape *et al.,* 2006; Tremblay *et al.,* 2012). While the overall area of North American Arctic tundra has not changed significantly, habitat has been degraded biome-wide due to high-latitude concentration of atmospheric pollutants (Hung *et al.,* 2010; Krachler *et al.,* 2005; Quinn *et al.,* 2007) and regionally due to road construction (Auerbach *et al.,* 1997; Hinkel *et al.,* 2017; Raynolds *et al.,* 2014; Walker & Everett, 1987).

Above-ground standing biomass has increased at low latitudes in the Arctic (Epstein *et al.,* 2012), and vegetation cover has increased in mid- to high-latitudes, possibly due to increased maritime climate moderation linked with sea ice decline (Bhatt *et al.,* 2010). Despite elevated productivity, overall carbon storage across North American Arctic tundra has decreased since 1970 due to warming-induced carbon losses from soil (Hayes *et al.,* 2014; Hinzman *et al.,* 2005; Oechel *et al.,* 2000; Schuur *et al.,* 2009). Recent trends in water balance are uncertain, though there has been a general acceleration of the hydrologic cycle across North American Arctic tundra due to changes in precipitation, evapotranspiration, and drainage conditions (Andresen & Lougheed, 2015; Bring *et al.,* 2016; Cherry *et al.,* 2014; Hinzman *et al.,* 2005; Liljedahl *et al.,* 2016; Oechel *et al.,* 2000; Rawlins *et al.,* 2010; Vihma *et al.,* 2016; Young *et al.,* 2015).

Greater variability in the timing and magnitude of precipitation events in North American Arctic tundra has decreased accessibility and yield for subsistence hunters (Berkes & Jolly, 2002; Rennert *et al.,* 2009). Further, atmospheric deposition of pollutants in North American Arctic tundra has threatened the health of local communities through the bioaccumulation of toxins in organisms used for food (Kelly & Gobas, 2001). To improve community resilience to these changes, Chapin *et al.,* (2006) suggest diversifying the economies of indigenous communities by reinvesting tax revenue from natural resource extraction into local education and infrastructure.

The extent of alpine habitat in western North America has decreased due to warming-induced treeline advance, though rates of advance are spatially variable (Elliott, 2011; Harsch *et al.,* 2009). Some degradation from logging, pasturing, and recreation is evident, but these disturbances have been minor compared to in alpine zones on other continents (Bowman & Seastedt, 2001). Recent changes include increased shrub cover and diminished species richness, likely in response to a combination of climatic change, and high levels of nitrogen deposition from anthropogenic pollution (Elmendorf *et al.,* 2012; Formica *et al.,* 2014; Sproull *et al.,* 2015). The most notable change among alpine fauna populations is the rapid decline of the American pika, a small alpine mammal experiencing an upslope range contraction in response to climate warming (Beever *et al.,* 2011, 2016; Stewart *et al.,* 2015).

South American subregion

Status. South American high elevation habitats occur principally along the entire length of the Andes (Arroyo & Cavieres, 2013). These habitats, found under a variety of climatic conditions, are remarkably rich in plant species **(Table 3.4.)** and evolutionary lineages (Sklenář *et al.,* 2011) and support the richest tropical alpine flora in the world (Sklenář *et al.,* 2014). High-elevation habitats support many species of large mammals (Ojeda *et al.,* 2003), lizards (Pincheira-Donoso *et al.,* 2015), birds (Arbeláez-Cortés *et al.,* 2011; Fjeldså & Rahbek, 2006; Fjeldså, 2002) and pollinating insects (Arroyo *et al.,* 1982). Puna lakes supports 58 species of native fishes (Vila *et al.,* 2007), diverse waterfowl (Cendrero *et al.,* 1993), and rich assemblages of gastropods (Kroll *et al.,* 2012), while hot springs and periglacial soils fascinating assemblages of microorganisms (Costello *et al.,* 2009; Schmidt *et al.,* 2009).

Species-level endemism and turnover in the high tropical Andes can be very high (Londoño *et al.,* 2014; Sklenář *et al.,* 2014). Mountain-top vegetation is richer in plant genera and species in páramo compared to puna (Cuesta *et al.,* 2017) but the puna and southern Andean steppe house more endemic genera than páramo (Arroyo & Cavieres, 2013). Páramo and puna have long been under human influence **(Box 3.6)**, but more intensely so as of colonial times (Vásquez, *et al.,* 2015). In the high southern Andes, human influence has never been very great. Today it is principally via low-intensity transhumance summer grazing, skiing, and mining. Some Andean threatened species rely heavily or partially on páramo, among them the Andean condor (*Vultur gryphus*), the mountain tapir (*Tapirus pinchaque*), the Andean bear (*Tremarctos ornatus*), and several deer species (*Pudu mephistophiles, Mazama rufina, M. americana* and *Odocoileus virginianus*) (Muñoz *et al.,* 2000). In general, South American high elevation habitats have garnered few alien plant species (Alexander *et al.,* 2016; Barros & Pickering, 2014; Luteyn, 1999; Urbina & Benavides, 2015) and these are mostly confined to disturbed areas. A few serious recent invasions have now been recorded for páramo, as for example *Ulex europaeus* in Colombian páramos (see **Table 3.2**) and more can be expected in the future given trends in alteration **(Box 3.6)**.

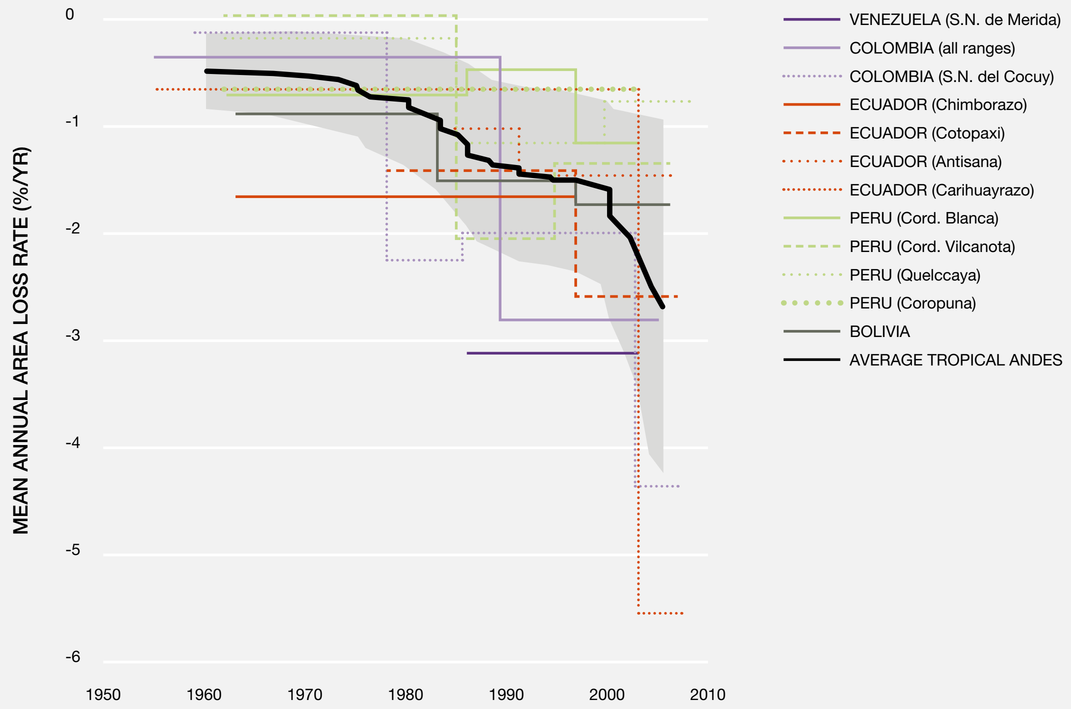
Páramo and wet puna are notable for rapid water absorption but slow water release (Buytaert *et al.,* 2005), which is important for the support of agriculture and the delivery of water to lowland areas. For example, 60% of water in Colombia derives from páramo (Cadena-Vargas & Sarmiento, 2016). Carbon storage is páramo is important (Forero *et al.,* 2015). In particular, it is very high in páramo peatlands (Hribljan *et al.,* 2015; 2016). Soils under older pine plantations in páramo have lower carbon content and retain less water compared with natural grasslands (Farley *et al.,* 2004, 2005, 2013) and the loss of water retention after afforestation may be the dominant factor in carbon loss (Farley *et al.,* 2004).

Recent trends. Páramo and puna have seen an increasing trend for afforestation with fast-growing exotic trees and intensive agriculture. Both afforestation and cultivation have been found to increase streamflow variability and decrease catchment regulation capacity and water yield.(Ochoa-

Box 3 6 The role of páramo and puna for human well-being.

Humans were living at 4,480 m.a.s.l. some 11,500 years ago in the puna of Peru (Rademaker *et al.,* 2014) and at 3,000–3,600 m.a.s.l some 13,000 calibrated years before the present on the Chilean altiplano (Núñez *et al.,* 2002). High altitude indigenous peoples of the páramo and puna have accumulated a wealth of ILK on high Andean biodiversity, especially useful plants (Aldunate *et al.,* 1983; Brandt *et al.,* 2013; Califano & Echazú, 2013; Huamantupa *et al.,* 2011; Monigatti *et al.,* 2013; Pauro *et al.,* 2011; Ramos *et al.,* 2013; Thomas *et al.,* 2008, Villagrán *et al.,* 2003) and have developed resilience to climatic extremes by managing alternative crop varieties Local inhabitants have developed their own taxonomic systems reflecting thousands of years of interchange between different linguistic groups (Aldunate *et al.,* 1983; Villagrán & Castro, 2003). High Andean bogs in the arid puna are key habitats for native camelids which sustain the livelihoods of high altitude peoples (Borgnia *et al.,* 2008; Tirado *et al.,* 2016). The integrity of the páramo and puna thus is critical to conserving ILK and for the livelihoods of local inhabitants. Páramo and wet puna play critical roles in supplying water supply to lowland Andean areas. Water availability is threatened on two counts. First, severe glacier dieback has occurred over the past decades **(Figure 3.21)**. Second, páramo and puna are increasingly being converted to other land use types involving higher water-demanding trees (c.f., Hofstede *et al.,* 2002) and crops. Around 16% of Colombian páramos have been now been transformed (Bello *et al.,* 2014) mainly due to cropping and pastures. Peruvian Jalca grasslands were transformed at the rate of 1.5% per year over a 20 year period starting 1987 due mostly to more intensive agriculture and afforestation (Tovar *et al.,* 2013). Rapid glacier melt also portends landslides on unconsolided deglaciated substrates following heavy rains.

Figure 3 21 Compilation of mean annual area loss rates for different time periods for glaciated areas in the northern Andes between Venezuela and Bolivia. Source: Rabatel *et al.* (2013).



Tocachi *et al.,* 2016). Moreover, shifts to agriculture lead to a loss of microbial functional diversity in páramo, which is reflected in lower metabolic activity. Fishing, based on native species, is a longstanding tradition in some large high Andean lakes. However, the introduction of trout (and silversides) together with more invasive fishing techniques has seen a decline in endemic native fish (Vila *et al.,* 2007).

High elevation areas have warmed in the southern (Falvey & Garreaud, 2009) and northern (Hofstede *et al.,* 2014) Andes. Whether and the degree to which anthropogenic warming has affected tree growth and the position of the treeline along the Andes are still somewhat unclear. Anthropogenic warming seems to have affected tree growth and increased recruitment above treeline in some places, but not in others

(Aravena *et al.,* 2002; Daniels & Veblen, 2004; Fajardo & McIntire, 2012; Lutz *et al.,* 2013; Rehm & Feeley, 2013; Villalba *et al.,* 1997). Some tree species have been moving upward below treeline (Feeley *et al.,* 2011). Lack of or very slow upward movement of the treeline might reflect recruitment difficulties in high altitude grasslands (Rehm & Feeley, 2013, 2015, 2016) or under reduced precipitation in some parts of the Andes. Historical comparisons suggest upward movement in some plant and beetle species in the northern Andes (Moret *et al.,* 2016; Morueta-Holme *et al.,* 2015; but see Sklenář, 2016). In the longer term, contractions of total area occupied can be expected in high elevation species under warming given that land area decreases with increasing elevation throughout much of the Andes. Warmer soil conditions in the páramo are expected to cause faster organic carbon turnover thereby decreasing below-ground organic carbon storage (Buytaert *et al.,* 2011).

3.4.1.6 Tropical savannas and grasslands

South American subregion

Status. In South America, this biome occurs mainly in Brazil, Paraguay, Argentina, Venezuela, Colombia, and Bolivia. The largest extents are the Cerrado, originally covering around 2 million km^2, and the Dry Chaco, originally over ¾ of a million square kilometers (Salazar *et al.,* 2015).

Comprising a mosaic of tall savanna woodlands, gallery forests and treeless grasslands, Cerrado is a recognized Biodiversity Hotspot (Myers *et al.,* 2000). It is characterized by high plant and bird species richness and endemism **(Table 3.4)**. Birds use many habitats, especially forested areas (Carmignotto *et al.,* 2012), lizards prefer open interfluvial habitats (Nogueira *et al.,* 2009), while large mammals use a wide range of habitats (Lyra-Jorge *et al.,* 2008), including converted land (Cárceres *et al.,* 2010). Over half of Cerrado mammals and birds consume fruits with about one-third of Cerrado plants depending on birds and mammals for fruit and seed dispersal (Kuhlmann & Ribeiro, 2016). Mammals and birds thus are fundamental for natural Cerrado regeneration.

Some 52% of all South American Cerrado has been converted (Salazar *et al.,* 2015) **(Figure 3.19)**. According to (Beuchle *et al.,* 2015), 47% of Brazilian Cerrado has been transformed. Remaining Cerrado is highly fragmented with the landscape dominated by crops and pastures (Carvalho *et al.,* 2009). Fragmentation reduces species richness and alters the composition of small mammals land (Cáceres *et al.,* 2010), and birds (Marini, 2001). However, large mammals, which tend to use the entire the landscape, appear less susceptible (Cáceres *et al.,* 2010; Vynne *et al.,* 2014). Shrubby pastures in Cerrado hold far more bird species than cleared ones and obligate natural grassland bird species do not adapt well to pastures (Tubelis & Cavalcanti, 2000). Butterfly richness and beta diversity are lower in disturbed riparian Cerrado forest (Cabette *et al.,* 2017). Additional threats to Cerrado biodiversity are fire suppression (Durigan & Ratter, 2016) and woody encroachment (Stevens *et al.,* 2017). Cerrado is resilient to fire, expressed in rapid post-fire recuperation and fire aids in maintaining the mosaic structure of Cerrado. Replacement of grassy savannas with forests is also considered a threat (Veldman *et al.,* 2015) because dense tree cover severely limits the richness and productivity of light-demanding herbaceous plants while reducing habitat for animals adapted to open environments. Several African grasses which were introduced into Brazil for cattle grazing are now highly invasive in the Cerrado leading to reductions in native plant species (Almeida-Neto *et al.,* 2010). In the phosphorus-poor Cerrado, the addition of phosphorus tends to increase the biomass of alien C4 grasses (Lannes *et al.,* 2016).

Some 34% of dry Chaco habitat has been converted **(Figure 3.19)** (Salazar *et al.,* 2015). The Gran Chaco has a long history of colonization and land use change, beginning with subsistence hunting by native people. Over the past 200 years, dry Chaco has experienced drastic land use changes as a result of intensive agriculture, livestock production and logging (Eva *et al.,* 2004; Hoyos *et al.,* 2013). Moreover, deforestation and the introduction of domestic cattle have led to the elimination of fire-climax grasslands and altered forest composition and structure (Bucher, 1982; Gasparri & Grau, 2009). Chaco conversion has had negative effects on biodiversity. Almost 50% of the largest frugivorous mammals and 80% of the largest herbivores in the Argentine Chaco are threatened and exhibit declining populations; this is expected to change vegetation composition since more than half of Chacoan woody plant species display endozoochory as their seed dispersal mechanism (Periago *et al.,* 2015).

Recent trends. The South American tropical and subtropical savannas and grasslands assessed here are strongly imperiled. As of around the 1970s, pasture development for cattle grazing and extensive and mechanized agriculture intensified, leading to the transformation of Cerrado into a vast commercial production landscape with concomitant charcoal production for the steel industry. Brazilian Cerrado suffered a gross loss of around 266,000 km^2 of natural vegetation between 1990 and 2010, but with a significant amount of regrowth also occurring (Beuchle *et al.,* 2015). Although the annual net rate of loss (total loss adjusted for regrowth) slowed in the last decade of the past century, overall conversion occurred an average annual net rate of –0.6% between 1990 and 2010 (Beuchle *et al.,* 2015). Between 2003 and 2013, the northeast agricultural frontier in Brazil more than doubled from 1.2 to 2.5 million ha, with

74% of new croplands sourced from intact Cerrado (Spera *et al.,* 2016). Shifts from Cerrado to cultivation have resulted in huge soil losses under erosive storms (12.4 t/ha/yr for bare soil compared to 0.1 t/ha/yr for Cerrado) (Oliveira *et al.,* 2015).The Paraná river basin suffered a 66% decrease in forest cover between 1977 and 2008, with a 3.5% annual rate of forest loss (Bianchi & Haig, 2013). A recent review (Hunke *et al.,* 2015) concluded that while conversion of Cerrado did not alter total soil nitrogen, nitrogen enrichment in agricultural catchments has increased, indicating fertilizer impacts and potential susceptibility to eutrophication; moreover, pesticides are consistently found throughout the entire aquatic system. Part of the loss of woody cover in the Cerrado is due to charcoal production (Ratter *et al.,* 1997). For example, 34.5% of around 5.5 milion tons of charcoal produced in the Brazil in 2005 still came from native Cerrado species in spite of efforts to transition to planted forests (Duboc *et al.,* 2007).

Like Cerrado, the Chaco has recently undergone extensive transformation (c.f., **Figure 3.19**). Rapid loss of chacoan dry forest has been documented in Bolivia, Paraguay and Argentina (Gasparri & Grau, 2009; Grau *et al.,* 2005; Zak *et al.,* 2004), mostly due to agriculture (mainly, soybean). For the Cordoba area in Argentina, Zak *et al.* (2004) estimated clearing of 1.2 million ha between 1969 and 1999. For North West Argentina between 1972 and 2007, another 1.4 million ha were removed (Gasparri & Grau, 2009). According to Fehlenberg *et al.* (2017), some 7.8 million ha out of a total of 110 million ha of dry Chaco in all countries was converted between 2000-2012, (principally to support soybean production and cattle ranching).

Conversion of vegetation has facilitated the spread of invasive species, like *Pyracantha angustifolia* (Rosaceae), which is now widely spread in the Chaco Serrano of Argentina (Tecco *et al.,* 2006). According to these authors, this species can potentially enhance the recruitment of forest species. However, a considerable number of other exotic woody species, and especially *Ligustrum*, are also favored by the presence of this exotic shrub (Tecco *et al.,* 2006).

3.4.1.7 Temperate grasslands

North American subregion

Status. Grasslands were once widespread in midwestern North America, occurring in a mosaic of tallgrass prairie and savanna (Nuzzo, 1986). Prior to European settlement, the central prairie of North America is thought to have ranged from southern Alberta, Saskatchewan and Manitoba south to mid-Texas, and from the foothills of the Rocky Mountains eastward into Indiana, Kentucky and Ohio and southwestern Ontario, covering about 2.4 million km^2 (The Nature Conservancy, 2009; USDA & USDOI, 2012). Diverse grasslands are major reservoirs for belowground carbon storage and prevention of soil loss due to erosion. Grasslands also serve as buffers increasing ecosystem nutrient uptake reducing runoff of agricultural waste and fertilizer into water bodies. Declines have been greatest in the mixed-and tall-grass prairie, with estimates of less than 5% (Sampson and Knopf, 1994) to 0.5% (The Nature Conservancy, 2009; USDA & USDOI, 2012) of the pre-European settlement tall-grass prairie remaining. Currently, approximately 50% of the Great Plains - about 148 million hectares in total - remains in grassland (i.e., not in annual crops or developed land) (WWF, 2017a).

Grassland vegetation structure is strongly influenced by fire frequency, driven by topographic barriers to the spread of fire (rivers, lakes, and bluffs), with oak savannas and prairies occurring on sites exposed to frequent fire (Peterson & Reich, 2008). Prior to modern settlement, fires annually burned large areas of the tallgrass prairie biome of North America (Gleason, 1913). Most prairie and savanna ecosystems were plowed under for agricultural uses or succeeded to forest following reductions in fire frequency. Prairie and savanna ecosystems are now exceedingly rare and mostly restricted to sites with infertile sandy soils that were unattractive for agricultural uses or where succession to woodland was slow following reductions in fire frequency (Nuzzo, 1986; Peterson & Reich, 2001; Will-Wolf & Stearns, 1999).

Fire suppression and agricultural land uses are important causes of habitat and biodiversity loss. For example, after conversion of all but 0.1% of tallgrass prairie in the USA state of Iowa, recent surveys found only 55% (491) of the original plant species formally known to be present there (Smith, 1998; Wilsey *et al.,* 2005). Fire suppression, exacerbated by fragmentation has caused a decline in small seeded and short stature species, as well as legumes, many of which are fire-dependent or require open areas (Leach & Givnish, 1996). In experimentally restored prairie/savanna systems, plant species richness and phylogenetic diversity are significantly higher in frequently burned grasslands than in unburned forests on the same soil conditions (Cavender-Bares & Reich, 2012; Peterson & Reich, 2008). Efforts to restore biodiversity and ecosystem services often fall short of the levels observed in remnant grasslands and other ecosystems (Benayas *et al.,* 2009; Martin *et al.,* 2005).

Bison were formerly dominant herbivores and a keystone species throughout the Great Plains (Knapp *et al.,* 1999). During the mid-1800s bison were reduced from tens of millions to only a few thousand individuals, subsequently recovering to more than 100,000 individuals. Bison grazing maintains grassland plant diversity by suppressing dominant warm-season grasses that would otherwise out-compete many rare wildflowers (Collins, 1998). Many populations of other animals dependent on prairie systems, including mammals and birds, have declined or are now absent from large portions of their historical range.

Recent trends. In the Great Plains region, 21.4 million ha of grassland have been converted to cropland since 2009. This loss represents almost 13% of the 170 million ha that remained intact (i.e., not in annual crops) in 2009. The average annual rate of loss of grasslands was 2% between 2009 and 2015. In 2016, only 148 million ha of grassland remained intact in the Great Plains (Northern Great Plains Program, 2016; WWF, 2016). A report based on data from the USA and Canadian governments, indicates that more than 21 million ha of land in the Great Plains have been converted to cropland since 2009. From 2014 to 2015 alone, approximately 1.5 million ha were lost. Endemic grassland bird species have shown steeper declines than any other group of North American bird species (USGS, 2003). Since the 1960s, populations of four key species have declined by as much as 80% with annual declines as follows: McCown's Longspur (*Rhynchophanes mccownii*), 6.2% decline per year; the chestnut-collared longspur (*Calcarius ornatus*), 4.4% decline per year; lark bunting (*Calamospiza melanocorys*) 4.1% decline per year and Sprague's pipit (*Anthus spragueii*), 3.5% decline per year. The decline of these grassland species has been attributed directly to the loss of intact grasslands throughout the Great Plains region (Northern Great Plains Program, 2016; WWF, 2016). Loss of prairie plant diversity (Leach & Givnish, 1996; Wilsey *et al.,* 2005) causes loss at higher trophic levels, including numerous insects and other organisms above- and belowground (Knops *et al.,* 1999; Lind *et al.,* 2015; Scherber *et al.,* 2010; Siemann *et al.,* 1998). Bees, important for pollination services, have declined; the rusty-patched Bumble Bee (*Bombus affinis*) which was declared an endangered species under the USA Endangered Species Act in 2017, once extended from the Dakotas and Nebraska, east across the Midwest and south to the Carolinas. Its population declined by 87% between 2011 and 2016. Other species that were once common in the Great Plains such as the western bumble bee (*Bombus occidentalis*) and the American bumble bee (*Bombus pensylvanicus*) are also declining severely (Northern Great Plains Program, 2016; WWF, 2016).

South American subregion

Status. This biome, as defined in the assessment, includes the Río de la Plata grasslands, Patagonian steppe and semi-desertic Monte vegetation, and thus includes many different vegetation types. Here, in our detailed analysis, we focus on the Río de la Plata grasslands, found principally in Argentina and Uruguay and extending into southern Brazil. These grasslands sustained grazing as of the 1600s. Fully 70% of the grasslands, formerly occupying an estimated ¾ of million square km, have been replaced (Salazar *et al.,* 2015) by crops, pastures or afforestations. In Argentina, about one in every three plant species growing in natural or semi-natural pampas is non-native. Although there are still very few natural parks protecting the Río de la Plata grasslands, some recent efforts on grassland conservation have been notorious (e.g. Sistema Nacional de Áreas Protegidas-Uruguay, Alianza del Pastizal).

Recent trends. Profound changes, affecting key ecosystem functions and ultimately human well-being, have occurred in South American temperate grasslands. Livestock grazing for over 400 years has reduced soil organic carbon stocks by an estimated 22% (a reduction of 1.5 picograms of carbon for the whole region) and net primary production by 24% (Piñeiro *et al.,* 2006). Cropping reduced soil organic carbon stocks by 20 to 30% in a few decades (Alvarez, 2001, 2005). Soil nutrients have been also depleted in croplands (near 30% of soil nitrogen and 80% of soil phosphorus) and rangelands (19% of soil nitrogen. Nutrient losses have triggered large increases in fertilizer use with beneficial effects for food production, but detrimental effects on air and water pollution (Portela *et al.,* 2006, 2009). Crop rotation with pasture in the past helped maintain elevated soil organic matter stocks and replenish nutrient losses (Morón & Sawchik, 2003). However, crop rotation was abandoned over the last 15 years due to soybean expansion (García-Préchac *et al.,* 2004). Nevertheless, more recently, new regulations for soil conservation have been successfully established in some countries of the region (e.g. Uruguay), with elevated adoption by farmers. Several parts of the region are experiencing decreases in light interception, and potentially their net primary production, with cascading effects on trophic networks. For example, large and consistent negative trends in net primary production have been observed in some parts of Uruguay and Argentina, associated with land use and climate change (Paruelo *et al.,* 2004).

Southern temperate grasslands have been strongly invaded by plants and animals. The grass, *Eragrostis plana* was accidentally introduced into southern Brazil from Africa in the late 1950s (Guido & Pillar, 2017). Later planted as a potentially promising forage grass, it has now invaded over 1 million ha of grasslands (Medeiros *et al.,* 2014). *Eragrostis plana* turned out to have low digestibility for cattle and causes economic losses by outcompeting more palatable native grasses. This is a very good example of how things can go wrong. Meanwhile, *Braquiaria* grasses (see also *Urochloa* spp.) are becoming adjusted to the local climate and could become a serious and widespread invasion problem in the future. The same grasses affect Uruguayan grasslands (Aber & Ferrari, 2010), so, without action, these invasions can be expected to expand in the coming years, encroaching on natural grassland areas. Exotic trees (e.g. *Gleditsia, Thriacanthos pines*) are also invading large areas of the region, altering grasslands physiognomy and displacing the local flora and fauna. Woody invasive species such as brooms (*Spartium junceum, Genista monspessulana* and *Ulex europaeus*), spiny rosaceous shrubs (*Rosa* spp. and *Rubus* spp.) and pines (*Pinus halepensis, P. radiata*) fit particularly well in a highly altered landscape matrix. Net forest cover in temperate grasslands increased from 2000 to 2012 **(Figure 3.19)**, but this increase is attributed mainly to exotic tree plantations (Salazar *et al.,* 2015).

Invasive vertebrates include wild boar (*Sus scrofa*). This species threatens key conservation habitats, affects agriculture and acts as a reservoir for diseases affecting pig farming, chital (*Axis axis*), and feral horses. European carp (*Cyprinus carpio*) has colonized most freshwater habitats, while common starlings (*Sturnus vulgaris*) and the red-bellied tree squirrel (*Callosciurus erythraeus*) are currently undergoing range expansion. Pet trade, forestry and aquaculture are emerging as new vectors of species introduction and expansion (see also Chapter 4). Other invasive animals include European pigeons, deer, and bullfrogs.

3.4.1.8 Drylands and deserts

North American and Mesoamerican subregions

Drylands are ranked as one of the most important biomes for the biodiversity of species and endemics both globally and in the Americas (Goudie & Seely, 2011; Le Saout *et al.,* 2013; Millennium Ecosystem Assessment, 2005). Much of the rich biodiversity and endemism **(Table 3.4)** found in these regions in the Americas and elsewhere is likely due to the high climate variability, which can drive speciation. High levels of endemism occur both at the species **(Table 3.4)** and generic levels. For example, 44% of seed plant genera in Mexican drylands under a broad definition are endemic (Challenger & Soberón, 2008). Animal biodiversity in North American can closely rival that found in tropical regions: Arizona alone contains 203 snake species (Southwestern Center for Herpetological Research, n.d.), almost two-thirds of the number found in the entire Amazon Basin. Unfortunately, many of these species have small home ranges, placing them at a high risk of extinction (Pimm *et al.,* 1988). Biodiversity of lichens and mosses in dryland biological soil crusts, critical to soil stability and fertility, often exceeds vascular plants (Belnap *et al.,* 2016).

Current habitat fragmentation, number of globally threatened animal species, and altered fire cycles in these drylands are rated moderate to very high (Hoekstra *et al.,* 2010). In Mexican drylands, fragmentation is greatest in coastal deserts (Arriaga, 2009). One fragmentation index indicates that the largest mean parcel size of intact habitat in North America is only about 4% of the total extent of the dryland ecoregion **(Figure 3.22)**. Nearly all drylands in North America and Mexico have been grazed by livestock relatively heavily at some point since European settlement; it is thus difficult to know how current ecosystems differ from those present before then. Estimates of departure of current vegetation conditions in the dryland biome relative to undisturbed dryland conditions based on the vegetation departure index are high in many areas of the biome, frequently more extreme than in agricultural or urban environments **(Figure 3.22)**.

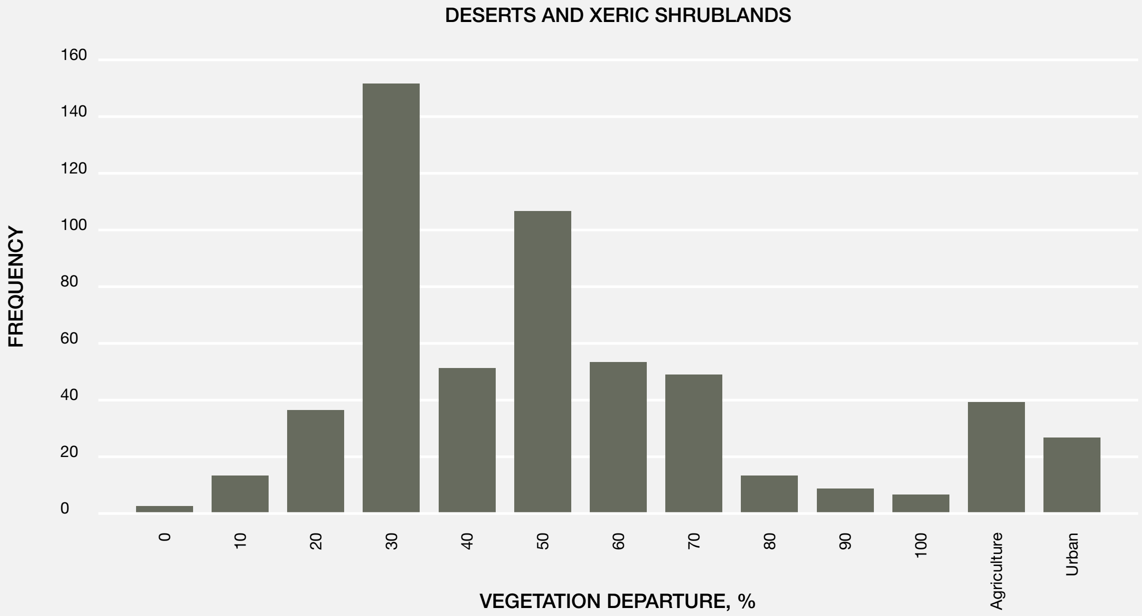
Dryland regions contain significant numbers of species that occupy habitats that have always had a very restricted range and thus are at high risk to disturbance. Reptile declines are associated with habitat loss. Individual desert tortoises occasionally move long distances between populations (Edwards *et al.,* 2004), but movement is increasingly difficult for tortoises due to habitat fragmentation. The main cause of a decline in the bunchgrass lizard (*Sceloporus scalaris*) in the Chiricahua Mountains in southeastern Arizona, USA, has been attributed to the loss of native bunchgrasses due to cattle grazing (Ballinger & Congdon, 1996). This lizard requires bunchgrasses for cover and protection from predators and harsh winter conditions.

Recent trends. Habitat loss between 2000 and 2009 is estimated at 15–60% in North America (Challenger & Dirzo, 2009; Hoekstra *et al.,* 2010). Biodiversity, soil health, and most associated ecosystem functions have declined over the past 50 years across these regions (e.g. http://www.biodiversitymapping.org; Goudie & Seely, 2011; Kéfi *et al.,* 2007; Sarukhán *et al.,* 2015). Biodiversity loss can be severe, as in the case of the highly specialized dune sagebrush lizard (*Sceloporus arenicolus*) of sandy depressions of dunes semi-stabilized by Shinnery oak (*Quercus havardii*) (Ryberg *et al.,* 2014). These dunes are currently experiencing a large amount of energy exploration and development, resulting in their mobilization and thus severe loss of lizards and their habitat. A 13-year study of the twin-spotted rattlesnake (*Crotalus pricei*) found that the age class structure has been skewed toward younger snakes, probably due to illegal collection of snakes for the pet trade (Prival & Schroff, 2012). Unique ecosystems like the Cuatro Ciénegas Basin in Coahuila, Mexico **(Box 3.7)** have experienced recent losses of microbial biodiversity found nowhere else on Earth.

Loss of sagebrush habitat in the western USA has also impacted biodiversity, including the sage-grouse. This bird was once widespread and common, inhabiting, at the time of European settlement, what was a relatively uninterrupted vast (~46,521 km^2) sea of sagebrush (*Artemisia tridentata tridentata*) (http://sagemap.wr.usgs.gov). Due to agricultural cropping, fire, grazing, and energy extraction, this bird now occupies about 1/10 of its original range (~4,787 km^2) and is believed to be in peril of extinction. Rehabilitation of the sagebrush habitat has proven very difficult especially with

Figure 3 22 **Percentage of departure between current vegetation conditions and reference vegetation conditions of dryland desert and xeric shrublands (aridity index < 0.05 extracted from 30 arc second (~1 km^2) resolution) and based on the VDEP index of the USA Forest Service and USA Geological Survey.**

Higher values indicate a greater departure from potential, or undisturbed vegetation. Agricultural and urban areas are grouped on the right for comparison. Source: Original data from The Nature Conservancy (2009) and The Nature Conservancy Terrestrial Ecosystems.



Box 3 7 The Cuatro Ciénegas Basin in Coahuila, Mexico.

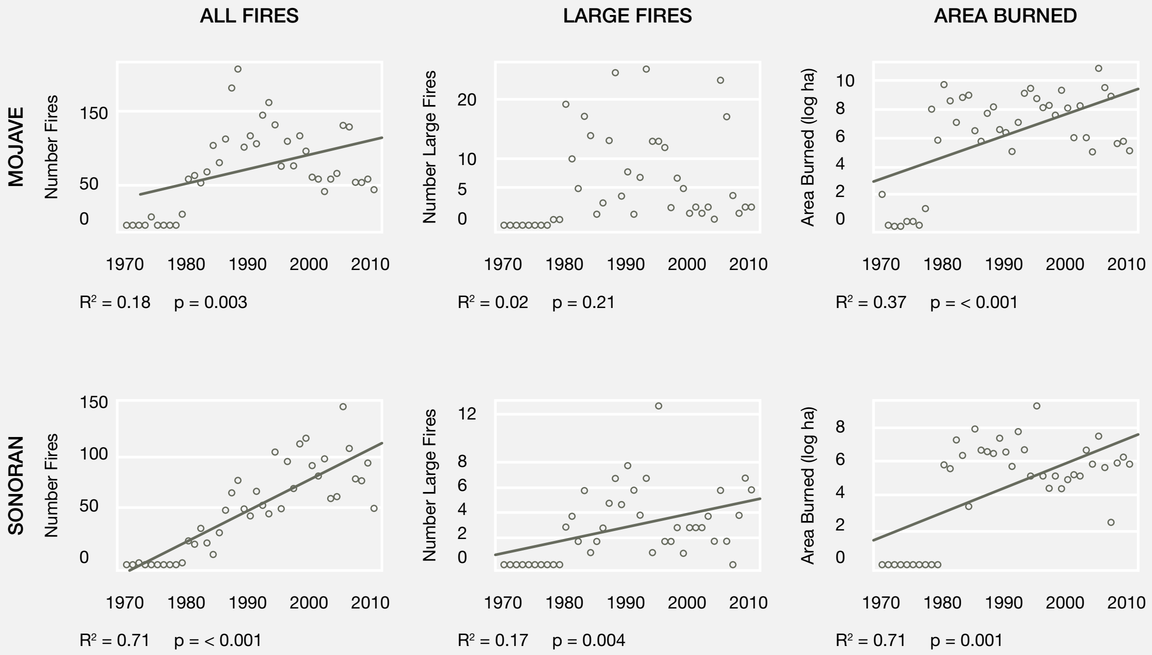
This ultra-low nutrient oasis in the Chihuahuan desert is extremely diverse, hosting at least 99 micro-endemic species and an equally wide array of microbial mats and stromatolites with ancestral marine lineages (CONABIO database, n.d.). The water's extremely low phosphorus content is characteristic of ancient ocean chemistry, earning it the nickname "Precambrian Park" (Redfield, 1934; Souza *et al.,* 2012). It exceeds diversity of other aquatic pools within desert systems globally by several orders of magnitude in the case of microbes and manyfold for other groups, such as spiders. Viral diversity is higher here than any other site in the world, reflecting the diversity of their bacterial and eukaryotic prey. The level of macrofauna endemism is equal to that of the Galápagos and is higher than anywhere in North or Mesoamerica (Stein *et al.,* 2000). Many species are new to science and still in the process of being described. The unusual geological history of this area explains its biodiversity: a large portion of the ancient Tethys Sea became entrapped by the regional uplift of the Sierra Madre Oriental and Occidental, isolating ancient seawater communities and leaving them to evolve independently (Ferrusquía-Villafranca *et al.,* 2005; Souza *et al.,* 2006, 2008, 2012). Due to intensive agriculture, 90% of Churince, the most widely studied part of the basin, has disappeared since 2006, with most of the loss occurring in 2017; the remaining 10% is unique since most of the species are microendemic to the basin and their unique site. The whole Cuatro Ciénegas Basin is threatened by the intensified use of the deep aquifer for agriculture, causing water to be drained at a very rapid pace.

the invasion of exotic annual Mediterranean grass *Bromus* (Germino *et al.,* 2016) which accelerates fire cycles, leading to further loss of sagebrush on a large scale (Germino *et al.,* 2016).

Exotic plants have increased in North American drylands due to several causes, but especially increased fire and soil surface disturbances; this invasion negatively impacts plant and animal communities (Brooks, 2009). Fire frequency and area burned increased in the Californian portions of the Mojave and Sonoran deserts between 1970 and 2010 **(Figure 3.23)**. Exotic grasses, which burn easier than other vegetation types, were an important explanatory variable for large fires in the Mojave, but the amount of native perennial vegetation was more important in the Sonora (Syphard *et al.,* 2017).

Figure 3 23 **Recent trends in fire frequency and area burned in the Mojave (upper) and Sonoran (lower) deserts based on data for southern California.**

Left: all fires; middle: large fires. Source: Modified from Syphard *et al.* (2017).



In the Sonoran desert, biological soil crusts have shown a dramatic decline in cover over the past 50 years, as they are highly vulnerable to fire and the disturbance of soil surfaces (Belnap & Eldridge, 2003). The loss of native plants, animals, and biological soil crusts has led to increased soil erosion via wind and water erosion; decreased soil albedo over large regions; and had a strong negative impact on water, carbon, and nutrient cycles (Ahlström *et al.,* 2015; Fields *et al.,* 2009; Hoekstra *et al.,* 2010; Painter *et al.,* 2010; Neff *et al.,* 2005).

South American subregion

Status. Notwithstanding increasing intensive agriculture and urban encroachment, large parts of the Atacama and Sechura deserts in western South America remain fairly intact (Luebert & Pliscoff, 2006). The western deserts, although in large part seemingly barren, are an area of unexpected richness, especially in plants and microorganisms. Plant species-richness and endemism are especially high in the narrow coastal loma vegetation band (Dillon *et al.,* 2011; Rundel *et al.,* 1991; Squeo *et al.,* 1998) **(Table 3.4)**. Cactaceae are important and highly threatened (Goettsch *et al.,* 2015; Guerrero *et al.,* 2011; Larridon *et al.,* 2014; Ortega-Baes & Godínez-Alvarez, 2006). Saline lakes and barren areas of the Atacama contain fascinating assemblages of Archaea, bacteria, and cyanobacteria (Crits-Christoph *et al.,* 2016; Fernandez *et al.,* 2016; Lester *et al.,* 2007; Navarro-González *et al.,* 2003; Wierzchos *et al.,* 2006). Western deserts are subject to flash floods, and thus vegetation integrity plays a critical role in containing water erosion. Western coastal desert loma vegetation in particular, is highly susceptible to invasion when disturbed (Aponte & Cano, 2013). While some areas of the western deserts are under threat, a growing appreciation of the rich so-called "flowering desert" in Chile as a tourist resource has greatly heightened public awareness of the value of biodiversity.

Caatinga vegetation in eastern Brazil, part of this biome, is also rich **(Table 3.4)**. Caatinga is poorly known in comparison to Brazilian Cerrado and tropical rainforest. The caatinga woody matrix is estimated to comprise around 63% of the original cover (Beuchle *et al.,* 2015, but see Schulz *et al.,* 2017) and thus is better conserved than Cerrado. Ten mammals are strictly endemic to caatinga and 11 more are endemic to the caatinga and Cerrado (Gutiérrez & Marinho-Filho, 2017). While most alien plant species in western drylands were accidentally introduced, the Caatinga is home to many intentionally introduced tropical forage grass species (Almeida *et al.,* 2015).

Recent trends. Urban encroachment into the western loma vegetation has affected a highly endemic, range-restricted rodent to the point of likely global extinction of (Mena *et al.,* 2007) warning that other local endemics in loma could be at risk with coastal development. A recent wave of private coastal development in Chile has greatly reduced the habitat of a globally threatened plant species (García-Guzman *et al.,* 2012) and other species are likely affected. The production and the illegal extraction of Cactaceae continues (Estevez *et al.,* 2010; Larridon *et al.,* 2015). Extensive vegetation dieback, accompanied by declining guanaco populations, has been reported repeatedly over the last 20 years in the arid-most part of the western coastal desert (Schulz *et al.,* 2011 and references therein). This trend coincides with a tendency for reduced precipitation, extended drought periods and reduced coastal cloud, notwithstanding typical El Niño variation. On the transition to Mediterranean shrublands, continuous monitoring has revealed El Niño Southern Oscillation-related fluctuations in the abundances in small mammals (Armas *et al.,* 2016; Meserve *et al.,* 2011) and alien plant species, but with significant recovery of native plants in wetter years (Jiménez *et al.* 2011), indicating high resilience at least in less arid areas. A noticeable shift in small mammals followed the last major El Niño Southern Oscillation event in 2000-2002 with their numbers becoming less fluctuating. This appears to have been caused by a shift in rainfall periodicity from strong interannual fluctuations, to a more equitable pattern with more consistent annual rainfall. These trends may be indicative of ongoing climate change in the Chilean semiarid region (Armas *et al.,* 2016).

Biome-scale studies agree that the Caatinga has seen recent large-scale vegetation turnover and cover changes. However, both increases and decreases in woodland and woody vegetation have been reported. While studies based on Moderate Resolution Imaging Spectroradiometer data tend to find a net gain of woody vegetation, those based on Landsat data find a net decrease (discussed in Schulz *et al.,* 2017). The impacts on caatinga species and populations of this highly dynamic scenario, to which a fertilization effect of carbon dioxide might be relevant (Donohue *et al.,* 2013), are largely unknown. In addition to many introduced forage grasses, a serious ongoing invasion in caatinga concerns *Prosopis juliflora* which forms monospecific stands that outcompete native woody species and now covers over one million hectares (Gonçalves *et al.,* 2015). As in the western deserts, selective biomass removal for fuel continues in the Caatinga, even though many households now possess gas stoves (Cavalcanti *et al.,* 2015; Ramos & Albuquerque, 2012).

3.4.1.9 Wetlands: peatlands, mires, bogs

North American subregion

Status. North America houses approximately 240 millions of hectares of wetlands comprising 12.6% of the total land area. Some of the largest North American wetland landscapes are the peatlands of the Hudson Bay Lowlands, the peatlands of the Mackenzie River Watershed (Vitt, 2016), the Prairie Pothole region of the glaciated midcontinent of Canada and the USA, covering 7.7 million ha, and The Everglades and Great Cypress Swamp, covering 1 million hectares located on the southern part of the Florida peninsula. The boreal peatlands of Canada (110 million hecares), store an estimated one-third of the world's global carbon and 10% of the world's soil nitrogen (Vitt, 2016). The cold anaerobic conditions of boreal peatlands favor the accretion of undecomposed mosses, sedges, and other plants, together, resulting in deep organic deposits of 2m or more. Canadian peatlands support exceptional bryophyte diversity, with a recorded 294 species of mosses and related species (about one-third of the world's moss species) (Junk *et al.,* 2006). The Prairie Potholes and the Everglades have outstanding biodiversity **(Table 3.4)**. The Prairie Potholes provide critical breeding and migratory habitat for North America's waterfowl. The Everglades serve as a wintering area for 249 migratory bird species, as well as 100 resident species and critical habitat for species of global conservation concern.

From the 1800s to the 1980s Canada sustained losses of about 20 million hectares of wetland habitat. The conterminous USA sustained wetland losses of 53% (117 million hectares) from the 1780s to 1980s; Alaska lost less than 1% (Dahl, 1990). Despite these losses and much regional variation, wetlands still cover 12% of North America (240 million ha) (Dahl, 1990, 2011; Federal Provincial and Territorial Governments of Canada, 2010).

Recent trends. Losses to drainage for agriculture over the past 40–60 years has been the most important cause of wetland loss; conversion for development has also been locally significant near urban centers. An estimated 350,000 ha of wetland habitat in Canada was lost over the past 40–60 years, to drainage for agriculture in the prairie pothole region (Government of Canada, 2009). Wetland losses in Greenland are presumed to have been negligible. Compared to historic rates of wetland conversion, loss rates in both the USA and Canada have likely been lower in recent years because federal policies create disincentives for filling and draining wetlands (i.e., US Clean Water Act of 1972), Canada's Federal Policy on Wetland Conservation of 1991 (Government of Canada, 1991). Unfortunately, policies that allow compensatory restoration to offset conversions have not been effective at preventing losses of forested wetlands, which are costly and difficult to restore (Dahl, 2011).

Many North American wetlands have undergone extensive eutrophication. The associated changes of eutrophication include changes in the composition of aquatic life and recreational uses, in the effectiveness of wetlands as effective filters that protect downstream and groundwater resources, and in accumulation rates of bulk sediments (Brenner *et al.,* 2001).

Wetland alteration has favored the expansion of invasive species and displacement of native species. Some serious wetland invaders in North America include common reed (*Phragmites australis*) in freshwater and brackish wetlands, cordgrass (*Spartina alterniflora*) in West Coast salt marshes and hybrid cattail (*Typha x glauca*), reed canary grass (*Phalaris arundinacea*) and purple loosestrife (*Lythrum salicaria*) in freshwater marshes. These invaders diminish wetland services in many ways including lost critical habitat for endangered species (e.g. *Phragmites*, central Platte River) and reduced wetland bird nesting (*Typha*, Great Lakes). Although a lot of attention and much funding have been devoted to managing and controlling these species, their spread is generally irreversible. Proximity to urban areas, as in the Everglades, has been associated with the escape and establishment of a large number of ornamental plants and pet animal species, including 221 plants, 32 fish, 30 amphibians and reptiles, and 10 mammals in the Everglades (Brown *et al.,* 2006).

Many wetlands in urban areas that have been modified by filling or dredging experience high pulses of stormwater from watersheds with diminished infiltration, and receive toxins from transportation (e.g. chloride from road de-icing salts) and industrial run-off (Brinson & Malvárez, 2002; Sanzo & Hecnar, 2006; Federal Provincial Territorial Governments of Canada, 2010). Simultaneously the combination of alterations from urban development and agriculture has caused radical changes to the water quality and water flow in places like the Everglades.

Mesoamerican subregion

Status. The Mesoamerican subregion possesses an outstandingly diverse contingent of large tropical wetland areas with abundant bird, fish and large mammals **(Table 3.4)**, among them the Centla Swamplands Biosphere Reserve south of the Gulf of Mexico, the Los Guatusos wetland area on the southern coast of Lake Nicaragua, and Palo Verde National Park and the Northeast Caribbean Wetland (Tortuguero) in Costa Rica (Hernández, 1999), all together summing to 141,470 km^2. The Centla Swamplands, located at sea level, constitute one of the world´s largest swamp areas. They are the home of gallery vegetation, mangroves, aquatic plants, manatees, jaguars, crocodiles, turtles and many fishes and birds. The Guatusos Wildlife Refuge, with many fish species, is inhabited by indigenous and mestizo peoples. Like wetlands in general, it is a very important area for migratory birds, in the dry season in Nicaragua. Palo Verde National Park includes deltas, estuaries, flood plains, swamp forests and seasonally flooded grasslands. Counts of more than 50,000 waterfowl have been made in the wetlands of Palo Verde National Park, including the endangered Jabiru stork (*Jabiru mycteria*) (Daniels & Cumming, 2008). Tortuguero is dominated by herbaceous swamps and wooded palm-dominated floodplains that run parallel to the coast. It is an important site for nesting green turtles and several threatened species. In general palm-dominated wetlands in Costa Rica and Nicaragua constitute 16-22% of all wetlands (Serrano-Sandi *et al.,* 2013); this type of wetland tends to be relatively poor in birds (Beneyto *et al.,* 2013) as well as reptiles and amphibians (Bonilla-Murillo *et al.,* 2013). An estimated 35% of Mexican wetlands have been transformed or suffered some level of deterioration (Hernández, 1999).

Recent trends. Contamination from heavy metals has been reported in the Centla swamplands (Pérez-Cruz *et al.,* 2013), while pesticides related to agriculture have been reported in the Palo Verde Wetlands (Mena-Torres *et al.,* 2014). Tabasco, where the Centla wetlands are located, is an area where petroleum extraction is currently occurring and is a threat to the reserve. The effect of these contaminants on aquatic biodiversity, however, is still unclear as baseline studies are only beginning. At the same time, the probability of wetland conversion increases as areas of wetlands are found closer to already converted land (Daniels & Cumming, 2008). Nevertheless, Landsat maps of Normalized Difference Vegetation Index suggest that the Palo Verde wetland has witnessed an overall increase in vegetation greenness and cover since 1986, matching the abandonment of cattle ranching and the known degradation of the wetland by cattail invasion (Alonso *et al.,* 2016). This study shows that large degraded tropical Mesoamerican wetlands have the capacity to recuperate when external pressures are removed. The Tortuguero wetland is threatened by subsistence, sports fishing, poaching, the illegal collection of turtle eggs (Hernández, 1999) and pesticides from banana plantations and packing plants (Castillo *et al.,* 2000). All these changes impact on human well-being. For example, total shrimp catch in El Salvador and Panamanian wetlands has dropped by 50% in the past decade or so (Hernández, 1999).

South American subregion

Status. South American wetlands are hugely diverse, spread over the entire continent, and found from sea level to above 5000 m altitude. The three largest wetland areas (Amazon river basin, Pantanal, Magellanic peatlands) in accordance with their sizes (Keddy *et al.,* 2009), comprise around 11% of South America. Other large wetlands are the Orinoco delta with large peatlands and the internal Venezuelan and Colombian deltas, which are pantanal-like areas. Total wetland extent is difficult to pin down, given lack of consensus over what constitutes a wetland and the fact that some wetlands tend to be overlooked. A case in point are the Veredas of Brazil, spread over the entire savanna biome and perhaps comprising as much 5% of that biome. Many wetland types are rich in bird species (Caziani *et al.,* 2001; Derlindati *et al.,* 2014; Mascitti & Bonaventura, 2002; Tellería *et al.,* 2006) **(Table 3.4)** and have important aesthetic value. Amazonian flooded forested wetlands and the Pantanal are especially rich in plants, birds, fishes, reptiles, and amphibians **(Table 3.4)**. Some wetlands are rich in planktonic assemblages (Küppers *et al.,* 2016; Muñoz-Pedreros *et al.,* 2015). In general, South American wetlands play a vital role in water regulation for surrounding forests and agricultural lands. For example, certain types of subantarctic peat bogs and mires, given the high water-holding capacity of *Sphagnum* species and of accumulated peat, discharge water slowly and have an important buffering effect on surrounding forest ecosystems (Iturraspe, 2010). Southern South American peatlands, including Amazonian and (and probably Orinocan peatlands), are important carbon sinks (Lähteenoja *et al.,* 2009; Loisel & Yu, 2013). In contrast, tropical floodplain lake ecosystems with a large amount of organic matter are considered important sources of carbon from the water to the atmosphere (carbon dioxide evasion) (Raymond *et al.,* 2013), although abundant macrophytes can counteract this effect locally (Peixoto *et al.,* 2016), indicating an important biodiversity link. High Andean bogs in the arid puna are important for the grazing of native camelid and other domestic animals which sustains the livelihoods of high altitude peoples (Borgnia *et al.,* 2008; Tirado *et al.,* 2016) (see also **Box 3.6**).

Recent trends. Many South American wetlands have been severely degraded over the past 30–40 years. High Andean bogs and associated salt lakes in arid areas of the Andes are now used extensively as a water source for mining (Aitken *et al.,* 2016), although this source is now being replaced by imported seawater from adjacent coastal areas in some cases. Roads built over these fragile ecosystems are an additional problem (Salvador *et al.,* 2014). Water levels in the bog complex in the arid Andes are critical for bird species maintenance (Tellería *et al.,* 2006). Water withdrawal also alters a key habitat for grazing animals. In the vast Pantanal, largely as a result of large-scale cattle ranching and cropping, between 1976 and 2008 loss of floodplain vegetation increased over 20-fold (Silva *et al.,* 2011) with some 12% lost. Loss of pantanal has lead to negative consequences for large animal species (Keuroghlian *et al.,* 2015). Nevertheless, absolute loss of pantanal is much lower than for Cerrado. In the lower Paraná delta in Argentina, between 1999 and 2013 one-third of the freshwater marshes (163,000 ha) were replaced by cattle pastures (70%) and forestry (18%) (Sica *et al.,* 2016) over a period of no more than 14 years. As of the 1970s, intensive commercial fisheries developed across the Amazon and the lower Paraná delta. Overexploitation of frugivorous fish species has depressed the quantity, quality, and diversity of seeds dispersed by fishes which could lead to overall reduced plant diversity in these habitats (Correa *et al.,* 2015). Long-term studies (1969–1987) in the extra-tropical Mar Chiquita in Argentina reveal that flamenco breeding is very susceptible to lake water levels, especially excessive flooding (Bucher *et al.,* 2000), making this kind of wetland vulnerable to surrounding land use changes affecting upstream flow. While commercial peat extraction is still limited in Magellanic peat bogs, abandoned peatlands show a significant invasive plant species component (Domínguez *et al.,* 2012). The integrity of southern peat bogs is further threatened by introduced beaver, which increased in number from 54,000 to 110,000 between 1999 and 2015 (Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas, 2016).

3.4.1.10 Summary biodiversity data for terrestrial biomes and overall trends for terrestrial biomes and other units of analysis

Table ③ ④ **Illustrative biodiversity data for principal terrestrial biomes in subregions of the Americas. The first number in parentheses gives richness; % value is endemism level where available.**

NORTH AMERICAN SUBREGION
Temperate and boreal forests and woodlands
USA forests: plants (9,195), mammals (234), birds (452), reptiles (218), amphibians (201), freshwater fishes (60), invertebrates (739), trees (~1,000) (U.S. Forestry Service, 2015).
Mediterranean forests, woodlands and scrub
California Floristic Province: plants (5,006; 37%) (Burge *et al.,* 2016); California: mammals (201), birds (653; 1%), reptiles (101;15%), amphibians (70; 46%) (Zavaleta *et al.,* 2016), bees (1,600) (Frankie *et al.,* 2014).
Tundra and high mountain habitats
North American tundra: vascular plants (1,486) (Elven *et al.,* 2011), mammals (41), birds (152), amphibians (1), insects (1,567), spiders (200), springtails (174), mites (368), white worms (73) (Meltofte, 2013); Western North American alpine: plants (> 1,400) (Malanson *et al.,* 2015).
Temperate grasslands
Midwestern grasslands: plants (897) (Wilsey *et al.,* 2005).
Drylands and deserts
Mojave Desert of southern California: plants (5,000) (USDA n.d).
Wetlands, peatlands and mires
Canadian peatlands: mosses and related species (294) (Junk *et al.,* 2006); Everglades: plants (1,033), birds (349, 249 migratory), fishes (432), reptiles (60), mammals (76), amphibians (38) (Brown *et al.,* 2006), macroinvertebrates: 290¬–400 (Trexler & Loftus, 2016).

MESOAMERICAN SUBREGION

Tropical and subtropical moist forests

Mexico lowland tropical broadleaf forest: seed plants (~5,000) (Challenger & Soberón, 2008); Mexican montane mesophyll forest: seed plants (~3000) (Challenger & Soberón, 2008); Mexican coniferous forest: pines (54), oaks (160) (CONABIO, 2014); Eastern Panama broadleaf forest: mammals (~165) (Voss & Emmons, 1996); Southern Mexico: mammals (~125) (Voss & Emmons, 1996).

Tropical and subtropical dry forests

Costa Rica: plants (~4,500), vertebrates (~1,100), arthropods (~150,000), fungi (~20,000) (Janzen, 1987; Janzen & Hallwachs, 2016); Mexico: seed plants (~6,000; 40%) (Challenger & Soberón, 2008), trees (1,072) (Banda-R *et al.,* 2016); Central America (northern South America included): trees (808) (Banda-R *et al.,* 2016).

Drylands and deserts

Mexico: seed plants (~6,000) (Challenger and Soberón 2008), endemic plants (3,600) (Arredondo Moreno & Huber-Sannwald, 2011), cacti (550 spp.; 78%) (Goettsch & Hernández, 2006).

Wetlands – peatlands, mires, bogs

Mexico, Centla Swampland: birds (213) (Santiago-Alarcon *et al.,* 2011), fishes (44) (Macossay-Cortez *et al.,* 2011); Nicaragua, Guatusos Wildlife Refuge: mammals (32), birds (>300), reptiles (10) (Hernández, 1999).

CARIBBEAN SUBREGION

Tropical and subtropical moist forests

Caribbean Islands (all terrestrial ecosystems): plants (11,000; 72%), mammals (69¬; 74%), birds (564; 26%), reptiles (520; 95%), amphibians (189; 100%), freshwater fishes (167; 39%) (Wege *et al.,* 2010).

Tropical and subtropical dry forests

Woody plants (611) (Banda-R *et al.,* 2016).

SOUTH AMERICAN SUBREGION

Tropical and subtropical moist forests

Amazonia: plant species (14,003), trees (6,727) (Cardoso *et al.,* 2017), trees (11,676) (ter Steege *et al.,* 2016), birds (1,300) (Marini & Garcia, 2005), reptiles (378), amphibians (428), fishes (>3,000) (Charity *et al.,* 2016); Amazonian lowland forest: mammals (434) (Mares, 1992); Atlantic Coastal forest: plants (~20,000), mammals (263), reptiles (306), amphibians (475) (Mittermeier *et al.,* 2005), birds (1,020) (Marini & Garcia, 2005); Andean Montane forest: trees (3,750) (Tejedor Garavito *et al.,* 2015), birds (many of a total of 1,160 species in all neotropical wet montane forests) (Stotz *et al.,* 1996), mammals (332) (Mares, 1992); Las Yungas, Bolivia: plants (6,073) (Jørgensen *et al.,* 2015).

Tropical and subtropical dry forests

Northern South America and Central America: tree species (808) (Banda-R *et al.,* 2016); Northern interandean Valleys: trees (418) (Banda-R *et al.,* 2016); Colombian dry forest: plants (2,569), birds (230), mammals (60) (Gómez *et al.,* 2016).

Temperate and boreal forests and woodlands

Temperate rainforests: plants (443–500) (Arroyo *et al.,* 1996; Villagrán & Hinojosa, 1997), mammals (58), birds (60) (Armesto *et al.,* 1996); Magellanic rainforest–tundra zone: bryophytes (450), liverworts (368) (Rozzi *et al.,* 2008); Tierra del Fuego and Patagonia: myxomycetes (67) (Wrigley *et al.,* 2010).

Mediterranean forests, woodlands and scrub

Central Chile: vascular plants (2,900; 30%) (Arroyo *et al.,* 2002), mammals (37), birds (200), reptiles (38), amphibians (12) (Simonetti, 1999), bees (~300) (Montalva & Ruz, 2010).

Tundra and high mountain habitats

Whole biome: plants (6,700) (Arroyo & Cavieres, 2013); Páramo: vascular plants (3,600) (Sklenář *et al.,* 2005), non-vascular plants (1,300) (Luteyn, 1999); Puna freshwater and salt lakes: fishes (60) (Vila *et al.,* 2007).

Tropical and subtropical savannas and grasslands

Brazilian Cerrado: plants (13,137), birds (837) (Overbeck *et al.,* 2015), mammals (251) (Paglia *et al.,* 2012), trees (2,916) ("Tree flora of the Neotropical Region," n.d.).

Temperate grasslands

Río de la Plata grasslands: grass species (550) (Bilenca & Miñarro, 2004).

Drylands and deserts

Chilean winter rainfall deserts (broadly): plants (1,893) (Arroyo & Cavieres, 1997); Pacific Coastal Lomas: plants (1,200) (Dillon *et al.,* 2011); Caatinga: plants (2,400-4,230) (Moro *et al.,* 2014), fishes (185), lizards (44), amphibians (8), snakes (47), turtles (4), crocodilians (3), amphibians (49) (WWF, 2017b), birds (519) (Silva *et al.,* 2003), mammals (148) (Oliveira, 2003).

Wetlands: peatlands, mires, bogs

Amazonian wetlands: plants (>1,390), endemic trees (68) (Junk *et al.,* 2014); Brazilian Pantanal: plants (1,863), aquatic and terrestrial mammals (170), bats (46-floodplain), birds (655 floodplain and uplands), herpetofauna (135 Plains), fishes (263) (Alho, 2011; Alho, *et al.,* 2011a; Alho *et al.,* 2011b; Pott *et al.,* 2011).

Figure ③ ㉔ **Historical and recent habitat change and recent species trends for terrestrial biomes and other units of analysis considered in the assessment for the four subregions of the Americas. Source: own representation.**

		HISTORICAL TRENDS	RECENT TRENDS (40 YRS)				
	Units of analysis	Habitat amount	Habitat amount	Habitat degradation	Native species diversity	Threatened species	Alien & Invasive species
NORTH AMERICA	Temperate and boreal forests and woodlands	↔••	↗••••	↗••••	↘•••	↔••••	↗•••
	Mediterranean forests, woodlands and scrub	↓••••	↘••••	↗••••	↘••••	↗••••	↗••••
	Tundra and high mountain habitats	↔••	↔••	↗••••	↔••••	↔••••	↗••
	Temperate grasslands	↓••••	↘••••	↗••••	↘••••	↗••••	↗••••
	Drylands and deserts	↓••••	↘••••	↗••••	↘••••	↗••••	↗••••
	Wetlands - peatlands, mires, bogs	↘••••	↘••••	↗••••	↘•	↗•	↗••••
	Inland surface waters and water bodies / freshwater	↘••••	↘•	↗••••	↘•••	↗•••	↗••••
	Coastal habitats and nearshore marine	↓••••	↘••••	↗••••	↘•••	↗••	↗••••
	Cryosphere / Sea Ice	↔•	↘••	↗•	↔••	↔••	↗••
MESOAMERICA	Tropical and subtropical moist forests	↓••••	↘••••	↗••	↔••	↗••	↔••
	Tropical and subtropical dry forests	↓••••	↘••••	↗•••	↘••	↗••	↗••
	Drylands and deserts	↘••	↘••	↗•	↘•	↗••	↗••
	Wetlands - peatlands, mires, bogs	↘••	↘•••	↗•	↘•	↗••	↗••
	Inland surface waters and water bodies/freshwater	↘••	↘•••	↗•	↘••	↗••	↗•
	Coastal habitats and nearshore marine	↘••	↘•••	↗••	↘••	↗•	↔•
	Marine/deepwater/ offshore systems	↘••	↔•	↗•	↔•	↗•	↔•
CARIBBEAN	Tropical and subtropical moist forests	↘••••	↗•••	↔•••	↔•••	↔••	↗••
	Tropical and subtropical dry forests	↓••••	↗••	↗•••	↔••	↗••	↗••
	Inland surface waters and water bodies/freshwater	↔•	↘••	↗••	↘••	↗•	↗•
	Coastal habitats and nearshore marine	↔••	↓••••	↑••••	↔•••	↗••	↗•••
	Marine/deepwater/ offshore systems	↔••	↔•••	↗•••	↔•••	↗••	↔•
SOUTH AMERICA	Tropical and subtropical moist forests	↘••••	↘•••	↗•••	↘•••	↗•••	↗•••
	Tropical and subtropical dry forests	↓••••	↘•••	↗•••	↘••	↗•••	↗•••
	Temperate and boreal forests and woodlands	↘•••	↘•••	↗•••	↘••	↗••	↗•••
	Mediterranean forests, woodlands and scrub	↓••••	↘••••	↗•••	↘•••	↗••••	↗••••
	Tundra and high mountain habitats	↘•••1 ↔•••2	↘••	↗•••	↔••	↔••	↔•••
	Tropical and subtropical savannas and grasslands	↘••••	↘••••	↗••	↘••	↗•••	↗•••
	Temperate grasslands	↓••••3 ↔••••4	↔••••	↗••••	↔••••	↔••••	↗••••
	Drylands and deserts	↔•••5 ↘6••••	↔••	↗•••	↔••	↗•••	↗•••
	Wetlands - peatlands, mires, bogs	↔••	↘•••	↗•••	↘•	↘•	↗••
	Inland surface waters and water bodies / freshwater	↘••	↘••	↗••	↔••	↗••	↗••

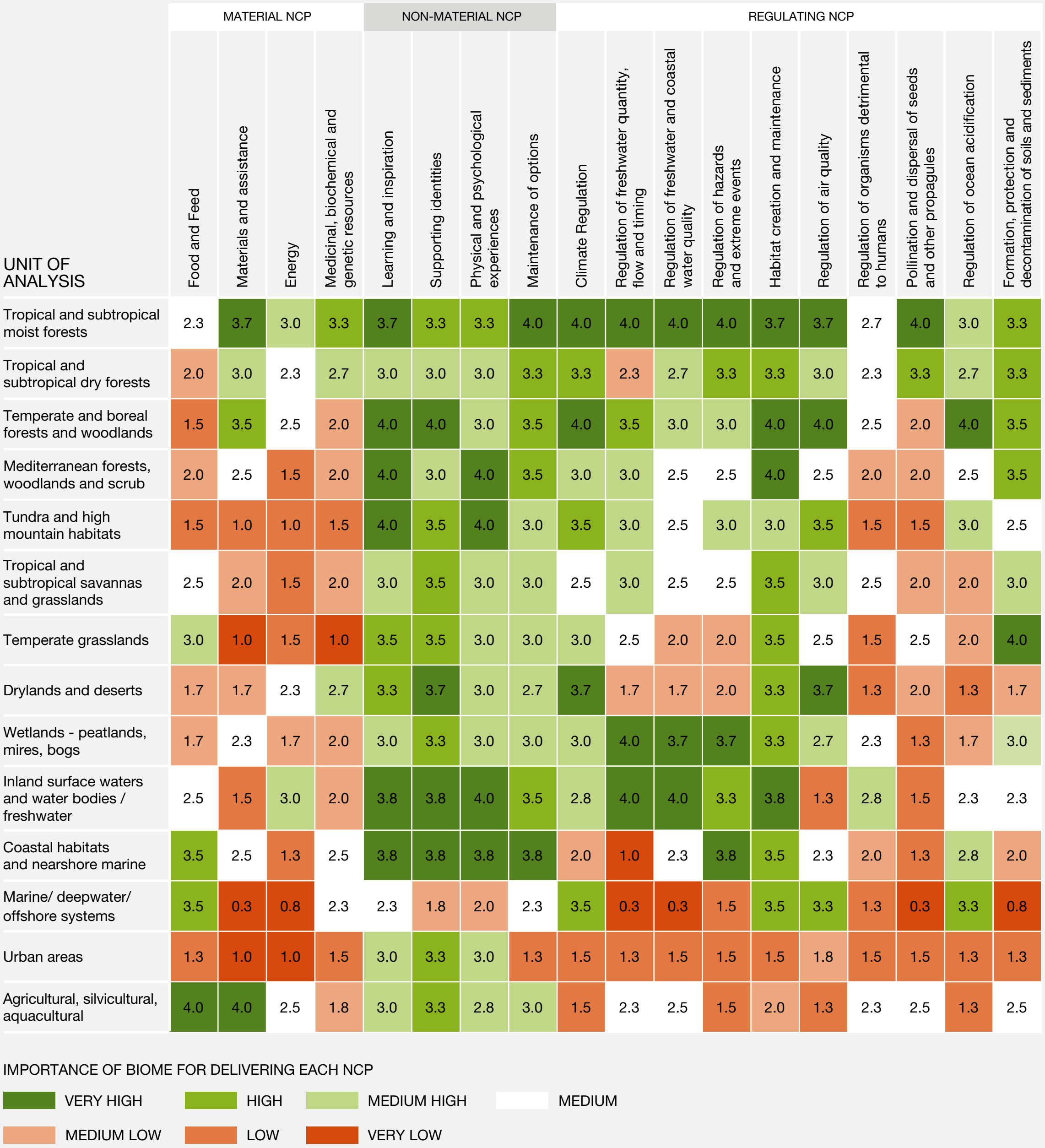
1: Páramo and puna; 2: Other areas of biome; 3: Río La Plata Grasslands; 4: Other areas of biome; 5: Western deserts; 6: Caatinga.

Figure 3 24

The historical habitat column indicates the proportion of intact habitat that remains compared with pre-European settlement (ca. 1600–1970): down arrow = around 50–100% decrease in spatial extent; diagonal down arrow = 10–50% decrease; diagonal up arrow = 10–50% increase; horizontal arrow = no or limited change (0–±10%). For recent trends, the columns give the general trends over the past 40 years, approximately 1970 to the present: diagonal up arrow = increased; diagonal down arrow = decreased; horizontal arrow = essentially no or very little change. Confidence levels: …. = well established; … = established but incomplete; .. = unresolved; . = speculative. The tendencies for threatened species are inferred tendencies based on the degree of habitat loss as well as formal assessment data where it exists. Trends were assigned by experts from each subregion following a modified Delphi process and in accordance with the literature reviewed for the assessment. Note: Coastal habitats and nearshore marine and Marine/ deepwater/ offshore systems were not considered for the South American subregion.

Figure 3 25 **Importance of each biome (unit of analysis) to Nature's Contributions to People (NCP: material, non-material and regulating) as defined by IPBES.**

Values are averaged across the four subregions for each unit of analysis, with values from each subregion equally weighted. Values between 0 (lowest) and 4 (highest) were assigned by panels of experts from each subregion and in accordance with the literature reviewed in the assessment. IPBES definitions of NCPs were used for all units of analysis in all subregions. Green colors indicate high importance of the biome/unit of analysis to the NCP. Red and orange colors indicate low importance. Values were assigned based on the proportions of the biomes/unit of analysis that have not been converted by humans to other land types. Values were assigned by experts from each subregion following a modified Delphi process. (Note: the cryosphere is not considered in this analysis). Source: own representation.



	MATERIAL NCP				NON-MATERIAL NCP				REGULATING NCP									
UNIT OF ANALYSIS	Food and Feed	Materials and assistance	Energy	Medicinal, biochemical and genetic resources	Learning and inspiration	Supporting identities	Physical and psychological experiences	Maintenance of options	Climate Regulation	Regulation of freshwater quantity, flow and timing	Regulation of freshwater and coastal water quality	Regulation of hazards and extreme events	Habitat creation and maintenance	Regulation of air quality	Regulation of organisms detrimental to humans	Pollination and dispersal of seeds and other propagules	Regulation of ocean acidification	Formation, protection and decontamination of soils and sediments
Tropical and subtropical moist forests	2.3	3.7	3.0	3.3	3.7	3.3	3.3	4.0	4.0	4.0	4.0	4.0	3.7	3.7	2.7	4.0	3.0	3.3
Tropical and subtropical dry forests	2.0	3.0	2.3	2.7	3.0	3.0	3.0	3.3	3.3	2.3	2.7	3.3	3.3	3.0	2.3	3.3	2.7	3.3
Temperate and boreal forests and woodlands	1.5	3.5	2.5	2.0	4.0	4.0	3.0	3.5	4.0	3.5	3.0	3.0	4.0	4.0	2.5	2.0	4.0	3.5
Mediterranean forests, woodlands and scrub	2.0	2.5	1.5	2.0	4.0	3.0	4.0	3.5	3.0	3.0	2.5	2.5	4.0	2.5	2.0	2.0	2.5	3.5
Tundra and high mountain habitats	1.5	1.0	1.0	1.5	4.0	3.5	4.0	3.0	3.5	3.0	2.5	3.0	3.0	3.5	1.5	1.5	3.0	2.5
Tropical and subtropical savannas and grasslands	2.5	2.0	1.5	2.0	3.0	3.5	3.0	3.0	2.5	3.0	2.5	2.5	3.5	3.0	2.5	2.0	2.0	3.0
Temperate grasslands	3.0	1.0	1.5	1.0	3.5	3.5	3.0	3.0	3.0	2.5	2.0	2.0	3.5	2.5	1.5	2.5	2.0	4.0
Drylands and deserts	1.7	1.7	2.3	2.7	3.3	3.7	3.0	2.7	3.7	1.7	1.7	2.0	3.3	3.7	1.3	2.0	1.3	1.7
Wetlands - peatlands, mires, bogs	1.7	2.3	1.7	2.0	3.0	3.3	3.0	3.0	3.0	4.0	3.7	3.7	3.3	2.7	2.3	1.3	1.7	3.0
Inland surface waters and water bodies / freshwater	2.5	1.5	3.0	2.0	3.8	3.8	4.0	3.5	2.8	4.0	4.0	3.3	3.8	1.3	2.8	1.5	2.3	2.3
Coastal habitats and nearshore marine	3.5	2.5	1.3	2.5	3.8	3.8	3.8	3.8	2.0	1.0	2.3	3.8	3.5	2.3	2.0	1.3	2.8	2.0
Marine/ deepwater/ offshore systems	3.5	0.3	0.8	2.3	2.3	1.8	2.0	2.3	3.5	0.3	0.3	1.5	3.5	3.3	1.3	0.3	3.3	0.8
Urban areas	1.3	1.0	1.0	1.5	3.0	3.3	3.0	1.3	1.5	1.3	1.5	1.5	1.5	1.8	1.5	1.5	1.3	1.3
Agricultural, silvicultural, aquacultural	4.0	4.0	2.5	1.8	3.0	3.3	2.8	3.0	1.5	2.3	2.5	1.5	2.0	1.3	2.3	2.5	1.3	2.5

IMPORTANCE OF BIOME FOR DELIVERING EACH NCP

🟩 VERY HIGH 🟩 HIGH 🟩 MEDIUM HIGH ⬜ MEDIUM

🟧 MEDIUM LOW 🟧 LOW 🟥 VERY LOW

3.4.2 Marine and ocean systems

Status. Considerable numbers of marine mammals are threatened in each of the four subregions **(Table 3.5)**. Extinctions in the Americas include Steller's sea cow (*Hydrodamalis gigas*) native to the Bering Sea; the Caribbean monk seal (*Neomonachus tropicalis*) native to the Caribbean Sea, Gulf of Mexico and West Atlantic Ocean; and the sea mink (*Neovison macrodon*), native to coastal eastern North America (Committee on Taxonomy, 2016).

Across subregions, trends in mammal populations are mixed (IUCN, 2017) **(Figure 3.26)**. For example, although some sea otter populations are stabilizing or increasing, abundances remain below carrying capacity (Doroff & Burdin, 2015). Both extant manatee species are considered vulnerable with decreasing populations. Because suitable sea ice habitat in the Arctic is degrading and/or disappearing rapidly with climate change, the polar bear is considered vulnerable; however the trends across the 11 populations of polar bear are mixed (4 increasing, 2 stable, 5 decreasing), and trends across eight other subpopulations are unknown (IUCN & SSC PBSG, 2017; Wiig *et al.,* 2015). Very little is known about population trends of most beaked whales and most dolphin species. Half of the turtle subpopulations that forage and/or nest in the Americas are endangered or critically endangered (IUCN, 2017).

Recent trends. In North America, protection under the US Endangered Species Act, the US Marine Mammal Protection Act, and the International Whaling Commission has led to increasing populations of some marine mammals (e.g. gray whales) and sea turtle species in USA waters, but habitat destruction and human activities continue to place other species in jeopardy. For example, the western North Atlantic right whale and Hawaiian monk seal continue to decline (Hourigan, 1999). Similarly, marine mammal populations in Canada are increasing, including grey seals in the Scotian Shelf and Gulf of St. Lawrence, harp seals in the Gulf of Maine and Scotian Shelf, western Arctic bowhead whales in the Beaufort Sea, Stellar sea lions, sea otters, and the Pacific harbour seal. Resident killer whale (*Orcinus orca*) populations off the coast of Vancouver Island have shown variable patterns since 2001, with the threatened northern population showing slight signs of recovery but the endangered southern population showing little recovery and listed as "endangered" in the USA "at risk" in Canada (Fisheries and Oceans Canada, 2017).

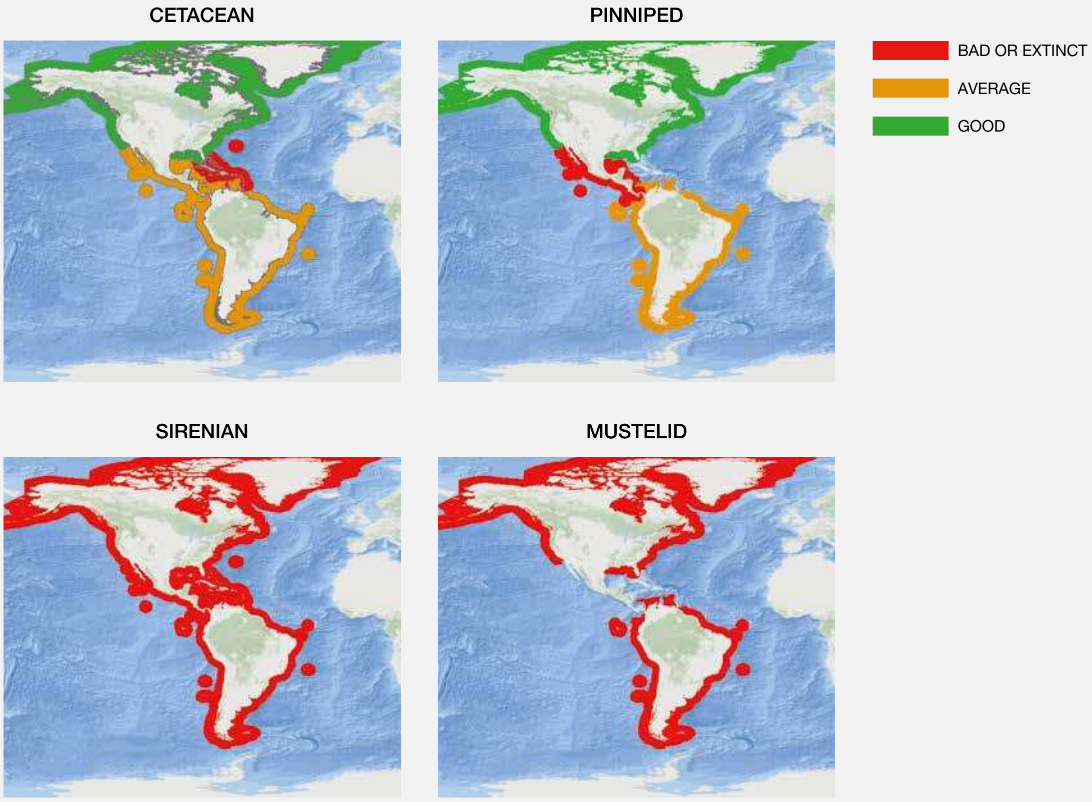
Across the Americas, despite bycatch reduction efforts, particularly in North America, some large whales are still endangered (e.g. North Atlantic blue whale, *Balaenoptera musculus*) as are cetacean populations with low abundances (Read, 2008). Some populations are small in number from previous anthropogenic impacts, such as false killer whales (*Pseudorca crassidens*). Sirenians (e.g. manatee, *Trichechus manatus*) and large whales are particularly vulnerable to fisheries bycatch and other types of removals because of inherent life history traits (e.g. slow maturation) that limit their potential population growth rate (Eberhardt & O'Shea, 1995). In Mesoamerica, the Caribbean and South America there is generally a lack of consistent, robust fisheries bycatch reduction management and/or enforcement, and fisheries bycatch remains a primary anthropogenic threat (Hucke-Gaete & Schlatte, 2004; Read, 2008). The vaquita, a small porpoise endemic to a small range in the northern Gulf of California, is an example of

Table ③ ⑤ **The number of marine mammal species found across the Americas, grouped by current IUCN Red List status and by subregion. DD = data deficient, LC = least concern, NT = near threatened, V = vulnerable, E = endangered, CE = critically endangered. Note three extinct species captured in these counts: Caribbean monk seal (*Neomonachus tropicalis*), Steller's sea cow (*Hydrodamalis gigas*), and the sea mink (*Neovison macrodon*). From IUCN Red List IUCN (2017).**

IUCN STATUS	Americas total	North America	Caribbean	Mesoamerica	South America
Data deficient (DD)	43	21	13	18	36
Least concern (LC)	35	27	10	10	18
Near threatened (NT)	4	3	0	0	1
Vulnerable (V)	7	6	2	2	3
Endangered (E)	10	7	3	3	6
Critically endangered (CE)	1	0	0	1	0
Extinct	3	2	1	0	0

Figure ③ ㉖ **Population status for each type of marine mammal categorized according to species population trends.**

"Bad or extinct" (red) indicates most or all species are declining; "Average" (orange) indicates some species are in decline, some are stable, some are increasing and some are unknown; "Good" (green) indicates most species are increasing, stable or unknown. Not shown are extinct species, or the polar bear (Ursus maritimus) only found in the North America region (IUCN Red List status is "vulnerable" and the population trend is unknown).
Source: Produced from status and trends species-level information in the IUCN Red List (2017).



CETACEAN

PINNIPED

SIRENIAN

MUSTELID

a small, critically endangered population subject to high bycatch rates; as such, this species is predicted to go extinct by 2022 or sooner (Taylor *et al.,* 2017).

Around 338 marine time series of change in the Americas have been collected. These studies are distributed inequitably and are geographically sparse, with only eight in South America (Dornelas *et al.,* 2014; Dunic, 2016; Elahi *et al.,* 2015). Further, most time series are less than 10 years, precluding a comprehensive picture of how marine biodiversity has changed in the Americas over the past 40-50 years. The only ecoregion (Spalding *et al.,* 2008) of the Americas with a sufficient sample size – the Southern California Bight, with 154 different available time series – shows a trend toward a general increase in local species diversity. Multiple studies show many marine species moving poleward, on average often in relation to shifts in ocean temperature (Cheung *et al.,* 2013; Pinsky *et al.,* 2013; Poloczanska *et al.,* 2013; Sorte *et al.,* 2010). Given high coastal diversity at low latitudes, this suggests that diversity in the future should increase outside of the tropics. Areas with extremely high cumulative human impacts (Halpern *et al.,* 2008) have tended to show losses in diversity over time (Elahi *et al.,* 2015).

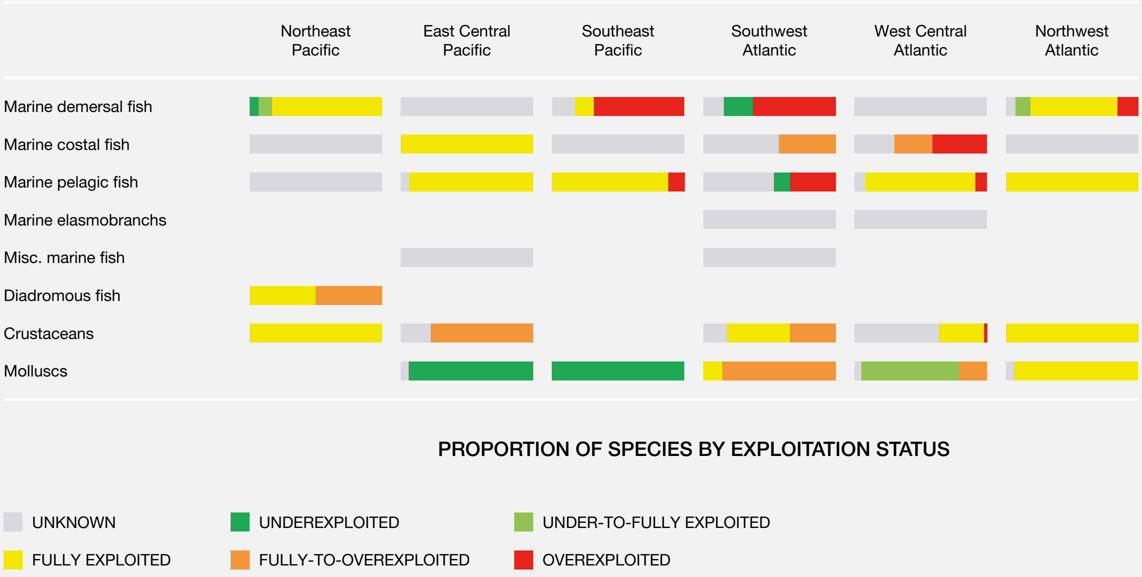
Fisheries species (fish and invertebrates). Commercial fisheries occur in all oceans surrounding the Americas. Nearly all marine animal phyla as well as seaweeds are harvested in commercial fisheries, but fished taxa and recorded landings data are heavily biased towards fishes—both ray-finned fishes and cartilaginous fishes, and invertebrate animals, especially crustaceans such as lobsters, crabs, and shrimps; molluscs such as clams, abalones and squids; and echinoderms such as sea cucumber and sea urchins. Major fishing countries in terms of total landings include Peru, the USA, Chile, Mexico, Canada, Argentina, and Brazil.

While extinction risk is generally very low for marine fishes, recovery of marine populations may take several decades to recover even when fishing intensity is relaxed (Neubauer *et al.,* 2013). In the Northeast Pacific and Northwest Atlantic, important fished species in most taxonomic groups are fully exploited **(Figure 3.27)**, i.e., they are fished at levels near maximum sustainable yield, with annual, sustainable catches near optimal levels (Costello *et al.,* 2016; Worm *et al.,* 2009). These evaluations are largely based on quantitative stock assessments, which yield relatively low uncertainty in estimates of exploitation status. Stock assessments are, however, typically conducted for species with large volumes of fishery landings or species with high ex-vessel prices so they are not representative of all marine taxa.

Fewer species from northern latitudes are considered to be either over-exploited or under-exploited **(Figure 3.27)**. A small proportion of Atlantic demersal fish species is overfished while a small proportion is underfished on both coasts of North America **(Figure 3.27)**. Moving towards the tropics, in the east-central Pacific most coastal and pelagic fish species are fully exploited, most crustaceans are fully-to-overexploited, and molluscs are underexploited. With the exception of pelagic fish species, however, many of these categorizations are highly uncertain (FAO, 2016). In the west-central Atlantic, a higher proportion of coastal fish are overexploited, fewer crustaceans are overexploited, and more molluscs appear to be overexploited compared to fisheries in the Pacific Ocean although the latter estimates are highly uncertain. Moving further south, we find that most important demersal fish species are overexploited on both coasts. Most pelagic fish tend to be exploited in the southeastern Pacific while all are exploited in the southwestern Atlantic. On the eastern coast of North America, many of the offshore fisheries exceed target levels and are not considered sustainable, especially those of elasmobranchs (Brick Peres *et al.,* 2012; Ministério do Meio Ambiente, 2006). Molluscs are underfished in the southeastern Pacific (though estimates are highly uncertain), while crustaceans and molluscs are fully exploited or overexploited in the southwestern Atlantic. The exploitation status of many species is unknown across several taxonomic groups, in particular, elasmobranchs (sharks, skates and rays) and coastal fishes.

Despite the collapse of certain fisheries, considerable efforts have been undertaken to manage fisheries in North America. Compilations of quantitative stock assessments (Costello *et al.,* 2016; Worm *et al.,* 2009) show that overfished populations usually recover after fishing pressure is reduced. In the USA, management actions have resulted in a number of successes, including Alaska groundfish, king and Spanish mackerel, striped bass, and ocean quahogs (Hourigan, 1999). Only a small percentage of USA fisheries are now considered overfished. However, fisheries impact nontarget species through bycatch and seafloor damage by trawls (Watling & Norse, 1998). In Canada, an expert panel (Hutchings *et al.,* 2012) concluded that marine fishes in Canada declined by an average of 52% from 1970 to the mid 1990s and then remained stable; most stocks, including some populations of groundfish, such as Atlantic and Pacific cod, lingcod and rockfish species, pelagic fish such as

Figure 3 27 **The proportion of fished species impacted by exploitation in different ocean regions adjacent to the Americas as determined by the FAO for individual species or species groups, which are subsequently aggregated into broad taxonomic groups. Source: Based on data provided by FAO (2012).**



herring and capelin, and anadromous fish such as coho, Chinook salmon, Atlantic salmon and Arctic char remain well below target levels.

Three of the seven global threat hotspots for neritic and epipelagic oceanic sharks in coastal waters are in the Americas (Gulf of California, southeast USA continental shelf, Patagonian shelf) (Dulvy *et al.,* 2014). Brazil, Mexico, Argentina and the USA are in the top 10 countries reporting the highest landings of chondrichthyans between 2003 and 2011 (Davidson *et al.,* 2016). Currently, despite decades of population declines for many chondrichthyans, only 18 sharks and rays have been listed by CITES (Convention on International Trade in Endangered Species). Stock assessments for a number of chondrichthyans in the Americas report declines of 20¬–80% from unfished conditions for multiple species (Highly Migratory Species Management Division, 2006). In the eastern central Pacific, chondrichthyan landings steadily increased throughout the latter half of the 20th century, peaking to ~50,000 tonnes in 2000, and declining to <40,000 tons in recent years (FAO, 2011). Mexican catches (which represent >60% of regional chondrichthyan landings) have continued to increase, and current fishing practices targeting elasmobranch aggregations on breeding and pupping grounds are posing increased threats to many species (Kyne *et al.,* 2012). Historic and current fishery landings data are limited, and the population status of most shark species throughout the region is poorly understood (Kyne *et al.,* 2012). However, fishery surveys suggest that two species of sawfish – the largetooth sawfish and the smalltooth sawfish – may have experienced local extinctions in Belize and possibly Guatemala (Kyne *et al.,* 2012).

Canada has become one the world's third-largest exporters of shark meat, and the USA has experienced the second greatest increase in chondrichthyan landings since 2003 (Davidson *et al.,* 2016). The FAO recently identified Brazil as having one of the largest and most rapidly expanding shark product consumer markets in the world (Barreto *et al.,* 2016). Some 32% of all Brazilian chondrichthyans are endangered and two species of shark are considered regionally extinct, according to IUCN Red List criteria (Reis *et al.,* 2016). The southeastern coast of South America is also considered a hotspot of deepwater threatened chondrichthyans (Dulvy *et al.,* 2014). Targeted shark fisheries have also expanded in Mexico and Venezuela (Tavares & Lopez, 2009).

Fisheries management plans are now in place for many elasmobranchs in the northwestern Atlantic, but lacking in most other areas. Recently, Chile, Colombia, Ecuador, and Peru have developed a regional action plan for protecting and managing chondrichthyans (Davidson *et al.,* 2016). Recently completed stock assessments for two shark species in the northeastern Pacific also revealed that all populations are either not overfished or are recovering from historical overfishing (Kleiber *et al.,* 2009; Tribuzio *et al.,* 2015; Young *et al.,* 2016).

3.4.2.1 Coastal habitats/Coastal and near shore marine/inshore ecosystems

Coastal marine habitats provide many ecosystem services, including food, protection against coastal erosion, recycling of pollutants, climate regulation and recreation.

Salt marshes

Status. Salt marshes are intertidal ecosystems that are regularly flooded with salt or brackish water and dominated by salt-tolerant plants. They remove sediment, nutrients and other contaminants from runoff and riverine discharge (Gedan *et al.,* 2009), protecting estuarine biota. They also protect coastal communities from storm waves (Costanza *et al.,* 2008) and are nursery areas for many commercial fish species. Many migratory shorebirds and ducks use salt marshes as stopovers during migration, and some birds winter in marshes. Wading birds, such as egrets and herons, feed in salt marshes during the summer. After European settlement, North American salt marshes were filled for urban or agricultural development or garbage dumps. Using historical maps, Bromberg & Bertness (2005) estimated the average loss in New England at 37%. Rhode Island has lost the most, 53%, since 1832. Salt marshes are estimated to have occupied 200,000 to 400,000 ha in pre-settlement Louisiana, with an estimated 50–75% remaining (Smith, 1993) as of two decades ago. San Francisco Bay has seen a 79% reduction in its salt marshes. Salt marshes in South America have been far less drained (6%) than in North America (~50%) (Zedler & Kercher, 2005). However, these marshes are threatened by agriculture, construction of flood control measures and hydroelectric power, pollution, and large-scale fish and shrimp aquaculture. Some marshes on the Atlantic Coast of South America have extensive bare areas dominated by high densities of the crab *Chasmagnathus granulata* (up to 60 individuals / m^2), which consumes the marsh grass *Spartina densiflora*. The bare areas, often comprising half of the habitat, are due to crab herbivory. It is suspected that the high densities of *Chasmagnathus* are at least in part due to the overfishing of predators (Bortolus *et al.,* 2009). In South America, invasive *Spartina* species are found in coastal marshes (Orensanz *et al.,* 2002).

Recent trends. In recent years, sea level rise has begun to impact many previously healthy marshes in the Americas (such as ponding, where water remains on the marsh surface during low tide and plants get waterlogged). The actual rate of sea level rise in the future will affect which marshes can persist. Other marshes are being restored, a very expensive procedure. There are some attempts to raise their elevations (Ford *et al.,* 1999), yet given accelerating sea level rise, extensive areas will most likely continue to be lost. The invasive reed, *Phragmites australis*, which has reduced plant diversity in many brackish marshes in the eastern coast of the USA and is often removed in restoration projects, allows marshes to increase their elevation more rapidly (Rooth & Stevenson, 2000) and might better enable marshes to keep up with sea level rise. While 50% of the salt marsh area in New England had been lost by the mid-1970s, recent loss rates have been lower because of awareness of their value and restoration projects (Valiela, 2006). Long-enrichment of coastal salt marshes has reduced belowground organic matter, contributing to subsidence (Turner *et al.,* 2009).

Mangroves

Status. In tropical and subtropical regions, intertidal mangroves perform similar ecological functions as salt marshes in temperate zones. The red mangrove, *Rhizophora* spp, lives at the water's edge with its aerial prop roots in the water, serving as the substrate for a community of attached invertebrates and shelter for fishes that swim among the roots. Caribbean mangroves are reported to host the world's richest mangrove-associated invertebrate fauna worldwide (Ellison & Farnsworth, 1996). Mangroves provide many NCP such as wood products, microclimate regulation, shoreline protection, nutrient cycling and carbon storage (Vo *et al.,* 2012).

Recent trends. Recently, use of mangroves has increased leading to substantial loss (Valiela *et al.,* 2001). Construction of shrimp and fishponds for aquaculture accounts for over 50% of the world's mangrove loss. In the Americas, losses average about 2.1% per year, with annual losses up to 3.6% per year. This is likely due to exploitation, deteriorating water quality, coastal development and climate change (Gilman *et al.,* 2008; McKee *et al.,* 2007; Polidoro *et al.,* 2010). In the Caribbean, mangrove area has declined by about 1% annually over the last 30yrs, the second highest rate of loss globally (FAO, 2007). In recent years, mangroves have been spreading northward in Florida, expanding their range in response to warming (Cavanaugh *et al.,* 2014). Since they are not likely to be harvested for wood or removed for aquaculture, this northward move may counterbalance some of the threats.

Submerged aquatic vegetation

Status. Seagrasses live submerged in salt or brackish water full-time and provide habitat for animals such as scallops, and, in tropical regions, juvenile coral reef fishes. USA populations crashed in the 1930s due to disease and slowly recovered over subsequent decades. Since the 1960s, much of the Submerged aquatic vegetation disappeared in North Atlantic estuaries, particularly in Chesapeake Bay. Loss of Submerged aquatic vegetation results in a loss of food and habitat for many species (US Fish and Wildlife Service, 2011). Of the seven native seagrass species in the Caribbean, two (*Halophila engelmannii* and *H. baillonii*) are considered to be near threatened and vulnerable. Elevated nutrient levels (eutrophication) is the biggest threat in the Americas and is particularly acute in developing nations with rapidly growing economies, where environmental legislation is weak. These local and regional threats exist with a backdrop of environmental change and sea level rise.

Recent trends. There was considerable loss, degradation, and fragmentation of seagrasses, as 2.6 km^2 in Biscayne Bay (Florida, USA) between 1938 and 2009 (Santos *et al.,* 2016). Extensive losses have been reported from Canada (Matheson *et al.,* 2016), and the Caribbean (Van Tussenbroek *et al.,* 2014). The Caribbean Coastal Marine Productivity program found that most study sites showed a decline in seagrass health between 1993 and 2007 (Van Tussenbroek *et al.,* 2014). However, in some areas that have undergone restoration and controls on nutrients, such as Chesapeake Bay in the USA, there has been some recovery (Chesapeake Bay Program, 2017). In cases where nutrient limitations are implemented, recovery is a very slow process, involving the replacement of fast-growing macroalgae with slower-growing plants. Simulation models predict recovery times of several years for fast-growing seagrasses to centuries for slow-growing seagrasses following nutrient reduction (Duarte, 1995).

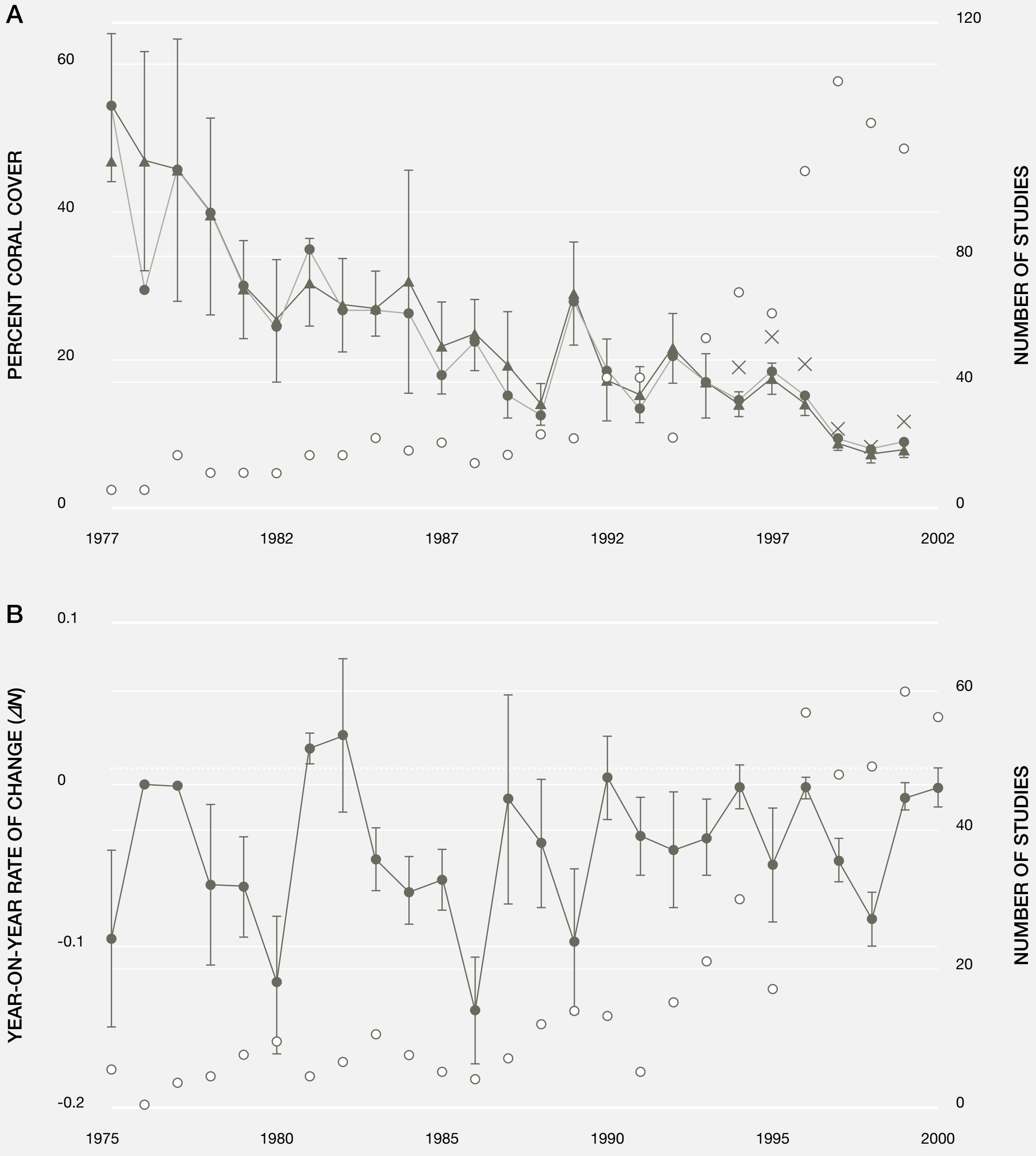
Coral reefs

Status. Coral reefs are one of the most productive and diverse ecosystems in the world. In addition to the many species of corals, they include populations of sponges, Echinoderms, mollusks such as giant clams, nudibranchs, and octopuses, crustaceans such as crabs, lobsters and shrimp, and a huge diversity of fishes, all of which are either directly or indirectly dependent on the foundation species, the corals. When corals degrade or disappear the rest of the community degrades or disappears. Coral reefs perform vital ecosystem services in tropical countries: they serve as protection against storms, attenuating wave intensity, their fisheries are a source of food for millions of people, and they are a source of considerable revenue from tourism.

Recent trends. Gardner *et al.* (2003) found that live coral cover in the Caribbean was reduced from more than 50% in the 1970s to just 10% today **(Figure 3.28)**. This decline was followed by widespread and severe coral bleaching in 2005, which was in turn followed by high coral mortality as a result of disease at many locations. Healthy corals are rare on the intensively studied reefs of the Florida reef tract, USA Virgin Islands and Jamaica (Gardner *et al.,* 2003). Furthermore, two of the formerly most abundant foundation

Figure 3 28 **Total observed change in percent coral cover across the Caribbean basin during the past three decades.**

Ⓐ Percent coral cover from 1977 to 2001. Annual coral cover estimates (▲) are weighted means with 95% bootstrap confidence intervals. Also shown are unweighted mean coral cover estimates for each year (●), the unweighted mean coral cover with the Florida Keys Coral Monitoring Project omitted (×), and the sample size (number of studies) for each year (○). Ⓑ Year-on-year rate of change [mean ΔN ± SE] in percent coral cover across all sites between 1975 and 2000 (●),which largely fall below the dotted line representing no change, and the number of studies for each period (○). Source: Gardner *et al.* (2003).



species of Caribbean reefs, the elkhorn coral (*Acropora palmata*) and staghorn coral (*Acropora cervicornis*), have been added to the US Endangered Species List. The decline of herbivorous species (e.g. parrotfish) in coastal marine areas has also been of consequence especially as many are vital to reef resilience (Mumby *et al.,* 2006). Many reef fish continue to be exploited (e.g. endangered *Nassau grouper, Epinephelus striatus*) (Sadovy & Eklund, 1999).

Jackson *et al.* (2014) found that the average coral cover for 88 locations in the Caribbean declined from 34.8% in 1984 to 19.1% in 1998 to 16.3% at the time of the report, but there was great disparity among sites. In contrast, macroalgal cover increased from 7% to 23.6% between 1984 and 1998 and held steady but with even greater disparity among locations since 1998. Differences among locations can be attributed to local factors such as human population density, overfishing of herbivorous fishes, and invasive species. The invasion of the predatory lionfish (see Chapter 1) has been particularly devastating to populations of herbivorous fish. The massive loss of corals in the Caribbean (see Chapter 4 for drivers) has been associated with increases in large seaweeds (macroalgae), outbreaks of coral bleaching and disease, and failure of corals to recover from natural disturbances like hurricanes (Jackson *et al.,* 2014). There are attempts to restore some Acropora reefs in the Caribbean with more tolerant strains. Bozec *et al.* (2016) concluded that reduced fishing for parrotfish and other herbivores would make reefs more resilient to warming.

Global warming is placing Caribbean coastal ecosystems under further stress (see Chapter 4). The predicted increased severity of hurricanes and greater rainfall seasonality here are also likely to increase stress (Fish *et al.,* 2009). In Brazilian reefs of the Southwestern Atlantic Ocean, long-term sea water thermal anomaly events, equal or higher than 1ºC, were responsible for more than 30% of bleached corals in the inshore reefs from 1998 to 2005, (Leão *et al.,* 2010).

3.5 PERILS AND OPPORTUNITIES FOR CONSERVATION

3.5.1 Threat status and temporal trends

Knowledge of threat status, temporal trends, and the main causes underlying extinction probability constitute useful information for policymakers for prioritizing recuperation plans and protection measures and for other stakeholders who wish to reap well-being benefits from particular species or contribute to biodiversity conservation.

Status. Overall, 14,184 species from taxonomic groups within which > 90% of species have been globally assessed by IUCN for extinction risk and synthesized by Brooks *et al.* (2016) are present in the Americas. Groups assessed cover mammals, birds, chameleons, amphibians, sharks and rays, selected bony fish groups (angelfishes and butterflyfishes, tarpons and ladyfishes, parrotfishes and surgeonfishes, groupers, wrasses, tunas and billfishes, hagfishes, sturgeon, blennies, pufferfishes, seabreams, porgies, picarels), freshwater caridean shrimps, cone snails, freshwater crabs, freshwater crayfish, lobsters, reef-building corals, conifers, cacti, cycads, seagrasses, and plant species occurring in mangrove ecosystems. Conspicuously absent are the majority of flowering plants. Recognizing that available data is strongly skewed towards animals, in total, 24.5% of assessed species are documented as threatened with a high risk of extinction in the wild in the medium term future. The inclusion of data-deficient species for these groups could shift this percentage to as high as 34.7% or as low as 21.2%. The great majority of species assessed for the taxonomic groups mentioned (92.3%, 13,096 species) are endemic to the Americas region.

Notable differences in extinction risk characterize the different subregions of the Americas **(Figure 3.29)**. Considering all species, North America shows much lower extinction risk than South America, Mesoamerica, and the Caribbean. With the exception of South America, extinction risk tends to be higher among endemic species. Especially high extinction risks for endemics are found in the Caribbean and Mesoamerica.

Recent trends. For mammals, birds, amphibians, corals, and cycads, global assessments of extinction risk against the Red List categories and criteria have been undertaken multiple times over the last three decades to derive Red List indices as indicators of the rate at which species groups are sliding towards extinction; these can be combined with species distribution data to produce geographically downscaled Red List indices (Rodrigues *et al.,* 2014). According to this criterion, overall the extinction risk has increased over the last 23 years in the Americas, but again there are notable subregional differences **(Figure 3.30)**. Extinction risk in the North America subregion has increased slightly, in Mesoamerica it has remained relatively steady, while in the Caribbean and South American it increased the fastest. Species in the Caribbean region are declining towards extinction the fastest of all but, of course, there are fewer overall species here (Brooks *et al.,* 2016).

The main threats in the North American subregion come under the IUCN category termed “Invasive & other problematic species (whose origins are uncertain), Genes & diseases” **(Figure 3.31)**. In the other three subregions, the main threats are “Agriculture & aquaculture” and “Biological resource use”. While it was seen earlier that there are many alien and invasive species in the Caribbean, the category of “Invasive & other problematic species, Genes & diseases” does not rank high as a threat, at least in the groups assessed to date. The relatively less importance still of the invasive species category in Mesoamerica and South America could relate to the fact that invasive species are less prevalent at tropical latitudes. The overall pattern for these last subregions mirrors the global threat trends (Maxwell *et al.,* 2016). Again, it should be borne in mind that species assessed are strongly skewed towards animal groups. Trends could change measurably with the inclusion of the many threatened plant species in the Americas. Throughout the Americas, biological resource use may be a primary concern in the marine environment (McCauley *et al.,* 2015).

Figure 3 ㉙ **Extinction risk for species in the Americas as a whole (Am) and by subregion (Caribbean: Ca, Mesoamerica: MA, North America: NA, South America: SA).**

Red lines show midpoint estimate of proportion of threatened species. The top 5 rows are all assessed species in the dataset, and the bottom 5 are the subset of endemic species. Source: Data synthesized by Brooks *et al.* (2016).

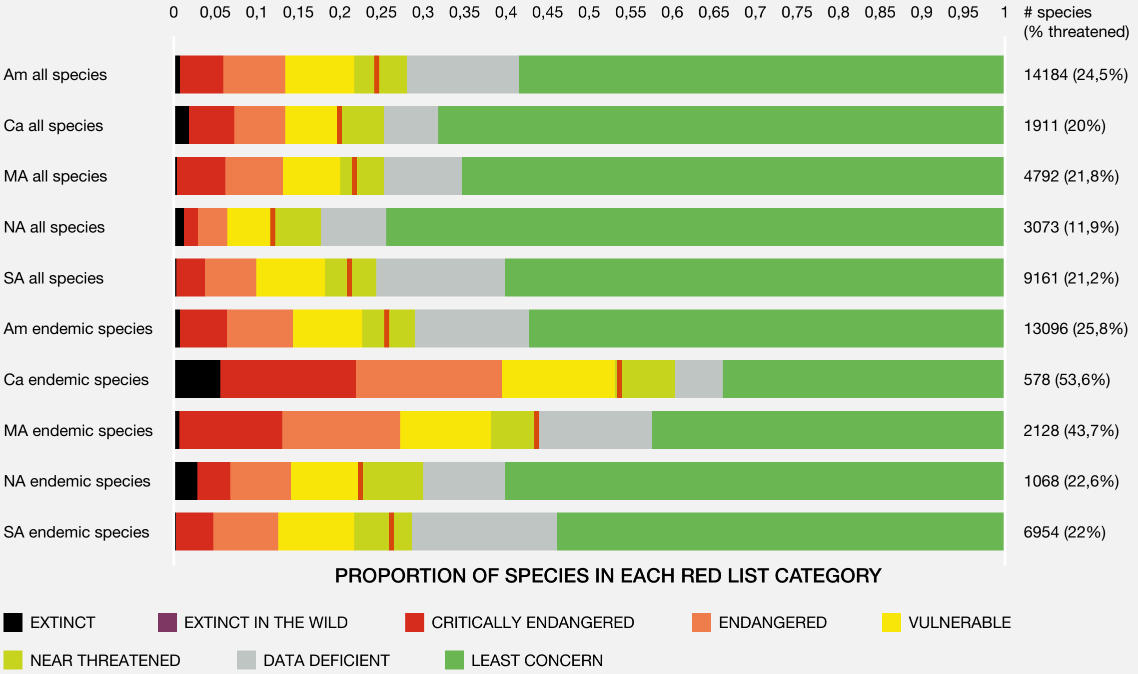


Figure 3 ㉚ **Red List indices of species survival for mammals, birds, amphibians, corals, and cycads, weighted by the fraction of each species' distribution occurring within each region/subregion.**

The position on the y-axis indicates the aggregate extinction risk facing species in the region overall. It ranges from 1 (if no species are threatened with extinction) to 0 (if all species are extinct). The horizontal axis shows time, so the slope of the lines for each subregion shows how the extinction risk of the species in that subregion has been changing. A declining slope indicates that extinction risk is growing; an increasing slope indicates that extinction risk is declining. Source: Data synthesized by Brooks *et al.* (2016).

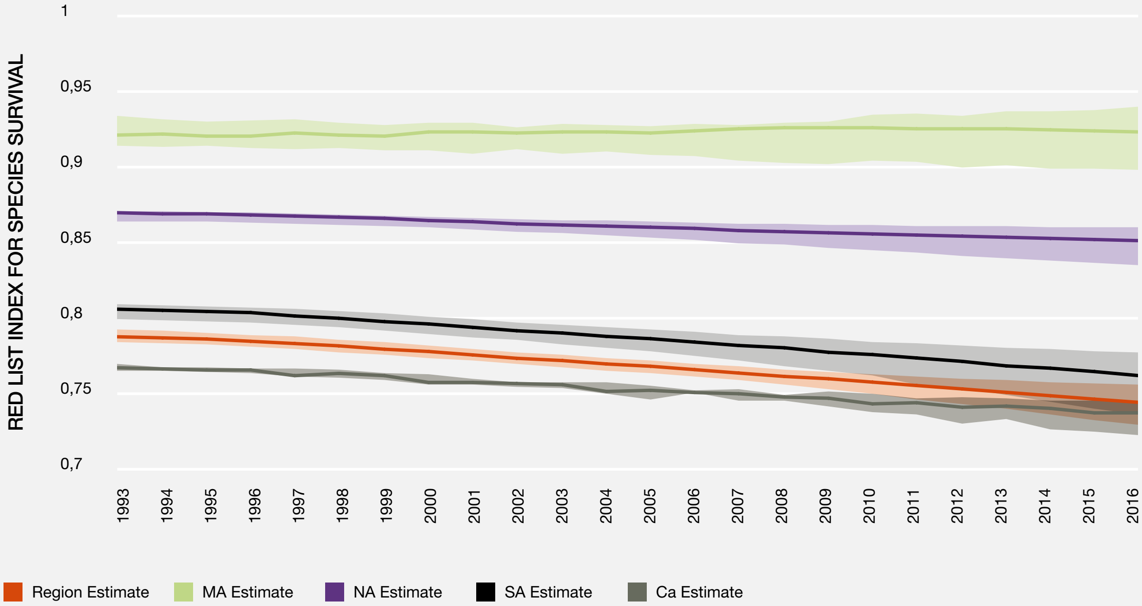
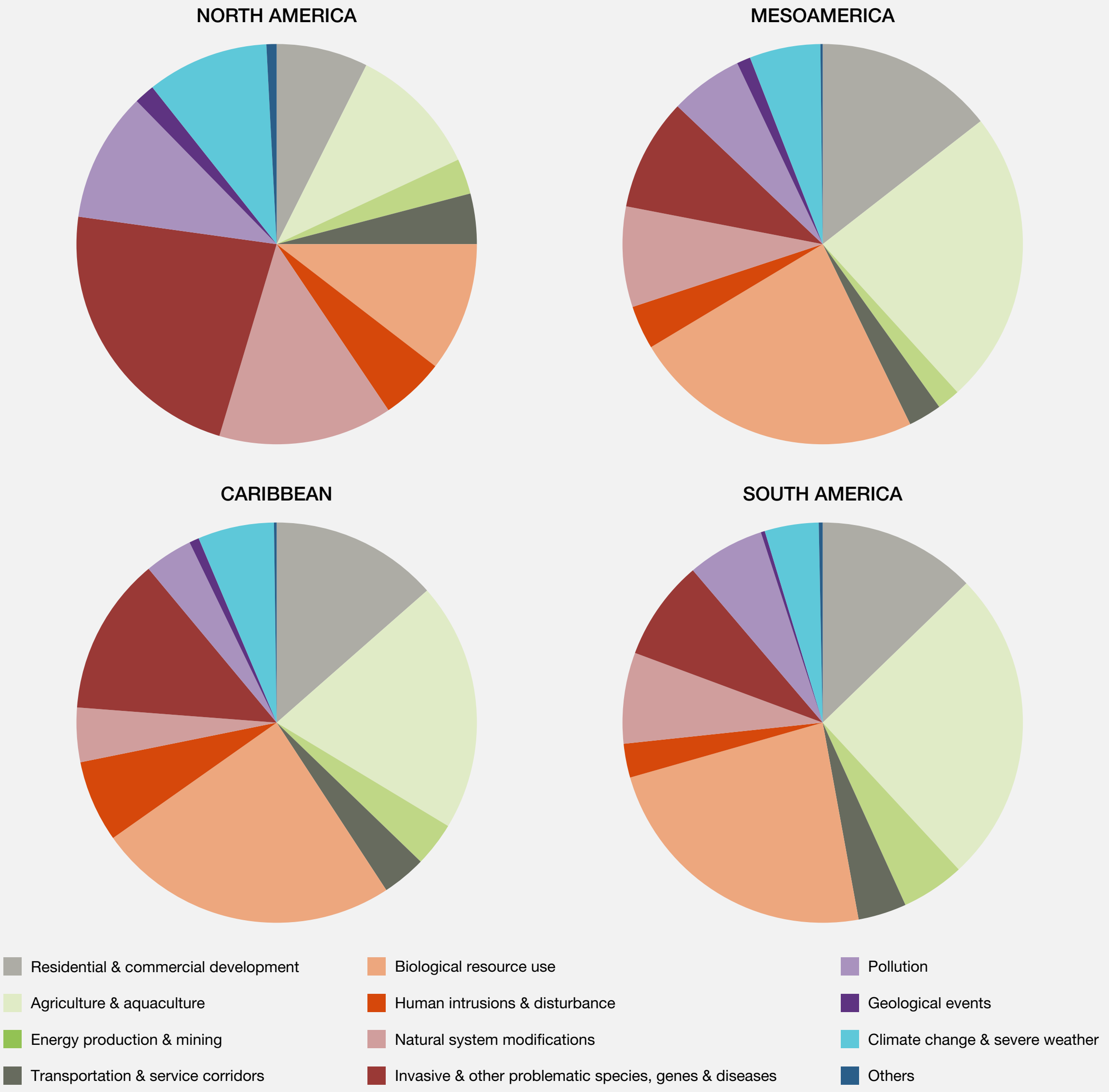


Figure 3 31 **Comparison of the main causes of extinction risk in the Americas.**

When a species is threatened by more than one cause, all causes were included to calculate the proportion. Source: Data from IUCN Red List threat classification, IUCN (2017).

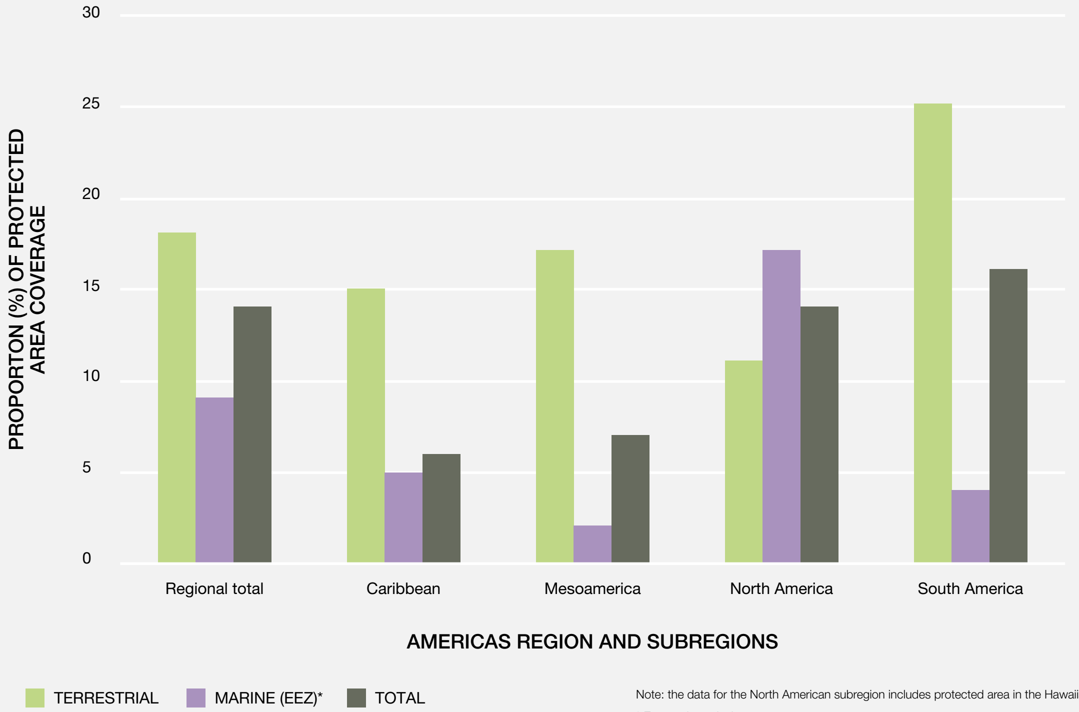


3.5.2 Protected areas

Most early protected areas in the Americas were established with the aim of protecting iconic landscapes. Heightened concern over environmental degradation and the importance of biodiversity led to changes in the motives for establishing protected areas, with an increasingly greater emphasis placed on *in situ* conservation, coverage of KBA (Key Biodiversity Areas), “hotspots”, ecosystem services and indigenous rights. Simultaneously, the range of stakeholders involved in establishing protected areas expanded to include private citizens, in addition to governments. Many early protected areas established in mountainous landscapes today perform important roles in protecting key ecosystem services such as water regulation and slope stability.

Status. Total protected area coverage for the Americas is 14%, with 18% of its terrestrial area and 9% of its marine area (within the Exclusive Economic Zone, EEZ) protected **(Figure 3.32)**. Protected area coverage shows variation both among subregions and in relation to the relative amount of land and the marine EEZ protected. For terrestrial habitats, South America has the highest fraction of land in protected areas, whereas for EEZ marine protection, North America has made the most advances **(Figure 3.32)**. Chile recently announced the creation of two new large marine

Figure 3 32 **Percentage of terrestrial, marine and total protected area coverage in the Americas region and subregions. Source: Based on UNEP-WCMC & IUCN (2015), synthesized by Brooks *et al.* (2016).**



Note: the data for the North American subregion includes protected area in the Hawaiian Islands.

* Economic exclusive zone.

protected areas (around the Juan Fernández Islands and in the Cape Horn-Isla Diego Ramírez area) (Ministerio del Medio Ambiente, 2017) in the South American subregion. Mexico announced the creation of Parque Nacional Revillagigedo (CONANP, 2017) in the Mesoamerican subregion. The Americas, thus are responding rapidly to the challenge of marine protection. These new marine protected areas are not included in **Figure 3.32**.

Recent trends. Over the past few decades, there has been an increase in the number of protected areas and the amount of land protected throughout the Americas region (see Chapter 2). In North America, the number of protected areas has almost tripled, and in the Caribbean, it has almost doubled. Protected areas came slowly to Mesoamerica, but have increased in number from 150 to more than 700 since the 1980s, and in South America, they have increased more than four-fold. In South America, over the past 10 years, an additional 683,000 km^2 of new protected areas were added to the Amazon Basin by different countries, increasing the amount of the Amazon protected by 10% (Charity *et al.,* 2016). According to the most recent analysis for terrestrial biomes, a large number of biomes in the Americas are better protected than the global average **(Table 3.6)**; however, despite advances, and of concern given the rapid rate of conversion in many (3.4), some fall well below the global rate. It should be pointed out that the exact level of protection in these biomes is constantly changing because of new initiatives and depends also on how the various biomes are defined, which is far from uniform. In general, it can be seen that closed forests are better protected in relation to the global rate than non-forested areas and wet forests better than dry forests.

With regard to priority areas for conservation, the Americas region hosts 20% of globally identified KBA **(Table 3.7)**. KBA include the 12,000 Important Bird and Biodiversity Areas (IBAs), identified by BirdLife International (2015), plus Alliance for Zero Extinction (AZE) sites (Ricketts *et al.,* 2005) and other KBA identified through hotspot profiles supported by Critical Ecosystem Partnership Fund (World Database of Key Biodiversity Areas, n.d.).

The total protected area coverage of KBA has increased significantly over the past 50 years **(Figure 3.33)**. Brooks *et al.* (2016) synthesize all three datasets for the Americas region. Currently (as of 2015) 17.0% of IBAs and 20.6% of AZE sites are fully covered in the Americas as a whole. At the subregional level, for IBAs South America lags strongly behind; for AZE sites the Caribbean takes the lead, while North America lags behind the most **(Figure 3.33)**.

With the increasing recognition of indigenous rights and public recognition of NCP, the establishment of indigenous

Table 3 6 **Percentage protection of terrestrial biomes in the Americas according to biogeographic realm. The North American realm (= Nearctic realm in Jenkins & Joppa, 2009) extends into Mexico and thus is larger than the corresponding IPBES subregion. The Neotropical realm includes South American and Caribbean subregions and part of the Mesoamerican subregion as defined by the IPBES. Biomes shown in bold enjoy a high level of protection relative to the global rate in at least one of the biogeographical realms. Based on data in Jenkins & Joppa (2009).**

BIOME	Global	North American	Neotropical
Tropical and subtropical moist broadleaf forests	21		**32**
Tropical and subtropical dry broadleaf forests	8	0	9
Tropical and subtropical coniferous forests	7	7	8
Temperate broadleaf and mixed forests	11	12	**29**
Temperate coniferous forest	25	**33**	
Boreal forests/taiga	9	10	
Tropical and subtropical grassland, savannas and shrubland	13	8	11
Temperate grasslands and savannas	4	3	2
Flooded grasslands and savannas	20		15
Montane grasslands and shrublands	25		14
Tundra	17	**22**	
Mediterranean forests, woodland and scrub	7	**21**	1
Deserts and xeric vegetation	9	**14**	9
Mangroves	21		**37**

Table 3 7 **Number and percentage of KBA by subregion in the Americas relative to the global total. Source: Data are from the World Database of Key Biodiversity Areas™, searched October 22, 2017. http://www.keybiodiversityareas.org/ site/search.**

REGION	# KBA	%
North America	985	6.35
Caribbean	419	2.70
Mesoamerica	305	1.96
South America	1,371	8.83
Americas	3,080	19.84
GLOBAL	**15,524**	**100**

and private reserves has increased notably. Indigenous reserves in South America tend to be concentrated in tropical forests where they contribute greatly to the integrity of ecosystem services, and the sustainable use of many plant and animal species used for human-well-being. Currently, indigenous reserves in Latin America and the Caribbean account for around 12% of all protected land (Nelson & Chomitz, 2011) (for more details on the contribution of indigenous reserves to human well-being see Chapter 2). In the Amazon, around 3000 indigenous lands (not all recognized) now cover over 2 million km^2 (Charity *et al.,* 2016; **Figure 3.34**). Both uninhabited protected areas (parks) and indigenous lands have proven to reduce deforestation and fire in South American wet tropical forest

Figure ③ ㉝ **Growth in the proportion of KBAs (Key Biodiversity Areas) completely covered by protected areas in the Americas between 1970 and 2015.**

Ⓐ Trends in the four American subregions for IBAs (Important Bird and Biodiversity Areas). Ⓑ Trends in the four American subregions for AZEs (Zero Extinction sites). Ⓒ Trends in the Americas as whole for both IBAs and AZEs. Source: IUCN & Birdlife International (2016) as synthesized by Brooks *et al.* (2016).

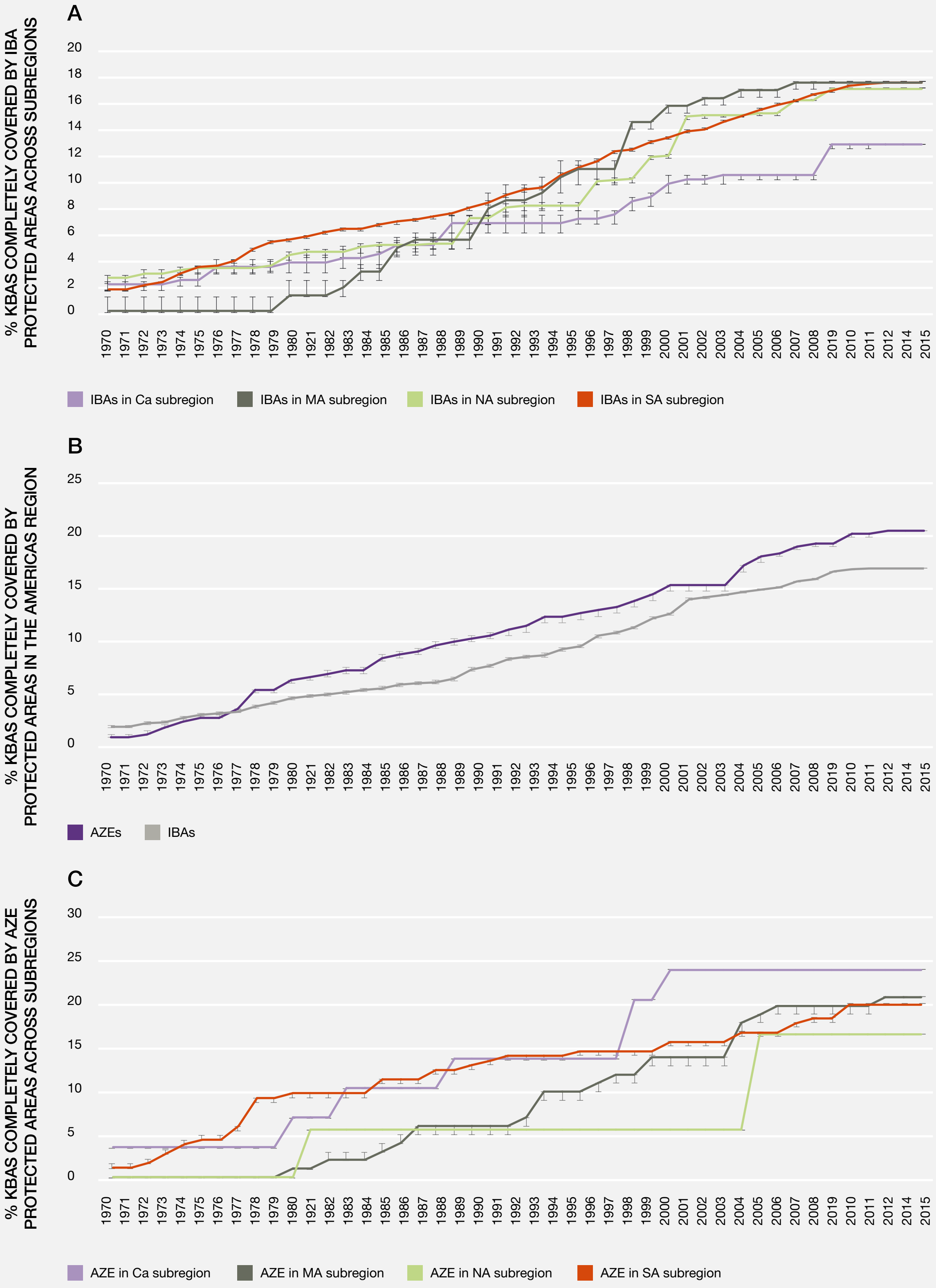
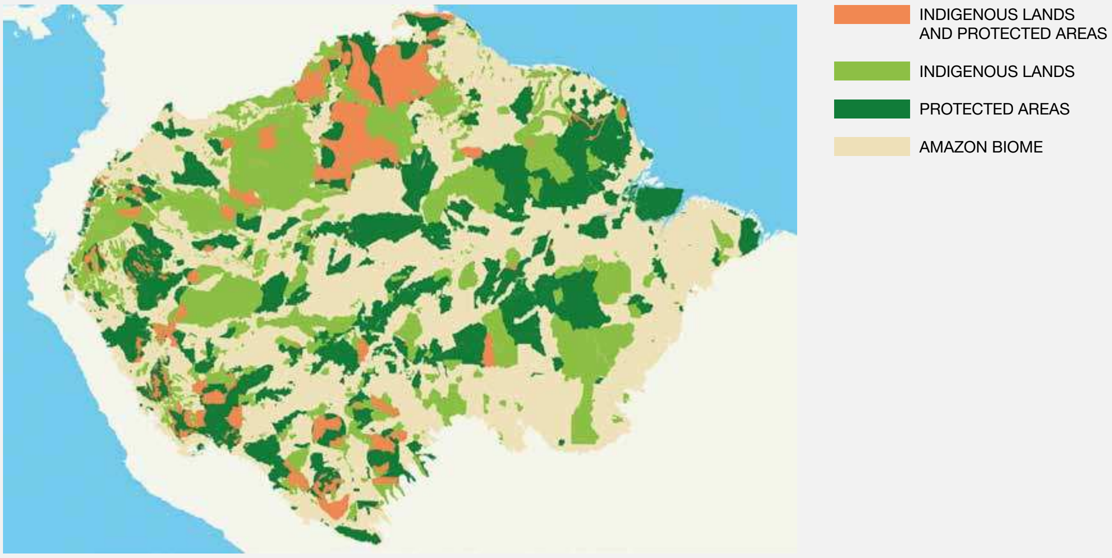


Figure ③ ㉞ **Location of protected indigenous lands, other indigenous lands and protected areas in the Amazon basin as of March 2016. Source: Charity *et al.* (2016).**



(Armenteras *et al.,* 2009; Nepstad *et al.,* 2006; Nelson & Chomitz, 2011), and contain viable populations of most threatened tree species (ter Steege *et al.,* 2015).

Private conservation efforts are now important in the temperate forests of southern South America (Pliscoff & Fuentes-Castillo, 2011), the Mediterranean forests, woodland and scrub biome in California (Paulich, 2010), and in Brazil in general (de Vasconcellos Pegas & Castley, 2016). Brazil's private reserves are distributed across seven biomes (six terrestrial and the marine); they are recognized under federal law and created to protect nature in perpetuity. Private conservation efforts in the USA have been stimulated by the fact that around two-thirds of the land in the continental USA is privately owned and three-quarters of all threatened or endangered species depend on private land for habitat, food or breeding (Paulich, 2010). A similar situation could occur in the South American Mediterranean biome. While private initiatives are noteworthy, they sometimes risk outcomes of the establishment of protected areas in places that are large and cheap but of less importance for biodiversity conservation (Barnes, 2015), or choices being made on purely aesthetic grounds increasing protection where it sometimes is perhaps less required. It is therefore essential to complement these measures with measures of safeguard of important sites (Butchart *et al.,* 2016) and encourage protection where it is most needed, regardless of aesthetic value.

Despite the overall increase in protection and notable conservation success stories (e.g. Carabias *et al.,* 2010), major conservation incongruencies within many biomes still remain. Incongruencies address both what and how much is conserved. With respect to what is conserved, as an example, although California has pioneered multiple species habitat conservation plans and other regional and multi-benefit approaches to enhance integrated planning of protected areas (Pincetl *et al.,* 2016), unprotected areas tend to harbor the highest numbers of rare plant taxa (Pavlik & Skinner, 1994), while important areas with high levels of plant neoendemism fall outside of protected lands (Kraft *et al.,* 2010). How common this trend is in other biomes remains to be seen and should be a priority question.

With respect to how much is conserved, as examples, the Central American system of protected areas currently includes 669 protected areas summing 129,640 km^2, the majority of which correspond to moist tropical and subtropical forest (Programa Estado de la Nacion, 2008; The Nature Conservancy, 2005). For Mesoamerica defined as the five southernmost states of Mexico to the Darien in eastern Panama, while 29% of tropical broad-leaved forest is protected, only 10% of coniferous forest comes under protection (DeClerck *et al.,* 2010). For South American moist tropical and subtropical forests, less than 2% of Atlantic rainforest is protected.

Incongruencies are even more extreme in other biomes. Overall, only 0.3% of Tropical dry forest in Mesoamerica, 7% in South America and 10% in the Caribbean is protected (Portillo-Quintero & Sánchez-Azofeifa, 2010); this percentage descends to 0.2% in Mexico and 1.0% in Venezuela, but is a much higher 15% in Costa Rica (Portillo-Quintero & Sánchez-Azofeifa, 2010), indicating notable differences in individual country efforts. Protection of Chaco is about 10% (Fehlenberg *et al.,* 2017), ranging from 36% in Bolivia to 6.5% in Paraguay. Although the amount of protected land tripled in the wider Mediterranean biome in South America between 1975 and 2017, less than 3% is currently protected (based on data in http://www.mma.gob.cl) with some particular ecosystems of the biome totally lacking protection (Pliscoff & Fuentes Castillo, 2011). Currently, 8.3% of the Brazilian Cerrado is considered to be under some kind of protection, with only 3.1% in strictly protected areas (National Database for Protected Areas/ Brazilian Ministry of the Environment - Cadastro Nacional de Unidades de Conservação - CNUC, updated February 7, 2017). South American drylands are very poorly protected ¬– 1% of land area of the Caatinga (Banda-R *et al.,* 2016; de Oliveira *et al.,* 2012), and 1–2% of Chilean western desert (Arroyo & Cavieres, 1997; Luebert & Pliscoff, 2006). Likewise, in the EEZ much variation is found for marine conservation (Watson *et al.,* 2014). All these incongruencies have many sources, but one obvious one is a lack of systematic planning among countries where a given biome is found.

3.6 KNOWLEDGE AND DATA GAPS

Biodiversity inventories. Basic inventoring of biodiversity is far from complete in the Americas. Accumulated species descriptions for vascular plants have not yet reached an asymptote **(Figure 3.35 A)**. Over the period 2004-2016, Brazil registered the largest number of new plant species names in the International Plant Names Index worldwide **(Figure 3.35 B)**. Over 2,000 new species of plants and vertebrates have been described from the Amazon alone since 1999 (Charity *et al.,* 2016). Even in well-known groups such as mammals, 42% of the new species described worldwide between 1993 to 2008 came from the Americas (Ceballos & Ehrlich, 2009), mostly from Mesoamerica and South America. These trends are likely to be repeated for other taxonomic groups. Knowledge of invertebrates is particularly deficient including for taxonomic groups of particular importance for human well-being, such as bees. This assessment has shown that high-quality information on species richness across the entire Americas is available for a very limited number of taxonomic groups. Some estimates of biodiversity, of course, might be exaggerated if care was not taken to remove synonyms. Overall, an accurate estimate of the total biodiversity in the Americas is currently not possible, and is unlikely to become available for a long time at the current rate of progress. Also, systematized knowledge on the use of biodiversity is still scarce, despite major efforts made in Mexico, Costa Rica, Brazil, and Colombia.

Similar and probably even much larger knowledge gaps occur in the marine (and probably freshwater) realms. Based on their studies, it is predicted that only about half of marine organisms have been described for the Atlantic and Pacific coasts of South America (Miloslavich *et al.,* 2011); as on the land, a severe lack of taxonomic expertise in the subregion is a major handicap.

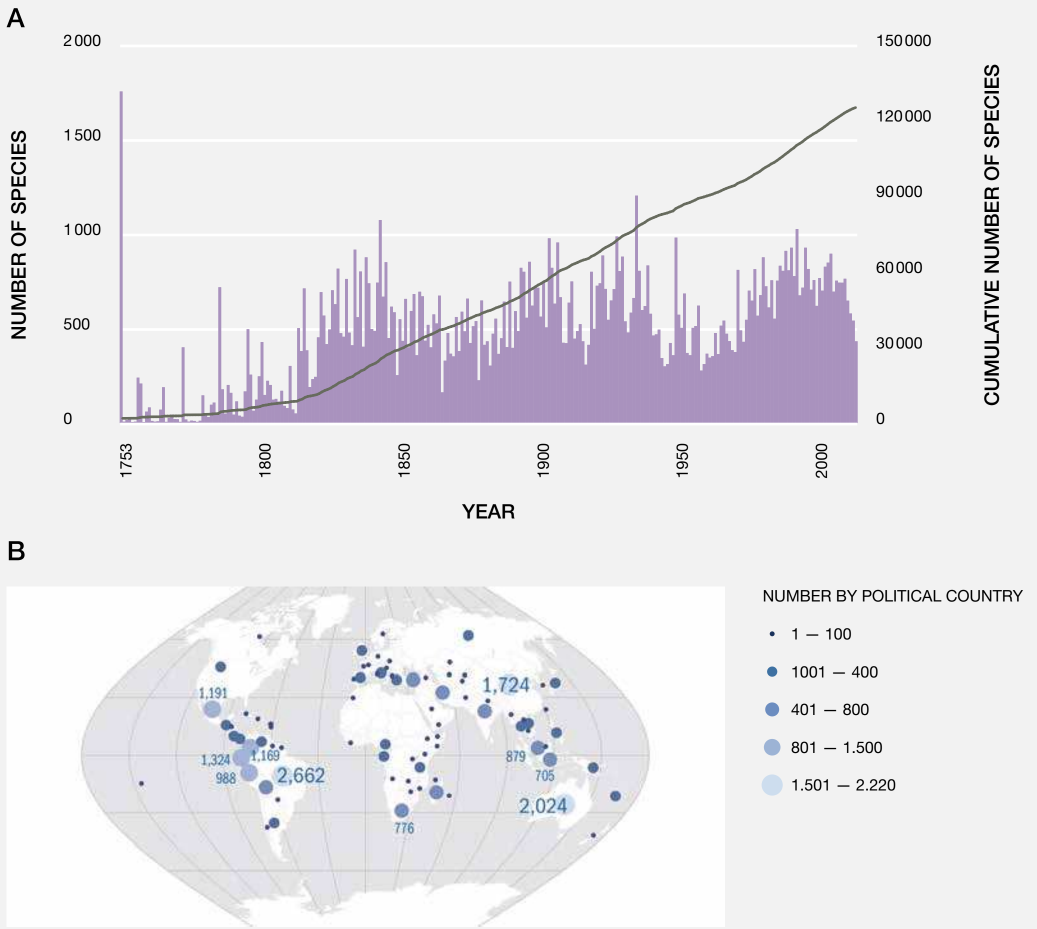
Mobility of biodiversity data. Progress in the detection of the impacts of climate change on biodiversity, conservation gaps, and areas with high concentrations of invasive species today depends heavily on georeferenced biodiversity occurrence data. Overall, 50% of georeferenced online occurrence data in the Global Biodiversity Information Facility pertains to the Americas. However, the density of georeferenced data varies widely among subregions **(Figure 3.36)** (and between countries within each subregion – not shown). Causes include differences in intrinsic richness among countries, a greater level of collaboration between foreign institutions and the tropical countries, differences in exploration intensity, lack of manpower to digitalize biodiversity data and some reticence still on the part of some institutions to incorporate their biodiversity data into the Global Biodiversity Information Facility. The South American subregion lags behind, but important efforts are getting underway. For example, specimens from several institutions in Argentina, thanks to support by the Argentinian National Science Council, can now be found in the Global Biodiversity Information Facility. Brazil is creating the Brazilian Information System on Biodiversity and the *"Portal da Biodiversidade"* which are first steps to consolidate biodiversity data and make it available online. The Chilean national science council is contemplating making it compulsory for grant-holders to place biodiversity data collected with national research funds in the Global Biodiversity Information Facility.

Importantly, efforts are being made to build comprehensive alien species databases at the country (e.g. USA, Brazil, Mexico, Chile) and regional (e.g. Invasives Information Network) levels. Not having access to all biodiversity data, in addition to hindering research progress, introduces uncertainty in the results of regional and global-scale studies that rely heavily on occurrence data and lowers the quality of environmental impact studies within countries.

Biome and ecosystem-level data. With very few exceptions, we currently lack accurate knowledge of biodiversity at the biome level. Where available, the information is limited to a few groups of better-known organisms and does not necessarily coincide with the spatial delineation of the World Wildlife Fund terrestrial biomes adopted by the assessment (see Chapter 1). These have been major obstacles in this assessment. Overall, studies, when present, are insufficient in number for performing biome-level meta-analyses. Thus the assessments of the units analysis in Chapter

Figure ③ ㉟ **Sources of new vascular plant species names entered into the International Plant Names Index.**

Ⓐ The number of plant species (basionyms) described per year from 1753 to 2015 for the Americas, and the cumulative number of accepted species.
Ⓑ Sources of new vascular plant species names entered into the International Plant Names Index between 2004 and 2016 for different countries. Source: Willis (2016) Original data as in updated for the years 2004 to 2016, Ulloa Ulloa *et al.* (2017).

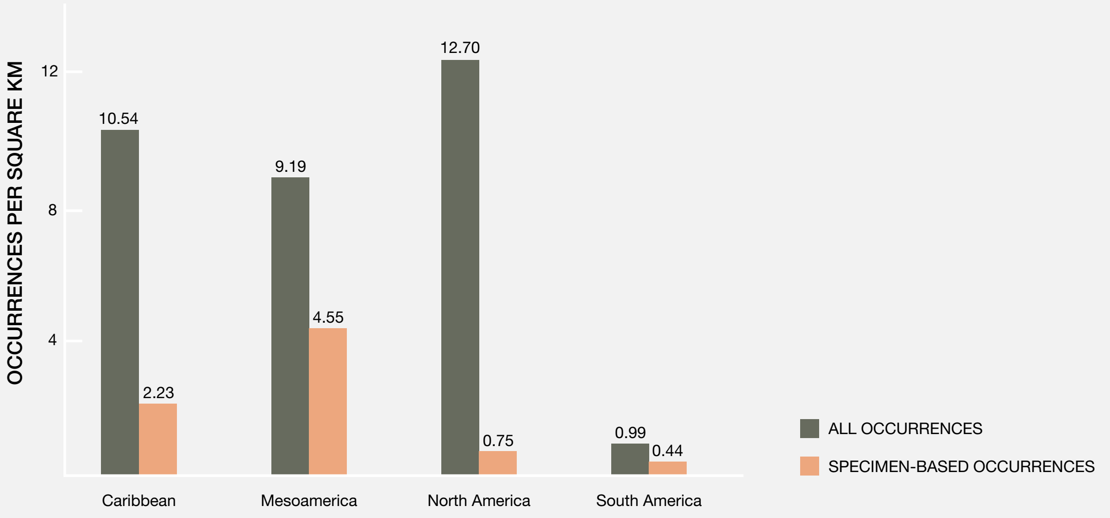


3 are necessarily descriptive and piecemeal. Revision of the World Wildlife Fund biomes based on a consensus is highly desirable now that more accurate vegetation mapping is possible and can be combined with verified species distribution data. If all countries were to adopt such a system, this would be an enormous step forward. One reason for a lack of biome-level data is that many biomes in the Americas cross country boundaries. For example, high elevation systems in South American are found in seven countries and span about 44 degrees of latitude, Mesoamerican dry tropical forest stretches over seven countries and the Amazonian basin over eight. This transnational problem is far less acute in the North American subregion composed of only three countries. Because governments are usually first concerned with the biodiversity of their respective countries, resources for undertaking cross-country, biome-level surveys are generally lacking, but of course, this is not the only reason. This represents a serious challenge for future regional and global IPBES assessments and undermines the efficiency of conservation measures in biomes.

Data on population sizes and genetic diversity is scarce outside the North American subregion. Likewise, long-term series data are few and far between making it difficult to detect temporal trends. Throughout the Americas, fishes and invertebrates differ in their population status, yet the exploitation status of many species is unknown across several taxonomic groups, in particular, elasmobranchs (sharks, skates and rays) and coastal fishes because of a lack of long-term series data. For terrestrial habitats, in the early 1990s, pioneering efforts in the US Long-Term

Figure 3 36 **Quantity of georeferenced biodiversity occurrences in the Global Biodiversity Information Facility for subregions of the Americas divided by land area.**

Note: Occurrence data for North America includes Hawaii. Source: GBIF (http://www.gbif.org/occurrence). Data accessed: March 26, 2017.



Ecological Research Network led to the International Long Term Ecological Research Network (Vanderbilt & Gaiser, 2017). Although many formally accredited sites are found in the Americas, these are strongly concentrated in the USA, Mexico, and Brazil. There are no high altitude International Long Term Ecological Research sites along the entire length of the high Andes where global warming is occurring faster than in adjacent lowlands. Nevertheless, the GLORIA program (www.mountainstudies.org/climate-change) has been active in setting up monitoring sites in the northern and central Andes, to be extended now to the southern Andes. For the marine domain, two North American marine sites were recently accredited by International Long Term Ecological Research Network.

Biodiversity-ecosystem functions-NCP linkages. Most work in this area in the Americas comes from the North American subregion and has involved plot-based studies with a strong focus on productivity. Some information exists in the agricultural, fisheries, pollination, and hydrological domains in the other subregions. Across the Americas, vascular plants comprise the only taxonomic group for which the coverage of functional trait data is abundant (Kattge *et al.,* 2011), yet gaps in functional trait data are highest precisely where diversity tends to be highest: i.e., tropical latitudes (Jetz *et al.,* 2016). Studies linking biodiversity and other less tangible kinds of NCP are incipient throughout. The health benefits of biodiversity and level of equity in terms of access to green areas in urban areas, for example, are fairly open fields. A major gap in our understanding, perhaps with the exception of carbon storage, are links between biodiversity and ecosystem services or NCP at large spatial scales. This requires replicated information across individual biomes/units of analysis and hence coordinated research, often in several countries. To advance in our knowledge here, also, greater collaboration between the traditional biodiversity research community and other disciplines is desirable.
Two major challenges for the future in the Americas are to standardize information and to make it available in a template that is usable by decision makers. In this sense, initiatives such as the Biodiversity Indicators Partnership (https://www.bipindicators.net/), which make suites of global indicators available to support national-level reporting and/or National Biodiversity Strategies and Action Plans updating and implementation, are promising.

3.7 CONCLUDING REMARKS

Biodiversity is linked to ecosystem functions and is highly relevant to NCP across the ecologically diverse and species-rich Americas. All units of analysis of the Americas considered contribute to human well-being. However, Tropical and subtropical moist forests, Temperate and boreal forests and woodlands, Tropical and subtropical dry forests, Mediterranean forests, woodlands and scrub, and Tundra and high elevation habitats stand out as particularly critical for NCP delivery. For aquatic systems, freshwater is considered somewhat more important for NCP than marine. Except in a limited number of cases, this chapter shows that the biodiversity in the Americas´ terrestrial biomes and freshwater and marine habitats continues to undergo serious erosion. The introduction and spread of alien species can be expected to continue causing direct and indirect impacts on human well-being and biodiversity. The subregions currently undergoing most dramatic land use change, considering their spatial extent, are South America and Mesoamerica, where conversion of vegetation to support pastures, agriculture and exotic plantation forestry is widespread. These changes are leading to major losses of habitat with concomitant population and species declines. In the marine and freshwater realms, the number of threatened species is high, and many fish species are over-exploited.

Climate change has begun to affect the distribution of biodiversity, but to a greater degree in North America than South America for the moment. Increased fire frequency in several biomes constitutes a growing threat. Despite significant progress in developing protective measures for the land and in the sea, they are often insufficient. The greatest challenges to policymakers and decision makers will be to: arrest or slow habitat loss; encourage more ecologically-friendly management practices to ensure long-term food- and water-security; and promote alternative biodiversity-based economic activities that are less destructive than current activities. These are not new challenges. Progress necessarily implies a conscious, collective societal effort. Many lessons can be learned from indigenous peoples who have succeeded in living in harmony on the land.

REFERENCES

Aber, A., & Ferrari, G. (2010). *Lineamientos para la gestión nacional de especies exóticas invasoras*. Montevideo, Uruguay: Comité Nacional de Especies Exóticas Invasoras, DINAMA (Dirección Nacional de Medio Ambiente), and UNESCO. Retrieved from http://unesdoc.unesco.org/images/0019/001906/190691s.pdf

Ackerly, D., Stock, W. D., & Slingsby, J. (2014). Geography, climate, and biodiversity: the history and future of Mediterranean-type ecosystems. In N. Allsopp, J. F. Colville, & G. A. Verboom (Eds.), *Fynbos: Ecology, Evolution, and Conservation of a Megadiverse Region* (pp. 361–375). Oxford, UK: Oxford University Press.

Ackerman, D., & Breen, A. (2016). Infrastructure development accelerates range expansion of trembling aspen (*Populus tremuloides*, Salicaceae) into the Arctic. *Arctic*, *69*(2), 130–136. http://doi.org/10.14430/arctic4560

Adams, C. D. (1997). Regional overview: Caribbean islands. In S. D. Davis, V. H. Heywood, O. Herrera-Macbryde, J. Villa-Lobos, & A. C. Hamilton (Eds.), *Centres of Plant Diversity. A Guide and Strategy for their Conservation. Vol. 3. The Americas.* (pp. 233–258). Cambridge, UK: IUCN and WWF.

Agard, J., Kishore, R., & Bayne, B. (1992). *Perna viridis* (Linnaeus, 1758): first record of the Indo-Pacific green mussel (Mollusca: Bivalvia) in the Caribbean. *Caribbean Marine Studies*, *3*, 59–60.

Aguirre-Muñoz, A., & Mendoza, R. (2009). Especies exóticas invasoras: impactos sobre las poblaciones de flora y fauna, los procesos ecológicos y la economía. In R. Dirzo, R. González, & I. J. March (Eds.), *Capital Natural de México, Vol 2: Estado de Conservación y Tendencias de Cambio.* (pp. 277–318). Mexico City, Mexico: CONABIO.

Ahlström, A., Raupach, M. R., Schurgers, G., Smith, B., Arneth, A., Jung, M., Reichstein, M., Canadell, J. G., Friedlingstein, P., Jain, A. K., Kato, E., Poulter, B., Sitch, S., Stocker, B. D., Viovy, N., Wang, Y. P., Wiltshire, A., Zaehle, S., & Zeng, N. (2015). The dominant role of semi-arid ecosystems in the trend and variability of the land CO2 sink. *Science*, *348*(6237), 895–899. https://doi.org/10.1126/science.aaa1668

Aide, T. M., Zimmerman, J. K., Pascarella, J. B., Rivera, L., & Marcano-Vega, H. (2000). Forest regeneration in a chronosequence of tropical abandoned pastures: Implications for restoration ecology. *Restoration Ecology*, *8*(4), 328–338. http://doi.org/10.1046/j.1526-100x.2000.80048.x

Aitken, D., Rivera, D., Godoy-Faúndez, A., & Holzapfel, E. (2016). Water scarcity and the impact of the mining and agricultural sectors in Chile. *Sustainability*, *8*(2), 128. http://doi.org/10.3390/su8020128

Aizen, M. A., Morales, C. L., Vázquez, D. P., Garibaldi, L. A., Sáez, A., & Harder, L. D. (2014). When mutualism goes bad: density-dependent impacts of introduced bees on plant reproduction. *New Phytologist*, *204*(2), 322–328. http://doi.org/10.1111/nph.12924

Alaska Department of Fish and Game (ADF&G). (2016). Hunting, Trapping, and Shooting. Retrieved January 1, 2016, from http://www.adfg.alaska.gov/index.cfm?adfg=hunting.main

Alayo, P. (1974). Los hemípteros acuáticos de Cuba. *Nueva Serie Torreia*, *36*, 9–64.

Alayón García, G. (1999). Biodiversidad de las arañas (Arachnida: Araneae): estado del conocimiento en Cuba. *Cocuyo*, *8*, 3–8.

Aldunate, C., Villagrán, C., Armesto, J. J., & Castro, V. (1983). Ethnobotany of pre-altiplanic community in the Andes of northern Chile. *Economic Botany*, *37*(1), 120–135. http://doi.org/10.1007/BF02859312

Alexander, J. M., Lembrechts, J. J., Cavieres, L. A., Daehler, C., Haider, S., Kueffer, C., Liu, G., McDougall, K., Milbau, A., Pauchard, A., Rew, L. J., & Seipel, T. (2016). Plant invasions into mountains and alpine ecosystems: current status and future challenges. *Alpine Botany*, *126*(2), 89–103. https://doi.org/10.1007/s00035-016-0172-8

Alho, C. J. R. (2011b). Biodiversity of the Pantanal: its magnitude, human occupation, environmental threats and challenges for conservation. *Brazilian Journal of Biology*, *71*(1 Suppl 1), 229–232. http://doi.org/10.1590/S1519-69842011000200001

Alho, C. J. R., Camargo, G., & Fischer, E. (2011a). Terrestrial and aquatic mammals of the Pantanal. *Brazilian Journal of Biology*, *71*(1 Suppl 1), 297–310. http://doi.org/10.1590/S1519-69842011000200009

Alho, C. J. R., Fischer, E., Oliveira-Pissini, L. F., & Santos, C. F. (2011). Bat-species richness in the Pantanal floodplain and its surrounding uplands. *Brazilian Journal of Biology*, *71*(1 suppl 1), 311–320. http://doi.org/10.1590/S1519-69842011000200010

Allen, C. D., Macalady, A. K., Chenchouni, H., Bachelet, D., McDowell, N., Vennetier, M., Kitzberger, T., Rigling, A., Breshears, D. D., Hogg, E. H., Gonzalez, P., Fensham, R., Zhang, Z., Castro, J., Demidova, N., Lim, J.-H., Allard, G., Running, S. W., Semerci, A., & Cobb, N. (2010). A global overview of drought and heat-induced tree mortality reveals emerging climate change risks for forests. *Forest Ecology and Management*, *259*(4), 660–684. https://doi.org/10.1016/j.foreco.2009.09.001

Almeida-Neto, M., Prado, P. I., Kubota, U., Bariani, J. M., Aguirre, G. H., & Lewinsohn, T. M. (2010). Invasive grasses and native Asteraceae in the Brazilian Cerrado. *Plant Ecology*, *209*(1), 109–122. http://doi.org/10.1007/s11258-010-9727-8

Almeida, W. R., Lopes, A. V., Tabarelli, M., & Leal, I. R. (2015). The alien flora of Brazilian Caatinga: deliberate introductions expand the contingent of potential invaders. *Biological Invasions*, *17*(1), 51–56. http://doi.org/10.1007/s10530-014-0738-6

Alonso, A., Muñoz-Carpena, R., Kennedy, R. E., & Murcia., C. (2016).

Wetland landscape spatio-temporal degradation dynamics using the new Google Earth engine cloud-based platform: Opportunities for non-specialists in remote sensing. *Transactions of the ASABE*, *59*(5), 1331–1342. http://doi.org/10.13031/trans.59.11608

Altieri, M. A. (2000). Developing sustainable agricultural systems for small farmers in Latin America. *Natural Resource Forum*, *24*(2), 97–105. http://doi.org/10.1111/j.1477-8947.2000.tb00935.x

Altieri, M. A. (2009). The ecological impacts of large-scale agrofuel monoculture production systems in the Americas. *Bulletin of Science, Technology & Society*, *29*(3), 236–244. http://doi.org/10.1177/0270467609333728

Alvarez, R. (2001). Estimation of carbon losses by cultivation from soils of the Argentine Pampa using the Century Model. *Soil Use and Management*, *17*(2), 62–66. http://doi.org/10.1079/SUM200165

Alvarez, R. (2005). Carbon stocks in pampean soils: a simple regression model for estimation of carbon storage under nondegraded scenarios. *Communications in Soil Science and Plant Analysis*, *36*(11–12), 1583–1589. http://doi.org/10.1081/CSS-200059082

Anacker, B. L. (2014). The nature of serpentine endemism. *American Journal of Botany*, *101*(2), 219–224. http://doi.org/10.3732/ajb.1300349

Anadón-Irizarry, V., Wege, D. C., Upgren, A., Young, R., Boom, B., León, Y. M., Arias, Y., Koenig, K., Morales, A. L., Burke, W., Pérez-Leroux, A., Levy, C., Koenig, S., Gape, L., & Moore, P. (2012). Sites for priority biodiversity conservation in the Caribbean Islands biodiversity hotspot. *Journal of Threatened Taxa*, *4*(8), 2806–2844. https://doi.org/10.11609/JoTT.o2996.2806-44

Anderson-Teixeira, K. J., Snyder, P. K., Twine, T. E., Cuadra, S. V., Costa, M. H., & DeLucia, E. H. (2012). Climate-regulation services of natural and agricultural ecoregions of the Americas. *Nature Climate Change*, *2*(3), 177–181. http://doi.org/10.1038/nclimate1346

Anderson, P. K., Cunningham, A. A., Patel, N. G., Morales, F. J., Epstein, P. R., & Daszak, P. (2004). Emerging infectious diseases of plants: Pathogen pollution, climate change and agrotechnology drivers. *Trends in Ecology and Evolution*, *19*(10), 535–544. http://doi.org/10.1016/j.tree.2004.07.021

Andréassian, V. (2004). Waters and forests: from historical controversy to scientific debate. *Journal of Hydrology*, *291*(1), 1–27. http://doi.org/10.1016/j.jhydrol.2003.12.015

Andresen, C. G., & Lougheed, V. L. (2015). Disappearing Arctic tundra ponds: fine-scale analysis of surface hydrology in drained thaw lake basins over a 65 year period (1948–2013). *Journal of Geophysical Research: Biogeosciences*, *120*(3), 466–479. http://doi.org/10.1002/2014JG002778

Antunes, A. P., Fewster, R. M., Venticinque, E. M., Peres, C. A., Levi, T., Rohe, F., & Shepard, G. H. (2016). Empty forest or empty rivers? A century of commercial hunting in Amazonia. *Science Advances*, *2*(10), e1600936. http://doi.org/10.1126/sciadv.1600936

Aponte, H., & Cano, A. (2013). Estudio florístico comparativo de seis humedales de la costa de Lima (Perú): actualización y nuevos retos para su conservación. *Revista Latinoamericana de Conservación*, *3*(2), 15–27.

Aravena, J. C., Lara, A., Wolodarsky-Franke, A., Villalba, R., & Cuq, E. (2002). Tree-ring growth patterns and temperature reconstruction from *Nothofagus pumilio* (Fagaceae) forests at the upper tree line of southern Chilean Patagonia. *Revista Chilena de Historia Natural*, *75*(2), 361–376. http://doi.org/10.4067/S0716-078X2002000200008

Arbeláez-Cortés, E., Marín-Gómez, O. H., Baena-Tovar, O., & Ospina-González, J. C. (2011). Aves, Finca Estrella de Agua - Páramo de Frontino, municipality of Salento, Quindío, Colombia. *Check List*, *7*(1), 064–070. http://doi.org/10.15560/7.1.64

Arias-González, J. E., Rodríguez-Peña, O. N., Almeida-Leñero, L., Hernández Almeida, O. U., & Schmitter-Soto, J. J. (2016). Cambios en la biodiversidad y sus consecuencias en el funcionamiento de los ecosistemas y sus servicios. In P. Balvanera, E. Arias-González, R. Rodríguez-Estrella, L. Almeida-Leñero, & J. J. Schmitter-Soto (Eds.), *Una Mirada al Conocimiento de los Ecosistemas de México* (pp. 191–228). Mexico City, Mexico: Universidad Nacional Autónoma de México.

Arino, O., Ramos Perez, J. J., Kalogirou, V., Bontemps, S., Defourny, P., & Van Bogaert, E. (2012). Global Land Cover Map for 2009 (GlobCover 2009). European Space Agency (ESA), Université catholique de Louvain (UCL), PANGAEA. http://doi.org/10.1594/PANGAEA.787668

Armas, C., Gutiérrez, J. R., Kelt, D. A., & Meserve, P. L. (2016). Twenty-five years of research in the north-central Chilean semiarid zone: The Fray Jorge Long-Term Socio-Ecological Research (LTSER) site and Norte Chico. *Journal of Arid Environments*, *126*, 1–6. http://doi.org/10.1016/j.jaridenv.2015.12.008

Armenteras, D., Gast, F., & Villareal, H. (2003). Andean forest fragmentation and the representativeness of protected natural areas in the eastern Andes, Colombia. *Biological Conservation*, *113*(2), 245–256. http://doi.org/10.1016/S0006-3207(02)00359-2

Armenteras, D., Rodríguez, N., & Retana, J. (2009). Are conservation strategies effective in avoiding the deforestation of the Colombian Guyana Shield? *Biological Conservation*, *142*(7), 1411–1419. http://doi.org/10.1016/j.biocon.2009.02.002

Armenteras, D., & Rodríguez Eraso, N. (2014). Forest deforestation dynamics and drivers in Latin America: a review since 1990. *Colombia Forestal*, *17*(2), 233–246. http://doi.org/10.14483/udistrital.jour.colomb.for.2014.2.a07

Armesto, J. J. (2009). Annual reports to council Ecological Society of America August 2009. *Bulletin of the Ecological Society of America*, *90*(4), 360–431. http://doi.org/10.1890/0012-9623-90.4.360

Armesto, J. J., Rozzi, R., Smith-Ramirez, C., & Arroyo, M. T. K. (1998). Conservation targets in South American temperate forests. *Science*, *282*(5392), 1271–1272. http://doi.org/10.1126/science.282.5392.1271

Armesto, J. J., Villagrán, C., & Arroyo, M. K. (1996). *Ecología de los Bosques Nativos de Chile*. Santiago, Chile: Editorial Universitaria.

Aronson, M. F. J., La Sorte, F. A., Nilon, C. H., Katti, M., Goddard, M. A., Lepczyk, C. A., Warren, P. S., Williams, N. S. G., Cilliers, S., Clarkson, B., Dobbs, C., Dolan, R., Hedblom, M., Klotz, S., Kooijmans, J. L., Kühn, I., MacGregor-Fors, I., McDonnell, M. J., Mörtberg, U., Pyšek, P., Siebert, S., Sushinsky, J. R., Werner, P., & Winter, M. (2014). A global analysis of the impacts of urbanization on bird and plant diversity reveals key anthropogenic drivers. *Proceedings of the Royal Society B: Biological Sciences*, *281*(1780). https://doi.org/10.1098/rspb.2013.3330

Arredondo Moreno, T., & Huber-Sannwald, E. (2011). Impacts of drought on agriculture in Northern Mexico. In H. G. Brauch, U. Oswald Spring, C. Mesjasz, J. Grin, P. Kameri-Mboute, B. Chourou, P. Dunay, J. Birkmann (Eds.), *Coping with Global Environmental Change, Disasters and Security* (pp. 875–891). Berlin, Heidelberg: Springer-Verlag. Retrieved from http://link.springer.com/chapter/10.1007/978-3-642-17776-7_51

Arriaga, L. (2009). Implicaciones del cambio de uso de suelo en la biodiversidad de los matorrales xerófilos: un enfoque multiescalar. *Investigación Ambiental: Ciencia Y Política Pública*, *1*(1), 6–16.

Arriaga, L., Castellanos, A. E., Moreno, E., & Alarcón, J. (2004). Potential ecological distribution of alien invasive species and risk assessment: a case study of buffel grass in arid regions of Mexico. *Conservation Biology*, *18*(6), 1504–1514. http://doi.org/10.1111/j.1523-1739.2004.00166.x

Arroyo, M. T. K., & Cavieres, L. (1997). The Mediterranean-type climate flora of Central Chile - What do we know and how can we assure its protection. *Noticiero de Biología*, *5*(2), 48–56.

Arroyo, M. T. K., & Cavieres, L. A. (2013). High-elevation Andean ecosystems. In S. Levin (Ed.), *Encyclopedia of Biodiversity* (pp. 96–110). New Jersey, USA: Elsevier Science and Technology.

Arroyo, M. T. K., Maricorena, C., Matthei, O., & Cavieres, L. (2000). Plant invasions in Chile: present patterns and future predictions. In H. A. Mooney & R. Hobbs (Ed.), *Invasive species in a changing world* (pp. 385–421). New York, USA: Island Press.

Arroyo, M. T. K., Marticorena, C., Matthei, O., Muñoz, M., & Pliscoff, P. (2002). Analysis of the contribution and efficiency of the Santuario de la Naturaleza Yerba Loca, 33° S in protecting the regional vascular plant flora (Metropolitan and Fifth regions of Chile). *Revista Chilena de Historia Natural*, *75*(4), 767–792. http://doi.org/10.4067/S0716-078X2002000400012

Arroyo, M. T. K., Primack, R., & Armesto, J. (1982). Community studies in pollination ecology in the high temperate Andes of central Chile. I. Pollination mechanisms and altitudinal variation. *American Journal of Botany*, *69*(1), 82–97. http://doi.org/10.2307/2442833

Arroyo, M. T. K., Riveros, M., Penaloza, A., Cavieres, L., & Faggi, A. M. (1996). Phytogeographic relationships and regional richness patterns of the cool temperate rainforest flora of southern South America. In R. G. Lawford, E. Fuentes, & P. B. Alaback (Eds.), *High-Latitude Rainforests and Associated Ecosystems of the West Coast of the Americas.* (pp. 134–172). New York, USA: Springer.

Asner, G. P., & Archer, S. R. (2010). Livestock and the global carbon cycle. In H. Steinfeld, H. A. Mooney, F. Schneider, & L. E. Neville (Eds.), *Livestock in a Changing Landscape: Volume 1, Drivers, Consequences, and Responses* (pp. 69–82). Washington, DC., USA: Island Press.

Asner, G. P., Rudel, T. K., Aide, T. M., Defries, R., & Emerson, R. (2009). A contemporary assessment of change in humid tropical forests. *Conservation Biology*, *23*(6), 1386–1395. http://doi.org/10.1111/j.1523-1739.2009.01333.x

Association for Temperate Agroforestry (AFTA). (2017). What is Agroforestry? Retrieved from www.aftaweb.org/about/what-is-agroforestry.html

Astorga, A., Fernández, M., Boschi, E. E., & Lagos, N. (2003). Two oceans, two taxa and one mode of development: latitudinal diversity patterns of South American crabs and test for possible causal processes. *Ecology Letters*, *6*(5), 420–427. http://doi.org/10.1046/j.1461-0248.2003.00445.x

Ataroff, M., & Rada, F. (2000). Deforestation impact on water dynamics in a Venezuelan Andean cloud forest. *Ambio*, *29*(7), 440–444. http://doi.org/10.1579/0044-7447-29.7.440

Auerbach, N. A., Walker, M. D., & Walker, D. A. (1997). Effects of roadside disturbance on substrate and vegetation properties in arctic tundra. *Ecological Applications*, *7*(1), 218–235. http://doi.org/10.1890/1051-0761(1997)007%5B0218:EORDOS%5D2.0.CO;2

Austin, A. T., Piñeiro, G., & Gonzalez-Polo, M. (2006). More is less: Agricultural impacts on the N cycle in Argentina. *Biogeochemistry*, *79*, 45–60. http://doi.org/10.1007/s10533-006-9002-1

Bacheler, N. M., Neal, J. W., & Noble, R. L. (2004). Diet overlap between native bigmouth sleepers (*Gobiomorus dormitor*) and introduced predatory fishes in a Puerto Rico reservoir. *Ecology of Freshwater Fish*, *13*(2), 111–118. http://doi.org/10.1111/j.1600-0633.2004.00040.x

Baena, M. L., & Halffilter, G. (2008). Extinción de especies. In J. Sarukhán (Ed.), *Capital Natural de México, Vol. I: Conocimiento Actual de la Biodiversidad* (pp. 263–282). Mexico City, Mexico: CONABIO.

Baldwin, B. G. (2014). Origins of plant diversity in the California Floristic Province. *Annual Review of Ecology, Evolution, and Systematics*, *45*(1), 347–369. http://doi.org/10.1146/annurev-ecolsys-110512-135847

Baldwin, B. G., Goldman, D. H., Keil, D. J., Patterson, R., Rosatti, T. J., & Wilken, D. H. (Eds.). (2012). *The Jepson Manual: Vascular Plants of California* (2nd ed.). Berkeley, USA: University California Press.

Ballinger, R. E., & Congdon, J. D. (1996). Status of the bunch grass lizard, *Sceloporus scalaris*, in the Chiricahua mountains of southeastern Arizona. *Bulletin of the Maryland Herpetological Society*, *32*, 67–69.

Balvanera, P., Pfisterer, A. B., Buchmann, N., He, J.-S., Nakashizuka, T., Raffaelli, D., & Schmid, B. (2006). Quantifying the evidence for biodiversity effects on ecosystem functioning and services. *Ecology Letters*, *9*(10), 1146–1156. http://doi.org/10.1111/j.1461-0248.2006.00963.x

Balvanera, P., Siddique, I., Dee, L., Paquette, A., Isbell, F., Gonzalez, A., Byrnes, J., O'Connor, M. I., Hungate, B. A., & Griffin, J. N. (2014). Linking biodiversity and ecosystem services: Current uncertainties and the necessary next steps. *BioScience*, *64*, 49–57. https://doi.org/10.1093/biosci/bit003

Banda-R, K., Delgado-Salinas, A., Dexter, K. G., Linares-Palomino, R., Oliveira-Filho, A., Prado, D., Pullan, M., Quintana, C., Riina, R., Rodriguez M., G. M., Weintritt, J., Acevedo-Rodriguez, P., Adarve, J., Alvarez, E., Aranguren B., A., Arteaga, J. C., Aymard, G., Castano, A., Ceballos-Mago, N., Cogollo, A., Cuadros, H., Delgado, F., Devia, W., Duenas, H., Fajardo, L., Fernandez, A., Fernandez, M. A., Franklin, J., Freid, E. H., Galetti, L. A., Gonto, R., Gonzalez-M., R., Graveson, R., Helmer, E. H., Idarraga, A., Lopez, R., Marcano-Vega, H., Martinez, O. G., Maturo, H. M., McDonald, M., McLaren, K., Melo, O., Mijares, F., Mogni, V., Molina, D., Moreno, N. d. P., Nassar, J. M., Neves, D. M., Oakley, L. J., Oatham, M., Olvera-Luna, A. R., Pezzini, F. F., Dominguez, O. J. R., Rios, M. E., Rivera, O., Rodriguez, N., Rojas, A., Sarkinen, T., Sanchez, R., Smith, M., Vargas, C., Villanueva, B., & Pennington, R. T. (2016). Plant diversity patterns in neotropical dry forests and their conservation implications. *Science*, *353*(6306), 1383–1387. https://doi.org/10.1126/science.aaf5080

Barnes, M. (2015). Aichi targets: Protect biodiversity, not just area. *Nature*, *526*(7572), 195–195. http://doi.org/10.1038/526195e

Barreto, R., Ferretti, F., Flemming, J. M., Amorim, A., Andrade, H., Worm, B., & Lessa, R. (2016). Trends in the exploitation of South Atlantic shark populations. *Conservation Biology*, *30*(4), 792–804. http://doi.org/10.1111/cobi.12663

Barrett, K., & Guyer, C. (2008). Differential responses of amphibians and reptiles in riparian and stream habitats to land use disturbances in western Georgia, USA. *Biological Conservation*, *141*(9), 2290–2300. http://doi.org/10.1016/j.biocon.2008.06.019

Barrett, K., Helms, B. S., Guyer, C., & Schoonover, J. E. (2010). Linking process to pattern: Causes of stream-breeding amphibian decline in urbanized watersheds. *Biological Conservation*, *143*, 1998–2005. http://doi.org/10.1016/j.biocon.2010.05.001

Barros, A., & Pickering, C. M. (2014). Non-native plant invasion in relation to tourism use of Aconcagua Park, Argentina, the highest protected area in the Southern Hemisphere. *Mountain Research and Development*, *34*(1), 13–26. http://doi.org/10.1659/MRD-JOURNAL-D-13-00054.1

Bartolomé, M. A. (2006). *Procesos Interculturales. Antropología Política del Pluralismo Cultural en América Latina*. México, D.F., México: Siglo XXI.

Baruch, Z., & Nozawa, S. (2014). Abandoned coffee plantations: biodiversity conservation or path for non-native species ? Case study in a neotropical montane forest. *Interciencia*, *39*(8), 554–561. Retrieved from https://search.proquest.com/docview/1564773458?accountid=14621

Bawa, K. S., Kress, W. J., Nadkarni, N. M., Lele, S., Raven, P. H., Janzen, D. H., Lugo, A. E., Ashton, P. S., & Lovejoy, T. E. (2004). Tropical ecosystems into the 21st century. *Science*, *306*, 227–228. https://doi.org/10.1126/science.306.5694.227b

Beckage, B., Osborne, B., Gavin, D. G., Pucko, C., Siccama, T., & Perkins, T. (2008). A rapid upward shift of a forest ecotone during 40 years of warming in the Green Mountains of Vermont. *PNAS*, *105*, 4197–4202. http://doi.org/10.1073/pnas.0708921105

Beddow, J. M., Pardey, P. G., Koo, J., & Wood, S. (2010). The changing landscape of global agriculture. In J. M. Alston, B. A. Babcock, & P. G. Pardey (Eds.), *The Shifting Patterns of Agricultural Production and Productivity Worldwide*. Ames, USA: Iowa State University, The Midwest Agribusiness Trade Research and Information Center (MATRIC).

Beever, E. A., Perrine, J. D., Rickman, T., Flores, M., Clark, J. P., Waters, C. & Goehring, K. E. (2016). Pika (Ochotona princeps) losses from two isolated regions reflect temperature and water balance, but reflect habitat area in a mainland region. *Journal of Mammalogy*, *97*(6), 1495–1511. http://doi.org/10.1093/jmammal/gyw128

Beever, E. A., Ray, C., Wilkening, J. L., Brussard, P. F., & Mote, P. W. (2011). Contemporary climate change alters the pace and drivers of extinction. *Global Change Biology*, *17*(6), 2054–2070. http://doi.org/10.1111/j.1365-2486.2010.02389.x

Bellard, C., Cassey, P., & Blackburn, T. M. (2016). Alien species as a driver of recent extinctions. *Biology Letters*, *12*(2), 20150623. http://doi.org/10.1098/rsbl.2015.0623

Bello, C., Galetti, M., Pizo, M. A., Magnago, L. F. S., Rocha, M. F., Lima, R. A. F., Peres, C. A., Ovaskainen, O., & Jordano, P. (2015). Defaunation affects carbon storage in tropical forests. *Science Advances*, *1*(11), e1501105. https://doi.org/10.1126/sciadv.1501105

Bello, J. C., Báez, M., Gomez, M. F., Orrego, O., & Nägele, L. (Eds.). (2014). *Biodiversidad 2014. Estado y Tendencias de la Biodiversidad Continental de Colombia*. Bogotá, D.C., Colombia: Instituto Alexander von Humboldt. Retrieved from http://www.humboldt.org.co/es/estado-de-los-recursos-naturales/item/898-bio2015

Belnap, J., & Eldridge, D. (2003). Disturbance and recovery of biological soil crusts. In J. Belnap & O. L. Lange (Eds.), *Biological Soil Crusts: Structure, Function, and Management* (pp. 363–383). Berlin, Germany: Springer-Verlag.

Belnap, J., Weber, B., & Büdel, B. (2016). Biological soil crusts as an organizing principle in drylands. In J. Weber, B, Büdel, B, Belnap (Ed.), *Biological Soil Crusts: An Organizing Principle in Drylands* (pp. 3–13). Switzerland: Springer International Publishing. Retrieved from http://link.springer.com/10.1007/978-3-319-30214-0_1

Benayas, R. J. M., Newton, A. C., Diaz, A., & Bullock, J. M. (2009). Enhancement of biodiversity and ecosystem services by ecological restoration: a meta-analysis. *Science*, *325*(5944), 1121–1124. http://doi.org/10.1126/science.1172460

Beneyto, D., Monros, J. S., & Piculo, R. (2013). The Raffia-swamps as sources or sinks of avifauna: a first approach to the problem. *Revista de Biologia Tropical*, *61*(supl. 1), 131–142. http://doi.org/10.15517/rbt.v61i1.23184

Beninde, J., Veith, M., & Hochkirch, A. (2015). Biodiversity in cities needs space: a meta-analysis of factors determining intra-urban biodiversity variation. *Ecology Letters*, *18*, 581–592. http://doi.org/10.1111/ele.12427

Bennett, E., & Thorpe, A. (2008). Review of river fisheries valuation in Central and South America. In A. Neiland & C. Béné (Eds.), *Tropical River Fisheries Valuation: Background Papers to a Global Synthesis* (pp. 3–44). Penang, Malaysia: The WorldFish Center Studies and Reviews.

Benson, A. J. (2012). The Exotic Zebra Mussel. U.S. Fish and Wildlife Service. Retrieved from http://www.fws.gov/midwest/endangered/clams/zebra.html.

Benson, A. J., & Boydstun, C. P. (1999). Documenting over a century of aquatic introductions in the United States. In R. Claudi & J. H. Leach (Eds.), *Nonindigenous Freshwater Organisms: Vectors, Biology, and Impacts* (pp. 1–31). Boca Raton, USA: Lewis Publishers.

Benson, A. J., Marelli, D. C., Frischer, M. E., Danforth, J. M., & Williams, J. D. (2001). Establishment of the green mussel *Perna viridis* (Linnaeus 1758), (Mollusca: Mytilidae) on the west coast of Florida. *Journal of Shellfish Research*, *20*(1), 21–29.

Benson, J. F., Mahoney, P. J., Sikich, J. A., Serieys, L. E. K., Pollinger, J. P., Ernest, H. B., & Riley, S. P. (2016). Interactions between demography, genetics, and landscape connectivity increase extinction probability for a small population of large carnivores in a major metropolitan area. *Proceedings of the Royal Society B: Biological Sciences*, *283*(1837), 20160957. http://doi.org/10.1098/rspb.2016.0957

Berkes, F. (2008). *Sacred Ecology*. New York, USA: Routledge.

Berkes, F., & Jolly, D. (2002). Adapting to climate change: Social-ecological resilience in a Canadian western arctic community. *Conservation Ecology*, *5*(2), 18. Retrieved from http://www.consecol.org/vol5/iss2/art18/

Berthrong, S. T., Piñeiro, G., Jobbágy, E. G., & Jackson, R. B. (2012). Soil C and N changes with afforestation of grasslands across gradients of precipitation and plantation age. *Ecological Applications*, *22*(1), 76–86. http://doi.org/10.1890/10-2210.1

Bertolin, M. L., Urretavizcaya, M. F., & Defosse, G. E. (2015). Fire emissions and carbon uptake in severely burned lenga beech (Nothofagus pumilio) forests of Patagonia, Argentina. *Fire Ecology*, *11*(1), 32–54. http://doi.org/10.4996/fireecology.1101032

Beuchle, R., Grecchi, R. C., Shimabukuro, Y. E., Seliger, R., Eva, H. D., Sano, E., & Achard, F. (2015). Land cover changes in the Brazilian Cerrado and Caatinga biomes from 1990 to 2010 based on a systematic remote sensing sampling approach. *Applied Geography*, *58*, 116–127. http://doi.org/10.1016/j.apgeog.2015.01.017

Bhatt, U. S., Walker, D. A., Raynolds, M. K., Comiso, J. C., Epstein, H. E., Jia, G., Gens, R., Pinzon, J. E., Tucker, C. J., Tweedie, C. E., & Webber, P. J. (2010). Circumpolar Arctic tundra vegetation change is linked to sea ice decline. *Earth Interactions*, *14*(8), 1–20. https://doi.org/10.1175/2010EI315.1

Bianchi, C. A., & Haig, S. M. (2013). Deforestation trends of tropical dry forests in Central Brazil. *Biotropica*, *45*(3), 395–400. http://doi.org/10.1111/btp.12010

Bilenca, D., & Miñarro, F. (2004). *Identificación de Áreas Valiosas de Pastizal (AVPs) en las Pampas y Campos de Argentina, Uruguay y Sur de Brazil*. Buenos Aires, Argentina: Fundación Vida Silvestre.

BirdLife International. (2015). DataZone. Retrieved from http://www.birdlife.org/datazone/site/search

BirdLife International, & NatureServe. (2012). Bird species distribution maps of the world. Version 2.0. Cambridge, UK and Arlington, USA.

Birx-Raybuck, D. A., Price, S. J., & Dorcas, M. E. (2010). Pond age and riparian zone proximity influence anuran occupancy of urban retention ponds. *Urban Ecosystems*, *13*(2), 181–190. http://doi.org/10.1007/s11252-009-0116-9

Blair, R. B., & Launer, A. E. (1997). Butterfly diversity and human land use: Species assemblages along an urban gradient. *Biological Conservation*, *80*(1), 113–125. http://doi.org/10.1016/S0006-3207(96)00056-0

Bogan, A. E. (2008). Global diversity of freshwater mussels (Mollusca, Bivalvia) in freshwater. *Hydrobiologia*, *595*(1), 139–147. http://doi.org/10.1007/s10750-007-9011-7

Bonilla-Murillo, F., Beneyto, D., & Sasa, M. (2013). Amphibians and reptiles in the swamps dominated by the palm *Raphia taedigera* (Arecaceae) in northeastern Costa Rica. *Revista de Biologia Tropical*, *61*(Suppl. 1), 143–161. http://doi.org/10.15517/rbt.v61i1.23185

Bonoff, M. B., & Janzen, D. H. (1980). Small terrestrial rodents in eleven habitats in Santa Rosa National Park. Costa Rica. *Brenesia*, *17*, 163–174.

Borgnia, M., Vila, B. L., & Cassini, M. H. (2008). Interaction between wild camelids and livestock in an Andean semi-desert. *Journal of Arid Environments*, *72*(12), 2150–2158. http://doi.org/10.1016/j.jaridenv.2008.07.012

Bortolus, A., Schwindt, E., Bouza, P. J., & Idaszkin, Y. L. (2009). A characterization of Patagonian salt marshes. *Wetlands*, *29*(2), 772–780. http://doi.org/10.1672/07-195.1

Bosch, J. M., & Hewlett, J. D. (1982). A review of catchment experiments to determine the effect of vegetation changes on water yield and evapotranspiration. *Journal of Hydrology*, *55*(1–4), 3–23. http://doi.org/10.1016/0022-1694(82)90117-2

Bouchenak-Khelladi, Y., Muasya, A. M., & Linder, H. P. (2014). A revised evolutionary history of Poales: origins and diversification. *Botanical Journal of the Linnean Society*, *175*(1), 4–16. http://doi.org/10.1111/boj.12160

Bour, R. (2008). Global diversity of turtles (Chelonii; Reptilia) in freshwater. *Hydrobiologia*, *595*(1), 593–598. http://doi.org/10.1007/s10750-007-9244-5

Bowman, W. D., & Seastedt, T. R. (Eds.). (2001). *Structure and Function of an Alpine Ecosystem: Niwot Ridge, Colorado*. New York, USA: Oxford University Press.

Bozec, Y.-M., O'Farrell, S., Bruggemann, J. H., Luckhurst, B. E., & Mumby, P. J. (2016). Tradeoffs between fisheries harvest and the resilience of coral reefs. *PNAS*, *113*(16), 4536–4541. http://doi.org/10.1073/pnas.1601529113

Bradley, R. D., Ammerma, L. K., Baker, R. J., Bradley, L. C., Cook, J. A., Dowler, R. C., Jones, C., Schmidley, D. J., Stangl Jr, R., Van Den Bussche, R., & Würsig, B. (2014). Revised checklist of North American mammals North of Mexico. *Occasional Papers Museum of Texas Tech University*, *327*, 1–27.

Bradshaw, C. J. A., Leroy, B., Bellard, C., Roiz, D., Albert, C., Fournier, A., Barbet-Massin, M., Salles, J.-M., Simard, F., & Courchamp, F. (2016). Massive yet grossly underestimated global costs of invasive insects. *Nature Communications*, *7*, 12986. https://doi.org/10.1038/ncomms12986

Brandt, R., Mathez-Stiefel, S.-L., Lachmuth, S., Hensen, I., & Rist, S. (2013). Knowledge and valuation of Andean agroforestry species: the role of sex, age, and migration among members of a rural community in Bolivia. *Journal of Ethnobiology and Ethnomedicine*, *9*, 83. http://doi.org/10.1186/1746-4269-9-83

Brauman, K. A., Daily, G. C., Duarte, T. K., & Mooney, H. A. (2007). The nature and value of ecosystem services: An overview highlighting hydrologic services. *Annual Review of Environment and Resources*, *32*(1), 67–98. http://doi.org/10.1146/annurev.energy.32.031306.102758

Braun, A. C., & Koch, B. (2016). Estimating impacts of plantation forestry on plant biodiversity in southern Chile - a spatially explicit modelling approach. *Environmental Monitoring and Assessment*, *188*(10), 564. http://doi.org/10.1007/s10661-016-5547-1

Brenner, M., Schelske, C. L., & Keenan, L. W. (2001). Historical rates of sediment and nutrient accumulation in marshes of the Upper St. Johns River Basin, Florida. *Journal of Paleolimnology*, *26*(3), 241–257. http://doi.org/10.1023/A:1017578330641

Brick Peres, M., Barreto, R., Lessa, R., Vooren, C., Charvet, P., & Rosa, R. (2012). Heavy fishing puts Brazilian sharks and rays in great trouble. In *6th World Fisheries Congress, Sustainable Fisheries in a Changing World*. Edinburgh, Scotland.

Bridi, R., & Montenegro, G. (2017). The value of chilean honey: Floral origin related to their antioxidant and antibacterial activities. In V. A. A. Toledo (Ed.), *Honey Analysis* (pp. 63–78). InTech, Open Access. Retrieved from https://www.intechopen.com/books/honey-analysis/the-value-of-chilean-honey-floral-origin-related-to-their-antioxidant-and-antibacterial-activities

Briggs, C. J., Vredenburg, V. T., Knapp, R. A., & Rachowicz, L. J. (2005). Investigating the population-level effects of Chytridiomycosis: An emerging infectious disease of amphibieans. *Ecology*, *86*, 3149–3159. http://doi.org/10.1890/04-1428

Bring, A., Fedorova, I., Dibike, Y., Hinzman, L., Mard, J., Mernild, S. H., Prowse, T., Semenova, O., Stuefer, S. L., & Woo, M. K. (2016). Arctic terrestrial hydrology: A synthesis of processes, regional effects, and research challenges. *Journal of Geophysical Research*, *121*(3), 621–649.

Brinson, M. M., & Malvarez, A. I. (2002). Temperate freshwater wetlands: types, status, and threats. *Environmental Conservation*, *29*(2), 115–133. http://doi.org/10.1017/S0376892902000085

Bromberg, K. D., & Bertness, M. D. (2005). Reconstructing New England salt marsh losses using historical maps. *Estuaries*, *28*(6), 823–832. http://doi.org/10.1007/BF02696012

Brooker, R. W., Maestre, F. T., Callaway, R. M., Lortie, C. L., Cavieres, L. A., Kunstler, G., Liancourt, P., Tielbörger, K., Travis, J. M. J., Anthelme, F., Armas, C., Coll, L., Corcket, E., Delzon, S., Forey, E., Kikvidze, Z., Olofsson, J., Pugnaire, F., Quiroz, C. L., Saccone, P., Schiffers, K., Seifan, M., Touzard, B., & Michalet, R. (2008). Facilitation in plant communities: the past, the present, and the future. *Journal of Ecology*, *96*(1), 18–34. https://doi.org/10.1111/j.1365-2745.2007.01295.x

Brooks, M. (2009). Spatial and temporal distribution of nonnative plants in upland areas of the Mojave Desert. In R. Webb, L. Fenstermaker, J. Heaton, D. Hughson, E. McDonald, & D. Miller (Eds.), *The Mojave Desert: Ecosystem Processes and Sustainability* (pp. 101–124). Reno, USA: University of Nevada Press.

Brooks, T. M., Akçakaya, H. R., Burgess, N. D., Butchart, S. H. M., Hilton-Taylor, C., Hoffmann, M., Juffe-Bignoli, D., Kingston, N., MacSharry, B., Parr, M., Perianin, L., Regan, E. C., Rodrigues, A. S. L., Rondinini, C., Shennan-Farpon, Y., & Young, B. E. (2016). Analysing biodiversity and conservation knowledge products to support regional environmental assessments. *Scientific Data*, *3*. https://doi.org/10.1038/sdata.2016.7

Brown, A. D. (1990). Epiphytism in the montane forests of El Rey National Park in Argentina: Floristic composition and distribution pattern. *Revista de Biología Tropical*, *38*(2A), 155–166.

Brown, A. E., Zhang, L., McMahon, T. A., Western, A. W., & Vertessy, R. A. (2005). A review of paired catchment studies for determining changes in water yield resulting from alterations in vegetation. *Journal of Hydrology*, *310*(1–4), 28–61. http://doi.org/10.1016/j.jhydrol.2004.12.010

Brown, M. T., Cohen, M. J., Bardi, E., & Ingwersen, W. W. (2006). Species diversity in the Florida Everglades, USA: A systems approach to calculating biodiversity. *Aquatic Sciences*, *68*, 254–277. http://doi.org/10.1007/s00027-006-0854-1

Brown, S. J., & Maceina, M. J. (2002). The influence of disparate levels of submersed aquatic vegetation on largemouth bass population characteristics in a Georgia reservoir. *Journal of Aquatic Plant Management*, *40*, 28–35.

Bucher, E. (1982). Chaco and Caatinga - South American arid savannas, woodlands and thickets. In B. I. Huntley & B. H. Walker (Eds.), *Ecology of Tropical Savannas* (pp. 48–79). Berlin, Germany: Springer-Verlag.

Bucher, E. H., Echevarria, A. L., Juri, M. D., & Chani, J. M. (2000). Long-term survey of Chilean flamingo breeding colonies on Mar Chiquita lake, Córdoba, Argentina. *Waterbirds: The International Journal of Waterbird Biology*, *23*, 114–118. http://doi.org/10.2307/1522155

Buckley, L. B., & Jetz, W. (2007). Environmental and historical constraints on global patterns of amphibian richness. *Proceedings of the Royal Society B: Biological Sciences*, *274*(1614), 1167–1173. http://doi.org/10.1098/rspb.2006.0436

Buddo, D. S. A., Steele, R. D., & D'Oyen, E. R. (2003). Distribution of the invasive Indo-Pacific green mussel, *Perna viridis*, in Kingston Harbour, Jamaica. *Bulletin of Marine Science*, *73*(2), 433–441.

Bullock, J. M., Pywell, R. F., & Walker, K. J. (2007). Long-term enhancement of agricultural production by restoration of biodiversity. *Journal of Applied Ecology*, *44*(1), 6–12. http://doi.org/10.1111/j.1365-2664.2006.01252.x

Burge, D. O., Thorne, J. H., Harrison, S. P., O'Brien, B. C., Rebman, J. P., Shevock, J. R., Alverson, E. R., Hardison, L. K., Rodríguez, J. D., Junak, S. A., Oberbauer, T. A., Riemann, H., Vanderplank, S. E., & Barry, T. (2016). Plant diversity and endemism in the California Floristic Province. *Madroño*, *63*(2), 3–206. https://doi.org/10.3120/madr-63-02-3-206.1

Burkhead, N. M. (2012). Extinction rates in North American freshwater fishes, 1900–2010. *BioScience*, *62*, 798–808. http://doi.org/10.1525/bio.2012.62.9.5

Burridge, L., Weis, J. S., Cabello, F., Pizarro, J., & Bostick, K. (2010). Chemical use in salmon aquaculture: A review of current practices and possible environmental effects. *Aquaculture*, *306*(1–4), 7–23. http://doi.org/10.1016/j.aquaculture.2010.05.020

Buschmann, A. H., Riquelme, V. A., Hernández-González, M. C., Varela, D., Jiménez, J. E., Henríquez, L. A., Vergara, P. A., Guíñez, R., & Filún, L. (2006). A review of the impacts of salmonid farming on marine coastal ecosystems in the southeast Pacific. *ICES Journal of Marine Science*, *63*(7), 1338–1345. https://doi.org/10.1016/j.icesjms.2006.04.021

Bustamante, R. O., & Castor, C. (1998). The decline of an endangered temperate ecosystem: the ruil (*Nothofagus alessandrii*) forest in central Chile. *Biodiversity and Conservation*, *7*(12), 1607–1626. http://doi.org/10.1023/A:1008856912888

Butchart, S. H. M., Scharlemann, J. P. W., Evans, M. I., Quader, S., Aricò, S., Arinaitwe, J., Balman, M., Bennun, L. A., Bertzky, B., Besançon, C., Boucher, T. M., Brooks, T. M., Burfield, I. J., Burgess, N. D., Chan, S., Clay, R. P., Crosby, M. J., Davidson, N. C., de Silva, N., Devenish, C., Dutson, G. C. L., Fernández, D. F. D., Fishpool, L. D. C., Fitzgerald, C., Foster, M., Heath, M. F., Hockings, M., Hoffmann, M., Knox, D., Larsen, F. W., Lamoreux, J. F., Loucks, C., May, I., Millett, J., Molloy, D., Morling, P., Parr, M., Ricketts, T. H., Seddon, N., Skolnik, B., Stuart, S. N., Upgren, A., & Woodley, S. (2012). Protecting important sites for biodiversity contributes to meeting global conservation targets. *PLoS ONE*, *7*(3), e32529. https://doi.org/10.1371/journal.pone.0032529

Buytaert, W., Cuesta-Camacho, F., & Tobón, C. (2011). Potential impacts of climate change on the environmental services of humid tropical alpine regions. *Global Ecology and Biogeography*, *20*, 19–33. http://doi.org/10.1111/j.1466-8238.2010.00585.x

Buytaert, W., Wyseure, G., De Bievre, B., & Deckers, J. (2005). The effect of land-use changes on the hydrological behaviour of Histic Andosols in south Ecuador. *Hydrological Processes*, *19*(20), 3985–3997. http://doi.org/10.1002/hyp.5867

Caballero, J., & Cortés, L. (2001). Percepción, uso y manejo tradicional de los recursos vegetales en México. In B. Rendón Aguilar, S. Rebollar Domínguez, J. Caballero Nieto, & M. A. Martínez Alfaro (Eds.), *Plantas, Cultura y Sociedad* (pp. 79–100). México D.F., México: Universidad Autónoma Metropolitana y Secretaría del Medio Ambiente, Recursos Naturales y Pesca.

Cabette, H. S. R., Souza, J. R., Shimano, Y., & Juen, L. (2017). Effects of changes in the riparian forest on the butterfly community (Insecta: Lepidoptera) in Cerrado areas. *Revista Brasileira de Entomologia*, *61*(1), 43–50. http://doi.org/10.1016/j.rbe.2016.10.004

Cáceres, N. C., Nápoli, R. P., Casella, J., & Hannibal, W. (2010). Mammals in a fragmented savannah landscape in south-western Brazil. *Journal of Natural History*, *44*(7–8), 491–512. http://doi.org/10.1080/00222930903477768

Cadena-Vargas, C. E., & Sarmiento, C. E. (2016). Cambios en las coberturas paramunas. Las Amenazas de los páramos de Colombia. In M. F. Gómez, L. A. Moreno, G. I. Andrade, & C. Rueda (Eds.), *Biodiversidad 2015. Estado y Tendencias de la Biodiversidad Continental de Colombia*. Bogotá, D.C., Colombia: Instituto Alexander von Humboldt.

Califano, L. M., & Echazú, F. (2013). Etnobotánica en comunidades pastoriles: Conocimiento tradicional sobre especies tóxicas para el ganado en la cuenca del río Iruya (Salta, Argentina). *Boletín de La Sociedad Argentina de Botánica*, *48*(2), 365–375.

California Invasive Plant Council. (2017). California Invasive Plant Council. Retrieved from http://cal-ipc.org/

California Natural Resources Agency. (2015). *California Department of Fish and Wildlife*. Retrieved from https://www.wildlife.ca.gov/Conservation/Plants/Invasives

Callaghan, T. V., Björn, L. O., Chernov, Y., Chapin, T., Christensen, T. R., Huntley, B., … Zöckler, C. (2004). Biodiversity, distributions and adaptations of Arctic species in the context of environmental change. *Ambio*, *33*(7), 404–417. http://doi.org/10.1579/0044-7447-33.7.404

Calvo-Alvarado, J., McLennan, B., Sánchez-Azofeifa, A., & Garvin, T. (2009). Deforestation and forest restoration in Guanacaste, Costa Rica: Putting conservation policies in context. *Forest Ecology and Management*, *258*(6), 931–940. http://doi.org/10.1016/j.foreco.2008.10.035

Calvo-Alvarado, J., Sánchez-Azofeifa, A., & Portillo-Quintero, C. (2013). Neotropical seasonally dry forests. In S. Levin (Ed.), *Encyclopedia of Biodiversity* (pp. 488–500). Amsterdam, The Netherlands: Academic Press. Retrieved from http://linkinghub.elsevier.com/retrieve/pii/B9780123847195003543

Cameron, R. W., Blanuša, T., Taylor, J. E., Salisbury, A., Halstead, A. J., Henricot, B., & Thompson, K. (2012). The domestic garden – Its contribution to urban green infrastructure. *Urban Forestry & Urban Greening*, *11*(2), 129–137. http://doi.org/10.1016/j.ufug.2012.01.002

Canfield, D. E., Glazer, A. N., & Falkowski, P. G. (2010). The evolution and future of Earth's nitrogen cycle. *Science*, *330*(6001), 192–196. http://doi.org/10.1126/science.1186120

Cano-Ortiz, A., Musarella, C. M., Piñar-Fuentes, J. C., Pinto-Gomes, C., & Cano-Carmona, E. (2015). Forests and landscapes of Dominican Republic. *British Journal of Applied Science & Technology*, *9*(3), 231–242. http://doi.org/10.9734/BJAST/2015/17507

Carabias, J., Sarukhán, J., De la Maza, J., & Galindo, C. (2010). *Patrimonio Natural de México. Cien Casos de Éxito.* Mexico City, Mexico: México, Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Retrieved from http://www.biodiversidad.gob.mx/pais/cien_casos/pdf/Cien casos.pdf
PLoS ONE, *7*(3), e32529. https://doi.org/10.1371/journal.pone.0032529

Cardinale, B. J., Duffy, J. E., Gonzalez, A., Hooper, D. U., Perrings, C., Venail, P., Narwani, A., Mace, G. M., Tilman, D., A.Wardle, D., Kinzig, A. P., Daily, G. C., Loreau, M., Grace, J. B., Larigauderie, A., Srivastava, D. S., & Naeem, S. (2012). Erratum: Corrigendum: Biodiversity loss and its impact on humanity. *Nature*, *489*(7415), 326–326. https://doi.org/10.1038/nature11373

Cardinale, B. J., Matulich, K. L., Hooper, D. U., Byrnes, J. E., Duffy, E., Gamfeldt, L., Balvanera, P., O'Connor, M. I., & Gonzalez, A. (2011). The functional role of producer diversity in ecosystems. *American Journal of Botany*, *98*(3), 572–592. https://doi.org/10.3732/ajb.1000364

Cardinale, B. J., Wright, J. P., Cadotte, M. W., Carroll, I. T., Hector, A., Srivastava, D. S., Loreau, M., & Weis, J. J. (2007). Impacts of plant diversity on biomass production increase through time because of species complementarity. *PNAS*, *104*(46), 18123–18128. https://doi.org/10.1073/pnas.0709069104

Cardoso, D., Särkinen, T., Alexander, S., Amorim, A. M., Bittrich, V., Celis, M., Daly, D. C., Fiaschi, P., Funk, V. A., Giacomin, L. L., Goldenberg, R., Heiden, G., Iganci, J., Kelloff, C. L., Knapp, S., Cavalcante de Lima, H., Machado, A. F. P., dos Santos, R. M., Mello-Silva, R., Michelangeli, F. A., Mitchell, J., Moonlight, P., de Moraes, P. L. R., Mori, S. A., Nunes, T. S., Pennington, T. D., Pirani, J. R., Prance, G. T., de Queiroz, L. P., Rapini, A., Riina, R., Rincon, C. A. V., Roque, N., Shimizu, G., Sobral, M., Stehmann, J. R., Stevens, W. D., Taylor, C. M., Trovó, M., van den Berg, C., van der Werff, H., Viana, P. L., Zartman, C. E., & Forzza, R. C. (2017). Amazon plant diversity revealed by a taxonomically verified species list. *PNAS*, *114*(40), 10695–10700. https://doi.org/10.1073/pnas.1706756114

Carlton, J. T., Chapman, J. W., Geller, J. B., Miller, J. A., Carlton, D. A., McCuller, M. I., Treneman, N. C., Steves, B. P., & Ruiz, G. M. (2017). Tsunami-driven rafting: Transoceanic species dispersal and implications for marine biogeography. *Science*, *357*(6358), 1402–1406. https://doi.org/10.1126/science.aao1498

Carmignotto, A. P., De Vivo, M., & Langguth, A. (2012). Mammals of the Cerrado and Caatinga: distribution patterns of the tropical open biomes of central South America. In B. D. Patterson & L. P. Costa (Eds.), *Bones, Clones and Biomes. The History and Geography of Recent Neotropical Mammals* (pp. 307–350). Chicago, USA.: University of Chicago Press.

Carvalho, F. M., De Marco, P., & Ferreira, L. G. (2009). The Cerrado into-pieces: Habitat fragmentation as a function of landscape use in the savannas of central Brazil. *Biological Conservation*, *142*(7), 1392–1403.

Casas, A., Moreno-Calles, A. I., Vallejo, M., & Parra, F. (2016). Importancia actual y potencial de los recursos genéticos. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 51–74). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina.

Casas, A., Otero-Arnaiz, A., Perez-Negron, E., & Valiente-Banuet, A. (2007). In situ management and domestication of plants in Mesoamerica. *Annals of Botany*, *100*(5), 1101–1115. http://doi.org/10.1093/aob/mcm126

Casas, A., Parra-Rodinel, F., Rangel-Landa, S., Blancas, J., Vallejo, M., Moreno-Calles, A. I., Guillén, S., Torres-García, I., Delgado-Lemus, A., Pérez-Negrón, E., Figueredo, C. J., Cruse-Sanders, J. M., Farfán-Heredia, B., Solís, L., Aguirre-Dugua, X., Otero-Arnaiz, A., Alvarado-Sizzo, H., & Camou-Guerrero, A. (2017). Manejo y domesticación de plantas en Mesoamérica. Una estrategia de investigación y estado del conocimiento sobre los recursos genéticos en México. In A. Casas, J. Torres-Guevara, & F. Parra-Rondinel (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 69–102). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Casas, A., Parra-Rondinel, F., Torres-García, I., Rangel-Landa, S., Zarazúa, M., & Torres-Guevara, J. (2017). Estudios y patrones continentales de domesticación y manejo de recursos genéticos: perspectivas. In A. Casas, J. Torres-Guevara, & F. Parra-Rondinel (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 537–569). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Casas, A., Parra, F., Blancas, J., Rangel-Landa, S., Vallejo, M., Figueredo, C. J., & Moreno-Calles, A. I. (2016). Origen de la domesticación y la agricultura: cómo y por qué. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 189–224). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina.

Cashman, A. (2014). Water security and services in the Caribbean. *Water*, *6*(5), 1187–1203.http://doi.org/10.3390/w6051187

Casner, K. L., Forister, M. L., O'Brien, J. M., Thorne, J., Waetjen, D., & Shapiro, A. M. (2014). Contribution of urban expansion and a changing climate to decline of a butterfly fauna. *Conservation Biology*, *28*(3), 773–782. http://doi.org/10.1111/cobi.12241

Castello, L., McGrath, D. G., Hess, L. L., Coe, M. T., Lefebvre, P. A., Petry, P., Macedo, M. N., Renó, V. F., & Arantes, C. C. (2013). The vulnerability of Amazon freshwater ecosystems. *Conservation Letters*, *6*(4), 217–229. https://doi.org/10.1111/conl.12008

Castillo, L. E., Ruepert, C., & Solis, E. (2000). Pesticide residues in the aquatic environment of banana plantation areas in the north Atlantic zone of Costa Rica. *Environmental Toxicology and Chemistry*, *19*, 1942–1950. http://doi.org/10.1002/etc.5620190802

Cavalcanti, M. C. B. T., Ramos, M. A., Araújo, E. L., & Albuquerque, U. P. (2015). Implications from the use of non-timber forest products on the consumption of wood as a fuel source in human-dominated semiarid landscapes. *Environmental Management*, *56*(2), 389–401. http://doi.org/10.1007/s00267-015-0510-4

Cavanaugh, K. C., Kellner, J. R., Forde, A. J., Gruner, D. S., Parker, J. D., Rodriguez, W., & Feller, I. C. (2014). Poleward expansion of mangroves is a threshold response to decreased frequency of extreme cold events. *PNAS*, *111*, 723–727. http://doi.org/10.1073/pnas.1315800111

Cavender-Bares, J., & Reich, P. B. (2012). Shocks to the system: community assembly of the oak savanna in a 40-year fire frequency experiment. *Ecology*, *93*(Special Issue), 52–69. http://doi.org/10.1890/11-0502.1

Cavia, R., Cueto, G. R., & Suárez, O. V. (2009). Changes in rodent communities according to the landscape structure in an urban ecosystem. *Landscape and Urban Planning*, *90*(1–2), 11–19. http://doi.org/10.1016/j.landurbplan.2008.10.017

Cavicchia, M., & García, G. V. (2012). Riqueza y composición de especies de aves rapaces (Falconiformes y Strigiformes) de la ciudad de Buenos Aires, Argentina. *Hornero*, *27*(2), 159–166.

Caziani, S. M., Derlindati, E. J., Tálamo, A., Sureda, A. L., Trucco, C. E., & Nicolossi, G. (2001). Waterbird richness in Altiplano wetlands of northwestern Argentina. *Waterbirds*, *24*, 103–117. http://doi.org/10.2307/1522249

Ceballos, G., & Ehrlich, P. R. (2009). Discoveries of new mammal species and their implications for conservation and ecosystem services. *PNAS*, *106*(10), 3841–3846. http://doi.org/10.1073/pnas.0812419106

Ceballos, G., Garcia-Aguayo, A., Rodriguez, P., & Noguera, F. (1993). *Plan de Manejo de la Reserva Ecológica de Chamela-Cuixmala*. México City, Mexico: Fundacion Ecologia de Cuixmala.

Cendrero, A., Terán, D. de, Ramón, J., González, D., Mascitti, V., Rotondaro, R., & Tecchi, R. (1993). Environmental diagnosis for planning and management in the high Andean region: The biosphere reserve of Pozuelos, Argentina. *Environmental Management*, *17*(5), 683–703. http://doi.org/10.1007/BF02393729

CEPAL. (2014). *Panorama Social de America Latina (LC/G.2635-P)*. Santiago, Chile.

Chace, J. F., & Walsh, J. J. (2006). Urban effects on native avifauna: A review. *Landscape and Urban Planning*, *74*(1), 46–69. http://doi.org/10.1016/j.landurbplan.2004.08.007

Chacón, M., Pickersgill, S., & Debouck, D. (2005). Domestication patterns in common bean (*Phaseolus vulgaris* L.) and the origin of the Mesoamerican and Andean cultivated races. *Theoretical and Applied Genetics*, *110*(3), 432–444. http://doi.org/10.1007/s00122-004-1842-2

Challenger, A., & Dirzo, R. (2009). Factores de cambio y estado de la biodiversidad. In J. Sarukhán (Ed.), *Capital Natural de México, Vol. II: Estado de Conservación y Tendencias de Cambio* (pp. 37–73). Mexico City, Mexico: CONABIO.

Challenger, A., & Soberón, J. (2008). Los ecosistemas terrestres. In J. Sarukhán (Ed.), *Capital Natural de México, Vol. I: Conocimiento Actual de la Biodiversidad* (pp. 87–108). Sarukhán, José: CONABIO.

Chapin, F. S., Hoel, M., Carpenter, S. R., Lubchenco, J., Walker, B., Callaghan, T. V, … Zimov, S. A. (2006). Building resilience and adaptation to manage Arctic change. *Ambio*, *35*(4), 198–202. http://doi.org/10.1579/0044-7447(2006)35%5B198:braatm%5D2.0.co;2

Chapin, F. S., Matson, P. A., & Vitousek, P. M. (2012). *Principles of Terrestrial Ecosystem Ecology*. New York, USA: Springer-Verlag.

Chapin, F. S., & Shaver, G. R. (1985). Individualistic growth response of tundra plant species to environmental manipulations in the field. *Ecology*, *66*(2), 564–576. http://doi.org/10.2307/1940405

Chapin, F. S., Shaver, G. R., Giblin, A. E., Nadelhoffer, K. J., & Laundre, J. A. (1995). Responses of arctic tundra to experimental and observed changes in climate. *Ecology*, *76*(3), 694–711. http://doi.org/10.2307/1939337

Chapin, F. S., Sturm, M., Serreze, M. C., McFadden, J. P., Key, J. R., Lloyd, A. H., McGuire, A. D., Rupp, T. S., Lynch, A. H., Schimel, J. P., Beringer, J., Chapman, W. L., Epstein, H. E., Euskirchen, E. S., Hinzman, L. D., Jia, G., Ping, C.-L., Tape, K. D., Thompson, C. D. C., Walker, D. A., & Welker, J. M. (2005). Role of land-surface changes in Arctic summer warming. *Science*, *310*(5748), 657–660. Retrieved from http://science.sciencemag.org/content/310/5748/657.abstract

Chapin, F. S., Zavaleta, E. S., Eviner, V. T., Naylor, R. L., Vitousek, P. M., Reynolds, H. L., Hooper, D. U., Lavorel, S., Sala, O. E., Hobbie, S. E., Mack, M. C., & Díaz, S. (2000). Consequences of changing biodiversity. *Nature*, *405*(6783), 234–242. https://doi.org/10.1038/35012241

Charity, S., Dudley, N., Oliveira, D., & Stolton, S. (Eds.). (2016). *Living Amazon Report 2016: A Regional Approach to Conservation in the Amazon*. Brasília, Brazil and Quito, Ecuador: WWF Living Amazon Initiative.

Chaudhary, A., Burivalova, Z., Koh, L. P., & Hellweg, S. (2016). Impact of forest management on species richness: global meta-analysis and economic trade-offs. *Scientific Reports*, *6*(April), 23954. http://doi.org/10.1038/srep23954

Cherry, J. E., Déry, S. J., Cheng, Y., Stieglitz, M., Jacobs, A. S., & Pan, F. (2014). Climate and hydrometeorology of the Toolik Lake Region and the Kuparuk River Basin: Past, present, and future. In J. E. Hobbie & G. W. Kling (Eds.), *Alaska's Changing Arctic: Ecological Consequences for Tundra, Streams, and Lakes* (pp. 21–60). New York, USA: Oxford University Press.

Chesapeake Bay Program. (2017). Submerged Aquatic Vegetation. Retrieved from http://www.chesapeakebay.net/

Cheung, W. W., Watson, R., & Pauly, D. (2013). Signature of ocean warming in global fisheries catch. *Nature*, *497*(7449), 365–368. http://doi.org/10.1038/nature12156

Childs, J. E., & Seegar, W. S. (1986). Epidemiologic observations on infection with Toxoplasma gondii in three species of urban mammals from Baltimore, Maryland, USA. *International Journal of Zoonoses*, *13*(4), 249–261.

Choi, C. (2008). Tierra del Fuego: the beavers must die. *Nature*, *453*, 968. http://doi.org/10.1038/453968a

Cleland, E. E., Funk, J., & Allen., E. B. (2016). Coastal sage scrub. In E. Zavaleta & H. Mooney (Eds.), *Ecosystems of California* (pp. 429–448). Oakland, USA: University of California Press.

Clement, C., de Cristo-Araujo, M., D'Eeckenbrugge, G. C., Alves-Pereira, A., & Picanço-Rodrigues, D. (2016). 10,000 years of plant domestication: the origins of agrobiodiversity in indigenous Amazonia. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 253–282). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina.

Clement, C. R. (1999). 1492 and the loss of Amazonian crop genetic resources. II. Crop biogeography at contact. *Economic Botany*, *53*, 203–216. http://doi.org/10.1007/BF02866499

Clement, C. R. (2017). Panorama de los recursos genéticos de Brasil, con énfasis en la Amazonía. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 27–42). México D.F. and Lima, Peru: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Clement, C. R., de Cristo-Araújo, M., D'Eeckenbrugge, G. C., Pereira, A. A., & Picanço-Rodrigues, D. (2010). Origin and domestication of native Amazonian crops. *Diversity*, *2*(1), 72–106. http://doi.org/10.3390/d2010072

Cohen, A., & Carlton, J. (1998). Accelerating invasion rate in a highly invaded estuary. *Science*, *279*(5350), 555–558. http://doi.org/10.1126/science.279.5350.555

Collins, F. S. (1998). New goals for the U.S. Human Genome Project: 1998-2003. *Science*, *282*(5389), 682–689. http://doi.org/10.1126/science.282.5389.682

Comité Operativo para la Prevención el Control y la Erradicación de las Especies Exóticas Invasoras (COCEI). (2014). *Estrategia Nacional Integrada para la Prevención, el Control y/o Erradicación de las Especies Exóticas Invasoras*. Santiago, Chile: Ministerio del Medio Ambiente (MMA).

Committee on Taxonomy (COT). (2016). List of marine mammal species and subspecies. Retrieved January 14, 2017, from www.marinemammalscience.org

Conabio. (2014). Biodiversidad Mexicana. http://doi.org/10.2307/3503924

CONABIO. (2016). Sistema de Información sobre Especies Invasoras en México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (Available on request).

CONABIO database. (n.d.). Retrieved from https//www.conabio.gob.mx

CONANP. (2017). Firma el Presidente Enrique Peña Nieto el decreto del Parque Nacional Revillagigedo. Retrieved from https://www.gob.mx/conanp/prensa/firma-el-presidente-enrique-pena-nieto-el-decreto-del-parque-nacional-revillagigedo-136220

Contreras, T. E., Figueroa, J. A., Abarca, L., & Castro, S. A. (2011). Fire regimen and spread of plants naturalized in central Chile. *Revista Chilena de Historia Natural*, *84*(3), 307–323. http://doi.org/10.4067/S0716-078X2011000300001

Cooney, P. B., & Kwak, T. J. (2010). Development of standard weight equations for Caribbean and Gulf of Mexico amphidromous fishes. *North American Journal of Fisheries Management*, *30*(5), 1203–1209. http://doi.org/10.1577/M10-058.1

Cooper, C. B., Loyd, K. A. T., Murante, T., Savoca, M., & Dickinson, J. (2012). Natural history traits associated with detecting mortality within residential bird communities: can citizen science provide insights? *Environmental Management*, *50*(1), 11–20. http://doi.org/10.1007/s00267-012-9866-x

Cornwell, W. K., Schwilk, D. W., & Ackerly, D. D. (2006). A trait-based test for habitat filtering: convex hull volume. *Ecology*, *87*(6), 1465–1471. http://doi.org/10.1890/0012-9658(2006)87%5B1465:ATTFHF%5D2.0.CO;2

Corporación Nacional Forestal (CONAF). (2017). *Análisis de la Afectación y Severidad de los Incendios Forestales ocurridos en Enero y Febrero de 2017 sobre los Usos de Suelo y los Ecosistemas Naturales presentes entre las Regiones de Coquimbo y Los Ríos de Chile*. Santiago, Chile. Retrieved from http://www.conaf.cl/tormenta_de_fuego-2017

Correa, F. (2011). Desencializando lo indígena. In G. R. Nemogá (Ed.), *Naciones Indígenas en los Estados Contemporáneos* (pp. 83–111). Bogotá, Colombia: Universidad Nacional de Colombia.

Correa, S. B., Araujo, J. K., Penha, J. M. F., Nunes da Cunha, C., Stevenson, P. R., & Anderson, J. P. (2015). Overfishing disrupts an ancient mutualism between frugivorous fishes and plants in Neotropical wetlands. *Biological Conservation*, *191*, 159–167. http://doi.org/10.1016/j.biocon.2015.06.019

Costanza, R., Pérez-Maqueo, O., Martinez, M. L., Sutton, P., Anderson, S. J., & Mulder, K. (2008). The value of coastal wetlands for hurricane protection. *Ambio*, *37*(4), 241–248. http://doi.org/10.1579/0044-7447(2008)37%5B241:TVOCWF%5D2.0.CO;2

Costello, C., Ovando, D., Clavelle, T., Strauss, C. K., Hilborn, R., Melnychuk, M. C., Branch, T. A., Gaines, S. D., Szuwalski, C. S., Cabral, R. B., Rader, D. N., & Leland, A. (2016). Global fishery prospects under contrasting management regimes. *PNAS*, *113*(18), 5125–5129. https://doi.org/10.1073/pnas.1520420113

Costello, E. K., Halloy, S. R. P., Reed, S. C., Sowell, P., & Schmidt, S. K. (2009). Fumarole-supported islands of biodiversity within a hyperarid, high-elevation landscape on Socompa volcano, Puna de Atacama, Andes. *Applied and Environmental Microbiology*, *75*(3), 735–747. http://doi.org/10.1128/AEM.01469-08

Crandall, K. A., & Buhay, J. E. (2008). Global diversity of crayfish (Astacidae, Cambaridae, and Parastacidae - Decapoda) in freshwater. *Hydrobiologia*, *595*(1), 295–301. http://doi.org/10.1007/s10750-007-9120-3

Crawford, A. J., Alonso, R., Jaramillo, C. A. A., Sucre, S., & Ibáńez, R. D. (2011). DNA barcoding identifies a third invasive species of *Eleutherodactylus* (Anura: Eleutherodactylidae) in Panama City, Panama. *Zootaxa*, *67*(2890), 65–67.

Crimmins, S. M., Dobrowski, S. Z., Greenberg, J. A., Abatzoglou, J. T., & Mynsberge, A. R. (2011). Changes in climatic water balance drive downhill shifts in plant species' optimum elevations. *Science*, *331*(6015), 324–327. http://doi.org/10.1126/science.1199040

Crisp, Laffan, Linder, & Monro. (2001). Endemism in the Australian flora. *Journal of Biogeography*, *28*(2), 183–198. http://doi.org/10.1046/j.1365-2699.2001.00524.x

Crits-Christoph, A., Gelsinger, D. R., Ma, B., Wierzchos, J., Ravel, J., Davila, A., Casero, M. C., & DiRuggiero, J. (2016). Functional interactions of archaea, bacteria and viruses in a hypersaline endolithic community. *Environmental Microbiology*, *18*(6), 2064–2077. https://doi.org/10.1111/1462-2920.13259

Cuesta, F., Muriel, P., Llambí, L. D., Halloy, S., Aguirre, N., Beck, S., Carilla, J., Meneses, R. I., Cuello, S., Grau, A., Gámez, L. E., Irazábal, J., Jácome, J., Jaramillo, R., Ramírez, L., Samaniego, N., Suárez-Duque, D., Thompson, N., Tupayachi, A., Viñas, P., Yager, K., Becerra, M. T., Pauli, H., & Gosling, W. D. (2017). Latitudinal and altitudinal patterns of plant community diversity on mountain summits across the tropical Andes. *Ecography*, *40*(12), 1381–1394. https://doi.org/10.1111/ecog.02567

Cushman, S. A. (2006). Effects of habitat loss and fragmentation on amphibians: A review and prospectus. *Biological Conservation*, *128*(2), 231–240. http://doi.org/10.1016/j.biocon.2005.09.031

da Silva, F. R., Candeira, C. P., & Rossa-Feres, D. C. (2012). Dependence of anuran diversity on environmental descriptors in farmland ponds. *Biodiversity and Conservation*, *21*(6), 1411–1424. http://doi.org/10.1007/s10531-012-0252-z

da Silva, F. R., Gibbs, J. P., & Rossa-Feres, D. C. (2011). Breeding habitat and landscape correlates of frog diversity and abundance in a tropical agricultural landscape. *Wetlands*, *31*(6), 1079–1087. http://doi.org/10.1007/s13157-011-0217-0

Dahl, T. E. (1990). *Wetland Losses in the United States 1780's to 1980's*. Washington, D.C., USA: Department of the Interior, Fish and Wildlife Service.

Dahl, T. E. (2011). *Status and Trends of Wetlands in the Conterminous United States 2004 to 2009*. Washington D.C., USA: U.S. Department of the Interior; Fish and Wildlife Service.

Dai, D. (2011). Racial/ethnic and socioeconomic disparities in urban green space accessibility: Where to intervene? *Landscape and Urban Planning*, *102*(4), 234–244. http://doi.org/10.1016/j.landurbplan.2011.05.002

Dangles, O., Carpio, C., Barragan, A. R., Zeddam, J.-L., & Silvain, J.-F. (2008). Temperature as a key driver of ecological sorting among invasive pest species in the tropical Andes. *Ecological Applications*, *18*(7), 1795–1809. http://doi.org/10.1890/07-1638.1

Daniels, A. E., & Cumming, G. S. (2008). Conversion or conservation? Understanding wetland change in northwest Costa Rica. *Ecological Applications*, *18*(1), 49–63. http://doi.org/10.1890/06-1658.1

Daniels, L. D., & Veblen, T. T. (2004). Spatiotemporal influences of climate on altitudinal treeline in northern Patagonia. *Ecology*, *85*(5), 1284–1296. http://doi.org/10.1890/03-0092

Danielsen, F., Beukema, H., Burgess, N. D., Parish, F., Buhl, C. A., Donald, P. F., Murdiyarso, D., Phalan, B., Reijnders, L., Struebig, M., & Fitzherbert, E. B. (2009). Biofuel plantations on forested lands: double jeopardy for biodiversity and climate. *Conservation Biology*, *23*(2), 348–358. https://doi.org/10.1111/j.1523-1739.2008.01096.x

Darrigran, G., Damborenea, C., Drago, E. C., Ezcurra de Drago, I., Paira, A., & Archuby, F. (2012). Invasion process of *Limnoperna fortunei* (Bivalvia: Mytilidae): the case of Uruguay River and emissaries of the Esteros del Iberá Wetland, Argentina. *Zoología*, *29*(6), 531–539. http://doi.org/10.1590/S1984-46702012000600004

Darrigran, G., & Ezcurra de Drago, I. (2000). Invasion of *Limnoperna fortunei* (Dunker, 1857) (Bivalvia: Mytilidae) in America. *The Nautilus*, *114*, 69–73. http://doi.org/10.1007/s10530-005-0331-0

Davidson, E. A., de Araújo, A. C., Artaxo, P., Balch, J. K., Brown, I. F., Bustamante, M. M., Coe, M. T., DeFries, R. S., Keller, M., Longo, M., Munger, J. W., Schroeder, W., Soares-Filho, B. S., Souza, C. M., & Wofsy, S. C. (2012). The Amazon basin in transition. *Nature*, *481*(7381), 321–328. https://doi.org/10.1038/nature10717

Davidson, L. N. K., Krawchuk, M. A., & Dulvy, N. K. (2016). Why have global shark and ray landings declined: Improved management or overfishing? *Fish and Fisheries*, *17*(2), 438–458. http://doi.org/10.1111/faf.12119

Davies, T. J., & Buckley, L. B. (2011). Phylogenetic diversity as a window into the evolutionary and biogeographic histories of present-day richness gradients for mammals. *Philosophical Transactions of the Royal Society B: Biological Sciences*, *366*(1576), 2414–2425. http://doi.org/10.1098/rstb.2011.0058

Davis, F. W., Baldocchi, D. D., & Tyler, C. M. (2016). Oak woodlands. In H. Mooney & E. Zavaleta (Eds.), *Ecosystems of California* (pp. 509–529). Oakland, USA: University of California Press.

De Jong, W. (1996). Swidden-fallow agroforestry in Amazonia: diversity at close distance. *Agroforestry Systems*, *34*, 277–290. http://doi.org/10.1007/BF00046928

De la Cruz, R. (2011). Conocimientos tradicionales, biodiversidad y derechos de propiedad intelectual, patentes. In G. R. Nemogá (Ed.), *Naciones Indígenas en los Estados Contemporáneos* (pp. 211–230). Bogotá, Colombia: Universidad Nacional de Colombia.

De la Cruz, R., Muyuy Jacanamejoy, G., Viteri Gualinga, A., Flores, G., Humpire, J. G., Mirabal DÌaz, J. G., & Guimaraez, R. (2005). *Elementos para la Protección sui generis de los Conocimientos Tradicionales Colectivos e Integrales desde la Perspectiva Indigena*. Caracas, Venezuela: The Corporacion Andina de Fomento (CAF)/Comunidad Andina (CAN).

de Oliveira, G., Araújo, M. B., Rangel, T. F., Alagador, D., & Diniz-Filho, J. A. F. (2012). Conserving the Brazilian semiarid (Caatinga) biome under climate change. *Biodiversity and Conservation*, *21*(11), 2913–2926. http://doi.org/10.1007/s10531-012-0346-7

de Vasconcellos Pegas, F., & Castley, J. G. (2016). Private reserves in Brazil: Distribution patterns, logistical challenges, and conservation contributions. *Journal for Nature Conservation*, *29*, 14–24. http://doi.org/10.1016/j.jnc.2015.09.007

DeClerck, F. A. J., Chazdon, R., Holl, K. D., Milder, J. C., Finegan, B., Martinez-Salinas, A., Imbach, P., Canet, L., & Ramos, Z. (2010). Biodiversity conservation in human-modified landscapes of Mesoamerica: Past, present and future. *Biological Conservation*, *143*, 2301–2313. https://doi.org/10.1016/j.biocon.2010.03.026

Dee, L. E., Horii, S. S., & Thornhill, D. J. (2014). Conservation and management of ornamental coral reef wildlife: Successes, shortcomings, and future directions. *Biological Conservation*, *169*, 225–237.

Dee, L. E., Miller, S. J., Peavey, L. E., Bradley, D., Gentry, R. R., Startz, R., Gaines, S. D., & Lester, S. E. (2016). Functional diversity of catch mitigates negative effects of temperature variability on fisheries yields. *Proceedings of the Royal Society of London B: Biological Sciences*, *283*(1836), 20161435. https://doi.org/http://dx.doi.org/10.1098/rspb.2016.1435

Del Hoyo, J., Elliott, A., & Sargatal, J. (Eds.). (1992a). *Handbook of the Birds of the World, Volume 1: Ostrich to Ducks*. Barcelona, Spain: Lynx Edicions.

Del Hoyo, J., Elliott, A., & Sargatal, J. (Eds.). (1992b). *Handbook of the Birds of the World, Volume 3: Hoatzin to Auks*. Barcelona, Spain: Lynx Edicions.

Delis, P. R., Mushinsky, H. R., & McCoy, E. D. (1996). Decline of some west-central Florida anuran populations in response to habitat degradation. *Biodiversity and Conservation*, *5*(12), 1579–1595. http://doi.org/10.1007/BF00052117

Denslow, J. S., & DeWalt, S. J. (2008). Exotic plant invasions in tropical forests: patterns and hypotheses. In W. Carson & S. Schnitze (Eds.), *Tropical forest community ecology* (pp. 409–426). Oxford, UK: Wiley-Blackwell.

Derlindati, E. J., Romano, M. C., Cruz, N. N., Barisón, C., Arengo, F., & Barberis, I. M. (2014). Seasonal activity patterns and abundance of Andean flamingo (*Phoenicoparrus andinus*) at two contrasting wetlands in Argentina. *Ornitología Neotropical*, *25*(3), 317–331.

Díaz-Betancourt, M., Ghermandi, L., Ladio, A., López-Moreno, I. R., Raffaele, E., & Rapoport, E. H. (1999). Weeds as a source for human consumption. A comparison between tropical and temperate Latin America. *Revista de Biologia Tropical*, *47*(3), 329–338.

Díaz, I. A., & Armesto, J. J. (2003). La conservación de las aves silvestres en ambientes urbanos de Santiago. *Revista Ambiente Y Desarrollo de CIPMA*, *19*(2), 31–38.

Diaz, R. J., & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *Science*, *321*(5891), 926–929. http://doi.org/10.1126/science.1156401

Diffenbaugh, N. S., Swain, D. L., & Touma, D. (2015). Anthropogenic warming has increased drought risk in California. *PNAS*, *112*(13), 3931–3936. http://doi.org/10.1073/pnas.1422385112

Dillon, M. O., Leiva González, S., Zapata Cruz, M., Lezama Asencio, P., & Quipuscoa Silvestre, V. (2011). Floristic checklist of the Peruvian lomas formations. *Arnaldoa*, *18*(1), 7–32.

Dinerstein, E., Olson, D. M., Graham, D. J., Webster, A. L., Primm, S. A., Bookbinder, M. P., & Ledec, G. (1995). *A Conservation Assessment of the Terrestrial Ecoregions of Latin America and the Caribbean*. Washington, D.C., USA: The World Bank and World Wildlife Fund. Retrieved from http://documents.worldbank.org/curated/en/1995/09/697067/conservation-assessment-terrestrial-ecoregions-latin-america-caribbean

Dirzo, R., & Raven, P. (2003). Global state of biodiversity and loss. *Annual Reivew of the Environment and Resources*, *28*, 137–167.

Diaz, S., & Cabido, M. (2001). Vive la différence: plant functional diversity matters to ecosystem processes. *Trends in Ecology & Evolution*, *16*(11), 646–655. http://dx.doi.org/10.1016/S0169-5347(01)02283-2

Dolanc, C. R., Safford, H. D., Dobrowski, S. Z., & Thorne, J. H. (2014). Twentieth century shifts in abundance and composition of vegetation types of the Sierra Nevada, CA, US. *Applied Vegetation Science*, *17*(3), 442–455. http://doi.org/10.1111/avsc.12079

Dolanc, C. R., Safford, H. D., Thorne, J. H., & Dobrowski, S. Z. (2014). Changing forest structure across the landscape of the Sierra Nevada, CA, USA, since the 1930s. *Ecosphere*, *5*(8), art101. http://doi.org/10.1890/ES14-00103.1

Domínguez, E., Bahamonde, N., & Muñoz-Escobar, C. (2012). Efectos de la extracción de turba sobre la composición y estructura de una turbera de Sphagnum explotada y abandonada hace 20 años, Chile. *Anales Del Instituto de La Patagonia*, *40*(2), 37–45. http://doi.org/10.4067/S0718-686X2012000200003

Donohue, R. J., Roderick, M. L., McVicar, T. R., & Farquhar, G. D. (2013). Impact of CO2 fertilization on maximum foliage cover across the globe's warm, arid environments. *Geophysical Research Letters*, *40*(12), 3031–3035. http://doi.org/10.1002/grl.50563

Dornelas, M., Gotelli, N. J., McGill, B., Shimadzu, H., Moyes, F., Sievers, C., & Magurran, A. E. (2014). Assemblage time series reveal biodiversity change but not systematic loss. *Science*, *344*(6181), 296–299. http://doi.org/10.1126/science.1248484

Doroff, A., & Burdin, A. (2015). *Enhydra lutris*. The IUCN Red List of Threatened Species 2015: e.T7750A21939518. Retrieved from http://dx.doi.org/10.2305/IUCN.UK.2015-2.RLTS.T7750A21939518.en.

Duarte, C. M. (1995). Submerged aquatic vegetation in relation to different nutrient regimes. *Ophelia*, *41*(1), 87–112. http://doi.org/10.1080/00785236.1995.10422039

Duboc, E., Costa, C. J., Veloso, R. F., Oliveira, L. S., & Paludo, A. (2007). *Panorama Atual da Produção de Carvão Vegetal no Brasil e no Cerrado. Documentos - Embrapa Cerrados No. 197*. Retrieved from http://www.cpac.embrapa.br

Duffy, J. E., Lefcheck, J. S., Stuart-Smith, R. D., Navarrete, S. A., & Edgar, G. J. (2016). Biodiversity enhances reef fish biomass and resistance to climate change. *PNAS*, *113*(22), 6230–6235. http://doi.org/10.1073/pnas.1524465113

Dulvy, N. K., Fowler, S. L., Musick, J. A., Cavanagh, R. D., Kyne, P. M., Harrison, L. R., Carlson, J. K., Davidson, L. N., Fordham, S. V, Francis, M. P., Pollock, C. M., Simpfendorfer, C. A., Burgess, G. H., Carpenter, K. E., Compagno, L. J., Ebert, D. A., Gibson, C., Heupel, M. R., Livingstone, S. R., Sanciangco, J. C., Stevens, J. D., Valenti, S., & White, W. T. (2014). Extinction risk and conservation of the world's sharks and rays. *ELife*, *3*, e00590. https://doi.org/10.7554/eLife.00590

Dunic, J. C. (2016). *Contextualizing Local Biodiversity Change in the Face of Human Impacts.* Masters Dissertation, 415, University of Massachusetts Boston. Retrieved from https://scholarworks.umb.edu/masters_theses/415

Dunn, R. R., Agosti, D., Andersen, A. N., Arnan, X., Bruhl, C. A., Cerdá, X., Ellison, A. M., Fisher, B. L., Fitzpatrick, M. C., Gibb, H., Gotelli, N. J., Gove, A. D., Guenard, B., Janda, M., Kaspari, M., Laurent, E. J., Lessard, J. P., Longino, J. T., Majer, J. D., Menke, S. B., McGlynn, T. P., Parr, C. L., Philpott, S. M., Pfeiffer, M., Retana, J., Suarez, A. V., Vasconcelos, H. L., Weiser, M. D., & Sanders, N. J. (2009). Climatic drivers of hemispheric asymmetry in global patterns of ant species richness. *Ecology Letters*, *12*(4), 324–333. https://doi.org/10.1111/j.1461-0248.2009.01291.x

Durán, A., Gryzenhout, M., Slippers, B., Ahumada, R., Rotella, A., Flores, F., Wingfield, B. D., & Wingfield, M. J. (2008). *Phytophthora pinifolia* sp. nov. associated with a serious needle disease of *Pinus radiata* in Chile. *Plant Pathology*, *57*(4), 715–727. https://doi.org/10.1111/j.1365-3059.2008.01893.x

Durigan, G., & Ratter, J. A. (2016). The need for a consistent fire policy for Cerrado conservation. *Journal of Applied Ecology*, *53*(1), 11–15. http://doi.org/10.1111/1365-2664.12559

Dyer, E. E., Cassey, P., Redding, D. W., Collen, B., Franks, V., Gaston, K. J., Jones, K. E., Kark, S., Orme, C. D. L., & Blackburn, T. M. (2017). The global distribution and drivers of alien bird species richness. *PLoS Biol*, *15*(1), e2000942. https://doi.org/10.1371/journal.pbio.2000942

Early, R., Bradley, B. A., Dukes, J. S., Lawler, J. J., Olden, J. D., Blumenthal, D. M., Gonzalez, P., Grosholz, E. D., Ibañez, I., Miller, L. P., Sorte, C. J. B., & Tatem, A. J. (2016). Global threats from invasive alien species in the twenty-first century and national response capacities. *Nature Communications*, *7*, art12485. https://doi.org/10.1038/ncomms12485

Eberhardt, L. L., & O'Shea, T. J. (1995). Integration of manatee life history data and population modeling. In T. J. O'Shea, B. B. Ackerman, & H. F. Percival (Eds.), *Population Biology of the Florida Manatee* (pp. 269–273). Washington D.C., USA: National Biological Service, Information and Technology Report.

Eclesia, R. P., Jobbagy, E. G., Jackson, R. B., Biganzoli, F., & Piñeiro, G. (2012). Shifts in soil organic carbon for plantation and pasture establishment in native forests and grasslands of South America. *Global Change Biology*, *18*(10), 3237–3251. http://doi.org/10.1111/j.1365-2486.2012.02761.x

Edwards, T., Schwalbe, C. R., Swann, D. E., & Goldberg, C. S. (2004). Implications of anthropogenic landscape change on inter-population movements of the desert tortoise (*Gopherus agassizii*). *Conservation Genetics*, *5*(4), 485–499. http://doi.org/10.1023/B:COGE.0000041031.58192.7c

Eisemberg, C. C., Machado Balestra, R. A., Famelli, S., Pereira, F. F., Diniz Bernardes, V. C., & Vogt, R. C. (2016). Vulnerability of giant South American turtle (*Podocnemis expansa*) nesting habitat to climate-change-induced alterations to fluvial cycles. *Tropical Conservation Science*, (October-December issue), 1–12. http://doi.org/10.1177/1940082916667139

Elahi, R., O'Connor, M. I., Byrnes, J. E. K., Dunic, J., Eriksson, B. K., Hensel, M. J. S., & Kearns, P. J. (2015). Recent trends in local-scale marine biodiversity reflect community structure and human impacts. *Current Biology*, *25*(14), 1938–1943. http://doi.org/10.1016/j.cub.2015.05.030

Elliott, G. P. (2011). Influences of 20th-century warming at the upper tree line contingent on local-scale interactions: evidence from a latitudinal gradient in the Rocky Mountains, USA. *Global Ecology and Biogeography*, *20*(1), 46–57.

Ellison, A. M., & Farnsworth, E. J. (1996). Anthropogenic disturbance of Caribbean mangrove ecosystems: Past impacts, present trends, and future predictions. *Biotropica*, *28*(4), 549–565. http://doi.org/10.2307/2389096

**Elmendorf, S. C., Henry, G. H. R., Hollister, R. D., Björk, R. G., Boulanger-Lapointe, N., Cooper, E. J., Cornelissen, J. H. C., Day, T. A., Dorrepaal, E., Elumeeva, T. G., Gill, M., Gould, W. A., Harte, J., Hik, D. S., Hofgaard, A., Johnson, D. R., Johnstone, J. F., Jónsdóttir, I. S., Jorgenson, J. C.,

Klanderud, K., Klein, J. A., Koh, S., Kudo, G., Lara, M., Lévesque, E., Magnússon, B., May, J. L., Mercado-Díaz, J. A., Michelsen, A., Molau, U., Myers-Smith, I. H., Oberbauer, S. F., Onipchenko, V. G., Rixen, C., Schmidt, N. M., Shaver, G. R., Spasojevic, M. J., Þórhallsdóttir, Þ. E., Tolvanen, A., Troxler, T., Tweedie, C. E., Villareal, S., Wahren, C.-H., Walker, X., Webber, P. J., Welker, J. M., & Wipf, S. (2012). Plot-scale evidence of tundra vegetation change and links to recent summer warming. *Nature Climate Change*, *2*, 453–457. https://doi.org/10.1038/nclimate1465

Elsner, W. K., & Jorgenson, J. C. (2009). White spruce seedling (*Picea glauca*) discovered north of the Brooks Range along Alaska's Dalton Highway. *Arctic*, *62*(3), 342–344.

Elven, R., Murray, D. F., Razzhivin, V. Y., & Yurtsev, B. A. (2011). Annotated checklist of the panarctic flora (PAF): vascular plants. Retrieved from http://geo.abds.is/geonetwork/srv/eng/catalog.search - /metadata/ad3b16c9-9a29-460e-a515-d0a48e6d88ff

Emperaire, L., & Peroni, N. (2007). Traditional management of agrobiodiversity in Brazil: A case study of manioc. *Human Ecology*, *35*, 761–768.

Enquist, B. J., Condit, R., Peet, R. K., Schildhauer, M., & Thiers, B. (2016). The Botanical Information and Ecology Network (BIEN): Cyberinfrastructure for an integrated botanical information network to investigate the ecological impacts of global climate change on plant biodiversity. http://doi.org/10.7287/peerj.preprints.2615v2.

Epstein, H. E., Raynolds, M. K., Walker, D. A., Bhatt, U. S., Tucker, C. J., & Pinzon, J. E. (2012). Dynamics of aboveground phytomass of the circumpolar Arctic tundra during the past three decades. *Environmental Research Letters*, *7*(1), 15506.

Ernst, O., & Siri-Prieto, G. (2009). Impact of perennial pasture and tillage systems on carbon input and soil quality indicators. *Soil and Tillage Research*, *105*(2), 260–268. http://doi.org/10.1016/j.still.2009.08.001

Eskew, E. A., Price, S. J., & Dorcas, M. E. (2012). Effects of river-flow regulation on anuran occupancy and abundance in riparian zones. *Conservation Biology*, *26*(3), 504–512. http://doi.org/10.1111/j.1523-1739.2012.01842.x

Espinosa, F. J., & Vibrans, H. (2009). The need for a national weed management strategy. In T. van Devender, F. J. Espinosa-García, B. L. Harper-Lore, & T. Hubbard (Eds.), *Invasive Plants on the Move: Controlling them in North America* (pp. 23–32). Tucson, USA: Arizona-Sonora Desert Museum.

Estay, S. A. (2016). Invasive insects in the mediterranean forests of Chile. In T. D. Paine & F. Lieutier (Eds.), *Insects and Diseases of Mediterranean Forest Systems* (pp. 379–396). New York, USA: Springer. http://doi.org/10.1007/978-3-319-24744-1_13

Estevez, R. A., Squeo, F. A., Arancio, G., & Erazo, M. B. (2010). Production of charcoal from native shrubs in the Atacama Region, Chile. *Gayana Botánica*, *67*(2), 213–222.

Eva, H. D., Belward, A. S., De Miranda, E. E., Di Bella, C. M., Gond, V., Huber, O., Jones, S., Sgrenzaroli, M., & Fritz, S. (2004). A land cover map of South America. *Global Change Biology*, *10*(5), 731–744. https://doi.org/10.1111/j.1529-8817.2003.00774.x

Evans, P., & Brown, C. D. (2017). The boreal-temperate forest ecotone response to climate change. *Environmental Reviews*, *25*(4), 423–431. http://doi.org/10.1139/er-2017-0009

Ewing, H. A., Tuininga, A. R., Groffman, P. M., Weathers, K. C., Fahey, T. J., Fisk, M. C., Bohlen, P. J., & Suarez, E. (2015). Earthworms reduce biotic 15-nitrogen retention in northern hardwood forests. *Ecosystems*, *18*(2), 328–342. https://doi.org/10.1007/s10021-014-9831-z

Faeth, S. H., Marussich, W. A., Shochat, E., & Warren, P. S. (2005). Trophic dynamics in urban communities. *BioScience*, *55*(5), 399–407. http://doi.org/10.1641/0006-3568(2005)055%5B0399:TDIUC%5D2.0.CO;2

Faggi, A., & Perepelizin, P. V. (2006). Riqueza de aves a lo largo de un gradiente de urbanización en la ciudad de Buenos Aires. *Revista Del Museo Argentino de Ciencias Naturales*, *8*(2), 289–297.

Fahrig, L. (2003). Effects of habitat fragmentation on biodiversity. *Annual Review of Ecology, Evolution, and Systematics*, *34*(1), 487–515. http://doi.org/10.1146/annurev.ecolsys.34.011802.132419

Fahrig, L., Pedlar, J. H., Pope, S. E., Taylor, P. D., & Wegner, J. F. (1995). Effect of road traffic on amphibian density. *Biological Conservation*, *73*(3), 177–182. http://doi.org/10.1016/0006-3207(94)00102-V

Faith, D. P. (1992). Conservation evaluation and phylogenetic diversity. *Biological Conservation*, *61*(1), 1–10. http://doi.org/10.1016/0006-3207(92)91201-3

Fajardo, A., & McIntire, E. J. B. (2012). Reversal of multicentury tree growth improvements and loss of synchrony at mountain tree lines point to changes in key drivers. *Journal of Ecology*, *100*(3), 782–794. http://doi.org/10.1111/j.1365-2745.2012.01955.x

Falvey, M., & Garreaud, R. D. (2009). Regional cooling in a warming world: recent temperature trends in the southeast Pacific and along the west coast of subtropical South America (1979-2006). *Journal of Geophysical Research: Atmospheres*, *114*(D4). http://doi.org/10.1029/2008JD010519

FAO. (2007). *The State of Food and Agriculture 2007*. Rome, Italy. Retrieved from http://www.fao.org/docrep/010/a1200e/a1200e00.htm

FAO. (2011). *The State of Forest Resources- a Regional Analysis. In State of the World's Forests*. *Food and Agriculture Organization of the United Nations.* Rome, Italy. Retrieved from http://www.fao.org/docrep/013/i2000e/i2000e00.htm

FAO. (2012). The State of World Fisheries and Aquaculture 2012. *Aquaculture Newsletter*, 209. Retrieved from http://www.fao.org/3/a-i2727e.pdf

FAO. (2016). *The State of World Fisheries and Aquaculture 2016: Contributing to Food Security and Nutrition for All*. *Food and Agriculture Organization of the United Nations*. Rome, Italy. Retrieved from http://www.fao.org/3/a-i5555e.pdf

Fargione, J., Tilman, D., Dybzinski, R., Lambers, J. H. R., Clark, C., Harpole,

W. S., Knops, J. M. H., Reich, P. B., & Loreau, M. (2007). From selection to complementarity: shifts in the causes of biodiversity-productivity relationships in a long-term biodiversity experiment. *Proceedings of the Royal Society Biological Sciences*, *274*(1611), 871–876. https://doi.org/10.1098/rspb.2006.0351

Farley, K. A., Bremer, L. L., Harden, C. P., & Hartsig, J. (2013). Changes in carbon storage under alternative land uses in biodiverse Andean grasslands: implications for payment for ecosystem services. *Conservation Letters*, *6*(1), 21–27. http://doi.org/10.1111/j.1755-263X.2012.00267.x

Farley, K. A., Jobbágy, E. G., & Jackson, R. B. (2005). Effects of afforestation on water yield: a global synthesis with implications for policy. *Global Change Biology*, *11*(10), 1565–1576. http://doi.org/10.1111/j.1365-2486.2005.01011.x

Farley, K. A., Kelly, E. F., & Hofstede, R. G. M. (2004). Soil organic carbon and water retention after conversion of grasslands to pine plantations in the Ecuadorian Andes. *Ecosystems*, *7*(7), 729–739. http://doi.org/10.1007/s10021-004-0047-5

Federal Provincial and Territorial Governments of Canada. (2010). *Canadian Biodiversity: Ecosystem Status and Trends 2010*. Ottawa, Canada: Canadian Councils of Resource Ministers. Retrieved from http://www.biodivcanada.ca/A519F000-8427-4F8C-9521-8A95AE287753/EN_CanadianBiodiversity_FULL.pdf

Federal Register. (2016). *Indian Entities Recognized and Eligible to Receive Services From the United States Bureau of Indian Affairs*. Bureau of Indian Affairs. Retrieved from https://www.gpo.gov/fdsys/pkg/FR-2016-01-29/pdf/2016-01769.pdf

Feeley, K. J., Silman, M. R., Bush, M. B., Farfan, W., Cabrera, K. G., Malhi, Y., Meir, P., Revilla, N. S., Quisiyupanqui, M. N. R., & Saatchi, S. (2011). Upslope migration of Andean trees. *Journal of Biogeography*, *38*(4), 783–791. https://doi.org/10.1111/j.1365-2699.2010.02444.x

Fehlenberg, V., Baumann, M., Gasparri, N. I., Piquer-Rodriguez, M., Gavier-Pizarro, G., & Kuemmerle, T. (2017). The role of soybean production as an underlying driver of deforestation in the South American Chaco. *Global Environmental Change*, *45*(April), 24–34. http://doi.org/10.1016/j.gloenvcha.2017.05.001

Fernandez, A. B., Rasuk, M. C., Visscher, P. T., Contreras, M., Novoa, F., Poire, D. G., Patterson, M. M., Ventosa, A., & Farias, M. E. (2016). Microbial diversity in sediment ecosystems (evaporites domes, microbial mats, and crusts) of hypersaline Laguna Tebenquiche, Salar de Atacama, Chile. *Frontiers in Microbiology*, *7*, 1284. https://doi.org/10.3389/fmicb.2016.01284

Ferrusquía-Villafranca, I., González-Guzmán, L. I., & Cartron, J. E. (2005). Northern Mexico's landscape, part I: the physical setting and constraints on modeling biotic evolution. In J. E. Cartron, G. Ceballos, & R. S. Felger (Eds.), *Biodiversity, Ecosystems, and Conservation in Northern Mexico* (pp. 11–38). New York, USA: Oxford University Press.

Fields, J. P., Belnap, J., Breshears, D. D., Neff, J., Okin, G. S., Whicker, J. J., Painter, T. H., Ravi, S., Reheis, M. C., & Reynolds, R. L. (2009). The ecology of dust. *Frontiers in Ecology and the Environment*, *8*(8), 423–430. https://doi.org/10.1890/090050

Figueroa, J. A., Teillier, S., & Castro, S. A. (2011). Diversity patterns and composition of native and exotic floras in central Chile. *Acta Oecologica*, *37*(2), 103–109. http://doi.org/10.1016/j.actao.2011.01.002

Fish, M. R., Lombana, A., & Drews., C. (2009). *Regional Climate Projections: Climate Change and Marine Turtles in the Wider Caribbean*. San Jose, Costa Rica: WWF report. Retrieved from http://awsassets.panda.org/downloads/climate_change_and_marine_turtles_in_the_wider_caribbean_1.pdf

Fisheries and Oceans Canada. (2017). Action plan for the Northern and Southern resident Killer Whale (*Orcinus orca*) in Canada. Species at Risk Action Plan Series. Ottawa, Canada: Fisheries and Oceans Canada. Retrieved from http://www.registrelep-sararegistry.gc.ca/virtual_sara/files/plans/Ap-ResidentKillerWhale-v00-2017Mar-Eng.pdf

Fitzpatrick, S. M., & Keegan, W. F. (2007). Human impacts and adaptations in the Caribbean Islands: an historical ecology approach. *Earth and Environmental Science Transactions of the Royal Society of Edinburgh*, *98*(1), 29–45.

Fjeldså, J. (2002). *Polylepis* forests - vestiges of a vanishing ecosystem in the Andes. *Ecotropica*, *8*(2), 111–123.

Fjeldså, J., & Rahbek, C. (2006). Diversification of tanagers, a species rich bird group, from lowlands to montane regions of South America. *Integrative and Comparative Biology*, *46*(1), 72–81. http://doi.org/10.1093/icb/icj009

Flather, C. H., Brady, S. J., & Knowles, M. S. (1999). Wildlife Resource Trends in the United States: A Technical Document Supporting the 2000 RPA Assessment. Gen. Tech. Rep. RMRS-GTR-33. Fort Collins, USA: U.S. Department of Agriculture, Forest Service, Rocky Mountain Research Station.

Flecker, A. S., McIntyre, P. B., Moore, J. W., Anderson, J. T., Taylor, B. W., & Hall Jr., R. O. (2010). Migratory fishes as material and process subsidies in riverine ecosystems. *American Fisheries Society Symposium*, *73*(2), 559–592. Retrieved from http://www.dartmouth.edu/~btaylor/index/Publications_files/flecker_etal_2010_AFSS_migrator-fishes.pdf

Flores-Villela, O., & Gerez, P. (1994). *Biodiversidad y Conservación en México: Vertebrados, Vegetación y Uso del Suelo* (2nd ed.). México D.F., México: Universidad Nacional Autónoma de México.

Fonseca, C. R., Ganade, G., Baldissera, R., Becker, C. G., Boelter, C. R., Brescovit, A. D., Campos, L. M., Fleck, T., Fonseca, V. S., Hartz, S. M., Joner, F., Käffer, M. I., Leal-Zanchet, A. M., Marcelli, M. P., Mesquita, A. S., Mondin, C. A., Paz, C. P., Petry, M. V., Piovensan, F. N., Putzke, J., Stranz, A., Vergara, M., & Vieira, E. M. (2009). Towards an ecologically-sustainable forestry in the Atlantic Forest. *Biological Conservation*, *142*(6), 1209–1219. https://doi.org/10.1016/j.biocon.2009.02.017

Forbes, B. C., & Jefferies, R. L. (1999). Revegetation of disturbed arctic sites: constraints and applications. *Biological Conservation*, *88*(1), 15–24. http://doi.org/10.1016/S0006-3207(98)00095-0

Ford, A., & Nigh, R. (2015). *Maya Forest Garden: Eight Millennia of Sustainable Cultivation of the Tropical Woodlands*. New York, USA: Routledge Taylor & Francis Group.

Ford, M. A., Cahoon, D., & Lynch, J. C. (1999). Restoring marsh elevation in a rapidly subsiding salt marsh by thin layer deposition of dredged material. *Ecological Engineering*, *12*(3), 189–205.

Forero Ulloa, F. E., Cely R., Germán, E., & Palacios Pacheco, L. S. (2015). *Dinámica del Páramo como Espacio para la Captura de Carbono*. Tunja, Colombia: Universidad Pedagógica y Tecnológica de Colombia (UPTC).

Forest, F., Grenyer, R., Rouget, M., Davies, T. J., Cowling, R. M., Faith, D. P., Balmford, A., Manning, J. C., Proches, S., Bank, M. van der, Reeves, G., Hedderson, T. A. J., & Savolainen, V. (2007). Preserving the evolutionary potential of floras in biodiversity hotspots. *Nature*, *445*(7129), 757–760. https://doi.org/10.1038/nature05587

Forister, M. L., & Shapiro, A. M. (2003). Climatic trends and advancing spring flight of butterflies in lowland California. *Global Change Biology*, *9*(7), 1130–1135. http://doi.org/10.1046/j.1365-2486.2003.00643.x

Formica, A., Farrer, E. C., Ashton, I. W., & Suding, K. N. (2014). Shrub expansion over the past 62 years in Rocky Mountain alpine tundra: possible causes and consequences. *Arctic, Antarctic, and Alpine Research*, *46*(3), 616–631.

Foster, R. B., & Hubbell, S. P. (1990). The floristic composition of the Barro Colorado Island forest. In A. H. Gentry (Ed.), *Four Neotropical Rainforests* (pp. 85–98). New Haven, USA: Yale University Press.

Frankie, G. W., Mata, A., & Vinson, S. B. (Eds.). (2004). *Biodiversity Conservation in Costa Rica: Learning the Lessons in a Seasonal Dry Forest.* Berkeley, USA: University of California Press.

Frankie, G. W., Thorp, R. W., Coville, R. E., & Ertter, B. (2014). *California Bees and Blooms: A Guide for Gardeners and Naturalists*. Berkeley, USA: Heyday.

Frankie, G. W., Thorp, R. W., Hernandez, J., Rizzardi, M., Ertter, B., Pawelek, J. C., Witt, S. L., Schindler, M., Coville, R., & Wojcik, V. A. (2009). Native bees are a rich natural resource in urban California gardens. *California Agriculture*, *63*(3), 113–120. https://doi.org/10.3733/ca.v063n03p113

Franzluebbers, A. J. (2005). Soil organic carbon sequestration and agricultural greenhouse gas emissions in the southeastern USA. *Soil and Tillage Research*, *83*(1), 120–147. http://doi.org/10.1016/j.still.2005.02.012

Fraser, J. A., Junqueira, A. B., Kawa, N. C., Moraes, C. P., & Clement, C. R. (2011). Crop diversity on anthropogenic dark earths in central Amazonia. *Human Ecology*, *39*(4), 395–406.

Fraser, R. H., Lantz, T. C., Olthof, I., Kokelj, S. V., & Sims, R. A. (2014). Warming-induced shrub expansion and lichen decline in the Western Canadian Arctic. *Ecosystems*, *17*(7), 1151–1168.

Freitas, L., Salino, A., Menini Neto, L., Almeida, T. E., Mortara, S. R., Stehmann, J. R., Amorim, A. M., Guimaraes, E. F., Coelho, M. N., Zanin, A., & Forzza, R. C. (2016). A comprehensive checklist of vascular epiphytes of the Atlantic Forest reveals outstanding endemic rates. *PhytoKeys*, *58*, 65–79. https://doi.org/10.3897/phytokeys.58.5643

Frelich, L. E. (1995). Old forest in the Lake States today and before European settlement. *Natural Areas Journal*, *15*(2), 157–167.

Frelich, L. E., Hale, C. M., Scheu, S., Holdsworth, A. R., Heneghan, L., Bohlen, P. J., & Reich, P. B. (2006). Earthworm invasion into previously earthworm-free temperate and boreal forests. *Biological Invasions*, *8*(2066), 1235–1245. http://doi.org/10.1007/s10530-006-9019-3

Frelich, L. E., & Reich, P. B. (2009). Wilderness conservation in an era of global warming and invasive species: a case study from Minnesota’s Boundary Waters Canoe Area Wilderness. *Natural Areas Journal*, *29*(4), 385–393.

Frick, W. F., Cheng, T. L., Langwig, K. E., Hoyt, J. R., Janicki, A. F., Parise, K. L., Foster, J. T., & Kilpatrick, A. M. (2017). Pathogen dynamics during invasion and establishment of white-nose syndrome explain mechanisms of host persistence. *Ecology*, *98*(3), 624–631. https://doi.org/10.1002/ecy.1706/suppinfo

Fritz, S. A., Bininda-Emonds, O. R. P., & Purvis, A. (2009). Geographical variation in predictors of mammalian extinction risk: big is bad, but only in the tropics. *Ecology Letters*, *12*(6), 538–549.

Fuentes, N., Saldaña, A., Kühn, I., & Klotz, S. (2015). Climatic and socio-economic factors determine the level of invasion by alien plants in Chile. *Plant Ecology and Diversity*, *8*(3), 371–377. http://doi.org/10.1080/17550874.2014.984003

Fuller, P., & Neilson, M. (2015). The U.S. Geological Survey’s Nonindigenous Aquatic Species Database: over thirty years of tracking introduced aquatic species in the United States (and counting). *Management of Biological Invasions*, *6*(2), 159–170. http://doi.org/10.3391/mbi.2015.6.2.06

Gaines, S. D., & Lubchenco, J. (1982). A unified approach to marine plant-herbivore interactions II. Biogeography. *Annual Review of Ecology and Systematics*, *13*(1), 111–138. http://doi.org/10.1146/annurev.es.13.110182.000551

Galinier, J. (1997). Pueblos de la Sierra Madre. México: Centro de Estudios Mexicanos y Centroamericanos, Instituto Nacional Indigenista.

Galloway, J. N., Dentener, F. J., Capone, D. G., Boyer, E. W., Howarth, R. W., Seitzinger, S. P., Asner, G. P., Cleveland, C. C., Green, P. A., Holland, E. A., Karl, D. M., Michaels, A. F., Porter, J. H., Townsend, A. R., & Vörösmarty, C. J. (2004). Nitrogen cycles: past, present, and future. *Biogeochemistry*, *70*(2), 153–226. https://doi.org/10.1007/s10533-004-0370-0

Galloway, J. N., Townsend, A. R., Erisman, J. W., Bekunda, M., Cai, Z., Freney, J. R., Martinelli, L. A., Seitzinger, S. P., & Sutton, M. A. (2008). Transformation of the nitrogen cycle: recent trends, questions, and potential solutions. *Science*, *320*(5878), 889–892. https://doi.org/10.1126/science.1136674

Galluzzi, G., Eyzaguirre, P., & Negri, V. (2010). Home gardens: Neglected hotspots of agro-biodiversity and cultural diversity. *Biodiversity and Conservation*, *19*(13), 3635–3654. http://doi.org/10.1007/s10531-010-9919-5

Galluzzi, G., & López Noriega, I. (2014). Conservation and use of genetic resources of underutilized crops in the Americas—a continental analysis. *Sustainability*, *6*(2), 980–1017.

Gamfeldt, L., Snäll, T., Bagchi, R., Jonsson, M., Gustafsson, L., Kjellander, P., Ruiz-Jaen, M. C., Fröberg, M., Stendahl, J., Philipson, C. D., Mikusiński, G., Andersson, E., Westerlund, B., Andrén, H., Moberg, F., Moen, J., & Bengtsson, J. (2013). Higher levels of multiple ecosystem services are found in forests with more tree species. *Nature Communications*, *4*, 1340. https://doi.org/10.1038/ncomms2328

García-Guzman, P., Loayza, A. P., Carvajal, D. E., Letelier, L., & Squeo, F. A. (2012). The ecology, distribution and conservation status of *Myrcianthes coquimbensis* : a globally endangered endemic shrub of the Chilean Coastal Desert. *Plant Ecology and Diversity*, *5*(2), 197–204. http://doi.org/10.1080/17550874.2011.583286

García-Morales, R., Badano, E. I., & Moreno, C. E. (2013). Response of neotropical bat assemblages to human land use. *Conservation Biology*, *27*(5), 1096–1106. http://doi.org/10.1111/cobi.12099

García-Préchac, F., Ernst, O., Siri-Prieto, G., & Terra, J. A. (2004). Integrating no-till into crop-pasture rotations in Uruguay. *Soil and Tillage Research*, *77*(1), 1–13. http://doi.org/10.1016/j.still.2003.12.002

García, R. A., Engler, M. L., Peña, E., Pollnac, F. W., & Pauchard., A. (2015). Fuel characteristics of the invasive shrub *Teline monspessulana* (L.) K. Koch. *International Journal of Wildland Fire*, *24*(3), 372–379.

Gardner, T. A., Côté, I. M., Gill, J. A., Grant, A., & Watkinson, A. R. (2003). Long-term region-wide declines in Caribbean corals. *Science*, *301*(5635), 958–960. http://doi.org/10.1126/science.1086050

Garibaldi, L. A., Carvalheiro, L. G., Leonhardt, S. D., Aizen, M. A., Blaauw, B. R., Isaacs, R., Kuhlmann, M., Kleijn, D., Klein, A. M., Kremen, C., Morandin, L., Scheper, J., & Winfree, R. (2014). From research to action: enhancing crop yield through wild pollinators. *Frontiers in Ecology and the Environment*, *12*(8), 439–447. https://doi.org/10.1890/130330

Garitano-Zavala, Á., & Gismondi, P. (2003). Variación de la riqueza y diversidad de la ornitofauna en áreas verdes urbanas de las ciudades de La Paz y El Alto (Bolivia). *Ecología En Bolivia*, *38*(1), 65–78.

Gärtner, E., Rojas, G., & Castro, S. A. (2015). Compositional patterns of ruderal herbs in Santiago, Chile. *Gayana Botánica*, *72*(2), 192–202.

Gasparri, N. I., & Grau, H. R. (2009). Deforestation and fragmentation of Chaco dry forest in NW Argentina (1972-2007). *Forest Ecology and Management*, *258*(6), 913–921. http://doi.org/10.1016/j.foreco.2009.02.024

Gedan, K. B., Silliman, B. R., & Bertness, M. D. (2009). Centuries of human-driven change in salt marsh ecosystems. *Annual Review of Marine Science*, *1*(1), 117–141. http://doi.org/10.1146/annurev.marine.010908.163930

Germaine, S. S., & Wakeling, B. F. (2001). Lizard species distributions and habitat occupation along an urban gradient in Tucson, Arizona, USA. *Biological Conservation*, *97*(2), 229–237. http://doi.org/10.1016/S0006-3207(00)00115-4

Germino, M. J., Chambers, J. C., & Brown, C. S. (2016). Introduction: Exotic annual *Bromus* in the Western USA. In M. J. Germino, J. C. Chambers, & C. S. Brown (Eds.), *Exotic Brome-Grasses in Arid and Semiarid Ecosystems of the Western US: Causes, Consequences, and Management Implications.* (pp. 1–7). New York, USA: Springer.

Geslin, B., & Morales, C. L. (2015). New records reveal rapid geographic expansion of *Bombus terrestris* Linnaeus, 1758 (Hymenoptera: Apidae), an invasive species in Argentina. *Check List*, *11*(3), 1620. http://doi.org/10.15560/11.3.1620

Gibbs, H. K., Ruesch, A. S., Achard, F., Clayton, M. K., Holmgren, P., Ramankutty, N., & Foley, J. A. (2010). Tropical forests were the primary sources of new agricultural land in the 1980s and 1990s. *PNAS*, *107*(38), 16732–16737. http://doi.org/10.1073/pnas.0910275107

Gill, F., & Donsker, D. (Eds.). (2017). *IOC World Bird List (v 7.3)*. Retrieved from www.worldbirdnames.org

Gilman, E. L., Ellison, J., Duke, N. C., & Field, C. (2008). Threats to mangroves from climate change and adaptation options. *Aquatic Botany*, *89*(2), 237–250.

Gleason, H. A. (1913). The relation of forest distribution and prairie fires in the Middle West. *Torreya*, *13*(8), 173–181. Retrieved from http://www.jstor.org/stable/40595396%5Cnhttp://about.jstor.org/terms

Global Forest Watch. (2017). Retrieved from http://www.globalforestwatch.org/map

Goettsch, B., & Hernández, H. M. (2006). Beta diversity and similarity among cactus assemblages in the Chihuahuan Desert. *Journal of Arid Environments*, *65*(4), 513–528. http://doi.org/10.1016/j.jaridenv.2005.08.008

Goettsch, B., Hilton-Taylor, C., Cruz-Piñón, G., Duffy, J. P., Frances, A., Hernández, H. M., Inger, R., Pollock, C., Schipper, J., Superina, M., Taylor, N. P., Tognelli, M., Abba, A. M., Arias, S., Arreola-Nava, H. J., Baker, M. A., Bárcenas, R. T., Barrios, D., Braun, P., Butterworth, C. A., Búrquez, A., Caceres, F., Chazaro-Basañez, M., Corral-Díaz, R., del Valle Perea, M., Demaio, P. H., Duarte de Barros, W. A., Durán, R., Faúndez Yancas, L., Felger, R. S., Fitz-Maurice, B., Fitz-Maurice, W. A., Gann, G., Gómez-Hinostrosa, C., Gonzales-Torres, L. R., Griffith, M. P., Guerrero, P. C., Hammel, B., Heil, K. D., Guadalupe Hernández-Oria, J., Hoffmann, M., Ishihara, M. I., Kiesling, R., Larocca, J., Luis León-de la Luz, J., Loaiza S., C. R., Lowry, M., Machado, M. C., Majure, L. C., Martínez Ávalos, J. G., Martorell, C., Maschinski, J.,

Méndez, E., Mittermeier, R. A., Nassar, J. M., Negrón-Ortiz, V., Oakley, L. J., Ortega-Baes, P., Ferreira, A. B. P., Pinkava, D. J., Porter, J. M., Puente-Martinez, R., Roque Gamarra, J., Saldivia Pérez, P., Sánchez Martínez, E., Smith, M., Sotomayor M. del C., J. M., Stuart, S. N., Tapia Muñoz, J. L., Terrazas, T., Terry, M., Trevisson, M., Valverde, T., Van Devender, T. R., Véliz-Pérez, M. E., Walter, H. E., Wyatt, S. A., Zappi, D., Alejandro Zavala-Hurtado, J., & Gaston, K. J. (2015). High proportion of cactus species threatened with extinction. *Nature Plants*, *1*(10), 15142. https://doi.org/10.1038/nplants.2015.142

Goijman, A. P., Conroy, M. J., Bernardos, J. N., & Zaccagnini, M. E. (2015). Multi-season regional analysis of multi-species occupancy: implications for bird conservation in agricultural lands in east-central Argentina. *PLoS ONE*, *10*(6), e0130874. http://doi.org/10.1371/journal.pone.0130874

Gómez-González, S., & Cavieres, L. A. (2009). Litter burning does not equally affect seedling emergence of native and alien species of the Mediterranean-type Chilean matorral. *International Journal of Wildland Fire*, *18*(2), 213–221. http://doi.org/10.1071/WF07074

Gómez-González, S., Torres-Díaz, C., Valencia, G., Torres-Morales, P., Cavieres, L. A., & Pausas, J. G. (2011). Anthropogenic fires increase alien and native annual species in the Chilean coastal matorral. *Diversity and Distributions*, *17*(1), 58–67. http://doi.org/10.1111/j.1472-4642.2010.00728.x

Gómez, M. F., Moreno, L. A., Andrade, G. G., & Rueda, C. (Eds.). (2016). *Biodiversidad 2015. Estado y Tendencias de la Biodiversidad Continental de Colombia*. Bogotá, D.C., Colombia: Instituto Alexander von Humboldt.

Gonçalves, F., Fischer, E., & Dirzo, R. (2017). Forest conversion to cattle ranching differentially affects taxonomic and functional groups of Neotropical bats. *Biological Conservation*, *210*, 343–348. http://doi.org/10.1016/j.biocon.2017.04.021

Gonçalves, G. S., Andrade, L. A. D., Xavier, K. R. F., & Silva, J. F. D. (2015). Control methods for *Prosopis juliflora* (Sw.) D.C. (Fabaceae) in invaded areas in the semiarid region of Brazil. *Ciência Florestal*, *25*(3), 645–653. http://doi.org/10.5902/1980509819615

González-Urrutia, M. (2009). Avifauna urbana de América Latina: estudio de casos. *Gestión Ambiental*, *17*, 55–68.

González Martínez, A. I., Barrios Caballero, Y., & De Jesus, S. (2017). Análisis de riesgo para especies invasoras en México. In *El Impacto de las Especies Exóticas en México* (pp. 24–29). Mexico City, Mexico: Centro de Estudios Sociales y de Opinión Pública de la Cámara de Diputados. Retrieved from https://www.diputados.gob.mx/cesop

González Oreja, J. A., Bonache Regidor, C., Buzo Franco, D., De La Fuente Díaz Ordaz, A., & Hernández Satín, L. (2007). Caracterización ecológica de la avifauna de los parques urbanos de la ciudad de Puebla (México). *Ardeola*, *54*(1), 53–67.

Goudie, A., & Seely, M. (2011). *World Heritage Desert Landscapes: Potential Priorities for the Recognition of Desert Landscapes and Geomorphological Sites on the World Heritage List*. Gland, Switzerland: International Union for Conservation of Nature (IUCN).

Gould, W. A., Martinuzzi, S., & Parés-Ramos, I. K. (2012). Land use, population dynamics, and land-cover change in eastern Puerto Rico. In S. F. Murphy & R. F. Stallard (Eds.), *Water Quality and Landscape Processes of Four Watersheds in Eastern Puerto Rico* (pp. 25–42). Reston, USA: U.S. Geological Survey / Professional Paper 1789.

Goulson, D., Nicholls, E., Botías, C., & Rotheray, E. L. (2015). Bee declines driven by combined stress from parasites, pesticides, and lack of flowers. *Science*, *347*(6229), 1255957. http://doi.org/10.1126/science.1255957

Government of Canada. (1991). *The Federal Policy on Wetland Conservation*. Ottawa, Canada. Retrieved from http://publications.gc.ca/collections/Collection/CW66-116-1991E.pdf

Government of Canada. (2009). *Canada's Fourth National Report to the United Nations Convention on Biological Diversity*. Retrieved from https://www.cbd.int/doc/world/ca/ca-nr-04-en.pdf

Gowda, J. H., Kitzberger, T., & Premoli, A. C. (2012). Landscape responses to a century of land use along the northern Patagonian forest-steppe transition. *Plant Ecology*, *213*(2), 259–272. http://doi.org/10.1007/s11258-011-9972-5

Grace, J. B., Anderson, T. M., Seabloom, E. W., Borer, E. T., Adler, P. B., Harpole, W. S., Hautier, Y., Hillebrand, H., Lind, E. M., Pärtel, M., Bakker, J. D., Buckley, Y. M., Crawley, M. J., Damschen, E. I., Davies, K. F., Fay, P. A., Firn, J., Gruner, D. S., Hector, A., Knops, J. M. H., MacDougall, A. S., Melbourne, B. A., Morgan, J. W., Orrock, J. L., Prober, S. M., & Smith, M. D. (2016). Integrative modelling reveals mechanisms linking productivity and plant species richness. *Nature*, *529*(7586), 390–393. https://doi.org/10.1038/nature16524

Gradinger, R., Bluhm, B. A., Hopcroft, R. R., Gebruk, A. V., Kosobokova, K., Sirenko, B., & Węsławski, J. M. (2010). Marine life in the Arctic. In A. D. McIntyre (Ed.), *Life in the World's Oceans: Diversity, Distribution, and Abundance* (pp. 183–202). New Jersey, USA: Willey Blackwell Publishing.

Grau, H. R., Gasparri, N. I., & Aide, T. M. (2005). Agriculture expansion and deforestation in seasonally dry forests of north-west Argentina. *Environmental Conservation*, *32*(2), 140–148.

Grau, H. R., Pérez, M., Martinuzzi, S., Encarnación, X., & Aide, T. M. (2008). Cambios socioeconómicos y regeneración del bosque en la República Dominicana. In M. González-Espinosa, J. M. Rey-Benayas, & N. Ramírez-Marcial (Eds.), *Restauración de Bosques en América Latina* (pp. 211–227). Ciudad de México, México: Fundación Internacional para la Restauración de Ecosistemas (FIRE) and Editorial Mundi-Prensa México.

Grez, A., Zaviezo, T., Gonzalez, G., & Rothman, S. (2010). *Harmonia axyridis* in Chile: a new threat. *Ciencia E Investigacion Agraria*, *37*(3), 145–149. http://doi.org/10.4067/S0718-16202010000300013

Grez, A., Zaviezo, T., Roy, H. E., Brown, P. M. J., & Bizama, G. (2016). Rapid spread of *Harmonia axyridis* in Chile and its effects on local coccinellid biodiversity. *Diversity and Distributions*, *22*(9), 982–994. http://doi.org/10.1111/ddi.12455

Griffin, D., & Anchukaitis, K. J. (2014). How unusual is the 2012-2014 California drought? *Geophysical Research Letters*, *41*(24), 9017–9023. http://doi.org/10.1002/2014GL062433

Grimm, N. B., Cook, E. M., Hale, R. L., Iwaniec, D. M., Seto, K. C., Solecki, W., & Griffith, C. A. (2015). A broader framing of ecosystem services in cities: benefits and challenges of built, natural, or hybrid system function. In K. C. Seto, W. D. Solecki, & C. A. Griffith (Eds.), *The Routledge Handbook of Urbanization and Global Environmental Change* (pp. 203–212). New York, USA: Routeledge Taylor & Francis Group. Retrieved from https://www.routledgehandbooks.com/doi/10.4324/9781315849256.ch14

Grimm, N. B., Faeth, S. H., Golubiewski, N. E., Redman, C. L., Wu, J., Bai, X., & Briggs, J. M. (2008). Global change and the ecology of cities. *Science*, *319*(5864), 756–760. http://doi.org/10.1126/science.1150195

Groffman, P. M., Bain, D. J., Band, L. E., Belt, K. T., Brush, G. S., Grove, J. M., Pouyat, R. V., Yesilonis, I. C., & Zipperer, W. C. (2003). Down by the riverside: urban riparian ecology. *Frontiers in Ecology and the Environment*, *1*(6), 315–321. https://doi.org/10.2307/3868092

Groffman, P. M., Cavender-Bares, J., Bettez, N. D., Grove, J. M., Hall, S. J., Heffernan, J. B., Hobbie, S. E., Larson, K. L., Morse, J. L., Neill, C., Nelson, K., O'Neil-Dunne, J., Ogden, L., Pataki, D. E., Polsky, C., Chowdhury, R. R., & Steele, M. K. (2014). Ecological homogenization of urban USA. *Frontiers in Ecology and the Environment*, *12*(1), 74–81. https://doi.org/10.1890/120374

Gudynas, E. (2009). *El Mandato Ecológico. Derechos de la Naturaleza y Políticas Ambientales en la Nueva Constitución*. Quito, Ecuador: Abya Yala.

Gudynas, E. (2011). Buen vivir: germinando alternativas al desarrollo. *Revista América Latina En Movimiento*, *462*, 1–20.

Guerrero, P. C., Durán, A. P., & Walter, H. E. (2011). Latitudinal and altitudinal patterns of the endemic cacti from the Atacama desert to Mediterranean Chile. *Journal of Arid Environments*, *75*(11), 991–997. http://doi.org/10.1016/j.jaridenv.2011.04.036

Guido, A., & Pillar, V. D. (2017). Invasive plant removal: assessing community impact and recovery from invasion. *Journal of Applied Ecology*, *54*(4), 1230–1237. http://doi.org/10.1111/1365-2664.12848

Güneralp, B., & Seto, K. C. (2013). Futures of global urban expansion: uncertainties and implications for biodiversity conservation. *Environmental Research Letters*, *8*(1), 14025. http://doi.org/10.1088/1748-9326/8/1/014025

Guns, M., & Vanacker, V. (2013). Forest cover change trajectories and their impact on landslide occurrence in the tropical Andes. *Environmental Earth Sciences*, *70*(7), 2941–2952. http://doi.org/10.1007/s12665-013-2352-9

Guo, L. B., & Gifford, R. M. (2002). Soil carbon stocks and land use change: a meta analysis. *Global Change Biology*, *8*(4), 345–360. http://doi.org/10.1046/j.1354-1013.2002.00486.x

Gutiérrez, E. E., & Marinho-Filho, J. (2017). The mammalian faunas endemic to the Cerrado and the Caatinga. *ZooKeys*, (644), 105–157. http://doi.org/10.3897/zookeys.644.10827

Gutiérrez, R., & Schafleitner, R. (2007). *Caracterización Morfofisiológica, Molecular y de Procesamiento para Cultivares de Papas Nativas en la Provincia de Canchis, Cusco. Reporte de Investigación para Soluciones Prácticas-ITDG y Centro Internacional de la Papa*. Lima, Perú.

Haag, W. R. (2009). Past and future patterns of freshwater mussel extinctions in North America during the Holocene. In S. Turvey (Ed.), *Holocene Extinctions* (pp. 107–128). Oxford, UK: Oxford University Press.

Haberl, H., Erb, K. H., Krausmann, F., Gaube, V., Bondeau, A., Plutzar, C., Gingrich, S., Lucht, W., & Fischer-Kowalski, M. (2007). Quantifying and mapping the human appropriation of net primary production in earth's terrestrial ecosystems. *PNAS*, *104*(31), 12942–12947. https://doi.org/10.1073/pnas.0704243104

Haddad, N. M., Brudvig, L. A., Clobert, J., Davies, K. F., Gonzalez, A., Holt, R. D., Lovejoy, T. E., Sexton, J. O., Austin, M. P., Collins, C. D., Cook, W. M., Damschen, E. I., Ewers, R. M., Foster, B. L., Jenkins, C. N., King, A. J., Laurance, W. F., Levey, D. J., Margules, C. R., Melbourne, B. A., Nicholls, A. O., Orrock, J. L., Song, D.-X., & Townshend, J. R. (2015). Habitat fragmentation and its lasting impact on Earth's ecosystems. *Science Advances*, *1*(2), e1500052. https://doi.org/10.1126/sciadv.1500052

Hale, C. M., Frelich, L. E., Reich, P. B., & Pastor, J. (2005). Effects of European earthworm invasion on soil characteristics in northern hardwood forests of Minnesota, USA. *Ecosystems*, *8*(8), 911–927. http://doi.org/10.1007/s10021-005-0066-x

Hall, S. J., Learned, J., Ruddell, B., Larson, K. L., Cavender-Bares, J., Bettez, N., Groffman, P. M., Grove, J. M., Heffernan, J. B., Hobbie, S. E., Morse, J. L., Neill, C., Nelson, K. C., ONeil-Dunne, J. P. M., Ogden, L., Pataki, D. E., Pearse, W. D., Polsky, C., Chowdhury, R. R., Steele, M. K., & Trammell, T. L. E. (2016). Convergence of microclimate in residential landscapes across diverse cities in the United States. *Landscape Ecology*, *31*(1), 101–117. https://doi.org/10.1007/s10980-015-0297-y

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M. P., Perry, M. T., Selig, E. R., Spaldin, M., Steneck, R., & Watson, R. (2008). A global map of human impact on marine ecosystems. *Science*, *319*(5865), 948–952. https://doi.org/10.1126/science.1149345

Halsey, R., & Syphard, A. (2016). High-severity fire in chaparral: cognitive dissonance in the shrublands. In D. A. DellaSala & C. T. Hanson (Eds.), *The Ecological Importance of Mixed-Severity Fires: Nature's Phoenix* (p. 177–209.). New York, USA: Elsevier.

Hamer, A. J., & McDonnell, M. J. (2008). Amphibian ecology and conservation in the urbanising world: A review. *Biological Conservation*, *141*(10), 2432–2449. http://doi.org/10.1016/j.biocon.2008.07.020

Hammel, B. (1990). The distribution of diversity among families, genera and habitat types in the La Selva Flora. In A. H. Gentry (Ed.), *Four Neotropical Rainforests* (pp. 75–84). New Haven, USA: Yale University Press.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O., & Townshend, J. R. G. (2013). High-resolution global maps of 21st-century forest cover change. *Science*, *342*(6160), 850–853. https://doi.org/10.1126/science.1244693

Harlan, J. (1961). Geographic origin of plants useful to agriculture. In *Germplasm Resources* (pp. 3–19). Washington D.C., USA: American Association for the Advancement of Science.

Harlan, J. R. (1971). Agricultural origins: centers and noncenters. *Science*, *174*(4008), 468–474. http://doi.org/10.1126/science.174.4008.468

Harris, J. B. C., Reid, J. L., Scheffers, B. R., Wanger, T. C., Sodhi, N. S., Fordham, D. A., & Brook, B. W. (2012). Conserving imperiled species: a comparison of the IUCN Red List and U.S. Endangered Species Act. *Conservation Letters*, *5*(1), 64–72.

Harsch, M. A., Hulme, P. E., McGlone, M. S., & Duncan, R. P. (2009). Are treelines advancing? A global meta-analysis of treeline response to climate warming. *Ecology Letters*, *12*(10), 1040–1049.

Harvell, C. D. (1999). Emerging marine diseases--climate links and anthropogenic factors. *Science*, *285*(5433), 1505–1510. http://doi.org/10.1126/science.285.5433.1505

Hatcher, M. J., Dick, J. T. A., & Dunn, A. M. (2012). Disease emergence and invasions. *Functional Ecology*, *26*(6), 1275–1287. http://doi.org/10.1111/j.1365-2435.2012.02031.x

Hautier, Y., Seabloom, E. W., Borer, E. T., Adler, P. B., Harpole, W. S., Hillebrand, H., Lind, E. M., MacDougall, A. S., Stevens, C. J., Bakker, J. D., Buckley, Y. M., Chu, C., Collins, S. L., Daleo, P., Damschen, E. I., Davies, K. F., Fay, P. A., Firn, J., Gruner, D. S., Jin, V. L., Klein, J. A., Knops, J. M. H., La Pierre, K. J., Li, W., McCulley, R. L., Melbourne, B. A., Moore, J. L., O'Halloran, L. R., Prober, S. M., Risch, A. C., Sankaran, M., Schuetz, M., & Hector, A. (2014). Eutrophication weakens stabilizing effects of diversity in natural grasslands. *Nature*, *508*(7497), 521–525. https://doi.org/10.1038/nature13014

Hayes, D. J., Kicklighter, D. W., McGuire, A. D., Chen, M., Zhuang, Q., Yuan, F., Melillo, J. M., & Wullschleger, S. D. (2014). The impacts of recent permafrost thaw on land-atmosphere greenhouse gas exchange. *Environmental Research Letters*, *9*(4), 045005. https://doi.org/10.1088/1748-9326/9/4/045005

Heath, L. S., Smith, J. E., Skog, K. E., Nowak, D. J., & Woodall, C. W. (2011). Managed forest carbon estimates for the US greenhouse gas inventory, 1990-2008. *Journal of Forestry*, *109*(3), 167–173.

Hector, A., Bell, T., Hautier, Y., Isbell, F., Kéry, M., Reich, P. B., van Ruijven, J., & Schmid, B. (2011). BUGS in the analysis of biodiversity experiments: species richness and composition are of similar importance for grassland productivity. *PLoS One*, *6*(3), e17434. https://doi.org/10.1371/journal.pone.0017434

Hendrix, P. F., Baker, G. H., Calaham, M. A., Damoff, G. A., Fragoso, C., Gonzalez, G., James, S. W., Lachnicht, S. L., Winsome, T., & Zou, X. (2006). Invasion of exotic earthworms into ecosystems inhabited by native earthworms. *Biological Invasions*, *8*(6), 1287–1300. https://doi.org/10.1007/978-1-4020-5429-7_9

Herms, D. A., & McCullough, D. G. (2014). Emerald ash borer invasion of North America: history, biology, ecology, impacts, and management. *Annual Review of Entomology*, *59*, 13–30. http://doi.org/10.1146/annurev-ento-011613-162051

Hernández, A., Miranda, M. D., Arellano, E. C., & Dobbs, C. (2016). Landscape trajectories and their effect on fragmentation for a Mediterranean semi-arid ecosystem in Central Chile. *Journal of Arid Environments*, *127*, 74–81.

Hernández, G. (Ed.). (1999). *Mesoamerican Wetlands. Ramsar sites in Central America and Mexico*. San José, Costa Rica: Unión Mundial para la Naturaleza (UICN)/ ORMA.

Hernández, X. (1985). Biología agrícola: los conocimientos biológicos y su aplicación a la agricultura. México: CECSA.

Hicke, J. A., Asner, G. P., Randerson, J. T., Tucker, C. J., Los, S., Birdsey, R. A., Jenkins, J. C., & Field, C. B. (2002). Trends in North American net primary productivity derived from satellite observations, 1982-1998. *Global Biogeochemical Cycles*, *16*(2), 1019. https://doi.org/10.1029/2001GB001550

Higgins, S. N., & Vander Zanden, M. J. (2010). What a difference a species makes: a meta-analysis of dreissenid mussel impacts on freshwater ecosystems. *Ecological Monographs*, *80*(2), 179–196.

Highly Migratory Species Management Division. (2006). *SEDAR 11: Stock Assessment Report: Large Coastal Shark Complex, Blacktip and Sandbar Shark*. Silver Spring, USA: National Marine Fisheries Service (NMFS). Retrieved from http://sedarweb.org/docs/sar/Final_LCS_SAR.pdf

Hijmans, R. J. (2016). Raster: Geographic Data Analysis and Modeling. R package version 2.5-8. Retrieved from https://cran.r-project.org/package=raster

Himsworth, C. G., Parsons, K. L., Jardine, C., & Patrick, D. M. (2013). Rats, cities, people, and pathogens: A systematic review and narrative synthesis of literature regarding the ecology of rat-associated zoonoses in urban centers. *Vector-Borne and Zoonotic Diseases*, *13*(6), 349–359. http://doi.org/10.1089/vbz.2012.1195

Hinkel, K. M., Eisner, W. R., & Kim, C. J. (2017). Detection of tundra trail damage near Barrow, Alaska using remote imagery. *Geomorphology*, *293*, 360–367. Retrieved from https://doi.org/10.1016/j.geomorph.2016.09.013

Hinzman, L. D., Bettez, N. D., Bolton, W. R., Chapin, F. S., Dyurgerov, M. B., Fastie, C. L., Griffith, B., Hollister, R. D., Hope, A., Huntington, H. P., Jensen, A. M., Jia, G. J., Jorgenson, T., Kane, D. L., Klein, D. R., Kofinas, G., Lynch, A. H., Lloyd, A. H., McGuire, A. D., Nelson, F. E., Oechel, W. C., Osterkamp, T. E., Racine, C. H., Romanovsky, V. E., Stone, R. S., Stow, D. A., Sturm, M., Tweedie, C. E., Vourlitis, G. L., Walker, M. D., Walker, D. A., Webber, P. J., Welker, J. M., Winker, K. S., & Yoshikawa, K. (2005). Evidence and implications of recent climate change in northern Alaska and other Arctic regions.

Climatic Change, *72*(3), 251–298. https://doi.org/10.1007/s10584-005-5352-2

Hobbs, R. J., & Mooney, H. A. (1998). 1998 report. Retrieved from http://www.dfg.ca.gov/wildlife/nongame/t_e_spp/

Hoekstra, J. M., Boucher, T. M., Ricketts, T. H., & Roberts, C. (2005). Confronting a biome crisis: global disparities of habitat loss and protection. *Ecology Letters*, *8*(1), 23–29. http://doi.org/10.1111/j.1461-0248.2004.00686.x

Hoekstra, J. M., Molnar, J. L., Jennings, M., Revenga, C., Spalding, M. D., Boucher, T. M., Robertson, J. C., Heibel, T. J., & Ellison, K. (2010). *The Atlas of Global Conservation: Changes, Challenges, and Opportunities to Make a Difference*. (J. L. Molnar, Ed.). Berkeley, USA: University of California Press.

Hofstede, R., Calles, J., López, V., Polanco, R., Torres, F., Ulloa, J., Vásquez, A., & Cerra, M. (2014). *Los Páramos Andinos ¿Qué Sabemos? Estado de Conocimiento sobre el Impacto del Cambio Climático en el Ecosistema Páramo*. Quito, Ecuador: UICN. Retrieved from https://portals.iucn.org/library/sites/library/files/documents/2014-025.pdf

Hofstede, R. G. M., Groenendijk, J. P., Coppus, R., Fehse, J. C., & Sevink, J. (2002). Impact of pine plantations on soils and vegetation in the Ecuadorian high Andes. *Mountain Research and Development*, *22*(2), 159–167. http://doi.org/10.1659/0276-4741(2002)022%5B0159:IOPPOS%5D2.0.CO;2

Holdsworth, A. R., Frelich, L. E., & Reich, P. B. (2007). Effects of earthworm invasion on plant species richness in northern hardwood forests. *Conservation Biology*, *21*(4), 997–1008. http://doi.org/10.1111/j.1523-1739.2007.00740.x

Holmlund, C. M., & Hammer, M. (1999). Ecosystem services generated by fish populations. *Ecological Economics*, *29*(2), 253–268. http://doi.org/10.1016/S0921-8009(99)00015-4

Hooper, D. U., Adair, E. C., Cardinale, B. J., Byrnes, J. E. K., Hungate, B. A., Matulich, K. L., Gonzalez, A., Duffy, J. E., Gamfeldt, L., & O'Connor, M. I. (2012). A global synthesis reveals biodiversity loss as a major driver of ecosystem change. *Nature*, *486*(7401), 105–108. https://doi.org/10.1038/nature11118

Hope, D., Gries, C., Zhu, W., Fagan, W. F., Redman, C. L., Grim, N. B., Nelson, A. L., Martin, C., & Kinzig, A. (2003). Socioeconomics drive urban plant diversity. *PNAS*, *100*(15), 8788–8792. https://doi.org/10.1073/pnas.1537557100

Hourigan, T. F. (1999). Conserving ocean biodiversity: trends and challenges. In B. Cicin-Sain, R. W. Knecht, & N. Foster (Eds.), *Trends and Future Challenges for U.S. National Ocean and Coastal Policy*. Washington D.C., USA: Proceedings of a Workshop.

Houspanossian, J., Giménez, R., Jobbágy, E., & Nosetto, M. (2017). Surface albedo raise in the South American Chaco: combined effects of deforestation and agricultural changes. *Agricultural and Forest Meteorology*, *232*, 118–127. http://doi.org/10.1016/j.agrformet.2016.08.015

Houtart, F. (2014). El concepto de sumak kawsay (Buen vivir) y su correspondencia con el bien común de la humanidad. In G. C. Delgado (Ed.), *Buena Vida, Buen Vivir: Imaginarios Alternativos Para el Bien Común de la Humanidad* (pp. 97–123). México D.F., México: Universidad Nacional Autónoma de México.

Hoverman, J. T., Mihaljevic, J. R., Richgels, K. L. D., Kerby, J. L., & Johnson, P. T. J. (2012). Widespread co-occurrence of virulent pathogens within California amphibian communities. *EcoHealth*, *9*(3), 288–292. http://doi.org/10.1007/s10393-012-0778-2

Howard, B. C. (2016). Pablo Escobar's escaped hippos are thriving in Colombia. *National Geographic*. Retrieved from https://news.nationalgeographic.com/2016/05/160510-pablo-escobar-hippos-colombia/

Hoyos, L. E., Cingolani, A. M., Zak, M. R., Vaieretti, M. V., Gorla, D. E., & Cabido, M. R. (2013). Deforestation and precipitation patterns in the arid Chaco forests of central Argentina. *Applied Vegetation Science*, *16*(2), 260–271. http://doi.org/10.1111/j.1654-109X.2012.01218.x

Hribljan, J. A., Cooper, D. J., Sueltenfuss, J., Wolf, E. C., Heckman, K. A., Lilleskov, E. A., & Chimner, R. A. (2015). Carbon storage and long-term rate of accumulation in high-altitude Andean peatlands of Bolivia. *Mires and Peat*, *15*(12), 1–14. Retrieved from http://www.mires-and-peat.net/.

Hribljan, J. A., Suárez, E., Heckman, K. A., Lilleskov, E. A., & Chimner, R. A. (2016). Peatland carbon stocks and accumulation rates in the Ecuadorian páramo. *Wetlands Ecology and Management*, *24*(2), 113–127. http://doi.org/10.1007/s11273-016-9482-2

Huamantupa, I., Cuba, M., Urrunaga, R., Paz, E., Ananya, N., Callalli, M., Pallquil, N., & Coasaca, H. (2011). Richness, use and origin of expended medicinal plants in the markets of the Cusco City. *Revista Peruana de Biologia*, *18*(3), 283–291.

Hucke-Gaete, R., Crespo, E., & Schlatter, R. P. (Eds.). (2004). *Aquatic Mammals in Latin America: Proceedings of a Workshop on Identifying High-Priority Conservation Needs and Actions*. Bonn, Germany: UNEP/CMS Secretariat. Retrieved from http://www.iucn-csg.org/wp-content/uploads/2010/03/final-report-cms-latam-workshop.pdf

Hugelius, G., Tarnocai, C., Broll, G., Canadell, J. G., Kuhry, P., & Swanson, D. K. (2013). The northern circumpolar soil carbon database: Spatially distributed datasets of soil coverage and soil carbon storage in the northern permafrost regions. *Earth System Science Data*, *5*(1), 3–13. http://doi.org/10.5194/essd-5-3-2013

Hung, H., Kallenborn, R., Breivik, K., Su, Y., Brorström-Lundén, E., Olafsdottir, K., Thorlacius, J. M., Leppänen, S., Bossi, R., Skov, H., Manø, S., Patton, G. W., Stern, G., Sverko, E., & Fellin, P. (2010). Atmospheric monitoring of organic pollutants in the Arctic under the Arctic Monitoring and Assessment Programme (AMAP): 1993-2006. *Science of the Total Environment*, *408*(15), 2854–2873. https://doi.org/10.1016/j.scitotenv.2009.10.044

Hunke, P., Mueller, E. N., Schröder, B., & Zeilhofer, P. (2015). The Brazilian Cerrado: assessment of water and soil degradation in catchments under intensive agriculture use. *Ecohydrology*, *8*(6), 1154–1180.

Hutchings, J. A., Côté, I. M., Dodson, J. J., Fleming, I. A., Jennings, S., Mantua, N. J., Peterman, R. M., Riddell, B. E., Weaver, A. J., & VanderZwaag, D. L. (2012). *Sustaining Canada's Marine Biodiversity: Responding to the Challenges Posed by Climate Change, Fisheries, and Aquaculture*. Ottawa, Canada: Royal Society of Canada. Retrieved from http://www.ianas.org/docs/books/wbp08.pdf

I3N. (2016). I3N National Database: Argentina, Brasil, Chile, Colombia, Costa Rica, Jamaica, and Uruguay. I3N National Database.

Ide, S. M., Ruiz, C. G., Sandoval, A. C., & Valenzuela, J. E. (2011). Detección de *Thaumastocoris peregrinus* (Hemiptera: Thaumastocoridae) asociado a *Eucalyptus* spp. en Chile. *Bosque*, *32*(3), 309–313. http://doi.org/10.4067/S0717-92002011000300012

Ingrao, D. A., Mikkelson, P. M., & Hicks, D. W. (2001). Another introduced marine mollusk in the Gulf of Mexico: the indo-Pacific green mussel, *Perna viridis*, in Tampa Bay, Florida. *Journal of Shellfish Research*, *20*(1), 13–19.

Instituto de Asuntos Públicos-Centro de Análisis de Políticas Públicas. (2016). *Informe País: Estado del Medio Ambiente en Chile. Comparación 1999 - 2015*. Santiago, Chile: Universidad de Chile. Retrieved from http://www.uchile.cl/publicaciones/129607/informe-pais-estado-del-medio-ambiente-en-chile-1999-2015

International Labour Organization. (1989). *Indigenous and Tribal Peoples Convention, C169*. International Labour Organization (ILO). Retrieved from http://www.refworld.org/docid/3ddb6d514.html

Iñiguez-Armijos, C., Leiva, A., Frede, H. G., Hampel, H., & Breuer, L. (2014). Deforestation and benthic indicators: how much vegetation cover is needed to sustain healthy Andean streams? *PLoS One*, *9*(8), e105869. http://doi.org/10.1371/journal.pone.0105869

Iriarte, J. A., Lobos, G. A., & Jaksic, F. M. (2005). Invasive vertebrate species in Chile and their control and monitoring by governmental agencies. *Revista Chilena de Historia Natural*, *78*(1), 143–154. http://doi.org/10.4067/s0716-078x2005000100010

Iroumé, A., & Palacios, H. (2013). Afforestation and changes in forest composition affect runoff in large river basins with pluvial regime and Mediterranean climate, Chile. *Journal of Hydrology*, *505*, 113–125. http://doi.org/10.1016/j.jhydrol.2013.09.031

Isaac, N. J. B., Turvey, S. T., Collen, B., Waterman, C., & Baillie, J. E. M. (2007). Mammals on the EDGE: conservation priorities based on threat and phylogeny. *PLoS ONE*, *2*(3), e296. http://doi.org/10.1371/journal.pone.0000296

Isbell, F. I., Polley, H. W., & Wilsey, B. J. (2009). Species interaction mechanisms maintain grassland plant species diversity. *Ecology*, *90*(7), 1821–1830. http://doi.org/10.1890/08-0514.1

Iturraspe, R. (2010). *Las turberas de Tierra del Fuego y el Cambio Climático Global*. Buenos Aires, Argentina: Fundación Humedales / Wetlands International.

IUCN. (2009). The IUCN Red list of Threatened Species Version 2009-1. Retrieved from http://www.iucnredlist.org/

IUCN. (2014). IUCN Red List of Threatened Species. Version 2014.2. Retrieved August 31, 2017, from http://www.iucnredlist.org

IUCN. (2017). The IUCN Red List of Threatened Species. Version 2016-3. Retrieved January 1, 2017, from http://www.iucnredlist.org

IUCN, & SSC PBSG. (2017). Summary of polar bear population status per 2017. Retrieved from http://pbsg.npolar.no/en/status/status-table.html

Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. (Eds.). (2014). *Status and Trends of Caribbean Coral Reefs: 1970-2012*. Gland, Switzerland: Global Coral Reef Monitoring Network, IUCN.

Jackson, R. B., Banner, J. L., Jobbágy, E. G., Pockman, W. T., & Wall, D. H. (2002). Ecosystem carbon loss with woody plant invasion of grasslands. *Nature*, *418*(6898), 623–626. http://doi.org/10.1038/nature00910

Jackson, R. B., Jobbágy, E. G., & Avissar, R. (2005). Trading water for carbon with biological carbon sequestration. *Science*, *310*(5756), 1944–1947. http://doi.org/10.1126/science.1119282

Jackson, R. B., Lajtha, K., Crow, S. E., Hugelius, G., Kramer, M. G., & Piñeiro, G. (2017). The ecology of soil carbon: pools, vulnerabilities, and biotic and abiotic controls. *Annual Review of Ecology, Evolution, and Systematics*, *48*(1), 419–445. http://doi.org/10.1146/annurev-ecolsys-112414-054234

Jackson, R. B., Randerson, J. T., Canadell, J. G., Anderson, R. G., Avissar, R., Baldocchi, D. D., Bonan, G. B., Caldeira, K., Diffenbaugh, N. S., Field, C. B., Hungate, B. A., Jobbágy, E. G., Kueppers, L. M., Nosetto, M. D., & Pataki, D. E. (2008). Protecting climate with forests. *Environmental Research Letters*, *3*(4), 044006. https://doi.org/10.1088/1748-9326/3/4/044006

Jaksic, F. M. (1998). Vertebrate invaders and their ecological impacts in Chile. *Biodiversity and Conservation*, *7*(11), 1427–1445. http://doi.org/10.1023/A:1008825802448

Jaksic, F. M., Pavez, E. F., Jiménez, J. E., & Torres-Mura, J. C. (2001). The conservation status of raptors in the Metropolitan Region, Chile. *Journal of Raptor Research*, *35*(2), 151–158.

Janzen, D. H. (1987). Insect diversity of a Costa Rican dry forest: why keep it, and how? *Biological Journal of the Linnean Society*, *30*(4), 343–356. http://doi.org/10.1111/j.1095-8312.1987.tb00307.x

Janzen, D. H. (1988). Tropical dry forests: the most endangered major tropical ecosystem. In E. O. Wilson & F. M. Peters (Eds.), *Biodiversity* (pp. 130–137). Washington, D.C.: National Academy Press.

Janzen, D. H. (2002). Tropical Dry Forest: Area de Conservación Guanacaste, northwetern Costa Rica. In M. R. Perrow & A. J. Davy (Eds.), *Handbook of Ecological Restoration. Vol 2 Restoration in Practice* (pp. 559–583). Cambridge, UK: Cambridge University Press.

Janzen, D. H., & Hallwachs, W. (2016). Biodiversity conservation history and future in Costa Rica: The case of Área de Conservación Guanacaste (ACG). In M. Kappelle (Ed.), *Costa Rican Ecosystems* (pp. 290–341). Chicago, USA.: University

Of Chicago Press. http://doi.org/10.7208/chicago/9780226121642.003.0010

Jenkins, C. N., & Joppa, L. (2009). Expansion of the global terrestrial protected area system. *Biological Conservation*, *142*(10), 2166–2174. http://doi.org/10.1016/j.biocon.2009.04.016

Jetz, W., Cavender-Bares, J., Pavlick, R., Schimel, D., Davis, F. W., Asner, G. P., Guralnick, R., Kattge, J., Latimer, A. M., Moorcroft, P., Schaepman, M. E., Schildhauer, M. P., Schneider, F. D., Schrodt, F., Stahl, U., & Ustin, S. L. (2016). Monitoring plant functional diversity from space. *Nature Plants*, *2*(3), 16024. https://doi.org/10.1038/nplants.2016.24

Jetz, W., Thomas, G. H., Joy, J. B., Hartmann, K., & Mooers, A. O. (2012). The global diversity of birds in space and time. *Nature*, *491*(7424), 444–448.

Jetz, W., Thomas, G. H., Joy, J. B., Redding, D. W., Hartmann, K., & Mooers, A. O. (2014). Global distribution and conservation of evolutionary distinctness in birds. *Current Biology*, *24*(9), 919–930. http://doi.org/10.1016/j.cub.2014.03.011

Jiménez, A., Pauchard, A., Cavieres, L. A., Marticorena, A., & Bustamante, R. O. (2008). Do climatically similar regions contain similar alien floras? A comparison between the mediterranean areas of central Chile and California. *Journal of Biogeography*, *35*(4), 614–624. http://doi.org/10.1111/j.1365-2699.2007.01799.x

Jiménez, J. E., Crego, R. D., Soto, G. E., Roman, I., Rozzi, R., & Vergara, P. M. (2014). Potential impact of the alien American mink (*Neovison vison*) on Magellanic woodpeckers (Campephilus magellanicus) in Navarino Island, Southern Chile. *Biological Invasions*, *16*(4), 961–966. http://doi.org/10.1007/s10530-013-0549-1

Jiménez, M. A., Jaksic, F. M., Armesto, J. J., Gaxiola, A., Meserve, P. L., Kelt, D. A., & Gutiérrez, J. R. (2011). Extreme climatic events change the dynamics and invasibility of semi-arid annual plant communities. *Ecology Letters*, *14*(12), 1227–1235. http://doi.org/10.1111/j.1461-0248.2011.01693.x

Johnson, A. M., & Karels, T. J. (2016). Partitioning the effects of habitat fragmentation on rodent species richness in an urban landscape. *Urban Ecosystems*, *19*(2), 547–560. http://doi.org/10.1007/s11252-015-0513-1

Johnson, E. A., & Miyanishi, K. (2008). Creating new landscapes and ecosystems. *Annals of the New York Academy of Sciences*, *1134*(1), 120–145. http://doi.org/10.1196/annals.1439.007

Joly, C. A., Metzger, J. P., & Tabarelli, M. (2014). Experiences from the Brazilian Atlantic Forest: ecological findings and conservation initiatives. *New Phytologist*, *204*(3), 459–473.

Jones, K. E., Patel, N. G., Levy, M. A., Storeygard, A., Balk, D., Gittleman, J. L., & Daszak, P. (2008). Global trends in emerging infectious diseases. *Nature*, *451*(7181), 990–993. http://doi.org/10.1038/nature06536

Jørgensen, P. M., Nee, M. H., & Beck, S. G. (2015). Catálogo de plantas vasculares de Bolivia. In *Monographs in Systematic Botany from the Missouri Botanical Garden* (4th ed., Vol. 127). Missouri,USA: Missouri Botanical Garden Press.

Jose, S., Gold, M. A., & Garrett, H. E. (2012). The future of temperate agroforestry in the United States. In P. K. R. Nair & D. Garrity (Eds.), *Agroforestry - The Future of Global Land Use* (pp. 217–245). New York, USA: Springer.

Junk, W. J. (2007). Freshwater fishes of South America: their biodiversity, fisheries, and habitats- a synthesis. *Aquatic Ecosystem Health & Management*, *10*(2), 228–242.

Junk, W. J., Brown, M., Campbell, I. C., Finlayson, M., Gopal, B., Ramberg, L., & Warner, B. G. (2006). The comparative biodiversity of seven globally important wetlands: a synthesis. *Aquatic Sciences*, *68*(3), 400–414. http://doi.org/10.1007/s00027-006-0856-z

Junk, W. J., Piedade, M. T. F., Lourival, R., Wittmann, F., Kandus, P., Lacerda, L. D., Bozelli, R. L., Esteves, F. A., Nunes da Cunha, C., Maltchik, L., Schöngart, J., Schaeffer-Novelli, Y., & Agostinho, A. A. (2014). Brazilian wetlands: their definition, delineation, and classification for research, sustainable management, and protection. *Aquatic Conservation: Marine and Freshwater Ecosystems*, *24*(1), 5–22. https://doi.org/10.1002/aqc.2386

Junk, W. J., Soares, M. G. M., & Bayley, P. B. (2007). Freshwater fishes of the Amazon River basin: their biodiversity, fisheries, and habitats. *Aquatic Ecosystem Health & Management*, *10*(2), 153–173. http://doi.org/10.1080/14634980701351023

Juri, M. D., & Chani, J. M. (2009). Variación estacional en la composición de las comunidades de aves en un gradiente urbano. *Ecología Austral*, *19*(3), 175–184.

Juzwik, J., Appel, D. N., MacDonald, W. L., & Burks, S. (2011). Challenges and successes in managing oak wilt in the United States. *Plant Disease*, *95*(8), 888–900. http://doi.org/10.1094/PDIS-12-10-0944

Kairo, M., Ali, B., Cheesman, O., Haysom, K., & Murphy, S. (2003). *Invasive Species Threats in the Caribbean Region, Report to the Nature Conservancy*. Arlington, USA: CAB International.

Kane, V. R., Cansler, C. A., Povak, N. A., Kane, J. T., McGaughey, R. J., Lutz, J. A., Churchill, D. J., & North, M. P. (2015). Mixed severity fire effects within the Rim fire: relative importance of local climate, fire weather, topography, and forest structure. *Forest Ecology and Management*, *358*, 62–79. https://doi.org/10.1016/j.foreco.2015.09.001

**Kattge, J., Díaz, S., Lavorel, S., Prentice, I. C., Leadley, P., Bönisch, G., Garnier, E., Westoby, M., Reich, P. B., Wright, I. J., Cornelissen, J. H. C., Violle, C., Harrison, S. P., Van Bodegom, P. M., Reichstein, M., Enquist, B. J., Soudzilovskaia, N. A., Ackerly, D. D., Anand, M., Atkin, O., Bahn, M., Baker, T. R., Baldocchi, D., Bekker, R., Blanco, C. C., Blonder, B., Bond, W. J., Bradstock, R., Bunker, D. E., Casanoves, F., Cavender-Bares, J., Chambers, J. Q., Chapin, F. S., Chave, J., Coomes, D., Cornwell, W. K., Craine, J. M., Dobrin, B. H., Duarte, L., Durka, W., Elser, J., Esser, G., Estiarte, M., Fagan, W. F., Fang, J., Fernández-Méndez, F., Fidelis, A., Finegan, B., Flores, O., Ford, H., Frank, D., Freschet, G. T., Fyllas,

N. M., Gallagher, R. V., Green, W. A., Gutierrez, A. G., Hickler, T., Higgins, S. I., Hodgson, J. G., Jalili, A., Jansen, S., Joly, C. A., Kerkhoff, A. J., Kirkup, D., Kitajima, K., Kleyer, M., Klotz, S., Knops, J. M. H., Kramer, K., Kühn, I., Kurokawa, H., Laughlin, D., Lee, T. D., Leishman, M., Lens, F., Lenz, T., Lewis, S. L., Lloyd, J., Llusià, J., Louault, F., Ma, S., Mahecha, M. D., Manning, P., Massad, T., Medlyn, B. E., Messier, J., Moles, A. T., Müller, S. C., Nadrowski, K., Naeem, S., Niinemets, Ü., Nöllert, S., Nüske, A., Ogaya, R., Oleksyn, J., Onipchenko, V. G., Onoda, Y., Ordoñez, J., Overbeck, G., Ozinga, W. A., Patiño, S., Paula, S., Pausas, J. G., Peñuelas, J., Phillips, O. L., Pillar, V., Poorter, H., Poorter, L., Poschlod, P., Prinzing, A., Proulx, R., Rammig, A., Reinsch, S., Reu, B., Sack, L., Salgado-Negret, B., Sardans, J., Shiodera, S., Shipley, B., Siefert, A., Sosinski, E., Soussana, J. F., Swaine, E., Swenson, N., Thompson, K., Thornton, P., Waldram, M., Weiher, E., White, M., White, S., Wright, S. J., Yguel, B., Zaehle, S., Zanne, A. E., & Wirth, C. (2011). TRY- a global database of plant traits. *Global Change Biology*, *17*(9), 2905–2935. https://doi.org/10.1111/j.1365-2486.2011.02451.x

Keddy, P. A., Fraser, L. H., Solomeshch, A. I., Junk, W. J., Campbell, D. R., Arroyo, M. T. K., & Alho, C. J. R. (2009). Wet and wonderful: the world's largest wetlands are conservation priorities. *BioScience*, *59*(1), 39–51. http://doi.org/10.1525/bio.2009.59.1.8

Keeley, J. E., & Zedler, P. H. (1998). Characterization and global distribution of vernal pools. In C. W. Witham (Ed.), *Ecology, Conservation, and Management of Vernal Pool Ecosystems* (pp. 1–14). Sacramento, USA: California Native Plant Society.

Keenan, R., Reams, G., Achard, F., De-Freitas, J., Grainger, A., & Lindquist, E. (2015). Dynamics of global forest area: results from the FAO Global Forest Resources Assessment 2015. *Forest Ecology and Management*, *352*, 9–20. http://doi.org/10.1016/j.foreco.2015.06.014

Keesing, F., Belden, L. K., Daszak, P., Dobson, A., Harvell, C. D., Holt, R. D., Hudson, P., Jolles, A., Jones, K. E., Mitchell, C. E., Myers, S. S., Bogich, T., & Ostfeld, R. S. (2010). Impacts of biodiversity on the emergence and transmission of infectious diseases. *Nature*, *468*(7324), 647–652. https://doi.org/10.1038/nature09575

Kéfi, S., Rietkerk, M., Alados, C. L., Pueyo, Y., Papanastasis, V. P., ElAich, A., & De Ruiter, P. C. (2007). Spatial vegetation patterns and imminent desertification in Mediterranean arid ecosystems. *Nature*, *449*(7159), 213–217. http://doi.org/10.1038/nature06111

Kelehear, C., Saltonstall, K., & Torchin, M. E. (2015). An introduced pentastomid parasite (*Raillietiella frenata*) infects native cane toads (*Rhinella marina*) in Panama. *Parasitology*, *142*(5), 675–679. http://doi.org/10.1017/S0031182014001759

Kelly, B. C., & Gobas, F. A. P. C. (2001). Bioaccumulation of persistent organic pollutants in lichen-caribou-wolf food chanins of Canada's central and western Arctic. *Environmental Science & Technology*, *35*(2), 325–334. http://doi.org/10.1021/es0011966

Kelly, D. W. (2007). *Vectors and Pathways for Nonindigenous Aquatic Species in the Great Lakes*. Washington, DC., USA: Prepared for Committee on the St. Lawrence Seaway: Options to Eliminate Introduction of Nonindigenous Species into the Great Lakes, Phase 2 Transportation Research Board and Division on Earth and Life Studies.

Kelly, D. W., Lamberti, G. A., & MacIsaac, H. J. (2009). The Laurentian Great Lakes as a case study of biological invasion. In R. P. Keller, D. M. Lodge, M. A. Lewis, & J.F. Shogren (Eds.), *Bioeconomics of Invasive Species: Integrating Ecology, Economics, Policy and Management* (pp. 205–225). Oxford, UK: Oxford University Press.

Kembel, S. W., Cowan, P. D., Helmus, M. R., Cornwell, W. K., Morlon, H., Ackerly, D. D., Blomberg, S. P., & Webb, C. O. (2010). Picante: R tools for integrating phylogenies and ecology. *Bioinformatics*, *26*(11), 1463–1464.

Kerswell, A. P. (2006). Global biodiversity patterns of benthic marine algae. *Ecology*, *87*(10), 2479–2488. http://doi.org/10.1890/0012-9658(2006)87%5B2479:GBPOBM%5D2.0.CO;2

Kessler, M. (2001). Patterns of diversity and range size of selected plant groups along an elevational transect in the Bolivian Andes. *Biodiversity and Conservation*, *10*(11), 1897–1921. http://doi.org/10.1023/A:1013130902993

Keuroghlian, A., Andrade Santos, M. D. C., & Eaton, D. P. (2015). The effects of deforestation on white-lipped peccary (*Tayassu pecari*) home range in the southern Pantanal. *Mammalia*, *79*(4), 491–497. http://doi.org/10.1515/mammalia-2014-0094

Khoury, C. K., Bjorkman, A. D., Dempewolf, H., Ramirez-Villegas, J., Guarino, L., Jarvis, A., Rieseberg, L. H., & Struik, P. C. (2014). Increasing homogeneity in global food supplies and the implications for food security. *PNAS*, *111*(11), 4001–4006. https://doi.org/10.1073/pnas.1313490111

Kier, G., Kreft, H., Lee, T. M., Jetz, W., Ibisch, P. L., Nowicki, C., Mutke, J., & Barthlott, W. (2009). A global assessment of endemism and species richness across island and mainland regions. *PNAS*, *106*(23), 9322–9327.

Klauke, N., Schaefer, H. M., Bauer, M., & Segelbacher, G. (2016). Limited dispersal and significant fine- scale genetic structure in a tropical montane parrot species. *PLoS ONE*, *11*(12), e0169165.

Kleiber, P., Clarke, S., Bigelow, K., Nakano, H., Mcallister, M., & Takeuchi, Y. (2009). *North Pacific Blue Shark Stock Assessment*. *U.S. Department of Commerce, NOAA Technical Memorandum NMFS-PIFSC-17*. Honolulu, USA. Retrieved from https://www.pifsc.noaa.gov/tech/NOAA_Tech_Memo_PIFSC_17.pdf

Kloppenburg, J. R. (1991). *First the Seed. The political economy of plant biotechnology 1492-2000*. New York, USA: Cambridge University Press.

Knapp, A. K., Blair, J. M., Briggs, J. M., Collins, S. L., Hartnett, D. C., Johnson, L. C., & Towne, E. G. (1999). The keystone role of bison in North American tallgrass prairie: bison increase habitat heterogeneity and alter a broad array of plant, community, and ecosystem processes. *Bioscience*, *49*(1), 39–50.

Knapp, S., Dinsmore, L., Fissore, C., Hobbie, S. E., Jakobsdottir, I., Kattge, J., King, J. Y., Klotz, S., McFadden, J. P., & Cavender-Bares, J. (2012). Phylogenetic and functional characteristics of household yard floras and their changes along an urbanization gradient. *Ecology*, *93*(sp8), S83–S98. https://doi.org/10.1890/11-0392.1

Knops, J. M. H., Tilman, D., Haddad, N. M., Naeem, S., Mitchell, C. E., Haarstad, J., Ritchie, M. E., Howe, K. M., Reich, P. B., Siemann, E., & Groth, J. (1999). Effects of plant species richness on invasion dynamics, disease outbreaks, insect abundances and diversity. *Ecology Letters*, *2*(5), 286–293.

Koh, I., Lonsdorf, E. V., Williams, N. M., Brittain, C., Isaacs, R., Gibbs, J., & Ricketts, T. H. (2016). Modeling the status, trends, and impacts of wild bee abundance in the United States. *PNAS*, *113*(1), 140–145. http://doi.org/10.1073/pnas.1517685113

Köhl, M., Lasco, R., Cifuentes, M., Jonsson, Ö., Korhonen, K. T., Mundhenk, P., de Jesus Navar, J., & Stinson, G. (2015). Changes in forest production, biomass and carbon: results from the 2015 UN FAO Global Forest Resource Assessment. *Forest Ecology and Management*, *352*, 21–34. https://doi.org/10.1016/j.foreco.2015.05.036

Köhler, G. (2008). *Reptiles of Central America* (2nd ed.). Offenbach, Germany: Herpeton-Verlag.

Köhler, G. (2011). *Amphibians of Central America*. Offenbach, Germany: Herpeton-Verlag.

Kohon, J. (2011). *La Infraestructura en el Desarrollo Integral de América Latina. Diagnóstico Estratégico y Propuestas para una Agenda Prioritaria. Transporte IDeAL 2011*. Bogotá, Colombia: Corporación Andina de Fomento (CAF).

Kort, J., Richardson, J., Soolanayakanahally, R., & Schroede, W. (2014). Innovations in temperate agroforestry: the 13th North American Agroforestry Conference. *Agroforestry Systems*, *88*(4), 563–567.

Kothari, A. (2009). Protected areas and people: the future of the past. *Parks*, *17*(2), 23–34.

Kowarik, I. (2008). On the role of alien species in urban flora and vegetation. In J. M. Marzluff, E. Shulenberger, W. Endlicher, M. Alberti, G. Bradley, C. Ryan, … U. Simon (Eds.), *Urban Ecology* (pp. 321–338). Boston, USA: Springer. http://doi.org/10.1007/978-0-387-73412-5_20

Krachler, M., Zheng, J., Koerner, R., Zdanowicz, C., Fisher, D., & Shotyk, W. (2005). Increasing atmospheric antimony contamination in the northern hemisphere: snow and ice evidence from Devon Island, Arctic Canada. *Journal of Environmental Management*, *7*(12), 1169–1176. http://doi.org/10.1039/b509373b

Kraft, N. J. B., Baldwin, B. G., & Ackerly, D. D. (2010). Range size, taxon age and hotspots of neoendemism in the California flora. *Diversity and Distributions*, *16*(3), 403–413. http://doi.org/10.1111/j.1472-4642.2010.00640.x

Kramer, T., Kessler, M., Gradstein, S. R., & Acebey, A. (2005). Diversity patterns of vascular epiphytes along an elevational gradient in the Andes. *Journal of Biogeography*, *32*(10), 1799–1809. http://doi.org/10.1111/j.1365-2699.2005.01318.x

Kremen, C., & Miles, A. (2012). Ecosystem services in biologically diversified versus conventional farming systems: benefits, externalities, and trade-offs. *Ecology and Society*, *17*(4), 40. http://doi.org/10.5751/ES-05035-170440

Kroll, O., Hershler, R., Albrecht, C., Terrazas, E. M., Apaza, R., Fuentealba, C., Wolff, C., & Wilke, T. (2012). The endemic gastropod fauna of Lake Titicaca: correlation between molecular evolution and hydrographic history. *Ecology and Evolution*, *2*(7), 1517–1530. https://doi.org/10.1002/ece3.280

Krömer, T., Kessler, M., & Herzog, S. K. (2006). Distribution and flowering ecology of bromeliads along two climatically contrasting elevational transects in the Bolivian Andes. *Biotropica*, *38*(2), 183–195. http://doi.org/10.1111/j.1744-7429.2006.00124.x

Kuhlmann, M., & Ribeiro, J. F. (2016). Fruits and frugivores of the Brazilian Cerrado: ecological and phylogenetic considerations. *Acta Botanica Brasilica*, *30*(3), 495–507. http://doi.org/10.1590/0102-33062016abb0192

Küper, W., Kreft, H., Nieder, J., Köster, N., & Barthlott, W. (2004). Large-scale diversity patterns of vascular epiphytes in Neotropical montane rain forests. *Journal of Biogeography*, *31*(9), 1477–1487. http://doi.org/10.1111/j.1365-2699.2004.01093.x

Küppers, G. C., González Garraza, G. C., Quiroga, M. V., Lombardo, R., Marinone, M. C., Vinocur, A., & Mataloni, G. (2016). Drivers of highly diverse planktonic ciliate assemblages in peat bog pools from Tierra del Fuego (Argentina). *Hydrobiologia*, *773*(1), 117–134. http://doi.org/10.1007/s10750-016-2686-x

Kurz, W. A., Shaw, C. H., Boisvenue, C., Stinson, G., Metsaranta, J., Leckie, D., Dyk, A., Smyth, C., & Neilson, E. T. (2013). Carbon in Canada's boreal forest—a synthesis. *Environmental Reviews*, *21*(4), 260–292.

Kwak, M., Kami, J. A., & Gepts, P. (2009). The putative Mesoamerican domestication center of *Phaseolus vulgaris* is located in the Lerma-Santiago basin of Mexico. *Crop Science*, *49*(2), 554–563. http://doi.org/10.2135/cropsci2008.07.0421

Kyne, P. M., Carlson, J. K., Ebert, D. A., Fordham, S. V., Bizzarro, J. J., Graham, R. T., Kulka, D. W., Tewes, E. E., Harrison, L. R., & Dulvy, N. K. (Eds.). (2012). *The Conservation Status of North American, Central American, and Caribbean Chondrichthyans*. Vancouver, Canada: IUCN Species Survival Commission Shark Specialist Group.

La Sorte, F. A., & McKinney, M. L. (2007). Compositional changes over space and time along an occurrence-abundance continuum: anthropogenic homogenization of the North American avifauna. *Journal of Biogeography*, *34*(12), 2159–2167. http://doi.org/10.1111/j.1365-2699.2007.01761.x

Lähteenoja, O., Ruokolainen, K., Schulman, L., & Oinonen, M. (2009). Amazonian peatlands: an ignored C sink and potential source. *Global Change Biology*, *15*(9), 2311–2320. http://doi.org/10.1111/j.1365-2486.2009.01920.x

Lamanna, C., Blonder, B., Violle, C., Kraft, N. J. B., Sandel, B., Imova, I., Donoghue, J. C., Svenning, J.-C., McGill, B. J., Boyle, B., Buzzard, V., Dolins, S., Jorgensen, P. M., Marcuse-

Kubitza, A., Morueta-Holme, N., Peet, R. K., Piel, W. H., Regetz, J., Schildhauer, M., Spencer, N., Thiers, B., Wiser, S. K., & Enquist, B. J. (2014). Functional trait space and the latitudinal diversity gradient. *PNAS*, *111*(38), 13745–13750. https://doi.org/10.1073/pnas.1317722111

Lamoreux, J. F., McKnight, M. W., & Hernandez, R. C. (2015). *Amphibian Alliance for Zero Extinction Sites in Chiapas and Oaxaca*. Gland, Switzerland: IUCN.

Lanfranco, D., & Dungey, H. S. (2001). Insect damage in *Eucalyptus*: a review of plantations in Chile. *Austral Ecology*, *26*(5), 477–481. http://doi.org/10.1046/j.1442-9993.2001.01131.x

Lannes, L. S., Bustamante, M. M. C., Edwards, P. J., & Olde Venterink, H. (2016). Native and alien herbaceous plants in the Brazilian Cerrado are (co-) limited by different nutrients. *Plant and Soil*, *400*(1–2), 231–243. http://doi.org/10.1007/s11104-015-2725-9

Lara, A., Little, C., Urrutia, R., McPhee, J., Álvarez-Garretón, C., Oyarzún, C., Soto, D., Donoso, P., Nahuelhual, L., Pino, M., & Arismendi, I. (2009). Assessment of ecosystem services as an opportunity for the conservation and management of native forests in Chile. *Forest Ecology and Management*, *258*(4), 415–424. https://doi.org/10.1016/j.foreco.2009.01.004

Larios, C., Casas, A., Vallejo, M., Moreno-Calles, A. I., & Blancas, J. (2013). Plant management and biodiversity conservation in Náhuatl homegardens of the Tehuacán Valley, Mexico. *Journal of Ethnobiology and Ethnomedicine*, *9*(74), 16. http://doi.org/10.1186/1746-4269-9-74

Larridon, I., Shaw, K., Cisternas, M. A., Paizanni Guillén, A., Sharrock, S., Oldfield, S., Goetghebeur, P., & Samain, M. S. (2014). Is there a future for the Cactaceae genera *Copiapoa*, *Eriosyce* and *Eulychnia*? A status report of a prickly situation. *Biodiversity and Conservation*, *23*(5), 1249–1287. https://doi.org/10.1007/s10531-014-0664-z

Larridon, I., Walter, H. E., Guerrero, P. C., Duarte, M., Cisternas, M. A., Hernández, C. P., Bauters, K., Asselman, P., Goetghebeur, P., & Samain, M. S. (2015). An integrative approach to understanding the evolution and diversity of *Copiapoa* (Cactaceae), a threatened endemic Chilean genus from the Atacama desert. *American Journal of Botany*, *102*(9), 1506–1520. https://doi.org/10.3732/ajb.1500168

Laurance, W. F. (2010). Habitat destruction: Death by a thousand cuts. In N. S. Sodhi & P. R. Ehrlich (Eds.), *Conservation Biology for All* (pp. 73–87). New York, USA: Oxford University Press.

Laurance, W. F., Camargo, J. L. C., Fearnside, P. M., Lovejoy, T. E., Williamson, G. B., Mesquita, R. C. G., Meyer, C. F. J., Bobrowiec, P. E. D., & Laurance, S. G. W. (2017). An Amazonian rainforest and its fragments as a laboratory of global change. *Biological Review*. https://doi.org/10.1111/brv.12343

Laurance, W. F., Goosem, M., & Laurance, S. G. W. (2009). Impacts of roads and linear clearings on tropical forests. *Trends in Ecology and Evolution*, *24*(12), 659–669. http://doi.org/10.1016/j.tree.2009.06.009

Lawrence, D., & Vandecar, K. (2015). Effects of tropical deforestation on climate and agriculture. *Nature Climate Change*, *5*(1), 27–36. http://doi.org/10.1038/nclimate2430

Leach, M. K., & Givnish, T. J. (1996). Ecological determinants of species loss in remnant prairies. *Science*, *273*(5281), 1555–1558. http://doi.org/10.1126/science.273.5281.1555

Leal-Flórez, J. (2008). *Impacts of non-native fishes on the fish community and the fishery of the Ciénaga Grande de Santa Marta estuary, northern Colombia*. Doctoral dissertation, Ph. D. Thesis, University of Bremen.

Leão, Z., R. Kikuchi, and M. Oliveira. (2010). Status of Eastern Brazilian coral reefs in time of climate changes. *Pan-American Journal of Aquatic Sciences,* 5:224-235.

Le Maitre, D. C., Gush, M. B., & Dzikiti, S. (2015). Impacts of invading alien plant species on water flows at stand and catchment scales. *AoB Plants*, *7*(1), plv043. http://doi.org/10.1093/aobpla/plv043

Le Saout, S., Hoffmann, M., Shi, Y., Hughes, A., Bernard, C., Brooks, T. M., Bertzky, B., Butchart, S. H. M., Stuart, S. N., Badman, T., & Rodrigues, A. S. L. (2013). Protected areas and effective biodiversity conservation. *Science*, *342*(6160), 803–805. https://doi.org/10.1126/science.1239268

Lefcheck, J. S., Byrnes, J. E. K., Isbell, F., Gamfeldt, L., Griffin, J. N., Eisenhauer, N., Hensel, M. J. S., Hector, A., Cardinale, B. J., & Duffy, J. E. (2015). Biodiversity enhances ecosystem multifunctionality across trophic levels and habitats. *Nature Communications*, *6*, 6936. https://doi.org/10.1038/ncomms7936

Leishman, M. R., Westoby, M., & Jurado, E. (1995). Correlates of seed size variation: a comparison among five temperate floras. *Journal of Ecology*, *83*(3), 517–529. http://doi.org/10.2307/2261604

León, O. A., & Vargas-Ríos, O. (2011). Estrategias para el control, manejo y restauración de áreas invadidas por retamo espinoso (Ulex europaeus) en la vereda El Hato, localidad de Usme, Bogotá D.C. In O. Vargas & S. Reyes (Eds.), *La Restauración Ecológica en la Practica: Memorias del I Congreso Colombiano de Restauración Ecológica* (pp. 474–490). Bogotá, D.C., Colombia: Universidad Nacional de Colombia.

Leslie, H. M. (2008). Global coastal change. *Quarterly Review of Biology*, *83*(1), 136–137. http://doi.org/10.4031/002533206787353278

Lester, E. D., Satomi, M., & Ponce, A. (2007). Microflora of extreme arid Atacama Desert soils. *Soil Biology and Biochemistry*, *39*(2), 704–708. http://doi.org/10.1016/j.soilbio.2006.09.020

Letourneau, D. K., Armbrecht, I., Rivera, B. S., Lerma, J., Carmona, E. J., Daza, M. C., Escobar, S., Galindo, V., GutiéRrez, C., LóPez, S. D., Mejía, J. L., Rangel, A. M. A., Rangel, J. H., Rivera, L., Saavedra, C. A., Torres, A. M., & Trujillo, A. R. (2011). Does plant diversity benefit agroecosystems? A synthetic review. *Ecological Applications*. https://doi.org/10.1890/09-2026.1

Leveau, L. M. (2013). Bird traits in urban–rural gradients: how many functional groups are there? *Journal of Ornithology*, *154*(3), 655–662. http://doi.org/10.1007/s10336-012-0928-x

Liang, J., Crowther, T. W., Picard, N., Wiser, S., Zhou, M., Alberti, G., Schulze, E.-D., McGuire, A. D., Bozzato, F., Pretzsch, H., De-Miguel, S., Paquette, A., Herault, B., Scherer-Lorenzen, M., Barrett, C. B., Glick, H. B., Hengeveld, G. M., Nabuurs, G.-J., Pfautsch, S., Viana, H., Vibrans, A. C., Ammer, C., Schall, P., Verbyla, D., Tchebakova, N., Fischer, M., Watson, J. V., Chen, H. Y. H., Lei, X., Schelhaas, M.-J., Lu, H., Gianelle, D., Parfenova, E. I., Salas, C., Lee, E., Lee, B., Kim, H. S., Bruelheide, H., Coomes, D. A., Piotto, D., Sunderland, T., Schmid, B., Gourlet-Fleury, S., Sonke, B., Tavani, R., Zhu, J., Brandl, S., Vayreda, J., Kitahara, F., Searle, E. B., Neldner, V. J., Ngugi, M. R., Baraloto, C., Frizzera, L., Ba azy, R., Oleksyn, J., Zawi a-Nied wiecki, T., Bouriaud, O., Bussotti, F., Finer, L., Jaroszewicz, B., Jucker, T., Valladares, F., Jagodzinski, A. M., Peri, P. L., Gonmadje, C., Marthy, W., OBrien, T., Martin, E. H., Marshall, A. R., Rovero, F., Bitariho, R., Niklaus, P. A., Alvarez-Loayza, P., Chamuya, N., Valencia, R., Mortier, F., Wortel, V., Engone-Obiang, N. L., Ferreira, L. V., Odeke, D. E., Vasquez, R. M., Lewis, S. L., & Reich, P. B. (2016). Positive biodiversity-productivity relationship predominant in global forests. *Science*, *354*(6309), aaf8957. https://doi.org/10.1126/science.aaf8957

Liljedahl, A. K., Boike, J., Daanen, R. P., Fedorov, A. N., Frost, G. V., Grosse, G., Hinzman, L. D., Iijma, Y., Jorgenson, J. C., Matveyeva, N., Necsoiu, M., Raynolds, M. K., Romanovsky, V. E., Schulla, J., Tape, K. D., Walker, D. A., Wilson, C. J., Yabuki, H., & Zona, D. (2016). Pan-Arctic ice-wedge degradation in warming permafrost and its influence on tundra hydrology. *Nature Geoscience*, *9*(4), 312–318. https://doi.org/10.1038/ngeo2674

Linares-Palomino, R., Oliveira-Filho, A. T., & Pennington, R. T. (2011). Neotropical seasonally dry forests: diversity, endemism and biogeography of woody plants. In R. Dirzo, H. Young, H. Mooney, & G. Ceballos (Eds.), *Seasonally Dry Tropical Forests: Ecology and Conservation* (pp. 3–21). Washington, DC, USA: Island Press.

Lind, E. M., Vincent, J. B., Weiblen, G. D., Cavender-Bares, J. M., & Borer, E. T. (2015). Trophic phylogenetics: evolutionary influences on body size, feeding, and species associations in grassland arthropods. *Ecology*, *96*(4), 998–1009.

Lira, P. K., Tambosi, L. R., Ewers, R. M., & Metzger, J. P. (2012). Land-use and land-cover change in Atlantic Forest landscapes. *Forest Ecology and Management*, *278*, 80–89. http://doi.org/10.1016/j.foreco.2012.05.008

Liu, Y., Pan, X., & Li, J. (2015). A 1961-2010 record of fertilizer use, pesticide application and cereal yields: a review. *Agronomy for Sustainable Development*, *35*(1), 83–93. http://doi.org/10.1007/s13593-014-0259-9

Lloyd, A. H., & Bunn, A. G. (2007). Responses of the circumpolar boreal forest to 20th century climate variability. *Environmental Research Letters*, *2*(4), 45013. Retrieved from http://stacks.iop.org/1748-9326/2/i=4/a=045013

Loehle, C. (2000). Strategy space and the disturbance spectrum: a life-history model for tree species coexistence. *The American Naturalist*, *156*(1), 14–33. http://doi.org/10.1086/303369

Loh, E. H., Zambrana-Torrelio, C., Olival, K. J., Bogich, T. L., Johnson, C. K., Mazet, J. A. K., Karesh, W., & Daszak, P. (2015). Targeting transmission pathways for emerging zoonotic disease surveillance and control. *Vector-Borne and Zoonotic Diseases*, *15*(7), 432–437. https://doi.org/10.1089/vbz.2013.1563

Loisel, J., & Yu, Z. (2013). Holocene peatland carbon dynamics in Patagonia. *Quaternary Science Reviews*, *69*, 125–141. http://doi.org/10.1016/j.quascirev.2013.02.023

Londoño, C., Cleef, A., & Madriñán, S. (2014). Angiosperm flora and biogeography of the páramo region of Colombia, Northern Andes. *Flora: Morphology, Distribution, Functional Ecology of Plants*, *209*(2), 81–87. http://doi.org/10.1016/j.flora.2013.11.006

López-Austin, A. (1990). *Cuerpo Humano e Ideología. Las Concepciones de los Antiguos Nahuas (2 vols)*. Mexico City, Mexico: Universidad Nacional Autónoma de México, Instituto de Investigaciones Antropológicas.

Loreau, M. (2004). Does functional redundancy exist? *Oikos*, *104*(3), 606–611. http://doi.org/10.1111/j.0030-1299.2004.12685.x

Loreau, M. (2010). *From Populations to Ecosystems: Theoretical Foundations for a New Ecological Synthesis*. New Jersey, USA: Princeton University Press.

Loreau, M., & Hector, A. (2001). Partitioning selection and complementarity in biodiversity experiments. *Nature*, *412*(6842), 72–76. http://doi.org/10.1038/35083573

Lucifora, L. O., García, V. B., & Worm, B. (2011). Global diversity hotspots and conservation priorities for sharks. *PLoS ONE*, *6*(5), e19356. http://doi.org/10.1371/journal.pone.0019356

Luebert, F., & Pliscoff, P. (2006). *Sinopsis Bioclimática y Vegetacional de Chile*. Santiago, Chile: Editorial Universitaria.

Lughadha, E. N., Govaerts, R., Belyaeva, I., Black, N., Lindon, H., Allkin, R., Magill, R. E., & Nicolson, N. (2016). Counting counts: revised estimates of numbers of accepted species of flowering plants, seed plants, vascular plants and land plants with a review of other recent estimates. *Phytotaxa*, *272*(1), 82–88. https://doi.org/10.11646/phytotaxa.272.1.5

Lugo, A. E., Helmer, E. H., & Valentín, E. S. (2012). Caribbean landscapes and their biodiversity. *Interciencia*, *37*(9), 705–710.

Luteyn, J. L. (1999). *Páramos: A Checklist of Plant Diversity, Geographical Distribution, and Botanical Literature*. New York, USA: The New York Botanical Garden Press.

Lutz, D. A., Powell, R. L., & Silman, M. R. (2013). Four decades of Andean timberline migration and implications for biodiversity loss with climate change. *PLoS ONE*, *8*(9), e74496. http://doi.org/10.1371/journal.pone.0074496

Lyra-Jorge, M. C., Ciocheti, G., & Pivello, V. R. (2008). Carnivore mammals in a fragmented landscape in northeast of Sao Paulo State, Brazil. *Biodiversity and Conservation*, *17*(7), 1573–1580.

MacGregor-Fors, I., & Ortega-Álvarez, R. (Eds.). (2013). Ecología Urbana: Experiencias en América Latina. Retrieved from www1.inecol.edu.mx/libro_ecologia_urbana

Macossay-Cortez, A., Sánchez, A. J., Florido, R., Huidobro, L., & Montalvo-Urgel, H. (2011). Historical and environmental distribution of ichthyofauna in the tropical wetland of Pantanos de Centla, southern Gulf of Mexico. *Acta Ichthyologica et Piscatoria*, *41*(3), 229–245. http://doi.org/10.3750/AIP2011.41.3.11

Maestre, F. T., Quero, J. L., Gotelli, N. J., Escudero, A., Ochoa, V., Delgado-Baquerizo, M., García-Gómez, M., Bowker, M. A., Soliveres, S., Escolar, C., García-Palacios, P., Berdugo, M., Valencia, E., Gozalo, B., Gallardo, A., Aguilera, L., Arredondo, T., Blones, J., Boeken, B., Bran, D., Conceição, A. A., Cabrera, O., Chaieb, M., Derak, M., Eldridge, D. J., Espinosa, C. I., Florentino, A., Gaitán, J., Gatica, M. G., Ghiloufi, W., Gómez-González, S., Gutiérrez, J. R., Hernández, R. M., Huang, X., Huber-Sannwald, E., Jankju, M., Miriti, M., Monerris, J., Mau, R. L., Morici, E., Naseri, K., Ospina, A., Polo, V., Prina, A., Pucheta, E., Ramírez-Collantes, D. A., Romão, R., Tighe, M., Torres-Díaz, C., Val, J., Veiga, J. P., Wang, D., & Zaady, E. (2012). Plant species richness and ecosystem multifunctionality in global drylands. *Science*, *335*(6065), 214–218. https://doi.org/10.1126/science.1215442

Maitner, B. S., Boyle, B., Casler, N., Condit, R., Donoghue, J., Durán, S. M., Guaderrama, D., Hinchliff, C. E., Jørgensen, P. M., Kraft, N. J. B., McGill, B., Merow, C., Morueta-Holme, N., Peet, R. K., Sandel, B., Schildhauer, M., Smith, S. A., Svenning, J.-C., Thiers, B., Violle, C., Wiser, S., & Enquist, B. J. (2017). The BIEB r package: A tool to access the Botanical Information and Ecology Network (BIEN) database. *Methods in Ecology and Evolution*, (March), 1–7. https://doi.org/10.1111/2041-210X.12861

Malanson, G. P., Cheney, A. B., & Kinney, M. (2015). Climatic and geographic relations of alpine tundra floras in western North America. *Alpine Botany*, *125*(1), 21–29. http://doi.org/10.1007/s00035-014-0144-9

Malanson, G. P., Zimmerman, D. L., & Fagre, D. B. (2015). Floristic similarity, diversity and endemism as indicators of refugia characteristics and needs in the West. *Biodiversity*, *16*(4), 237–246. http://doi.org/10.1080/14888386.2015.1117989

Manhães, M. A., & Loures-Ribeiro, A. (2005). Spatial distribution and diversity of bird community in an urban area of southeast Brazil. *Brazilian Archives of Biology and Technology*, *48*(2), 285–294. http://doi.org/10.1590/S1516-89132005000200016

Maragliano, R. E., Marti, L. J., Ibañez, L. M., & Montalti, D. (2009). Comunidades de aves urbanas de Lavallol, Buenos Aires, Argentina. *Acta Zoologica Lilloana*, *53*(1–2), 108–114.

Marateo, G., Grilli, P., Bouzas, N., Jensen, R., Ferretti, V., Juarez, M., & Soave, G. (2013). Uso de hábitat por aves en rellenos sanitarios del noreste de la provincia de Buenos Aires, Argentina. *Ecologia Austral*, (Diciembre 2013), 202–208.

Mares, M. (1992). Neotropical mammals and the myth of Amazonian biodiversity. *Science*, *255*(5047), 976–979. http://doi.org/10.1126/science.255.5047.976

Marini, M. Â., & García, F. I. (2005). Bird conservation in Brazil. *Conservation Biology*, *19*(3), 665–671. http://doi.org/10.1111/j.1523-1739.2005.00706.x

Marini, M. Â. (2001). Effects of forest fragmentation on birds of the cerrado region, Brazil. *Bird Conservation International*, *11*(1), 13–25. http://doi.org/10.1017/S0959270901001034

Marquard, E., Weigelt, A., Temperton, V. M., Roscher, C., Schumacher, J., Buchmann, N., Fischer, M., Weisser, W. W., & Schmid, B. (2009). Plant species richness and functional composition drive overyielding in a six-year grassland experiment. *Ecology*, *90*(12), 3290–3302. https://doi.org/10.1890/09-0069.1

Martín-Forés, I., Sánchez-Jardón, L., Acosta-Gallo, B., del Pozo, A., Castro, I., de Miguel, J. M., Ovalle, C., & Casado, M. A. (2015). From Spain to Chile: environmental filters and success of herbaceous species in Mediterranean-climate regions. *Biological Invasions*, *17*(5), 1425–1438. https://doi.org/10.1007/s10530-014-0805-z

Martin, L. M., Moloney, K. A., & Wilsey, B. J. (2005). An assessment of grassland restoration success using species diversity components. *Journal of Applied Ecology*, *42*(2), 327–336. http://doi.org/10.1111/j.1365-2664.2005.01019.x

Martin, S. (2008). Global diversity of crocodiles (Crocodilia, Reptilia) in freshwater. *Hydrobiologia*, *595*(1), 587–591. http://doi.org/10.1007/s10750-007-9030-4

Martyniuk, N. A., Morales, C. L., & Aizen, M. A. (2015). Invasive conifers reduce seed set of a native Andean cedar through heterospecific pollination competition. *Biological Invasions*, *17*(4), 1055–1067. http://doi.org/10.1007/s10530-014-0775-1

Mascitti, V., & Bonaventura, S. M. (2002). Patterns of abundance, distribution and habitat use of flamingos in the high Andes, South America. *Waterbirds*, *25*, 358–365. http://doi.org/10.1675/1524-4695(2002)025%5B0358:POADAH%5D2.0.CO;2

Masera, O. R., Bellon, M. R., & Segura, G. (1995). Forest management options for sequestering carbon in Mexico. *Biomass and Bioenergy*, *8*(5), 357–367. http://doi.org/10.1016/0961-9534(95)00028-3

Matheson, K., McKenzie, C. H., Gregory, R. S., Robichaud, D. A., Bradbury, I. R., Snelgrove, P. V. R., & Rose, G. A. (2016). Linking eelgrass decline and impacts on associated fish communities to European green crab *Carcinus maenas* invasion. *Marine Ecology Progress Series*, *548*, 31–45. http://doi.org/10.3354/meps11674

Maxwell, S. L., Fuller, R. A., Brooks, T. M., & Watson, J. E. M. (2016). Biodiversity: The ravages of guns, nets and bulldozers. *Nature*, *536*(7615), 143–145. http://doi.org/10.1038/536143a

McCarthy, A. J., Shaw, M.-A., & Goodman, S. J. (2007). Pathogen evolution and disease emergence in carnivores. *Proceedings of the Royal Society B: Biological Sciences*, *274*(1629), 3165–3174. http://doi.org/10.1098/rspb.2007.0884

McCauley, D. J., Pinsky, M. L., Palumbi, S. R., Estes, J. A., Joyce, F. H., & Warner, R. R. (2015). Marine defaunation: Animal loss in the global ocean. *Science*, *347*(6219), 1255641 (1-7). http://doi.org/10.1126/science.1255641

McCleery, R. (2010). Urban mammals. In J. Aitkenhead-Peterson & A. Volder (Eds.), *Urban Ecosystem Ecology, Agronomy Monographs 55* (pp. 87–102). Madison, USA: American Society of Agronomy, Crop Science Society of America, Soil Science Society of America.

McIntyre, N. E. (2000). Ecology of urban arthropods: A review and a call to action. *Annals of the Entomological Society of America*, *93*(4), 825–835. http://doi.org/10.1603/0013-8746(2000)093%5B0825:EOUAAR%5D2.0.CO;2

McIntyre, N. E., Rango, J., Fagan, W. F., & Faeth, S. H. (2001). Ground arthropod community structure in a heterogeneous urban environment. *Landscape and Urban Planning*, *52*(4), 257–274. http://doi.org/10.1016/S0169-2046(00)00122-5

McIntyre, P. B., Reidy Liermann, C. A., & Revenga, C. (2016). Linking freshwater fishery management to global food security and biodiversity conservation. *PNAS*, *113*(45), 12880–12885. http://doi.org/10.1073/pnas.1521540113

McIntyre, P. J., Thorne, J. H., Dolanc, C. R., Flint, A. L., Flint, L. E., Kelly, M., & Ackerly, D. D. (2015). Twentieth-century shifts in forest structure in California: Denser forests, smaller trees, and increased dominance of oaks. *PNAS*, *112*(5), 1458–1463. http://doi.org/10.1073/pnas.1410186112

McKane, R. B., Johnson, L. C., Shaver, G. R., Nadelhoffer, K. J., Rastetter, E. B., Fry, B., Giblin, A. E., Kielland, K., Kwiatkowski, B. L., Laundre, J. A., & Murray, G. (2002). Resource-based niches provide a basis for plant species diversity and dominance in arctic tundra. *Nature*, *415*(6867), 68–71. https://doi.org/10.1038/415068a

McKee, K. L., Cahoon, D. R., & Feller, I. C. (2007). Caribbean mangroves adjust to rising sea level through biotic controls on change in soil elevation. *Global Ecology and Biogeography*, *16*, 545–556. http://doi.org/10.1111/j.1466-8238.2007.00317.x

McKinney, M. L. (2002). Urbanization, biodiversity, and conservation. *BioScience*, *52*(10), 883–890. http://doi.org/10.1641/0006-3568(2002)052%5B0883:UBAC%5D2.0.CO;2

McKinney, M. L. (2006). Urbanization as a major cause of biotic homogenization. *Biological Conservation*, *127*, 247–260. http://doi.org/10.1016/j.biocon.2005.09.005

McKinney, M. L. (2008). Effects of urbanization on species richness: A review of plants and animals. *Urban Ecosystems*, *11*(2), 161–176. http://doi.org/10.1007/s11252-007-0045-4

McNaughton, S. J., Oesterheld, M., Frank, D. A., & Williams, K. J. (1989). Ecosystem-level patterns of primary productivity and herbivory in terrestrial habitats. *Nature*, *341*, 142–144. http://doi.org/10.1038/341142a0

McPhearson, T., Auch, R., & Alberti, M. (2013). Regional assessment of North America: Urbanization trends, biodiversity patterns, and ecosystem services. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, … C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 279–286). New York, USA: Springer.

MD Sea Grant. (n.d.). Bait Worm Study. Retrieved from http://www.mdsg.umd.edu/topics/aquatic-invasive-species/bait-worm-study

Medan, D., Torretta, J. P., Hodara, K., de la Fuente, E. B., & Montaldo, N. H. (2011). Effects of agriculture expansion and intensification on the vertebrate and invertebrate diversity in the Pampas of Argentina. *Biodiversity and Conservation*, *20*(13), 3077–3100. http://doi.org/10.1007/s10531-011-0118-9

Medeiros, R. B. de, Focht, T., Menegon, L. L., & Freitas, M. R. (2014). Seed longevity of *Eragrostis plana* Nees buried in natural grassland soil. *Revista Brasileira de Zootecnia*, *43*(11), 561–567. http://doi.org/10.1590/S1516-35982014001100001

Meltofte, H. (Ed.). (2013). *Arctic Biodiversity Assessment. Status and Trends in Arctic Biodiversity*. Akureyri, Iceland: Conservation of Arctic Flora and Fauna.

Mena-Torres, F., Fernández-San Juan, M., Campos, B., Sánchez-Avila, J., Faría, M., Pinnock-Branford, M. V., de la Cruz-Malavassi, E. M., Lacorte, S., Soares, A. M. V. M., & Barata., C. (2014). Pesticide residue analyses and biomarker responses of native Costa Rican fish of the Poeciliidae and Cichlidae families to assess environmental impacts of pesticides in Palo Verde National Park. *Journal of Environmental Biology*, *35*(1), 19–27.

Mena, J. L., Williams, M., Gazzolo, C., & Montero, F. (2007). Estado de conservación de *Melanomys zunigae* (Sanborn 1949) y de los mamíferos pequeños en las Lomas de Lima. *Revista Peruana de Biología*, *14*(2), 201–207.

Mendenhall, C. D., Karp, D. S., Meyer, C. F. J., Hadly, E. A., & Daily, G. C. (2014). Predicting biodiversity change and averting collapse in agricultural landscapes. *Nature*, *509*(7499), 213–217. http://doi.org/10.1038/nature13139

Mendoza, R., & Koleff, P. (2014). *Especies acuáticas invasoras en México*. Mexico City, Mexico: Comisión Nacional para el Conocimiento y Uso de la Biodiversidad, México.

Menominee Tribal Enterprises. (2012). Forest Management Plan (Revised 1973) 2012-2027. November 29, 2012. Retrieved from http://www.mtewood.com/Forestry/FMP/finalfmp06112012.pdf

Meserve, P. L., Kelt, D. A., Previtali, M. A., Milstead, W. B., & Gutiérrez, J. R. (2011). Global climate change and small mammal populations in north-central Chile. *Journal of Mammalogy*, *92*(6), 1223–1235. http://doi.org/10.1644/10-MAMM-S-267.1

Metzger, J. P. (2009). Conservation issues in the Brazilian Atlantic forest. *Biological Conservation*, *142*(6), 1138–1140. http://doi.org/10.1016/j.biocon.2008.10.012

Michaelson, G. J., Ping, C. L., & Kimble, J. M. (1996). Carbon storage and distribution in tundra soils of Arctic Alaska, U.S.A. *Arctic and Alpine Research*, *28*(4), 414–424. http://doi.org/10.2307/1551852

Millennium Ecosystem Assessment. (2005). *Ecosystems and Human Well-being*. Washington, D.C., USA: Island Press.

Milliman, J. D., Farnsworth, K. L., Jones, P. D., Xu, K. H., & Smith, L. C. (2008). Climatic and anthropogenic factors affecting river discharge to the global ocean, 1951-2000. *Global and Planetary Change*, *62*(3), 187–194. http://doi.org/10.1016/j.gloplacha.2008.03.001

Miloslavich, P., Díaz, J. M., Klein, E., Alvarado, J. J., Díaz, C., Gobin, J., Escobar-Briones, E., Cruz-Motta, J. J., Weil, E., Cortés, J., Bastidas, A. C., Robertson, R., Zapata, F., Martín, A., Castillo, J., Kazandjian, A., & Ortiz, M. (2010). Marine biodiversity in the Caribbean: regional estimates and distribution patterns. *PLoS ONE*, *5*(8), e11916. https://doi.org/10.1371/journal.pone.0011916

Miloslavich, P., Klein, E., Diaz, J. M., Hernandez, C. E., Bigatti, G., Campos, L., Artigas, F., Castillo, J., Penchaszadeh, P. E., Neill, P. E., Carranza, A., Retana, M. V., de Astarloa, J. M. D., Lewis, M., Yorio, P., Piriz, M. L., Rodriguez, D., Yoneshigue-Valentin, Y., Gamboa, L., & Martin, A. (2011). Marine biodiversity in the Atlantic and Pacific Coasts of South America: Knowledge and gaps. *PLoS ONE*, *6*(1), e14631. https://doi.org/10.1371/journal.pone.0014631

Ministerio del Ambiente. (2014). *Plan Nacional de Restauración Forestal 2014-2017*. Quito, Ecuador. Retrieved from http://sociobosque.ambiente.gob.ec/files/images/articulos/archivos/amrPlanRF.pdf

Ministerio del Medio Ambiente. (2017). No Title.

Ministério do Meio Ambiente (MMA). (2006). *Programa REVIZEE. Avaliação do Potencial Sustentável de Recursos Vivos na Zona Econômica Exclusiva*. Brasília, Brazil: Relatório Executivo.

Mites, M. (2008). Criteria used to set export quotas for Appendix I and II orchid species from Ecuador. NDF Workshop WG 4 – Geophytes and epiphytes. Case study 3 summary. Retrieved from https://cites.org/sites/default/files/ndf_material/WG4-CS3-S.pdf

Mittermeier, R. A., Gil, P. R., Hoffmann, M., Pilgrim, J., Brooks, T., Mittermeier, C. G., Lamoreux, J., & Fonseca, G. A. (2005). *Hotspots Revisited: Earth's Biologically Richest and Most Endangered Terrestrial Ecoregions*. Mexico City, Mexico: CEMEX Books on Nature Series.

Moffat, N. D., Lantz, T. C., Fraser, R. H., & Olthof, I. (2016). Recent vegetation change (1980–2013) in the tundra ccosystems of the Tuktoyaktuk Coastlands, NWT, Canada. *Arctic, Antarctic, and Alpine Research*, *48*(3), 581–597. http://doi.org/10.1657/AAAR0015-063

Moles, A. T., Ackerly, D. D., Webb, C. O., Tweddle, J. C., Dickie, J. B., & Westoby, M. (2005). A brief history of seed size. *Science*, *307*(5709), 576–580. http://doi.org/10.1126/science.1104863

Molnar, J. L., Gamboa, R. L., Revenga, C., & Spalding, M. D. (2008). Assessing the global threat of invasive species to marine biodiversity. *Frontiers in Ecology and the Environment*, *6*(9), 485–492. http://doi.org/10.1890/070064

Monigatti, M., Bussmann, R. W., & Weckerle, C. S. (2013). Medicinal plant use in two Andean communities located at different altitudes in the Bolivar Province, Peru. *Journal of Ethnopharmacology*, *145*(2), 450–464. http://doi.org/10.1016/j.jep.2012.10.066

Montalva, J., Castro, B., & Allendes, J. L. (2010). Las abejas (Hymenoptera: Apoidea) del Jardín Botánico Chagual. Estudio de caso de abejas nativas en zonas urbanas de Santiago de Chile. *Revista Del Jardín Botánico Chagual*, *No. 8*, 13–23.

Montalva, J., Dudley, L., Arroyo, M. K., Retamales, H., & Abrahamovich, A. H. (2011). Geographic distribution and associated flora of native and introduced bumble bees (*Bombus* spp.) in Chile. *Journal of Apicultural Research*, *50*(1), 11–21. http://doi.org/10.3896/IBRA.1.50.1.02

Montalva, J., & Ruz, L. (2010). Actualización a la lista sistemática de las abejas chilenas. *Revista Chilena de Entomología*, *8*, 13–23.

Montecino, V., Molina, X., Kumar, S., Castillo, M. L. C., & Bustamante, R. O. (2014). Niche dynamics and potential geographic distribution of *Didymosphenia geminata* (Lyngbye) M. Schmidt, an invasive freshwater diatom in Southern Chile. *Aquatic Invasions*, *9*(4), 507–519. http://doi.org/10.3391/ai.2014.9.4.09

Montenegro, G., Gómez, M., Díaz, F., & Ginocchio, R. (2003). Regeneration potencial of Chilean matorral after fire. In T. T. Veblen, W. L. Baker, G. Montenegro, & T. W. Swetnam (Eds.), *Fire and Climatic Change in Temperate Ecosystems of the Western Americas* (pp. 381–409). New York: Springer.

Montes-Leyva, L., Téllez-Valdés, O., Bojorquez, L., Dávila, P., & Lira, R. (2017). Potential areas for conservation of useful flora of the Tehuacán-Cuicatlán Valley, Mexico. *Genetic Resources and Crop Evolution*, *65*(1), 343–354. http://doi.org/10.1007/s10722-017-0538-9

Moore, A. A., & Palmer, M. A. (2005). Invertebrate biodiversity in agricultural and urban headwater streams: Implications for conservation and management. *Ecological Applications*, *15*(4), 1169–1177. http://doi.org/10.1890/04-1484

Morales, C. L., Arbetman, M. P., Cameron, S. A., & Aizen., M. A. (2013). Rapid ecological replacement of a native bumble bee by invasive species. *Frontiers in Ecology and the Environment*, *11*(10), 529–534. http://doi.org/10.1890/120321

Moreno-Calles, A. I., Casas, A., Rivero-Romero, A. D., Romero-Bautista, Y. A., Rangel-Landa, S., Fisher-Ortíz, R. A., Alvarado-Ramos, F., Vallejo-Ramos, M., & Santos-Fita, D. (2016). Ethnoagroforestry: integration of biocultural diversity for food sovereignty in Mexico. *Journal of Ethnobiology and Ethnomedicine*, *12*, 54. https://doi.org/10.1186/s13002-016-0127-6

Moreno-Calles, A. I., Casas, A., Toledo, V. M., & Vallejo-Ramos, M. (Eds.). (2016). *Etnoagroforestería en México*. México D.F., México: Universidad Nacional Autónoma de México.

Moret, P., Aráuz, M., Gobbi, M., & Barragán, Á. (2016). Climate warming effects in the tropical Andes: first evidence for upslope shifts of Carabidae (Coleoptera) in Ecuador. *Insect Conservation and Diversity*, *9*(4), 342–350. http://doi.org/10.1111/icad.12173

Moritz, C. (2002). Strategies to protect biological diversity and the evolutionary processes that sustain it. *Systematic Biology*, *51*(2), 238–254. http://doi.org/10.1080/10635150252899752

Moritz, C., Patton, J. L., Conroy, C. J., Parra, J. L., White, G. C., & Beissinger, S. R. (2008). Impact of a century of climate change on small-mammal communities in Yosemite National Park, USA. *Science*, *322*(5899), 261–264. http://doi.org/10.1126/science.1163428

Moro, M. F., Lughadha, E., Filer, D., Soares de Araújo, F., & Martins, F. (2014). A catalogue of the vascular plants of the Caatinga Phytogeographical Domain: a synthesis of floristic and phytosociological surveys. *Phytotaxa*, *160*(1), 1–118. http://doi.org/10.11646/phytotaxa.160.1.1

Morón, A., & Sawchik, J. (2003). Soil quality indicators in a long-term crop-pasture rotation experiment in Uruguay. In *17th World Congress of Soil Science Symposium n° 32* (p. 1327). Serie Técnica 134. INIA La Estanzuela.

Morón Zambrano, V. I., García Rangel, S., & Yerena, E. (2015). *Deforestación en Venezuela: Una Comparación de las Evaluaciones Existentes*. Caracas, Venezuela. Retrieved from https://www.researchgate.net/profile/Vilisa_Moron_Zambrano/publication/301889738_Deforestacion_en_Venezuela_Una_comparacion_de_las_evaluaciones_existentes/links/572b32f608ae057b0a094add/Deforestacion-en-Venezuela-Una-comparacion-de-las-evaluaciones-exist

Morueta-Holme, N., Engemann, K., Sandoval-Acuña, P., Jonas, J. D., Segnitz, R. M., & Svenning, J.-C. (2015). Strong upslope shifts in Chimborazo's vegetation over two centuries since Humboldt. *PNAS*, *112*(41), 12741–12745. http://doi.org/10.1073/pnas.1509938112

Müller, N., Ignatieva, M., Nilon, C. H., Werner, P., & Zipperer, W. C. (2013). Patterns and trends in urban biodiversity and landscape design. In *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 123–174). http://doi.org/10.1007/978-94-007-7088-1_10

Mumby, P. J., Dahlgren, C. P., Harborne, A. R., Kappel, C. V., Micheli, F., Brumbaugh, D. R., Holmes, K. E., Mendes, J. M., Broad, K., Sanchirico, J. N., Buch, K., Box, S., Stoffle, R. W., & Gill, A. B. (2006). Fishing, trophic cascades, and the process of grazing on coral reefs. *Science*, *311*(5757), 98–101. https://doi.org/10.1126/science.1121129

Muñoz-Concha, D., Farías, C., & Méndez, J. (2015). Notes on a new population of the endangered Chilean tree *Gomortega keule*. *New Zealand Journal of Botany*, *53*(4), 224–230. http://doi.org/10.1080/0028825X.2015.1064974

Muñoz-Pedreros, A., de los Rios-Escalante, P., & Möller, P. (2015). Zooplankton of the highland bogs of Putana, a desert wetland of the high puna, northern Chile. *Crustaceana*, *88*(10–11), 1235–1244. https://doi.org/10.1163/15685403-00003482

Muñoz, A. A., & Cavieres, L. A. (2008). The presence of a showy invasive plant disrupts pollinator service and reproductive output in native alpine species only at high densities. *Journal of Ecology*, *96*(3), 459–467. http://doi.org/10.1111/j.1365-2745.2008.01361.x

Muñoz, Y., Cadena, A., & Rangel-Ch., J. O. (2000). Mamíferos. In J. O. Rangel Ch. (Ed.), *Colombia Diversidad Biótica III* (pp. 599–611). Santafé de Bogotá, D.C., Colombia: Universidad Nacional de Colombia. Retrieved from http://www.uneditorial.net/pdf/TomoIII.pdf

Myers-Smith, I. H., Elmendorf, S. C., Beck, P. S., Wilmking, M., Hallinger, M., Blok, D., Tape, K. D., Rayback, S. A., Macias-Fauria, M., Speed, J. D., & Vellend, M. (2015). Climate sensitivity of shrub growth across the tundra biome. *Nature Climate Change*, *5*(9), 887–891. https://doi.org/10.1038/NCLIMATE2697

Myers-Smith, I. H., Forbes, B. C., Wilmking, M., Hallinger, M., Lantz, T., Blok, D., Tape, K. D., Macias-Fauria, M., Sass-Klaassen, U., Lévesque, E., Boudreau, S., Ropars, P., Hermanutz, L., Trant, A., Siegwart Collier, L., Weijers, S., Rozema, J., Rayback, S. A., Martin Schmidt, N., Schaepman-Strub, G., Wipf, S., Rixen, C., Ménard, C. B., Venn, S., Goetz, S., Andreu-Hayles, L., Elmendorf, S., Ravolainen, V., Welker, J., Grogan, P., Epstein, H. E., & Hik, D. S. (2011). Shrub expansion in tundra ecosystems: dynamics, impacts and research priorities. *Environmental Research Letters*, *6*(4), 15pp. https://doi.org/10.1088/1748-9326/6/4/045509

Myers, N., Mittermeier, R. A., Mittermeier, C. G., da Fonseca, G. A. B., & Kent, J. (2000). Biodiversity hotspots for conservation priorities. *Nature*, *403*(6772), 853–858. http://doi.org/10.1038/35002501

Nair, P. K. R. (1985). Classification of agroforestry systems. *Agroforestry Systems*, *3*(2), 97–128. http://doi.org/10.1007/BF00122638

Nair, P. K. R., Gordon, A. M., & Mosqueda-Losada, M. R. (2008). Agroforestry. In S. E. Jorgensen & B. Fath (Eds.), *Encyclopedia of Ecology. Vol. 1*. Amsterdam, The Netherlands: Elsevier.

Naito, A. T., & Cairns, D. M. (2015). Patterns of shrub expansion in Alaskan arctic river corridors suggest phase transition. *Ecology and Evolution*, *5*(1), 87–101. http://doi.org/10.1002/ece3.1341

Navarro-González, R., Rainey, F. A., Molina, P., Bagaley, D. R., Hollen, B. J., de la Rosa, J., Small, A. M., Quinn, R. C., Grunthaner, F. J., Cáceres, L., Gomez-Silva, B., & McKay, C. P. (2003). Mars-like soils in the Atacama Desert, Chile, and the dry limit of microbial life. *Science*, *302*(5647), 1018–1021. https://doi.org/10.1126/science.1089143

Naylor, R. L. (2006). Environmental safeguards for open-ocean aquaculture. *Issues in Science and Technology*, *22*(3), 53–58.

Neff, J., Reynolds, R., Belnap, J., & Lamothe, P. (2005). Multi-decadal impacts of grazing on soil physical and biogeochemical properties in southeast Utah. *Ecological Applications*, *15*, 87–95. http://doi.org/10.1890/04-0268

Nelson, A., & Chomitz, K. M. (2011). Effectiveness of strict vs. multiple use protected areas in reducing tropical forest fires: a global analysis using matching methods. *PLoS ONE*, *6*(8), e22722. http://doi.org/10.1371/journal.pone.0022722

Nelson, E., Helmus, M. R., Cavender-Bares, J., Polasky, S., Lasky, J. R., Zanne, A. E., Pearse, W. D., Kraft, N. J.

B., Miteva, D. A., Fagan, W. F., Miteva, D., & Fagan, W. (2016). Commercial plant production and consumption still follow the latitudinal gradient in species diversity despite economic globalization. *PLoS ONE*, *11*(10), e0163002. https://doi.org/https://doi.org/10.1371/journal.pone.0163002

Nelson, M. D., Flather, C. H., Riitters, K. H., Sieg, C., & Garner, J. D. (2016). National report on sustainable forests—2015: Conservation of biological diversity. In S. M. Stanton & G. A. Christensen (Eds.), *Pushing Boundaries: New Directions in Inventory Techniques and Applications: Forest Inventory and Analysis (FIA) Symposium 2015* (p. 375). Portland, USA: United States Department of Agriculture. http://doi.org/10.2737/PNW-GTR-931

Nemogá Soto, G. R. (2011). Conocimientos tradicionales, biodiversidad y derechos de propiedad intelectual: Necesidad de una perspectiva de análisis propia. In G. R. Nemogá (Ed.), *Naciones Indígenas en los Estados Contemporáneos* (pp. 231–252). Bogotá, Colombia: Universidad Nacional de Colombia.

Nepstad, D., McGrath, D., Stickler, C., Alencar, A., Azevedo, A., Swette, B., Bezerra, T., DiGiano, M., Shimada, J., Seroa da Motta, R., Armijo, E., Castello, L., Brando, P., Hansen, M. C., McGrath-Horn, M., Carvalho, O., & Hess, L. (2014). Slowing Amazon deforestation through public policy and interventions in beef and soy supply chains. *Science*, *344*(6188), 1118–1123.https://doi.org/10.1126/science.1248525

Nepstad, D., Schwartzman, S., Bamberger, B., Santilli, M., Ray, D., Schlesinger, P., Lefebvre, P., Alencar, A., Prinz, E., Fiske, G., & Rolla, A. (2006). Inhibition of Amazon deforestation and fire by parks and indigenous lands. *Conservation Biology*, *20*(1), 65–73. https://doi.org/10.1111/j.1523-1739.2006.00351.x

Neubauer, P., Jensen, O. P., Hutchings, J. A., & Baum, J. K. (2013). Resilience and recovery of overexploited marine populations. *Science*, *340*(6130), 347–349. http://doi.org/10.1126/science.1230441

Niemeyer, H. M. (1995). Biologically active compounds from Chilean medicinal plants. In J. T. Arnason, R. Matta, & J. T. Romeo (Eds.), *Recent Advances in Phytochemistry. Phytochemistry of Medicinal Plants, Vol. 29* (pp. 137–159). New York, USA: Springer. http://doi.org/10.1007/978-1-4899-1778-2_7

Nipperess, D., & Wilson, P. (2017). PDcalc: An implementation of the Phylogenetic Diversity (PD) calculus in R. R package version 0.3.0.9000. Retrieved from https://github.com/davidnipperess/PDcalc

Nobles, T., & Zhang, Y. (2011). Biodiversity loss in freshwater mussels: Importance, threats, and solutions. In O. Grillo & G. Venora (Eds.), *Biodiversity Loss in a Changing Planet* (pp. 137–162). Open Access: Intech. http://doi.org/10.5772/25102

Nogueira, C., Colli, G. R., & Martins, M. (2009). Local richness and distribution of the lizard fauna in natural habitat mosaics of the Brazilian Cerrado. *Austral Ecology*, *34*(1), 83–96. http://doi.org/10.1111/j.1442-9993.2008.01887.x

Norambuena, H., Escobar, S., & Rodriguez, F. (2000). The biocontrol of Gorse, *Ulex europaeus*, in Chile: a progress report. In N. R. Spencer (Ed.), *Proceedings of the International Symposium on Biological Control of Weeds* (pp. 955–961). Bozeman, Montana, USA: Montana State University. Retrieved from https://www.researchgate.net/profile/Fernando_Rodriguez12/publication/237442332_The_Biocontrol_of_Gorse_Ulex_europaeus_in_Chile_A_Progress_Report/links/575ab24008aec91374a614e5.pdf

Northern Great Plains Program, W. (2016). *Grasslands*. Retrieved from https://www.worldwildlife.org/habitats/grasslands

Noss, R., LaRoe, E., & Scott, J. (1995). *Endangered Ecosystems of the United States: A Preliminary Assessment of Loss and Degradation*. Washington D.C., USA: US Department of the Interior. National Biological Service.

Nowak, D., Hirabayashi, S., Bodine, A., & Hoehn, R. (2013). Modeled PM2.5 removal by trees in ten U.S. cities and associated health effects. *Environmental Pollution*, *178*, 395–402. http://doi.org/10.1016/j.envpol.2013.03.050

Núñez, L., Grosjean, M., & Cartajena, I. (2002). Human occupations and climate change in the Puna de Atacama, Chile. *Science*, *298*(5594), 821–824. http://doi.org/10.1126/science.1076449

Nuzzo, V. A. (1986). Extent and status of midwest oak savanna: Presettlement and 1985. *Natural Areas Journal*, *6*(2), 6–36.

O'Connor, M. I., Gonzalez, A., Byrnes, J. E. K., Cardinale, B. J., Duffy, J. E., Gamfeldt, L., Griffin, J. N., Hooper, D., Hungate, B. A., Paquette, A., Thompson, P. L., Dee, L. E., & Dolan, K. L. (2017). A general biodiversity–function relationship is mediated by trophic level. *Oikos*, *126*(1), 18–31. https://doi.org/10.1111/oik.03652

O'Donnell, J. A., Romanovsky, V. E., Harden, J. W., & McGuire, A. D. (2009). The effect of moisture content on the thermal conductivity of moss and organic soil horizons from black spruce ecosystems in interior Alaska. *Soil Science*, *174*(12), 646–651. http://doi.org/10.1097/SS.0b013e3181c4a7f8

OBIS. (n.d.). No Title. Retrieved from http://www.iobis.org/

Ocampo-Peñuela, N., & Pimm, S. L. (2015). Elevational ranges of montane birds and deforestation in the Western Andes of Colombia. *PLoS ONE*, *10*(12), e0143311. http://doi.org/10.1371/journal.pone.0143311

Ochoa-Tocachi, B. F., Buytaert, W., & De Bièvre, B. (2016). Regionalization of land-use impacts on streamflow using a network of paired catchments. *Water Resources Research*, *52*(9), 6710–6729. http://doi.org/10.1002/2016WR018596

OECD/ECLAC. (2016). *OECD Environmental Performance Reviews: Chile 2016* (OECD Environmental Performance Reviews). Paris, France: OECD Publishing. Retrieved from http://www.oecd-ilibrary.org/environment/oecd-environmental-performance-reviews-chile-2016_9789264252615-en

Oechel, W. C., Vourlitis, G. L., Hastings, S. J., Zulueta, R. C., Hinzman, L., & Kane, D. (2000). Acclimation of ecosystem CO2 exchange in the Alaskan Arctic in response to decadal climate warming.

Nature, *406*(6799), 978–981. http://doi.org/10.1038/35023137

Ojeda, R. A., Stadler, J., & Brandl, R. (2003). Diversity of mammals in the tropical-temperate Neotropics: hotspots on a regional scale. *Biodiversity and Conservation*, *12*(7), 1431–1444. http://doi.org/10.1023/A:1023625125032

Oliveira, J. A. (2003). Diversidade de mamíferos e o estabelecimento de áreas prioritárias para a conservação do bioma Caatinga. In J. M. C. da. Silva, M. Tabarelli, T. M. Fonseca, & L. V. Lins (Eds.), *Biodiversidade da Caatinga: Áreas e Ações Prioritárias para a Conservação* (pp. 263–282). Brasilia D.F., Brazil: Ministério do Meio Ambiente/Universidade Federal de Pernambuco.

Oliveira, M. D., Campos, M. C. S., Paolucci, E. M., Mansur, M. C. D., & Hamilton, S. K. (2015). Colonization and spread of *Limnoperna fortunei* in South America. In D. Boltovskoy (Ed.), *Limnoperna fortunei: the Ecology, Distribution and Control of a Swiftly Spreading Invasive Fouling Mussel* (pp. 333–355). Springer International Publishing. http://doi.org/10.1007/978-3-319-13494-9_19

Oliveira, P. T. S., Nearing, M. A., & Wendland, E. (2015). Orders of magnitude increase in soil erosion associated with land use change from native to cultivated vegetation in a Brazilian savannah environment. *Earth Surface Processes and Landforms*, *40*(11), 1524–1532. http://doi.org/10.1002/esp.3738

Omaeqnomenew Masenahekan. (2004). *Menominee Indian Tribe of Wisconsin. Facts and Figures Reference* (3rd ed.). Menominee Indian Tribe of Wisconsin Department of Administration Community Resource Planner – Brian Kowalkowski. Retrieved from https://www.menominee-nsn.gov/CulturePages/Documents/FactsFigureswithSupplement.pdf

Orensanz, J. M., Schwindt, E., Pastorino, G., Bortolus, A., Casas, G., Darrigran, G., Elías, R., López Gappa, J. J., Obenat, S., Pascual, M., Penchaszadeh, P., Piriz, M. L., Scarabino, F., Spivak, E. D., & Vallarino, E. A. (2002). No longer the pristine confines of the world ocean: a survey of exotic marine species in the southwestern Atlantic. *Biological Invasions*, *4*(1–2), 115–143. https://doi.org/10.1023/A:1020596916153

Ortega-Álvarez, R., & MacGregor-Fors, I. (2011). Spreading the word: The ecology of urban birds outside the United States, Canada, and western Europe. *The Auk: Ornithological Advances*, *128*(2), 415–418. http://doi.org/10.1525/auk.2011.10082

Ortega-Baes, P., & Godínez-Alvarez, H. (2006). Global diversity and conservation priorities in the Cactaceae. *Biodiversity and Conservation*, *15*(3), 817–827. http://doi.org/10.1007/s10531-004-1461-x

Orwig, D. A., Foster, D. R., & Mausel, D. L. (2002). Landscape patterns of hemlock decline in New England due to the introduced hemlock woolly adelgid. *Journal of Biogeography*, *29*(10–11), 1475–1487. http://doi.org/10.1046/j.1365-2699.2002.00765.x

Overbeck, G. E., Vélez-Martin, E., Scarano, F. R., Lewinsohn, T. M., Fonseca, C. R., Meyer, S. T., Muller, S. C., Ceotto, P., Dadalt, L., Durigan, G., Ganade, G., Gossner, M. M., Guadagnin, D. L., Lorenzen, K., Jacobi, C. M., Weisser, W. W., & Pillar, V. D. (2015). Conservation in Brazil needs to include non-forest ecosystems. *Diversity and Distributions*, *21*(12), 1455–1460. https://doi.org/10.1111/ddi.12380

Pacheco, P., Aguilar-Støen, M., Börner, J., Etter, A., Putzel, L., & Vera Diaz, M. del C. (2011). Landscape transformation in tropical Latin America: assessing trends and policy implications for REDD+. *Forests*, *2*(1), 1–29. http://doi.org/10.3390/f2010001

Pagad, S., Hayes, K., Katsanevakis, S., & Costello, M. J. (2017). World Register of Introduced Marine Species (WRIMS). Retrieved March 15, 2017, from http://www.marinespecies.org/introduced

Paglia, A. P., da Fonseca, G. A., Rylands, A. B., Herrmann, G., Aguiar, L. M., Chiarello, A. G., Leite, Y. L., Costa, L. P., Siciliano, S., Kierulff, M. C. M., Mendes, S. L., Tavares, V., Mittermeier, R. A., & Patton, J. L. (2012). *Lista Anotada dos Mamíferos do Brasil/Annotated Checklist of Brazilian Mammals. Occasional Papers in Conservation Biology N°6* (2nd ed.). Arlington, USA: Conservation International.

Painter, T. H., Deems, J. S., Belnap, J., Hamlet, A. F., Landry, C. C., & Udall, B. (2010). Response of Colorado River runoff to dust radiative forcing in snow. *PNAS*, *107*(40), 17125–17130. http://doi.org/10.1073/pnas.0913139107

Paquette, A., & Messier, C. (2011). The effect of biodiversity on tree productivity: from temperate to boreal forests. *Global Ecology and Biogeography*, *20*(1), 170–180. http://doi.org/10.1111/j.1466-8238.2010.00592.x

Parés-Ramos, I. K., Gould, W. A., & Aide, T. M. (2008). Agricultural abandonment, suburban growth, and forest expansion in Puerto Rico between 1991 and 2000. *Ecology and Society*, *13*(2).

Paritsis, J., & Aizen, M. A. (2008). Effects of exotic conifer plantations on the biodiversity of understory plants, epigeal beetles and birds in *Nothofagus dombeyi* forests. *Forest Ecology and Management*, *255*(5–6), 1575–1583. http://doi.org/10.1016/j.foreco.2007.11.015

Parra, F., & Casas, A. (2016). Origen y difusión de la domesticación y la agricultura en el Nuevo Mundo. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 159–188). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México /Universidad Nacional Agraria La Molina.

Paruelo, J. M., Garbulsky, M. F., Guerschman, J. P., & Jobbágy, E. G. (2004). Two decades of normalized difference vegetation index changes in South America: identifying the imprint of global change. *International Journal of Remote Sensing*, *25*(14), 2793–2806. http://doi.org/10.1080/01431160310001619526

Pauchard, A., Aguayo, M., Peña, E., & Urrutia, R. (2006). Multiple effects of urbanization on the biodiversity of developing countries: the case of a fast-growing metropolitan area (Concepción, Chile). *Biological Conservation*, *127*(3), 272–281. http://doi.org/10.1016/j.biocon.2005.05.015

Pauchard, A., & Barbosa, O. (2013). Regional assessment of Latin America: Rapid urban development and social economic inequity threaten biodiversity hotspots. In T. Elmqvist, M. Fragkias, J. Goodness, B. Güneralp, P. J. Marcotullio, R. I. McDonald, … C. Wilkinson (Eds.), *Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities: A Global Assessment* (pp. 589–608). New York, USA: Springer.

Pauchard, A., García, R. A., Peña, E., González, C., Cavieres, L. A., & Bustamante, R. O. (2008). Positive feedbacks between plant invasions and fire regimes: *Teline monspessulana* (L.) K. Koch (Fabaceae) in central Chile. *Biological Invasions*, *10*(4), 547–553. http://doi.org/10.1007/s10530-007-9151-8

Pauchard, A., Kueffer, C., Dietz, H., Daehler, C. C., Alexander, J., Edwards, P. J., Ramon Arevalo, J., Cavieres, L. A., Guisan, A., Haider, S., Jakobs, G., McDougall, K., Millar, C. I., Naylor, B. J., Parks, C. G., Rew, L. J., & Seipel, T. (2009). Ain't no mountain high enough: plant invasions reaching new elevations. *Frontiers in Ecology and the Environment*, *7*(9), 479–486. https://doi.org/10.1890/080072

Paul, M. J., & Meyer, J. L. (2001). Streams in the urban landscape. *Annual Review of Ecology and Systematics*, *32*(2001), 333–365. http://doi.org/10.1146/annurev.ecolsys.32.081501.114040

Paulich, N. (2010). Increasing private conservation through incentive mechanisms. *Journal of Animal Law and Policy*, *3*, 106–158.

Pauro, J., Gonzáles, M., Gamarra C, B., Pauro R, J., Mamani M, F., & Huerta, R. (2011). Plantas alimenticias, medicinales y biocidas de la comuna de Muñani y Suatia, provincia de Lampa (Puno-Perú). *Ecología Aplicada*, *10*(1), 41–49.

Pavez, E. F., Lobos, G. A., & Jaksic, F. M. (2010). Long-term changes in landscape and in small mammal and raptor assemblages in central Chile. *Revista Chilena de Historia Natural*, *83*(1), 99–111.

Pavlik, B. M., & Skinner, M. W. (1994). Ecological characteristics of California's rare plants. In *Inventory of Rare and Endangered Vascular Plants of California* (5th ed., pp. 4–6). Sacramento, California: California Native Plant Society.

Pearse, W. D., Cavender-Bares, J., Hobbie, S. E., Avolio, M. L., Bettez, N., Roy Chowdhury, R., Darling, L. E., Groffman, P. M., Grove, J. M., Hall, S. J., Heffernan, J. B., Learned, J., Neill, C., Nelson, K. C., Pataki, D. E., Ruddell, B. L., Steele, M. K., & Trammell, T. L. E. (2018). Homogenization of plant diversity, composition, and structure in North American urban yards. *Ecosphere*, *9*(2), e02105. https://doi.org/10.1002/ecs2.2105

Pedersen, A. B., Jones, K. E., Nunn, C. L., & Altizer, S. (2007). Infectious diseases and extinction risk in wild mammals. *Conservation Biology*, *21*(5), 1269–1279. http://doi.org/10.1111/j.1523-1739.2007.00776.x

Peixoto, R. B., Marotta, H., Bastviken, D., & Enrich-Prast, A. (2016). Floating aquatic macrophytes can substantially offset open water CO2 emissions from tropical floodplain lake ecosystems. *Ecosystems*, *19*(4), 724–736. http://doi.org/10.1007/s10021-016-9964-3

Pell, A. N. (1999). Animals and agroforestry in the tropics. In L. E. Buck, J. P. Lassoie, & E. C. M. Fernandes (Eds.), *Agroforestry in Sustainable Agricultural Systems* (pp. 33–46). Boca Raton, USA: CRC Press and Lewis Publishers.

Peña-Gómez, F. T., Guerrero, P. C., Bizama, G., Duarte, M., & Bustamante, R. O. (2014). Climatic niche conservatism and biogeographical non-equilibrium in *Eschscholzia californica* (Papaveraceae), an invasive plant in the Chilean Mediterranean region. *PLoS ONE*, *9*(8), e105025. http://doi.org/10.1371/journal.pone.0105025

Peña, E., Hidalgo, M., Langdon, B., & Pauchard., A. (2008). Patterns of spread of Pinus contorta Dougl. ex Loud. invasion in a Natural Reserve in southern South America. *Forest Ecology and Management*, *256*(5), 1049–1054. http://doi.org/10.1016/j.foreco.2008.06.020

Peralvo, M. F., Cuesta, F., & van Manen, F. (2005). Delineating priority habitat areas for the conservation of Andean bears in northern Ecuador. *Ursus*, *16*(2), 222–233. http://doi.org/10.2192/1537-6176(2005)016%5B0222:DPHAFT%5D2.0.CO;2

Pereira-Garbero, R., Barreneche, J. M., Laufer, G., Achaval, F., & Arim, M. (2013). Mamíferos invasores en Uruguay, historia, perspectivas y consecuencias. *Revista Chilena de Historia Natural*, *86*(4), 403–421. http://doi.org/10.4067/S0716-078X2013000400003

Perepelizin, P. V., & Faggi, A. M. (2009). Diversidad de aves en tres barrios de la ciudad de Buenos Aires, Argentina. *Multequina*, *18*(2), 71–85.

Peres, C. A., Emilio, T., Schietti, J., Desmoulière, S. J. M., & Levi, T. (2016). Dispersal limitation induces long-term biomass collapse in overhunted Amazonian forests. *PNAS*, *113*(4), 892–897.http://doi.org/10.1073/pnas.1516525113

Pérez-Cruz, Y. G., Rangel-Ruiz, L. J., & Gamboa-Aguilar, J. (2013). Metals in clams and sediments in the marshes of Centla Biosphere Reserve, Tabasco, Mexico. *Hidrobiologica*, *23*(1), 1–8.

Pérez-Ruzafa, A., Alvarado, J. J., Solís-Marín, F. A., Hernández, J. C., Morata, A., Marcos, C., Abreu-Pérez, M., Aguilera, O., Alió, J., Bacallado-Aránega, J. J., Barraza, E., Benavides-Serrato, M., Benítez-Villalobos, F., Betancourt-Fernández, L., Borges, M., Brandt, M., Brogger, M. I., Borrero-Pérez, G. H., Buitrón-Sánchez, B. E., Campos, L. S., Cantera, J. R., Clemente, S., Cohen-Renfijo, M., Coppard, S. E., Costa-Lotufo, L. V., Del Valle-García, R., Díaz De Vivar, M. E., Díaz-Martínez, J. P., Díaz, Y., Durán-González, A., Epherra, L., Escolar, M., Francisco, V., Freire, C. A., García-Arrarás, J. E., Gil, D. G., Guarderas, P., Hadel, V. F., Hearn, A., Hernández-Delgado, E. A., Herrera-Moreno, A., Herrero-Pérezrul, M. D., Hooker, Y., Honey-Escandón, M. B. I., Lodeiros, C., Luzuriaga, M., Manso, C. L. C., Martín, A., Martínez, M. I., Martínez, S., Moro-Abad, L., Mutschke, E., Navarro, J. C., Neira, R., Noriega, N., Palleiro-Nayar, J. S., Pérez, A. F., Prieto-Ríos, E., Reyes, J., Rodríguez-Barreras, R., Rubilar, T., Sancho-Mejías, T. I., Sangil, C., Silva, J. R. M. C., Sonnenholzner, J. I., Ventura, C. R. R., Tablado, A., Tavares, Y., Tiago, C. G., Tuya, F., & Williams, S. M. (2013). Latin America echinoderm biodiversity and biogeography: patterns and affinities. In J. J. Alvarado & F. A. Solís-Marín (Eds.), *Echinoderm Research and Diversity in Latin America* (pp. 511–542). Berlin, Germany: Springer.

Pérez, L. (2008). Aportes para la comprensión del imaginario Otomí. In C. H. Durand (Ed.), *El Derecho al Desarrollo Social: Una Visión Desde el Multiculturalismo: El Caso de los Pueblos Indígenas* (pp. 273–292). Ciudad de México, México: Editorial Porrúa.

Perfecto, I., & Vandermeer, J. (2008). Biodiversity conservation in tropical agroecosystems. *Annals of the New York Academy of Sciences*, *1134*(1), 173–200. http://doi.org/10.1196/annals.1439.011

Peri, P. L., Dube, F., & Varella, A. C. (2016). Silvopastoral systems in the subtropical and temperate zones of South America: an overview. In P. L. Peri, F. Dube, & A. Varella (Eds.), *Silvopastoral Systems in Southern South America* (pp. 1–8). Cham, Switzerland: Springer.

Peri, P. L., Ladd, B., Pepper, D. A., Bonser, S. P., Laffan, S. W., & Amelung, W. (2012). Carbon (δ 13C) and nitrogen (δ 15N) stable isotope composition in plant and soil in Southern Patagonia's native forests. *Global Change Biology*, *18*(1), 311–321. http://doi.org/10.1111/j.1365-2486.2011.02494.x

Periago, M. E., Chillo, V., & Ojeda, R. A. (2015). Loss of mammalian species from the South American Gran Chaco: empty savanna syndrome? *Mammal Review*, *45*(1), 41–53. http://doi.org/10.1111/mam.12031

Peroni, N., & Hanazaki., N. (2002). Current and lost diversity of cultivated varieties, especially cassava, under swidden cultivation systems in the Brazilian Atlantic Forest. *Agriculture Ecosystems & Environment*, *92*(2), 171–183. http://doi.org/10.1016/S0167-8809(01)00298-5

Perry, L., Dickau, R., Zarrillo, S., Holst, I., Pearsall, D. M., Piperno, D. R., Berman, M. J., Cooke, R. G., Rademaker, K., Ranere, A. J., Raymond, J. S., Sandweiss, D. H., Scaramelli, F., Tarble, K., & Zeidler, J. A. (2007). Starch fossils and the domestication and dispersal of chili peppers (*Capsicum* spp. L.) in the Americas. *Science*, *315*(5814), 986–988. https://doi.org/10.1126/science.1136914

Perry, W. L., Feder, J. L., & Lodge, D. M. (2001). Implications of hybridization between introduced and resident Orconectes crayfishes. *Conservation Biology*, *15*(6), 1656–1666. http://doi.org/10.1046/j.1523-1739.2001.00019.x

Petchey, O. L., & Gaston, K. J. (2002). Functional diversity (FD), species richness and community composition. *Ecology Letters*, *5*(3), 402–411. http://doi.org/10.1046/j.1461-0248.2002.00339.x

Petermann, J. S., Fergus, A. J. F., Turnbull, L. A., & Schmid, B. (2008). Janzen-Connell effects are widespread and strong enough to maintain diversity in grasslands. *Ecology*, *89*(9), 2399–2406. http://doi.org/10.1890/07-2056.1

Peterson, D. W., & Reich, P. B. (2001). Prescribed fire in oak savanna: fire frequency effects on stand structure and dynamics. *Ecological Applications*, *11*(3), 914–927. http://doi.org/10.1890/1051-0761(2001)011%5B0914:PFIOSF%5D2.0.CO;2

Peterson, D. W., & Reich, P. B. (2008). Fire frequency and tree canopy structure influence plant species diversity in a forest-grassland ecotone. *Plant Ecology*, *194*(1), 5–16. http://doi.org/10.1007/s11258-007-9270-4

Petitpas, R., Ibarra, J. T., Miranda, M., & Bonacic, C. (2016). Spatial patterns over a 24-year period show an increase in native vegetation cover and decreased fragmentation in Andean temperate landscapes, Chile. *Ciencia E Investigación Agraria*, *43*(3), 384–395. http://doi.org/10.4067/S0718-16202016000300005

Philpott, S. M., Arendt, W. J., Armbrecht, I., Bichier, P., Diestch, T. V., Gordon, C., Greenberg, R., Perfecto, I., Reynoso-Santos, R., Soto-Pinto, L., Tejeda-Cruz, C., Williams-Linera, G., Valenzuela, J., & Zolotoff, J. M. (2008). Biodiversity loss in Latin American coffee landscapes: Review of the evidence on ants, birds, and trees. *Conservation Biology*, *22*(5), 1093–1105. https://doi.org/10.1111/j.1523-1739.2008.01029.x

Pimentel, D. (2002). *Biological Invasions – Economic and Environmental Costs of Alien Plant, Animal, and Microbe Species*. (D. Pimentel, Ed.). Boca Raton, USA: CRC Press.

Pimentel, D., Zuniga, R., & Morrison, D. (2005). Update on the environmental and economic costs associated with alien-invasive species in the United States. *Ecological Economics*, *52*(3), 273–288. http://doi.org/10.1016/j.ecolecon.2004.10.002

Pimm, S. L., Jones, H. L., & Diamond, J. (1988). On the risk of extinction. *The American Naturalist*, *132*(6), 757–785. http://doi.org/10.1086/284889

Pincetl, S., Watt, T., & Santos, M. J. (2016). Land use regulation for resource conservation. In H. Mooney & E. Zavaleta (Eds.), *Ecosystem of California* (pp. 899–924). Oakland, USA: University of California Press.

Pincheira-Donoso, D., Harvey, L. P., & Ruta, M. (2015). What defines an adaptive radiation? Macroevolutionary diversification dynamics of an exceptionally species-rich continental lizard radiation. *BMC Evolutionary Biology*, *15*(1), 153. http://doi.org/10.1186/s12862-015-0435-9

Pinsky, M. L., Worm, B., Fogarty, M. J., Sarmiento, J. L., & Levin, S. A. (2013). Marine taxa track local climate velocities. *Science*, *341*(6151), 1239–1242. http://doi.org/10.1126/science.1239352

Piñeiro, G., Paruelo, J. M., & Oesterheld, M. (2006). Potential long-term impacts of livestock introduction on carbon and nitrogen cycling in grasslands of southern South America. *Global Change Biology*, *12*(7), 1267–1284. http://doi.org/10.1111/j.1365-2486.2006.01173.x

Piovia-Scott, J., Pope, K., Worth, S. J., Rosenblum, E. B., Poorten, T., Refsnider, J., … Foley, J. (2015). Correlates of virulence in a frog-killing fungal pathogen: evidence from a California amphibian decline. *The ISME Journal*, *9*(7), 1570–1578. http://doi.org/10.1038/ismej.2014.241

Pizano, C., Barón, A. F., Schuur, E. A. G., Crummer, K. G., & Mack, M. C. (2014). Effects of thermo-erosional disturbance on surface soil carbon and nitrogen dynamics in upland arctic tundra. *Environmental Research Letters*, *9*(7), 75006. http://doi.org/10.1088/1748-9326/9/7/075006

Pizano, C., & García, H. (2014). *El Bosque Seco Tropical de Colombia*. Bogota, D.C.: Instituto de Recursos Biológicos Alexander Von Humboldt.

Pizarro Tapia, R., M., Tapia Cornejo, M., Román Arellano, L., Jordán Díaz, C., & Farías Daza, C. (2006). Coeficientes de escorrentía instantáneos para la cuenca del río Tutuvén, VII Región del Maule, Chile. *Bosque*, *27*(2), 83–91.

Pliscoff, P., & Fuentes-Castillo, T. (2011). Representativeness of terrestrial ecosystems in Chile's protected area system. *Environmental Conservation*, *38*(3), 303–311. http://doi.org/10.1017/S0376892911000208

Poeplau, C., & Don, A. (2015). Carbon sequestration in agricultural soils via cultivation of cover crops – A meta-analysis. *Agriculture Ecosystems & Environment*, *200*, 33–41. http://doi.org/10.1016/j.agee.2014.10.024

Polidoro, B. A., Carpenter, K. E., Collins, L., Duke, N. C., Ellison, A. M., Ellison, J. C., Farnsworth, E. J., Fernando, E. S., Kathiresan, K., Koedam, N. E., Livingstone, S. R., Miyagi, T., Moore, G. E., Vien, N. N., Ong, J. E., Primavera, J. H., Salmo III, S. G., Sanciangco, J. C., Sukardjo, S., Wang, Y., & Yong, J. W. H. (2010). The loss of species: mangrove extinction risk and geographic areas of global concern. *PLoS ONE*, *5*(4), e10095. https://doi.org/10.1371/journal.pone.0010095

Poloczanska, E. S., Brown, C. J., Sydeman, W. J., Kiessling, W., Schoeman, D. S., Moore, P. J., Brander, K., Bruno, J. F., Buckley, L. B., Burrows, M. T., Duarte, C. M., Halpern, B. S., Holding, J., Kappel, C. V., O'Connor, M. I., Pandolfi, J. M., Parmesan, C., Schwing, F. B., Thompson, R. C., & Richardson, A. J. (2013). Global imprint of climate change on marine life. *Nature Climate Change*, *3*(10), 919–925. https://doi.org/10.1038/Nclimate1958

Portela, S. I., Andriulo, A. E., Jobbágy, E. G., & Sasal, M. C. (2009). Water and nitrate exchange between cultivated ecosystems and groundwater in the Rolling Pampas. *Agriculture, Ecosystems and Environment*, *134*(3–4), 277–286. http://doi.org/10.1016/j.agee.2009.08.001

Portela, S. I., Andriulo, A. E., Sasal, M. C., Mary, B., & Jobbágy, E. G. (2006). Fertilizer vs. organic matter contributions to nitrogen leaching in cropping systems of the Pampas: 15N application in field lysimeters. *Plant and Soil*, *289*(1–2), 265–277. http://doi.org/10.1007/s11104-006-9134-z

Portillo-Quintero, C. A., & Sánchez-Azofeifa, G. A. (2010). Extent and conservation of tropical dry forests in the Americas. *Biological Conservation*, *143*(1), 144–155. http://doi.org/10.1016/j.biocon.2009.09.020

Posey, D. A. (1999). *Cultural and Spiritual Values of Biodiversity*. London, UK: UNEP and Intermediate Technology Publications.

Pott, A., Oliveira, A. K. M., Damasceno-Junior, G. A., & Silva, J. S. V. (2011). Plant diversity of the Pantanal wetland. *Brazilian Journal of Biology*, *71*(1), 265–273. http://doi.org/10.1590/S1519-69842011000200005

Potts, S. G., Biesmeijer, J. C., Kremen, C., Neumann, P., Schweiger, O., & Kunin, W. E. (2010). Global pollinator declines: trends, impacts and drivers. *Trends in Ecology and Evolution*, *25*(6), 345–353. http://doi.org/10.1016/j.tree.2010.01.007

Price, S. J., Cecala, K. K., Browne, R. A., & Dorcas, M. E. (2011). Effects of urbanization on occupancy of stream salamanders. *Conservation Biology*, *25*(3), 547–555. http://doi.org/10.1111/j.1523-1739.2010.01627.x

Prival, D. B., & Schroff, M. J. (2012). A 13-year study of a northern population of twin-spotted rattlesnakes (*Crotalus pricei*): growth, reproduction, survival, and conservation. *Herpetological Monographs*, *26*(1), 1–18.

Programa Estado de la Nacion. (2008). *Estado de la Región en Desarrollo Humano Sostenible: Un Informe desde Centroamérica y para Centroamérica*. San José, Costa Rica: Estado de la Nación.

Pyron, M., Becker, J. C., Broadway, K. J., Etchison, L., Minder, M., DeColibus, D., Chezem, M., Wyatt, K. H., & Murry, B. A. (2017). Are long-term fish assemblage changes in a large US river related to the Asian Carp invasion? Test of the hostile take-over and opportunistic dispersal hypotheses. *Aquatic Sciences*, *79*(3), 631–642. https://doi.org/10.1007/s00027-017-0525-4

Pyron, R. A. (2014). Biogeographic analysis reveals ancient continental vicariance and recent oceanic dispersal in amphibians. *Systematic Biology*, *63*(5), 779–797. http://doi.org/10.1093/sysbio/syu042

Quijas, S., Schmid, B., & Balvanera, P. (2010). Plant diversity enhances provision of ecosystem services: a new synthesis. *Basic and Applied Ecology*, *11*(7), 582–593. http://doi.org/10.1016/j.baae.2010.06.009

Quinn, P. K., Shaw, G., Andrews, E., Dutton, E. G., Ruoho-Airola, T., & Gong, S. L. (2007). Arctic haze: current trends and knowledge gaps. *Tellus, Series B: Chemical and Physical Meteorology*, *59*(1), 99–114. http://doi.org/10.1111/j.1600-0889.2006.00238.x

Rabatel, A., Francou, B., Soruco, A., Gomez, J., Cáceres, B., Ceballos, J. L., Basantes, R., Vuille, M., Sicart, J.-E., Huggel, C., Scheel, M., Lejeune, Y., Arnaud, Y., Collet, M., Condom, T., Consoli, G., Favier, V., Jomelli, V., Galarraga, R., Ginot, P., Maisincho, L., Mendoza, J., Ménégoz, M., Ramirez, E., Ribstein, P., Suarez, W., Villacis, M., & Wagnon, P. (2013). Current state of glaciers in the tropical Andes: a multi-century perspective on glacier evolution and climate change. *The Cryosphere*, *7*(1), 81–102. https://doi.org/10.5194/tc-7-81-2013

Rademaker, K., Hodgins, G., Moore, K., Zarrillo, S., Miller, C., Bromley, G. R. M., Leach, P., Reid, D. A., Álvarez, W. Y., & Sandweiss, D. H. (2014). Paleoindian settlement of the high-altitude Peruvian Andes. *Science*, *346*(6208), 466–469. https://doi.org/10.1126/science.1258260

R Development Core Team. (2017). *R: A Language and Environment for Statistical Computing*. Vienna, Austria: R Foundation for Statistical Computing. Retrieved from https://cran.r-project.org/

RAISG. (2015). *Deforestación en la Amazonía (1970-2013)*. São Paulo: Instituto Socioambiental. Retrieved from www.raisg.socioambiental.org

Ramankutty, N., Evan, A. T., Monfreda, C., & Foley, J. A. (2010). Global Agricultural Lands: Pastures, 2000 Palisades, NY: NASA Socioeconomic Data and Applications Center (SEDAC). Retrieved fromhttp://

sedac.ciesin.columbia.edu/data/set/aglands-pastures-2000

Ramankutty, N., Foley, J. A., & Olejniczak, N. J. (2002). People on the land: changes in global population and croplands during the 20th century. *Ambio*, *31*(3), 251–257. http://doi.org/10.1579/0044-7447-31.3.251

Ramjohn, I. A., Murphy, P. G., Burton, T. M., & Lugo, A. E. (2012). Survival and rebound of Antillean dry forests: role of forest fragments. *Forest Ecology and Management*, *284*, 124–132. http://doi.org/10.1016/j.foreco.2012.08.001

Ramos, M. A., & Albuquerque, U. P. (2012). The domestic use of firewood in rural communities of the Caatinga: how seasonality interferes with patterns of firewood collection. *Biomass and Bioenergy*, *39*, 147–158. http://doi.org/10.1016/j.biombioe.2012.01.003

Ramos, R. S., Hilgert, N. I., & Lambaré, D. A. (2013). Traditional agriculture and the richness of maize (Zea mays). A case study in Caspalá, Jujuy province, Argentina. *Boletin de La Sociedad Argentina de Botánica*, *48*(3–4), 607–621.

Ratter, J., Ribeiro, J. F., & Bridgewater, S. (1997). The Brazilian cerrado vegetation and threats to its biodiversity. *Annals of Botany*, *80*(3), 223–230. http://doi.org/10.1006/anbo.1997.0469

Raupp, M. J., Shrewsbury, P. M., & Herms, D. A. (2010). Ecology of herbivorous arthropods in urban landscapes. *Annual Review of Entomology*, *55*(1), 19–38. http://doi.org/10.1146/annurev-ento-112408-085351

Rawlins, M. A., Steele, M., Holland, M. M., Adam, J. C., Cherry, J. E., Francis, J. A., Groisman, P. Y. A., Hinzman, L. D., Huntington, T. G., Kane, D. L., Kimball, J. S., Kwok, R., Lammers, R. B., Lee, C. M., Lettenmaier, D. P., Mcdonald, K. C., Podest, E., Pundsack, J. W., Rudels, B., Serreze, M. C., Shiklomanov, A., Skagseth, Ø., Troy, T. J., Vörösmarty, C. J., Wensnahan, M., Wood, E. F., Woodgate, R., Yang, D., Zhang, K., & Zhang, T. (2010). Analysis of the Arctic system for freshwater cycle intensification: observations and expectations. *Journal of Climate*, *23*(21), 5715–5737. https://doi.org/10.1175/2010JCLI3421.1

Ray, D. K., Welch, R. M., Lawton, R. O., & Nair, U. S. (2006). Dry season clouds and rainfall in northern Central America: implications for the Mesoamerican Biological Corridor. *Global and Planetary Change*, *54*(1–2), 150–162. http://doi.org/10.1016/j.gloplacha.2005.09.004

Ray, G. L. (2005). *Invasive Animal Species in Marine and Estuarine Environments: Biology and Ecology*. Vicksburg, USA: US Army Corps of Engineers.

Raymond, P. A., Hartmann, J., Lauerwald, R., Sobek, S., McDonald, C., Hoover, M., Butman, D., Striegl, R., Mayorga, E., Humborg, C., Kortelainen, P., Durr, H., Meybeck, M., Ciais, P., & Guth, P. (2013). Global carbon dioxide emissions from inland waters. *Nature*, *503*(7476), 355–359. https://doi.org/10.1038/nature12760

Raynolds, M. K., Walker, D. A., Ambrosius, K. J., Brown, J., Everett, K. R., Kanevskiy, M., Kofinas, G. P., Romanovsky, V. E., Shur, Y., & Webber, P. J. (2014). Cumulative geoecological effects of 62 years of infrastructure and climate change in ice-rich permafrost landscapes, Prudhoe Bay Oilfield, Alaska. *Global Change Biology*, *20*(4), 1211–1224. https://doi.org/10.1111/gcb.12500

Read, A. J. (2008). The looming crisis: interactions between marine mammals and fisheries. *Journal of Mammalogy*, *89*(3), 541–548. http://doi.org/10.1644/07-MAMM-S-315R1.1

Redfield, A. C. (1934). On the proportions of organic derivatives in sea water and their relation to the composition of plankton. In *James Johnstone Memorial Volume* (pp. 176–192). Liverpool, UK: University Press of Liverpool.

Redman, C. L., Grove, J. M., & Kuby, L. H. (2004). Integrating social science into the long-term ecological research (LTER) network: social dimensions of ecological change and ecological dimensions of social change. *Ecosystems*, *7*(2), 161–171. http://doi.org/10.1007/s10021-003-0215-z

Rehm, E. M., & Feeley, K. J. (2013). Forest patches and the upward migration of timberline in the southern Peruvian Andes. *Forest Ecology and Management*, *305*, 204–211. http://doi.org/10.1016/j.foreco.2013.05.041

Rehm, E. M., & Feeley, K. J. (2015). The inability of tropical cloud forest species to invade grasslands above treeline during climate change: Potential explanations and consequences. *Ecography*, *38*(12), 1167–1175. http://doi.org/10.1111/ecog.01050

Rehm, E. M., & Feeley, K. J. (2016). Seedling transplants reveal species-specific responses of high-elevation tropical treeline trees to climate change. *Oecologia*, *181*(4), 1233–1242. http://doi.org/10.1007/s00442-016-3619-0

Reich, P. B., Tilman, D., Isbell, F., Mueller, K., Hobbie, S. E., Flynn, D. F. B., & Eisenhauer, N. (2012). Impacts of biodiversity loss escalate through time as redundancy fades. *Science*, *336*(6081), 589–592. http://doi.org/10.1126/science.1217909

Reich, P. B., Tilman, D., Naeem, S., Ellsworth, D. S., Knops, J., Craine, J., Wedin, D., & Trost, J. (2004). Species and functional group diversity independently influence biomass accumulation and its response to CO2 and N. *PNAS*, *101*(27), 10101–10106. https://doi.org/10.1073/pnas.0306602101

Reichard, S. H., & White, P. (2001). Horticulture as a pathway of invasive plant introductions in the United States most invasive plants have been introduced for horticultural use by nurseries, botanical gardens, and individuals. *BioScience*, *51*(2), 103–113. http://doi.org/10.1641/0006-3568(2001)051%5B0103:HAAPOI%5D2.0.CO;2

Reis, R. E., Albert, J. S., Di Dario, F., Mincarone, M. M., Petry, P., & Rocha, L. A. (2016). Fish biodiversity and conservation in South America. *Journal of Fish Biology*, *89*(1), 12–47. http://doi.org/10.1111/jfb.13016

Rejmánek, M. (1996). Species richness and resistance to invasions. In G. Orians, R. Dirzo, & J. H. Cushman (Eds.), *Biodiversity and Ecosystem Processes in Tropical Forests. Ecological Studies (Analysis and Synthesis)* (Vol. 122, pp. 153–172). Berlin, Germany: Springer Verlag.

Rejmánek, M. (2017). Vascular plant extinctions in California: a critical assessment. *Diversity and Distributions*, *24*(1), 129–136. http://doi.org/10.1111/ddi.12665

Rennert, K. J., Roe, G., Putkonen, J., & Bitz, C. M. (2009). Soil thermal and ecological impacts of rain on snow events in the circumpolar Arctic. *Journal of Climate*, *22*(9), 2302–2315. http://doi.org/10.1175/2008JCLI2117.1

Resner, K., Yoo, K., Sebestyen, S. D., Aufdenkampe, A., Hale, C., Lyttle, A., & Blum, A. (2015). Invasive earthworms deplete key soil inorganic nutrients (Ca, Mg, K, and P) in a northern hardwood forest. *Ecosystems*, *18*(1), 89–102. http://doi.org/10.1007/s10021-014-9814-0

Revenga, C., & Kura, Y. (2003). *Status and Trends of Biodiversity of Inland Water Ecosystems*. *Technical Series No. 11*. Montreal, Canada: Secretariat of the Convention on Biological Diversity.

Reyes, S., & Figueroa, I. M. (2010). Distribución, superficie y accesibilidad de las áreas verdes en Santiago de Chile. *EURE Revista Latinoamericana de Estudios Urbanos Regionales*, *36*(109), 89–110. http://doi.org/10.4067/S0250-71612010000300004

Rezende, V. L. (2017). Tree species distribution and phylogenetic diversity across southern South America. *Frontiers of Biogeography*, *9*(2), 0–6. http://doi.org/10.21425/F59232082

Ribeiro, M. C., Martensen, A. C., Metzger, J. P., Tabarelli, M., Scarano, F., & Fortin, M. J. (2011). The Brazilian Atlantic Forest: a shrinking biodiversity hotspot. In F. Zachos & J. Habel (Eds.), *Biodiversity Hotspots: Distribution and Protection of Conservation Priority Areas* (pp. 405–434). Berlin, Germany: Springer.

Ricciardi, A. (2006). Patterns of invasion in the Laurentian Great Lakes in relation to changes in vector activity. *Diversity and Distributions*, *12*(4), 425–433. http://doi.org/10.1111/j.1366-9516.2006.00262.x

Richardson, D. M., Williams, P. A., & Hobbs, R. J. (1994). Pine invasions in the southern hemisphere: determinants of spread and invadability. *Journal of Biogeography*, *21*(5), 511–527.

Ricketts, T. H., Dinerstein, E., Boucher, T., Brooks, T. M., Butchart, S. H. M., Hoffmann, M., Lamoreux, J. F., Morrison, J., Parr, M., Pilgrim, J. D., Rodrigues, A. S. L., Sechrest, W., Wallace, G. E., Berlin, K., Bielby, J., Burgess, N. D., Church, D. R., Cox, N., Knox, D., Loucks, C., Luck, G. W., Master, L. L., Moore, R., Naidoo, R., Ridgely, R., Schatz, G. E., Shire, G., Strand, H., Wettengel, W., & Wikramanayake, E. (2005). Pinpointing and preventing imminent extinctions. *PNAS*, *102*(51), 18497–18501. https://doi.org/10.1073/pnas.0509060102

Ricketts, T. H., Regetz, J., Steffan-Dewenter, I., Cunningham, S. A., Kremen, C., Bogdanski, A., Gemmill-Herren, B., Greenleaf, S. S., Klein, A. M., Mayfield, M. M., Morandin, L. A., Ochieng', A., & Viana, B. F. (2008). Landscape effects on crop pollination services: are there general patterns? *Ecology Letters*, *11*(5), 499–515. https://doi.org/10.1111/j.1461-0248.2008.01157.x

Rimski-Korsakov, H., Zubillaga, M. S., Landriscini, M. R., & Lavado, R. S. (2016). Maize and cover crop sequence in the Pampas: effect of fertilization and water stress on the fate of nitrogen. *Journal of Soil and Water Conservation*, *71*(1), 12–20. http://doi.org/10.2489/jswc.71.1.12

Roberts, P., Hunt, C., Arroyo-Kalin, M., Evans, D., & Boivin, N. (2017). The deep human prehistory of global tropical forests and its relevance for modern conservation. *Nature Plants*, *3*(8), 17093. http://doi.org/10.1038/nplants.2017.93

Robinson, R. A., Lawson, B., Toms, M. P., Peck, K. M., Kirkwood, J. K., Chantrey, J., Clatworthy, I. R., Evans, A. D., Hughes, L. A., Hutchinson, O. C., John, S. K., Pennycott, T. W., Perkins, M. W., Rowley, P. S., Simpson, V. R., Tyler, K. M., & Cunningham, A. A. (2010). Emerging infectious disease leads to rapid population declines of common british birds. *PLoS ONE*, *5*(8), e12215. https://doi.org/10.1371/journal.pone.0012215

Robson, J., & Berkes, F. (2012). Sacred nature and community conserved areas. In S. Pilgrim & J. Pretty (Eds.), *Nature and Culture: Rebuilding Lost Connections* (pp. 197–216). London, UK and Washington DC., USA: Earthscan.

Rodewald, P. (Ed.). (2015). *The Birds of North America*. Ithaca, USA: Cornell Laboratory of Ornithology. Retrieved from https://birdsna.org

Rodrigue, J. P., Slack, B., & Blank, S. (2017). Gateways and transport corridors in North America. In J. P. Rodrigue, C. Comotois, & B. Slack (Eds.), *The Geography of Transport Systems* (4th ed.). New York, USA: Routledge Taylor & Francis Group.

Rodrigues, A. S. L., Brooks, T. M., Butchart, S. H. M., Chanson, J., Cox, N., Hoffmann, M., & Stuart, S. N. (2014). Spatially explicit trends in the global conservation status of vertebrates. *PLoS One*, *9*(11), e113934.http://doi.org/10.1371/journal.pone.0113934

Rodrigues da Silva, Ú. S., & Matos, D. M. D. S. (2006). The invasion of *Pteridium aquilinum* and the impoverishment of the seed bank in fire prone areas of Brazilian Atlantic Forest. *Biodiversity and Conservation*, *15*(9), 3035–3043. http://doi.org/10.1007/s10531-005-4877-z

Rodríguez-Catón, M., Villalba, R., Morales, M., & Srur, A. (2016). Influence of droughts on *Nothofagus pumilio* forest decline across northern Patagonia, Argentina. *Ecosphere*, *7*(7), e01390. http://doi.org/10.1002/ecs2.1390

Rodríguez-Estrella, R., Donázar, J. A., & Hiraldo, F. (1998). Raptors as indicators of environmental change in the scrub habitat of Baja California Sur, Mexico. *Conservation Biology*, *12*(4), 921–925.

Rohde, K., Heap, M., & Heap, D. (1993). Rapoport's rule does not apply to marine teleosts and cannot explain latitudinal gradients in species richness. *The American Naturalist*, *142*(1), 1–16. http://doi.org/10.1086/285526

Rojas, R. T. (1991). La agricultura en la época prehispánica. In T. Rojas (Ed.), *La Agricultura en Tierras Mexicanas desde sus Orígenes hasta Nuestros Días* (pp. 15–138). Ciudad de México, México: Grijalbo.

Román-Cuesta, R. M., Salinas, N., Asbjornsen, H., Oliveras, I., Huaman, V., Gutiérrez, Y., Puelles, L., Kala, J., Yabar, D., Rojas, M., Astete, R., Jordán, D. Y., Silman, M., Mosandl, R., Weber, M., Stimm, B., Günter, S., Knoke, T., & Malhi, Y. (2011). Implications of fires on carbon budgets in Andean cloud montane forest: the importance of peat soils and tree resprouting. *Forest Ecology and Management*, *261*(11), 1987–1997. https://doi.org/10.1016/j.foreco.2011.02.025

Rooth, J. E., & Stevenson, J. C. (2000). Sediment deposition patterns in *Phragmites australis* communities: implications for coastal areas threatened by rising sea-level. *Wetlands Ecology and Management*, *8*(2), 173–183. http://doi.org/10.1023/a:1008444502859

Roque, J., & León, B. (2006). Endemic Orchidaceae of Peru. *Revista Peruana de Biologia*, *13*(2), 759S–878S. http://doi.org/10.15381/rpb.v13i2.1953

Rosauer, D. F., & Jetz, W. (2015). Phylogenetic endemism in terrestrial mammals. *Global Ecology and Biogeography*, *24*(2), 168–179. http://doi.org/10.1111/geb.12237

Rosauer, D., Laffan, S. W., Crisp, M. D., Donnellan, S. C., & Cook, L. G. (2009). Phylogenetic endemism: a new approach for identifying geographical concentrations of evolutionary history. *Molecular Ecology*, *18*(19), 4061–4072. http://doi.org/10.1111/j.1365-294X.2009.04311.x

Roy, K., Jablonski, D., Valentine, J. W., & Rosenberg, G. (1998). Marine latitudinal diversity gradients: tests of causal hypotheses. *PNAS*, *95*(7), 3699–3702. http://doi.org/10.1073/pnas.95.7.3699

Rozzi, R., Armesto, J. J., Goffinet, B., Buck, W., Massardo, F., Silander, J., Arroyo, M. T. K., Russell, S., Anderson, C. B., Cavieres, L. A., & Callicott, J B. (2008). Changing lenses to assess biodiversity: patterns of species richness in sub-Antarctic plants and implications for global conservation. *Frontiers in Ecology and the Environment*, *6*(3), 131–137. https://doi.org/10.1890/070020

Ruiz, G. M., Fofonoff, P. W., Carlton, J. T., Wonham, M. J., & Hines, A H. (2000). Invasion of coastal marine communities in North America: apparent patterns, processes, and biases. *Annual Review of Ecology and Systematics*, *31*(1), 481–531. http://doi.org/10.1146/annurev.ecolsys.31.1.481

Ruiz, G. M., Fofonoff, P. W., Steves, B. P., & Carlton, J. T. (2015). Invasion history and vector dynamics in coastal marine ecosystems: a North American perspective. *Aquatic Ecosystem Health & Management*, *18*(3), 299–311. http://doi.org/10.1080/14634988.2015.1027534

Rundel, P. W., Arroyo, M. T. K., Cowling, R. M., Keeley, J. E., Lamont, B. B., & Vargas, P. (2016). Mediterranean biomes: evolution of their vegetation, floras and climate. *Annual Review of Ecology, Evolution, and Systematics*, *47*, 383–407. http://doi.org/10.1146/annurev-ecolsys-121415-032330

Rundel, P. W., Dillon, M. O., Palma, B., Mooney, H. A., Gulmon, S. L., & Ehleringer, J. R. (1991). The phytogeography and ecology of the coastal Atacama and Peruvian deserts. *Aliso: A Journal of Systematic and Evolutionary Botany*, *13*(1), 1–49. http://doi.org/10.5642/aliso.19911301.02

Rutherford, S., D'Hondt, S., & Prell, W. (1999). Environmental controls on the geographic distribution of zooplankton diversity. *Nature*, *400*(6746), 749–753. http://doi.org/10.1038/23449

Ryberg, W. A., Hill, M. T., Painter, C. W., & Fitzgerald, L. A. (2014). Linking irreplaceable landforms in a self-organizing landscape to sensitivity of population vital rates for an ecological specialist. *Conservation Biology*, *29*(3), 888–898.

Rylander, K., Perez, J., & Gomez, J. (1996). Status of the green mussel, *Perna viridis* (Linnaeus, 1758)(Mollusca: Mytilidae), in northeastern Venezuela. *Caribbean Marine Studies*, *5*, 86–87.

Rzedowski, J. (1991). Diversidad y orígenes de la flora fanerogámica de México. *Acta Botánica Mexicana*, *14*, 3–21.

Saavedra, B., & Simonetti, J. A. (2005). Small mammals of Maulino forest remnants, a vanishing ecosystem of south-central Chile. *Mammalia*, *69*(3–4), 337–348. http://doi.org/10.1515/mamm.2005.027

Sacco, A. G., Bergmann, F. B., & Rui, A. M. (2013). Assembleia de aves na área urbana do município de Pelotas, Rio Grande do Sul, Brasil. *Biota Neotropica*, *13*(2), 153–162. http://doi.org/10.1590/S1676-06032013000200014

Sadovy, Y., & Eklund, A. (1999). *Synopsis of Biological Information on the Nassau Grouper, Epinephelus striatus (Bloch, 1792), and the Jewfish, E. itajara (Lichtenstein, 1822). NOAA Technical Report NMFS 146. Technical Report of the Fishery Bulletin. FAO Fisheries Synopsis 157.* Seattle, USA: US Department of Commerce. Retrieved from https://www.nrc.gov/docs/ML1224/ML12240A298.pdf

Sahin, V., & Hall, M. J. (1996). The effects of afforestation and deforestation on water yields. *Journal of Hydrology*, *178*(1), 293–309. http://doi.org/10.1016/0022-1694(95)02825-0

Sala, E., & Knowlton, N. (2006). Global marine biodiversity trends. *Annual Review of Environment and Resources*, *31*(1), 93–122. http://doi.org/10.1146/annurev.energy.31.020105.100235

Salazar, A., Baldi, G., Hirota, M., Syktus, J., & McAlpine, C. (2015). Land use and land cover change impacts on the regional climate of non-Amazonian South America: A review. *Global and Planetary Change*, *128*, 103–119. http://doi.org/10.1016/j.gloplacha.2015.02.009

Salvador, F., Monerris, J., & Rochefort, L. (2014). Peatlands of the Peruvian Puna ecoregion: types, characteristics and disturbance. *Mires and Peat*, *15*(4), 1–17.

Samson, F. and F. Knopf. (1994). Prairie Conservation in North America. BioScience 44:418-421.

Sandoval, M. L., & Barquez, R. M. (2013). The Chacoan bat fauna identity: Patterns of distributional congruence and conservation implications. *Revista Chilena de Historia Natural*, *86*(1), 75–94. http://doi.org/10.4067/S0716-078X2013000100007

Sanguinetti, A., & Singer, R. B. (2014). Invasive bees promote high reproductive success in Andean orchids. *Biological Conservation*, *175*, 10–20. http://doi.org/10.1016/j.biocon.2014.04.011

Santiago-Alarcon, D., Arriaga-Weiss, S. L., & Escobar, O. (2011). Bird community composition of Centla Marshes Biosphere Reserve, Tabasco, Mexico. *Ornitologia Neotropical*, *22*, 229–246.

Santos, R. O., Lirman, D., & Pittman, S. J. (2016). Long-term spatial dynamics in vegetated seascapes: fragmentation and habitat loss in a human-impacted subtropical lagoon. *Marine Ecology*, *37*(1), 200–214. http://doi.org/10.1111/maec.12259

Sanzo, D., & Hecnar, S. J. (2006). Effects of road de-icing salt (NaCl) on larval wood frogs (*Rana sylvatica*). *Environmental Pollution*, *140*(2), 247–256. http://doi.org/10.1016/j.envpol.2005.07.013

Sarukhán, J., Urquiza-Haas, T., Koleff, P., Carabias, J., Dirzo, R., Ezcurra, E., Cerdeira-Estrada, S., & Soberón., J. (2015). Strategic actions to value, conserve, and restore the natural capital of megadiversity countries: the case of Mexico. *BioScience*, *65*(2), 164–173. https://doi.org/10.1093/biosci/biu195

Scherber, C., Eisenhauer, N., Weisser, W. W., Schmid, B., Voigt, W., Fischer, M., Schulze, E.-D., Roscher, C., Weigelt, A., Allan, E., Bessler, H., Bonkowski, M., Buchmann, N., Buscot, F., Clement, L. W., Ebeling, A., Engels, C., Halle, S., Kertscher, I., Klein, A.-M., Koller, R., König, S., Kowalski, E., Kummer, V., Kuu, A., Lange, M., Lauterbach, D., Middelhoff, C., Migunova, V. D., Milcu, A., Muller, R., Partsch, S., Petermann, J. S., Renker, C., Rottstock, T., Sabais, A., Scheu, S., Schumacher, J., Temperton, V. M., & Tscharntke, T. (2010). Bottom-up effects of plant diversity on multitrophic interactions in a biodiversity experiment. *Nature*, *468*, 553–556. https://doi.org/10.1038/nature09492

Scherer-Lorenzen, M. (2014). The functional role of biodiversity in the context of global change. In D. A. Coomes, F. R. Burslem, & W. D. Simonson (Eds.), *Forests and Global Change* (pp. 195–238). Cambridge, UK: Cambridge University Press.

Scherson, R. A., Thornhill, A. H., Urbina-Casanova, R., Freyman, W. A., Pliscoff, P. A., & Mishler, B. D. (2017). Spatial phylogenetics of the vascular flora of Chile. *Molecular Phylogenetics and Evolution*, *112*, 88–95. http://doi.org/10.1016/j.ympev.2017.04.021

Schmid-Hempel, R., Eckhardt, M., Goulson, D., Heinzmann, D., Lange, C., Plischuk, S., Escudero, L. R., Salathé, R., Scriven, J. J., & Schmid-Hempel, P. (2014). The invasion of southern South America by imported bumblebees and associated parasites. *Journal of Animal Ecology*, *83*(4), 823–837. https://doi.org/10.1111/1365-2656.12185

Schmidt, S. K., Nemergut, D. R., Miller, A. E., Freeman, K. R., King, A. J., & Seimon, A. (2009). Microbial activity and diversity during extreme freeze-thaw cycles in periglacial soils, 5400 m elevation, Cordillera Vilcanota, Perú. *Extremophiles: Microbial Life under Extreme Conditions*, *13*(5), 807–816. http://doi.org/10.1007/s00792-009-0268-9

Schulz, C., Koch, R., Cierjacks, A., & Kleinschmit, B. (2017). Land change and loss of landscape diversity at the Caatinga phytogeographical domain – Analysis of pattern-process relationships with MODIS land cover products (2001–2012). *Journal of Arid Environments*, *136*, 54–74. http://doi.org/10.1016/j.jaridenv.2016.10.004

Schulz, J. J., Cayuela, L., Echeverria, C., Salas, J., & Rey Benayas, J. M. (2010). Monitoring land cover change of the dryland forest landscape of Central Chile (1975-2008). *Applied Geography*, *30*(3), 436–447. http://doi.org/10.1016/j.apgeog.2009.12.003

Schulz, N., Aceituno, P., & Richter, M. (2011). Phytogeographic divisions, climate change and plant dieback along the coastal desert of northern Chile. *Erdkunde*, *65*(2), 169–187. http://doi.org/10.3112/erdkunde.2011.02.05

Schuur, E. A. G., McGuire, A. D., Schädel, C., Grosse, G., Harden, J. W., Hayes, D. J., Hugelius, G., Koven, C. D., Kuhry, P., Lawrence, D. M., Natali, S. M., Olefeldt, D., Romanovsky, V. E., Schaefer, K., Turetsky, M. R., Treat, C. C., & Vonk, J. E. (2015). Climate change and the permafrost carbon feedback. *Nature*, *520*(January 2016), 171–179. https://doi.org/10.1038/nature14338

Schuur, E. A. G., Vogel, J. G., Crummer, K. G., Lee, H., Sickman, J. O., & Osterkamp, T. E. (2009). The effect of permafrost thaw on old carbon release and net carbon exchange from tundra. *Nature*, *459*(7246), 556–559. http://doi.org/10.1038/nature08031

Schwartz, M. W., Thorne, J. H., & Viers, J. H. (2006). Biotic homogenization of the California flora in urban and urbanizing regions. *Biological Conservation*, *127*(3), 282–291. http://doi.org/10.1016/j.biocon.2005.05.017

Seebens, H., Blackburn, T. M., Dyer, E. E., Genovesi, P., Hulme, P. E., Jeschke, J. M., Pagad, S., Pyšek, P., Winter, M., Arianoutsou, M., Bacher, S., Blasius, B., Brundu, G., Capinha, C., Celesti-Grapow, L., Dawson, W., Dullinger, S., Fuentes, N., Jäger, H., Kartesz, J., Kenis, M., Kreft, H., Kühn, I., Lenzner, B., Liebhold, A., Mosena, A., Moser, D., Nishino, M., Pearman, D., Pergl, J., Rabitsch, W., Rojas-Sandoval, J., Roques, A., Rorke, S., Rossinelli, S., Roy, H. E., Scalera, R., Schindler, S., Štajerová, K., Tokarska-Guzik, B., van Kleunen, M., Walker, K., Weigelt, P., Yamanaka, T., & Essl, F. (2017). No saturation in the accumulation of alien species worldwide. *Nature Communications*, *8*, 14435. https://doi.org/10.1038/ncomms14435

Seebens, H., Schwartz, N., Schupp, P. J., & Blasius, B. (2016). Predicting the spread of marine species introduced by global shipping. *PNAS*, *113*(20), 5646–5651. http://doi.org/10.1073/pnas.1524427113

Serrano-Sandi, J., F. Bonilla-Murillo, & Sasa, M. (2013). Distribution, surface and protected area of palm-swamps in Costa Rica and Nicaragua. *Revista de Biologia Tropical*, *61*(Suppl. 1), 25–33.

Seto, K. C., Güneralp, B., & Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *PNAS*, *109*, 16083–16088. http://doi.org/10.1073/pnas.1211658109

Shaver, G. R., Laundre, J. A., Syndonia Bret-Harte, M., Stuart Chapin, F., Mercado-Díaz, J. A., Giblin, A. E., Gough, L., Gould, W. A., Hobbie, S. E., Kling, G. W., Mack, M. C., Moore, J. C., Nadelhoffer, K. J., Rastetter, E. B., & Schimel, J. P. (2014). Terrestrial ecosystems at Toolik Lake, Alaska. In J. E. Hobbie & G. W. Kling (Eds.), *Alaska's Changing Arctic: Ecological Consequences for Tundra, Streams, and Lakes*. New York, USA: Oxford University Press.

Shaver, I., Chain Guadarrama, A., Cleary, K., Sanfiorenzo, A. R., Santiago-Garcia, R., Finegan, B., Hormel, L., Sibelet, N., Vierling, L. A., Bosque-Perez, N., DeClerck, F., Fagan, M. E., & Waits, L. (2015). Coupled social and ecological outcomes of agricultural intensification in Costa Rica and the future of biodiversity conservation in tropical agricultural regions. *Global Environmental Change*, *32*, 74–86. https://doi.org/10.1016/j.gloenvcha.2015.02.006

Shochat, E., Lerman, S. B., Anderies, J. M., Warren, P. S., Faeth, S. H., & Nilon, C. H. (2010). Invasion, competition, and biodiversity loss in urban ecosystems. *BioScience*, *60*(3), 199–208. http://doi.org/10.1525/bio.2010.60.3.6

Sica, Y. V., Quintana, R. D., Radeloff, V. C., & G. I. Gavier-Pizarro. (2016). Wetland loss due to land use change in the Lower Paraná River Delta, Argentina. *Science of the Total Environment*, *568*, 967–978. http://doi.org/10.1016/j.scitotenv.2016.04.200

Siemann, E., Tilman, D., Haarstad, J., & Ritchie, M. (1998). Experimental tests of the dependence of arthropod diversity on plant diversity. *The American Naturalist*, *152*(2), 738–750. http://doi.org/10.1086/286204

Sierra, R. (2013). *Patrones y Factores de Deforestación en el Ecuador Continental, 1990-2010. Y un Acercamiento a los Próximos 10 Años*. Quito, Ecuador: Conservación Internacional Ecuador y Forest Trends.

Silva, J. M. C., Souza, M. A., Bieber, A. G. D., & Carlos, C. J. (2003). Aves da Caatinga: status, uso do habitat e sensitividade. In I. R. Leal, M. Tabarelli, & J. M. C. Silva (Eds.), *Ecologia e Conservação da Caatinga* (pp. 237–273). Racife, Brazil: Editora Universitaria, Universidade Federal de Pemambuco.

Silva, J. S. V., Abdon, M., Silva, S. M. A., & Moraes, J. A. (2011). Evolution of deforestation in the Brazilian pantanal and surroundings in the timeframe 1976-2008. *Geografia (Rio Claro)*, *36*(Especial), 35–55.

Simonetti, J. A. (1999). Diversity and conservation of terrestrial vertebrates in mediterranean Chile. *Revista Chilena de Historia Natural*, *72*(4), 493–500.

Simonetti, J. A., & Lazo, W. (1994). *Lepiota locaniensis*: an extinct Chilean fungus. *Revista Chilena De Historia Natural*, *67*(3), 351–352.

Singapore Index. (n.d.). Retrieved from http://www.cbd.int/authorities/gettinginvolved/cbi.shtml

Sklenář, P. (2016). Advance of plant species on slopes of the Chimborazo volcano (Ecuador) calculated based on unreliable data. *PNAS*, *113*(4), E407–E408. http://doi.org/10.1073/pnas.1522531113

Sklenář, P., Dušková, E., & Balslev, H. (2011). Tropical and temperate: evolutionary history of páramo flora. *The Botanical Review*, *77*(2), 71–108. http://doi.org/10.1007/s12229-010-9061-9

Sklenář, P., Hedberg, I., & Cleef, A. M. (2014). Island biogeography of tropical alpine floras. *Journal of Biogeography*, *41*(2), 287–297. http://doi.org/10.1111/jbi.12212

Sklenář, P., Luteyn, J., Ulloa, C., Jorgensen, P., & Dillon, M. (2005). *Flora Genérica de los Páramos: Guía Ilustrada de las Plantas Vasculares* (Vol. 92). New York, USA: The New York Botanical Garden Press.

Slippers, B., Burgess, T., Pavlic, D., Ahumada, R., Maleme, H., Mohali, S., Rodas, C., & Wingfield, M. J. (2009). A diverse assemblage of Botryosphaeriaceae infect *Eucalyptus* in native and non-native environments. *Southern Forests: A Journal of Forest Science*, *71*(2), 101–110. https://doi.org/10.2989/SF.2009.71.2.3.818

Smith, B. D. (1994). The origins of agriculture in the Americas. *Evolutionary Anthropology: Issues, News, and Reviews*, *3*(5), 174–184. http://doi.org/10.1002/evan.1360030507

Smith, D. D. (1998). Iowa Prairie: Original extent and loss, preservation and recovery attempts. *The Journal of the Iowa Academy of Science*, *105*(3), 94–108.

Smith, K. F., Acevedo-Whitehouse, K., & Pedersen, A. B. (2009). The role of infectious diseases in biological conservation. *Animal Conservation*, *12*(1), 1–12. http://doi.org/10.1111/j.1469-1795.2008.00228.x

Smith, K. F., Behrens, M. D., Max, L. M., & Daszak, P. (2008). U.S. drowning in unidentified fishes: Scope, implications, and regulation of live fish import. *Conservation Letters*, *1*(2), 103–109. http://doi.org/10.1111/j.1755-263X.2008.00014.x

Smith, L. M. (1993). *Estimated Presettlement and Current Acres of Natural Plant Communities in Louisiana*. Baton Rouge, LA: Louisiana Natural Heritage Program, Louisiana Department of Wildlife and Fisheries.

Snapp, S. S., Mafongoya, P. L., & Waddington, S. (1998). Organic matter technologies for integrated nutrient management in smallholder cropping systems of southern Africa. *Agriculture, Ecosystems and Environment*, *71*(1–3), 185–200. http://doi.org/10.1016/S0167-8809(98)00140-6

Solar, R. R. de C., Barlow, J., Ferreira, J., Berenguer, E., Lees, A. C., Thomson, J. R., Louzada, J., Maués, M., Moura, N. G., Oliveira, V. H. F., Chaul, J. C. M., Schoereder, J. H., Vieira, I. C. G., Mac Nally, R., & Gardner, T. A. (2015). How pervasive is biotic homogenization in human-modified tropical forest landscapes? *Ecology Letters*, *18*(10), 1108–1118. https://doi.org/10.1111/ele.12494

Somarriba, E., Beer, J., Alegre-Orihuela, J., Andrade, H., Cerda, R., DeClerck, F., Detlefsen, G., Escalante, M., Giraldo, L., Ibrahim, M., Krishnamurthy, L., Mena-Mosquera, V., Mora- Degado, J., Orozco, L., Scheelje, M., & Campos, J. (2012). Mainstreaming agroforestry in Latin America. In P. K. R. Nair & D. Garrity (Eds.), *Agroforestry the Future of Global Land Use, Advances in Agroforestry* (pp. 429–453). Dordrecht, Netherlands: Springer.

Sorte, C. J. B., Williams, S. L., & Carlton, J. T. (2010). Marine range shifts and species introductions: comparative spread rates and community impacts. *Global Ecology and Biogeography*, *19*(3), 303–316. http://doi.org/10.1111/j.1466-8238.2009.00519.x

Soto-Azat, C., Peñafiel-Ricaurte, A., Price, S. J., Sallaberry-Pincheira, N., García, M. P., Alvarado-Rybak, M., & Cunningham, A. A. (2016). *Xenopus laevis* and emerging amphibian pathogens in Chile. *EcoHealth*, *13*(4), 775–783. http://doi.org/10.1007/s10393-016-1186-9

Soto-Azat, C., Valenzuela-Sánchez, A., Collen, B., Rowcliffe, J. M., Veloso, A., & Cunningham, A. A. (2013). The population decline and extinction of Darwin's frogs. *PLoS ONE*, *8*(6), e66957. http://doi.org/10.1371/journal.pone.0066957

Southwestern Center for Herpetological Research. (n.d.). Snakes of the American Southwest. Retrieved from http://www.southwesternherp.com/snakes

Souza, V., Eguiarte, L. E., Siefert, J., & Elser, J. J. (2008). Microbial endemism: does phosphorus limitation enhance speciation? *Nature Reviews Microbiology*, *6*(7), 559–564. http://doi.org/10.1038/nrmicro1917

Souza, V., Espinosa-Asuar, L., Escalante, A. E., Eguiarte, L. E., Farmer, J., Forney, L., Lloret, L., Rodríguez-Martínez, J. M., Soberón, X., Dirzo, R., & Elser, J. J. (2006). An endangered oasis of aquatic microbial biodiversity in the Chihuahuan desert. *PNAS*, *103*(17), 6565–6570.https://doi.org/10.1073/pnas.0601434103

Souza, V., Siefert, J. L., Escalante, A. E., Elser, J. J., & Eguiarte, L. E. (2012). The Cuatro Ciénegas Basin in Coahuila, Mexico: an astrobiological Precambrian park. *Astrobiology*, *12*(7), 641–647. http://doi.org/10.1089/ast.2011.0675

Spalding, M. D., Fish, L., & Wood, L. J. (2008). Toward representative protection of the world's coasts and oceans-progress, gaps, and opportunities. *Conservation Letters*, *1*(5), 217–226. http://doi.org/10.1111/j.1755-263X.2008.00030.x

Spera, S. A., Galford, G. L., Coe, M. T., Macedo, M. N., & Mustard, J. F. (2016). Land-use change affects water recycling in Brazil's last agricultural frontier. *Global Change Biology*, *22*(10), 3405–3413. http://doi.org/10.1111/gcb.13298

Spracklen, D. V., & Righelato, R. (2014). Tropical montane forests are a larger than expected global carbon store. *Biogeosciences*, *11*(10), 2741–2754. http://doi.org/10.5194/bg-11-2741-2014

Spriggs, E. L., Christinb, P. A., & Edwards, E. J. (2014). C4 photosynthesis promoted species diversification during the miocene grassland expansion. *PLoS ONE*, *9*(5), e97722. http://doi.org/10.1371/journal.pone.0097722

Sproull, G. J., Quigley, M. F., Sher, A., & González, E. (2015). Long-term changes in composition, diversity and distribution patterns in four herbaceous plant communities along an elevational gradient. *Journal of Vegetation Science*, *26*(3), 552–563. http://doi.org/10.1111/jvs.12264

Squeo, F. A., Cavieres, L. A., Arancio, G., Novoa, J. E., Matthei, O., Marticorena, C., Rodriguez, R., Arroyo, M. T. K., & Munoz, M. (1998). Biodiversity of vascular flora in the Antofagasta Region, Chile. *Revista Chilena de Historia Natural*, *71*(4), 571–591.

Stapp, P., Antolin, M. F., & Ball, M. (2004). Patterns of extinction in prairie dog metapopulations: plague outbreaks follow El Niño events. *Frontiers in Ecology and the Environment*, *2*(5), 235–240. http://doi.org/10.1890/1540-9295(2004)002%5B0235:POEIPD%5D2.0.CO;2

Stattersfield, A. J., Crosby, M. J., Long, A. J., & Wege, D. C. (1998). *Endemic Bird Areas of the World: Priorities for Biodiversity Conservation*. Cambridge, UK: BirdLife International.

Steele, M. K., Heffernan, J. B., Bettez, N., Cavender-Bares, J., Groffman, P. M., Grove, J. M., Hall, S., Hobbie, S. E., Larson, K., Morse, J. L., Neill, C., Nelson, K. C., O'Neil-Dunne, J., Ogden, L., Pataki, D. E., Polsky, C., & Roy Chowdhury, R. (2014). Convergent surface water distributions in U.S. cities. *Ecosystems*, *17*(4), 685–697. https://doi.org/10.1007/s10021-014-9751-y

Stein, B. A., Kutner, L. S., & Adams, J. S. (Eds.). (2000). *Precious Heritage: The Status of Biodiversity in the United States*. New York, USA: Oxford University Press.

Stephenson, N. L., & Das, A. J. (2011). Comment on "Changes in climatic water balance drive downhill shifts in plant species' optimum elevations." *Science*, *334*(6053), 177c. http://doi.org/10.1126/science.1205740

Stevens, N., Lehmann, C. E. R., Murphy, B. P., & Durigan, G. (2017). Savanna woody encroachment is widespread across three continents. *Global Change Biology*, *23*(1), 235–244. http://doi.org/10.1111/gcb.13409

Stewart, J. A., Perrine, J. D., Nichols, L. B., Thorne, J. H., Millar, C. I., Goehring, K. E., & Wright, D. H. (2015). Revisiting the past to foretell the future: summer temperature and habitat area predict pika extirpations in California. *Journal of Biogeography*, *42*(5), 880–890. http://doi.org/10.1111/jbi.12466

Stöhr, S., O'Hara, T. D., & Thuy, B. (2012). Global diversity of brittle stars (Echinodermata: Ophiuroidea). *PLoS ONE*, *7*(3), e31940. http://doi.org/10.1371/journal.pone.0031940

Stotz, D. F., Fitzpatrick, J. W., Parker III, T., & Moskovits, D. K. (1996). *Neotropical Birds: Ecology and Conservation*. Chicago, USA: The University of Chicago Press.

Stranko, S. A., Gresens, S. E., Klauda, R. J., Kilian, J. V., Ciccotto, P. J., Ashton, M. J., & Becker, A. J. (2010). Differential effects of urbanization and non-natives on imperiled stream species. *Northeastern Naturalist*, *17*(4), 593–614. http://doi.org/10.1656/045.017.0406

Stuart, S. N., Hoffmann, M., Chanson, J. S., Cox, N. A., Berridge, R. J., Ramani, P., & Young, B. E. (Eds.). (2008). *Threatened Amphibians of the World*. Barcelona, Spain; Gland Switzerland; Arlington, USA: Lynx Edicions, IUCN and Conservation International.

Sturm, M., Racine, C., & Tape, K. (2001). Climate change: increasing shrub abundance in the Arctic. *Nature*, *411*(6837), 546–547. http://doi.org/10.1038/35079180

Suarez, M. L., Ghermandi, L., & Kitzberger, T. (2004). Factors predisposing episodic drought-induced tree mortality in *Nothofagus* - site, climatic sensitivity and growth trends. *Journal of Ecology*, *92*(6), 954–966. http://doi.org/10.1111/j.1365-2745.2004.00941.x

Sutherland, K. P., Porter, J. W., Turner, J. W., Thomas, B. J., Looney, E. E., Luna, T. P., Meyers, M. K., Futch, J. C., & Lipp, E. K. (2010). Human sewage identified as likely source of white pox disease of the threatened Caribbean elkhorn coral, *Acropora palmata*. *Environmental Microbiology*, *12*(5), 1122–1131. https://doi.org/10.1111/j.1462-2920.2010.02152.x

Sutherland, R. W., Dunning, P. R., & Baker, W. M. (2010). Amphibian encounter rates on roads with different amounts of traffic and urbanization. *Conservation Biology*, *24*(6), 1626–1635. http://doi.org/10.1111/j.1523-1739.2010.01570.x

Suzán, G., Marcé, E., Giermakowski, J. T., Mills, J. N., Ceballos, G., Ostfeld, R. S., Armién, B., Pascale, J. M., & Yates, T. L. (2009). Experimental evidence for reduced rodent diversity causing increased hantavirus prevalence. *PLoS ONE*, *4*(5), e5461. https://doi.org/10.1371/journal.pone.0005461

Svriz, M., Damascos, M. A., Zimmermann, H., & Hensen, I. (2013). The exotic shrub *Rosa rubiginosa* as a nurse plant. Implications for the restoration of disturbed temperate forests in Patagonia, Argentina. *Forest Ecology and Management*, *289*, 234–242. http://doi.org/10.1016/j.foreco.2012.09.037

Swenson, N. G., Enquist, B. J., Pither, J., Kerkhoff, A. J., Boyle, B., Weiser, M. D., Elser, J. J., Fagan, W. F., Forero-Montana, J., Fyllas, N., Kraft, N. J. B., Lake, J. K., Moles, A. T., Patino, S., Phillips, O. L., Price, C. A., Reich, P. B., Quesada, C. A., Stegen, J. C., Valencia, R., Wright, I. J., Wright, S. J., Andelman, S., Jorgensen, P. M., Lacher, T. E., Monteagudo, A., Nunez-Vargas, M. P., Vasquez-Martinez, R., & Nolting, K. M. (2012). The biogeography and filtering of woody plant functional diversity in North and South America. *Global Ecology and Biogeography*, *21*(8), 798–808. https://doi.org/10.1111/j.1466-8238.2011.00727.x

Syphard, A. D., Keeley, J. E., & Abatzoglou, J. T. (2017). Trends and drivers of fire activity vary across California aridland ecosystems. *Journal of Arid Environments*, *144*, 110–122. http://doi.org/10.1016/j.jaridenv.2017.03.017

Tabarelli, M., Aguiar, A. V., Ribeiro, M. C., Metzger, J. P., & Peres, C. A. (2010). Prospects for biodiversity conservation in the Atlantic Forest: lessons from aging human-modified landscapes. *Biological Conservation*, *143*(10), 2328–2340. http://doi.org/10.1016/j.biocon.2010.02.005

Tacón, A., Palma, J., Fernández, U., & Ortega, F. (2006). *El Mercado de los Productoss Forestales no Madereros y la Conservation de los Bosques del Sur de Chile y Argentina*. Valdivia, Chile: WWF Chile and Red de Productos Forestales No Madereros de Chile.

Tape, K., Sturm, M., & Racine, C. (2006). The evidence for shrub expansion in Northern Alaska and the Pan-Arctic. *Global Change Biology*, *12*(4), 686–702. http://doi.org/10.1111/j.1365-2486.2006.01128.x

Tavares, R., & Lopez, D. (2009). Fishery production trends of elasmobranchs from Venezuela: with emphasis on sharks. *Proceedings of the 62nd Gulf and Caribbean Fisheries Institute*, *62*, 178–183.

Taylor, B. L., Rojas-Bracho, L., Moore, J., Jaramillo-Legorreta, A., Ver Hoef, J. M., Cardenas-Hinojosa, G., Nieto-Garcia, E., Barlow, J., Gerrodette, T., Tregenza, N., Thomas, L., & Hammond, P. S. (2017). Extinction is imminent for Mexico's endemic porpoise unless fishery bycatch is eliminated. *Conservation Letters*, *10*(5), 588–595. https://doi.org/10.1111/conl.12331

Taylor, K. T., Maxwell, B. D., Pauchard, A., Nuñez, M. A., & Rew., L. J. (2016). Native versus non-native invasions: similarities and differences in the biodiversity impacts of *Pinus contorta* in introduced and native ranges. *Diversity and Distributions*, *22*(5), 578–588. http://doi.org/10.1111/ddi.12419

Tecco, P. A., Gurvich, D. E., Díaz, S., Pérez-Harguindeguy, N., & Cabido, M. (2006). Positive interaction between invasive plants: the influence of Pyracantha angustifolia on the recruitment of native and exotic woody species. *Austral Ecology*, *31*(3), 293–300. http://doi.org/10.1111/j.1442-9993.2006.01557.x

Tedersoo, L., Bahram, M., Põlme, S., Kõljalg, U., Yorou, N. S., Wijesundera, R., Ruiz, L. V., Vasco-Palacios, A. M., Thu, P. Q., Suija, A., Smith, M. E., Sharp, C., Saluveer, E., Saitta, A., Rosas, M., Riit, T., Ratkowsky, D., Pritsch, K., Poldmaa, K., Piepenbring, M., Phosri, C., Peterson, M., Parts, K., Partel, K., Otsing, E., Nouhra, E., Njouonkou, A. L., Nilsson, R. H., Morgado, L. N., Mayor, J., May, T. W., Majuakim, L., Lodge, D. J., Lee, S. S., Larsson, K.-H., Kohout, P., Hosaka, K., Hiiesalu, I., Henkel, T. W., Harend, H., Guo, L. D., Greslebin, A., Grelet, G., Geml, J., Gates, G., Dunstan, W., Dunk, C., Drenkhan, R., Dearnaley, J., De Kesel, A., Dang, T., Chen, X., Buegger, F., Brearley, F. Q., Bonito, G., Anslan, S., Abell, S., & Abarenkov, K. (2014). Global diversity and geography of soil fungi. *Science*, *346*(6213), 1256688. https://doi.org/10.1126/science.1256688

Teixeira, A. M. G., Soares-Filho, B. S., Freitas, S. R., & Metzger, J. P. (2009). Modeling landscape dynamics in an Atlantic Rainforest region: implications for conservation. *Forest Ecology and Management*, *257*(4), 1219–1230. http://doi.org/10.1016/j.foreco.2008.10.011

Tejedor-Garavito, N., Álvarez Dávila, E., Arango Caro, S., Araujo Murakami, A., Blundo, C., Boza Espinoza, T. E., La Torre Cuadros, M. A., Gaviria, J., Gutiérrez, N., Jørgensen, P. M., León, B., López Camacho, R., Malizia, L., Timaná de la Flor, M., Ulloa Ulloa, C., Vacas Cruz, O., & Newton, A. C. (2012). Evaluación del estado de conservación de los bosques montanos en los Andes tropicales. *Ecosistemas*, *21*(1–2), 148–166.

Tejedor Garavito, N., Álvarez Dávila, E., Arango Caro, S., Araujo Murakami, A., Baldeón, A., Beltrán, H., Blundo, C., Boza Espinoza, T. E., Fuentes Claros, A., Gaviria, J., Gutiérrez, N., Khela, S., León, B., La Torre Cuadros, M. A., López Camacho, R., Malizia, L., Millán, B., Moraes, R. M., Newton, A., Pacheco, S., Reynel, C., Ulloa Ulloa, C., & Vacas Cruz, O. (2014). *A Regional Red List of Montane Tree Species of the Tropical Andes: Trees at the top of the world*. Richmond, UK: Botanic Gardens Conservation International.

Tejedor Garavito, N., Newton, A. C., Golicher, D., & Oldfield, S. (2015). The relative impact of climate change on the extinction risk of tree species in the montane tropical Andes. *PLoS ONE*, *10*(7), e0131388. http://doi.org/10.1371/journal.pone.0131388

Tellería, J. L., Venero, J. L., & Santos, T. (2006). Conserving birdlife of Peruvian highland bogs: effects of patch-size and habitat quality on species richness and bird numbers. *Ardeola*, *53*(2), 271–283.

Temperton, V. M., Mwangi, P. N., Scherer-Lorenzen, M., Schmid, B., & Buchmann, N. (2007). Positive interactions between nitrogen-fixing legumes and four different neighbouring species in a biodiversity experiment. *Oecologia*, *151*(2), 190–205. http://doi.org/10.1007/s00442-006-0576-z

ter Steege, H., Pitman, N. C. A., Killeen, T. J., Laurance, W. F., Peres, C. A., Guevara, J. E., Salomão, R. P., Castilho, C. V., Amaral, I. L., de Almeida Matos, F. D., Valderrama Sandoval, E. H., & Valenzuela Gamarra, L. (2015). Estimating the global conservation status of more than 15,000 Amazonian tree species. *Science Advances*, *1*(10), e1500936. https://doi.org/10.1126/sciadv.1500936

ter Steege, H., Pitman, N. C. A., Sabatier, D., Baraloto, C., Salomão, R. P., Guevara, J. E., Phillips, O. L., Castilho, C. V., Magnusson, W. E., Molino, J. F., Monteagudo, A., Nunez Vargas, P., Montero, J. C., Feldpausch, T. R., Coronado, E. N. H., Killeen, T. J., Mostacedo, B., Vasquez, R., Assis, R. L., Terborgh, J., Wittmann, F., Andrade, A., Laurance, W. F., Laurance, S. G. W., Marimon, B. S., Marimon, B. H., Guimaraes Vieira, I. C., Amaral, I. L., Brienen, R., Castellanos, H., Cardenas Lopez, D., Duivenvoorden, J. F., Mogollon, H. F., Matos, F. D. D. A., Davila, N., Garcia-Villacorta, R., Stevenson Diaz, P. R., Costa, F., Emilio, T., Levis, C., Schietti, J., Souza, P., Alonso, A., Dallmeier, F., Montoya, A. J. D., Fernandez Piedade, M. T., Araujo-Murakami, A., Arroyo, L., Gribel, R., Fine, P. V. A., Peres, C. A., Toledo, M., Aymard C., G. A., Baker, T. R., Ceron, C., Engel, J., Henkel, T. W., Maas, P., Petronelli, P., Stropp, J., Zartman, C. E., Daly, D., Neill, D., Silveira, M., Paredes, M. R., Chave, J., Lima Filho, D. D. A., Jorgensen, P. M., Fuentes, A., Schongart, J., Cornejo Valverde, F., Di Fiore, A., Jimenez, E. M., Penuela Mora, M. C., Phillips, J. F., Rivas, G., van Andel, T. R., von Hildebrand, P., Hoffman, B., Zent, E. L., Malhi, Y., Prieto, A., Rudas, A., Ruschell, A. R., Silva, N., Vos, V. A., Zent, S., Oliveira, A. A., Schutz, A. C., Gonzales, T., Trindade Nascimento, M., Ramirez-Angulo, H., Sierra, R., Tirado, M., Umana Medina, M. N., van der Heijden, G., Vela, C. I. A., Vilanova Torre, E., Vriesendorp, C., Wang, O., Young, K. R., Baider, C., Balslev, H., Ferreira, C., Mesones, I., Torres-Lezama, A., Urrego Giraldo, L. E., Zagt, R., Alexiades, M. N., Hernandez, L., Huamantupa-Chuquimaco, I., Milliken, W., Palacios Cuenca, W., Pauletto, D., Valderrama Sandoval, E., Valenzuela Gamarra, L., Dexter, K. G., Feeley, K., Lopez-Gonzalez, G., & Silman, M. R. (2013). Hyperdominance in the Amazonian tree flora. *Science*, *342*(6156), 1243092. https://doi.org/10.1126/science.1243092

ter Steege, H., Vaessen, R. W., Cárdenas-López, D., Sabatier, D., Antonelli, A., de Oliveira, S. M., Pitman, N. C. A., Jørgensen, P. M., & Salomão, R. P. (2016). The discovery of the Amazonian tree flora with an updated checklist of all known tree taxa. *Scientific Reports*, *6*(1), 29549. https://doi.org/10.1038/srep29549

The Nature Conservancy. (2005). *Assessing Linkages between Agriculture and Biodiversity in Central America: Historical Overview and Future Perspectives*. San José, Costa Rica: Mesoamerican & Caribbean Region, Conservation Science Program.

The Nature Conservancy. (2009). TNC Terrestrial Ecoregions. Retrieved May 1, 2016, from http://maps.tnc.org/gis_data.html

Thébault, E., & Loreau, M. (2003). Food-web constraints on biodiversity-ecosystem functioning relationships. *PNAS*, *100*(25), 14949–14954. http://doi.org/10.1073/pnas.2434847100

Thomas, E., Vandebroek, I., Goetghebeur, P., Sanca, C., Arrázola, S., & Van Damme, P. (2008). The relationship between plant use and plant diversity in the Bolivian Andes, with special reference to medicinal plant use. *Human Ecology*, *36*(6), 861–879. http://doi.org/10.1007/s10745-008-9208-z

Thompson, D. Q., Stuckey, R. L., & Thompson, E. B. (1987). *Spread, Impact, and Control of Purple Loosestrife (Lythrum salicaria) in North American Wetlands*. Washington, D.C.: U.S. Fish and Wildlife Service.

Thorne, T., & Williams, E. S. (1988). Disease and endangered species: the black-footed ferret as a recent example. *Conservation Biology*, *2*(1), 66–74. http://doi.org/10.1111/j.1523-1739.1988.tb00336.x

Thornton, P. K. (2010). Livestock production: recent trends, future prospects. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, *365*(1554), 2853–2867. http://doi.org/10.1098/rstb.2010.0134

Tilman, D., Lehman, C. L., & Thomson, K. T. (1997). Plant diversity and ecosystem productivity: theoretical considerations. *PNAS*, *94*(5), 1857–1861.

Tilman, D., Reich, P. B., & Isbell, F. (2012). Biodiversity impacts ecosystem productivity as much as resources, disturbance, or herbivory. *PNAS*, *109*(26), 10394–10397. http://doi.org/10.1073/pnas.1208240109

Tingley, M. W., Monahan, W. B., Beissinger, S. R., & Moritz, C. (2009). Birds track their Grinnellian niche through a century of climate change. *PNAS*, *106*(Supplement 2), 19637–19643.http://doi.org/10.1073/pnas.0901562106

Tirado, C., Cortés, A., Carretero, M. A., & Bozinovic, F. (2016). Does the presence of livestock alter the trophic behaviour of sympatric populations of wild camelids *Vicugna vicugna* Molina 1782 and *Lama guanicoe* Müller 1976 (Artiodactyla: Camelidae)? Evidence from central Andes. *Gayana*, *80*(1), 29–39. http://doi.org/10.4067/S0717-65382016000100004

Tittensor, D. P., Mora, C., Jetz, W., Lotze, H. K., Ricard, D., Vanden Berghe, E., & Worm, B. (2010). Global patterns and predictors of marine biodiversity across taxa. *Nature*, *466*(7310), 1098–1101. http://doi.org/10.1038/nature09329

Tomasella, J., Neill, C., Figueiredo, R., & Nobre, A. D. (2009). Water and chemical budgets at the catchment scale including nutrient exports from intact forests and disturbed landscapes. In M. Keller, M. Bustamante, J. Gash, & P. S. Dias (Eds.), *Amazonia and Global Change* (pp. 505–524). Washington D.C., USA: American Geophysical Union.

Toral-Granda, M. V., Causton, C. E., Jäger, H., Trueman, M., Izurieta, J. C., Araujo, E., Cruz, M., Zander, K. K., Izurieta, A., & Garnett, S. T. (2017). Alien species pathways to the Galapagos Islands, Ecuador. *PLoS ONE*, *12*(9), e0184379. https://doi.org/10.1371/journal.pone.0184379

Tordoff, H., & Redig, P. T. (2001). Role of genetic background in the success of reintroduced peregrine falcons. *Conservation Biology*, *15*(2), 528–532. http://doi.org/10.1046/j.1523-1739.2001.015002528.x

Torres-Guevara, J., Parra-Rodinel, F., & Casas, A. (2017). Panorama de los recursos genéticos en Perú. In A. Casas, J. Torres-Guevara, & F. Parra-Rondinel (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 103–133). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Tovar, C., Seijmonsbergen, A. C., & Duivenvoorden, J. F. (2013). Monitoring land use and land cover change in mountain regions: An example in the Jalca grasslands of the Peruvian Andes. *Landscape and Urban Planning*, *112*, 40–49. http://doi.org/10.1016/J.LANDURBPLAN.2012.12.003

Trabucco, A., Zomer, R., Bossio, D., van Straaten, O., & Verchot, L. (2008). Climate change mitigation through afforestation/reforestation: a global analysis of hydrologic impacts with four case studies. *Agriculture, Ecosystems & Environment*, *126*(1–2), 81–97. http://doi.org/10.1016/j.agee.2008.01.015

Tracewski, Ł., Butchart, S. H. M., Di Marco, M., Ficetola, G. F., Rondinini, C., Symes, A., Wheatley, H., Beresford, A. E., & Buchanan, G. M. (2016). Toward quantification of the impact of 21st-century deforestation on the extinction risk of terrestrial vertebrates. *Conservation Biology*, *30*(5), 1070–1079. https://doi.org/10.1111/cobi.12715

Tree flora of the Neotropical Region. (n.d.). Retrieved from http://prof.icb.ufmg.br/treeatlan/treeatlanE_0_index.htm

Tremblay, B., Lévesque, E., & Boudreau, S. (2012). Recent expansion of erect shrubs in the Low Arctic: evidence from Eastern Nunavik. *Environmental Research Letters*, *7*(3), 35501. http://doi.org/10.1088/1748-9326/7/3/035501

Trexler, J. C., & Loftus, W. F. (2016). Invertebrates of the Florida Everglades. In D. Batzer & D. Boix (Eds.), *Invertebrates in Freshwater Wetlands* (pp. 321–356). Cham, Switzerlan: Springer.

Tribuzio, C. A., Rodgveller, C., Echave, K. B., & Hulson, P. J. (2015). Assessment of the shark stock complex in the Gulf of Alaska. In *Stock Assessment and Fishery Evaluation Report for the Groundfish Resources of the Gulf of Alaska for 2011* (pp. 1569–1642). Anchorage, USA.

Tuanmu, M. N., & Jetz, W. (2014). A global 1-km consensus land-cover product for biodiversity and ecosystem modellingxxx. *Global Ecology and Biogeography*, *23*(9), 1031–1045. http://doi.org/10.1111/geb.12182

Tubelis, D. P., & Cavalcanti, R. B. (2000). A comparison of bird communities in natural and disturbed non-wetland open habitats in the Cerrado's central region, Brazil. *Bird Conservation International*, *10*(4), 331–350.

Turnbull, L. A., Isbell, F., Purves, D. W., Loreau, M., & Hector, A. (2016). Understanding the value of plant diversity for ecosystem functioning through niche theory. *Proceedings of the Royal Society B: Biological Sciences*, *283*(1844), 20160536. https://doi.org/10.1098/rspb.2016.0536

Turner, R. E., Howes, B. L., Teal, J. M., Milan, C. S., Swenson, E. M., & Goehringer-Tonerb, D. D. (2009). Salt marshes and eutrophication: an unsustainable outcome. *Limnology and Oceanography*, *54*(5), 1634–1642. http://doi.org/10.4319/lo.2009.54.5.1634

U.S. Forestry Service. (2015). *National Report on Sustainable Forests*. United States Department of Agriculture.

Ulloa Ulloa, C., Acevedo-Rodríguez, P., Beck, S., Belgrano, M. J., Bernal, R., Berry, P. E., Brako, L., Celis, M., Davidse, G., Forzza, R. C., Gradstein, S. R., Hokche, O., León, B., León-Yánez, S., Magill, 1 R.E., Neill, D. A., Nee, M., Raven, P. H., Stimmel, S., Strong, M. T., Villaseñor, H. L., Zarucchi, J. L., Zuloaga, F. O., & Jørgensen, P. M. (2017). An integrated assessment of the vascular plants of the Americas. *Science*, *358*, 1614–1617.https://doi.org/10.1126/science.aao0398

UNEP-WCMC; IUCN. (2017). Protected Planet. Retrieved from https://www.protectedplanet.net/

UNESCO. (2002). Declaración Universal Sobre la Diversidad Cultural: Una Visión, una Plataforma Conceptual, un Semillero de Ideas, un Paradigma Nuevo. Johannesburgo, South Africa: Serie sobre la Diversidad Cultural. Documento preparado para la Cumbre Mundial sobre el Desarrollo Sostenible.

United Nations Development Programme. (2014). *Human Development Report 2014. Sustaining Human Progress: Reducing Vulnerabilities and Building Resilience*. New York, USA: United Nations Development Programme (UNDP).

Upham, N. S. (2017). Past and present of insular Caribbean mammals: understanding Holocene extinctions to inform modern biodiversity conservation. *Journal of Mammalogy*, *98*(4), 913–917. http://doi.org/10.1093/jmammal/gyx079

Urban, M. C., Skelly, D. K., Burchsted, D., Price, W., & Lowry, S. (2006). Stream communities across a rural-urban landscape gradient. *Diversity and Distributions*, *12*(4), 337–350. http://doi.org/10.1111/j.1366-9516.2005.00226.x

Urbina, J. C., & Benavides, J. C. (2015). Simulated small scale disturbances increase decomposition rates and facilitates invasive species encroachment in a high elevation tropical Andean peatland. *Biotropica*, *47*(2), 143–151. http://doi.org/10.1111/btp.12191

USA quickfacts census. (2012). Retrieved June 16, 2013, from https://web.archive.org/web/20120304192040/http://quickfacts.census.gov:80/qfd/states/00000.html

US Fish and Wildlife Service. (2011). Submerged Aquatic Vegetation: Where Have All the Grasses Gone? Retrieved from http://www.fws.gov/chesapeakebay/cbsav.html

US Forest Service. (2016). *New Aerial Survey Identifies More Than 100 Million Dead Trees in California*. Retrieved from https://www.fs.fed.us/news/releases/new-aerial-survey-identifies-more-100-million-dead-trees-california

USDA. (n.d.). Plant Database. Retrieved from http://plants.usda.gov

USDA. (2007). *Land Use Status and Trends 2007*. Retrieved from https://www.nrcs.usda.gov/wps/portal/nrcs/detail/national/home/?cid=stelprdb1083124

USDA. (2015). *Major Forest Insect and Disease Conditions in the United States: 2013*. Washington D.C., USA: US Forest service.

USDA, & USDOI. (2012). LANDFIRE: Vegetation Departure 1.3.0. Retrieved from http://www.landfire.gov

USGS. (2003). *PRAIRIEMAP, A GIS Database for Prairie Grassland Management in Western North America*. Washington D.C., USA: USGS Forest and Rangeland Ecosystem, Science Center, Snake River Field Station.

Valdivia, C. E., Simonetti, J. A., & Henríquez, C. A. (2006). Depressed pollination of *Lapageria rosea* Ruiz et Pav. (Philesiaceae) in the fragmented temperate rainforest of southern South America. *Biodiversity and Conservation*, *15*(5), 1845–1856. http://doi.org/10.1007/s10531-004-6683-4

Valiela, I. (2006). *Global Coastal Change*. Malden, USA: Blackwell Publishing.

Valiela, I., Bowen, J. L., & York, J. K. (2001). Mangrove forests: one of the world's threatened major tropical environments. *BioScience*, *51*(10), 807. http://doi.org/10.1641/0006-3568(2001)051%5B0807:MFOOTW%5D2.0.CO;2

Van Kleunen, M., Dawson, W., Essl, F., Pergl, J., Winter, M., Weber, E., Kreft, H., Weigelt, P., Kartesz, J., Nishino, M., Antonova, L. A., Barcelona, J. F., Cabezas, F. J., Cárdenas, D., Cárdenas-Toro, J., N, C., Chacón, E., Chatelain, C., Ebel, A. L., Figueiredo, E., Fuentes, N., Groom, Q. J., Henderson, L., Inderjit, L., Kupriyanov, A., S, M., Morozova, O., Moser, D., Nickrent, D. L., Patzelt, A., Pelser, P. B., Baptiste, M. P., Poopath, M., Schulze, M., Seebens, H., Shu, W.-S., Thomas, J., Velayos, M., Wieringa, J. J., & Pyšek, P. (2015). Global exchange and accumulation of non-native plants. *Nature*, *525*(7567), 100–103. https://doi.org/10.1038/nature14910

van Ruijven, J., & Berendse, F. (2005). Diversity-productivity relationships: initial effects, long-term patterns, and underlying mechanisms. *PNAS*, *102*(3), 695–700. http://doi.org/10.1073/pnas.0407524102

van Ruijven, J., & Berendse, F. (2009). Long-term persistence of a positive plant diversity-productivity relationship in the absence of legumes. *Oikos*, *118*(1), 101–106. http://doi.org/10.1111/j.1600-0706.2008.17119.x

van Tussenbroek, B. I., Cortés, J., Collin, R., Fonseca, A. C., Gayle, P. M. H., Guzmán, H. M., Jácome, G. E., Juman, R., Koltes, K. H., Oxenford, H. A., Rodríguez-Ramirez, A., Samper-Villarreal, J., Smith, S. R., Tschirky, J. J., & Weil, E. (2014). Caribbean-wide, long-term study of seagrass beds reveals local variations, shifts in community structure and occasional collapse. *PLoS ONE*, *9*(3), e90600. https://doi.org/10.1371/journal.pone.0090600

Van Vliet, N., Mertz, O., Heinimann, A., Langanke, T., Pascual, U., Schmook, B., Adams, C., Schmidt-Vogt, D., Messerli, P., Leisz, S., Castella, J.-C., Jørgensen, L., Birch-Thomsen, T., Hett, C., Bech-Bruun, T., Ickowitz, A., Vu, K. C., Yasuyuki, K., Fox, J., Padoch, C., Dressler, W., & Ziegler, A. D. (2012). Trends, drivers and impacts of changes in swidden cultivation in tropical forest-agriculture frontiers: a global assessment. *Global Environmental Change*, *22*(2), 418–429. https://doi.org/10.1016/j.gloenvcha.2011.10.009

Vanderbilt, K., & Gaiser, E. (2017). The International Long Term Ecological Research Network: a platform for collaboration. *Ecosphere*, *8*(2), e01697. http://doi.org/10.1002/ecs2.1697

Vandermeer, J. (1981). The interference production principle: an ecological theory for agriculture. *BioScience*, *31*(5), 361–364. http://doi.org/10.2307/1308400

Vásquez, D. L. A., Balslev, H., & Sklenář, P. (2015). Human impact on tropical-alpine plant diversity in the northern Andes. *Biodiversity and Conservation*, *24*(11), 2673–2683. http://doi.org/10.1007/s10531-015-0954-0

Vassallo, M. M., Dieguez, H. D., Garbulsky, M. F., Jobbágy, E. G., & Paruelo, J. M. (2013). Grassland afforestation impact on primary productivity: a remote sensing approach. *Applied Vegetation Science*, *16*(3), 390–403. http://doi.org/10.1111/avsc.12016

Vázquez, D. P. (2002). Multiple effects of introduced mammalian herbivores in a temperate forest. *Biological Invasions*, *4*(1–2), 175–191. http://doi.org/10.1023/A:1020522923905

Vega Thurber, R. L., Burkepile, D. E., Fuchs, C., Shantz, A. A., Mcminds, R., & Zaneveld, J. R. (2014). Chronic nutrient enrichment increases prevalence and severity of coral disease and bleaching. *Global Change Biology*, *20*(2), 544–554. http://doi.org/10.1111/gcb.12450

Veldman, J. W., Overbeck, G. E., Negreiros, D., Mahy, G., Le Stradic, S., Fernandes, G. W., Durigan, G., Buisson, E., Putz, F. E., & Bond, W. J. (2015). Where tree planting and forest expansion are bad for biodiversity and ecosystem services. *BioScience*, *65*(10), 1011–1018. https://doi.org/10.1093/biosci/biv118

Vences, M., & Köhler, J. (2008). Global diversity of amphibians (Amphibia) in freshwater. *Hydrobiologia*, *595*(1), 569–580. http://doi.org/10.1007/s10750-007-9032-2

Venegas-González, A., Juñent, F., Gutiérrez, A. G., & Tomazello Filho, M. (2018). Recent radial growth decline in response to increased drought conditions in the northernmost Nothofagus populations from South America. *Forest Ecology and Management*, *409*, 94–104. http://doi.org/10.1016/j.foreco.2017.11.006

Vergara, P. M., Pérez-Hernández, C. G., Hahn, I. J., & Soto, G. E. (2013). Deforestation in central Chile causes a rapid decline in landscape connectivity for a forest specialist bird species. *Ecological Research*, *28*(3), 481–492. http://doi.org/10.1007/s11284-013-1037-x

Vergara, P. M., & Simonetti, J. A. (2004). Avian responses to fragmentation of the Maulino forest in central Chile. *Oryx*, *38*(4), 383–388. http://doi.org/10.1017/S0030605304000742

Versveld, D. B., Le Maitre, D. C., & Chapman, R. A. (1998). *Alien Invading Plants and Water Resources in South Africa: a Preliminary Assessment*. Pretoria, South Africa: Water Research Commission.

Vihma, T., Screen, J., Tjernström, M., Newton, B., Zhang, X., Popova, V., Deser, C., Holland, M., & Prowse, T. (2016). The atmospheric role in the Arctic water cycle: A review on processes, past and future changes, and their impacts. *Journal of Geophysical Research: Biogeosciences*, *121*(3), 586–620. https://doi.org/10.1002/2015JG003132

Vila, I., Pardo, R., & Scott, S. (2007). Freshwater fishes of the Altiplano. *Aquatic Ecosystem Health & Management*, *10*(2), 201–211. http://doi.org/10.1080/14634980701351395

Vilela, B., & Villalobos, F. (2015). LetsR: a new R package for data handling and analysis in macroecology. *Methods in Ecology and Evolution*, *6*(10), 1229–1234.

Villagrán, C., & Castro, V. (2003). *Ciencia Indígena de los Andes del Norte de Chile*. Santiago. Chile: Editorial Universitaria.

Villagrán, C., & Hinojosa, L. F. (1997). History of the forests of southern South America. 2. phytogeographical analysis. *Revista Chilena de Historia Natural*, *70*(2), 241–267.

Villagrán, C., Romo, M., & Castro, V. (2003). Etnobotánica del sur de los Andes de la primera Región de Chile: un enlace entre las culturas altiplánicas y las de quebradas altas del Loa superior. *Chungará*, *35*(1), 73–124.

Villalba, R., Boninsegna, J. A., Veblen, T. T., Schmelter, A., & Rubulis, S. (1997). Recent trends in tree-ring records from high elevation sites in the Andes of northern Patagonia. *Climatic Change*, *36*(3–4), 425–454. http://doi.org/10.1023/a:1005366317996

Villamagna, A. M., & Murphy, B. R. (2010). Ecological and socio-economic impacts of invasive water hyacinth (Eichhornia crassipes): a review. *Freshwater Biology*, *55*(2), 282–298. http://doi.org/10.1111/j.1365-2427.2009.02294.x

Villarino, S. H., Studdert, G. A., Baldassini, P., Cendoya, M. G., Ciuffoli, L., Mastrángelo, M., & Piñeiro, G. (2016). Deforestation impacts on soil organic carbon stocks in the Semiarid Chaco Region, Argentina. *Science of The Total Environment*, *575*, 1056–1065. http://doi.org/10.1016/j.scitotenv.2016.09.175

Villegas, M., & Garitano-Zavala, Á. (2010). Bird community responses to different urban conditions in La Paz, Bolivia. *Urban Ecosystems*, *13*(3), 375–391. http://doi.org/10.1007/s11252-010-0126-7

Vitousek, P. M., Naylor, R., Crews, T., David, M. B., Drinkwater, L. E., Holland, E., Johnes, P. J., Katzenberger, J., Martinelli, L. A., Matson, P. A., Nziguheba, G., Ojima, D., Palm, C. A., Robertson, G. P., Sanchez, P. A., Townsend, A. R., & Zhang, F. S. (2009). Agriculture. Nutrient imbalances in agricultural development. *Science*, *324*, 1519–1520. https://doi.org/10.1126/science.1170261

Vitt, D. H. (2016). Peatlands of continental North America. In C. M. Finalyson, R. Milton, C. Prentice, & N. C. Davidson (Eds.), *The Wetland Book, II. Distribution, Description, and Conservation.* (pp. 1–6). Dordrecht, Netherlands: Springer.

Vo, Q. T., Kuenzer, C., Vo, Q. M., Moder, F., & Oppelt, N. (2012). Review of valuation methods for mangrove ecosystem services. *Ecological Indicators*, *23*, 431–446. http://doi.org/10.1016/j.ecolind.2012.04.022

Vörösmarty, C. J., Green, P., Salisbury, J., & Lammers, R. B. (2000). Global water resources: vulnerability from climate change and population growth. *Science*, *289*(5477), 284–288. http://doi.org/10.1126/science.289.5477.284

Vörösmarty, C. J., McIntyre, P. B., Gessner, M. O., Dudgeon, D., Prusevich, A., Green, P., Glidden, S., Bunn, S. E., Sullivan, C. A., Liermann, C. R., & Davies, P. M. (2010). Global threats to human water security and river biodiversity. *Nature*, *467*(7315), 555–561. https://doi.org/10.1038/nature09549

Voss, R. S., & Emmons, L. H. (1996). Mammalian diversity in neotropical lowland rainforests: a preliminary assessment. *Bulletin of the American Museum of Natural History*, *230*, 3–115. Retrieved from http://hdl.handle.net/2246/1671

Vuilleumier, F. (1985). Forest birds of Patagonia: ecological geography, speciation, endemism, and faunal history. *Ornithological Monographs*, *36*, 255–304. http://doi.org/10.2307/40168287

Vynne, C., Booth, R. K., & Wasser, S. K. (2014). Physiological implications of landscape use by free-ranging maned wolves (Chrysocyon brachyurus) in Brazil. *Journal of Mammalogy*, *95*(4), 696–706. http://doi.org/10.1644/12-MAMM-A-247

Walker, D. A., & Everett, K. R. (1987). Road dust and its environmental impact on Alaskan taiga and tundra. *Arctic and Alpine Research*, *19*(4), 479–489. http://doi.org/10.2307/1551414

Walker, J. S., Grimm, N. B., Briggs, J. M., Gries, C., & Dugan, L. (2009). Effects of urbanization on plant species diversity in central Arizona. *Frontiers in Ecology and the Environment*, *7*(9), 465–470. http://doi.org/10.1890/080084

Walker, M. D. (1995). Patterns and causes of Arctic plant community diversity. In F. S. Chapin III & C. Körner (Eds.), *Arctic and Alpine Biodiversity: Patterns, Causes and Ecosystem Consequences* (pp. 3–20). Berlin, Germany: Springer.

Walters, B. B., & Hansen, L. (2013). Farmed landscapes, trees and forest conservation in Saint Lucia (West Indies). *Environmental Conservation*, *40*(3), 211–221. http://doi.org/10.1017/S0376892912000446

Watling, L., & Norse, E. A. (1998). Disturbance of the seabed by mobile fishing gear: a comparison to forest clearcutting. *Conservation Biology*, *12*(6), 1180–1197. http://doi.org/10.1046/j.1523-1739.1998.0120061180.x

Watson, J. E. M., Dudley, N., Segan, D. B., & Hockings, M. (2014). The performance and potential of protected areas. *Nature*, *515*(7525), 67–73. http://doi.org/10.1038/nature13947

Webb, C. O., Ackerly, D. D., McPeek, M. A., & Donoghue, M. J. (2002). Phylogenies and community ecology. *Annual Review of Ecology and Systematics*, *33*(1), 475–505. http://doi.org/10.1146/annurev.ecolsys.33.010802.150448

Weed, A. S., Ayres, M. P., & Hicke, J. A. (2013). Consequences of climate change for biotic disturbances in North American forests. *Ecological Monographs*, *83*(4), 441–470. http://doi.org/10.1890/13-0160.1

Wege, D. C., Ryan, D., Varty, N., Anadon-Irizarry, V., & Perez-Leroux, A. (2010). Ecosystem Profile: The Caribbean Islands Biodiversity Hotspot. Arlington, USA: Critical Ecosystem Partnership Fund.

Wehrtmann, I. S., Ramírez, A., & Pérez-Reyes, O. (2016). Freshwater decapod diversity and conservation in Central America and the Caribbean. In T. Kawai &

N. Cumberlidge (Eds.), *A Global Overview of the Conservation of Freshwater Decapod Crustaceans* (pp. 267–301). Cham, Switzerlan: Springer.

Weigelt, A., Weisser, W. W., Buchmann, N., & Scherer-Lorenzen, M. (2009). Biodiversity for multifunctional grasslands: equal productivity in high-diversity low-input and low-diversity high-input systems. *Biogeosciences*, *6*(8), 1695–1706. http://doi.org/10.5194/bgd-6-3187-2009

Wester, L. (1991). Invasions and extinctions on Masatierra (Juan Fernandez Islands): a review of early historical evidence. *Journal of Historical Geography*, *17*(1), 18–34. http://doi.org/10.1016/0305-7488(91)90003-E

Westerling, A. L., Hidalgo, H. G., Cayan, D. R., & Swetnam, T. W. (2006). Warming and earlier spring increase western U.S. forest wildfire activity. *Science*, *313*(5789), 940–943.http://doi.org/10.1126/science.1128834

Westoby, M. (1998). A leaf-height-seed (LHS) plant ecology strategy scheme. *Plant and Soil*, *199*(2), 213–227. http://doi.org/10.1023/A:1004327224729

Wetmore, A., Friedmann, H., Amadon, D., Lincoln, F. C., Lowery, G. H., Miller, A. H., Peters, J. L., Pitelka, F. A., van Rossem, A. J., van Tyle, J., & Zimmer, J. T. (1957). *Check-list of North American Birds, 5th Edition*. Baltimore, USA: American Ornithologists' Union.

Wheeler, J. C. (2017). Evolución y domesticación en los camélidos sudamericanos. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 2. Investigación para el Manejo Sustentable de Recursos Genéticos en el Nuevo Mundo* (pp. 193–216). México D.F., México and Lima, Perú: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina/CONACYT.

Whitfield, S. M., Lips, K. R., & Donnelly, M. A. (2016). Amphibian decline and conservation in Central America. *Copeia*, *104*(2), 351–379. http://doi.org/10.1643/CH-15-300

Wiegmann, S. M., & Waller, D. M. (2006). Fifty years of change in northern upland forest understories: identity and traits of "winner" and "loser" plant species. *Biological Conservation*, *129*(1), 109–123. http://doi.org/10.1016/j.biocon.2005.10.027

Wierzchos, J., Ascaso, C., & McKay, C. P. (2006). Endolithic cyanobacteria in halite rocks from the hyperarid core of the Atacama Desert. *Astrobiology*, *6*(3), 415–422. http://doi.org/10.1089/ast.2006.6.415

Wiig, Ø., Amstrup, S., Atwood, T., Laidre, K., Lunn, N., Obbard, M., Regehr, E., & Thiemann, G. (2015). *Ursus maritimus*. *The IUCN Red List of Threatened Species 2015:* Retrieved from http://dx.doi.org/10.2305/IUCN.UK.2015-4.RLTS.T22823A14871490.en

Will-Wolf, S., & Stearns, F. (1999). Dry soil oak savanna in the Great Lakes region. In R. C. Anderson, J. S. Fralish, & J. M. Baskin (Eds.), *Savannas, Barrens, and Rock Outcrop Plant Communities of North America* (pp. 135–154). Cambridge, UK: Cambridge University Press.

Williams, A. P., Allen, C. D., Macalady, A. K., Griffin, D., Woodhouse, C. A., Meko, D. M., Swetnam, T. W., Rauscher, S. A., Seager, R., Grissino-Mayer, H. D., Dean, J. S., Cook, E. R., Gangodagamage, C., Cai, M., & McDowell, N. G. (2013). Temperature as a potent driver of regional forest drought stress and tree mortality. *Nature Climate Change*, *3*(3), 292–297. https://doi.org/10.1038/nclimate1693

Williams, A. P., Seager, R., Abatzoglou, J. T., Cook, B. I., Smerdon, J. E., & Cook, E. R. (2015). Contribution of anthropogenic warming to California drought during 2012–2014. *Geophysical Research Letters*, *42*(16), 6819–6828. http://doi.org/10.1002/2015GL064924

Williams, J. D., Warren Jr, M. L., Cummings, K. S., Harris, J. L., & Neves, R. J. (1993). Conservation status of freshwater mussels of the United States and Canada. *Fisheries*, *18*(9), 6–22. http://doi.org/10.1577/1548-8446(1993)018<0006:CSOFMO>2.0.CO;2

Willis, K. J. (Ed.). (2016). *State of the World´s Plants*. London, UK: Royal Botanic Gardens, Kew. Retrieved from https://stateoftheworldsplants.com/2016/

Wilsey, B. J., Martin, L. M., & Polley, H. W. (2005). Predicting plant extinction based on species-area curves in prairie fragments with high beta richness. *Conservation Biology*, *19*(6), 1835–1841.http://doi.org/10.1111/j.1523-1739.2005.00250.x

Wilsey, B. J., & Polley, H. W. (2004). Realistically low species evenness does not alter grassland species-richness-productivity relationships. *Ecology*, *85*(10), 2693–2700. http://doi.org/10.1890/04-0245

Wilson, B. S., Koenig, S. E., van Veen, R., Miersma, E., & Rudolph, D. C. (2011). Cane toads a threat to West Indian wildlife: mortality of Jamaican boas attributable to toad ingestion. *Biological Invasions*, *13*(1), 55–60. http://doi.org/10.1007/s10530-010-9787-7

Wilson, T. S., Sleeter, B. M., & Cameron, D. R. (2016). Future land-use related water demand in California. *Environmental Research Letters*, *11*(5), 1–12. http://doi.org/10.1088/1748-9326/11/5/054018

Winograd, M., Farrow, A., Aguilar, M., Clavijo, L. A., Debouck, D. G., Jones, P. G., Hyman, G., Lema, G., Nelson, A. N., Holmann, F. J., Tohme, M. J., Toro, C. O., & Voyset, V. O. (1999). *Agroecosystem Assessment for Latin America: Agriculture Extent, Production Systems and Agrobiodiversity*. Cali, Colombia: Centro Internacional de Agricultura Tropical (CIAT).

Wolf, A., Zimmerman, N. B., Anderegg, W. R. L., Busby, P. E., & Christensen, J. (2016). Altitudinal shifts of the native and introduced flora of California in the context of 20th-century warming. *Global Ecology and Biogeography*, *25*(4), 418–429. http://doi.org/10.1111/geb.12423

World Cities Culture Forum. (n.d.). % of public green space (parks and gardens). Retrieved from http://www.worldcitiescultureforum.com/data/of-public-green-space-parks-and-gardens

World Database of Key Biodiversity Areas. (n.d.). Retrieved from http://www.keybiodiversityareas.org

World Wildlife Fund (WWF). (2016). *Plowprint Report*. Retrieved from https://www.worldwildlife.org/projects/plowprint-report

World Wildlife Fund (WWF). (2017a). *2017 Plowprint Report*. Bozeman, USA. Retrieved from https://www.worldwildlife.org/projects/plowprint-report

World Wildlife Fund (WWF). (2017b). Northern South America: Northeastern Brazil. Retrieved from https://www.worldwildlife.org/ecoregions/nt1304

Worm, B., Hilborn, R., Baum, J. K., Branch, T. A., Collie, J. S., Costello, C., … Zeller, D. (2009). Rebuilding global fisheries. *Science*, *325*(5940), 578–585. http://doi.org/10.1126/science.1173146

Wright, I. J., Reich, P. B., Westoby, M., Ackerly, D. D., Baruch, Z., Bongers, F., Cavender-Bares, J., Chapin, T., Cornelissen, J. H. C., Diemer, M., Flexas, J., Garnier, E., Groom, P. K., Gulias, J., Hikosaka, K., Lamont, B. B., Lee, T., Lee, W., Lusk, C., Midgley, J. J., Navas, M.-L., Niinemets, Ü., Oleksyn, J., Osada, N., Poorter, H., Poot, P., Prior, L., Pyankov, V. I., Roumet, C., Thomas, S. C., Tjoelker, M. G., Veneklaas, E. J., & Villar, R. (2004). The worldwide leaf economics spectrum. *Nature*, *428*(6985), 821–827. https://doi.org/10.1038/nature02403

Wright, J. A., DiNicola, A., & Gaitan, E. (2000). Latin American forest plantations: opportunities for carbon sequestration, economic development, and financial returns. *Journal of Forestry*, *98*(9), 20–23. Retrieved from http://www.scopus.com/inward/record.url?eid=2-s2.0-0034283558&partnerID=40&md5=0cc1b0bb502bf3c27297f4b0452c9f0a

Wrigley de Basanta, D., Lado, C., Estrada-Torres, A., & Stephenson, S. L. (2010). Biodiversity of myxomycetes in subantarctic forests of Patagonia and Tierra del Fuego, Argentina. *Nova Hedwigia*, *90*(1–2), 45–79. http://doi.org/10.1127/0029-5035/2010/0090-0045

Young, B. E., Lips, K. R., Reaser, J. K., Ibáñez, R., Salas, A. W., Cedeño, J. R., Coloma, L. A., Ron, S., La Marca, E., Meyer, J. R., Muñoz, A., Bolaños, F., Chaves, G., & Romo, D. (2001). Population declines and priorities for amphibian conservation in Latin America. *Conservation Biology*, *15*(5), 1213–1223. https://doi.org/10.1111/j.1523-1739.2001.00218.x

Young, C. N., Carlson, J., Hutchinson, M., Kobayashi, D., McCandless, C., Miller, M. H., Teo, S., & T. Warren. (2016). *Status review report: common thresher shark (Alopias vulpinus) and bigeye thresher shark (Alopias superciliosus). Final Report to National Marine Fisheries Service, Office of Protected Resources*.

Young, K. L., Lafrenière, M. J., Lamoureux, S. F., Abnizova, A., & Miller, E. A. (2015). Recent multi-year streamflow regimes and water budgets of hillslope catchments in the Canadian High Arctic: evaluation and comparison to other small Arctic watershed studies. *Hydrology Research*, *46*(4), 533–550. http://doi.org/10.2166/nh.2014.004

Young, O. R., & Chapin III, F. S. (1995). Anthropogenic impacts on biodiversity in the Arctic. In F. S. Chapin III & C. Körner (Eds.), *Arctic and Alpine Biodiversity: Patterns, Causes and Ecosystem Consequences* (pp. 183–196). Berlin, Germany: Springer.

Zak, M. R., Cabido, M., & Hodgson, J. G. (2004). Do subtropical seasonal forests in the Gran Chaco, Argentina, have a future? *Biological Conservation*, *120*(4), 589–598. http://doi.org/10.1016/j.biocon.2004.03.034

Zamora Nasca, L., Montti, L., Grau, R., & Paolini, L. (2014). Effects of glossy privet's invasion on the water dynamics of the Argentinean Yungas forest. *Bosque*, *35*(2), 195–205. http://doi.org/10.4067/S0717-92002014000200007

Zamorano-Elgueta, C., Rey Benayas, J. M., Cayuela, L., Hantson, S., & Armenteras, D. (2015). Native forest replacement by exotic plantations in southern Chile (1985-2011) and partial compensation by natural regeneration. *Forest Ecology and Management*, *345*, 10–20. http://doi.org/10.1016/j.foreco.2015.02.025

Zanne, A. E., Tank, D. C., Cornwell, W. K., Eastman, J. M., Smith, S. A., FitzJohn, R. G., McGlinn, D. J., O'Meara, B. C., Moles, A. T., Reich, P. B., Royer, D. L., Soltis, D. E., Stevens, P. F., Westoby, M., Wright, I. J., Aarssen, L., Bertin, R. I., Calaminus, A., Govaerts, R., Hemmings, F., Leishman, M. R., Oleksyn, J., Soltis, P. S., Swenson, N. G., Warman, L., & Beaulieu, J. M. (2013). Three keys to the radiation of angiosperms into freezing environments. *Nature*, *506*, 89–92. https://doi.org/10.1038/nature12872

Zarazúa, M. (2016). Del guajolote a las chicatanas. Uso, manejo y domesticación de recursos genéticos animales en Mesoamérica. In A. Casas, J. Torres-Guevara, & F. Parra (Eds.), *Domesticación en el Continente Americano Vol. 1. Investigación Manejo de Biodiversidad y Evolución Dirigida por las Culturas del Nuevo Mundo* (pp. 283–316). México D.F., México and: Universidad Nacional Autónoma de México/Universidad Nacional Agraria La Molina.

Zarin, D. J., Harris, N. L., Baccini, A., Aksenov, D., Hansen, M. C., Azevedo-Ramos, C., Azevedo, T., Margono, B. A., Alencar, A. C., Gabris, C., Allegretti, A., Potapov, P., Farina, M., Walker, W. S., Shevade, V. S., Loboda, T. V, Turubanova, S., & Tyukavina, A. (2016). Can carbon emissions from tropical deforestation drop by 50% in 5 years? *Global Change Biology*, *22*(4), 1336–1347. https://doi.org/10.1111/gcb.13153

Zavaleta, E., Tershy, B., Harrison, S., Borker, A., Sinervo, B., Cornelisse, T., Li, C., Spatz, D., & Croll, D. (2016). Biodiversity. In H. Mooney and E. Zavaleta (Ed.), *Ecosystems of California* (pp. 187–212). Oakland, USA: University of California Press.

Zedler, J. B., & Kercher, S. (2005). Wetland resources: Status, trends, ecosystem services, and restorability. *Annual Review of Environment and Resources*, *30*, 39–74. http://doi.org/10.1146/annurev.energy.30.050504.144248

Zedler, P. H. (1995). Fire frequency in southern California shrublands: biological effects and management options. In J. E. Keeley & T. Scott (Eds.), *Brushfires in California: Ecology and Resource Management* (pp. 101–112). Fairfield, USA: International Association of Wildland Fire.

Zolla, C., & Zolla, E. (2004). *Los Pueblos Indígenas de México, 100 Preguntas*. México D.F., México: Universidad Nacional Autónoma de México.