CHAPTER 4

DIRECT AND INDIRECT DRIVERS OF CHANGE IN BIODIVERSITY AND NATURE'S CONTRIBUTIONS TO PEOPLE

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DIRECT AND INDIRECT DRIVERS OF CHANGE IN BIODIVERSITY AND NATURE'S CONTRIBUTIONS TO PEOPLE

4.1 EXECUTIVE SUMMARY

① **The most important indirect anthropogenic drivers of changes in nature, nature's contributions to people and good quality of life include unsustainable patterns of economic growth (including issues related to international trade and finances); population and demographic trends; weaknesses in the governance systems and inequity (*well established*).** Increasing human demand for food, water, and energy caused by increases in population, per capita Gross Domestic Product and international trade have had negative consequences for nature and many regulating and non-material nature's contributions to people.

② **Social inequity is a concern with adverse implications for nature, nature's contributions to people and good quality of life (*well established*). When the United Nations Development Program Human Development Index is adjusted for inequality, it is 22 per cent lower in Latin American and Caribbean countries and 11.1 per cent lower in North America {4.3.6}.** Seventy-two million people escaped income-poverty from 2003-2013 in Latin America; however, around 26.9 per cent of the Latin American population still lived in poverty in 2012: 40.6 per cent in Mesoamerica and 21 per cent in South America {4.3.6}. In many cases, poor people in the Americas tend to increase the pressures on nature merely to survive, while on the other hand, there is high per capita consumption of natural resources in affluent segments of the population.

③ **Economic growth (measured as Gross Domestic Product growth and Gross Domestic Product per capita) and international trade are major drivers of natural resource consumption in the Americas. Economic growth and trade can positively or negatively impact biodiversity and nature's contributions to people, but currently, on balance, they adversely impact biodiversity and nature's contributions to people when environmental and social development goals are insufficiently accounted for (*well established*).** Positive impacts of economic growth and international trade may include a stronger economy and increased employment, and social and environmental investments such as biodiversity protection. Negative impacts of economic growth include unsustainable conversion, use and exploitation of terrestrial, freshwater and marine ecosystems and resources, which threaten biodiversity and degrade nature's contributions to people by reducing species abundances below self-sustaining levels and by disrupting key ecosystem functions {4.6}. The Americas generates around 18 per cent of world exports, with 70 per cent of this from North America. The Latin American and Caribbean contributions to world exports is 5.4 per cent, and natural resource governance is strongly influenced by having economies dominated by commodity exports. Natural resources (oil, minerals, and agriculture) contribute more than 50 per cent to these Latin America and the Caribbean exports {4.3.3}. Globalization has catalyzed rapid growth of international trade and become an important motor for regional development, but it has also disconnected places of production, transformation and consumption of land-based products. This decoupling places significant challenges for socio-environmental governance and regulatory implementation for sectors rapidly changing in response to increases in the global demand for food, feed and fiber. Consequently, natural resource use policies often come into place only after fundamental shifts in the land-use system are already underway, and interventions have become costly and have limited influence {4.6}.

④ **Weaknesses in the governance systems and institutional frameworks in the Americas have had adverse implications for nature, nature's contributions to people and good quality of life in the Americas (*well established*).** In most countries in the region centralized modes of governance still prevail where decision-making regarding Nature and nature's contributions to people in reality falls on the State. Centralized command and control measures nonetheless, such as the establishment of protected areas, continue to be a pillar of biodiversity conservation. Significant progress has been made to include other actors and new hybrid governance modes such as public-private certification schemes or payment for ecosystem services, which are in line with the rising role of markets in environmental governance. These transformations from centralized to descentralized forms, however, have led to significant socioenvironmental conflicts in the region {4.3.1}.

5 **Value systems in the Americas differ among cultural groups and identities across the whole region and shape governance systems, in particular the ways of addressing development policies, land tenure and indigenous rights, and strongly influence decisions on land use and natural resources exploitation in the different subregions (*well established*).** Indigenous and traditional peoples throughout the Americas have developed many different socio-economic systems (nationally and locally). Indigenous and local knowledge are expressions of social articulations that can positively influence biodiversity and ecosystem services. While cases that conservation of biodiversity and nature's benefits to people are related to empowerment of indigenous and traditional communities are emerging in the region (for example, the role of indigenous land on deforestation control in tropical forests of South America), weak and less participatory governance systems are associated with cases of conflicts in managing land and natural resources in all of the Americas subregions (for example, conflicts related to infrastructure building in indigenous lands) {4.3.1, 4.3.6}.

6 **Habitat conversion, fragmentation and overexploitation/overharvesting are resulting in a loss of biodiversity and a loss of nature's contributions to people in all ecosystems. Habitat degradation due to land conversion and agricultural intensification; wetland drainage and conversion; urbanization and other new infrastructure, and resource extraction is the largest threat to fresh water, marine and terrestrial biodiversity and nature's contributions to people in the Americas (*well established*).** The resulting changes in terrestrial, freshwater and marine environments are interrelated and often lead to changes in biogeochemical cycles, pollution of ecosystems and eutrophication, and biological invasions, which are at the same time significant direct drivers of change in the region (*well established*). The expansion and intensification of agriculture and livestock production in the Americas are decreasing the area of and altering natural ecosystems (*well established*) {4.4.1}. Related changes include shifting drainage patterns (affecting infiltration and runoff), water quality degradation, soil disturbance, habitat loss, and release of chemicals that can be toxic to biota and human populations. Nitrogen and phosphorus fertilizer use have greatly contributed to increases in the amount of available nitrogen and phosphorus in the environment, doubling available nitrogen, for example, with negative consequences for ecosystem function, and air, soil and water quality {4.4.2}, including major contributions to coastal and freshwater oxygen depletion. Land-use changes, road and trail construction, waterways and domestic animals are common dispersal routes for invasive species (*well established*) {4.4.4}. Habitat conversion also decreases connectivity among, and diversity within, remaining fragments of natural ecosystems (*well established*). Wildlife, fisheries, and people, including many indigenous peoples, are exposed to residual pollution in the environment. Mining for trace metal ores and coal has left lasting legacies of toxic pollution across the region {4.4.2} (*well established*). Although unsustainable management of natural resources are threatening biodiversity and degrading nature's contributions to people by reducing populations below natural self-sustaining levels and disrupting ecosystem functions {4.4.5}, some sustainable practices have been identified and used in terrestrial and aquatic environments.

7 **Rapid urbanization is a key driver of loss of biodiversity and nature's contributions to people, but the nature and the magnitude of impacts vary substantially among subregions of the Americas (*established but incomplete*).** The Americas region is highly urbanized, with about 80 per cent of the region's population residing in urban settings {4.3.5}. Although urban population impacts depend on consumption patterns and lifestyles, which vary considerably from one subregion to another, in all subregions a large number of ecosystems have been affected. Urbanization driven by growing populations and internal migration acts as an indirect driver of land-use change through linear infrastructures. In Latin America and the Caribbean, 12 per cent of the urban population and 36 per cent of rural population do not have access to improved sanitation facilities, and only 50 per cent of the population in Latin America is connected to sewerage. The poor systematic waste management in Latin America and the Caribbean implies pollution of inland waters and coastal areas {4.4.2} affecting biodiversity and human health.

8 **Carbon dioxide emissions from fossil fuel production continue to increase, increasing 29 per cent from 2000 to 2008. The combustion of fossil fuels is not only the primary source of anthropogenic greenhouse gases that cause human-induced climate change, but fossil fuel combustion itself is also a major source of pollution adversely impacting most terrestrial and marine ecosystems and human health {4.4.2} (*well established*). Air pollution (especially particulates, ozone, mercury, and carcinogens) causes significant adverse health effects on infants, adults and biodiversity (*well established*), and carbon dioxide emissions cause ocean acidification.** For example, the combustion of fossil fuels account for 25 per cent of the direct anthropogenic mercury emissions that are increasing the mercury burden of polar and subpolar wildlife and indigenous people with diets dominated

by fish, eggs of fish-eating birds, and marine mammals, affecting wildlife reproduction and infant nervous systems. Ocean acidification from increased atmospheric carbon dioxide is increasing and is already impacting major components of the Pacific Ocean food web and contributing to a Caribbean-wide flattening of coral reefs. If current trends continue, coral reef systems will be further adversely affected. Ocean temperatures have become warmer, and together with nutrient run-off, are contributing to increasing ocean deoxygenation. Fossil fuel combustion also contributes to human-caused atmospheric nitrogen deposition, being responsible for 16 per cent of anthropogenic creations of reactive nitrogen, which shifts the species composition of ecosystems and makes groundwater toxic. Fossil fuel related nitrogen emissions have declined in North America.

9 **Marine plastic pollution is increasing, and it is expected to exacerbate stresses on the marine food web from warming temperatures, acidification and overexploitation (*establisehd but incomplete*).** In 2010, globally and from land-based sources alone, five to 13 million metric tons of plastic pollution entered the ocean. Two countries of the Americas are among the 20 top polluters. The environmental implications of microplastics at sea are still largely unknown, however the number of marine species known to be affected by this contaminant has gone from 247 to 680 {4.4.2}. New evidence indicates microplastics have a complex effect on marine life and are is transferred up the food chain to people. Impacts on marine wildlife include entanglement, ingestion, death and contamination to a wide variety of species.

10 **Human induced climate change caused by the emissions of greenhouse gases is becoming an increasingly more important direct driver, amplifying the impacts of other drivers (i.e. habitat degradation, pollution, invasive species and overexploitation) through changes in temperature, precipitation and frequency of extreme events and other variables (*well-established*).** Climate change has, and will continue to, adversely affect biodiversity at the genetic, species and ecosystem level. The majority of ecosystems in the Americas have already experienced increased mean and extreme temperatures and/or precipitation which have, for example, caused changes in species distributions and ecosystem boundaries, and caused mountain glaciers to retreat. However, the interaction between these direct impacts and other direct and indirect drivers are increasing vulnerability of sensitive ecosystems through the interaction of warming temperatures and pollution, as in the example of coral reefs in the Caribbean. The main impacts on terrestrial, freshwater and marine species are the shift in their geographic ranges, and changes in seasonal activities, migration patterns and abundances. Species affected by other drivers are less resilient to climate change and therefore have a high extinction risk.

11 **Although most ecosystems in the America's continue to be degraded, increases in conservation (e.g. protected areas), and in ecological restoration, are having positive effects. Ecological restoration significantly speeds up ecosystem recovery in some cases (*well established*), but costs can be significant, and full reversal of the adverse impacts of humans on nature is unlikely to be achievable (*well established*).** Evidence from different subregions indicates that structure and functionality of ecosystems recover faster than species richness (particularly in species-rich biomes). Non-material contributions of naature to people may not be restored for some people {4.4.1}.

12 **In spite of the pressures of drivers of change on nature and nature's contributions to people, there are management and policy options that can affect the drivers of change in order to mitigate, and most importantly, to avoid, impacts on different ecosystems (*establisehd but incomplete*)**. However, given the current status and trends of drivers, meeting the Aichi targets and Sustainable Development Goals will require stronger and more effective efforts on the parts of the countries across the region. These options and their implementation are context dependent and strongly influenced by values, governance and institutions {4.7}. Such conditions vary substantially across the Americas in relation to social and economic inequity.

4.2 INTRODUCTION

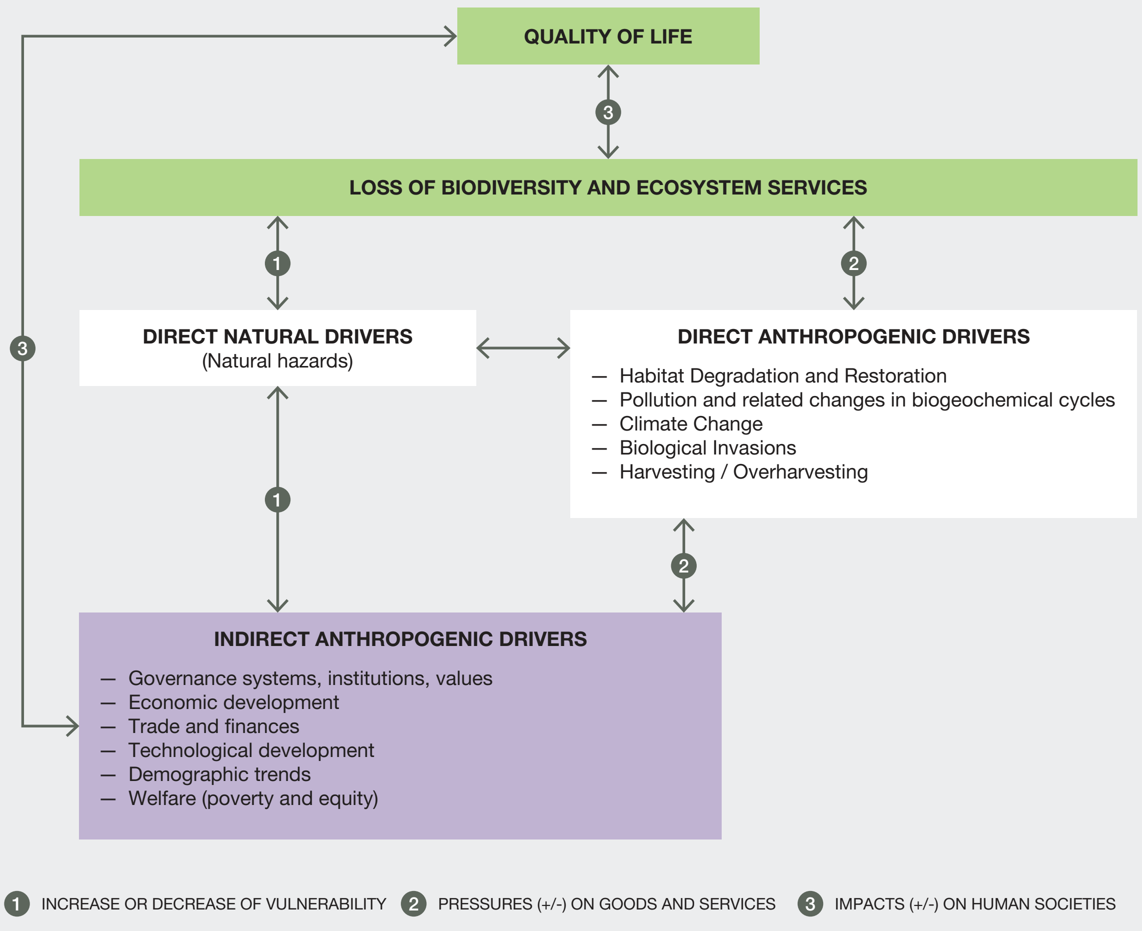
The Americas encompass seven magadiverse countries (one in North America, one in Mesoamerica and five in South America) of the 17 in the world (see Chapter 1 for more details). However, the degradation of critical ecosystems and loss of biodiversity in the region threaten human well-being by impacting important ecosystem functions and services, like clean air and water, flood and climate control, and soil regeneration, as well as food, medicines and raw materials (see Chapter 2 for more details).

As a function of the pressure on natural ecosystems, the Americas contain 10 of the 36 world biodiversity hotspots, i.e. areas with high biodiversity facing extreme threats and that have lost at least 70 percent of their original habitat: 1. California floristic province (USA), 2. North American coastal plain (USA), 3. madrean pine-oak woodlands (USA and Mexico), 4. Mesoamerica, 5. Caribbean islands, 6. Atlantic forest (Brazil), 7. Cerrado (Brazil), 8. Chilean winter rainfall-Valdivian forests (Chile), 9. Tumbes-Chocó-Magdalena (Colombia) and 10. Tropical Andes (Marchese, 2015, http://www.cepf.net/resources/hotspots/).

Environmental problems are also wide-ranging and vary between and within nations. Negative environmental trends are observed throughout the region, which are to a large extent the result of long historical patterns of growth induced by non-sustainable consumption. A significant feature of these environmental problems is that they are often shared among countries, including climate change and disaster risk management, sustainable management of land and ecosystems, water resources management, sustainable energy management, good governance for inclusive and sustainable development, such that regional cooperation is needed to tackle them (UNEP, 2016).

Social and economic inequality and weak environmental governance are common features in the Americas that are intricately linked with a deteriorating environment. Environmental and climate change issues are gaining

Figure 4 1 **Interactions among multiple indirect and direct drivers of biodiversity loss and ecosystem changes. Source: own representation.**



weight regionally, but unsustainable development models still predominate, with significant consequences for the environment and human well-being. Lack of security and equity in accessing basic resources (like land ownership or user rights, access to the natural commons and fundamental ecosystem services) do not provide incentives for sustainable management or increased efficiency. However, sustainable use might provide an opportunity to improve welfare for the people (UNEP, 2016)

Given the importance of the Americas' biodiversity and ecosystem services for human well-being (see Chapters 2 and 3 for more details), this chapter explores key drivers of changes in biodiversity and ecosystem services in the region. These include indirect and direct anthropogenic drivers as well as direct natural drivers.

A range of drivers, including environmental change and human uses of resources, induce changes in biodiversity and ecosystems. A driver is any natural or human-induced factor that directly or indirectly causes a change. A direct driver unequivocally influences ecosystem processes. An indirect driver operates more diffusely, by altering one or more direct drivers. **Box 4.1** summarizes the definitions on drivers included in the Intergovernmental Platform on Biodiversity and Ecosystem Services (IPBES) conceptual framework (Decision IPBES-2/4, available on http://www.ipbes.net).

The drivers examined in this chapter are primarily anthropogenic. Indirect anthropogenic drivers are aspects and patterns of human organization and socioeconomic activity (section 4.3) that produce aggregate outcomes that in turn bring about changes in biodiversity and ecosystem services. Direct anthropogenic drivers (section 4.4) are the aggregate outcomes, such as habitat change, pollution or climate change, from the indirect anthropogenic drivers that yield those changes. Direct natural drivers also produce changes in biodiversity and ecosystem services, and are thus also presented briefly in this chapter (section 4.5).

Box 4 1 Definitions of drivers of change of nature´s contributions to people and good quality of life, and partial representation of the IPBES conceptual framework according to IPBES Decision 2-4.

Drivers of change refers to all those external factors that affect nature, anthropogenic assets, nature's contributions to people and a good quality of life. They include institutions and governance systems and other indirect drivers and direct drivers (both natural and anthropogenic).

Institutions and governance systems and other indirect drivers are the ways in which societies organize themselves, and the resulting influences on other components. They are the underlying causes of environmental change that are exogenous to the ecosystem in question. Because of their central role, influencing all aspects of human relationships with nature, these are key levers for decision-making. Institutions encompass all formal and informal interactions among stakeholders and social structures that determine how decisions are taken and implemented, how power is exercised, and how responsibilities are distributed. Institutions determine, to various degrees, the access to, and the control, allocation and distribution of components of nature and anthropogenic assets and their benefits to people.

Direct drivers, both natural and anthropogenic, affect nature directly.

Natural drivers are those that are not the result of human activities and are beyond human control. These include earthquakes, volcanic eruptions and tsunamis, extreme weather or ocean-related events such as prolonged drought or cold periods, tropical cyclones and floods, the El Niño/La Niña Southern Oscillation and extreme tidal events.

The direct anthropogenic drivers are those that are the result of human decisions, namely, of institutions and governance systems and other indirect drivers. Anthropogenic drivers include habitat conversion, e.g. degradation of land and aquatic habitats, deforestation and afforestation, exploitation of wild populations, climate change, pollution of soil, water and air and species introductions. Some of these drivers, such as pollution, can have negative impacts on nature; others, as in the case of habitat restoration, or the introduction of a natural enemy to combat invasive species, can have positive effects. Institutions and governance systems and other indirect drivers affect all elements and are the root causes of the direct anthropogenic drivers that directly affect nature and also affect the interactions and balance between nature and human assets in the co-production of nature's benefits to people

Anthropogenic assets refer to built-up infrastructure, health facilities, knowledge (including indigenous and local knowledge systems and technical or scientific knowledge, as well as formal and non-formal education), technology (both physical objects and procedures), as financial assets, among others. Direct drivers also affect anthropogenic assets and in addition, anthropogenic assets directly affect the possibility of leading a good life through the provision of and access to material wealth, shelter, health, education, satisfactory human relationships, freedom of choice and action, and sense of cultural identity and security. These linkages are acknowledged but not addressed in depth because they are not the main focus of IPBES.

As **Figure 4.1** shows, the indirect and direct anthropogenic drivers are significantly interrelated. Even though sections 4.3, 4.4, and 4.5 describe these drivers sequentially and distinctly, important interactions are also presented in the specific sections (indicated in bold along the text). These interactions will be synthesized in section 4.6, while the effects of indirect drivers on direct drivers are further discussed in section 4.7. Section 4.8 provides a starting indication of where gaps in current scientific knowledge lie. The gaps in knowledge point to areas where data remain insufficient and to areas where further data collection and scientific inquiry and analysis are needed to produce a stronger understanding of the links between indirect and direct anthropogenic drivers, changes in biodiversity and ecosystem services, and human well-being.

Lastly, section 4.9 contains supplementary material that enrich the chapter by displaying additional content that add detail, background, or context by resources such as case studies, figures and tables.

4.3 INDIRECT ANTHROPOGENIC DRIVERS

Indirect drivers (also referred as underlying factors) play a major role in influencing direct drivers (proximate causes) of changes in nature, nature's contributions to people and good quality of life in different spatial and temporal scales, involving "anthropogenic assets" (encompassing infrastructure, knowledge systems, including indigenous and local knowledge (ILK), technology and financial assets, among others). Considering the concept and the nature of complex ecological systems, the role of indirect drivers is an integral aspect of natural resource use assessments, and needs to be considered to explain and study past and ongoing processes as well as for scenario development and subsequent analysis (IPBES, 2016).

The indirect anthropogenic drivers can be classified according to the origin of the driver, which for instance can be fed by predominantly local processes, like for example poor local governance and corruption. It is widely recognized that globalization in recent decades has led to "spatial decoupling of the local land uses from their most important driving forces" (Reenberg *et al.,* 2010). This recent observation has led to the establishment of the teleconnection framework (Friis *et al.,* 2015; Kastner *et al.,* 2015). For instance, changes in land systems at various spatial scales are influenced by long-distance flows of capital, energy, traded products, people and information. While locally driven processes have been studied for decades using perspectives from different disciplines (demography, anthropology, political economy), teleconnections have been assessed only in the last decade. Furthermore, it is only recently that the teleconnection framework has given birth to the concept of telecoupling (Liu *et al.,* 2013), which considers also the multiple feedbacks and teleconnected interactions in both socioeconomic and environmental terms. For example, climate risks may be transmitted to a region via trade networks, but also through migration flows into that region that can be triggered by climate risks elsewhere. In both cases local socio-economic conditions in that region are affected, and therefore its natural resource management. The complexity and multi-layered nature of these interactions hampers the design and implementation of governance measures. However, at the same time it may also allow the participation of a number of distal actors and processes, opening space for mobilizing resources and fostering a more coordinated, beyond borders and polycentric approach to natural resource governance (Godar *et al.,* 2016).

The discussion on the indirect anthropogenic drivers for changes in nature, nature´s contributions to people (NCP) and good quality of life is a relevant component of the Development Agenda 2030 and the Sustainable Development Goals (SDG). Equity, literacy level, share of population in extreme poverty, income distribution, access to public health, health care infrastructure, food security, political organization and socio-cultural aspects are relevant variables to define the critical mass of a country and the capacity of social debate, and hence its "anthropogenic assets". On the other hand, the worldviews and culture (attitudes to environment/sustainability/equity), life-styles (including diets) and the level of societal tension and conflict are other important drivers of opposition or consensus in the economic and political arena. The level of efficiency in governance systems, the legislation and the strength of the institutions involved in decision-making and their implementation capacity, and their level of credibility and transparency, are also drivers that will influence the status and trend of NCP.

This section describes the current status and trends of six broad indirect anthropogenic drivers of changes in NCP in the Americas: Governance systems and institutions (4.3.1); Economic growth (4.3.2); International trade and finance (4.3.3); Population and demographic trends (4.3.4); Technological development (4.3.5); and Welfare and human development (4.3.6). Internationally comparable socioeconomic data for Greenland is limited in regional sources of the Americas, considering that Greenland has been politically and to some extent culturally associated with Europe for more than a millennium. Systematic socioeconomic data of other Protectorates located in the Americas were also not included in the following sections.

4.3.1 Governance systems and institutions (formal and informal)

There is a widespread consensus that governance (see definition **Box 4.2**) has a strong effect on environmental outcomes (Smith *et al.,* 2003; Armitage *et al.,* 2012; Delmas & Young, 2009; de Castro *et al.,* 2016), although there is very limited empirical evidence relating governance measures to biodiversity and changes in ecosystem services.

In response to such consensus, there is a growing demand for governance arising from human-environment interactions, which nonetheless is escorted by a declining confidence in the capacity of governments to address such matters (Delmas & Young, 2009).

Rule of law, citizen's rights of access to information, community participation and even access to justice have been recognized as a basis for poverty reduction and sustainable development as reflected by SDG16 "Peace, justice and strong institutions". Evidence from the Americas reveals important differences across subregions for major governance indicators (defined **Box 4.3**) in the last two decades, as reported by the World Bank **Figure 4.2**.

Voice and accountability shows a decrease after 2004, except for the Caribbean islands. In turn, political stability and no violence fluctuated and decreased in North America until 2004 and then slightly recovered afterwards in all subregions. The other four indicators have remained largely stable over time according to public perception, with Mesoamerica and South America below the other two subregions. Yet, these aggregate figures hide particularities of specific countries and they should be taken carefully. These indicators have been criticized for their "construct validity", that is, whether the indicators measure what they intend to measure (Thomas, 2010), and for their methodology being too broad and biased (Langbein & Knack, 2010). These and previous critiques have been in turn contested (Kaufmann *et al.,* 2007), assuring the validity of the indicators and the methodological procedures.

Reinforcing the rule of law in the environmental domain from current levels is critical to the achievement of SDG and Aichi targets in the region. The importance of this matter was first

Box 4 2 The meaning of governance.

The broader definitions of governance are linked to international agencies (e.g. World Bank and Organization for Economic Cooperation and Development, OECD) and standards of "good" public governance (Armitage *et al.,* 2012). These standards encompass accountability, transparency, responsiveness, equity and inclusion, effectiveness and efficiency, following the rule of law, and participatory, consensus-oriented decision making (Crabbé & LeRoy, 2008).

Environmental governance, as a subclass of the broader governance concept, has been defined as "the set of regulatory processes, mechanisms and organizations through which political actors influence environmental actions and outcomes" (Lemos & Agrawal, 2006), and it "should be understood broadly so as to include all institutional solutions for resolving conflicts over environmental resources" (Paavola, 2007).

Box 4 3 Definitions of governance indicators (Reproduced from Kaufmann *et al.* (2010).

Voice and accountability, capturing perceptions of the extent to which a country's citizens are able to participate in selecting their government, as well as freedom of expression, freedom of association, and a free media.

Political stability and absence of violence/terrorism, capturing perceptions of the likelihood that the government will be destabilized or overthrown by unconstitutional or violent means, including politically-motivated violence and terrorism.

Government effectiveness, capturing perceptions of the quality of public services, the quality of the civil service and the degree of its independence from political pressures, the quality of policy formulation and implementation, and the credibility of the government's commitment to such policies.

Regulatory quality, capturing perceptions of the ability of the government to formulate and implement sound policies and regulations that permit and promote private sector development.

Control of corruption, capturing perceptions of the extent to which public power is exercised for private gain, including both petty and grand forms of corruption, as well as "capture" of the state by elites and private interests.

Rule of law, capturing perceptions of the extent to which agents have confidence in and abide by the rules of society, and in particular the quality of contract enforcement, property rights, the police, and the courts, as well as the likelihood of crime and violence (see Chapter 2, section 2.6).

Figure 4 2 **Trends in World Bank Governance Indicators for the Americas between 1996 and 2014/2015, expressed in percentile rank, where lowest is 0 and highest is 100.**

Source: own representation constructed from data available at http://info.worldbank.org/governance/wgi/#home.

recognized by the Rio Declaration and has been recently corroborated by the International Union for Conservation of Nature (IUCN) World Declaration on the Environmental Rule of Law in 2017. "Without the environmental rule of law and the enforcement of legal rights and obligations, environmental governance, conservation and protection may be arbitrary, subjective, and unpredictable" (IUCN, 2017).

On the other hand, the impacts of political instability on natural resources use have been tremendously negative in the region (Baud *et al.,* 2011; Ruyle, 2017), particularly in South America in the last decade (Arsel *et al.,* 2016). The most prominent conflicts concern mining in Brazil (see for example Tofóli *et al.,* 2017), Ecuador (Avci & Fernández-Salvador, 2016), Honduras (Middeldorp *et al.,* 2016) and Peru (Paredes, 2016), the use of rangelands for energy production (e.g. biofuels, solar) in the USA, Mexico and Canada (Kreuter *et al.,* 2016), water use in most countries (Philpot *et al.,* 2016), oil investments in Canada (Hebblewhite, 2017), and hydroelectricity projects on indigenous lands in Chile (Silva 2016), Colombia (Martínez & Castillo, 2016) and Canada.

Despite an impressive body of laws and institutions, the Region finds itself far off track in fulfilling the vision of sustainable development as indicated by the monitoring of the sustainable development goals (http://www.mdgmonitor.org). Political corruption (people exploiting public office for financial or other individual gain) is persistent in many countries and may have a significant impact on nature conservation by endorsing overexploitation of forests, wildlife, fisheries and other resources, and by impairing the effectiveness of conservation actions (Smith *et al.,* 2003; Laurence, 2004). Few studies conducted in the region show the effect of corruption on biodiversity loss. Bulte *et al.* (2007) find a positive association between corruption and expansion of agricultural land (by subsidies), which is detrimental to forests in Latin America. Miller (2011) examines how corruption among forestry regulators in Costa Rica is one important factor that leads them to allow people to log illicitly. Yet, more robust studies showing causality between weak governance and biodiversity and ecosystem services loss are clearly needed for the Region.

Evolution of governance modes in the Americas and effects on nature conservation

Governments and States are no longer the most important basis of decision-making in the environmental field of the Americas. Instead, new actors (e.g. Non-Govermental Organizations (NGO)), researchers, indigenous groups) are performing critical roles and new mechanisms and forums are arising (e.g. The Economics on Economics and Biodiversity and IPBES) **Figure 4.3** (Paavola, 2007; Armitage *et al.,* 2012).

Different perceptions and values are strongly contested by different actors according to their images of nature (Sténs *et al.,* 2016). Values, ideologies and sources of knowledge, which guide the manner in which nature is conceptualized,

Figure ④ ③ **Modes of environmental governance and actors involved. Source: Modified from Armitage *et al.* (2012).**

are key elements of environmental governance (de Castro *et al.,* 2016; Inoue & Moreira, 2016) and they seem to be in increasing dispute. They influence how environmental issues are problematized, how solutions are planned, and how priorities and agreements are established between conflicting objectives. Therefore, the more actors involved in environmental governance, the more complex and heterogeneous the images become (de Castro *et al.,* 2016; Tijoux, 2016).

Environmental governance in the Americas has gone through major transformations in the last decades **Figure 4.4** and yet biodiversity and ecosystem services continue to decline. From the mid-1980s onwards, most countries turned away from centralized, state-based institutional arrangements and direct regulation (Baud *et al.,* 2011). Common problems around centralized modes of governance are the usual institutional fragmentation and centralization. A prominent example of these transformations is the case of the Great Lakes in the USA regarding water quality and water supply as key dimensions to be governed (Jetoo *et al.,* 2015).

With the accent on privatization and decentralization, the new approaches towards management and conservation emphasized self-governance and higher levels of participation for civil society and private enterprises (Baud *et al.,* 2011; de Castro *et al.,* 2016).

Neoliberal policies guided the privatization of natural resources such as water (Molinos-Senate *et al.,* 2015) and forests (Manuschevich, 2016) as in the case of Chile, and fish as in the case of the USA (Pinkerton & Davis, 2015; Carothers, 2015), along with land grabbing as in Argentina for example (Coscieme *et al.,* 2016), producing major socio-environmental impacts (Liverman & Villas, 2006). In parallel, coalitions among civil society organizations, (international) NGOs and academic institutions established an alternative governance perspective for local communities, which was labeled participatory governance **Figure 4.4**. This new trend cemented the way for 'glocalization' processes linking local and global actors to develop local conservation approaches (Baud *et al.,* 2011).

By and large, the main governance arrangement towards nature conservation has been the centralized establishment of public protected areas (encompassing different levels of protection from total preservation to multiple uses). Comprising Mesoamerica, South America and the Caribbean the coverage of protected areas has increased by 8.9% with respect to the subregions' total area between 2000 and 2014, being the territory with the largest increase in area under protection worldwide (World Bank, 2017). The same three subregions show an increase between 2000 and 2014 of 5.2% of the total territorial waters protected with respect to the regions' total area. Conservation policy and implementation often assume that protected areas are enduring institutions, but some recent evidence suggests widespread protected areas downgrading, downsizing, and degazettement (Mascia *et al.,* 2014). Mascia *et al.* (2014) describe protected areas downgrading, downsizing, and

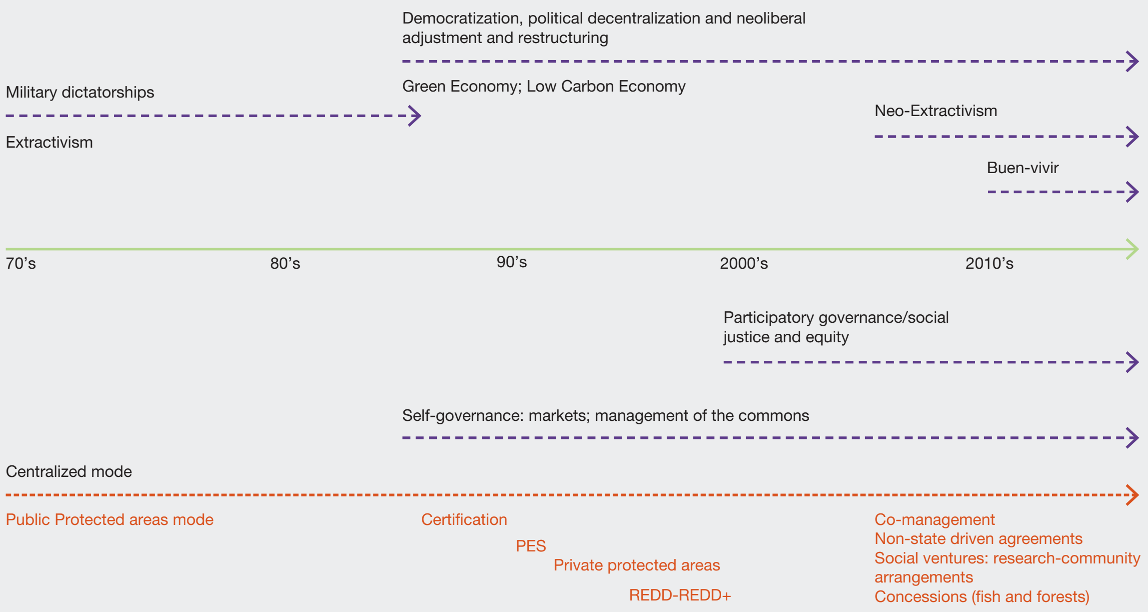


Figure 4 4 **Evolution of environmental governance modes in the Americas. Source: Synthesis by the authors based on Baud *et al.* (2011) and de Castro *et al.* (2016).**

degazettement as a "patchy, episodic phenomenon" which nonetheless suggests tradeoffs between conservation goals and other policy objectives and is linked to industrial-scale natural resource extraction and development, local land pressures and land claims, and conservation planning.

Another circumstance is that in several cases the creation of protected areas has displaced local communities (Cardozo, 2011; Jones *et al.,* 2017). For three case studies in Mexico, for example, García-Frapolli *et al.* (2009) identifies the most common difficulties in protected areas policy as: (1) uncoordinated public policies; (2) the usual conflict between environmental authorities and local people over the management of natural resources; and (3) the exclusion of local people's perspectives, values and beliefs in conservation policy development and implementation.

Aside from command and control arrangements such as protected areas, several hybrid modes have emerged in the region **Figure 4.4**. Among them the most notorious are: state private partnerships (certification), private-social partnerships (e.g. payment for ecosystem services), and co-management. forest certification is prominent in Brazil, Chile and Argentina (see Pinto and Mcdermontt, 2013; Cubbage *et al.,* 2010). Another iconic example is the certification of coffee in countries such as Colombia, Brasil, Costa Rica, Ecuador, and Honduras, among others (Pinto *et al.,* 2014; Ibañez and Blackman *et al.,* 2016). Certification has recently expanded to industrial and smallscale fisheries with promising results in several countries of the region (Perez-Ramírez *et al.,* 2016) **(Box 4.4)**.

Despite the increasing enthusiasm for ecosystem services based market mechanisms, the reality is that incentive allocation on private lands has relied on scarce knowledge of ecosystem service supply by different properties (Ferraro *et al.,* 2015). In the absence of supply data at the farm level for the entire region, the measurement of policy impacts has had to rely on imperfect proxies for additionality in terms of service provision (e.g. avoided deforestation) (Ferraro *et al.,* 2015). Undeniably, the lack of complete, high-resolution, updated spatial information to obtain ecosystem services indicators is a primary restriction on the development of conservation planning assessments in developing countries (Di Minin & Toivonen, 2015; Stephenson *et al.,* 2017) including the design of payment for ecosystem services mechanisms. In the domain of payments, Reduced Emissions from Deforestation and Forest Degradation and Reduced Emissions from Deforestation and Forest Degradation-Plus have emerged as a core climate change mitigation strategy. Nonetheless the mechanism has been harshly contested due to its undesirable social impacts and undetermined role in avoiding deforestation (Pirard & Belna, 2012).

The commitment by most countries to expand the area under protection in a representative and well-connected manner, as part of the Convention on Biological Diversity's (CBD) Aichi target 11, requires the inclusion of a range of protection mechanisms over a variety of tenures, including protected areas over private land (Woodley *et al.,* 2012). Despite their potentially important role in biodiversity conservation, recognition of the role of private protected areas has suffered from sparse data, loose definitions and lack of integration within the broader conservation arena (Stolton *et al.,* 2014) (see details in **Box 4.5**). The main challenges of private protected areas are the absence of recording and as a consequence there is no reliable

Box ④ ④ The promise of fisheries certification.

Ecolabelling and certification schemes are market-based tools to promote the sustainable use of natural resources. In the case of fisheries, ecolabels are a growing feature of international fish trade and marketing (Washington & Ababouch, 2011) in response to growing concerns about the state of the world's fish stocks, increased demand for fish and seafood, and a perception that many governments are failing to manage marine resources. The Marine Stewardship Council features as the most comprehensive fisheries certification scheme covering a range of species and dealing with all aspects of the management of a fishery. The Marine Stewardship Council has two standards: on sustainable fishing and on seafood traceability (Bush *et al.,* 2013; Agnew *et al.,* 2013; Washington & Ababouch, 2011). Although there are 10 Marine Stewardship Council-certified fisheries in Latin American and the Caribbean, this proportion is low (4%) compared to the total number of certified fisheries globally (Pérez Ramírez *et al.,* 2016). Fisheries participating in the Marine Stewardship Council program in the region may be classified into two groups: (1) large enterprises of industrial fisheries, especially multi-national ones that can afford the certification process (i.e., Argentine hoki); and (2) small-scale fisheries that are vital to the local livelihoods (i.e. lobsters). Among the latter a successful case is the Chilean rock lobster *(Jasus frontalis)* of The Juan Fernández Archipelago and Robinson Crusoe Island, Marine Stewardship Council certified in 2015. The success of fishery management over recent years relies on five key management measures that are implemented with the full support of the community (near 900 inhabitants): only licensed artisanal fishers who are residents may harvest lobster in the area; the use of relatively small vessels that can only tend a few traps per day; informal property rights on individual fishing grounds; a conservative minimum landing size (115 mm length); and a closed season of four and a half months.

information on how many there are, where they are located, what conservation activities they are engaged in. With private protected areas there is also an absence of clear guidelines for establishment and operation, and there are differences in the support and incentives given by government to the creation and maintenance of private protected areas (Bingham *et al.,* 2017). They also face the challenge of avoiding conflicts with local and indigenous communities, particularly those located on the private protected areas' buffer zones (Serenari *et al.,* 2017).

At the local level, there has been an emergence of community-based participatory conservation approaches seeking to engage local communities in management decisions, transfer rights to resources and allow sustainable use, to varying degrees. Many countries have introduced new policies and laws to support community-based conservation and there have been some successes **(Box 4.6)**. However, in most cases, community-based conservation remains small-scale and isolated and is weakly integrated within the formal conservation sector (Baud *et al.,* 2011; Lammers *et al.,* 2017; Redmore *et al.,* 2017) facing barriers such as a limited binding participation of communities in the development of conservation policies; insufficient devolution of authority and benefits to communities; and lack of support from other natural resource and economic sectors (Baud *et al.,* 2011).

On the opposite side of the green economy and the previous set of governance arrangements, new proposals arise that contemplate a fundamentally different ontology of nature, grouped under the label of *Buen Vivir* (Vanhulst & Bieling, 2014; Villalba-Eguiluz & Etxano, 2017) **Figure 4.4**. This trend includes a wide range of alternative conceptions of nature and of human-nature relations, starting with alternative, often indigenous, ideas about the relationship

Box 4 5 The challenges of private protected areas.

The declaration of private protected areas involves "a private intention to protect an area where government and other organizations do not play a pivotal role" (Stolton *et al.,* 2014). The motivations behind their creation vary widely from pure philanthropic motives to real state and tourism development and speculation. The following are examples for different countries of the region (Stolton *et al.,* 2014):

- USA. There is no formal private protected areas definition and no comprehensive reporting, but there is an active private protected areas community driven by land trust organizations and NGOs, with many thousand private protected areas.
- Canada. Private protected areas are primarily located on the country's southern border on land with high levels of species diversity and also species at risk.
- Mexico. Private protected areas, which protect 487,300 hectares (0.25%) of the country's land surface, play an important role in connecting government managed protected areas.
- Colombia. There are 280 registered national private protected areas organizations, most are small in area and many are in the Andes.
- Chile. The term private protected areas is legally recognized, although undefined and unregulated. The private protected areas vary widely in size (from a few hectares to over 300,000 hectares) and ownership (comprising private individuals; industrial forest companies; NGOs; and foundations). They represent over 10% to the national protected area system.
- Brazil. Brazil has a legislated and federated system of over 1,100 private reserves of natural heritage protecting approximately 703,700 ha.

Box 4 6 Los pueblos del bosque.

The socio-ecological struggles of traditional populations are what Martínez-Alier calls the "environmentalism of the poor" (Martínez-Alier, 2014). Within the multiple manifestations of this "ecology of the poor" in South America, Mesoamerica and the Caribbean, one of the first to have had an international echo was the movement of rubber tappers (*seringueiros*) who are not indigenous peoples but the first or second impoverished immigrants from northeastern Brazil, left in search of their own forms of subsistence long after the commercial exploitation of rubber on a large scale was over.

Acre rubber tappers formed unions, and in 1987 they joined the indigenous inhabitants of the Amazon to form an Alliance of Forest Peoples led by Francisco "Chico" Mendes who paid with his life for the cause of the Amazonian peoples (Tijoux, 2016). This movement was the forerunner of multiple expressions in the present as the Yasuní Park Project in Ecuador, which is considered one of the most important actions of the indigenous movements of the Americas. At present many of these actions are channeled through formal coalitions such as the Mesoamerican Alliance of Peoples and Forests http://www.alianzamesoamericana.org, among others.

between human production, the environment and the rights of nature (Gudynas, 2011; Bauhardt, 2014). They propose a perspective of environmental governance that claims a transformation or even the end of the hegemonic capitalist model that is considered as the source of environmental degradation and injustice (de Castro *et al.,* 2016; Inoue & Moreira, 2016).

These varied modes of governance do not necessarily coexist peacefully in the region and in many cases are antagonistic rather than synergistic, leading to severe social conflicts, which pose serious challenges for nature conservation and human well-being. Next to aspiration and creativity, attaining new modes to govern nature requires overcoming persistent barriers such as historical injustices, social inequalities and economic inefficiencies (Baud *et al.,* 2011).

Major challenges have been reported in the past and continue to be significant limitations in the present. Among them: i) the environment continues to be a low priority (e.g. underfunded environmental agencies; low political support); ii) the understanding of environment-poverty-development links is frail (e.g. environmental concerns are perceived as barrier to economic growth); iii) the rule of law is weak (e.g. implementation of environmental legislation is still insufficient); and iv) environmental authority is weak (e.g. taking a management view rather than a governance focus). A critical issue pointed out at several international conservation forums is the fact that the three pillars of sustainable development – environmental, economic, and social – are not well integrated in the United Nations system and in global, regional, and national policies. Lessons learned in the past 25 years since the Earth Summit have led civil society organizations to uphold human rights as the basis for sustainable development governance.

4.3.2 Economic growth

Economic growth (measured as Gross Domestic Product (GDP) growth) is one of the main drivers of resource consumption (Dietz *et al.,* 2007, quoted by IPBES, 2016). Virtually all socioeconomic and environmental scenarios for this century (i.e., up to the year 2050 and beyond) include economic growth as a key driver (IPBES, 2016).

Economic growth and trade can positively or negatively influence nature and NCP, but currently, on balance, they adversely impact nature and NCP when environmental and social development goals are insufficiently accounted for. Positive impacts of economic growth include, for instance, the resulting income availability for social and environmental investments, like biodiversity protection and conservation (Tlayie & Aryal, 2013), and greater environmental awareness. Negative impacts of economic growth mainly refer to the adverse consequences (e.g. habitat degradation, overharvesting, etc.) of those styles of economic growth that disregard social development and environmental goals.

Assessing relevant information on economic development includes consideration of key indicators, like regional and subregional GDP (and GDP per capita) growth trends; regional and subregional distribution of GDP purchasing power parity (PPP); as well as the sectoral structure of national economies (agriculture, industry, services). **Table 4.1** synthetizes historical (since 1960) and projected (until 2050) trends for GDP and population in the Americas. GDP and population increased by 5.9 and 2.4 times, respectively, in the Americas from 1960 to 2016. By 2050, GDP in the Americas is expected to double with respect to 2016, while population would increase by 20% in that period.

Economic growth has been identified as a key driver of global greenhouse gasses emissions (IPCC, 2014a). With around 5% of world population, North America produces 24.2% of global GDP[1] (16.8% of global GDP_{PPP}) and 16% of global greenhouse gasses emissions, while Latin America and Caribbean accounts for 8.7% of total population, 7.6% of world GDP[2] (8.1% of global GDP_{PPP}), and 5.2% of global greenhouse gasses emissions (**Table 4.1**, IEA, 2016).

The impact of the consumers' purchasing power on the demand of natural resources is receiving growing attention in the economic literature nowadays due to the emergence of new waves of affluent consumers who tend to increase the demand for the limited natural resources (Myers & Kent, 2003). Purchasing power parity dollars are between 1.5 and 2.6 times higher than conventional dollars in at least 27 developing countries of the Americas. For the USA, PPP dollars and conventional dollars are the same by definition.

The countries of the region with the largest economies overall are the USA, Brazil, Canada, and Mexico. Dominica, Grenada, and Antigua and Barbuda, all small States in the Caribbean, have the region's smallest economies overall. Factoring in countries' populations, the countries with the largest per capita incomes in the region are the USA and Canada. At around $50,000, their per capita incomes are considerably higher than all other countries in the region. The other countries' per capita incomes vary between Haiti's low of about $728 to The Bahamas $19,758. In general, per capita incomes are lowest in the Mesoamerica subregion, though other subregions exhibit a fair degree of variation (World Bank, 2017[3]).

The economies of the Americas vary widely in the sectoral composition of their national output. The contribution of agricultural production to national output has fallen to less

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1. Based on constant 2010 USA Dollars (see **Table 4.1**)
2. Based on constant 2010 USA Dollars (see **Table 4.1**)
3. Data available at http://databank.worldbank.org/data/reports.aspx

Table ④ ① **Gross Domestic Product (GDP) and population in the Americas: historical and projected trends. Sources: Based on The World Bank Database (2017). https://data.worldbank.org/indicator/; Worldometers (2017). Accessed 2 May 2017, and 3 September 2017 at: http://www.worldometers.info/world-population/population-by-region/; Foure *et al.* (2012).**

REGIONS	GDP PPP (*)	GDP (**)			POPULATION		
	% of world GDP, 2016	% of world GDP, 2016	Cumulative change, 1960-2016 (GDP_{2016}/GDP_{1960})	Expected cumulative change, 2016-2050 (GDP_{2050}/GDP_{2016})	% of world population, 2017	Cumulative change, 1960-2017 (Pop_{2017}/Pop_{1960})	Expected cumulative change, 2017-2050 (Pop^{2050}/Pop_{2017})
North America	16.8	24.2	5.51	1.71	4.8	1.77	1.19
Mesoamerica	2.3	1.9	8.86	3.19	2.4	3.44	1.29
Caribbean	0.4	0.3	7.56	3.71	0.6	2.11	1.09
South America	5.4	5.4	7.05	3.16	5.7	2.86	1.18
AMERICAS	**24.9**	**31.8**	**5.86**	**1.98**	**13.5**	**2.37**	**1.20**

Notes:

(*) GDP at purchasing power parity (PPP)
(**) Constant 2010 USA Dollars
Reference data: World GDP_{PPP} in 2016: $120.1 trillions; World GDP at Constant 2010 USA Dollar: $75.5 trillions; World population 2017: 7,515.1 millions

than 20% throughout the region with the exception of Haiti where agriculture's GDP share is 21.5%. The economies of the region are primarily service driven, although there is variation across the individual national economies between Paraguay's 51.2% to Barbados' 85.5%. Throughout the region, the countries with the higher per capita incomes are those whose economic output is driven more heavily by service sector activity. Industrial production is a significant driver of most of the economies of the region, ranging from contributing less than 15% of GDP in Barbados and Grenada, to more than 40% of Trinidad and Tobago's GDP. Most economies in the region derive 25% to 35% of their GDP from industrial production (World Bank, 2017[4]).

The Americas has experienced substantial economic growth since 1960. Although the worldwide recession of 2008-2009 temporarily reduced national incomes, GDP in the Americas has increased approximately six fold since 1960, although North American income grew from a substantially higher 1960 level. Despite increasing populations throughout the region, the pace of real GDP growth has been sufficient to raise per capita GDP more than twofold from 1960 to 2015 (World Bank, 2017[5]).

While overall growth has been sizable at the regional and subregional levels, individual countries within the Americas have experienced varying growth trends since 1960. Per capita incomes have increased substantially over time in some countries; in other countries, per capita incomes have increased more modestly, or in still other countries, barely at all. In North America, Canada and the USA each experienced large growth in per capita income from already high 1960 levels. In Mesoamerica, GDP per capita grew significantly in Panama, Costa Rica, and Mexico, while it increased much more slowly in other countries. In the Caribbean subregion, The Bahamas has consistently had significantly higher per capita income than the rest of the subregion, followed by Trinidad and Tobago, Barbados, St. Kitts and Nevis, and Antigua and Barbuda. Incomes in a handful of the subregion's countries barely grew at all. In South America, per capita GDP shows varying growth by country. Venezuela's was higher on average (partly due to oil endowments). In 1960, per capita incomes in the subregion (excluding Venezuela) ranged from about $1,000 in Paraguay to about $5,600 in Argentina. By 2015, the range had widened considerably, from about $2,400 in Bolivia to almost $15,000 in Chile. Countries with strong growth since about 1990 include Chile, Uruguay, Brazil, Argentina, and Suriname. Peru's growth has been steady but slower, with a recent acceleration (World Bank, 2017[6]).

The GDP growth rate for the USA fell from an average of 3.3% per year in 1997-2006 to 1.2% in 2007-2015; while the economic dynamics for Latin America and the Caribbean also diminished from an average of 3.1% to 2.9% in those years.

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[4] Data available at http://databank.worldbank.org/data/reports.aspx

[5] Data available at http://data.worldbank.org/indicator/

[6] Data available at http://data.worldbank.org/indicator/

These trends partially reflect the interconnections between the USA market and Latin America and the Caribbean economies, particularly those of Mesoamerica and the Caribbean. The period 2007-2015 was characterized by the effects of the global economic crisis, with absolute reductions of GDP for the USA in 2008 (by -0.3%) and in 2009 (by -2.8%), and for Latin America and the Caribbean region in 2009 (by -1.2%) and 2015 (by -0.3%)[7].

Growing pressures on natural resources are expressed in different ways in different country groupings and regions, due to patterns indicating high per capita consumption of natural resources, growing dependency on commodities exports and other conditions **(Table 4.2)**. Per capita consumption of natural resources is particularly high in North America. For instance, total primary energy consumption per capita for North America was 6.1 tons oil equivalent versus 2.39 tons oil equivalent for non-OECD Americas in 2013 (IEA, 2015; Pichs, 2008). According to WWF (2014) the nitrogen loss indicator[8] is largest in North America (81kg/capita/year), more than twice the world average (29 kg/capita/year).

Commodities (including, for instance, hydrocarbons, mineral raw materials, food and other agricultural products) represent more than 50% of Latin America and the Caribbean exports (for the years 2012-2014) and 9% of the regional GDP, reflecting a clear extractivist bias in the regional economic growth. South America is the most commodity-intensive subregion in the Americas, with commodities accounting for more than 70% of goods exports, and nearly 10% of GDP. Mesoamerica is considerably less commodity dependent than South America, but commodities still account for about one quarter of exports there, and 7.5% of GDP (World Bank, 2016). North American economies are more diversified than Latin America and the Caribbean economies and consequently less vulnerable to price shocks in the global commodity markets. The export diversification index[9] for North America in 2015 was 0.213, while this indicator averaged 0.584 for the Caribbean, 0.549 for South America, and 0.375 in Mesoamerica (UNCTAD, 2016).

Rapid economic growth generates growing pressures on nature and NCP, particularly when the economic growth is heavily dependent on increasing use of natural resources and carbon intensity. Economic crisis also increases pressures on natural resources when economic agents tend to compensate low commodity prices with higher export volumes.

In recent decades, the increase in household income in Latin America and the Caribbean has resulted in a striking rise in consumption. Per capita private consumption for Latin America and the Caribbean, in USA dollars at constant 2010 prices, rose by a cumulative annual rate of 2% between 1990 and 2000 and 2.5% between 2000 and 2016, while the corresponding rates for North America were 2.5% and 1.1%, respectively in those periods. Since 2010, the average per capita private consumption in Latin America and the Caribbean have surpassed the world average, but by 2016 it was only 17.1% of the corresponding level for North America **(Table 4. 3)**.

Within Latin America, consumption trends have followed differentiated patterns across the various subregions in the last three decades. The expansion of private consumption in South America, for instance, has been supported, to

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[7] Based on IMF (2014 & 2015).

[8] The nitrogen loss indicator was developed for the CBD and represents the potential nitrogen pollution from all sources within a country or region as a result of the production and consumption of food and the use of energy (WWF, 2014).

[9] The export diversification index is calculated by measuring the absolute deviation of the export structure of a country from world structure. This index takes values between 0 and 1. A value closer to 1 indicates greater divergence from the world pattern (UNCTAD, 2016).

Table 4 2 Combining GDP growth and GDP intensity in natural resources (including energy / carbon intensity) and assessing the level of pressure on biodiversity and ecosystem services

Note: GDP intensity in natural resources refers to the consumption of natural resources required to produce a unit of GDP, with fossil fuel intensity (measured as volume of oil equivalent for monetary unit of GDP), for instance, being a subset of GDP intensity in natural resources. Source: Elaborated by the authors based on ECLAC (2014); CEPAL (2017); IMF (2017); The World Bank Database (2017); UNCTAD (2016); WWF (2014, 2016); GFN (2017).

Low GDP growth / High GDP intensity in natural resources High pressures, due to situations like economic stagnation (e.g. Extractivist policies with economic crisis)	**Low GDP growth / High GDP intensity in natural resources** Very high pressures, through very high GHG emissions (reinforcing climate change), land use change (deforestation) and general overexploitation of natural resources (e.g. Extractivist policies with economic expansion).
Low GDP growth / High GDP intensity in natural resources Low pressures associated, for instance, to low technological development.	**Low GDP growth / High GDP intensity in natural resources** Low pressures due to de-coupling between GDP growth and GDP intensity.

Table 4 3 **Household final consumption expenditure per capita. Source: The World Bank Database (2017) World Development Indicators. https://data.worldbank.org/indicator/NE.CON.PRVT.PC.KD?view=chart. Accessed 4 November 2017**

REGIONS	CONSTANT 2010 USA DOLLARS			AVERAGE ANNUAL % GROWTH	
	1990	2000	2016	1990-2000	2000-2016
North America	22,675	28,703	34,841	2.5	1.1
Latin America and the Caribbean	3,675	4,488	5,958	2.0	2.5
WORLD	**4,036**	**4,710**	**5,833**	**1.6**	**1.4**

a large extent, by the boom in exports of renewable and non-renewable natural resources, with highly favorable terms of trade up to 2014. In Central America, however, the consumption dynamics have been more closely associated with the stabilization of remittances, while Mexico combines both patterns: exports of natural resources (mainly oil) and significant flows of remittances (ECLAC, 2014). The prevailing consumption model in Latin America and the Caribbean is still what Fernando Fajnzylber termed *"showcase modernization"*, which may expand the population's access to goods and services but also tends to replicate the socio-environmentally unsustainable conditions seen in the developed countries (ECLAC, 2014).

On the one hand, private consumption dynamics in Latin America and the Caribbean during the recent decades has brought positive effects, as it has been partially associated with increased well-being in sectors that were deprived in the past, and it has contributed to better living standards, which in turn enable better use of time and more opportunities for capacity-building. On the other hand, growing private consumption has also brought negative consequences and externalities such as higher fossil fuel consumption, waste generation, air pollution, environmental destruction and increased exploitation of renewable and non-renewable natural resources. In addition to that, consumption in Latin America and the Caribbean is pro-cyclical and exposes economies to greater vulnerability. Recent regional consumption trends have also widened the gap between consumers of private and public services (ECLAC, 2014).

Another source of concern is that the upper income segments of the population in Latin America and the Caribbean, favoured by wealth concentration, tend to show a pattern of consumption very intensive in high-cost private services and luxury goods, with a high imported content. The region's highest income quintile spends between four and 12 times more than the lowest income quintile (ECLAC, 2014).

Scenarios that assume rapid economic growth in the coming decades are mainly based on prioritizing market goals and incentives under conventional market approaches, with adverse social and environmental implications, including negative impacts on biodiversity and ecosystems (e.g. Global Environmental Outlook 4 Market First (IPBES, 2016)).

Statistics on the composition of the ecological footprint for the Americas reveal that the carbon footprint accounts for 53% of the total ecological footprint of the Western Hemisphere (65% for North America). The second largest hemispheric contributor is cropland, which accounts for 19% (26% in South America), and the third position is shared by grazing land and forest products (12% each). The predominant role of the carbon footprint in the Americas is mainly associated with the high dependency on fossil fuels in the region **Table 4.4**.

The list of the top five countries with the highest ecological footprint includes two countries from the Americas, the USA (accounting for 13.7% of world total ecological footprint) and Brazil (with 3.7%) (WWF, 2014).

4.3.3 International trade and finances

Economic activities, international trade and financial flows are closely related, particularly in recent decades due to the expansion of economic globalization. Trends in economic growth, international trade and financial markets considerably influence changes nitrogen, NCP and good quality of life through various direct and indirect pathways. In turn, these pathways are influenced by a number of policy channels and mechanisms, like trade policies, including incentives (tax exemptions, subsidies) and trade barriers, the dynamics of foreign debt and foreign debt service, flows of foreign direct investments, and monetary policies (dynamic of exchange rates, interest rates).

Table 4 4 **Composition of the ecological footprint the regions of the Americas (%). Source: Based on WWF (2014, 2016), GFN (2017) (See Chapter 2, section 2.6).**

Notes: Ecological footprint data for 2013. Composition in % of the ecological footprint for 2010. 1. Information for Belize is not available, 2. Information available only for five countries: Cuba, Dominican Republic, Haiti, Jamaica and Trinidad and Tobago, 3. Information for Guyana and Suriname is not available.

REGIONS	Cropland	Grazing land	Forest products	Fishing Grounds	Building-up land	Carbon	Total ecological footprint
North America	16	5	11	2	1	65	100
Mesoamerica (1)	22	12	12	2	2	50	100
Caribbean (2)	25	10	7	4	2	52	100
South America (3)	26	30	16	1	4	23	100
Latin America & Caribbean	25	25	14	1	4	31	100
AMERICAS	**19**	**12**	**12**	**2**	**2**	**53**	**100**

The Americas generates around 18% of world exports, and most of this proportion (12.6%) is supplied by North America. Latin America and the Caribbean contribution to world exports (5.4%) is modest in relation to the region´s fraction of world population (8.7%).

The volumes of trade are directly related to economic size and openness. The USA has the highest trade volumes, with a substantial trade deficit. Canada and Mexico are in the next tier with respect to volumes, followed by Brazil. The composition of trade reflects countries' economic activity and natural resources. Fuel ranges between 10% and 23% of imports for all countries in the region except Costa Rica and fuel exporting countries. Over 50% of all countries' goods imports are manufactured goods. Manufactured goods form over ¾ of all imported goods for 11 of the countries with data. On the export side, agricultural raw material forms a very small part of each nation's trade. It is most important for Uruguay, comprising 12.7% of its merchandise exports. Fuel comprises over half of Venezuela's, Colombia's and Bolivia's exports and plays an important role in exports from Ecuador and Canada. Manufactured goods form an important component of most of the region's nations' exports, being most important for Mexico and El Salvador. Tourism is by far the most important export for The Bahamas, and is also important to other Caribbean nations (World Bank, 2017[10]).

As mentioned before, natural resources (oil, minerals, and agricultural products) contribute with more than 50% to Latin America and the Caribbean exports. Commodities account for more than 70% of exports in South America, and about one quarter of exports in Mesoamerica (World Bank, 2016). Tourism is also a key sector in several Latin America and the Caribbean countries, particularly for small Caribbean island States and some Central American countries. Drastic reduction of commodities prices in world markets since 2014 has severely affected commodities exporters in the region. In some cases, Latin America and the Caribbean countries have tried to compensate declining export prices of commodities with increasing export volumes, generating additional pressure on the natural environment. International export prices for Latin America (19 countries reported by ECLAC) declined by 8.7% in 2015 in relation to 2010, while export volume increased by 15.4% (CEPAL, 2015). As indicated before, the export structure of North America is more diversified, and therefore these developed economies are less vulnerable to market shocks, in relation to the Latin America and the Caribbean economies.

In contrast to North American economies, most Latin America and the Caribbean countries have very limited influence in world trade and financial markets and flows, with high vulnerability to abrupt changes in those markets **(Table 4.5)**.

The **Table 4.6** presents the potential pressures on nature and NCP due to the dynamics of trade and financial trends. In South America, for instance, export policies and currency exchange rates (Richards *et al.,* 2012) have created incentives to buy land for planting soybean, and this explains the high deforestation rate in ecosystems like the South American Chaco. This has generated not only high export revenues but also the devastation of nature as well as increasing poverty and social conflicts (Barbarán, 2015; Barbarán *et al.,* 2015; Weinhold *et al.,* 2013).

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[10] Data available at http://data.worldbank.org/indicator/

Table 4 5 **Relevant trade data for the Americas (2016). Source: The World Bank (2017). World Development Indicators (Last Updated Date: 08.02.2017): www.worldbank.org**

Country/Region	Number of economies	% of world exports of goods and services	Exports of goods and services as % of GDP
North America	2	12.6	14.0
Mesoamerica	8	2.2	37.0
Caribbean	13	0.4	22.3
South America	12	2.8	16.5
Latin America & Caribbean	32	5.4	21.7
AMERICAS	**34**	**18.0**	**15.6**

Table 4 6 **Potential pressures on biodiversity and ecosystem services due to the dynamics of trade and financial trends. Source: Elaborated by the authors. Based on ECLAC (2014), CEPAL (2017), IMF (2017), The World Bank Database (2017), UNCTAD (2016), WWF (2014, 2016).**

Note: Cases (1 and 2) correspond to each indicator of the first column (horizontal analysis).

Trade & Finance Indicators	Case 1	Case 2
Prices for relevant export products based on natural resources (including carbon intensive exports).	**High prices:** Potential pressures on biodiversity due to the incentive of having high export prices. New exporters can emerge.	**Low prices:** Potential pressures due to attempts to compensate losses in export prices with increasing export physical volumes.
Trade Policies for trading products based on natural resources.	**Restrictive policies (e.g. protectionist measures / trade barriers):** Potential pressures on biodiversity in the importing countries as non-efficient producers may be competitive. Growing pressures on biodiversity in exporting countries, due to efforts to find alternative export solutions with limited options.	**Non-restrictive polices (e.g. trade liberalization):** Significant pressures on biodiversity when these measures are not carried out in a sustainable development context, as they may encourage a massive flow of trade.
Foreign Debt (in proportion to key indicators like GDP and/or export. income).	**High levels:** Significant pressures on biodiversity in debtor countries, as they struggle for get additional income to serve the foreign debt, with one option being increasing export of products / services based on natural resources.	**Low levels:** Low pressure on biodiversity.
Foreign Direct Investments (particularly in sectors based on natural resources).	**Growing flows:** Significant pressures on biodiversity in the recipient country, particularly in absence of well- established local foreign direct investments laws to ensure sustainable use of natural resources.	**Declining flows:** Pressures on biodiversity would depend on local investment options as alternative to foreign direct investments.
Monetary Policies.	**E.g. Local currency devaluation:** This encourages exports, by making them more competitive. This could imply additional pressures on biodiversity.	**E.g. Local currency revaluation:** This makes exports less competitive. This could imply pressures on biodiversity in exporting countries, due to efforts to find alternative export solutions with limited options.

The cumulative foreign debt for Latin America and the Caribbean countries reached $2,062 billion in 2016, with a per capita foreign debt for the region of $3,250. Total cumulative payments of foreign debt service (interests and amortization) increased to $3,461 billion during 2008-2016. The regional payments to cover the foreign debt service accounted for 51.4% of Latin America and the Caribbean export income (including goods and services) in 2016 (based on IMF, 2014, 2015, 2016, 2017). South America absorbs 70% of regional Latin America and the Caribbean foreign debt (corresponding 22% to Brazil); Mesoamerica, 27% (with Mexico absorbing 21%); and the Caribbean, 3% (based on CEPAL, 2016).

Foreign debt for North America reached around $20.6 trillion in 2016 / early 2017 (corresponding 89% of this amount to the USA[11]). Approximately 80% of USA foreign debt is denominated in USA dollars. Foreign lenders have been willing to hold USA dollar denominated debt instruments because they perceive the dollar as the world's reserve currency. With the USA dollar being the national currency of the USA, this makes a significant qualitative difference between the foreign debt status of North America with regard to other regions of the Americas.

The flow of foreign direct investments to the Latin America and the Caribbean region totaled $134.8 billion in 2015 (8% below the average flow for the period 2011- 2014). This trend has been influenced to a great extent by the declining tendency of prices for commodities exported by the region. South America hosted 73% of foreign direct investments flows to Latin America and the Caribbean in 2015 (only Brazil, 46%); Mesoamerica, 24% (only Mexico 16%) and the Caribbean, 3% (based on CEPAL, 2016). Foreign direct investments inflows to North America reached $428.5 billion in 2015 (only USA 89%) (UNCTAD, 2016).

4.3.4 Technological development

Human development has been historically related to technological change, with historical epochs named after the key technologies: the Stone, Bronze and Iron Ages, the industrial revolution, the age of steam, and the information age. The way of orienting the development, dissemination, and use of technology is crucial to find just, equitable, and sustainable solutions for present and future generations. Political, social, cultural, and economic factors determine the way new technologies are developed and used (Trace, 2016).

The rate of technological change is considered as an indirect driver of changes in nature, NCP and good quality of life because it affects the efficiency by which ecosystem services are produced or used (Alcamo *et al.,* 2005, quoted by IPBES, 2016). The impact of technological innovation on biodiversity and ecosystem change is exerted through its influence on direct drivers (e.g. land use change), as well as through interactions and synergies with other indirect drivers (e.g. economic growth, see 4.3.2).

Finding indicators of the status and trends in the Americas region's or any given country's technological development is difficult due to data shortcomings. The Americas, with 13.6% of world population (2013 data) accounted for 22.5% of the total amount of researchers, 33.1% of world investments in research and development, 34.8% of world publications and 53.2% of patents submitted to the US Patent and Trademark Office. Regional information reveals the persisting gaps regarding science, technology and innovation in the Americas **Table 4.7**.

Most of the scientific and technological potential of the Americas corresponds to North America, with 18.8% of researchers, 29.6% of global research and development, 29.6% of world publications, and 52.9% of patents submitted

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11. http://ticdata.treasury.gov/Publish/debta2017q1.html; http://www.indexmundi.com/united_states/debt_external.html; http://www.statcan.gc.ca/tables-tableaux/sum-som/l01/cst01/indi01j-eng.htm.

Table ④ ⑦ **Selected science and technology indicators in the Americas (2013)[1]. Source: UNESCO (2016).**

Countries	% of world population, 2013	% of world R&D, 2013	Per capita R&D (USD), 2013	R&D/ GDP, 2013 (%)	Researchers / thousand inhabitants, 2013	% total researchers, 2013	% of global increase in R&D 2007-2013	% of world publications, 2014	% of total patents submitted to USPTO, 2013[4]
USA	4.3	28.1	1249.3	2.81	4.0	16.7	10.8	25.3	50.1 [5]
Canada	0.5	1.5	612.0	1.63	4.5	2.1	[2]	4.3	2.8
Latina America	8.1	3.4	87.2	0.69	0.5	3.6	4.2	5.1	0.3
Caribbean	0.6	0.1	40.8	0.34	0.2	0.1	0.0 [3]	0.1	0.0
WORLD	**100**	**100**	**206.3**	**1.70**	**1.1**	**100**	**100**	**100**	**100**

Notes:
[1]. This information does not separate non-military and military research and development (R&D).
[2]. Canadian investments in R&D reduced from $23.3 billion in 2007 to $21.5 billion in 2013.
[3]. Caribbean investments in R&D marginally increased from $1.6 billion in 2007 to $1.7 billion in 2013.
[4]. UPSTO: United States Patent and Trademark Office.
[5]. This is used as an international indicator considering the attractiveness of the USA market also for foreign investors.

to US Patent and Trademark Office. Latin America and the Caribbean only account for 3.7% of researchers, 3.5% of global research and development, 5.2% of publications and 0.3% of US Patent and Trademark Office patents. The USA accounted for 10.8% of the global increase of research and development during 2007-2013, while the contribution to that increase from Latin America and the Caribbean hardly reached 4.2% **Table 4.7**, UNESCO, 2016).

The availability of secure internet servers in the Americas has increased rapidly since the early 2000s. The North American subregion significantly outpaces the Latin America and the Caribbean subregion, however. In North America, there are currently almost 1,600 servers per million people, while in Latin America and the Caribbean there are only 59 per million people. Individual countries within subregions also exhibit wide variation in both the current number and increase in the number of secure internet servers per million people (World Bank, 2017[12]).

Technological innovation can catalyze paradigm shifts in production systems (Pérez, 2004, quoted by IPBES, 2016) that cause biodiversity loss and adverse ecosystem changes (i.e. technologies as part of the problem), or conversely reduce biodiversity loss and improve ecosystems health (technologies as part of the solution).

Technology offers important positive solutions to resource conservation, sustainable use and development, and management, but technological change can also increase pressure on ecosystem services through increasing resource demand and leading to unforeseen ecological risks, particularly for technologies associated with agriculture and other land uses (e.g. first generation of biofuels when produced unsustainably).

As part of the solution space, technological change can increase agriculture efficiency and replace unsustainable production patterns (e.g. improvements in crop yields and resilience, sustainable livestock, fishing, and aquaculture practices). Although technology can significantly increase the availability of some ecosystem services, and improve the efficiency of provision, management, and allocation of different ecosystem services, it cannot serve as a substitute for all ecosystem services (Carpenter *et al.,* 2006, quoted by IPBES, 2016).

In some cases, technological developments and agricultural practices may combine positive and negative implications for biodiversity and ecosystems as revealed by the agricultural intensification of the "green revolution". On the one hand the "green revolution" led to higher crop yields and lower food prices, partially mitigating the expansion of agricultural land and resulting in a net decrease of greenhouse gasses emissions. On the other hand, excessive nitrogen and phosphorous use through fertilizers, associated with the "green revolution" led to substantial degradation of freshwater and marine habitats. In addition, the shift from traditional crop varieties to industrial monocultures resulted in a loss of crop genetic diversity as well as increased susceptibility to disease and pests (IPBES, 2016, chapter 3). This confirms the importance of promoting sustainable practices with an integrative approach concerning the linkages between environment and socioeconomic development.

Those production technologies and practices that are based on increasing dependence on external inputs like chemical fertilizers, pesticides, herbicides and water for crop production and artificial feeds, supplements and antibiotics for livestock and aquaculture production have adverse implications in terms of sustainability. These technologies damage the environment, undermine the nutritional and health value of foods, lead to reduced function of essential ecosystem services and result in the loss of biodiversity (FAO, 2011, quoted by Trace, 2016).

When the technological changes in agriculture are implemented in accordance with the principles of sustainable development, these transformations may imply greater equity within and between generations, including with regard to food security (FAO, 1996).

Agroecological food production systems are considered as one approach to addressing the loss of biodiversity and the consequent unsustainability of industrialized food production, because they recognize the interdependencies between the sources of food and the wider environment, and the overlapping needs to provide sustainable food systems and sustainable livelihoods (Trace, 2016). Local knowledge and culture can be considered as integral parts of agricultural biodiversity (FAO, 2004, quoted by Trace, 2016). Agroecology considers productive processes in a broad and integral manner, taking into account the complexity of local forms of production. It is based on sustainability criteria, resource conservation and social equity (Vos *et al.,* 2015).

The misappropriation of traditional biodiversity knowledge or 'biopiracy' has been considered as one of the most 'complex problems facing the future of traditional knowledge' (Khor, 2002, quoted by Trace, 2016). The system of community sharing and collaborative innovation is being challenged by intellectual property rights and the trade-related aspects of intellectual property rights regime, which together create a new system to exert private ownership rights over knowledge (Trace, 2016).

The intersection between agriculture, trade, and intellectual property governance is marked by a diversity of institutions involved, including the World Trade

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[12] Data available at http://data.worldbank.org/indicator/

Organization, the World Intellectual Property Organization, the CBD, and the Food and Agriculture Organisation. On balance, the corporations have the upper hand in this complicated game (Sell, 2009).

A combination of expanded intellectual property rights and relaxed antitrust enforcement facilitated a recent shift from public to private provision of seeds, which is undermining small farmers' tradition of saving seeds and reusing seeds. In this and other ways, the current situation is marked by underinvestment in crops and technologies suitable for smallholder farmers. In agri-biotechnology, six companies alone hold 75% of all USA patents granted to the top thirty patent-holding firms (Dutfield, 2003; Fowler, 1994). The top ten seed companies control over half of the global seed market (ETC Group, 2008) and are contributing to monoculture and associated loss of biodiversity in Latin America. This institutional dominance of transnational corporation facilitates "gene grab" (Sell, 2009), with negative effects on biodiversity, competition, and food security to the extent that it prevents resource sharing and locks out potential user-innovators by preventing small farmers from breeding, saving and reusing seeds to feed themselves and their communities (Rajotte, 2012). This is especially consequential considering that small farmers provide the majority of the food consumed by national populations. In Brazil, small farmers occupy 30% of agricultural land yet produce 70% of the food consumed by Brazilians.

4.3.5 Population and demographic trends

Assessing human demographic trends and their implications for nature, NCP and good quality of life includes consideration of total population and age structure; urban vs. rural populations and urban forms; information on locations, like coastal versus inland, migration flows, among other indicators **Table 4.8** and present data on population and demographic trends in the Americas for the period 1960-2017 and expected future trends to 2050.

The Americas accounted for 13.5% of the world's estimated population in 2017. Subregionally, while having nearly equal areas[13], North America accounts for 4.8% of world population, while Latin America and the Caribbean accounts for nearly twice that at 8.7% of world population. This is reflected in population density, with Latin America and the Caribbean being much more densely settled (32 people per km^2) than Northern America (20 people per km^2). The population of the Americas is highly urbanized, with 80.8% of the region's population residing in urban settings (82.8% for North America, and 79.7% for Latin America and the Caribbean) **Table 4.8** (Index Mundi, 2017).

Urbanization, driven by growing populations and internal migration, acts as an indirect driver of land-use change through linear infrastructures like transportation networks, as well as through synergies with other forms of infrastructure development (Seiler, 2001, quoted by IPBES, 2016, see also section 4.4.1). In Latin America and Caribbean 35% of the population (year-basis 2015) gained access to sanitation since 1990, but still 12% of the urban population and 36% of rural population do not have access to improved sanitation facilities (UN-Habitat, 2016). On average, only 50% of the population in Latin America is connected to sewerage and 30% of those households receive any treatment. The poor systematic waste management in Latin America and the Caribbean implies in pollution of inland waters and coastal areas (4.4.2), affecting biodiversity and human health.

Current population growth rates are 0.75% per year in North America and 1.02% per year in Latin America and the Caribbean. Migration and fertility rates combine differently in these two subregions. In Latin America, an above-replacement fertility rate of 2.15 outweighs net outmigration from the subregion, such that population growth is positive and relatively high compared to the world community there. In the North American subregion, net in-migration outweighs lower-than-replacement fertility rate to produce that

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[13] Area data are not corrected for inhabitable spaces.

Table 4 8 **Population in the Americas by region in 2017. Source: Authors' compilation from Worldometers (2017). Accessed 2 May 2017, and 3 September 2017 at http://www.worldometers.info/world-population/population-by-region/.**

Regions	Population 2017	Yearly Change, %	Migrants (net)	Median Age	Fertility Rate	Density (P/km²)
North America	363,224,006	0.75	1,219,564	38.4	1.86	20
Mesoamerica	177,249,493	1.28	-192,495	26.9	2.34	72
Caribbean	43,767,545	0.64	-120,068	30.5	2.27	194
South America	426,548,298	0.95	-63,786	30.6	2.03	24
AMERICAS	**1,010,789,342**					

subregion's positive population growth rate. North America has among the world's oldest median population, while Latin America and the Caribbean has among the world's youngest median population.

The USA, Brazil, and Mexico are by far the most populous countries of the region. Population densities vary widely throughout the region, as do population growth rates.

Population growth rates throughout the region have generally fallen substantially since 1960. This is less true for the Caribbean subregion as a whole. Several countries' annual population growth rates have been more volatile than their subregion's overall trend: Greenland in North America, Belize in Mesoamerica, Grenada and Antigua & Barbuda in the Caribbean, and Guyana and Suriname in South America (World Bank, 2017[14]).

Population trends have an important role in explaining changes in natural resources and biodiversity (**Table 4.9** and **Table 4.10**). Population growth has been identified as a key driver of global greenhouse gasses emissions (IPCC, 2014a). However, the analysis of population growth, as an indirect driver of changes in nature and NCP needs to be completed by including the consumption patterns and life-styles considerations (Pichs, 2008, 2012).

The global middle class is expected to grow from 1.8 billion in 2009 to 4.9 billion by 2030. Much of this will occur in developing countries (including Latin America and the Caribbean) where 70% of global economic activity will emerge by 2050. With this trend comes increasing demand for energy, infrastructure, and consumer goods (Runde and Magpile, 2014; Myers & Kent, 2003).

Population growth projections for the Americas range from around 10% (in the Caribbean) to near 30% (in Mesoamerica) between the years 2017 and 2050. At the same time, GDP projections range from 3.1 to 3.7 times in the developing regions of the Americas (around 70% in North America) in relation to the 2017 levels by 2050. Consequently, core baseline scenarios regarding the consumption of natural resources and energy in the Americas would be mainly driven by GDP growth, and population growth, as relevant drivers (Ruijven *et al.,* 2016).

4.3.6 Human development

Analysis of the various dimensions of human development is critical for assessing the wide range of indirect drivers for changes in nature and NCP. Several social indicators and aggregated indexes may be useful for achieving that assessment purpose, including the Human Development Index (HDI) that can provide information on the share of population in extreme poverty, income distribution (e.g. Gini coefficient), educational attainment (e.g. access, literacy level), health (e.g. access to public health, health care infrastructure, expectancy of life), social expenditure / GDP (e.g. education, health), and food security (e.g. number and % of hungry people) (see Chapter 2, section 2.6).

Social inequity is still a concern for the various subregions of the Americas, with adverse implications for nature, NCP and good quality of life. On the one hand, poor people in the Americas often increase the demand

Table ④ ⑨ **Population in the Americas by region: present (2017), past (1960-2017) and future (2017-2050) trends. Source: Based on Worldometers (2017). Accessed May 2, 2017, and September 3, 2017 at http://www.worldometers.info/world-population/population-by-region/.**

Countries	**Region's share of world pop, 2017**	**Region's share of Americas pop, 2017**	**Total pop change, 1960- 2017 (Pop_{2017} / Pop_{1960})**	**Total pop change, 2017-2050 (Pop_{2050} / Pop_{2017})**	**Urban, % of total pop 2017**	**Urban pop, change, 1960-2017 ($UrbPop_{2017}$ / $UrbPop_{1960}$)**	**Urban pop, change, 2017-2050 ($UrbPop_{2050}$ / $UrbPop_{2017}$)**
North America	4.8	35.9	1.77	1.19	82.8	2.11	1.30
Mesoamerica	2.4	17.5	3.44	1.29	74	5.44	1.43
Caribbean	0.6	4.3	2.11	1.10	71.2	3.78	1.23
South America	5.7	42.2	2.86	1.19	83	4.64	1.27
AMERICAS	**13.5**	**100**	**2.38**	**1.20**	**80.8**	**3.25**	**1.30**

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[14] Data available at http://data.worldbank.org/indicator/

Table 4 10 Combining population growth with per capita consumption of natural resources and assessing the level of pressures on biodiversity and ecosystem services.

Source: Elaborated by the authors based on ECLAC (2014), CEPAL (2017), IMF (2017), UNDP (2016), The World Bank Database (2017), UNCTAD (2016), Worldometers (2017), WWF (2014, 2016); GFN (2017).

Low Population Growth / High per Capita Consumption of Natural Resources High pressures on BD resources mainly due to high per capita ecological footprint. This is a typical pattern of several industrialized countries. Critical role of international trade.	**High Population Growth / High per Capita Consumption of Natural Resources** Very high pressures on BD, due to the combined effect of increasing population / density and growing per capita ecological footprint. Critical role of international trade, and adverse implications in terms of high GHG emissions, land use changes (deforestation) and general overexploitation of natural resources.
High Population Growth / High per Capita Consumption of Natural Resources Low pressures on BD due to low population and population density, as well as low per capita ecological footprint.	**High Population Growth / High per Capita Consumption of Natural Resources** Low pressures on BD mainly due to survival reasons of growing population. Typical pattern of least developed countries and poor communities.

pressures on nature merely to survive. On the other hand, high per capita consumption by affluent segments of the population also increases pressure on natural resources in. This discussion is very relevant in the context of the global debate on multidimensional progress (PNUD, 2016) and the SDG, particularly for key areas of social development like poverty and hunger eradication, as well as access to education, health, safe water and sustainable energy.

In 2015, Mesoamerica showed the lowest regional HDI in the Western Hemisphere, which was below the average levels for Latin America and the Caribbean countries (0.7310), the Americas (0.7418), and the world (0.7170). Haiti had the lowest country-specific HDI in the Americas (0.4930), even below the corresponding level for Sub-Saharan Africa (0.5230). Inequality Adjusted HDI was considerably lower than HDI in the Americas (by 21%), in Latin America and the Caribbean countries (by 22%) and in North America (by 11.1%) **(Table 4.11)**.

Country-specific HDI values and trends indicate that most countries of the Americas rank as "very high" or "high" human development within the world community. However, four Mesoamerican and three South American countries have HDI values that rate their human development as "medium" within the world community, while Haiti's HDI falls very low in the world rankings (UNDP, 2016).

Table 4 11 HDI and inequality adjusted HDI in the Americas (*), 2015.

Regions	No. of countries	HDI 2015	No. of countries	Inequality Adjusted HDI, 2015 (IA-HDI)	IA-HDI / HDI (change in %)
North America	2	0.9200	2	0.8175	-11.1
South America	12	0.7438	12	0.5854	-21.3
Caribbean	13	0.7365	5 (**)	0.5502	-20.5
Mesoamerica	8	0.7028	8	0.5345	-23.9
Americas	35	0.7418	27	0.5810	-21.0
Latin America and the Caribbean	33	0.7310	25	0.5621	-22.0
WORLD	**188**	**0.7170**	**151**	**0.5570**	**-22.3**

Notes:

(*) The HDI is a statistic constructed by combining a range of indicators thought to capture human potential and development: per capita income, education, and life expectancy. The inequality-adjusted HDI statistically adjusts the HDI to account for income inequality, in order to reflect the potential for human development in the absence of inequality. Higher HDI and inequality-adjusted -HDI scores indicate better conditions in these areas combined; that is, greater human well-being and potential for human well-being, respectively.

(**) Trinidad and Tobago, Jamaica, Saint Lucia, Dominican Republic and Haiti. Sources: Based on UNDP (2016).

Average HDI values for all regions of the Americas improved from 2010 to 2015, representing widespread regional gains in incomes, education, and socioeconomic factors that increase life expectancy. Despite those overall improvements, HDI scores for 18 countries in the region dropped in the worldwide rankings between 2010 and 2015, indicating a failure to match gains in human development at a more international level. Of these 18 countries, half are in the Caribbean subregion.

Cuba (with 48 points) and Barbados (20 points) lead the list of countries of the Western Hemisphere where the "gross national income ranks minus HDI rank" shows positive results, indicating that their human development achievements go far beyond those derived from their gross national income. These results may be associated, for instance, with more efficient allocation of economic resources to social goals like education and health (UNDP, 2016).

Income inequality is high in the Americas overall. Most countries in the region have a degree of income inequality (reflected in low international ranks in terms of equality and high Gini coefficients) that ranks among the world's 50 most unequal nations. This is particularly true of countries in the Mesoamerican and South American subregions (Index Mundi, 2017[15]). The ratio of inequality-adjusted -HDI/HDI shows that inequality is constraining the region's societies from realizing their human development potential **(Table 4.11)**.

The prevalence of extreme poverty in the Americas has decreased considerably since 1981. The World Bank data show that the portion of the population of Latin America and the Caribbean living below the international "income poverty" line of $1.90 per day fell from 23.9% in 1981 to 5.6% in 2012, and that living below the international "working poor" poverty line of $3.10 per day fell from 38.0% to 12.0% over the same period (World Bank, 2017[16]).

Nevertheless, poverty in the Latin America and the Caribbean region remains a concern. First, the proportion of the population facing extreme poverty varies considerably throughout Latin America and the Caribbean at the country level. More than a quarter of the populations of El Salvador and Honduras live on less than $3.10 per day. Second, extreme income poverty in the Latin America and the Caribbean region, even at reduced levels, affects millions of people, including many children (World Bank, 201717). Third, 38% of the Latin America and the Caribbean region's population is socioeconomically vulnerable due to a persistent inability to enter the middle class (PNUD, 2016). Fourth, the recent worldwide economic slowdown exacerbates this susceptibility.

The percentage populations living in poverty in 2012 was approximately 26.9% in Latin America, 40.6% in Mesoamerica, and 21% in South America (CEPAL, 2014). Around 72 million people exited the condition of income-poverty during 2003-2013 in Latin America; however, 25-30 million people are at risk of falling into that condition again as a result of economic vulnerability and social fragility (PNUD, 2016).

Poverty not only affects the developing countries in the Americas. The percentage of poor people recently reached 13.9% in the USA population (43.1 million people)[18]; and those living in households below statistics Canada's low income threshold represented 9.7% in 2013; incidence of low income tended to be higher among children, seniors, and persons in single-parent families (Lammam & MacIntyre, 2016).

Historically, the needs and priorities of indigenous peoples in the Americas have been largely ignored, mainly affecting indigenous women. This situation has started to change in recent past. By 2010, about 45 million indigenous people (8.3% of the regional population) lived in Latin America, compared with an estimated 30 million in 2000, an increase that is partially a result of population growth but also from the greater visibility of this population in the national censuses. On average, without distinguishing educational levels, the labor income of non-indigenous and Afro-descendant men quadrupled those of indigenous women and almost doubled those of Afro-descendant women. Between 2009 and 2013, around 235 conflicts were identified in Latin America, which were generated by projects of extractive industries (mining and hydrocarbons) in indigenous territories (CEPAL, 2016).

The population of American indians and Alaska natives in the USA, including those of more than one race, comprised approximately 2.0% of the total population (6.6 millions) in 2015[19]. Data from the National Household Survey in Canada show that 1,400,685 people had an Aboriginal identity in 2011, representing 4.3% of the total population[20].

Another set of broader societal factors deserving special consideration when dealing with the implications of social development on biodiversity and ecosystem services include worldviews and culture (attitudes to environment/

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15. Available at https://www.indexmundi.com/facts/indicators/SI.POV.GINI/rankings/central-america

16. Available at: Povcal Net, Online Database – http://go.worldbank.org

17. Available at http://povertydata.worldbank.org/poverty/region/LAC

18. According to data from the Center for American Progress (2017). Available at https://www.census.gov/content/dam/Census/library/publications/2016/demo/p60-256.pdf, quoted by https://talkpoverty.org

19. Vintage 2015 Population Estimates: http://nativenewsonline.net/currents/u-s-census-bureau-native-american-statistics/

20. http://www12.statcan.gc.ca/nhs-enm/2011/as-sa/99-011-x/99-011-x2011001-eng.cfm

sustainability/equity), life-styles (including diets), and societal tensions and conflict levels.

Culture in the form of the values, norms, and beliefs of a group of people can act as an indirect driver of ecosystem change by affecting environmentally relevant attitudes and behaviours (IPBES, 2016).

Biodiversity and linguistic diversity are threatened globally. They are declining at different rates in different regions, with the most rapid losses in linguistic diversity occurring in the Americas, which is in parallel to biodiversity loss (Maffi, 2005; Harmon & Loh, 2010; Gorenflo *et al.,* 2012).

In this context, indigenous and local communities' traditional knowledge provides a comprehensive reflection of prevailing conditions and other key inputs and incorporates methods and approaches that capture holistic values that people place on nature, while internalizing principles and ethical values specific to their world views and realities (Illescas and Riqch'arina, 2007; Medina, 2014, quoted by IPBES, 2016).

Traditional ecological knowledge can be found all over the world, particularly within indigenous traditions across diverse geographical regions from the Artic to the Amazon, and represents various understandings of ecological relationships, spirituality, and traditional systems of resource management (Alexander *et al.,* 2011). In recent decades, resource managers have gradually begun to embrace the usefulness of applying that knowledge to contemporary stewardship issues in various parts of the world.

Indigenous peoples in multiple geographical contexts, including the Americas, have been pushed into marginalized territories that are more sensitive to environmental challenges, in turn limiting their access to food, cultural resources, traditional livelihoods and place-based knowledge. All this disrupts their ability to respond to environmental changes and undermines aspects of their socio–cultural resilience (Ford *et al.,* 2016) **(Box 4.7)**.

The broad ways in which indigenous knowledge and experiences are framed mirror common portrayals of indigenous peoples as "victim–heroes"; "victims" through the framing that indigenous peoples are highly vulnerable and "heroes" through the framing that indigenous peoples possess knowledge that can help address the problem (Ford *et al.,* 2016). The complexity and diversity of indigenous experiences and their understanding and responses to environmental challenges are not well captured in many of the cases where indigenous content is documented by peer review literature.

Some studies identify the ongoing effects of colonialism, marginalization, power relations, land dispossession and land rights to be central to understanding the human dimensions of global environmental change for indigenous peoples in diverse contexts (Ford *et al.,* 2016).

4.4 DIRECT ANTHROPOGENIC DRIVERS

4.4.1 Habitat degradation and restoration

Nature of the driver, its recent status and trends, and what influences its intensity

Habitat degradation includes land conversion and intensification of croplands and rangelands; wetland drainage and conversion; construction of roads, dams, pipelines, and transmission lines; sprawl; pollution, and resource extraction. Physical alterations of freshwater habitats also include change in hydrological regime (flow regime and water withdrawals). Marine environment degradation is increasing in some areas with increased shipping and bottom trawling, coastal construction (ports, marinas, housing and other development, and pollution with various forms of sediment and chemical discharges. Aquaculture (farming of marine flora and fauna) also can contribute to habitat degradation (for ponds, access and infrastructure; for feed: fishing to produce fish meal, hormone and antibiotic additives; discharges in the form of fecal pollution, etc.). Pollution as a driver of change will be discussed in the section 4.4.2.

Habitat loss and degradation are considered the greatest threats to biodiversity (Wilcove *et al.,* 1998, Sala *et al.,* 2000, Hanski *et al.,* 2013, Murphy & Romanuk, 2014; Haddad *et al.,* 2015; Newbold *et al.,* 2015). Worldwide, nearly half of tropical dry forests, temperate broadleaf forests, and temperate grasslands, savannas, and shrublands have been converted to human uses (Hoekstra *et al.,* 2005). Land use change affects biodiversity and ecosystems not only by reducing population sizes and movements, but also by reducing habitat area, increasing habitat isolation, and increasing habitat edge (Haddad *et al.,* 2015). Reducing area or increasing isolation decreases both species persistence and species richness (Haddad *et al.,* 2015).

Forests covered 1.6 billion hectares of land in the Americas, which is approximately 41% of its land area and 40% of worldwide forest area (FAO, 2013a). This forest includes 722 million hectares of relatively undisturbed old-growth forest, 57 million hectares of planted forest, and 818 million

Box ④ ⑦ Indigenous and local knowledge and values: Implications for natural resources management.

The Americas are populated by many indigenous nations, from the Artic to Patagonia, with a variety of cultures and languages that have developed many different socio-economic systems (nationally and locally). Increasing numbers of historically marginalized groups are joining transnational networks and alliances that promote indigenous mobilization and demand recognition and rights from their respective nation-states and the international community. These rights include protection of and control over their property and possessions (like territories, resources, material culture, genetic material, and sacred sites), practices (cultural performances, arts, and literature), and knowledge (cultural, linguistic, environmental, medical, and agricultural). By linking issues of representation, recognition, resources, and rights, these movements engage and often challenge current theories of culture, power, and difference in sociocultural anthropology (Hodgson, 2002). Indigenous and local knowledge are expressions of social capital and may act as a driver of biodiversity and ecosystem services supply because of direct influences on land use change (direct influences), as well as its ability to modify the influence of other drivers (interactive influences). Some cases illustrating the role of ILK as drivers of land use change in the Americas, hence on biodiversity and ecosystem services, are presented below:

1. The Isobore Sécure National park and indigenous territory case in Bolivia (McNeish, 2013). In August 2011, 2000 marchers left the city of Trinidad, the lowland regional capital of the department of Beni, to follow a route that would take them 66 days and 600 kilometers of walking to the capital city of La Paz. The central demand of the protest march was founded on the cessation of a road-building project planned to go through the Isobore Sécure National Park and Indigenous Territory. Following a series of meetings between the protesters and the president, the government agreed to pass a legal decree on 24 October 2011 guaranteeing that the road would not pass through the Isobore Sécure National Park and Indigenous Territory. Furthermore, the law stated that the Isobore Sécure National Park and Indigenous Territory would be protected by the state as an 'intangible' territory, effectively making the territory out of bounds for all forms of future state or development projects.

2. Shrimp farming versus mangroves in coastal Ecuador (Veuthey & Gerber, 2012). Over the last two decades, the global production of farm-raised shrimps has increased at a faster rate than any other aquacultural product, leading to massive socio-ecological damages in the mangrove areas where shrimp farming often takes place. Consequently, an increasing number of conflicts pitting coastal populations against shrimp farmers have been reported; although, very few conflicts have been studied in detail. According to the authors, the development of shrimp farming can be understood as a modern case of enclosure movement whereby customary community mangroves are privatized for the building of shrimp ponds. As a result, local mangrove-dependent populations – especially women – mobilized and protested against a form of ecologically unequal exchange. While only some mangroves could be saved or reforested as a result of the movement, women's mobilization has had the unexpected effect of challenging gender relations in their communities.

3. Oil frontiers and indigenous resistance in the Peruvian Amazon (Orta-Martínez & Finer, 2010). The Peruvian Amazon is culturally and biologically one of the most diverse regions on Earth. Since the 1920s oil exploration and extraction in the region have threatened both biodiversity and indigenous peoples, particularly those living in voluntary isolation. Modern patterns of production and consumption and high oil prices are forcing a new oil exploratory boom in the Peruvian Amazon. While conflicts spread on indigenous territories, new forms of resistance appear and indigenous political organizations are born and become more powerful.

4. Indigenous land and deforestation control in Amazon (Nepstad *et al.,* 2006). Indigenous lands occupy one-fifth of the Brazilian Amazon. Analyses of satellite-based maps of land cover and fire occurrence in the Brazilian Amazon compared the performance of large (>10,000 ha) un-inhabited (parks) and inhabited (indigenous lands, extractive reserves, and national forests) reserves. Reserves significantly reduced both deforestation and fire. There was no significant difference in the inhibition of deforestation or fire between parks and indigenous lands, but uninhabited reserves tended to be located away from areas of high deforestation and burning rates. In contrast, indigenous lands were often created in response to frontier expansion, and many prevented complete deforestation despite high rates of deforestation along their boundaries.

hectares of forest that regenerated after human disturbance. From 1990 to 2015, forest area expanded in North America by nearly three million hectares and the Caribbean by more than two million hectares but declined in Central America by nearly seven million hectares and in South America by more than 88 million hectares (Keenan *et al.,* 2015). Approximately 34% of forest area is protected in South America (where the percentage of protected forest area doubled from 1990-2005) and less than 9% of forest area is protected in North America (in accordance with the IUCN definition, excluding categories V and VI) (Morales-Hidalgo *et al.,* 2015). Brazil has a much higher proportion of its forest protected (41.8%, 206 million hectares) than any other country and the USA has protected the second greatest forest area (33 million hectares, 10.6% of forests; Morales-Hidalgo *et al.,* 2015).

Conversion to croplands and pasturelands is the main driver of terrestrial habitat change in the region. In 2013, agriculture covered 1.23 billion hectares of land in the Americas, which is approximately 32% of its land area and 25% of the worldwide agricultural land (FAO, 2013a). This agriculture included 828 million hectares of permanent meadows or pastures and rangelands used for livestock grazing (68%), 28 million hectares of permanent crops, and 370 million hectares of arable land (~2%), which includes land covered by temporary crops, pasture, or hay meadows (~30%). Conversion patterns differ among subregions. Most land conversion in Mesoamerica and North America occurred more than one century ago, whereas in South America most occurred within the last century. Since 1961, the area of agricultural land has increased by 13% across the Americas, which is the net result of a 40% increase in South America, a 29% increase in the Caribbean, an 11% increase in Central America, and a 9% decrease in North America. From 2001 to 2013, 17% of new cropland and 57% of new pastureland replaced forests throughout Latin America (Aide *et al.,* 2013). Cropland expansion from 2001 to 2013 was less (44.27 million hectares) than pastureland (96.9 million hectares), but 44% of cropland in 2013 was new, versus 27% of pastureland, revealing row crop expansion. Most cropland expansion was into pastureland within agricultural regions of Argentina, Brazil, Bolivia, Paraguay, and Uruguay (Graesser *et al.,* 2015, Volante *et al.,* 2015). Commodity crop expansion, for both global and domestic urban markets, follows multiple land change pathways entailing direct and indirect deforestation, and has various social and environmental impacts (Meyfroidt *et al.,* 2014, see Chapter 2, section 2.2.1).

Agricultural practices associated with land conversion significantly change biogeochemical cycles contributing to pollution of terrestrial and aquatic ecosystems and to climate change (sections 4.4.2 and 4.4.3). Each year, land conversion results in emissions of approximately one billion metric tonnes of carbon (1 Pg C per year), which is 10% of emissions from all human activities (Friedlingstein *et al.,* 2010). Soil carbon losses also diminish crop yields and degrade water quality. Nitrogen fertilization also contributes to climate change by emitting the greenhouse gas of nitrous oxide (Compton *et al.,* 2011; Sutton *et al.,* 2011; Keeler *et al.,* 2016). In the Americas, approximately 23 million tonnes of nitrogen fertilizer and 22 million tonnes of phosphorus (phosphate + potash) were consumed in 2013; and about 52 million hectares of land were under irrigation. Increasing anthropogenic nitrogen inputs are also likely driving loss of diversity (Bobbink *et al.,* 2010) and polluting freshwater supplies (section 4.4.2). Nutrient imbalances due to agriculture are related to depletion or accumulation depending on the balance between inputs and outputs of nutrients. Nitrogen depletion occurred in the southern parts of South America (e.g. Argentina), the Amazon region, Central America, and some parts of the Midwest of the USA, partially attributable to the high crop yields (Liu *et al.,* 2010). Soil nitrogen depletion occurs regardless of how high the nitrogen input once crop nitrogen uptake, along with other nitrogen losses, exceeds the inputs (Liu *et al.,* 2010).

Croplands also affect migratory species through habitat degradation and pesticide use along their migratory routes (e.g. neotropical migratory birds like dicksisels, bobolinks, and Swainson's hawks) (Basili & Temple, 1999; Hooper *et al.,* 2002; Lopez-Lanus *et al.,* 2007). Habitat conversion leads to not only many native species losses, but also to gains in some exotic species (section 4.4.4). Exotic species are often introduced for particular human uses and are not necessarily functionally equivalent to the native species they displace (Wardle *et al.,* 2011).

Urbanization can also directly and indirectly threaten biodiversity and services from surrounding ecosystems. In 2016, while the degree of urbanization worldwide was around 54%, it was around 80% in the Americas. In Latin America and the Caribbean, the urbanization rate has declined over the past six decades (UN, 2014). Cities in Latin America exhibit extreme social and economic differences, which generate a complex mosaic of urban settlement structures and ecosystem management systems. In addition, conservation of ecosystems and biodiversity, and ecosystem services provisioning, are not prioritized in urban planning (Pauchard & Barbosa, 2013). Direct impacts include land occupation by buildings and roads. Indirect impacts result from the provisioning of services to urban populations, like food, building materials, energy, water, and other resources. This requires infrastructure such as dams, pipelines, transmission lines, and roads, timber harvesting, and land cover conversion for grazing and cropping. (e.g. McDonald *et al.,* 2014; Bhattacharya *et al.,* 2012). Roads help deliver benefits from where they are supplied to where they are demanded and consumed. However, they also threaten biodiversity (Laurance *et al.,* 2014) by fragmenting habitat and facilitating resource extraction activities like cropping; grazing; timber harvesting and extraction of water, minerals, oil, and gas. For example, over the last 60 years, there have been at least 238 notable oil spills along mangrove shorelines worldwide. In total, at least 5.5 million tonnes of oil has been released into mangrove-lined, coastal waters, oiling possibly up to around 1.94 million hectares of mangrove habitat and killing at least 126,000 hectares of mangrove vegetation since 1958 (Duke, 2016). Mangroves and other coastal "blue carbon" ecosystems also have high ecosystem carbon stocks and are undergoing significant conversion at a great cost in terms of greenhouse gas emissions, as well losses of other important ecosystem services (Kauffman *et al.,* 2016).

Despite declines in the density of species, cities can have unique assemblages of plants and animals and retain some endemic native species, thus providing opportunities for regional and global biodiversity conservation, restoration and education (Aronson *et al.,* 2014). Habitat conversion has also resulted in increases in food, mineral, timber, and energy production. For example, global cereal production has more than doubled since 1960 (Tilman *et al.,* 2002; Wik *et al.,* 2008). Few studies have weighed such benefits against the costs of habitat degradation described above. In some cases, however, the financial costs of habitat conversion for non-provisioning ecosystem services, like carbon storage and sequestration, can outweigh the benefits of conversion for supply of provisioning services (Nelson *et al.,* 2009).

The intensity of land degradation depends on indirect drivers (section 4.3), like governance (zoning, incentive policies, management policies), social development (education, technology), economic development (markets, trade, technology, land tenure, corporate pressures), and interactions among land degradation and other direct drivers, including climate change and changes in fire regimes. With economic development, human diets have shifted toward more meat and dairy consumption (Foley *et al.,* 2011, Tilman *et al.,* 2011). Continuing this trend in coming decades would require further pasture expansion, intensification of livestock production, or both. Maintaining or increasing future food, energy and water production without compromising biodiversity and ecosystem services can involve multiple strategies, including land sharing and land sparing (Fisher *et al.,* 2014); closing yield gaps on underperforming lands (Mueller *et al.,* 2012); improving efficiency of agricultural input application, reducing food waste (Foley *et al.,* 2011) and changing diets (Tilman *et al.,* 2011, Tilman & Clark, 2014; Vranken *et al.,* 2014).

After abandonment from human uses, some habitats gradually recover while others fail to do so (Benayas *et al.,* 2009; Jones & Schmitz, 2009; Barral *et al.,* 2015). Over the past 15 years, total global pasture area decreased by 2%, with much of that land likely abandoned, rather than converted to other agriculture (Poore, 2016). There is substantial potential for biomass recovery of Neotropical secondary forests, with most forests recovering 90% of biomass in less than a century (Poorter *et al.,* 2016). Based on well documented evidence of the negative impacts of deforestation on surface water quality (Baker *et al.,* 2004; Scanlon *et al.,* 2007) it is possible that the reverse of deforestation will improve stream water quality in freshwater systems, especially with active forest restoration.

Even with active ecosystem restoration, however, it is rarely possible to fully restore lost biodiversity and ecosystem services (Benayas *et al.,* 2009). Habitat restoration often significantly increases biodiversity and ecosystem services above levels observed in degraded ecosystems, but levels of biodiversity and ecosystem services in restored ecosystems often remain significantly lower than levels in reference remnant ecosystems. Compared with reference ecosytems, recovering ecosystems exhibit annual deficits of 46–51% for organism abundance, 27–33% for species diversity, 32–42% for carbon cycling and 31–41% for nitrogen cycling (Moreno-Mateos *et al.,* 2017). Although degradation of ecosystems is ongoing, there is also a significant increase in conservation and restoration efforts in the Americas (Wortley *et al.,* 2013; Echeverría et a. 2015). Some examples of restoration of terrestrial and freshwater ecosystems are presented in **Box 4.8** and **Box 4.9**.

Box 4 8 Examples of restoration initiatives in the Americas – Great Lakes.

The five Laurentian Great Lakes – Superior, Huron, Michigan, Erie and Ontario – comprise 20% of the world's available freshwater supply. The Great Lakes cover an area of about 246 million km^2. The draining basin extends from roughly 41 to 51° N, and from 75 to 93° W, and includes parts of eight USA states and two Canadian provinces. Human activity has had deleterious impacts on the Great Lakes ecosystem. The logging boom of the late 1800s altered the basin's hydrologic regime. Shipping traffic introduced non-native species and untreated waste discharge of nutrients and other chemical pollutants led to a virtual ecological collapse in the mid-1900s (Rankin, 2002).

Since 2009, the Great Lakes have been the focus of a major restoration initiative by the USA government (expenditures of greater than $1 billion over five yers), targeting invasive species, nonpoint run-off, chemical pollution, and habitat alteration. The current initiative specifically targets key classes of environmental stressors that were identified through a planning process involving numerous government agencies and environmental groups (Allan *et al.,* 2013). For example, Great Lakes Restoration Initiative resources have been used to double the acreage enrolled in agricultural conservation programs in watersheds where phosphorus runoff contributes to harmful algal blooms in western Lake Erie, Saginaw Bay and Green Bay (https://www.glri.us).

The Great Lakes sand dunes constitute the most extensive freshwater dunes in the world, covering over 1,000 km^2 in Michigan alone (Albert, 2000). In the region, traditional dune restoration efforts involving monoculture plantings of *A. breviligulata* (American beach grass) restore many measures of diversity and ecosystem function over the past 20-30 years (Emery & Rudgers, 2009). Plant and insect diversity, vegetation structure (plant biomass and cover), and ecological processes (soil nutrients and mycorrhizal fungi abundance) in restored sites were similar to reference sites. Differences were mostly attributed to the relative age of the sites, where the younger sites supported slightly lower plant diversity and mycorrhizal spore abundance than older sites (Emery & Rudgers, 2009).

Box 4 9 Examples of restoration initiatives in the Americas – Tropical forests and pastures.

The presence of degraded areas, many of them already abandoned, in almost all types of land use, generate further degradation and impacts on natural remnants, like effects on pollinators through uncontrolled application of pesticides. The persistence of these practices will lead to the emergence of additional degraded areas. Two different and coordinated actions could be considered in order to provide potential solutions for these environmental problems: 1) actions to avoid, stop, minimize or reverse the ongoing environmental degradation (e.g. fire management, erosion control, reduction of pesticide use, among others) which could be generically called sustainable management practice, and 2) specific actions for the recovery of already degraded areas, that is, restoration. Productive and environmental landscape optimization, in addition to the actions forementioned, is also intended to change land-use economic practices, locally increasing productivity, thereby reducing pressures to use areas that have more value for conservation. Effective actions have been taken in many regions of the world that correspond to sustainable management practices (FAO *et al.,* 2011; FAO, 2011 and 2013b), rehabilitation (Buckingham & Hanson, 2015) and restoration of degraded areas (Nellemann & Corcoran, 2010; Goosem & Tucker, 2013; Hanson *et al.,* 2015).

In the Americas there are already important examples of the successful implementation of sustainable management practices (e.g. ITTO, 2011; Calle et al*.*, 2012; Calle & Murgueitio, 2015; FAO, 2013b), rehabilitation (e.g. Brancalion *et al.,* 2012), and restoration (e.g. Calvo-Alvarado *et al.,* 2009; Rodrigues *et al.,* 2009, 2011; Murcia & Guariguata, 2014; Hanson *et al.,* 2015). Restoring distinct vegetation types that have very different levels of resilience, species richness and complexity of interactions and are inside landscapes with different degrees of fragmentation have demanded different methods. Although the degree of success achieved for each one varies between vegetation types and socioeconomic conditions considered, there are already examples in Brazil where restoration in large-scale and high-biodiversity tropical forests have been achieved (Rodrigues *et al.,* 2011) and whose principles could be adapted to other vegetation types and countries. An example is the intensive silvopastoral systems, which have been implemented in Colombia (Calle *et al.,* 2012). Livestock grazing, a common practice in the Americas and around the world, results in soil compaction, soil erosion, reduction of water infiltration, and silting of springs and streams. This degraded land condition can maintain very few animals and produces less income. Grazing also favors continuous land abandonment and migration, inducing deforestation to create new pastures. Converting extensive pastures to intensive silvopastoral systems allowed for, in 4-5 years, increases in production, productivity, and rural incomes and jobs, as well as the elimination of all sources of degradation. This change resulted in increases of environmental services and rural biodiversity and allowed for the release of farm margins to be used for forest restoration or rehabilitation.

North America

Oil and gas development in Alaska and Canada has focused on tundra in North America since the 1960s (Maki *et al.,* 1992). Its effects on birds and mammals can extend beyond the area occupied by oil and gas industrial infrastructure. Cameron *et al.* (2005) found that calving caribou abundance was lower within 4 km of roads in an oil and gas development area and declined exponentially with road density. With increasing infrastructure, high-density calving shifted inland, despite the lower forage biomass there (see also Wolfe *et al.,* 2000). Similarly, passerine bird nests are at greater predation risk within 5 km of infrastructure (Liebezeit *et al.,* 2009; see also Weiser & Powell, 2010). Substantial tundra habitat changes are expected with climate change that may have substantially greater impacts on habitat than human infrastructure, including increases in shrub-dominated ecosystems and changes in wetland abundance and distribution (section 4.4.3).

Boreal forest disturbance (tree cover loss), due largely to fire and forestry, was globally the second largest in both absolute and proportional extent from 2000-2010 (Hansen *et al.,* 2013). North America presented the higher overall rate of forest loss in comparison with other boreal coniferous and mountain ecozones in the world. In boreal forest, fire is the primary natural disturbance (see also section 4.5). Fire creates a complex mosaic of stands of varying age, composition, and structure, within which other disturbances and processes interact. Thus, it has been suggested to attenuate the impacts of logging on a managed landscape; logging should create patterns and processes resembling those of fire. However, logging has already shifted forest age-class distributions to younger stands, with a concurrent decrease in old-growth stands, and is quickly forcing the landscape outside of its long-term natural range of variability (Cyr *et al.,* 2009). Fire severity is a key component of regeneration trajectory (Johnstone *et al.,* 2010). Increases in boreal fires severity with climate warming may catalyze shifts toward deciduous-dominated forests, altering landscape dynamics and ecosystem services (see also sections 4.4.3 and 4.5). Besides climate impacts, other anthropogenic environmental changes like changes in biogeochemical cycles (section 4.4.2) and exotic invasive species (section 4.4.4) can interact with heat and drought (Millar & Stephenson, 2015) to negatively affect temperate and boreal forests.

The traditional fire knowledge of many native American cultures of North America was lost during European settlement. Many groups experienced declining the traditional fire knowledge systems abruptly and for several generations as most indigenous peoples in the subregion were forced from their ancestral lands, punished for speaking their native languages, and forbidden to use fire in open native vegetation. Some tribes, however, retained enough traditional fire knowledge although they did not practice traditional burning continuously on the landscape (Huffman, 2013).

Many temperate forests have at some time been used for agriculture. Large-scale deforestation first occurred during the 18th-19th centuries (Flinn & Vellend, 2005). Particularly across northeastern North America, phases of forest clearance were followed by agricultural use, agricultural abandonment, old-field succession, and then forest regeneration. Generally, species richness within forest stands (alpha diversity) remains lower in recent compared to ancient forests, even when recent forests are decades or centuries old (Flinn & Vellend, 2005). This biotic homogenization is legacy of human land-use that may endure for decades if not centuries (Leps & Rejmánek, 1991; Vellend, 2007; Thompson *et al.,* 2013; Deines *et al.,* 2016). Additionally, fire once shaped many North American ecosystems, but Euro–American settlement and 20th-century fire suppression drastically altered historic fire regimes, shifting forest composition and structure (McEwan *et al.,* 2011; Ryan *et al.,* 2013).

Earlier in the 20th century, USA land cover was on a trajectory of forest expansion after agricultural abandonment (Drummond & Loveland, 2010). The expansion of forest cover since 2000 has been offset by forest loss, with forest loss evenly divided among cropland, pasture and urban/suburban land (Masek *et al.,* 2011). The potential for forest regeneration has slowed, however, because forest conversion to urban/suburban land is less reversible. In addition, in some regions, like the eastern USA, tree cover has declined because forest harvest rates have outpaced reforestation (Drummond & Loveland, 2010, Masek *et al.,* 2011, Hansen *et al.,* 2013). Currently, according to Hansen *et al.* (2013) the northwestern USA is an area of intensive forestry, as is all of temperate Canada. Land-use pressures significantly impact the extent and condition of eastern USA forests, causing a regional-scale decline in tree cover, mainly from urban expansion. Annual forest loss accelerated from approximately 56,000 hectares from 1973-1980 to 90,000 hectares by 1992-2000 (Drummond & Loveland, 2010).

Prairie grasslands dominated central North America for millennia, until the mid- to late-1800s when European settlers converted them to croplands and rangelands (Ellis *et al.,* 2010). North American grasslands are now some of the planet's most heavily converted ecosystems (Isbell *et al.,* 2015). As a result of this dramatic habitat loss and fragmentation, these grasslands are rapidly losing plant species (Leach & Givnish, 1996; Wilsey *et al.,* 2005). Even more notable, nearly all of them have lost their keystone herbivores, including bison and elk. For example, during the mid-1800s, bison populations declined from tens of millions to a few thousand individuals (Knapp *et al.,* 1999). Since that time, bison numbers have increased to more than 100,000 individuals in public and private herds that are maintained for prairie restoration or meat production. Rangeland degradation in the west, grassland conversion to croplands, and afforestation of old fields in the east have together caused North American songbirds to sharply decline in recent decades (Brennan & Kuvlesky, 2005). Increased use of prescribed fire and grazing as sources of disturbance, and sowing of seeds to overcome dispersal limitation in fragmented agricultural landscapes, have improved prairie grassland restoration, preventing woody encroachment and restoring native plant diversity (Martin *et al.,* 2005).

A second wave of conversion of remaining fragments of North American grasslands to croplands, including 530,000 hectares from 2006-2011 in the upper Midwestern USA alone, has resulted from the recent doubling of crop prices following increased demand for biofuel feedstocks. These grasslands escaped conversion until only recently because they are particularly vulnerable to erosion and drought, or because they are adjacent to wetlands (Wright & Wimberly, 2013). The relationship between biofuel production and food prices is controversial in the scientific literature and depends on several factors as increased demand, decreased supply, and increased production costs driven by higher energy and fertilizer costs. Disentangling these factors and providing a precise quantification of their contributions is difficult but there is a convergence that analysis should include short and long-run effects, type of crops and technology (first or second-generation biofuels) as different biofuels have different impacts (Rathman *et al.,* 2010; Ajanovic, 2011; Mueller *et al.,* 2011; Zilberman *et al.,* 2013; Koizumi, 2015; Filip *et al.,* 2017).

Drylands in North America (the hot Sonoran, Mojave, and Chihuahuan deserts and the cool Columbia Plateau, Great Basin, and Colorado Plateau deserts) have experienced moderately low to high appropriation of land by humans; degraded to very degraded fire cycles; very high to extremely high habitat fragmentation; and habitat losses between 2000 and 2009 of up to 11% (Hoesktra *et al.,* 2010). Intensive cropping in many areas has lowered water tables and the amount of fertilized and salinized soil, leading to land abandonment with ensuing invasion by exotic annual grasses and reduced biodiversity and ecosystem function (Gelt, 1993). Most of these lands have been grazed by livestock since the early 1800s, and as most current grasses did not evolve with large mammal herds, this grazing has caused native species losses, altering plant and animal community composition, (Mack & Thompson, 1982). Climate change models are predicting higher temperatures and reduced precipitation for North American drylands (Cook *et al.,* 2004; Christensen *et al.,* 2007), likely leading to long-term declines in soil moisture, which will negatively affecting shallow-rooted plants (Fernandez & Reynolds, 2000; Munson *et al.,* 2011; Wertin *et al.,* 2015). Increasing carbon dioxide loss of grass, and altered climate and fire regimes favor woody plant encroachment, further reducing biodiversity and affecting animals that depend on native plants that are lost (Archer *et al.,* 1995). Grasses are vital to these ecosystems; they form the base of the food web, providing forage for livestock and small mammals, promoting soil carbon sequestration, stability and fertility and thus their loss affects ecosystem function (Sala & Paruelo, 1997). These landscapes are also seeing dramatic increases in soil surface disturbance from recreation and energy and mineral exploration and extraction (Weber *et al.,* 2016). Disturbance of the soil surface compromises the cover and function of biological soil crusts, a community of organisms that are critical to water, nutrient, and carbon cycles in drylands (Weber *et al.,* 2016) and they may not return to their pre-disturbance state or function (Concostrina-Zubiri *et al.,* 2014). Reduction in plant and biocrust cover increases soil erosion, which itself directly drives biodiversity loss and alters ecosystem function. Erosion reduces source soil carbon and nutrients (e.g. Neff *et al.,* 2008; Belnap & Büdel, 2016; Weber *et al.,* 2016; Ahlström, 2015); increases dust deposition on nearby snowpacks, which reduces the amount of water entering major rivers (Painter *et al.,* 2010); and threatens human economic, health, and social well-being (Fields *et al.,* 2009). Roads, pipelines, transmission lines, vegetation change, and energy developments continue to heavily fragment and degrade many drylands, especially the Mojave and Great Basin deserts (Knick *et al.,* 2003; Hoesktra *et al.,* 2010).

The wetlands of North America include many different wetland types, ranging from the expansive peatlands of boreal Canada and Alaska to the seasonally flooded marshes of the subtropical Florida Everglades. Wetlands of North America continue to be threatened by drainage for agriculture and urban development, extreme coastal and river management, water pollution from upstream watersheds, peat mining, waterfowl management, and more recently climate change. From 1780-1980, from 65 to 80% of wetlands in Canada were lost, while 53% of wetlands in the continental USA were lost (Mitsch & Hernandez, 2013). The middle Atlantic coastal plain experienced vast land cover change compared with other Eastern USA ecoregions, ranking third in the proportion of area changed. Two of the dominant land-cover types, forest and wetlands, experienced considerable net change (Auch, 2016). Urban development almost always increases in area, as it tends to be permanent, whereas other land-cover types, like forest, agriculture, wetlands, and mechanically disturbed lands, may fluctuate in area as part of cyclic land-use changes (Auch, 2016). Probably as a result of enforcing Clean Water Act requirements to mitigate wetland losses, as well as program such as the Wetlands Reserve Program (Wiebusch & Lant, 2017), wetland restoration and creation may have partially offset losses in rural and suburban areas since the mid-1980s (Mitsch & Hernandez, 2013).

North America contains some of the most urbanized landscapes in the world. In the USA and Canada, approximately 80% of the population is urban (Kaiser Family Foundation, 2013 in McPhearson *et al.,* 2013). Population growth combined with economic growth has fueled this recent urban land expansion. Between 1970 and 2000, urban land area expanded annually by 3.31% (Seto *et al.,* 2011), which was mostly cropland and forest conversion (Alig *et al.,* 2004), creating unique challenges for conserving biodiversity and maintaining regional and local ecosystem services. Urban areas in the USA could increase by 79% by 2025, which would mean that 9.2% of USA land will be urban (Alig *et al.,* 2004). A large portion of this increase is expected in coastal areas where populations will be exposed to issues associated with predicted sea level rise. Changes in development density will have an impact on how populations are distributed and will affect land use and land cover. Some of the projected changes in developed areas will depend on assumptions about changes in household size and how concentrated urban development will be. While higher population density means less land is converted from forests or grasslands, it can result in larger extents of paved areas and an increase in low-density exurban areas, which will lead to a greater area affected by development and increase commuting times and infrastructure costs (Brown *et al.,* 2014).

Mesoamerica

Drivers of change in biodiversity and ecosystem function in Mesoamerican drylands (Sonoran and Chihuahuan deserts) are similar to those in North America, though they differ in relative importance (CONABIO, 2014). Livestock have grazed Mexican deserts and semi-deserts for hundreds of years. Again, lack of resistance to this herbivory has altered plant community composition, decreased native species cover, and altered nutrient, carbon, and hydrologic cycles. (Mack & Thompson, 1982). Climate models predict warmer temperatures and reduced precipitation for this region (Cook *et al.,* 2004, Christensen *et al.,* 2007). These changes, along with natural drought will cause loss of grasses and other shallow-rooted plants (Fernandez & Reynolds, 2000; Moreno & Huber-Sannwald, 2011) and facilitate woody plant encroachment, which is already underway (Archer *et al.,* 1995). Loss of grasses will reduce food availability for livestock and wildlife, reduce an already limited soil carbon sequestration, reduce limited soil nutrients, alter plant and animal community composition and change ecosystem functions (Sala & Paruelo, 1997). Loss of biological soil crusts[21] and plant cover reduction with soil disturbance negatively influences water, nutrient, and carbon cycles and increases soil erosion in these ecosystems (Weber *et al.,* 2016). Disturbed biological soil crusts may not recover to a pre-disturbance state, altering their ecosystem role (Concostrina-Zubiri *et al.,* 2014). Grazing, cropping, energy and mineral exploration and development, and recreation are the major drivers of land degradation of Mexican deserts and semi-deserts (Sarukhan *et al.,* 2015; Sala *et al.,* 2000). These changes generally result in loss of biological soil crust and plant cover, resulting in soil erosion, which is a major issue in Mexican deserts and semi-desert areas (Balvanera *et al.,* 2009). Hoesktra *et al.* (2010) report that these areas have experienced moderately low to moderate appropriation of land by humans, fire cycles that are degraded, very high to extremely high fragmentation, and up to 3.3% habitat losses between 2000 and 2009.

Mesoamerican forests are the third largest among the global biodiversity hotspots and are one of the most endangered ecosystems in the tropics (Sánchez-Azofeifa *et al.,* 2014) due to high rates of forest loss and fragmentation (Chacon, 2005).

Drivers of change in Mesoamerican tropical dry forests are both negative and positive, but they still contribute to significant forest loss. Dry forests now exist as fragments of what was once a large, contiguous forest extending from Mexico to northern Argentina. The timber industry, indigenous fuel–wood extraction, and cattle ranching expansion are the main drivers of dry forest loss (Fajardo *et al.,* 2005; Calvo-Alvarado *et al.,* 2009). These forests now cover 519,597 km^2 across North and South America. Mexico contains the largest extent at 181,461 km^2 (38% of the total), although it remains poorly represented within protected areas (Portillo-Quintero & Sánchez-Azofeifa, 2010).

In general, tropical dry forest area in Mexico is declining, with cattle ranching driving most of this deforestation, particularly along the Pacific coast (Sanchez-Azofeifa *et al.,* 2009), even though the forest loss rate in Mexico was halved between 2010 and 2015 (Keenan *et al.,* 2015). Unfortunately, the protected tropical dry forest in Costa Rica represents less than 1% of the total extent of this ecosystem in the Americas and is continentally less significant. Low extent and high fragmentation of dry forests in Guatemala, El Salvador, and Nicaragua mean that these forests are at high risk from human disturbance and deforestation.

There are many wetlands and freshwater systems in Mesoamerica that are each integral to a system of life, culture, a means of economic support and habitat. Tourism income represents 20.4% of the foreign earnings in Mesoamerica (Agencia EFE, 1998). The location and topographic complexity of Mesoamerica makes it unique in its water availability, with an average of 27,200 m^3 inhabitants per year. The World Meteorological Organization cites that Mesoamerican countries have few real problems with water supply, using on average less than 10% of the available water resources. However, countries like Mexico, Guatemala and El Salvador experience water shortages (IUCN 1999, https://portals.iucn.org/library/efiles/documents/1999-012.pdf). In Mexico, water shortages occur because water resources are not located close to human settlements, producing an imbalance between supply and demand and leading to overexploitation of aquifers and water transfer between basins (Arriaga *et al.,* 2000). According to the National Water Commission Atlas (CONAGUA, 2012), 101 of the 282 most important aquifers are currently overexploited, mainly because of excessive water extraction for agricultural irrigation. These overexploited aquifers provide 49% of subterranean water. The most serious environmental impacts include droughts in semi-arid areas that reduce flow and its timing, saline intrusion into aquifers, and wetlands ecosystem deterioration (Ávila *et al.,* 2005).

Continuous groundwater pumping irreversibly affects natural water discharge flowing into aquatic ecosystems and riparian areas, even those that are far from mining areas. There are several cases in Mexico where the loss of fresh water that previously came from groundwater threatens the ecosystem. Such is the case of wetlands in Xochimilco, springs high Lerma and Aguascalientes, several major lakes in central Mexico (Chapala, Cuitzeo and Patzcuaro) or wildlife protected area Cuatrociénegas, among many others (Carabias *et al.,* 2010).

In El Estor, a wetland area in Guatemala, only small wetland remnants remain; most wetlands in the area have been transformed to large-scale oil palm, sugar cane, and other crops, displacing communities and causing land conflicts among other problems (Guatemala Ramsar National Report, 2015).

The Honduras Wetland Inventory (SERNA, 2009) notes that the most affected and currently endangered systems in Honduras are humid forests and the freshwater systems within them; due to replacement with monocultures like oil palm and banana or urban lands. Honduras has implemented agreements of understanding with the private sector to carry out international certification and develop

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[21] Biological soil crusts result from an intimate association between soil particles and cyanobacteria, algae, microfungi, lichens, and bryophytes (in different proportions) which live within, or immediately on top of, the uppermost millimeters of soil. Soil particles are aggregated through the presence and activity of these biota, and the resultant living crust covers the surface of the ground as a coherent layer.

programs of good practices considering the policy and strategy of cleaner production for oil palm because it is affecting large areas of wetlands in the country. On the other hand, regulations including subsidies and incentives promoting monocultures in protected areas are under review that will, in some cases, allow for excessive development within these areas (Honduras Ramsar National Report, 2015)

Mangroves in Mesoamerica are also threatened by deforestation and aquaculture. Mexico has 5.4% of the global extent of mangroves (Giri *et al.,* 2011), but many of those forests are being replaced with shrimp farms, agro-industrial plantations, or tourism enterprises. The threats to mangroves are similar along the Nicaraguan Pacific coast, which is unique as it marks the transition from dry to moist. The total destruction of the Estero Real mangrove in the Fonseca Gulf (between Nicaragua and El Salvador) is a clear example of the impact of uncontrolled shrimp-farm development in the region.

Caribbean

Humid and dry tropical forests are increasing overall across the Caribbean as agriculture has declined. In Puerto Rico and the Lesser Antilles, forest cover has been increasing since the 1950s (Helmer *et al.,* 2008a,b), starting with emigration to more developed countries after the Second World War and continuing with emigration from rural to urban areas as local economies shifted from agriculture to industry and services. This shift is largely the result of sugar cane cultivation becoming less profitable due to the rise of mechanized sugar cane cultivation in South America and cessation of European price supports for banana cultivation in the Lesser Antilles (Helmer *et al.,* 2008b; Walters, 2016). In a subset of four islands of the Lesser Antilles, cultivated land area declined 60-100% from 1950-2000, while forest cover increased 50-950% and urban land areas increased 90 to 2400% (Helmer *et al.,* 2008b). Forest recovery will likely continue on islands like St. Kitts, Barbados, and Trinidad, where local government subsidies for sugar cane cultivation stopped only in the last decade (Helmer *et al.,* 2008a, b; Helmer *et al.,* 2012; Walters, 2016).

Forest recovery is most extensive in the least accessible places: at higher elevations, further from roads and urban centers, and in protected areas (Helmer *et al.,* 2008a; Chai *et al.,* 2009; Newman *et al.,* 2014a). Deforestation and forest fragmentation continue in some places, including for small-scale agriculture where there is underemployment, when coffee prices are high, or in protected areas where protection is not enforced (Chai *et al.,* 2009; Newman *et al.,* 2014 a, b). Haiti, the poorest country in the Caribbean, lost forest cover from 2001-2010 (Alvarez-Berrios *et al.,* 2013).

In the Caribbean, expansion of tourism and urbanization drive land-cover change rather than agriculture and cattle ranching expansion. The attraction of Caribbean islands for the development of exclusive resorts and golf courses targeted at the North American and European markets drives this land-cover change. Such tourism development plus urbanization often most severely impact tropical dry forests on Caribbean islands, because these forests are located at lower elevations and in coastal areas (Helmer *et al.,* 2008b; Portillo-Quintero & Sanchez-Azofeifa, 2010; van Andel *et al.,* 2016).

Development also affects water quality in freshwater and coastal systems (see 4.4.2). In the Lesser Antilles, much of the urban and residential development is for tourism and for former emigrants returning to retire (Walters, 2016). Mangrove area has declined in the Caribbean from 1980-2010 (Angelelli & Saffache, 2013), and mangrove forests continue to undergo clearing for land development (Schleupner, 2008); although, mangroves have recovered in some places where they were previously cleared for agriculture (Chinea & Agosto, 2007). Cuba alone has 3.1% of the global extension of mangroves (Giri *et al.,* 2011).

Over 180 million people live in or travel to coastal areas of the Caribbean Sea and Gulf of Mexico annually, not counting USA coastal areas. Urban habitats have been changing rapidly in the Caribbean, with unforeseen consequences on the quality of life. An important issue has been the rapid spread of diseases, like those borne by mosquito vectors. For example, in the municipality of San Juan, Puerto Rico, the incidence of dengue fever has increased along with sea surface temperatures and sea level, as more areas for breeding become available along the shoreline and because of increasing rainfall (Mendez-Lazaro *et al.,* 2014).

Caribbean marine ecosystems are among the most severely impacted globally (Halpern *et al.,* 2007), mainly due to impacts on coastal systems: mangroves, coral reefs, seagrass beds and beaches (see also section 4.4.2). Live coral cover declined by 80% in 25 years in the wider Caribbean to 2001 (Gardner *et al.,* 2003), and further declined following mass coral bleaching in 2005 (Wilkinson & Souter, 2008).

South America

Net forest loss from 2010 to 2015 in South America was dominated by forest loss in Brazil (984,000 hectares per year) and, to a lesser extent, Paraguay (325, 000 hectares per year), Argentina (297,000 hectares per year), Bolivia (289,000 hectares per year) and Peru (187,000 hectares per year) (Keenan *et al.,* 2015). Despite the net loss of forest in South America, there has been a decline in the net rate of forest loss in some countries of the Americas (for example, in Brazil, the net loss rate between 2010 and 2015 was only 40% of that in the 1990s) and forest area increased in other countries in the last five years (for example, in Chile partly due to an increase in planted forest areas) (Keenan *et al.,* 2015).

Deforestation and degradation of tropical rainforest are important global issues due to their role in carbon emissions, biodiversity loss, and reduction of other ecosystem services (Foley *et al.,* 2007). Of global gross forest cover loss from 2000 to 2012, 32% occurred within tropical rainforests (Hansen *et al.,* 2013). Almost half of rainforest loss was found in South America, primarily in the Amazon basin. Large-scale (e.g. cattle ranching) and small-scale farming were historically the most significant drivers of deforestation in the Amazon. These farming activities resulted from favorable incentives received by cattle ranchers in the 1960s–1980s. More recently, the establishment of soy farming has become a land-demanding economic activity (Kirby *et al.,* 2006; Rudel *et al.,* 2009). Deforestation influences Amazonian fire regimes because it results in increased sources of ignition, increased forest edge lengths, and alterations of regional climates (Alencar *et al.,* 2015). Droughts linked to the El Niño and human-related activities were associated with large forest fires (Alencar *et al.,* 2006; Morton *et al.,* 2013). If climate change and increased forest degradation continue, fires may burn more frequently and expand to larger areas, perhaps including landscapes that otherwise are fire resistant (Alencar *et al.,* 2015).

Together with lowland tropical forests, mountain areas represent an important percentage of South America (Armenteras *et al.,* 2011). Andean forests are particularly susceptible and highly vulnerable to climate change because of their location on steep slopes and because of their altitudinal and climatic gradients (Karmalkar *et al.,* 2008). In addition to climate change, tropical mountains are subject to high pressure from other natural and anthropogenic drivers of change like land use and land cover change, soil erosion, landslides and habitat destruction (Achard *et al.,* 2002; Bush *et al.,* 2004; Grau & Aide, 2008).

Together with Mexico, Brazil and Bolivia harbor the largest and best-preserved tropical dry forest fragments. The Chiquitano dry forests of Bolivia and Brazil alone extend over 142,941 km^2 (27.5% of total dry forest area in the region) (Portillo-Quintero & Sánchez-Azofeifa, 2010). Of the 23,000 km^2 of dry forest under legal protection, 15,000 km^2 are in Bolivia and Brazil. In fact, Bolivia protects 10,609 km^2 of dry forests, including 7,600 km^2 in a single park. In other countries, like Ecuador and Peru, however, low extent and high fragmentation of dry forests were observed.

Woodlands and savannas in South America are also under strong conversion rates related to the expansion of soybean and pasture (Barona *et al.,* 2010). The Brazilian Cerrado is the second largest biome in South America and is considered a biodiversity hotspot. By 2010, approximately 50% of the original vegetative cover of the Brazilian Cerrado has been converted. Land use changes in the Cerrado, often coupled with increased fire frequency and invasion of exotic species, have generated profound changes in the vegetation structure and functioning of these ecosystems (Bustamante *et al.,* 2012). Alterations in land cover from natural to rural and urban are also changing stream water chemistry in the Cerrado (Silva *et al.,* 2011).

Fire is an important factor in maintaining grassland ecosystems. It prevents woody encroachment, removes dead herbaceous material, and recycles nutrients. Without fire, organic matter and litter would accumulate and tree densities would increase, leading eventually to forested areas. The timing, frequency, and intensity of fires determine specific effects of these events on the functioning of grassland ecosystems. Indigenous people in the Cerrado region have been using fire for multiple purposes (**Table 4.12** and **Box 4.10**).

Similarly, vegetation cover loss in the dry Chaco from 2002 to 2006 was associated to the rapid expansion soybean and planted pastures (Clark *et al.,* 2010). During this period a net loss of 6.9 million hectares of closed-canopy (≥80% cover) was detected in the dry Chaco ecoregion. Some of the loss of woody vegetation can be attributed to forest degradation, where forests have trees and shrubs removed as an intermediate step to agriculture or pastures (Clark *et al.,* 2010).

Change in South American grasslands (distinguished from grasslands found in dryland regions that generally did not evolve with large mammalian herds) has been brought about primarily by conversion of these ecosystems to agriculture. The Río de la Plata grasslands are one of the largest temperate grasslands regions of the world, covering nearly 700,000 km^2 of eastern Argentina, southern Brazil and Uruguay (Paruelo *et al.,* 2007). This region plays a key role in international crop production and land use change rates in some areas and are among the highest detected nowadays. Agricultural activities have undergone important changes during the last 20 years because of technological improvements and new national and international market conditions for commodities (mainly soybean, sunflower, wheat, and maize) (Baldi & Paruelo, 2008).

Wild ungulates are also an essential component of energy and nutrient flows in grassland ecosystems that evolved with grazing. By contrast, domestic livestock generate effects that are disputed as either positive or negative, particularly in relation to different stocking densities, different grassland environments and whether the different environments evolved with large mammalian herds (Mack & Thompson, 1982). The economic and environmental sustainability of beef cattle from pasture use and preservation in specific

Box 4 10 Traditional fire management in the South America.

Traditional fire knowledge is as fire-related knowledge, beliefs, and practices that have been developed and applied on specific landscapes for specific purposes by long time inhabitants (Huffman, 2013). Across the Americas indigenous people have managed fire for different purposes. The articulation of traditional and scientific knowledge can be a valuable strategy for the formulation of environmental policies for effective fire management.

Indigenous peoples have been using fire in the Cerrado (savannas) of Brazil as a form of management for thousands of years. Mistry *et al.* (2005) studied the traditional use of fire as a management tool by the Krahˆo indigenous group living in the northeastern region of Tocantins state, Brazil. The results indicate that the Krahˆo burn for a variety of reasons throughout the dry season, thereby producing a mosaic of burned and unburned patches in the landscape **Table 4.12**. Similarly, in Canaima National Park, Venezuela, a protected area inhabited by the Pemón people, ecological studies have revealed that the creation of a mosaic of patches with different fire histories could be used to create firebreaks that reduce the risk of the wildfires that threaten the vulnerable and diverse savanna-forest transition areas (Bilbao *et al.,* 2010). In the Amazon region, particularly along large and small rivers, are numerous patches of Amazonian dark earth (Junqueira *et al.,* 2010). These are anthropogenic soils associated with archaeological sites, created mostly between 1000 BC and the European conquest around 500 years ago and managed with the use of fire (Rebellato *et al.,* 2009). Pre-conquest Amazonian peoples used fire for most of their landscape management. Small areas were weeded with wooden digging sticks and wooden machetes, while occasional small trees were cut with stone axes and burned well before being completely dry and/or with low oxygen availability, leaving large amounts of charcoal instead of easily eroded ash (Denevan, 2004). The combination of fire management and plant cultures improved soil fertility and once a plot was abandoned growth of secondary forests was rapid (Junqueira *et al.,* 2010).

Table 4 12 **The different burning regimes used by the Krahˆo. Source: Mistry *et al.* (2005).**

BURNING REGIMES FOR DIFFERENT PURPOSES	
Protection of roça (swidden plots)	Early dry season, around April/May
Protection of certain fruiting trees	Early dry season, around April/May
Hunting	April is perceived as the best time—small patches of Cerrado are burnt over a number of days during a hunting trip
Protection of *carrasco*	Burnt April/May every 5–6 years
Livestock	Grazing Pasture burnt in mid-May—small areas burnt each year
Protection of areas of *Cerrado* from later, more intense fires	Early to mid dry season
Clearing and preparing land for planting	Roças are burnt at the very end of August or in September
Honey extraction	September and October
Keep clean and increase visibility	Throughout the dry season—fires are set when walking to villages, hunting and travelling to roças
Eliminate pests	Throughout dry season
Outsider fires	Occur throughout dry season

biomes is still not well evaluated. The study of the feasibility of beef production in the pampa biome suggests it is possible to optimize low greenhouse gases emission of beef production with a significant economic return under certain feed conditions. Actually, studies suggest it is possible to obtain beef production increases without the need of new livestock areas, which can contribute to the proper use and preservation of the pampa biome (Ruviaro *et al.,* 2016, see also Modernel *et al.,* 2016).

Afforestation of some of the most productive native grasslands of the region is currently undergoing, and might be further promoted by carbon markets (Paruelo *et al.,* 2007) posing a new threat to these ecosystems. Interestingly, grasslands store approximately 34% of the global stock of carbon in terrestrial ecosystems while forests store approximately 39% and agroecosystems approximately 17%. Unlike tropical forests, most of the grassland carbon stocks are in the soil.

Drylands cover more than 50% of South America. The region possesses tropical, highland, coastal and continental drylands (Cabrera & Willink, 1980). In South America, humans have appropriated much of the Sechura Desert (Peru) for their use, and the habitat is highly fragmented (Hoekstra *et al.,* 2010). Similarly, the Atacama Desert (Chile) has experienced moderate land appropriation for human use and moderately high habitat fragmentation (Hoekstra *et al.,* 2010). In Patagonia, heavy sheep grazing has locally extirpated preferred forage species, thus altering plant community composition and resulting in the endangerment of 76 grass species (Cibils & Borrelli, 2005). Aside from grazing, this region has experienced a relatively low appropriation of land for human use, but has very high habitat fragmentation (Hoekstra *et al.,* 2010). As with the other deserts, it does not have a natural fire cycle. Habitat loss in all three regions has been relatively low (0.1% for Atacama Desert, 0.5% for Sechura Desert, and 1.6% for the Patagonia steppe) (Hoekstra *et al.,* 2010).

From 2001 to 2013, 17% of new cropland and 57% of new pastureland replaced forests throughout Latin America (Aide *et al.,* 2013). Cropland expansion from 2001 to 2013 was less (44.27 Millions of hectares) than pastureland (96.9 Millions of hectares), but 44% of the 2013 cropland total was new cropland, versus 27% of the 2013 pastureland total, revealing higher regional expansion rates of row crop agriculture. The majority of cropland expansion was into pastureland within core agricultural regions of Argentina, Brazil, Bolivia, Paraguay, and Uruguay (Graesser *et al.,* 2015; Volante *et al.,* 2015). Commodity crop expansion, for both global and domestic urban markets, follows multiple land change pathways entailing direct and indirect deforestation, and results in various social and environmental impacts (Meyfroidt *et al.,* 2014).

Forested wetlands in the western Amazon, have declined only moderately in area in recent years but local deforestation is more intense in the eastern Amazon. Habitat loss in that region is mostly concentrated in the vicinity of very large cities and in the Amazon estuary (Magalhães *et al.,* 2015). The anthropization of these wetlands involves the forest cover removal, or alternatively, sudden changes in forest composition (Freitas *et al.,* 2015). Natural wetland habitats are continually transformed into croplands and pastures (Junk *et al.,* 2014).

In recent years many new large dams have been planned for the Amazon and its connection to the Andes (Finer & Jenkins, 2012; Fearnside, 2013), causing deforestation and habitat loss (mainly riverine habitats, forming wetland patches along the river side) as main impacts (among others) (Lima *et al.,* 2014, Cunha & Ferreira, 2012; Ferreira *et al.,* 2013). Further, dam construction comes with huge social and economic costs involved (Fearnside 2005 and 2015). About 60% of the rural population lives inside várzeas (basin), and all major large cities are inside or on the border of flooded environments. Most timber and a significant part of the beef, fruits and vegetables consumed in urban areas are produced in these wetlands. Additionally, most of the fish consumed come from the white-water rivers and their floodplains (Junk *et al.,* 2012). Wetlands also provide other benefits to people (Castello *et al.,* 2013b, Junk *et al.,* 2014), particularly because they retain nutrient rich sediment that forms new soil, control erosion, and sequester carbon dioxide.

The intense loss of natural habitats and associated biodiversity is causing the slow degradation of South American wetlands, reducing natures benefits to people by reducing the number of commercial fish species, total fish stocks, and a persistent "fishing-down" process Castello *et al.,* 2013; Cella-Ribeiro *et al.,* 2015), as well as the loss of carbon dioxide sinks where land-use change has been intense (Schöngart *et al.,* 2010; Vogt *et al.,* 2015).

Unregulated markets for timber and fish (Soares-Filho *et al.,* 2006; Junk *et al.,* 2007), among other natural resources harvested from the Amazonian wetlands, are the main source of illegal pressure on the extraction rates of those resources. Rural-urban migration in the Amazon, closely related to wetlands, has contributed to urban degradation, and also puts pressure on rural exploitation, affecting forest extent, since important rural patterns of consumption are maintained (Padoch *et al.,* 2008).

The marine areas of South America include almost 30,000 km of coastline and encompass three different oceanic domains—the Caribbean, the Pacific, and the Atlantic (latitude range from 12°N to 55°S) (Miloslavich *et al.,* 2011). Habitat transformation (for infrastructure expansion, aquaculture, agriculture, etc.), and sewage and garbage disposal are among the most recurrent problems in South America coastal zones. As such, these areas undergo fast and frequently drastic transformation. When compared to other tropical regions like Southeast Asia, the importance of aquaculture in South America is relatively small. Nonetheless its importance is growing in countries like Ecuador, where a significant shrimp mariculture industry has developed mostly in mangrove converted areas and salt ponds and in Peru and Chile (Humbolt Current region) with the cultivation of introduced salmonid species (Campuzano *et al.,* 2013). In the tropical west Atlantic major threats are industrial (trawling) and artisanal (line and longline) fishing, urban development, agriculture development, dredging and flow navigation, water pollution (runoff from smaller rivers as in terms of volume the Orinoco and Amazon discharge is relatively pristine), mangrove deforestation, activities related to oil and gas exploitation, port activities, and maritime shipping (Klein *et al.,* 2009).

Mangroves in South America correspond to 11% of the global mangrove extent (Giri *et al.,* 2011). In the Brazilian shelf, mangrove ecosystems cover 16 of the 17 Brazilian coastal States, representing 85% of the coastline (about 7,300 km), and the extent of mangroves along the Brazilian coastline from east of the Amazon River mouth (Pará) to the Bay of São José (Maranhão) constitutes the largest continuous belt globally (Nascimento *et al.,* 2013). Although almost 83% of mangrove areas are protected, human settlements along the coast have dramatically increased, impacting mangroves by diverting freshwater flows and degrading water quality. Mangroves also undergo salt extraction and conversion to agriculture, aquaculture (mainly shrimp farms), or built-up lands, all of which contribute to mangrove degradation and deforestation (Magris & Barreto, 2010). Despite its value, the mangrove ecosystem is one of the most threatened on the planet. Mangroves are being destroyed at rates three to five times greater than average rates of forest loss and over a quarter of the original mangrove cover has already disappeared; this destruction is driven by land conversion for aquaculture and agriculture, coastal development, pollution and overexploitation of mangrove resources. As mangroves become smaller and more fragmented, important ecosystem goods and services will be diminished or lost. The consequences of further mangrove degradation will be particularly severe for the well-being of coastal communities in developing countries, especially where people rely heavily on mangrove goods and services for their daily subsistence and livelihoods (Valiela *et al.,* 2001; Duke *et al.,* 2007; UNEP, 2014).

South America's west coast is home to approximately 40 million people. In Chile, three quarters of the population lives and works along a 500 kilometer stretch of coastline between Valparaiso and Concepcion, representing 15% of the country's land area. In the east coast, over 15 million people live in the Buenos Aires-La Plata-Montevideo coastal region. The coastal area between Sao Paulo and Rio de Janeiro, Brazil, hosts over 30 million people. Each of these areas continues to grow in population. The marine and inland waters are used for food production, transportation, tourism, and water supply and are important for the economic and social vitality of these communities. These aquatic ecosystems are exposed to resource use and extraction by a range of activities, from oil and gas to fisheries, from urbanization to agriculture. These activities lead to sediment, nutrient, or other pollutant inputs from the watershed (section 4.4). Many coastal, estuarine, and fresh water systems in the region have in the past seen intense outbreaks of cholera and other water-borne diseases, dengue fever and other mosquito-borne diseases, as well as an increase in the occurrence of harmful algal blooms. Some of these are due to population growth and eutrophication, but climate variability complicates the situation.

An important factor that affects the coasts and shelf environments is riverine discharge. Discharge affects the amount of sediment and nutrients that may be delivered to the coastal zone, and this in part depends on uses of the land in the watershed. As weather patterns of the future are still uncertain, the impact on global coastal systems is also a matter of speculation. Many rivers are intervened by damming, and many have different nutrient inputs due to point and non-point sources of nutrients and pollutants (section 4.4.2)

4.4.2 Pollution and related changes in biogeochemical cycles

Nature of the driver, its recent status and trends, and what influences its intensity

In its pursuit of food, water and civilization, humanity mobilizes chemicals that impact biodiversity and NCP. Pollutants **(Table 4.13)** are a major driver of declinesin freshwater systems, which are now, in many cases, severely degraded (Dudgeon *et al.,* 2006). Besides changing climate (section 4.4.3), increased concentrations of atmospheric carbon dioxide adversely impacts marine species through ocean acidification. Pollutants also affect biodiversity because their human use to increase food, energy or minerals alters air, water and soil chemistry, or disturbs watersheds, causing soil erosion and sediment movement into water bodies. Other pollutants are toxic to organisms.

Ocean acidification, deoxygenation and plastics pollution

As atmospheric carbon dioxide increases, mainly from fossil fuel combustion, pH and calcium carbonate saturation in ocean water decrease (Fabry *et al.,* 2008). This is adversely impacting marine ecosystems and biota (Cooper *et al.,* 2008; Fabry *et al.,* 2008; Albright & Langdon, 2011; Anthony *et al.,* 2011; Pandolfi *et al.,* 2011; Bramanti *et al.,* 2013; Courtney *et al.,* 2013; Webster *et al.,* 2013; Hall-Spencer *et al.,* 2008). Many marine animals, like plankton, mollusks, sea stars, corals, snails and other groups, extract calcium carbonate from seawater to form their skeletal structures or shells. Ocean acidification reduces the calcium carbonate availability. The ocean is also undergoing deoxygenation. Ocean oxygen content declined 2% since 1960 and with climate change could decline an additional 1 to 7% by 2100. In the upper water column, warmer waters from global climate change drive this deoxygenation by reducing oxygen solubility; at lower depths, reduced mixing is the chief driver. Along coastlines, rivers with large nitrogen and phosphorus loads draining from fertilized agricultural watersheds, or from

Table 4 13 **Examples of ubiquitous water pollutants (A) micropollutants; (B) macropollutants and fluxes to world rivers. Source: modified from Schwarzenbach *et al.* (2006) and references therein.**

A ORIGIN/USAGE	CLASS	SELECTED EXAMPLES	RELATED PROBLEMS
Industrial chemicals	Solvents	Tetrachloromethane	Drinking-water contamination
	Intermediates	Methyl-t-butylether	
	Petrochemicals	BTEX (benzene, toluene, xylene)	
Industrial products	Additives	Phthalates	
	Lubricants	PCBs (polychlorinated biphenyls)	Biomagnification, long-range transport
	Flame retardants	Polybrominated diphenylethers	
Consumer products	Detergents	Nonylphenol ethoxylates	Endocrine active transformation product
	Pharmaceuticals	Antibiotics	Bacterial resistance, nontarget effects
	Hormones	Ethinyl estradiol	Feminization of fish
	Personal-care products	Ultraviolet filters	Multitude of (partially unknown) effects
Biocides	Pesticides	Dichlorodiphenyltrichloroethane (DDT)	Toxic effects and persistent metabolites
		Atrazine	Effects on primary producers
	Nonagricultural biocides	Tributyltin	Endocrine effects
		Triclosan	Nontarget effects, persistent degradation product (methyl-triclosan)
Geogenic/natural	Heavy metals	Lead, cadmium, mercury	
	Inorganics	Arsenic, selenium, fluoride, uranium	Risks for human health
	Taste and odor	Geosmin, methylisoborneol	Drinking-water–quality problems
	Cyanotoxines	Microcystins	
	Human hormones	Estradiol	Feminization of fish
Disinfection/oxidation	Disinfection by-products	Trihalomethanes, haloacetic acids, bromate	Drinking-water–quality, human health
Transformation prods.	Metabolites from all above	Metabolites of perfluorinated compounds	Bioaccumulation despite low hydrophobicity
		Chloroacetanilide herbicide metabolites	Drinking-water–quality problems

B EXAMPLES OF AQUATIC MACROPOLLUTANTS AND FLUXES OR MASS OF ANTHROPOGENIC PRODUCTION MILLION METRIC TONS YR^{-1}

	Total inorganic nitrogen fluxes to world rivers (~75% anthropogenic)	21
	Total phosphorus fluxes to world rivers (60% anthropogenic)	5.6
	Anthropogenic inputs of heavy metals Zn, Cr, Ni, Pb, Cu, Cd, Hg	0.3-1
	Global fertilizer production (2000)	140
	Global pesticide production	5
	Synthetic organic chemicals production	300
	Oil spills (average 1980-2000)	0.4
	Plastics, Microplastics	*5-13

*Clark *et al.* (2016)

sewage and atmospheric nitrogen depostion, cause low oxygen levels and hypoxic "dead zones" (Diaz & Rosenberg, 2008; Rabalais *et al.*, 2014; Schmidko *et al.*, 2017). Hypoxic coastal waters have grown exponentially (Vaquer-Sunyer & Dwarte, 2008). The intensity and duration of hypoxia controls its impacts on biodiversity. The combination of warmer water, acidification and deoxygenation are likely interacting to negatively impact marine organisms (Bednarsk *et al.*, 2016).

Plastic pollution enters the ocean via rivers, sewage, fishing and other sources. Plastic characteristics, like lower natural resource use and costs, and resistance to degradation, drive consumer plastics use. Although waves and sunlight break plastics to smaller pieces including microplastics (<5 mm), non-bouyant plastics take hundreds of years to degrade in ocean waters and comprise 90 to 99% of ocean plastic pollution. Plastics kill or harm biodiversity, from zooplankton, to fish, shellfish, sea turtles, seabirds and marine mammals: animals frequently consume plastics or are suffocated or maimed by them. Impacts on marine wildlife include entanglement, ingestion, and contamination to a wide variety of species. Reductions in plastics use and disposal into the oceans wouold require policy development as well as consumer-driven changes in plastics use and disposal (Wilcox *et al.,* 2016). Many of the environmental implications of microplastics at sea are still largely unknown, however the number of marine species known to be affected by this contaminant has increased from 247 to 680 (Gall & Thompson, 2015). Microplastics have a complex effect on marine life. They adsorb legacy persistent organic pollutants and are passed up the food chain to higher trophic levels including to people, exposing humans and animals that consume marine biota to carcinogens and teratogens (toxic to embryos) (Clark *et al.,* 2016; Worm *et al.,* 2017). By fouling boats and fishing nets and equipment, plastic pollution imposes costs to the fishing industry and society for related cleaning and rescue (Clark *et al.,* 2016; Kershaw *et al.,* 2011). The top 20 countries' mismanaged plastic waste encompass 83% of the total in 2010 with Brazil in 16th position and the USA in the 20th position in the global ranking (Jambeck *et al.,* 2015).

Fertilization of Earth with nitrogen, phosphorus and other nutrients from human activities.

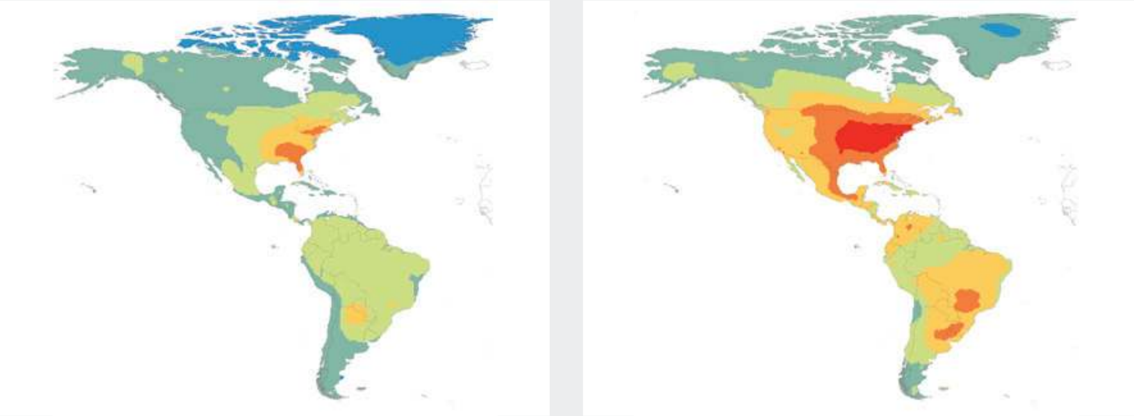
Food, fiber and energy production are changing the biogeochemical cycles of major nutrients (nitrogen, carbon, phosphorus, sulfur). The use of nitrogen, phosphate and potash fertilizer is increasing by 1.9% per year in the Americas, contributing to increasing nitrogen deposition onto ecosystems **(Figure 4.5)**. Demand for these agrichemicals will continue to increase, mainly due to increased demand in Latin America (FAO, 2011 and 2017). Increased biologically available, reactive nitrogen (all nitrogen forms except molecular nitrogen, N_2) is the most dramatic change (Rockström *et al.,* 2009; Jia *et al.,* 2016). Nitrogen is central to ecosystem productivity (LeBauer & Treseder, 2008; Elser *et al.,* 2009). In terrestrial systems, direct toxicity of nitrogen gases, ozone and aerosols, increased nitrogen availability, soil-dependent acidification, and secondary stress and disturbance, are ecosystem- and site-specific impacts that can contribute to species composition changes and reduced plant diversity (Valliere *et al.,* 2017; Bobbink *et al.,* 2010). Inorganic nitrogen fertilizer use releases reactive nitrogen to the atmosphere. In addition, concentrated animal feeding operations have emerged across the Americas. Animals (pigs, chickens, cows, fish and other animals) are confined, with large amounts of waste and ammonia produced. Applying this manure to agricultural fields can lead to pathogen and nutrient runoff into ground and surface waters. Increasing fossil fuel combustion, particularly coal burning for electricity, has also increased emissions of reactive nitrogen, including nitric oxide and ammonia, and sulfur dioxide. Emissions from large portions of North America have increased by more than 1,000% (van Aardenne *et al.,* 2001).

Nitrogen release can change ecosystem structure and function, affecting plant or microbial community composition, production, soil properties and susceptibility to fire or disease (Porter *et al.,* 2013). These changes can affect recreation, drinking water quality, timber production, fisheries, wildlife viewing, climate stability, fire risk, and "non-use" values of intact, natural ecosystems (Compton *et al.,* 2011). Runoff from agricultural fields, point sources of municipal waste (from human waste and manufacturing), and urban runoff, can transport nutrients and sediment to rivers and streams. This can increase nutrient (phosphorus, nitrogen, and carbon) concentrations and promote algal and aquatic vegetation growth causing eutrophication (**Box 4.17** and **Box 4.20**). In aquatic eutrophication, high levels of organic matter from fertilizer and sediment run-off, and organisms decomposing it, deplete water oxygen, killing organisms including fish. It can also shift primary producer communities, alter species composition and decrease plant diversity **(Box 4.17)**. Increased organic matter can also affect drinking water suitability and cause algal blooms that release toxins (Bushaw-Newton & Sellner, 1999; Lopez *et al.,* 2008; Michalak, 2015; Glibert *et al.,* 2006). Urea from fertilizer is also associated with increased paralytic shellfish poisoning along Americas coasts (Glibert *et al.,* 2006; Glibert, 2017). These nutrient flows increase as per capita GDP, food crop and meat and milk production increase **(Figure 4.6)**.

Rivers and streams naturally carry some uncontaminated sediment. However, increased land disturbance, primarily from agriculture and urbanization, can mobilize excessive amounts of fine sediment into streams. Excessive sedimentation can directly harm organisms. With mussels, for example, it buries adults and juveniles or interrupts respiration or feeding. In rivers, suspended sediments and sediment deposits may also bury eggs, displace host fish, or disrupt host fish/mussel interactions leading to declines of some species. Excessive sediment may also block light penetration into water, reducing primary production and causing the need for river channel dredging for ship traffic. Conversely, on many major rivers, dams for hydroelectric power and irrigation water have reduced river sediment loads. Lack of sediment can reduce habitat, excessively scour river channels and banks, and cause losses of coastal wetlands that depend upon a steady sediment supply (Morang *et al.,* 2013).

Figure ④ ⑤ **Total nitrogen deposition (wet and dry deposition of nitrogen oxides and reduced nitrogen) derived from the multi-model global datasets for nitrogen deposition from Lamarque *et al.* (2013).**

Data at resolution of 0.5*0.5 degrees and in units of kg N/ha/yr. 1850, 1980, 2000 and 2030 (rcp4.5). Nitrogen deposition is greatest in major agricultural regions. Source: Lamarque *et al.* (2013).

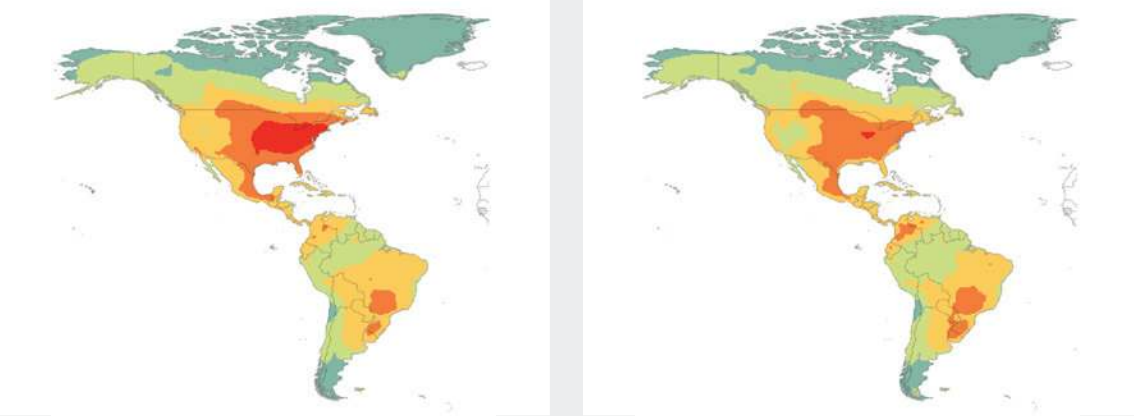


NITROGEN DEPOSITION 1850
(KG/HA/Y)

🟦	0.08 — 0.2
🟩	0.2 — 1
🟨	1 — 2.5
🟧	2.5 — 5
🟥	5 — 9

NITROGEN DEPOSITION 1980
(KG/HA/Y)

🟦	0.18 — 0.2
🟩	0.2 — 1
🟨	1 — 2.5
🟧	5 — 10
🟥	10 — 14.7



NITROGEN DEPOSITION 2000
(KG/HA/Y)

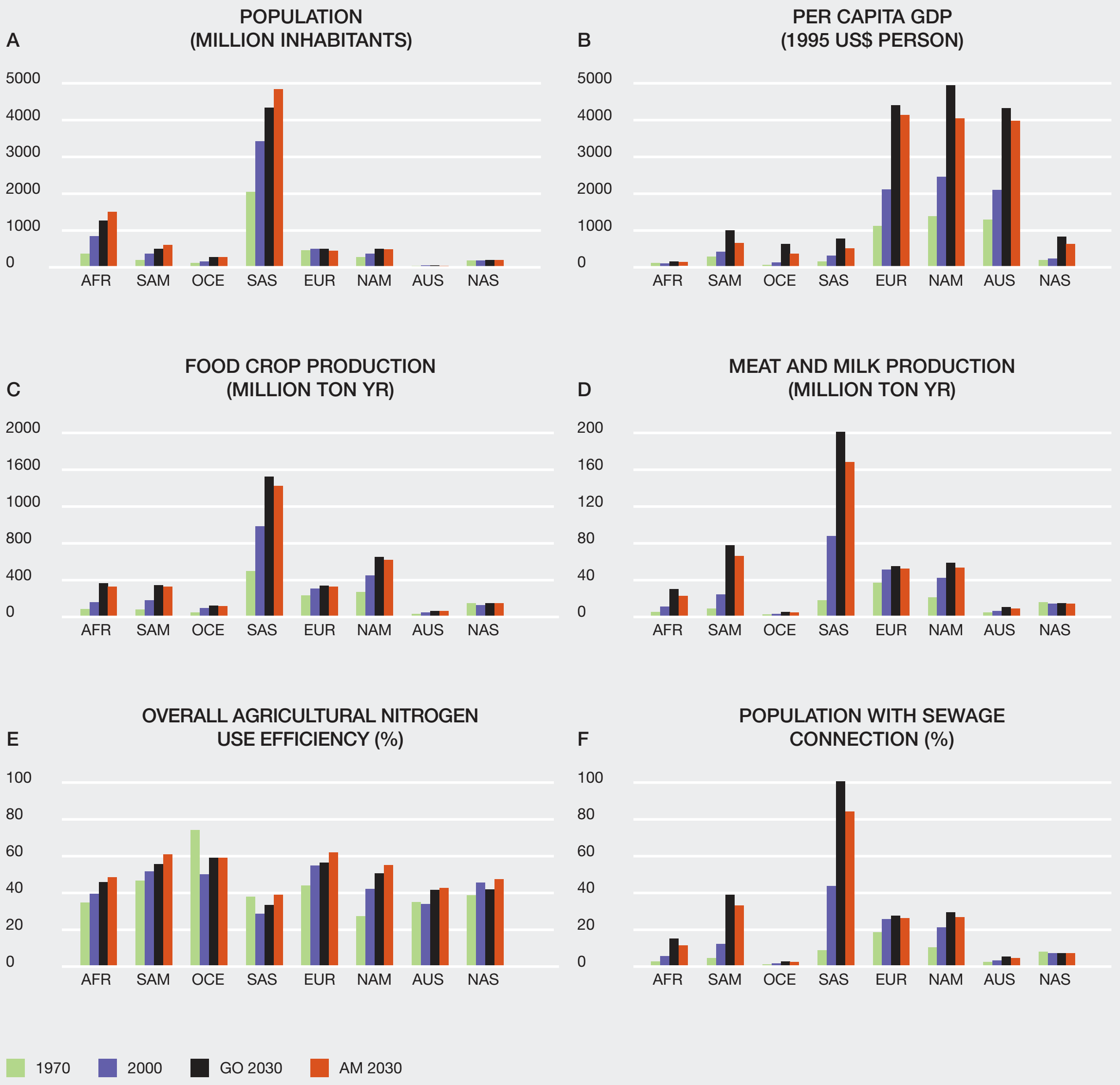
🟦	0.2 — 1
🟩	1 — 2.5
🟨	2.5 — 5
🟧	5 — 10
🟥	10 — 14.9

NITROGEN DEPOSITION 2030
(KG/HA/Y)

🟦	0.23 — 1
🟩	1 — 2.5
🟨	2.5 — 5
🟧	5 — 10
🟥	10 — 10.3

Figure ④ ⑥ **Anthropogenic drivers of nutrient flows for eight world regions for 1970, 2000, and 2030 for two scenarios:**

Global Orchestration (GO) (supra-national environmental regulation) and Adapting Mosaic (AM) (localized ecosystem management). Source: Seitzinger *et al.* (2010).



AFR, Africa; SAM, South America; OCE, Oceania; SAS, South Asia; EUR, Europe; NAM, North America; AUS, Australia; NAS, North Asia

Toxicants

Ecosystems throughout the world have experienced low-level exposure to many different toxicants due to human activities. Low-level exposure to toxicants may occur via air (e.g. tropospheric ozone), water (e.g. trace metals, methyl mercury, pharmaceuticals), soil or sediments (e.g. lead, polycyclic aromatic hydrocarbons), or food (pesticides, microplastics, bioaccumulative toxics). Toxicants released to the air are disseminated the longest distances and affect the most species.

Because biota experience toxicants in combination with other stressors (water stress, altered thermal regime, habitat destruction, etc.), toxicant effects are often difficult to ascertain. Much evidence of the adverse effects of low-level toxicant exposure on biodiversity is in the literature on point sources of trace metals to aquatic habitats. We have known since the 1980’s that changes in community composition occur at metal concentrations much lower than water quality criteria (Clements *et al.,* 1988, 2000, 2013). Restoration of streams contaminated by mine drainage is often unsuccessful because the sediments have accumulated trace metals that continue low-level exposure sufficient to inhibit numerous bottom-dwelling organisms (Clements *et al.,* 2010a, b). Metal concentrations below the chronic toxicity values on which water quality criteria are based can inhibit important ecosystem functions (e.g. photosynthesis) (Twiss *et al.,* 2004; Sunda, 2012). These effects of low-level exposure to toxicants are consistent with the observations that abrupt changes in community composition (loss of sensitive species, loss of functional groups, decreased abundance of some species and increases in others) occur at low levels of disturbance, including low levels of pollutants (Fleeger *et al.,* 2003; Dodds *et al.,* 2010; King & Baker, 2010).

Atmospheric ozone occurs where emissions from fossil fuel combustion (energy utilities, industry, motor vehicle exhaust) or biomass burning interact with vapors from solvents, gasoline or vegetation. Ozone damages plant tissues, decreases plant primary production, and changes plant and insect communities (Hillstrom & Lindroth 2008; Volk *et al.,* 2006), but its effects on biodiversity remain poorly studied.

Major sources of atmospheric mercury include fossil fuel (primarily coal) combustion (the largest source), artisanal gold mining, non-ferrous metal manufacturing, cement production, waste disposal, caustic soda production, and emissions from soils, sediment, water, and biomass burning, including re-emissions from past anthropogenic emissions (Pacnya *et al.,* 2006; Pirrone *et al.,* 2010). Legacy releases from commercial products and contaminated sites contribute to re-emissions (Horowitz *et al.,* 2014; Kocman *et al.,* 2013). In the vicinity of past or current mining, at higher latitudes, at mid latitudes with soft water ecosystems, or in regions downwind of coal fired power plants, consumers of aquatic foods may suffer high exposure to methyl mercury (Mahaffey & Mergler, 1998; Després *et al.,* 2005; Fujimora *et.al.*, 2012; Driscoll *et al.,* 2007). Methyl mercury is a potent neurotoxin, and it is particularly toxic to human and other vertebrate embryos.

The discovery and development of synthetic herbicides during World War II has increased crop yields, enhanced crop quality, and reduced production and harvesting costs (Coupe *et al.,* 2012). Possible health effects from exposure to pesticides include cancer, reproductive or nervous-system disorders, and acute toxicity. Recent studies suggest that some pesticides disrupt endocrine systems and affect reproduction by interfering with natural hormones (García *et al.,* 2017; Gilliom *et al.,* 2006). The amounts, types, and use of pesticides for agriculture change over time, but their worldwide use increases. Persistent organic pollutants, like organochlorine pesticides, polycyclic aromatic hydrocarbons, polychlorinated biphenyl compounds, polybrominated biphenyl ethers, and others, by being semi-volatile and resistant to degradation, are transported in the atmosphere or ocean to remote places where they can bioaccumulate and biomagnify in food webs (supplementary material: **Box 4.18** and **Box 4. 19**). Being detectable in most global ecosystems (Bartrons *et al.,* 2016), persistent organic pollutants should always be considered in total toxic burdens. Like methyl mercury, deposition from the atmosphere to water, soils, or sediment can be greater at colder-latitude or montane ecosystems where temperatures are colder or precipitation greater (Macdonald *et al.,* 2000; Blackwell & Driscoll, 2015; Kirchner *et al.,* 2009).

Agroecology is an alternative to conventional agriculture that builds on local knowledge and innovation, which could complement other agricultural approaches to contribute to sustainable intensification on farms. Organic agriculture comprises 0.8% of North American agriculture (Willer & Lernoud, 2016). In much of Latin America, agricultural fields are still managed by small farmers, despite rapid increases in industrial agriculture. Many of them practice diversified agriculture, using hand or animal power and zero or little agricultural chemicals, preserving soils and biodiversity while supplying much of the food for their countries. Networks like *Campesino a Campesino* (Farmer to Farmer) further Agroecology – the science of sustainable agriculture - by promoting exchanges of traditional knowledge and experience among farmers. Perhaps the most famous example of small-scale farmer success is Cuba. Following the Soviet Union collapse in the 1990s and the USA embargo, food production in Cuba collapsed with the loss of imported fertilizers, pesticides, tractors, parts, and petroleum. Cubans developed alternative methods of growing food. Sustainable agriculture, organic farming, urban gardens, smaller farms, animal traction, and biological pest control all became part of Cuban agriculture. They were so successful that from 1996 to 2005 Cuba sustained a 4.2% growth in per capita food production. In southern Brazil in 2008 - 2009, conventional maize farmers lost 50% of their crops in a severe drought, but farmers who followed agroecological systems lost just 20% of their maize. In Honduras, soil conservation practices introduced via *Campesino a Campesino* helped triple or quadruple the yields of hillside farmers. Many other examples of successful agroecology exist (Altieri *et al.,* 2012; Altieri & Funes-Monzote, 2012).

North America

Atlantic and Pacific Ocean waters are more acidic since 1991, except for the subpolar Pacific (Lauvset *et al.,* 2015; Ríos *et al.,* 2015; Feeley *et al.,* 2012). Arctic Ocean pH trends are not significant, but undersaturation with calcium minerals, colder waters that absorb more carbon dioxide, and low-alkaline freshwater inputs from rivers and melting sea ice, contribute to North American Arctic Ocean vulnerability to ocean acidification, including the Pacific Arctic, home to one of the world's largest commercial and subsistence fisheries (Steiner *et al.,* 2014; Mathis *et al.,* 2015). Large areas off the USA Pacific coast are now acidic enough to dissolve the shells of free-swimming snails (sea butterflies/pteropods), which are important in ocean food webs (Bednaršek *et al.,* 2016). Cod larvae are highly sensitive to ocean acidification (Frommel *et al.,* 2012).

In the USA ozone pollution from fossil fuel combustion increases human morbidity and mortality (Li *et al.,* 2016). Springtime ozone levels are increasing in North America, which may in part be attributable to Asia (Cooper *et al.,* 2010; Law, 2010). Emissions from motor vehicles and other fossil fuel combustion are large contributors to atmospheric fine particulate matter (Lee *et al.,* 2003). Particulate matter is associated with premature mortality and lung cancer (Apte *et al.,* 2015). In the USA increased infant mortality from respiratory complications, increasing the odds of sudden infant death syndrome by 25% in some studies (Woodruff *et al.,* 1997; Son *et al.,* 2017). Even where air meets USA standards, rates of low human birthweights increase with increasing air particulate matter (Ebisu & Bell, 2012; Hao *et al.,* 2016). Regulations to reduce industrial and other particulate matter release to the atmosphere since the 1970s improved life expectancies in the USA (Pope *et al.,* 2009).

Since nitrogen fertilizer production from atmospheric nitrogen gas began with the Haber-Bosch process in the early 1900s, inorganic nitrogen fertilizer use across the USA has increased (Erisman *et al.,* 2008). Agricultural fertilizers, nitrogen deposition and nitrogen-fixing crops dominate reactive nitrogen sources, with limited areas driven by centralized sewage (point sources), manure application or urban run-off **(Box 4.17)**. Ammonia emissions, mainly from fertilizer use, increased 9% in Canada from 1995-2000 (Schindler *et al.,* 2006). Where oil is extracted from oil sands in North American prairie grasslands, nitrogen oxides and Sulfur emissions are increasing (McLinden *et al.,* 2015). In the eastern USA, power plant upgrades through Clean Air Act regulations since the 1970s reduced Sulfur and nitrogen oxides deposition (though ammonia levels are increasing) (Li *et al.,* 2015), reducing acidification of acid-sensitive lakes and rivers (Garmo *et al.,* 2014). Recently low natural gas prices caused USA power plants to use less coal, reducing emissions of carbon dioxide (by ~23%), nitrogen oxides and sulfur dioxide (de Gouw *et al.,* 2014). Natural gas is a potent greenhouse gas, however; leaks during its extraction, transportation and storage must be minimized (Howarth, 2014; Zimmerle *et al.,* 2015).

Both nitrogen and sulfur atmospheric deposition can affect growth, species composition, biodiversity and ecosystem function in temperate and boreal forests of North America (Pardo *et al.,* 2011). Nitrogen deposition's clearest impact on species is to reduce lichen and mycorrhizal diversity. They respond quickly to changes in nitrogen availability. Where soils lack minerals to neutralize acidic inputs, sulfur deposition has acidified soils, decreasing tree growth and health, and acidified runoff to aquatic ecosystems, affecting aquatic species. Atmospheric nitrogen and Sulfur deposition is also reducing diversity and increasing fire risk in some temperate grasslands and deserts of North America (Pardo *et al.,* 2011), and it can alter diversity and ecosystem function in wetlands and freshwater systems that are naturally low in nitrogen. Nitrogen deposition may be responsible for declines in endangered species in some areas of the USA (Hernández *et al.,* 2016).

In the USA from 1992 to 2011, pesticide concentrations exceeded aquatic-life benchmarks in many rivers and streams in agricultural, urban, and mixed-land use watersheds. The proportions of assessed streams with one or more pesticides that exceeded an aquatic-life benchmark were very similar between the two decades for agricultural (69% during 1992–2001 versus 61% during 2002–2011) and mixed-land-use streams (45% versus 46%). Urban streams, in contrast, increased from 53% during 1992–2011 to 90% during 2002–2011, largely because of fipronil and dichlorvos. The potential for adverse effects on aquatic life is likely greater than these results indicate, because potentially important pesticide compounds were not assessed. Widespread trends in pesticide concentrations, some downward and some upward, occurred in response to shifts in use patterns primarily driven by regulatory changes and new pesticide introductions (Stone *et al.,* 2014).

In the USA agricultural use of glyphosate [N-(phosphonomethyl) glycine] has increased from less than 10,000 to more than 70,000 metric tons per year from 1993 to 2006 (active ingredient), primarily due to the introduction of genetically modified crops, particuallry corn and soybean, and is still increasing. In 2009, glyphosate accounted for >80 percent of all herbicide use on more than 31 million hectares of soybean (by weight of active ingredient). On 31.1 million hectares of corn, glyphosate accounted for about a third of herbicide use (Coupe & Capel, 2016). Glyphosate is also used in homes, and along rights of way. Glyphosate was considered more "environmentally benign" than herbicides it replaced because it has lower toxicity and mobility or environmental persistence. However, results from >2,000 samples across the USA indicate that glyphosate is more mobile and occurs more widely in the environment than was thought. Glyphosate and aminomethylphosphonic acid (a glyphosate degradation product) were detected in surface water, groundwater, rainfall, soil water, and soil, at concentrations from <0.1 to >100 micrograms per liter. Most concentrations were below adverse effects criteria, however, the effects of chronic low-level exposures to mixtures of pesticides are uncertain. Studies have attributed toxic effects to surfactants or other additives to common glyphosate formulations.

New classes of pesticides have been developed and introduced and are now widely used, but have documented environmental issues such as the persistent, systemic and neurotoxic neonicotinoids and fipronil, introduced in the early 1990s. Insecticide use has been related to the disappearance of honey bees and other insects and insect eating birds. Neonicotinoids and fipronil are found in nectar and pollen of treated crops such as maize, oilseed rape and sunflower and also in flowers of wild plants growing in farmland. They have also been detected at much higher concentrations in guttation drops exuded by many crops (van Lexmond *et al.,* 2015).

The Laurentian Great Lakes and Greenland illustrate aspects of persistent organic pollutants in North America (**Box 4.18** and **Box 4.19**). Persistent organic pollutants concentrations in air and fish samples in the North American Great Lakes and in some Arctic Ocean biota have slowly declined in recent decades. Polycyclic aromatic hydrocarbons decreases are from improved emissions controls (Carlson *et al.,* 2010; Venier & Hites, 2010). Since their ban, levels of polybrominated biphenyl ethers, used as fire retardants, have declined in fish, bivalves and bird eggs in San Francisco Bay (Sutton *et al.,* 2014). Persistent organic pollutants persist, however, and new ones are emerging. Across North America, polychlorinated biphenyls in air samples increase along a remote-rural-urban gradient. Lighter congeners are more common at higher latitudes. Polychlorinated biphenyls loadings have not declined in the Canadian Arctic, as heavier polychlorinated biphenyls are moving northwards more slowly. For polybrominated diphenyl ethers, and other emerging persistent organic pollutants, few trends have emerged (Shen *et al.,* 2006; Braune *et al.,* 2005; Macdonald *et al.,* 2000).

In North America, fish mercury levels, even in remote places, are often unsafe for regular consumption by humans and wildlife in North America (Driscoll *et al.,* 2007). Decreased reproduction in common loons, which are fish-eating birds, is correlated with female tissue mercury levels (Evers *et al.,* 2008). In contaminated areas where fish consumption is high, human populations are at risk (Mahaffey & Mergler, 1998; Cole *et al.,* 2004). Industrialization increased atmospheric mercury loads to remote northern lakes in North America (Swain *et al.,* 1992; Driscoll *et al.,* 2007; Fitzgerald *et al.,* 1998; Durnford *et al.,* 2010). Decreases in USA coal combustion, and environmental regulations, have reduced mercury loads to the eastern and midwestern USA have decreased, reducing mercury levels in the environment and fish (Engstrom & Swain, 1997; Evers *et al.,* 2007; Munthe *et al.,* 2007; Cross *et al.,* 2015). However, the decrease in atmospheric mercury deposition in the USA has slowed, particularly in the western and central USA, which is attributed to mercury deposition from elsewhere, possibly China (Weiss-Penzias *et al.,* 2015). In Arctic North America, mercury levels in seabird eggs and feathers, marine mammals and lake sediments are increasing (Braune *et al.,* 2005). Emissions from Asia account for one-third of atmospheric mercury there (Durnford *et al.,* 2010). Total mercury emissions from China increased by about 3% per year from 1995 to 2003, mostly from increasing coal burning and non-ferrous metal smelting (Wu *et al.,* 2006). The mercury burden in the Arctic marine food web is now 92% from man-made sources (Dietz *et al.,* 2009), increasing an order-of-magnitude since the industrial revolution and accelerating in the 20th century. It may now cause subtle neurological or other toxic effects in many fish-eating Arctic wildlife, including Arctic toothed whales, polar bears, pilot whales, hooded seal, some bird species and landlocked Arctic char (Dietz *et al.,* 2009). The effects of multiple pollutants, including persistent organic pollutants and mercury, are a concern among Arctic indigenous groups that frequently consume fish, marine mammals or sea bird eggs, particularly where local persistent organic pollutants sources add to background atmospheric burdens (Burger *et al.,* 2007; Hardell *et al.,* 2010; Hoover *et al.,* 2012; Byrne *et al.,* 2015). Lead contamination has also reached the Arctic from coal combustion (McConnell & Edwards, 2008). Després *et al.* (2005) detected correlations among tremor amplitude or other neuromotor effects and blood mercury or lead, in Inuit children in Canada. Although fish consumption increases human blood lipids that reduce cardiovascular risk and increase cognition, mercury exposure diminishes these advantages and increases cardiovascular disease indicators (Virtanen *et al.,* 2005; Oken *et al.,* 2005; Guallar *et al.,* 2002).

Pollution from past and ongoing coal mining, hard-rock mining, and metal-ore smelting, expose humans, fish and wildlife to toxicants (e.g. toxic metals and selenium) across North America; thousands of mines are abandoned, and bankrupcies of mining companies are common, leaving neither public nor private funds available to to mitigate or restore these sites and allowing toxic releases and exposures to continue (Woody *et al.,* 2010; Palmer *et al.,* 2010; Lewis *et al.,* 2017; Gorokhovich *et al.,* 2003; Clements *et al.,* 2000; Maret & MacCoy, 2002; Maret *et al.,* 2003; Dudka & Adriano, 1997; Lovingood *et al.,* 2004; Surber & Simonton, 2017; Hughes *et al.,* 2016). Near past lead mining and smelting operations, ground-feeding songbirds are exposed to lead at toxic concentrations (Beyer *et al.,* 2013). The costs to contain pollution from hard rock mining sites in the USA have spiraled upwards from tens of billions of dollars in 1993 (Lyon *et al.,* 1993) to $75 to $240 billion today (Hughes *et al.,* 2016).

Mesoamerica

Basin-wide acidification is increasing in oceans surrounding Mesoamerica, with pH decreasing from 1991-2011 (Lauvset *et al.,* 2015; Bates *et al.,* 2014). If increases in atmospheric carbon dioxide continue, many Pacific coral reef systems may no longer be viable (Feely *et al.,* 2012). As for nitrogen deposition, studies in Mesoamerica suggest it could affect tropical forest composition by increasing soil nitrate levels that could then alter the competitive ability of nitrogen-fixing legumes or alter soil cation exchange capacity, making nutrients like calcium or potassium scarcer (Sayer *et al.,* 2012; Hietz *et al.,* 2011).

There are no systematic studies of agricultural chemicals in the Mesoamerica, but it appears that pesticides are frequently found in the environment. For example, glyphosate is the most commonly used pesticide in Mexico, and it was detected in water from all 23 locations sampled in one study, including protected and agricultural areas, and was higher during the dry season (up to 36.7 ug/L) (Ruiz-Toledo *et al.,* 2014).

Pesticide use in Costa Rica more than quadrupled from 1977 to 2006, from approximately 2,650 metric tons of active ingredient to 11,600. In a study from late 2005 to 2006, pesticides were measured invarious media throughout Costa Rica (Shunthirasingham *et al.,* 2011). Because of the variety of crops grown in Costa Rica (coffee, bananas, rice, and sugar cane) many different pesticides are used and were detected in this program, including some from fog and air samples in remote areas.

In Mesoamerica, past rather than current use appears to drive organochlorine pesticides contamination. A Costa Rican location with limited past organochlorine pesticides use has low air and soil organochlorine pesticides levels (Daly *et al.,* 2007; UNEP, 2009). Air and soil from four Mesoamerican sites had low polychlorinated biphenyls and polybrominated diphenyl ethers levels (Shen *et al.,* 2006), but in Mexican communities where past agricultural and antimalarial DDT (dichlorodiphenyltrichloroethane) use was high, human exposure to DDT components and dichlorodiphenyldichloroethylene is high. Children had polychlorinated biphenyls in their blood. Risk assessments should consider multiple persistent organic pollutants exposures. Metal mining concessions cover 28% of Mexico and 8% of Mexico's protected land (Armendariz-Villegas *et al.,* 2015). Limited studies suggest that mercury levels are not elevated in sharks and rays; freshwater and marine forage fish for migratory aquatic raptors; or Pacific coastal water and sediment (Sandoval-Herrera *et al.,* 2016; Gutierrez-Galindo *et al.,* 2007; Elliot *et al.,* 2015). Soils at former mining sites in Mexico have high mercury levels. (Santos-Santos *et al.,* 2006). Though mercury may be stable in some soils (Gavilán-Garía *et al.,* 2008), it is most toxic when methylated in wet environments, warranting surveys of mercury contamination in nearby waters. Artisanal mining still releases mercury to aquatic environments in Mesoamerica.

Caribbean

Worth almost $2 billion in 2003, the annual net benefit from Caribbean island coral reefs, excluding USA reefs, was more than the GDP of some eastern Caribbean island nations. The difference between the income they generate and their maintenance cost was almost $50 billion (Cesar *et al.,* 2003). Forest cover increases on Caribbean islands (section 4.4.1) should reduce sedimentation to coral reefs, but concurrent urbanization could offset those benefits (Ramos-Scharron *et al.,* 2015). Ocean acidification, pollution from human sewage, other nutrient pollution sources, sedimentation and temperature increases all contribute to Caribbean coral reef declines **(Box 4.11)**. In addition, decreases in aragonite (calcium carbonate) saturation levels across the region **(Figure 4.8)** (Gledhill *et al.,* 2008) due to acidification damages coral reef structure (Webster *et al.,* 2013).

Few studies examine nutrient and sediment in Caribbean rivers and streams, but Puerto Rico provides an example. Beginning in the 1800s, land clearing for agriculture and urban development increased sediment and nutrient fluxes to coral reefs. A study examining sediment flux from different land uses (forest, pasture, cropland, and urban) showed that the sediment flux was higher on disturbed land and depended on the storm hydrograph, previous storms, location in the watershed, and underlying geology (Gellis, 2013). Despite much reforestation since the mid-1940s, sediment transported to river valleys from previous agriculture is still being transported through river systems. Nitrogen and phosphorous concentrations in river waters are within regulatory limits but up to 10 times higer than estimated pre-settlement levels, negatively affecting coral reefs, especially near shores. Nitrogen deposition in in Puerto Rico was associated with more soil nitrates (Cusak *et al.,* 2015). Other anthropogenic sources of nitrogen to Caribbean ecosystems come from reforestation with molecular nitrogen-fixing trees, including exotic species (Erickson *et al.,* 2015).

Caribbean island cloud forests and biota can have high mercury levels (Townsend *et al.,* 2013), suggesting that global atmospheric mercury burdens are affecting them, given that these forests are cooler, wetter and intercept fog. Caribbean cloud forest soils are often waterlogged (Silver *et al.,* 1999), which could spur mercury methylation. As in Mesoamerica, past use of legacy organochlorine pesticides is associated with high concentrations in streams, coastal environments and biota. Past chlordecone use in Martinique and Guadeloupe is associated with current concentrations in freshwater and coastal ecosystems (Coat *et al.,* 2006, 2011; Charlotte *et al.,* 2016). Low-level chronic exposure of developing infants and infants to chlordecone negatively impacts infant cognitive and motor developments in Guadeloupe (Dallaire *et al.,* 2012).

South America

Ocean acidification is increasing around South America; pH decreased from 1991-2011 in the southern and equatorial Atlantic and Pacific Oceans and the subpolar southern Ocean (Lauvset *et al.,* 2015). Southern Ocean systems are highly vulnerable to ocean acidification. Colder waters hold more carbon dioxide and dissolve more calcium carbonate.

Box 4 11 Regional flattening of Caribbean Sea coral reefs.

All four subregions of the Americas border the Caribbean Sea. Caribbean coral reefs have undergone a region-wide "flattening", in which an objective measure of their structural complexity, their "rugosity", which is directly related to their species diversity (Newman *et al.,* 2015) greatly decreased from 1969 to 2008 (Álvarez-Filip *et al.,* 2009) (see **Figure 4.7**). Caribbean reefs are among the marine ecosystems most impacted by humans (Halpern *et al.,* 2008). Globally, Caribbean coral reefs have the most critically endangered species as a proportion of total species (Carpenter *et al.,* 2008). Models suggest that ocean acidification and warming alone are enough to cause widespread coral mortality and reduced growth (Anthony *et al.,* 2011). Further, overfishing that reduces populations of the fish that graze sponges or algae can also degrade Caribbean reefs (Anthony *et al.,* 2011; Loh *et al.,* 2015), and these same models suggest overfishing of the fish that eat algae or elevated nutrient levels will lessen coral reef resilience to ocean acidification or warming (Anthony *et al.,* 2011). Caribbean coral reefs are subject to a variety of other stressors that reduce reef resistance to acidification (Anthony *et al.,* 2011; Woodridge & Done, 2009). Pollution sources include sedimentation, which represents a severe disturbance (Fabricius, 2005), and nutrient-laden runoff including sewage. Reefs are exposed to elevated nitrogen from runoff and discharges off the coast of Mexico when tourist numbers are higher (Sanchez *et al.,* 2013). In experiments, nitrogen enrichment decreases calcification rates including for at least two dominant reef-building Caribbean corals species, and likely contributes to coral overgrowth by algae (Marubini & Davies, 1996; Fabricius, 2005). Various diseases are also devastating Caribbean reefs (Sutherland *et al.,* 2004; Carpenter *et al.,* 2008), including one that rapidly spreads and kills a primary reef building species in the Caribbean, Elkhorn coral (*Acropora palmata*) and that is linked to human sewage (Patterson *et al.,* 2002; Sutherland *et al.,* 2010).

Acidification in the greater Caribbean Sea is demonstrated by a clear long-term decrease in pH and an increase in surface water dissolved carbon dioxide between 1996 and 2016 (see Bates *et al.,* 2014; Astor *et al.,* 2013) and a strong decrease in aragonite (calcium carbonate) saturation levels across the region **(Figure 4.8)** (Gledhill *et al.,* 2008). Decreases in aragonite saturation due to acidification can inhibit maintenance and recovery of coral reef structure (Webster *et al.,* 2013), and for coral reefs to remain in coastal Caribbean areas, they will have to recover from local- to large-scale physical and other disturbances like those from hurricanes or coral bleaching (Goreau, 1992; Carpenter *et al.,* 2008), both of which can kill coral, and from ocean warming (Yee *et al.,* 2008; Pandolfi *et al.,* 2011), which leads to bleaching. Increasing atmospheric carbon dioxide depresses metabolism, settlement and growth of larvae of the important Caribbean reef-building species *Porites astreoides* (mustard hill coral) (Albright & Langdon, 2011). Related *Porites sp.* of the Indo-Pacific show declining calcification rates over the past 16 years, and Cooper *et al.* (2008) attribute this change to ocean acidification. Experiments with other Caribbean species, like the reef urchin (*Echinometra viridis*), also show impaired calcification of Caribbean reef species (Courtney *et al.,* 2013).

Species critical to the pelagic or benthic southern Ocean food web, including Antarctic krill (*Euphausia superba*), some pteropods, and benthic marine invertebrates, could collapse from ocean acidification alone, ignoring temperature changes (Kawaguchi *et al.,* 2013; McNeil & Matear, 2008; McClintoc *et al.,* 2009). Experiments show that species from subtropical southern Pacific Ocean waters are vulnerable to ocean acidification (Vargas *et al.,* 2015). Upwelling, rainfall, tides and river flows (Vargas *et al.,* 2016; Manzello, 2010) affect seawater carbon dioxide levels, upwelling around the Galapagos Islands, cause high carbon dioxide levels and low calcium carbonate, places its waters near the distributional limits for coral reefs, making them particularly vulnerable to ocean acidification (Manzello, 2010).

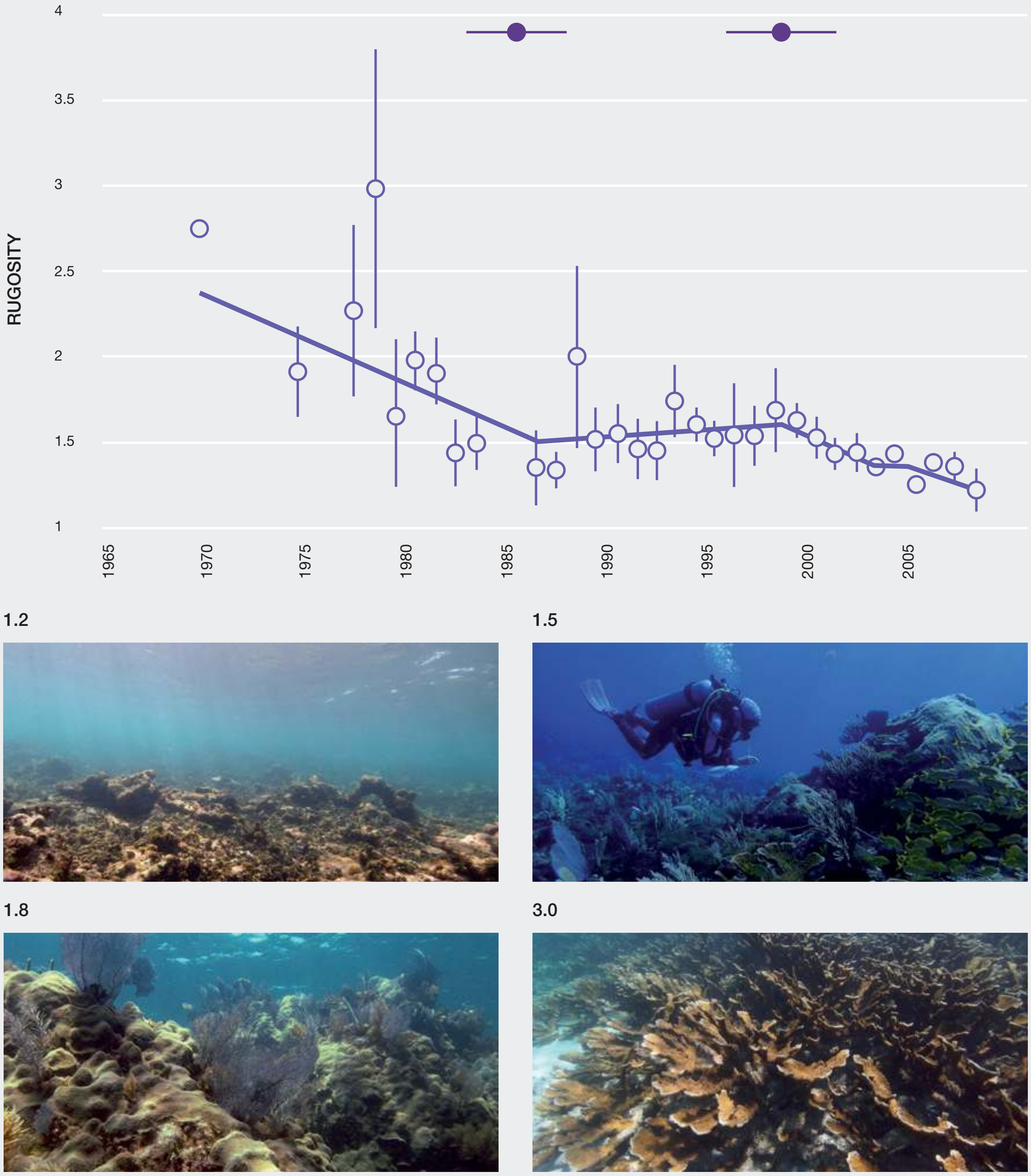
The worldwide need for food and increased rainfall as led to agricultural expansion and change over recent decades in South America. Rapid adoption of genetically modified crops has occurred, particularly glyphosate tolerant soybean and corn and Bt-corn and cotton (De la Casa & Ovando, 2014; Brookes & Barfoot, 2011). Between 1996 and 2009, the area planted to soybeans in Argentina increased by 215% (from 5.9 to 18.6 million hectares) (Lapola *et al.,* 2014). Agriculture has intensified over the same period, with one field producing two to three crops per year. Water-quality degradation in Brazilian rivers is directly proportional to agricultural extent in watersheds and riparian zones.

There are no systemic studies of agricultural chemicals in the South American environment, but given the large use of glyphosate on genetically modified soybean it can be assumed that conditions are similar to the USA where glyphosate can be found in every environmental compartment (Coupe *et al.,* 2012; Battaglin *et al.,* 2014; Rios *et al.,* 2010).

Total dissolved nitrogen yields in major South American rivers, including the Río de la Plata and Amazon, are less than many major world rivers. Rivers with the highest total dissolved nitrogen yields in South America pass through heavily populated areas - they lack of municipal and industrial treatment plants. Rivers impacted by agriculture have lower total dissolved nitrogen yields. Water pollution in South America is dominated by municipal and industrial sewage (Bustamente *et al.,* 2015). In all countries of the Amazon and Orinoco River basins, wetlands and major rivers show pollutant impacts on biodiversity (Crema *et al.,* 2011; Gomez-Salazar *et al.,* 2012; Lopes & Piedade,

Figure 4 7 **Changes in reef rugosity on reefs across the Caribbean from 1969 to 2008 with examples of four different values of rugosity index of architectural complexity on Caribbean reefs.**

Black dots at the top of the figure indicate the significant breakpoint in 1985 and 1998 (+1 s.e.) for the segmented regression. Source: Álvarez-Filip *et al.* (2009). Photos courtesy of Lorenzo Álvarez-Filip.



1.2

1.5

1.8

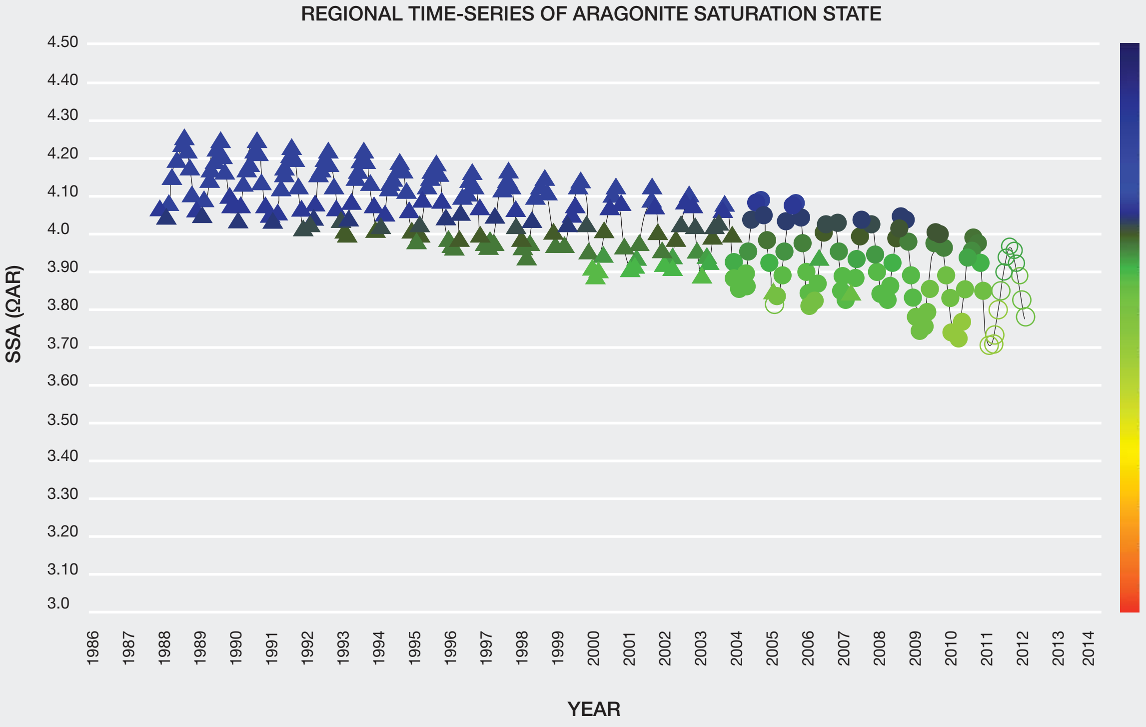
3.0

2014). Where Amazonian wetlands (forested floodplains, marshes, wet meadows, peatlands, tidal wetlands, etc.) are densely populated, conversion to agriculture, accompanied by fertilizer organic matter loads, cause super or even hypereutrophic areas in the mid-lower course of the Amazonas River (Affonso *et al.,* 2011). Increased nitrogen availability from agriculture, mining, sewage pollution, shrimp farming and solid waste disposal threaten South American mangroves (Lacerda *et al.,* 2002; Castellanos-Gallindo *et al.,* 2014; Rodríguez-Rodríguez *et al.,* 2016) (supplementary material: **Box 4.20**).

Petroleum drilling is increasing in the Amazon; repeated spills contaminate water, sediment and soils with toxic

Figure 4 8 **Trends in mean monthly sea-surface aragonite saturation (SSA) (aragonite is a form of $CaCO_3$) in the Caribbean sea since the late 1980s.**

Declines in SSA are related to declines in coral reef accretion. Sources: Gledhill *et al.* (2008), figure updated data downloaded from: https://coralreefwatch.noaa.gov/satellite/oa/saturationState_GCR.php.



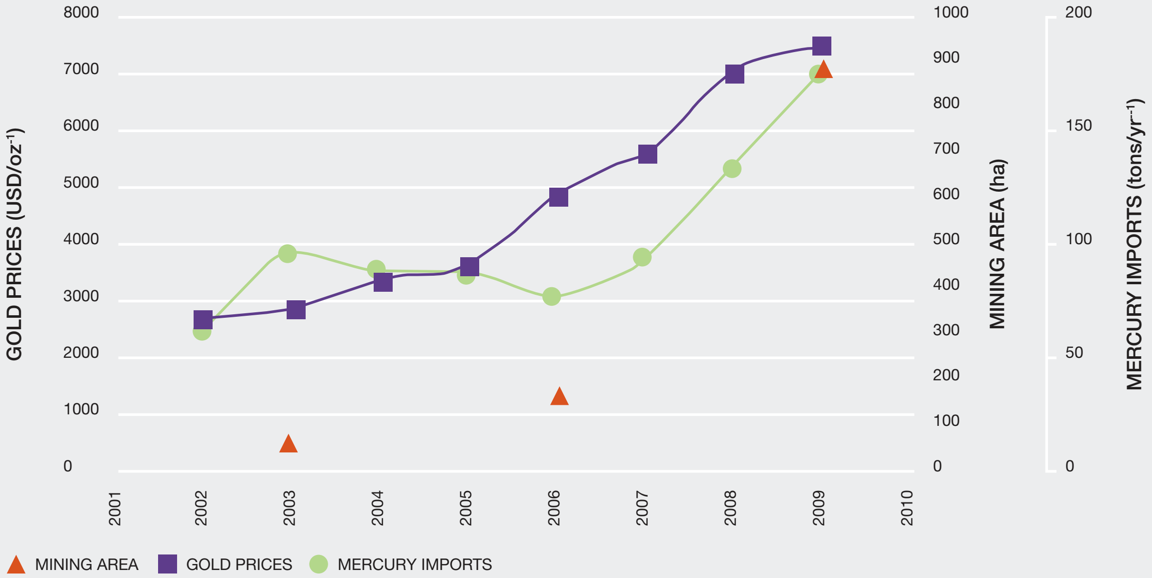
hydrocarbons or metals (Frazer, 2016; Marínez *et al.,* 2007) in many indigenous communities. This income source is also a public health concern: childhood leukemia and spontaneous abortion are higher among people living near oil drilling, and stream water exceeds allowable limits for petroleum hydrocarbons (San Sebastián & Hurtig, 2004; San Sebastián *et al.,* 2002). Despite such concerns, little related research is available (Orta Martínez *et al.,* 2007; Orta-Martínez & Finer, 2010), but water and sediment near oil-related activities can be highly contaminated with polycyclic aromatic hydrocarbons and mutagenic (Reátegui-Zirena *et al.,* 2013), and drilling fluids have high toxic metal concentrations. Oil exploration is a source of spills that affect wetlands (Lopes & Piedade, 2014). In general, metal-polycyclic aromatic hydrocarbons mixtures have a more than additive toxicity effect on aquatic invertebrates (Gauthier *et al.,* 2014). Oil and dispersants are toxic to Amazonian fish (Pinto *et al.,* 2013). As of 2008, around180 concessions for oil exploration or extraction, involving ≥35 companies, cover much of the most species-rich part of the Amazon (Finer *et al.,* 2008), subjecting the area to pollution and opening it to deforestation and hunting (Butt *et al.,* 2013).

Amazonian countries are large and increasing sources of mercury emissions from artisanal gold mining (Telmer & Veiga, 2009). Mining area correlates with gold prices (Swenson *et al.,* 2011) **(Figure 4.9)**. Although some mercury leaches from soils (Fadini & Jardim, 2001), most mercury contamination is anthropogenic, and seasonal flooding disperses it. Higher mercury concentrations occur downstream from mining sites in fish, sediment and humans (Malm, 1998; Mol *et al.,* 2001; Cordy *et al.,* 2011; Fujimura *et al.,* 2012). Its adverse effects on vertebrate embryos and the human nervous system are well known (e.g. Passos & Mergler, 2008).

In South America also, higher legacy of persistent organic pollutants levels occur where past use was high. In a Patagonian watershed of Argentina, river water, sediments and wetland soils had higher polychlorinated biphenyls and organochlorine pesticides concentrations near agriculture, urban areas and hydroelectric facilities (Miglioranza *et al.,* 2013), and raptors may have high organochlorine pesticides levels (Martínez-López *et al.,* 2015). In coastal areas, a protected estuary receiving sediment from nearby urban and industrial areas had high polychlorinated biphenyls concentrations (Pozo *et al.,* 2013). Like the Arctic, long-range transport of polychlorinated biphenyls is still increasing in remote mountain lakes in Chile (Pozo *et al.,* 2007).

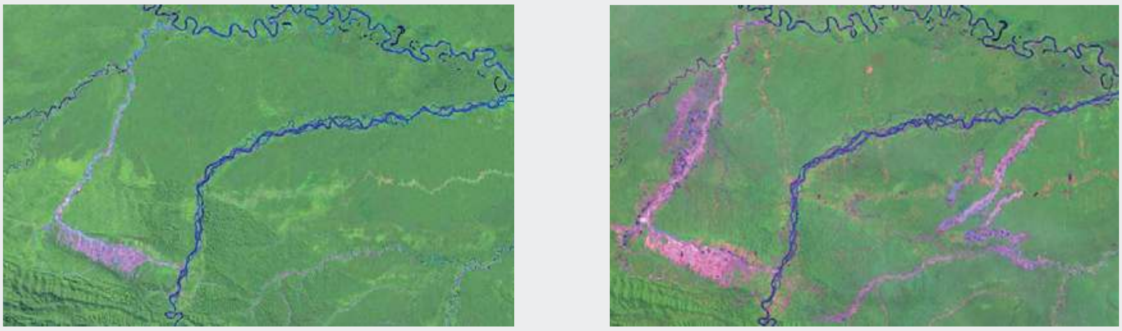
Figure ④ ⑨ **Relationships among gold prices, mercury imports to Peru, and forest clearing for gold mining from 2000 to 2016 in a region of Peru. Source: Swenson *et al*. (2011); USA Geological Survey.**

A. From 2000 to 2009



B. 2000

C. 2016



In air samples from the Cauca valley of Colombia, higher persistent organic pollutants compared with other places in Latin America are presumably associated with the extensive urban and agricultural areas (Álvarez *et al.* 2016). Sediment cores from the Santos estuary of Brazil show that polycyclic aromatic hydrocarbons increased over time with development (Martins *et al.,* 2011).

4.4.3 Climate Change

Nature of the driver, its recent status and trends, and what influences its intensity

Climate change is defined as "Any change of climate which is attributed directly or indirectly to human activity that alters the composition of the global atmosphere greenhouse gases (carbon dioxide, methane, methane and nitrous oxide) over comparable time periods." (IPCC, 2013).

Earth's climate, as well as the atmospheric greenhouse gases of its atmosphere, has changed throughout its history. During the pre-industrial period, the ice core shows that the greenhouse gases concentration stayed within well-defined natural limits with a maximum concentration of approximately 300 parts per million, 800 parts per billion and 300 parts per billion for carbon dioxide, methane and nitrous oxide, respectively, and a minimum concentration of approximately 180 parts per million, 350 parts per billion and 200 parts per million.

The last report of the Intergovernmental Panel on Climate Change (IPCC) (IPCC, 2014a) indicates that greenhouse gasses from anthropogenic sources have significantly increased since the pre-industrial era because of economic and population growth. This has led to atmospheric concentrations of carbon dioxide, methane and nitrous oxide that are unprecedented in at least the last 800,000 years. The IPCC reports that this significant increase in greenhouse gasses has caused a warming of 0.85ºC on average globally (land and ocean surface combined) over the period 1880 to 2012. The most recent report of the World Meteorological Organization stated that the warming has now exceeded 1ºC.

As shown in **Figure 4.10**, the economic sectors that contributes the most to greenhouse gasses are the electricity and heat production sector, agriculture, forestry and other land use, the industry sector, and the transport sector (emissions are converted into carbon dioxide-equivalents based on Global Warming Potential (100) from the IPCC Second Assessment Report) (IPCC, WGIII, 2014).

The IPCC developed the representative concentration pathways (RCPs) as a way of projecting how factors like population size, economic activity, lifestyle, energy use, land use patterns, technology and climate policy, will have an impact in the concentration of atmospheric greenhouse gasses. There are four RCPs: a stringent mitigation scenario (RCP2.6) (this scenario is based on the goal of maintaining global warming below 2ºC above pre-industrial temperatures), two intermediate scenarios (RCP4.5 and RCP6.0) and one scenario with very high greenhouse gasses emissions (RCP8.5) (IPCCC, 2014b).

The IPCC (Stocker *et al.,* 2013) reported that in all of these scenarios, except RCP2.6, global surface temperature change for the end of the 21st century is likely to exceed 1.5 °C relative to 1850 to 1900. Furthermore, under two scenarios (RCP6.0 and RCP8.5) it is likely that global surface temperature change will exceed 2°C (the upper limit of the goal of the Paris Agreement), and more likely than not to exceed 2°C for RCP4.5. (IPCC, 2013).

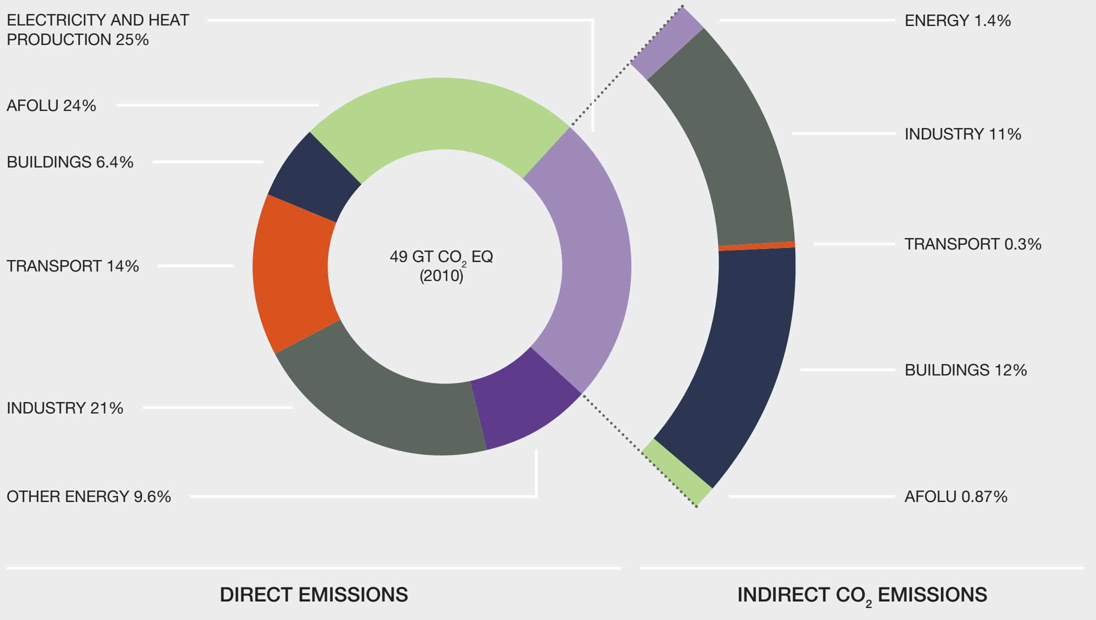
Mean surface temperatures for 2081-2100 relative to 1986-2005 is likely to increase in the following ranges for each scenario: 0.3°C to 1.7°C (RCP2.6), 1.1°C to 2.6°C (RCP4.5), 1.4°C to 3.1°C (RCP6.0), 2.6°C to 4.8°C (RCP8.5) (IPCC, 2013).

Moreover, it is very likely that heat waves will occur more often and last longer and that extreme precipitation events, both floods and droughts, will become more intense and frequent in many regions (IPCC, 2013).

The ocean will continue to warm. In the top hundred meters, ocean warming is expected to be about 0.6°C (RCP2.6) to 2.0°C (RCP8.5), and about 0.3°C (RCP2.6) to 0.6°C (RCP8.5) at a depth of about 1,000 meters by the end of the 21st century (IPCC, 2013).

Global mean sea level will continue to rise during the 21st century, with the rate of rise very likely exceeding that observed during 1971 to 2010 due to increased ocean warming and increased loss of mass from glaciers and ice sheets. Sea level rise for 2081–2100 relative to

Figure 4 10 **Greenhouse gas emissions by economic sectors. Source: IPCC (2014).**



1986–2005 will likely be in the ranges of 0.26 to 0.55 meters for RCP2.6, 0.32 to 0.63 meters for RCP4.5, 0.33 to 0.63 meters for RCP6.0, and 0.45 to 0.82 meters for RCP8.5. For RCP8.5, the rise by the year 2100 is 0.52 to 0.98 meters, with a rate during 2081 to 2100 of 8 to 16 millimeters per year (IPCC, 2013).

Biodiversity is impacted significantly by climate change in a wide range of ways and scales (i.e. ecosystems, species, genes). Scheffers *et al.* (2016) identified a set of 32 core terrestrial ecological processes and 31 each in marine and freshwater ecosystems that supports ecosystem functions and its capability in providing benefits to people. From this set of 94 processes, the authors state that 82% show evidence of impact from climate change like shifts in species ranges, changes in phenology and population dynamics, and disruptions from the gene to the ecosystem scale (Scheffers *et al.,* 2016) **(Figure 4.11)**.

In order to illustrate the impact of climate change on biodiversity, the following is a summary based on the findings of the last report of the IPCC on impacts, adaptation and vulnerability. In general, many terrestrial, freshwater, and marine species have shifted their geographic ranges, seasonal activities, migration patterns, abundances, and species interactions in response to climate change (IPCC, 2014a).

Certain naturally occurring factors, like the El Niño Southern Oscillation, have the potential to exacerbate the effects that climate change is already having in many parts of the Americas region. The El Niño Southern Oscillation warming and cooling phases (i.e., El Niño and La Niña, respectively) are known to predictably alter precipitation and temperature patterns both spatially and temporally throughout the region. Between December and January, El Niño generally causes wetter conditions in southwestern portions of North America

Figure 4 11 **Ecological process in terrestrial, marine and freshwater ecosystems that will be impacted by climate change.**

Impacts are measured on multiple processes at different levels of biological organization within ecosystems (i.e. organism, species, population and community). Source: Scheffers *et al.* (2016).

(northwestern Mexico and southwestern USA), northwestern portions of South America (Colombia, Ecuador, and Peru), drier conditions in the Amazon basin, and warmer conditions in southeastern Brazil and the northeastern and northwestern portions of North America (Lindsey, 2016). Between June and August, El Niño can be associated with drier and warmer conditions in Central America, wetter conditions in central Chile and the northwestern USA, and warmer conditions on the east and west coasts of central South America (Lindsey, 2016). Consequently, areas experiencing drier conditions as a result of climate change, like the tropical dry forest in Central America (Fuentes-Franco *et al.,* 2015), may experience intensified conditions during El Niño years.

Extreme weather events, like coastal storms, can intensify the effects that climate change-related sea-level rise is already having on many coastal areas. Specifically, coastal regions that exist in low-lying areas and are already experiencing inundation from sea-level rise are especially vulnerable to storm surge from tropical cyclones (i.e. hurricanes, typhoons), which increases flooding and land subsidence (Yang *et al.,* 2014). Areas in the Americas region that are particularly susceptible to both sea-level rise and tropical cyclones include coastlines and island nations/territories in the Caribbean Sea, Gulf of Mexico, north Atlantic Ocean (along the southeastern coast of the USA), and northeast Pacific Ocean (along the western coast of Mexico).

Terrestrial and freshwater ecosystems

Under all the RCP scenarios, the extinction risk of a large fraction of terrestrial and freshwater species by climate change in the 21st century and beyond is increased by the interaction of other drivers of biodiversity loss like pollution, habitat modification, over exploitation, and invasive species. These ecosystems will be at risk of abrupt and irreversible regional-scale change in the composition, structure, and function under medium- to high-emissions scenarios.

Climate changes exceeding those projected under RCP2.6 in high-altitude and high-latitude ecosystems will lead to significant changes in species distributions and ecosystems function. The increase in water temperature due to global warming will lead to shifts in freshwater species distributions.

For the second half of the 21st century, all the RCP scenarios indicate that the composition of communities will change due to a change (decrease or increase) in abundance of some species, and that the seasonal activity of many species will change differentially, causing the disruption of life cycles and interactions between species. In addition, human health will be affected as a consequence of the change in the distribution (in altitude and latitude) and/or abundance of certain organisms that are important disease vectors (in fewer cases the capacity of vectors will be reduced) (IPCC, 2014b).

Climate change will reduce the populations, vigour, and viability of species with spatially restricted populations (e.g. small and insulated habitats and mountaintops). Extinctions of endemic species could be as high as 39-43% (i.e. >50,000 plant and vertebrate species) under worst case scenarios (Malcom *et al.,* 2005)

Marine ecosystems

As in terrestrial and freshwater species, some marine species will change their distribution due to the projected warming of the planet, causing high-latitude invasions and local-extinction rates in the tropics and semi-enclosed seas (Muhling *et al.,* 2015; Liu *et al.,* 2015). The economic dimension of these changes is different across the world, where species richness and fisheries catch potential are projected to increase (on average) at mid and high latitudes, contrary to what would happen in tropical latitudes**.**

For example, the IPCC (Field *et al.,* 2014) states that in North America there is going to be a shift in distribution of the northwest Atlantic fish species, changes in mussel beds along the west coast of the USA, and a change in migration and survival of salmon in northeast Pacific. In South America, mangrove degradation on the north coast will be impacted in a minor scale by climate change (pollution and land use are the main drivers of change). In the polar regions, climate change will significantly impact Arctic non-migratory species, the reproductive success of Arctic seabirds, populations (decrease) of southern ocean seals and seabird populations, thickness of foraminiferal shells (reduction) in southern oceans due to ocean acidification, and the density of krill (reduced) in the Scotia Sea.

Three main drivers related to climate change and emissions of carbon dioxide will have a negative impact on coastal ecosystems: 1. Sea level rise, which is related to the capacity of animals (e.g. corals) and plants (e.g. mangroves) to keep up with the vertical rise of the sea; 2. Ocean temperature, which has a direct impact on species adjusted to specific and sometimes narrow temperature ranges (e.g. coral bleaching). As a response to warmer temperatures, many marine species change their distributions towards the poles; 3. Ocean acidity, caused by the absorption of carbon dioxide that produces carbonic acid. An increase of acidity in seawater diminishes the ability of "calcifiers" (e.g. shellfish, corals) to produce carbonate to make their shells and skeletons.

The physical, chemical, and biological properties of the ocean will be altered by climate change, causing a change in the physiological performance of marine biodiversity.

Shifts in populations, geographic distribution, migration patterns, and phenology of species caused by climate change, have been and will be paralleled by a reduction in their maximum body size. Furthermore, this has caused and will continue causing a change in the interaction between species (e.g. competition and predator-prey dynamics).

Regional changes in the temperature of the atmosphere and the ocean will be accompanied by changes in glacial extent, rainfall, river discharge, wind, ocean currents, and sea level, among many other environmental parameters. There are large fluctuations in ocean conditions in each ocean basin, like the El Niño Southern Oscillation, the North Atlantic Oscillation, and the Atlantic Multidecadal Oscillation, each leading to major changes that have impacts on the coastal zone. There are, on the other hand, very large differences in freshwater supply in different coastal locations, and processes in the watershed, including the balance of different human activities, are different in all watersheds. All of these factors work together in different ways to affect any one coastal habitat.

North America

Climate in the Arctic is harsh, characterized by cold winters and cool summers. Consequently, plant growth is restricted to a relatively short growing season on the order of three months or less during the boreal summer. The tundra biome is home to approximately 1,800 species of vascular plants and has less species diversity than more temperate biomes (Callaghan *et al.,* 2005) (see Chapter 3 for more details). Alpine tundra can also occur at high elevations in mountain ranges of North America.

Global temperature increases during the twentieth century have been amplified in the Arctic, with mean annual temperature increases approximately twice that of the global increase. For example, over the past 60 years, Alaska has warmed more than twice as rapidly as the rest of the USA, with state-wide average annual air temperature increasing by 1.7 °C and average winter temperature by 3.4 °C, with substantial year-to-year and regional variability (Chapin *et al.,* 2014). The overall warming has involved more extremely hot days and fewer extremely cold days.

There is increasing evidence that physical and ecological changes are already occurring throughout the tundra biome (Hinzman *et al.,* 2005; McGuire *et al.,* 2006), and includes increases in photosynthetic activity (Bunn & Goetz, 2006) and an expansion of shrub tundra at the expense of graminoid tundra (Myers-smith *et al.,* 2011).

Average annual temperatures in the northern tundra region of Alaska are projected to rise by an additional 2.5 ºC to 5 ºC by the end of this century depending on fossil fuel emissions (Chapin *et al.,* 2014). Annual precipitation is projected to increase about 15% to 30% by late this century if global emissions continue to increase (Chapin *et al.,* 2014). However, increases in evaporation due to higher air temperatures and longer growing seasons are expected to reduce water availability.

The changes in climate are projected to increase the area occupied by shrub tundra in northern Alaska by 2% to 21% by the end of this century, largely at the expense of graminoid tundra, which is projected to decrease by 8% to 24% (Rupp *et al.,* 2016). Treeline is projected to move slightly northward in some climate scenarios (see Chapter 3 for more details). Climate change is also expected to have significant consequences for the distribution and diversity of Alpine tundra ecosystems in mountain ranges of North America, as tundra ecosystems may shift to higher elevations and lose biodiversity (Lesica, 2014).

Notably, the acceleration in ice sheet loss over the last 18 years was 21.9 ± 1 Gt/yr^2 for Greenland (Rignot *et al.,* 2011). In July 2012, over 97% of the Greenland ice sheet experienced surface melt, the first widespread melt during the era of satellite remote sensing. Since Arctic temperatures are expected to rise with climate change, the authors' results suggest that widespread melt events on the Greenland ice sheet may begin to occur almost annually by the end of century (Keegana *et al.,* 2014). Lenton (2011) included the irreversible melt of the Greenland ice sheet as one of the eight candidates of human-induced climate change tipping points. Biodiversity and ecosystem services of Greenland are highly vulnerable to anthropogenic climate change (Larsen *et al.,* 2014).

Boreal forests and temperate forests: warming in the boreal forest area of Alaska has occurred throughout the 20th century, with mean annual temperatures increasing between 0.5 and 3.0 ºC in regions south of 60 ºN (Price *et al.,* 2013). Since 1900, annual precipitation amounts appear to have increased by 10% to 20% throughout much of the boreal zone of Canada, although drought conditions have existed in western Canada since 1995 (Price *et al.,* 2013). In the temperate zone of North America, warming has also been substantial (~0.9 ºC since 1895, Melillo *et al.,* 2014). In recent decades, moisture availability has decreased in the southeast and west, while the northeastern USA has experienced more extreme precipitation events (Melillo *et al.,* 2014). These changes in climate in recent decades have generally increased tree mortality of both boreal and temperate forests through fire, insect infestations, drought, and disease outbreaks (Price *et al.,* 2013; Chapin *et al.,* 2014; Joyce *et al.,* 2014).

Annual mean temperatures across the Canadian and Alaska boreal zones are projected to be 4 to 5 ºC warmer by 2100 (Price *et al.,* 2013; Chapin *et al.,* 2014). Although annual precipitation is projected to increase in Canada and Alaska, increases in evaporation due to higher air temperatures and longer growing seasons are expected to reduce water availability to these forests. In the temperate zone, another 1 to 2 ºC warming is expected by 2100, with continued reduced water availability in the southeast and western USA (Melillo *et al.,* 2014). Although climate envelope models for individual species suggest that these changes could potentially result in substantial shifts in species ranges in response to climate change, they generally do not account for limiting factors such as soil suitability, geographic barriers, and seed dispersal distances, which all limit the rate at which new areas can be colonized (Price *et al.,* 2013). The application of models that do consider these limiting factors indicate that northward migration of boreal forest into tundra regions will be very limited during the remainder of this century (Rupp *et al.,* 2016). However, the projected climate changes for North America are expected to increase the vulnerability of boreal and temperate forest to increased mortality through fire, insect infestations, drought, and disease outbreaks, particularly in areas where water availability is already a concern (Price *et al.,* 2013; Chapin *et al.,* 2014; Joyce *et al.,* 2014). For example, the analyses of Rupp *et al.* (2016) estimate that changes in the fire regime will decrease late successional boreal conifer forest by 8% to 44% by the end of this century, with a concomitant increase in early successional deciduous forest. In lowland forest areas of the boreal zone underlain by ice-rich permafrost, forest mortality could increase because of subsidence and inundation associated with permafrost thaw (Price *et al.,* 2013). However, in both boreal and temperate forests with well-drained soils and adequate water availability, it is expected that forest productivity may increase (Price *et al.,* 2013; Joyce *et al.,* 2014).

Increasing temperatures and changes in the amount and timing of precipitation are expected to affect the temperate grasslands of North America. However, despite potential increases in aridity, particularly during summer, the fractional cover of green foliage may increase under future climate scenarios (Hufkens *et al.,* 2016). This increase is likely to occur from earlier spring green-up and later autumn senescence, which may more than compensate for any reduction of fractional cover during hot, dry summers (Hufkens *et al.,* 2016).

Many of the dryland regions of North America are experiencing changes in climate. The Great Basin, Colorado Plateau, Mojave in the USA and Sonoran Desert in northwestern Mexico and the southwestern USA have experienced a warming trend, particularly during winter and spring, and the freeze-free season has lengthened (Weiss & Overpeck, 2005; Cook & Seager, 2013). These temperature changes have the potential to shift vegetation types northward and eastward and upward in elevation (Weiss & Overpeck, 2005), having implications for the adjacent deserts (Notaro *et al.,* 2012).

Wetlands in the Prairie Pothole region (freshwater marshes, wet meadows, etc.) are experiencing increased temperatures and variability in precipitation, which may have implications for waterfowl and important ecosystem services. Projected changes in temperature and precipitation of more than 1.5-2.0 °C may diminish wetland function across the majority of the Prairie Pothole region (Johnson & Poiani, 2016).

Northern portions of the Everglades in South Florida are dominated by peatlands that depend on adequate amounts of precipitation to balance the constant loss of water through evapotranspiration, but increased periods of drought have the potential to cause large shifts in plant and animal communities (Nungesser *et al.,* 2015). In southern portions of the Everglades, plant communities are threatened by increased salinity from sea level rise, which can create physiological drought and a shift from freshwater to saltwater-tolerant species (Nungesser *et al.,* 2015). In the Florida region, models and field data indicate that mangrove forests will continue to expand their latitudinal range as temperature and atmospheric carbon dioxide concentrations increase (Alongi, 2015).

Average annual temperatures have increased by as much as 0.25°C per decade since the middle of the twentieth century in some parts of the Great Lakes region of North America (Hayhoe *et al.,* 2010). Those temperature changes have advanced the timing of spring, lengthened the growing season (Robeson, 2002), and produced low lake levels (Notaro *et al.,* 2015a).

The frequency of heavy rainfall events has nearly doubled since the 1930's (Angel & Huff, 1997; Kunkel *et al.,* 1999; Villarini *et al.,* 2011) and is associated with hydrologic flooding in some areas of the midwest (Peterson *et al.,* 2013). Increased lake surface temperatures, frequent and intense cyclones, and reduced ice cover have been associated with more occurrences of lake-effect snow (Burnett *et al.,* 2003; Kunkel *et al.,* 2009), which can affect hydrologic systems and species that are sensitive to changing moisture regimes (Davis *et al.,* 2000; Burnett *et al.,* 2003). Warming lake temperatures have been shown to generate low oxygen conditions in deeper portions of the lakes and extreme precipitation and drought events may play a role in harmful algae growth (Zhou *et al.,* 2015), both affecting fish growth, reproduction, and survival (Scavia *et al.,* 2014). Additionally, warming lakes have been shown to alter the extent and duration of temperature preferences for some commercially important fish species, potentially intensifying competition and food-web interactions (Cline *et al.,* 2013).

Ice cover in the Great Lakes is projected to continue declining and will eventually be restricted to the northern lake shores in mid- to late winter (Notaro *et al.,* 2015b). Enhanced evaporation from lack of ice cover will increase lake-effect precipitation, but it will consist primarily of rain due to increasing temperatures (Notaro *et al.,* 2015b). However, because both precipitation and evaporation over lakes is expected to increase, the influence on lake levels is still unclear (Angel & Kunkel, 2010; Notaro *et al.,* 2015a).

The pelagic ocean is presenting changes in major wind patterns, ocean currents, temperature, and pH (e.g. Bates *et al.,* 2014; Muller-Karger *et al.,* 2015). For example, it is expected that the north Atlantic Ocean will continue the warming trend that has been observed there over the past decade (Liu *et al.,* 2015, 2016). These changes are expected to have an impact on suitable habitat of a number of valuable fish and affect fisheries that depend on them (Kerr *et al.,* 2009; Hare *et al.* 2010, Lenoir *et al.,* 2011; Muhling *et al.,* 2015, 2017). Warming off the Alaska coast since the late 1970s triggered a decline in forage species (e.g. shrimp and capelin) and an increase in high-trophic level groundfish (Anderson & Piatt, 1999). This community reorganization negatively affected seabirds, marine mammals, and other species that depend on forage species (Anderson & Piatt, 1999). A warm-water anomaly (i.e. "the blob") was detected off the Alaska coast during the winter of 2013-2014, with near-surface temperatures 2.5°C greater than normal that eventually stretched south to Baha, California (Bond *et al.,* 2015; Cavole *et al.,* 2016). The cause of the anomaly is believed to be the result of reduced heat exchange between the ocean and the atmosphere and weak horizontal advection in the upper ocean, which may have been triggered by a much higher than normal sea level pressure (Bond *et al.,* 2015). The anomaly negatively affected commercially-important fisheries, including tuna, and was responsible for marine mammal and seabird strandings (Cavole *et al.,* 2016).

Mesoamerica

Precipitation is projected to decline during the wet season throughout the region and mountainous areas in Costa Rica and Panama, which generally receive a large amount of orographic moisture, will see a decline in precipitation (Karmalkar *et al.,* 2011). Differential warming of the Pacific and Atlantic sea surface temperatures, which causes a stronger Caribbean low level jet, will lead to drier conditions in Mexico and Central America (as much as 50% drier) during summer (Fuentes-Franco *et al.,* 2015) and has the potential to lead to water stress in many regions. Additionally, severe and extended dry seasons are likely to lead to forest species turnover and loss of many tree species (Condit, 1998). However, Prieto-Torres *et al.* (2015) found that while tropical dry forests are projected to decline in many areas of Mexico, they may increase in other areas by moving upward in elevation.

Changes in temperature and precipitation have the potential to affect the climate-sensitive cloud forests of Mesoamerica by causing biodiversity loss and shifts from the unique ecosystems to lower-altitude vegetation types (Foster, 2001). Additionally, climate changes may result in changes in cloud formations, which are already being observed in certain parts of Costa Rica (Foster, 2001). Although sea evaporation is likely to increase with increasing sea surface temperatures, pumping more water into the atmosphere, cloud formation is expected to increase in height, which will alter the relative humidity and amount of sunlight the forests are exposed to (Foster, 2001). The total area of cloud forests in Mexico is expected to decline by as much as 70% by 2080 (Ponce-Reyes *et al.,* 2013). However, models suggest that minimizing land-use change and developing protected areas in remaining cloud forests may promote dispersal and allow some species to persist despite changes to climate (Ponce-Reyes *et al.,* 2013). In addition, protected areas can have other benefits, such as the ability to capture and reduce carbon dioxide emissions into the atmosphere (Uribe, 2015).

The Mesoamerican tropical dry forests are experiencing increased warming (Aguilar *et al.,* 2005, Karmalkar *et al.,* 2011). Between 1961 and 2003, the percentage of warm minimum and maximum temperatures have increased by 1.7% and 2.5% per decade, respectively, whereas the percentage of cool minimum and maximum temperatures have decreased by 2.4% and 2.2% per decade, respectively (Aguilar *et al.,* 2005). Most of the precipitation in the tropical dry forests occurs during the summer (Fuentes-Franco *et al.,* 2015) and is likely an important factor in the distribution of tropical tree species richness (Somers *et al.,* 2015). Although no trend in the amount of precipitation has been observed, the intensity of rainfall events has increased over the last 40 years (Aguilar *et al.,* 2005).

Karmalkar *et al.* (2011) projected that warming in the region will vary both spatially and temporally, with higher temperatures in the Yucatan Peninsula and during the wet season. Increased temperatures in the tropical dry forest has implications for carbon sequestration, as carbon uptake is likely to decline substantially under warming conditions (Dai *et al.,* 2015). Additionally, because understory microsite variability is low in some portions of the tropical dry forests, future warming could have serious implications for neotropical birds (Pollock *et al.,* 2015). A temperature increase >3°C has the potential to cause a 15% decline in potential species richness (Golicher *et al.,* 2012) (see Chapter 3 for more details).

Most wetlands in Mexico are found along the Gulf of Mexico or Pacific Ocean (Mitsch & Hernandez, 2013). Similarly, mangrove swamps are common on both coastlines in Central America (Mitsch & Hernandez, 2013). Consequently, sea level rise is by far one of the largest concerns with regards to climate change impacts on wetland resources in those regions (Mitsch & Hernandez, 2013). The effects of sea level rise on mangrove ecosystems, for example, could have implications for fish, mollusks, and aquatic mammals (Botero, 2015). However, feedbacks between plant growth and geomorphology may allow for wetlands to maintain stability and resist the negative impacts of sea level rise. This resiliency likely depends on human interference, such as groundwater withdrawal or artificial drainage of wetland soils, which can lead to more rapid subsidence (Kirwan & Megonigal, 2013). Additionally, the construction of dams and reservoirs may prevent sediments needed for wetland building from reaching coastal areas, which can minimize the likelihood for wetland sustainability under sea level rise (Kirwan & Megonigal, 2013).

Caribbean

Most insular ecosystems in the Caribbean Sea have experienced a warming trend in recent decades, with increases in both daily minimum and maximum temperatures (Karmalkar *et al.,* 2013). However, those trends vary by region as Puerto Rico has experienced an increase in daily minimum temperatures, but a decrease in daily maximum temperatures (Van Beusekom *et al.,* 2015).

Ecosystems found in Caribbean regions may be particularly vulnerable to rising sea levels; Bellard *et al.* (2014) projected that 63 out of 723 Caribbean islands would be completely submerged with 1 m of sea-level rise and 356 islands submerged with 6 meters of sea-level rise, which may have implications for hundreds of endemic species inhabiting the islands. Additionally, tropical cyclones are expected to increase in intensity (as well as frequency of intense storms) as a result of climate change (Michener *et al.,* 1997; Reyer *et al.,* 2015). Some regions of the Caribbean may receive a large proportion of their annual rainfall from hurricanes (Scatena & Larsen, 1991), which may be important given droughts increased between 1950 and 2010 (Dai, 2012). The frequency of droughts is also expected to increase in the future (Reyer *et al.,* 2015). Karmalkar *et al.* (2013) estimated that precipitation is likely to decline by 5.7% to 24.6% (depending on the model) between the years of 2080 and 2089 compared with 1970 and 1989. Reduced preciptitation, along with warmer temperatures, have the potential increase evapotranspiration and drought risk (Reyer *et al.,* 2015).

The region's forests and terrestrial biodiversity are also threatened by climate change (see Chapter 3 for details). While hurricanes are part of the Caribbean's "normal" environment and ecosystems have adapted to them, the repeated and compounding impacts of frequent extreme weather events has been shown to reduce their ability for recovery. The flash floods and mudslides that caused the many fatalities during the devastating 2008 hurricane season in Haiti, would probably not have been so severe had the mountains not been deforested. Protecting forests and improving their resilience will be an important adaptation strategy both for the conservation of biodiversity and for the future wellbeing of Caribbean communities (Day, 2009).

Warming of coastal areas has had marked impacts on the population, diversity, and health of coral reef resources in the Caribbean Sea and Gulf of Mexico (Eakin *et al.,* 2010; Vega-Rodriguez *et al.,* 2015; van Hooidonk *et al.,* 2015). Increased water temperatures have the potential to affect fisheries in Caribbean countries. Cheung *et al.* (2010) estimated that catch potential off Caribbean coasts may decrease as much as 5% to 50% between 2050 (2°C of warming) and 2100 (4°C of warming).

The global net value of the coral reefs of the Caribbean Sea services related with fishery, coastal protection, tourism, and biodiversity, were estimated $29,800 million per year. Currently two thirds of the Caribbean coral reefs are impacted detrimentally by human activities, including climate change, (GEO 4, UNEP, 2007).

Mass coral bleaching events have also become more frequent and more severe in recent years as a result of increasing sea surface temperatures and aragonite saturation, in particular the widespread and catastrophic bleaching event of 2005 in the Caribbean. This is presenting a new challenge to islands dependent on reefs for fisheries, dive tourism and coastal protection (Day, 2009). By 2050, with 1.5°C to 2°C, there is 20-40% to 60-80% probability, respectively, that coral reefs in the Caribbean and western Atlantic will undergo yearly bleaching events (Meissner *et al.,* 2012). Nearly all coral reefs are expected to undergo severe bleaching by 2100, with exception to areas with upwellings (Meissner *et al.,* 2012).

The IPCC (2014) considers the small island states, like those of the Caribbean, to be among the most vulnerable to the projected impacts of climate change, like rising sea levels, intensifying storms, mass coral bleaching events, ocean acidification, and potential water and food shortages.

South America

Although the Amazon basin has experienced periodic warming and cooling since the 1900s, which may be associated with the Pacific Decadal Oscillation (Malhi & Wright, 2004; Gloor *et al.,* 2015), annual mean temperature has steadily increased since the 1970s (Victoria *et al.,* 1998, Malhi & Wright, 2004; Vincent *et al.,* 2005) and is more intense during the dry season than the wet season (Gloor *et al.,* 2015). Trends in long-term precipitation patterns and their link to climate change (as opposed to Pacific Decadal Oscillation and El Niño Southern Oscillation) are less clear (Marengo, 2004; Satyamurty *et al.,* 2010). However, Gloor *et al.* (2015) showed that although annual net rainfall has increased in the area, the amount of rainfall during the dry season has decreased since the 1970s. Those trends are concerning given that droughts in the tropical forests have been associated with reduced vegetation growth and browning (de Moura *et al.,* 2015), slow canopy recovery times (Saatchi *et al.,* 2013), reduced above ground live biomass (Saatchi *et al.,* 2013), and accelerated tree mortality over large areas (Phillips *et al.,* 2009). During the wet season, the frequency of heavy rainfall events and severity of Amazon flood pulses has increased (Donat *et al.,* 2013; Gloor *et al.,* 2015), potentially affecting the ecology of floodplain and swamp forests in the Amazon basin.

Climate projections suggest that both temperature and precipitation trends are likely to continue, with a substantial lengthening of the dry season by the end of the twenty-first century (Boisier *et al.,* 2015). Those conditions have the potential to prevent the tropical forest distribution from moving upslope (staying restricted to wet areas) and persisting along ecotones, and could eventually cause it to convert to savannah-type vegetation in eastern portions of the basin (Olivares *et al.,* 2015). Additionally, species richness and plant productivity are likely to decline, altering the Amazon basin from a carbon sink to a source (Olivares *et al.,* 2015). Finally, the severity of wet-season flood pulses is projected to increase and may have implications for movement and reproduction of many Amazon River-associated species (Zulkafli *et al.,* 2016).

There is no climatic assessment devoted exclusively to the Amazonian wetlands. However, the IPCC Regional Assessment for Central and South America (Magrin *et al.,* 2014) covers the entire distribution of this environment. Based exclusively on this assessment in the northern part of South America, some inferences can be drawn in regard of these wetlands. The trends are:

- Temperatures: In general terms, with the exception of interior Venezuela, 30% to 50% increase in temperature is expected in northern South America, representing +5°C to +7°C. And for the period of 2071 to 2100 another increase from +4°C to +5°C is expected (Marengo *et al.,* 2012). This problem is exacerbated in urban environments, even in small island developing states (Mendez-Lazaro *et al.,* 2017).

- Precipitation: In general, an increase from 30% to 50% in precipitation is expected in northern South America. However, while a decrease of 20% to 30% in rainfall in central and eastern Amazonia, is expected, an increase from 10% to 30% in rainfall in western Amazonia is expected (Giorgi & Diffenbaugh, 2008; Mendes & Marengo, 2010; Sorensson *et al.,* 2010; Marengo *et al.,* 2012). This increase in rainfall for western Amazonia will be observed both in summer and winter. This, in turn, will deeply affect flooding patterns in wetlands in northern and western Amazonia. Effects of precipitation on current flows, rivers discharge and potential flooding was observed for most of the large rivers (Dai *et al.,* 2009; Dai *et al.,* 2004)

- Sea level: In coastal areas an increase in sea level is expected, with increase in flood probabilities (>40%). Impacts of flooding can be costly and coastal communities should evaluate possible solutions to cope with this problem (Marengo *et al.,* 2017). Extreme events: Longer dry periods, or consecutive dry days, are expected for the region, with an increase of up to 8% (or 5 more dry days). Heavier precipitation in northern and western Amazonia (from 1 to 10mm) is also expected.

All impact analysis available indicates that these extreme events and the trends of climate change in Amazonian wetlands and rivers will be very strong (Marengo & Espinoza, 2016). Extreme events will be more frequent and more intense, and floods and droughts will impact both natural and human systems in the region. Although with a large range of uncertainty, wetlands in the northern and western Amazonia may experience more frequent floods, while wetlands in eastern Amazonia might be under more intense and severe droughts. These effects might cause great changes on the biota of all wetlands affected. Intense floods can bring losses in crops (inundation of small farms and gardens), in local and regional fisheries, and even in human lives. Intense droughts are associated with fire incidence, and additional aerosol emissions, public health problems, and other losses in agriculture and fisheries (Marengo & Espinoza, 2016).

Most areas in the Andes Mountains have experienced a warming trend (Vuille *et al.,* 2015), particularly during winter (Barros *et al.,* 2015). Magrin *et al.* (2014; and references therein) showed that temperatures have increased by 0.1°C to 0.6°C per decade across different regions of the Andes since the 1950s and 1960s. The warming conditions have caused many of the Andean glaciers to retreat, creating a loss of important water reserves (Barros *et al.,* 2015). Additionally, snow is melting earlier in the spring and has affected the timing of maximum stream flows, which are peaking as much as a month earlier in recent years than when compared to the early twentieth century (Barros *et al.,* 2015). Reduced river flows in Argentina have suggested a decrease in precipitation (Barros *et al.,* 2015), but the precipitation trends are less clear in other regions of the Andes Mountains (Marengo *et al.,* 2009). Vuille *et al.* (2003) found that precipitation was greater north of approximately 11°S, whereas stations found south of that mark showed decreasing precipitation between 1959 and 1994.

Projected temperatures suggest increases of 2.0-3.5°C by the end of the 21st century, which has the potential to cause glaciers to retreat substantially or disappear altogether (Barros *et al.,* 2015). Precipitation is most likely going to increase between the latitudes of 5°N and 20°S, particularly in northern Peru where precipitation could increase as much as 70% (Marengo *et al.,* 2011). However, precipitation is most likely going to decrease (as much as 10%) in the subtropical Andes south to Patagonia and on the altiplano (Marengo *et al.,* 2011). Additionally, Andes snowfall will be less common in the mountains of Argentina and melt earlier in the spring, affecting the amount of water available for summer irrigation (Barros *et al.,* 2015). Important tropical Andes ecosystems, like páramos, punas, and evergreen montane forests, are projected to undergo a large amount of species turnover or loss of species richness (Ramirez-Villegas *et al.,* 2014). The páramo grasslands, glaciers, and cryoturbated areas, which are found at the highest elevations, may be at greatest risk (Tovar *et al.,* 2013). Species found in the cloud forests of the Andes may be at risk of extinction due to observed upward shifts in ecotones, which could serve as barriers to species migration (Lutz *et al.,* 2013) (see Chapter 3 for more details).

The Brazilian Cerrado, a large area of tropical dry forest, savanna, and grasslands found on the Brazilian Central Plateau, has been trending warmer, with an annual maximum temperature increase of 0.79°C between 1980 and 2004 (Santos, 2014). Additionally, the number of days with temperatures >25°C increased at a rate of 4.4 days per year during that same time period (Santos, 2014). Precipitation trends are less clear, with the exception of the number of days with heavy precipitation (>10mm), which showed a decrease of 0.43 days per year between 1980 and 2004 (Santos 2014). Projected temperature increases may increase as much as 2.5°C to 5.5°C over tropical and subtropical latitudes and precipitation is expected to decrease during most seasons (with exception to winter) by the end of the 21st century (Cabré *et al.,* 2016). This warming trend along with reduced precipitation (Marengo *et al.,* 2009) could have implications for fire activity. Fire is an important factor in the grassland regions of the Cerrado, and has increased in frequency since European settlement (Pivello, 2011). Although fire is often anthropogenic in nature, it can occur naturally through lightning strikes and is particularly destructive in areas where fire is actively suppressed, having important implications for biodiversity (Pivello, 2011). For example, small mammal communities, which play important roles in a variety of ecosystem processes (e.g. plant composition, soil structure; Sieg, 1987), have shown to be sensitive to severe fires, particularly in the savanna woodland regions of the Cerrado ("Cerradão"; Mendonca *et al.,* 2015). Although sustainable use of fire is appropriate in the Cerrado, careful management is needed to avoid land degradation and loss of biological diversity and ecosystem processes (Pivello, 2011).

Many tropical grasslands have been targeted for reforestation to help offset carbon dioxide emissions. However, not all grassland regions are the result of deforestation and converting them to plantations has the potential to cause substantial losses in biodiversity (Bond, 2016).

Temperature are expected to increase in the Río de la Plata grasslands, particularly during spring (Cabré *et al.,* 2016). Although precipitation in many areas of the region has been linked with El Niño Southern Oscillation (Ropelewski & Halpert, 1987), trends suggest that rainfall has increased in Uruguay, Paraguay, northern Argentina, and southern Brazil between 1960 and 2000 (Haylock *et al.,* 2006). However, Haylock *et al.* (2006) found that those precipitation trends closely align with a trend towards a more negative southern oscillation index, suggesting that more frequent El Niño Southern Oscillation-like events are responsible for recent changes in precipitation. Rainfall is expected to increase in southern Brazil, particularly in summer and fall, and will decrease during winter and spring (Cabré *et al.,* 2016). Precipitation is associated with net primary productivity in some areas of the Río de la Plata region, particularly in native forests and afforested areas, but other land use activities can interact with climate factors and cause carbon storage to decline (Texeira *et al.,* 2015). An increase in precipitation may cause flooding, erosion, and increased nutrient runoff, which can affect biological communities in pampean rivers and streams by increasing the number of species that better tolerate turbid and enriched environments (Capitulo *et al.,* 2010).

Climate change is likely to have a substantial impact on mangrove ecosystems (Ellison, 2015), through processes including sea level rise, changing ocean currents, increased storminess, increased temperature, changes in precipitation, and increased carbon dioxide. Exposure to disturbances induces dynamism on annual and decadal scales that is reflected in changes in the populations, biomass, and spatial distribution of the mangrove ecosystem (Schaeffer-Novelli *et al.,* 2016). Sea level rise is likely to influence mangroves in all regions, although local impacts are likely to be more varied. Mangroves are likely to be less affected by sea level rise in areas with high sediment availability, uplifting or stable coasts, high productivity, and large tidal ranges (Ward *et al.,* 2016), as well as along wet tropical coasts and/or in areas adjacent to significant river input (Alongi, 2008), like the Amazon estuary and Parnaiba delta.

These factors combined with increased temperatures at the latitudinal extremes of mangrove distribution, a predicted increase in the strength and frequency of El Niño events that lead to below normal rainfall and a decrease in extreme precipitation events in most of tropical South America, and a resultant decrease in the cooling and drying influence of the Humboldt Current in western South America, could provide an increase in the distribution of mangroves within South

America. However, in semiarid regions of South America, where mangroves typically occur in estuaries, and irrigation and damming are more prevalent, mangroves are likely to suffer from increases in salt-stress and resultant decreases in productivity combined with decreases in sediment input (Ward *et al.,* 2016).

Climate change mitigation and adaptation strategies

Because of the substantial increase of atmospheric greenhouse gases in recent decades, it is important to identify actions that may reduce emissions through mitigation efforts. Many mitigation policies have already been implemented in the Americas region. For example, although no national climate legislation exists, a variety of policies and measures that lower emissions have been implemented at multiple governmental levels in the USA (U.S. National Climate Assessment, 2014). Additionally, developing countries, like Brazil, are also making strides with regards to mitigation, pledging to reduce greenhouse gas emissions by as much as 40% below 2005 levels by 2030 (Brazil Intended Nationally Determined Contribution, 2015). Some communities are taking the important step of talking about possible impacts of sea level rise, for example (Marengo *et al.,* 2017). However, because climate change is a global issue, it is important that countries work collaboratively to develop emission reduction strategies as opposed to each country approaching the problem independently (IPCC, 2014a).

Mitigation can also refer to enhancing the capacity for carbon storage in regions that may be able to remove greenhouse gases from the atmosphere (IPCC, 2014a). Both oceans and vegetated regions have the potential to serve as carbon dioxide sinks, and improving our understanding of the various physical and biological processes that can increase carbon uptake will assist with developing better estimates of potential carbon offsets. For example, it is well known that vegetated coastal regions (e.g. salt marshes, mangroves) can be important regions for carbon sequestration, but recent work has indicated that microalgae may also sequester substantial amounts of carbon and is able to deliver it to sediments and the deep sea for long-term storage (Krause-Jensen & Duarte, 2016). Similarly, calculating more accurate carbon offsets in forests requires consideration of both the ability to regulate greenhouse gases, as well as regulation of water and energy (Anderson-Taixeira *et al.,* 2012).

Although mitigation is critical for reducing greenhouse gas emissions, the IPCC has warned that projected climate change is expected to affect human and natural systems despite the scale of mitigation policies that are adopted in the next few years (IPCC, 2007). Therefore, developing and implementing effective adaptation strategies will be needed to minimize those potential climate change impacts (IPCC, 2007). Adaptation planning is occurring in both the public and private sectors throughout many regions of the Americas. For example, many municipalities in North America are considering incremental changes to their planning efforts as a result of climate change and some regions in Central and South America are considering ecosystem-based approaches, such as developing protected areas (IPCC, 2014a). Despite increased recognition of the importance of adaptation planning in response to climate change, few measures have actually been implemented on the ground (IPCC, 2014a). Barriers to implementation include limited funding, policy and legal impediments, and difficulty in anticipating climate related changes at local scales (U.S. National Climate Assessment, 2014).

The majority of adaptation planning is focused on risk and water management and the importance of ecosystem-based adaptation is only recently being recognized (IPCC, 2014a). Vignola *et al.* (2009) found that developing countries, in particular, depend heavily on ecosystem services and it is critical that they be mainstreamed into national and international adaptation policies. Additionally, those authors suggested that adaptation needs to be more closely linked with mitigation to ensure certain mitigation policies are less likely to have negative impacts on the well-being of certain communities (Vignola *et al.,* 2009). Ongoing monitoring is therefore crucial to develop a better understanding of, and adaptation to future changes. This will also allow for more effective incorporation of ecosystems into spatial planning, including disaster risk reduction strategies (UNEP, 2014). Indigenous and local knowledge also contribute to climate change mitigation and adaptation as presented in **Box 4.12**.

Climate change is a central element of the Aichi targets of the CBD Strategic Plan for 2011-2020 **(Box 4.13)**.

4.4.4 Biological Invasions

Nature of the driver, its recent status and trend, and factors that influence its intensity

Invasive alien species have gone from scientific curiosity to a real societal concern due to their ecological, social, and economic impacts (Mack *et al.,* 2000). Invasive plants and animals cause changes in the composition and function of ecosystems, affecting biodiversity, ecosystem services, and human welfare. invasive alien species have become a major component of global change and pose a serious threat to local and global biodiversity (Hobbs, 2000; Mack *et al.,* 2000; Vilà & Ibañez, 2011).

Box 4 ⑫ Indigenous and traditional knowledge on climate change.

The Millennium Ecosystems Assessment (2005) considers the traditional knowledge, or practitioners' knowledge held by local resource managers, can be of equal or greater value for ecosystem management, not only the formal scientific information.

For North America, the government agencies incorporated the indigenous communities into established initiatives to develop no-regrets and co-benefits climate change adaptation strategies. Rural and indigenous community members possess valuable local and experiential knowledge regarding NCP (Romero-Lankao, 2014).

For the Caribbean islands, the preservation of the traditional knowledge of biodiversity is crucial to the sustainable use of NCP. The loss of such traditional knowledge, for example that related to medicine plants and agriculture, has had a direct negative effect on biodiversity and on the degradation of ecosystems (Suárez *et al.,* 2008). There is continuing strong support for the incorporation of indigenous knowledge into adaptation planning on small islands (Nurse *et al.,* 2014).

There is a growing acknowledgement that indigenous and traditional knowledge has the potential to bring solutions to face the rapidly changing climate and that land ownership and authority of indigenous groups can help better manage many natural areas and reduce deforestation of the Central and South American region. Linking indigenous knowledge with scientific knowledge is crucial for the adaptation process, currently there is limited scientific literature discussing that subject (Magrin *et al.,* 2014). The concept of "mother earth" (madre tierra in Spanish) as a living system has emerged in different forms in recent years, as a key sacred entity on the view of indigenous nations and as a system that may be affected by and also resilient to climate change.

Box 4 ⑬ Climate change and the Aichi targets of the CBD Strategic Plan for 2011-2020.

The CBD recognizes the urgency of addressing climate change in order to halt the rate of biodiversity loss, and this is reflected in its Strategic Plan for 2011-2020. Because of the broad impact of climate change, this driver is covered and/or impacted indirectly by many of the Aichi targets of the Plan, in targets like number 5 on the half of natural habitats rate loss, number 11 on terrestrial and coastal and marine areas protection and number 14 on the restoration and protection of ecosystem services, to mention a few. The achievement of these targets will help to mitigate and adapt to climate change, both from an anthropocentric and biodiversity perspective.

Nevertheless, targets 10 and 15 refers directly to climate change.

Aichi target 10: By 2015, the multiple anthropogenic pressures on coral reefs, and other vulnerable ecosystems impacted by climate change or ocean acidification are minimized, so as to maintain their integrity and functioning. The aim of the target is to reduce the impact of other drivers (like the ones covered in this chapter) on vulnerable ecosystems in order to make them more resilient to the unavoidable effects of climate change. This target has a link with target 12 on the conservation of endangered species and target 15 on ecosystems resilience and carbon stocks[1].

Aichi target 15: By 2020, ecosystem resilience and the contribution of biodiversity to carbon stocks have been enhanced, through conservation and restoration, including restoration of at least 15 per cent of degraded ecosystems, thereby contributing to climate change mitigation and adaptation and to combating desertification. Carbon sequestration refers in this target to the carbon taken and stored in biomass and soils of ecosystems like tropical forests, mangroves, wetlands, peatlands and seagrass beds. Therefore, a key mitigation strategy is to recover these ecosystems that have been degraded, damaged or destroyed[2].

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1. CBD. Quick guide to the Aichi Biodiversity Targets, pressures on vulnerable ecosystems reduced. Available at: https://www.cbd.int/doc/strategic-plan/targets/T10-quick-guide-en.pdf Accessed on 11/16/2016.

2. CBD. Quick guide to the Aichi Biodiversity Targets, ecosystems restored and resilience enhanced. Available at: https://www.cbd.int/doc/strategic-plan/targets/T15-quick-guide-en.pdf

For a species to become an invasive species, it must successfully transit three distinct stages, often called the "invasion process" (Blackburn *et al.,* 2014; Canning-Clode, 2015). The first stage of this process is the "transport phase" where individuals of a species are transported (intentionally or unintentionally) from their native range and released outside their native range. These individuals are termed "non-native" (synonymous term with the terms "non-indigenous", "exotic", and "alien"). Second, these individuals may establish a viable self-sustaining population ("establishment phase") and become "naturalized" species in the new environment. In the third and final stage, a naturalized non-native population might increase in abundance and expand its geographic range ("spread phase"), with the potential to alter the environment in which they have become established, causing ecological and economic harm ("impact phase") and becoming what is considered an "invasive species". This report uses the definition of invasive alien species of the CBD (see Deliverable 3b on invasive alien species), which defines the term (https://www.cbd.int/invasive/terms.shtml) as "plants, animals, pathogens and other organisms that are non-native to an ecosystem, and which may cause economic or environmental harm or adversely affect human health. In particular, they impact adversely upon biodiversity, including decline or elimination of native species - through competition, predation, or transmission of pathogens - and the disruption of local ecosystems and ecosystem functions."

Invasive alien species as drivers and passengers of global change

Unlike other drivers of biodiversity, biological invasions are considered both drivers and passengers of human-driven global change (MacDougall & Turkington, 2005). Biological invasions are by definition caused by the human movement of species and their magnitudes are highly associated with the intensity of changes caused by human activities (Mack & Lonsdale, 2001). Some invasive alien species may be considered passengers of global change because they only persist in an ecosystem through continued human disturbance (e.g. some European weeds associated to roadsides; Seipel *et al.,* 2011). However, many invasive alien species also cause substantial alterations to biodiversity and ecosystem function (e.g. plants increase fire regimes or top-predators). Thus, estimating and forecasting the effects of invasive alien species on biodiversity and ecosystem services has an additional layer of complexity compared to other drivers of global change.

Invasive alien species may act synergistically with each other or with other forces of global change (e.g. climate and land use change) to produce more intense consequences for biodiversity and NCP (Sala *et al.,* 2000; Newbold *et al.,* 2015). Land use changes have long been recognized as a main promoter of invasive alien species across taxa (Hobbs, 2000). Changes in the dominant cover type cause shifts in species composition creating important opportunities for invasive alien species that are well adapted to human disturbances (Didham *et al.,* 2007). From tropical to cold environments, land use changes are associated with roads and other human corridors, which are the main route for dispersal of invasive plants and animals (Seipel *et al.,* 2012). In the last two decades, climate change has been shown to promote invasive alien species by disrupting ecosystems, but also by changing conditions so that they are more suitable to the invader than to the native community (Bellard *et al.,* 2012).

The movement of species by humans and its successful naturalization has increased exponentially in the last two centuries (Seebens *et al.,* 2017). In the Americas, the onset of biological invasions is marked by the arrival of Europeans in the 1500s, which resulted in the massive introduction of non-native species, and the reduction of the natural biogeographical barriers of a continent that had been isolated for thousands of years (i.e. since last glaciation). The influx of non-native species caused by European colonization is still visible today as most invasive alien species in Mediterranean and Temperate regions of the continent are from Eurasia. For example, naturalized plants in Chile and California are mostly Eurasian species (Jimenez *et al.,* 2008). Increase in trade and connectivity, in the last two centuries, and especially since the 1900s, have facilitated the arrival of non-native species from other continents including Australia, Asia and Africa (Jimenez *et al.,* 2008; Van Kleunen *et al.,* 2015).

The introduction of new non-native species into the Americas is expected to continue with increasing trade and transportation by land, sea and air, increasing biological invasions and their potential impacts on biodiversity and NCP in the Americas (Early *et al.,* 2016). Furthermore, the consequences of recent additions of non-native species to the Americas may not yet be visible because it takes time for species to reach high population numbers and wide distributions to cause detectable ecological or economic impacts (i.e. "invasion debt", Essl *et al.,* 2011).

Significant knowledge gaps of invasive species in the Americas exist (Pysek *et al.,* 2008). While countries such as the USA and Canada have been leaders in recording and studying invasive species, most countries in the Americas have only recently directed efforts to record invasive alien species and their impacts (Pysek *et al.,* 2008; Pauchard *et al.,* 2010). Auspiciously, national inventories of invasive species and research on invasive species and their impacts is now being promoted across the Americas to reduce this knowledge gap (e.g. Mexico, Chile, Brazil, Argentina; Zenni *et al.,* 2017).

In the following sections, we review some of the most relevant impacts causes by invasive species in each of the regions of the Americas and their main ecosystem units and we emphasize their role as drivers of changes in biodiversity and their interactions with other drivers of global change.

North America

North America is one of the most invaded regions of the world and one of the most studied in terms of the numbers and impacts of biological invasions (Jeschke & Strayer, 2005; Pysek *et al.,* 2009). Since the 1500s, trade and land use change drivers, in this region, has consistently promoted the establishment of some of the most damaging plant and animal invasive alien species (Stohlgren *et al.,* 2006). The advance of the chestnut rust that decimated the natural populations of the American chestnut (*Castanea dentata*) exemplifies the magnitude of the species, community and ecosystem level impacts of biological invasions in North America (Jacobs *et al.,* 2013). Reductions of plant diversity caused by direct competition between native and non-native plants have been extensively reported in grasslands of North America (Vilà *et al.,* 2003). Plant invasions have also caused enormous changes in ecosystems processes such as hydrological and fire regimes. For example, cheatgrass (*Bromus tectorum*) invasion in arid grasslands has resulted in more frequent and more damaging fires (Pawlak *et al.,* 2014). In addition, some of the most well-known examples of animal invasions have occurred in North America. Vertebrate predators such as rats (e.g. *Rattus rattus*), carps (*Cyprinidae* spp.), and snakes (e.g. *Python bivittatus*) have substantially altered native animal populations driving some to near extinction (Dorcas *et al.,* 2012). Non-native insects, such as ants and mosquitoes, have had a large impact on human well-being (Juliano & Lounibos, 2005).

Tundra and mountain grasslands show relatively low number of plant invasions because of the climatic barrier and the relatively low levels of human disturbances (Pauchard *et al.,* 2009; Bellard *et al.,* 2013). However, some species, mostly European ruderals, are widely distributed in mountains and alpine ecosystems (Alexander *et al.,* 2016). Because of the low abundance and frequency, few impacts have been reported of these plant invasions. Similarly, other taxa invasions have been scarcely reported in these cold ecosystems, partly because the lack of surveys and studies. Climate change and increasing human pressure will likely change this scenario, also causing unexpected shifts in native species distributions (Pauchard *et al.,* 2016).

Boreal and temperate forests and woodlands pose a significant barrier to plant invasions because of the high competition for light (Martin *et al.,* 2008). Thus, most ruderal plant invaders, which invade roadsides and disturbed areas, are not able to succeed in the forest understory (Martin *et al.,* 2008). Nonetheless, in eastern North America, species that are shade tolerant are now entering forested areas. For example, garlic mustard (*Alliara petiolata*) is now occupying deciduous forests generating monospecific patches and displacing native understory species (Kurtz & Hansen, 2014). On the other hand, these forests have been heavily impacted by animals and pathogens. For example, earthworms are now considered a major driver of change in temperate forests (Bohlen *et al.,* 2004). Invasive insects, such wooly adelgids, have devastated forests in eastern USA, having broad range impacts including indirect impacts on fish in streams due to loss of shading (Ellison *et al.,* 2005).

Temperate grasslands have suffered extreme transformations in North America, being replaced by agricultural lands or when maintained, have gone intense grazing pressure and heavy disturbance (e.g. plowing). Thus, the remaining grasslands in North America are being intensively affected by plant invasions. Ruderal species of Eurasian origin such as *Centaurea* spp. *Euphorbia* spp. and *Bromus* spp. have replaced native grasses and herbs across the North American grasslands (Stohlgren *et al.,* 1999). Their impacts not only include changes in plant cover but also long-term shifts in soil processes, food webs and fire regimes (Simberloff *et al.,* 2013).

Mediterranean forests, woodlands and scrub in North America are one of the hotspots for invasive plant species (Seabloom *et al.,* 2006). The high level of trade and human-caused disturbance in this area, and the close climatic match with Mediterranean Europe are responsible for the high levels of invasive plant species (Seabloom *et al.,* 2006). Some of these species have caused irreversible ecosystem change by replacing native species and creating a positive feedback with fire (see example of *Bromus* above). Fungi pathogens have also affected the health of these ecosystems (e.g. Oak Death, Rizzo & Gargelotto, 2003).

Drylands and deserts in North America have been invaded by non-native grasses, shrubs and trees. Invasive species capable of standing desert conditions have thrived in the shrubland and grassland vegetation competing directly for water with native species and creating a continuous fuel layer that promotes more intense and larger fires (Brooks & Chambers, 2011). *Tamarix* invasion in riparian corridors have displaced native riparian vegetation and altered ecosystem structure (Merritt & Poff, 2010).

Wetlands in North America show the highest levels of plant invasions due to the intense purposeful or accidental introductions of aquatic plants (Batzer & Baldwin, 2012). Many of these invasive aquatic plants have profound environmental and economic costs such as *Eichornia crassipes*, *Phragmites australis*, *Lythrum salicaria,* and *Egeria densa*.

In freshwater systems, the zebra mussel (*Dreissena polymorpha*), originally (1988) affected the Great Lakes area, but has now spread to all of the large navigable rivers in the eastern USA, extending along the Illinois River to the Mississippi River and into the Caribbean (Benson *et al.,* 2017). Human activities are important vectors of transport of this species between aquatic systems (Johnson & Padilla, 1996), which is notorious for their biofouling capabilities by colonizing different human aquatic infrastructure (e.g. water supplies for hydroelectric and nuclear power plants, public water plants and other industrial facilities), causing high economic costs and having profound effects on the aquatic ecosystems they invade (Griffiths *et al.,* 1991; Pimmentel *et al.,* 2000; Bykova et al*.*, 2006; Ward & Ricciardi, 2007). Invasive fish, such as round goby (*Neogobius* *melanostomus*) or Asian carp (*Cyprinus carpio*), have also impacted freshwater ecosystems and reduced native fish populations (Kolar *et al.,* 2007; Freedman *et al.,* 2012; Kornis *et al.,* 2013).

In coastal ecosystems of North America, 298 non-indigenous species of invertebrates and algae have been recorded as naturalized (Ruiz *et al.,* 2000). Most non-indigenous species are crustaceans and molluscs and have resulted from ballast water, inferring that source regions of non-indigenous species differ among coasts, corresponding to local and global trade patterns. Further, at least 100 species of non-indigenous fish and 200 species of non-indigenous vascular plants are known to be established within North America coastal area (Ruiz *et al.,* 2000). North American mangroves are considered to be protected from invasions due to the harsh hydrological and edaphic conditions in which they grow. However, there is an increasing number of invasive species being reported in mangrove ecosystems associated to anthropogenic and natural disturbances (Lugo, 1998), including the Brazilian pepper *Schinus terebinthifolius* raddi (Anacardiaceae) in Florida (Ferriter, 1997) and the Indo-Pacific lionfish *Pterois volitans* (Linnaeus, 1758) (Scorpaenidae) from North Carolina to Caribbean (Barbour *et al.,* 2010).

Urban sprawl in North America is a major driver of landscape change and cities are a contributing source of invasive species to the surrounding rural or natural matrix. Ornamental plants, pets and pests have higher chances to adapt and invade natural systems as the propagule pressure (i.e. events of introduction) increases. Insects such as the argentine ants have also exploited human disturbances around cities (Holway *et al.,* 2002).

Mesoamerica and the Caribbean

As of 2006, Mexico's National Commission for the Knowledge and Use of Biodiversity identified at least 800 invasive species in Mexico, including 665 plants, 77 fishes, 2 amphibians, 8 reptiles, 30 birds and six mammals, with significant ecological and economic impacts.

Buffel grass (*Pennisetum ciliare*) has invaded many of the drylands in Mexico (Marshall *et al.,* 2012) after being introduced in the 1970s into Sonora from the USA to bolster the cattle industry (Cox *et al.,* 1988; De La Barrera & Castellanos, 2007; Franklin *et al.,* 2006). From 1973 to 2000, Buffel grass pastures in Mexico increased from 7,700 hectares to 140,000 hectares (Franklin *et al.,* 2006). It is estimated to cover 53% of Sonora and up to 12% of Mexico overall (Arriaga *et al.,* 2004). Buffel grass invasion can devastate local ecosystems by increasing wildfire regimes, soil erosion rates, ground surface temperatures and supply of vital resources to surrounding life forms, compromising biodiversity (D'antonio & Vitousek, 1992). Buffel grass is also present in Central American countries like Nicaragua, El Salvador, Honduras, and, Panama (Global Biodiversity Information Facility, 2011).

The southern Yucatán peninsular region is the largest continuous expanse of tropical forests remaining in Central America and Mexico, it has been identified as a hotspot of forest and biotic diversity loss (Achard *et al.,* 1998). Bracken fern (*Pteridium aquilinum* (L.) Kuhn) invasion have spread under agriculture cultivation (Schneider, 2006). Frequent fires and land clearance for agriculture have facilitated the replacement of secondary vegetation with bracken fern (Schneider & Nelun Fernando, 2010). The feral pig (*Sus scrofa*), from the same species as the European wild pig, has invaded the Coco's Island Marine and Land Conservation Area, a national park in the Costa Rican Pacific (Hernández *et al.,* 2002). Because of their rooting activity, these animals alter approximately 20% of the island surface each year, leading up to eight times the erosion in the affected area. These animals also eat fruits, earthworms, roots, stems and leaves, reducing the layer of organic material in leaf litter and plant cover.

Invasive insects are also wide spread throughout Mesoamerica. The Mediterranean fruit fly *(Ceratitis capilata)*, heads the list of invasive alien species of economic importance in the Mesoamerican region, and is considered a genuine pest affecting all Central American countries. This insect, which entered the region in 1955, attacks fruit and fills it with worms. As a result, some fruit exports from Central America to the USA were suspended. Fruit trade with Europe and Japan has also been affected.

In freshwater ecosystems, African cichlid fish, *Oreochromis* spp., were accidentally introduced in Lake Chichancanab two decades ago, in the central Yucatán Peninsula in Mexico, causing change in the native fish diversity and in the transmission of endemic trematodes to the piscivorous birds (Strecker, 2006). Nile tilapia *(Oreochromis niloticus)* is currently found in the Apoyo, Nicaragua and Managua lakes (Nicaragua), Caño Negro Wildlife Refuge, and Lake Arenal (Costa Rica). This species has resulted in a decline of approximately 80% in the biomass of native cichlidic fish in Lake Nicaragua and has displaced native fish in Caño Negro due to increased competition and predation.

Introduced fish species often result in alteration of food webs. Two exotic fish, common carp (*Cyprinus carpio*) and tilapia (*Oreochromis niloticus*), were introduced for aquaculture more than 20 years ago into the Xochimilco wetlands, Mexico City and now dominate the system in terms of biomass and numbers. Over this period, wild populations of the microendemic axolotl salamander (*Ambystoma mexicanum*) have been dramatically reduced (Zambrano *et al.,* 2010).

In the Mexican Caribbean, the Indo-Pacific lionfish (*Pterois volitans*) has become a species of great concern because of their predatory habits and rapid proliferation throughout the Mesoamerican Barrier Reef, the second largest continuous reef system in the world (Valdez-Moreno *et al.,* 2012). Having few predators, this invasive predatory fish can greatly reduce native fish biomass and is a threat to the marine environment throughout the region (Green *et al.,* 2012) **(Box 4.21)**.

The seaweed flora of California, USA and Baja California, Mexico is highly diverse and is now being threatened by invasive species that are largely introduced unintentionally. Most of the 29 non-native seaweed species that have been recorded, originated in Asia and have been introduced within the last 30 years. The vectors that bring these plants or their propagules to the California and Baja California coasts (international shipping (e.g. ballast water) and shellfish aquaculture) may have not changed drastically in the last decades, but the conditions for the establishment of non-native species seem to have improved. Climate change, including the frequency and severity of El Niño Southern Oscillation events, may be responsible for creating space, diminishing competition, and permitting the persistence and spread of non-native species (Miller *et al.,* 2011; Kaplanis & Smith, 2016).

In the Caribbean islands, humans have introduced many plant and animal species (Kairo *et al.,* 2003; Rojas & Acevedo, 2015; van der Burg *et al.,* 2012; Jenkins *et al.,* 2014), and non-native species have often become ubiquitous there. Caribbean terrestrial ecosystems have been heavily invaded by plants and animals. For example, forest inventories of various Caribbean islands, based on plots or remote sensing, have found that forests dominated by non-native tree species are extensive (Chinea & Helmer, 2003; Brandeis *et al.,* 2009; Helmer *et al.,* 2012), although some of these new tree communities may have a beneficial role. For example, early successional species often dominate and catalyze understory colonization by native tree species (Parrotta, 1992; Parrotta *et al.,* 1997; Wolf & van Bloem, 2012), or when legumes or nutrient-rich leaves attract insects that provide more forage for insectivorous birds. Shade-tolerant non-native species, however, can be common in forest understories (Brown *et al.,* 2006) and could permanently change species composition by effectively competing with late successional native species.

The marabú, (*Dichrostachys cinerea L.*), an invasive Fabaceae, has invaded almost 800,000 hectares of Cuba's forests (Hernández *et al.,* 2002). This thorny bush grows in forests and abandoned agricultural fields, leaving infested areas unproductive. Nowadays, marabú has become Cuba's primary problem with respect to invasive alien species, in terms of both economic and environmental impacts. Environmentally, the most serious damage is inflicted on fields (livestock) and on forest plantations. Lands invaded by marabú remain unusable and thorny, impassable for livestock and human beings. In its juvenile state, marabú is practically impenetrable since it forms extremely dense thickets up to five meters high. In the case of forest plantations, this invasive bush is highly expensive to control. The country spends millions of USA dollars a year to combat this species, but its great capacity for reproducing through seeds, trunks and roots makes it very difficult to eliminate. More information on invasive species in Cuba is presented in supplementary material: **Box 4.22**.

Many of the problems of Mesoamerican invaders in ocean ecosystems are repeated throughout the Caribbean. The Indo-Pacific lionfish (*Pterois volitans* and *P. miles*) waslikely introduced in the USA state of Florida through aquarium releases, and has quickly spread to all tropical and subtropical coastal waters of the western Atlantic Ocean and Caribbean Sea (Schofield, 2010). In fact, this species may be the most damaging marine fish invasion to date (Hixon *et al.,* 2016) (Supplementary material, **Box 4.21** and Mesoamerica section above).

South America

South America, due to its relative isolation, was until recently, considered to be relatively less affected by biological invasions (Speziale *et al.,* 2012). However, evidence has shown that biological invasions are occurring in ecosystems that were considered protected, such as the Andes mountains (Pauchard *et al.,* 2009), the Amazon basin (Silvério *et al.,* 2013), and the Patagonian south Atlantic coast (Oresanz *et al.,* 2002). These large and diverse ecosystems harbor a number of invasive species, including some of the world's worst invaders (Speziale *et al.,* 2012). The mongoose (*Herpestes javanicus*), introduced as a predator of rats and snakes, spread preying on endemic fauna and transmitting rabies and leptospirosis (Ziller *et al.,* 2005). Other introduced species act as ecosystem engineers, transforming and threatening complete ecosystems (Speziale *et al.,* 2012), as well as changing their services (e.g. beavers *Castor canadienses*; Anderson *et al.,* 2006 and **Box 4.23** in supplementary material and *Limnoperna fortunei,* Boltovskoy *et al.* 2015 and **Box 4.24**, in supplementary material). Crop species with important commercial value, have also become invasive. Pines (Pinaceae family) for example, used widely as a forestry cultivar, are invasive in both temperate and tropical regions because they have been planted extensively and have biological attributes that promote their invasiveness (Pauchard *et al.,* 2015).

Invasive species in South America come from all continents, although Europe is a major donor of invasive species, especially for plants (Van Kleunen *et al.,* 2015). Undoubtedly, the number of new introductions is increasing annually because of intensified trade and transport routes which is diversifying the source of invasions (Speziale *et al.,* 2012). Harbors, roads, airports, and cities are major sources for the entry of new species. For example, big metropolitan areas such as Sao Paulo, Santiago, or Buenos Aires are centers for the introduction of new invaders (e.g. Masi *et al.,* 2010). Also, the increase human footprint in the landscape (section 4.4.1), and the introduction of new species for cultivation, is increasing the chances for new invasions.

Invasive species can also come from within the same country. For example, introduced marmosets in southeastern Brazil have been reported as a potential threat to local biodiversity. Marmosets compete with other primate species and birds for resources (Lyra-Neves *et al.,* 2007), depredate birds and eggs (Galetti *et al.,* 2009), hybridize with conspecifics (Begotti & Landesmann, 2008), and transport new pathogens (Sales *et al.,* 2010).

Tropical and subtropical humid and dry forests are one of the most extensive ecosystems in South America and are being impacted by several species that mostly originated from other tropical areas in Asia and Africa. While many tropical forests appear to be substantially free of invasive species, some species are able to invade mainland forest ecosystems where canopy structure is naturally open, rainforests are fragmented or disturbed, or forests are exploited for crops or timber (Denslow & DeWalt, 2008). In addition, fires reportedly interact with grass invasion through a positive feedback cycle, causing a decline in tree cover, facilitating grass invasions, and increasing the likelihood of future fires. In the tropical dry forests of Bolivia, grasses have invaded the forest where disturbance coincides with seed dispersal by motor vehicles involved in logging activities (Veldman & Putz, 2010). In the tropical and subtropical forests of Brazil, some of the most invasive plants known by their ability to outcompete native species, are *Artocarpus heterophyllus* and *Hedychium coronarium* in tropical ombrophilous forest, *Hovenia dulcis* in subtropical ombrophilous forest and subtropical semi-deciduous forest, *Pinus taeda* and *Pinus elliottii* in subtropical ombrophilous forest and steppe, and *Tecoma stans* in tropical and subtropical semi-deciduous forest (Zenni & Ziller, 2011). Tropical forest biotas are susceptible to taxonomic homogenization (i.e. increasing levels of similarity and reduce biotic differentiation) due to the increase of some generalist invaders that replace more specialized native species (e.g. the Atlantic forest of northeast Brazil, Lôbo *et al.,* 2011).

Mediterranean forests, woodlands and scrub are one of the invasion hotspots of South America because of their high human footprint and climatic similarities with biomes in Europe and North America. Ruderal agricultural weeds, native to the Mediterranean region of Europe, are widely distributed and invade natural ecosystems, increasing homogenization and affecting ecosystem dynamics (e.g. intensifying fire regimes) (Jimenez *et al.,* 2008; Castro *et al.,* 2005). Animal invasions are also affecting the processes of this ecosystem. For example, the European rabbit (*Oryctolagus cuniculus)* exerts a profound herbivore pressure in the Mediterranean scrub (Camus *et al.,* 2008, Iriarte *et al.,* 2005).

Tropical savannas and grasslands have been heavily affected by invasive African grasses. African grasses are used for pasture improvement, recovery of degraded areas, and slope cover along highway and railway embankments (Reis *et al.,* 2003; Martins, 2006). Invasive grasses have been identified as a degradation driver of Colombian wetlands (Ricaurte *et al.,* 2014), while in the Cerrado biome of Brazil, they constitute a serious problem because they invade open areas (Pivello, 2014). Molasses grass (*Melinis minutiflora* P. Beauv.) accumulates more biomass than do most other species of the herbaceous stratum vegetation native to the Cerrado (Rossi *et al.,* 2014). The effect of invasive grass cover is especially high on the Cerrado-specialist species, whose proportion has consistently declined with increasing invasive dominance. Thus, invasive grasses reduce the floristic uniqueness of pristine vegetation physiognomies (Almeida-Neto *et al.,* 2010). In savannas and grasslands, invasive trees have become problematic. For example, the invasion by *Pinus elliottii* is one of the most serious threats to the remaining native Cerrado vegetation causing biodiversity losses (Abreu & Durigan, 2011).

Temperate grasslands in South America are highly threatened by invasive species because of their long history of agriculture and livestock usage that has causedinvasive species to become widely distributed. For example, in the Argentina pampas, introduced forage grasses, such as *Festuca arundinacea* and *Lolium multiflorum*, and weedy forbs such as *Carduus acanthoides*, heavily dominate secondary grasslands on former arable fields (Tognetti *et al.,* 2010) and invade native grassland remnants grazed by cattle (Perelman *et al.,* 2007; Tognetti & Chaneton, 2015).

Drylands and deserts of South America show relatively low numbers of invasive plant species (Fuentes *et al.,* 2013). However, some succulent plant invaders such as *Mesembryanthemum* spp are invading desert islands in northern Chile (Madrigal-González *et al.,* 2013) and invasive animals such as rabbits and feral goats are having a strong effect on vegetation and overall ecosystem dynamics (Meserve *et al.,* 2016).

Temperate and boreal forests and woodlands have a relatively low area in South America (see Chapter 3 for more details). However, they show a high level of endemism and represent the most southern forests in the world (Rozzi *et al.,* 2008). These forests are being invaded by herbs, shrubs, and trees mostly brought to Chile for agricultural use, erosion control, forestry, and ornamental use (Pauchard *et al.,* 2015). For example, *Acacia* and *Pinus* species arewidely used in forestry, and are a problem in the temperate forests of south-central Chile where they outcompete native vegetation and increase fire regimes (Fuentes-Ramirez *et al.,* 2011; Le Maitre *et al.,* 2011; Langdon *et al.,* 2010; Cobar-Carranza *et al.,* 2015). Several invasive vertebrates are also invading these forests (e.g. wild boar, red deer, mink; Iriarte *et al.* 2005), with the most damaging being the North American beaver, which has decimated forests (i.e. cutting and flooding) in the southern tip of the continent (Anderson *et al.,* 2006; see **Box 4.23**, supplementary material).

Although tundra and mountain grasslands are considered less invaded than lowland ecosystems, recent evidence shows that there is an increasing number of invasive plant species being established at higher elevations in the Andes (Pauchard *et al.,* 2009; Alexander *et al.,* 2016). Species, such as *Taraxacum officinale*, may have important impacts on pollination, reaching high elevations beyond the treeline (Muñoz *et al.,* 2005). As climate warming progresses, there is a greater chance of higher latitude and elevation plant invasions (Lembrecht *et al.,* 2015).

Freshwater ecosystems are suffering strong transformation due to invasive species. For example, *Limnoperna fortunei,* commonly known as golden mussel, have invaded major rivers of the Río de la Plata basin and associated tributary basins via ballast water. Because of the ecological effects caused in aquatic ecosystems and expenses incurred in industrial infrastructure, it is considered a high priority aquatic invasive species to be addressed at the regional level (Boltovskoy, 2015) (see **Box 4.24**, supplementary material). *Lithobates catesbeianus* native frog from the southeast of USA has colonized more than 75% of South America where it has been reported to be a highly effective predator, competitor, and vector of amphibian diseases (Laufer *et al.,* 2018). Climate change may have a potential synergistic effect on the invasion of this frog throughout the Atlantic forest biodiversity hotspot (Nori *et al.,* 2011). The microalgae *Didymosphenia geminata*, an invasive freshwater benthic diatom native to rivers of the Circumboreal region of Europe, was reported in Argentinean and Chilean freshwater rivers. Thisalgae has been characterized as one of the most aggressive invasions in recent history, resulting in severe ecological and economic impacts due to the velocity of expansion and the number of rivers affected (Jaramillo *et al.,* 2015).

In marine ecosystems of South America during the decades 1990-2000, ballast water, biofouling, and aquaculture vectors moved several coastal marine species from distant biogeographic provinces (e.g. Indo-Pacific and Asia) to coastal environments of America (Orensanz *et al.,* 2002; Salles & Correa da Silva Luz de Souza, 2004). These species have become invasive, resulting in negative effects on ecosystem services provided by various aquatic ecosystems. The golden mussel (*Limnoperna fortunei*) in the Río de la Plata basin have modified the provision of freshwater services (potable and industrial uses) (Boltovskoy, 2015) and food services (malacological resources) due to effects of predation on native malacofauna by *Rapana venosa* in the Río de la Plata (Brugnoli*et al.,* 2014) (**Box 4.25**, supplementary material). Finally, the Indo-Pacific lionfish (*Pterois volitans* and *P. miles*) affects food (fisheries) and cultural (tourism, recreation: diving) services at the north coast of South America (Colombia, Venezuela) due to predation of indigenous fish fauna of megadiverse coastal marine ecosystems (e.g. coral reefs) (**Box 4.21**, supplementary material). However, because of euryhaline and eurythermal features of this species, their expansion has not been constrained by the Amazon-Orinoco plume (Luiz*et al.,* 2013), being recently reported in the southeastern coast of Brazil (Ferreira *et al.,* 2015).

In the marine environments off Patagonian shelf and Chilean Pacific coast, a series of biological invasions including algae, mollusks, hydroids, bryozoans, ascidiaceans, and crustaceans (at least 41 invasive alien species) occurred with severe consequences for local biodiversity with economic impact (Bigatti*et al.,* 2008; Orensanz *et al.,* 2002; Penchaszadeh *et al.,* 2005). *Undaria pinnatifida* is a successful invasive seaweed widespread along the coast of Patagonia. Its presence is associated with a dramatic decrease in species richness and diversity of native seaweeds (Casas *et al.,* 2004; Irigoyen *et al.,* 2011). For Brazilian shelves, Lopes *et al.* (2009) have compiled information on the threat of invasive species. Currently, 66 invasive species have been recorded for the marine environment in Brazil from the following groups: phytoplankton (3), macroalgae (10), zooplankton (10), zoobenthos (38), fish (4), and pelagic bacteria (1) with different ecological and economics impacts in marine Brazilian ecosystems (Lopes *et al.,* 2009).

4.4.5 Overexploitation

Nature of the driver, its recent status and trend, and factors that influence its intensity

Overharvesting, or overexploitation, occurs when humans extract more of a natural resource than can be replaced naturally. This unsustainable practice threatens biodiversity and can degrade ecosystem services by reducing species populations below natural self-sustaining levels and disrupting ecosystem functions and species interactions. Overharvesting can happen in hunting, fishing, logging, groundwater mining, overgrazing, or the collection of wild plants and animals for medicine, decoration or for the pet trade. Harvested species are used as food, building and other industrial materials, medicines, fibers for clothing, ornamental items, as well as in other social and cultural aspects.

Growing human populations, rising incomes, consumer demand, expanding markets, and improved technology all contribute to overharvesting. Individuals, communities or corporations that have open and unregulated access to public goods like forests, aquifers, fisheries, and grazing lands can overexploit a shared resource to maximize short-term profits until it eventually becomes unavailable for the whole (Hardin, 1968). Harvesting natural resources is an essential part of livelihoods and economies of all worldviews. When people act in their own self-interests, they tend to consume as much of a scarce resource as possible, leading to overharvesting and in some cases extinction or resource depletion. Early examples include, the Steller's sea cow (*Hydrodamalisgigas*), once found throughout the Bering Sea, was hunted into extinction within 27 years of discovery for its meat, fat, and hide; and the passenger pigeon (*Ectopistesmigratorius*), once considered the most abundant bird species on the planet, was hunted to extinction over a few decades throughout North America (Bucher, 1992). There are many examples linking extinction to joint effects of harvesting and habitat change as extensive areas in eastern North America were converted to agriculture and urbanization.

Overexploitation of species often leads to cascading effects with sometimes irreversible impacts on trophic-level functions and can negatively affect the structure, dynamics, or quality of an ecosystem. This is particularly true if a habitat loses an apex predator which can result in a dramatic increase in the population of a prey species. In turn, the unchecked prey can overexploit their own food resources to their own demise and impact other species (Frank *et al.,* 2005; Borrvall & Ebenman, 2006; Heithaus *et al.,* 2008). Fishing down the food chain, where larger predatory fish, such as cod, tuna, and grouper, are targeted first, followed by smaller fish in the food chain, causes trophic level dysfunction (Pauly *et al.,* 1998). Some species require a sufficient density of individuals to reproduce and when reduced to smaller populations, they become vulnerable, suffering from lower genetic diversity and an increased likelihood of being eliminated by natural disasters or diseases (Lacy, 2000).

When a species in not able to reproduce faster than it is harvested, it becomes increasingly rare which can drive its price higher in the illegal wildlife trade. This in turn, increases the incentive to extract which can cause the population to eventually collapse (Brook *et al.,* 2008). Wildlife trade poses the challenge of separating legal from illegal trade (Broad *et al.,* 2003) and governments can deter such illegal trade by measures such as policies that strengthen enforcement, curb the demand, and expand international cooperation to stop the illegal trade.

Many countries are responding by implementing strategies that mitigate or avoid negative impacts of overharvesting such as strengthening management regulations and enforcement, providing incentives to fishermen, foresters and others to become long term stewards of the resource, through the establishment of effectively managed protected areas and no-take zones, as well as strengthening institutions and regulations to eliminate illegal wildlife trade and put in place sound practices to regulate legal exports/imports of vulnerable species. Tenure rights and other means of co-management are also ways in which local communities can have more say over their natural resources and long-term conservation. For example, territorial user rights in fisheries, such as those set up in Chile for the small scale artisanal fishing sector, provide incentives to maximize economic benefits and encourage greater stewardship of the resource to local communities. Individual transferable quotas or other catch share strategies can also be applied to larger scale fisheries to prevent collapses and restore declining fisheries although critics point to them being exclusionary and involve trade-offs, such as changes in fleet capacity, employment, and aggregation of fishery shares (Costello *et al.,* 2008). However, many States have implemented measures to manage the potentially disruptive effects of individual transferable quotas. These practices should be accompanied with investments in sustainable alternative livelihoods and wide-spread education that can inspire conservation of local habitats and species and promotes the ability of local institutions to implement and sustain conservation programs.

Terrestrial

Overharvesting of terrestrial species and resources is often driven by the pursuit of quick short-term gains without regard to the long-term effects. Illegal logging, for example, can include overharvesting of large tracks of forests or the selling of rare wood species. It is pervasive throughout Mesoamerica and South America and impacts many different stakeholders and communities that rely on timber for their livelihoods (Richards *et al.,* 2003). Capital-endowed actors as well as poor forest dwellers may drive overharvesting, albeit for different reasons (Pokorny *et al.,* 2016). Poor governance, corruption, and rampant demands for space to carry out socio-economic activities (e.g. cattle grazing) contribute to the problem. Curbing this problem is difficult. For example, in the Amazon region, timber companies, as well as illegal harvesters, seeking to adopt sustainable practices face challenges such as high investment costs, large transport distances, lack of capacity, and resources to implement environmental regulations (Pokorny *et al.,* 2016). The pattern of deforestation can be exacerbated once timber companies provide road access and infrastructure to previously intact areas, allowing small landholders to continue to overharvest with often no management or enforcement.

Unsustainable hunting and collection of species driven by market demand is another contributing factor of overharvesting. The animal diversity that Central and South America holds and the limited enforcement of wildlife trading laws creates a magnet for wildlife traffickers and the lucrative exotic pet trade. However, the sustainability level of harvest for the majority of species is unknown. Birds are the most trafficked for pets, but reptiles like iguanas, snakes, and turtles are highly valued as pets as well as for their skin, shells, and eggs (Shirey *et al.,* 2013). Amphibians, scorpions, spiders, and insects are also collected (Ripple *et al.,* 2015; Broad *et al.,* 2003). Products are often sold for ornaments and furnishings include coral, turtle and mollusk shells, and reptile skins (Shirey *et al.,* 2013), many other products are sold as traditional "medicine" especially to Asian countries. In addition to the pet trade, there is an estimated eight million people in South America that rely regularly on bushmeat as a source of protein in their diets. While this represents only 1.4% to 2.2% of the total continental population, these people are likely to be some of the poorest in the region (Wilkie & Godoy, 2001). The distinction between subsistence and commercial use is often unclear and more research is needed on subsistence vs non-subsistence harvesting and how much of subsistence harvesting is optional but local (i.e. they have other sources but choose to eat bushmeat when available).

Plants and fungi provide people with food, medicine, building materials, and as raw materials for making other products. Some species are highly valued for their beauty or medicinal value. Thousands of medicinal and aromatic plants that are collected in the Americas are used in the international trade and are valued at over $1.3 billion (Lange, 1998). Many species of ornamental plants, like flowers, orchids, tree ferns, bromeliads, cycads, palms, and cacti, are commercially overexploited in both legal and illegal markets. For example, orchids throughout North and South America are one of the best-selling in the legal horticultural trade but are also traded illegally and make up 70% of all species listed by the Convention on the International Trade in Endangered Species (CITES). Research conducted by Hinsley *et al.* (2015) in the Americas indicates that two key consumer groups purchasing rare plants are either serious hobbyists, who prefer rare species, or mass market buyers whose preferences are based on aesthetic attributes.

Freshwater resources

The Americas show wide variation in overexploitation of surface and groundwater resources. Large portions of South and Central America, Canada, and Alaska are relatively water secure, while the western half of the USA, nearly all of Mexico and the Caribbean, and coastal portions of South America all experience seasonal and dry year water depletion (Brauman *et al.,* 2016). Climate change is expected to exacerbate water shortages in many parts of the Americas (UNEP, 2010; IPCC, 2014a).

Surface water depletion can have visible impacts as streams dry up, but groundwater depletion is no less serious and can have longer-term consequences. Sustained groundwater pumping can lead to drying up of wells, reduction of water in streams and lakes, deterioration of water quality, increased pumping costs, and land subsidence (Konikow, 2013). Depletion of ground water in the USA is a serious problem as aquifers provide drinking water for about half the total population and nearly all rural population as well as providing over 50 billion gallons per day for agricultural needs. The cumulative depletion of groundwater in the USA between 1900 and 2008 was about 1,000 km^3—equivalent to about twice the water volume of Lake Erie (Konikow, 2013).

Irrigation is by far the largest source of water consumption globally and in the Americas. Domestic use is the second largest consumer in North and Central America, while in South America livestock production is slightly higher (Brauman *et al.,* 2016). Overharvest of water in general has implications not only for human communities, both in terms of water quality and quantity, but also for aquatic and even terrestrial species whose life cycles are adapted to natural flow regimes (Poff *et al.,* 1997).

Impacts to species from overexploitation of water largely track where that overexploitation is greatest. An analysis of species listed as extinct through vulnerable in the IUCN Red List finds that only 5% of assessed species associated with South American inland wetlands are threatened by water abstraction, whereas the numbers rise to 17% in Mesoamerica and 32% in North America (IUCN Red List, 2016). These numbers should be interpreted with caution, given that comprehensive species assessments are lacking for much of Latin America. Overharvesting of freshwater species in the Americas is considered in general less of a threat to biodiversity and ecosystem services than the degradation and alteration of the habitats in which those species live (Welcomme *et al.,* 2011). However, overharvest can combine with those impacts, which include but are not limited to changes to hydrology, connectivity, and water quality, to impair species and services further (Allan *et al.,* 2005).

Freshwater species

Globally, most inland fisheries are comprised of small-scale fishers, whose catches are underreported by as much as a factor of two (Coates, 1995; Mills *et al.,* 2011). Even with underreporting the level of fisheries exploitation in Latin America has been judged to be lower than in Africa and Asia; however, specific fisheries show signs of overharvest (Welcomme *et al.,* 2011; Muller-Karger *et al.,* 2017). For instance, overfishing of valuable freshwater fish species and turtles has been documented in tributaries of the Amazon (Alho *et al.,* 2015). In general, national governments have underinvested in monitoring inland fisheries because those fisheries are assumed to be of low value. Consequently, the range of threats to those fisheries, including overexploitation, are poorly documented (FAO Committee on Fisheries, 2014).

Cascading effects of freshwater overharvesting are numerous and include the phenomenon of "fishing down", in which exploitation leads to depletion of high-value, large-bodied fish species and the consequent reduction of mean body size of harvested species (Welcomme, 1999; Pauly & Palomares, 2005). This has been documented in the Amazon and elsewhere, with implications for food web structure, water quality, and nutrient cycles; these changes, in turn, have been implicated in the ecological extinction of species like manatees (Castello *et al.,* 2013; Castello *et al.,* 2015).

Marine

The most significant driver of overharvesting in the marine environment is fishing. With population growth and incomes rising, the demand for seafood continues to grow for both human consumption and feed for livestock and aquaculture. Fishing remains a key source of food and employment for millions of people in the Americas and a significant factor in regional economies. About 2.4 million fishers and 10% of the world's motorized fishing vessels are in the Americas (FAO, 2016c), landing 18.5 million metric tons of seafood in 2013 (FAO, 2016b). From 1961 to 2013, the per capita annual seafood consumption in the Americas rose 26% from 7.9 to 10.7 kg (FAO, 2016a). Different large marine ecosystems of the Americas (Sherman *et al.,* 2005; Sherman & Hamukuaya, 2016) show different top-down pressures and strong regional differences in oceanographic properties which shape the diversity and abundance of the catch within these regions (Muller-Karger *et al.,* 2017). The adoption of more efficient fishing technologies has also contributed to the rapid depletion of fish stocks, the endangerment of charismatic marine species, and the loss and degradation of marine habitats. An estimated 34% of the assessed stocks in geographic areas surrounding the Americas (FAO regions 67, 77, 87, 21, 31, and 41) were deemed overexploited in 2009 (FAO, 2011). However, the adoption of fishing technologies has been documented to have positive effects as well, such as much lower bycatches and less habitat impacts.

Invertebrates like squids, shrimps, lobsters, crabs, oysters, and sea cucumbers account for roughly 20% – 3.7 million tons – of the seafood caught in the Americas in 2013 (FishStatJ, 2016). Many of these fisheries and their habitats are at risk from overexploitation. For example, 85% of the world's oyster reefs have disappeared since the late 19th century, largely due to habitat degradation, with many formerly prolific reefs rendered "functionally extinct." Overharvesting is the main cause of oyster reef loss, however direct habitat loss is also a significant problem caused by commercial ship traffic, pollution, and aquaculture, among others. Other invertebrates, like seas cucumbers, have plummeted across the Americas due to high demand from Asian markets.

A consequence of fishing is the unintended catch of fish and other marine organisms, also known as bycatch. Hundreds of thousands of sea turtles, seabirds, whales, dolphins, and porpoises die globally each year from being caught as bycatch in regular fishing operations. As many as 200,000 loggerhead turtles and 50,000 critically endangered leatherbacks were killed as bycatch on longlines in 2000 (Lewison *et al.,* 2004); longlining is also estimated to kill between 160,000 to 320,000 seabirds annually (Anderson *et al.,* 2011). Several studies report that the use of bycatch reduction devices can successfully reduce bycatch species while maintaining target catch rates (Favaro & Côté, 2013; Pelc *et al.,* 2015). The vaquita, a small porpoise in Mexico's Gulf of California, have been driven towards extinction as they are killed after getting entangled in gillnets used to catch shrimp and other fish; only 30 are estimated to remain (Morell, 2017).

Sharks and rays are severely overfished globally, with an estimated 97 million caught each year either in direct target fisheries or as bycatch in other fisheries (Clarke *et al.,* 2013). One-quarter of the 1,041 assessed sharks, rays, and chimaeras are threatened under the IUCN Red List criteria due to overfishing, however nearly half are considered too data-deficient to be classified. Many shark species are pelagic and migratory—some with a circumglobal distribution across temperate and tropical oceans—meaning that overharvesting of sharks in the Americas contribute to a global problem. Only 23 sharks and rays had been listed under CITES up to 2016, when an additional 13 species of sharks and rays were listed. Trade restrictions on listed species and bans on shark finning have increased during the last decade, however they have not significantly reduced shark mortality or risk to threatened species (Davidson *et al.,* 2016). Some countries, such as The Bahamas, have implemented a national ban on the harvest of sharks, protecting more than 40 species of sharks.

Additional drivers of overharvesting in the America's marine environment include hunting, aquarium trade, medicinal use, and entanglement in fishing and marine gears. Turtles, narwhals, and corals are harvested for ornamental and jewellery making, and live fish, corals, and invertebrates are harvested for the aquarium and pet trade. Some species like sea horses are also targeted for traditional medicinal use primarily in Asian markets. Direct harvest of non-fish species, like seals, otters and whales, has seen a reduction since the peak of these industries almost a century ago, but some of these species continue to be harvested, particularly in Canada. An estimated 308,000 whales and dolphins die each year from the consequences of entanglement in fishing gear, laceration, infection, and starvation) (International Whaling Commission https://iwc.int/entanglement).

North America

Terrestrial

An example of an overharvested plant in North America is American ginseng (*Panax quinquefolius*), a species found in the temperate eastern forests and is prized for its medicinal properties that has received increased scientific and commercial attention. Due to the plant's very specialized growing environment and demand in the commercial market, it has started to reach an endangered status in some areas (McGraw *et al.,* 2010). Acts, such as the Endangered Species Act, have succeeded in reducing the harvest of rare species, preventing the extinction of hundreds of additional American wildlife species since 1973 (Adkins, 2016).

Freshwater

While loss of spawning beds and pollution contributed, overfishing in the Great Lakes is a good example of inland surface water overharvesting that has caused whitefish, walleye, and sturgeon populations to decline. Recreational fisheries are also poorly documented, by and large; in Canada, however, the collapse of four inland fisheries has been associated with recreational fishing (Cooke & Cowx, 2004). Within coastal and inland rivers, the well-documented decline of Pacific salmon and other anadromous fish species as a result of overfishing, dams, and other threats has led to cascading effects including the loss of nutrient inputs to terrestrial systems (Marcarelli *et al.,* 2014). Four native freshwater turtle species (*Chelydra serpentine*, *Apalone ferox*, *Apalone mutica*, and *Apalone spinifera*) now require increased protection driven by trade to Asia (USFW, 2014).

Marine

In North America, fishing remains the primary driver of overharvesting in the marine environment. In the USA, fish stocks are generally well-managed, at least at the federal level. For the 233 stocks with known status only 16% are overharvested, while overharvesting occurs in only 9% of the 313 stocks with known status (NOAA, 2016). Several overharvested species have been well-documented, like the collapse of the Atlantic cod of the Scotian bank, which provides a classic example of overharvesting that resulted in the closure of a 9,600 square miles area in 1994 (Frank *et al.,* 2005). There has been a reduction in the direct harvest of marine mammals that have historically been overharvested, like seals, otters, and whales since the peak of these industries almost a century ago. For example, sea otter (*Enhydra lutris*) hunts peaked in the middle of the 1800s when the species was almost driven to extinction by the fur trade. Sea otters were listed under the U.S. Endangered Species Act in 1977 and designated endangered in Canada in 1978, and most of their historical range has been reoccupied, but their numbers are still considered low in some areas (Bodkin, 2014). For oyster reefs, overharvesting remains a serious problem as about three-quarters of the world's remaining wild oyster reefs are found in just five locations in North America, however only in one of these regions — the Gulf of Mexico — are oyster populations deemed relatively healthy as of 2011 (Beck *et al.,* 2011).

Several policies have reduced or eliminated the harvesting of selected species like the U.S. Marine Mammal Protection Act of 1972 that established a moratorium on the taking of marine mammals in USA waters and the USA passed the Endangered Species Act (1973) that restricts harvests of critically imperilled species. In 1973, CITES was established to ensure that international trade of animals and plants does not threaten their survival in the wild. Canada and the USA often use allocation of fishing rights and use of protected areas to manage fisheries in federal waters, with agencies establishing quotas using robust stock assessments and monitoring programs. Examples of overharvesting in North America Artic and Greenland are presented in **Box 4.14**.

Mesoamerica

Terrestrial

Mesoamerica provides an important corridor for many Neotropical migrant bird species and home to rare and charismatic species like the scarlet parrot, ocelot, beaded lizard, river turtle, and the iconic jaguar that are threatened by the illegal pet trade. Butterflies, reptile leather, shark fin are also popular items on the black market. In the tropical dry and humid forests, several valuable tree species like mahogany and black rosewood are increasingly in demand and being cut and smuggled into markets in India and China by organized crime (Dudley *et al.,* 2014; Blaser *et al.,* 2015). In 2016, rosewood species have been included in CITES. The southern border of the USA is also a hot zone for wildlife smuggling based on the nearly 50,000 illegal shipments of wildlife and wildlife products that were seized at ports of entry from 2005 through 2014. This included nearly 55,000 live animals and three million pounds of wildlife products (Defenders of Wildlife, 2016)

Marine

While most high migratory species are assessed and well-managed through multinational efforts in Mesoamerica, many coastal fish stocks are considered to be overfished or declining (FAO, 2011). Examples of locally overfished

Box 4 14 Overharvesting in North America Artic and Greenland.

Several fisheries studies in northeastern Canada and Greenland observe species overharvesting which can lead to cascading effects and modification of food webs (Jørgensen *et al.,* 2014; Shelton & Morgan, 2014; Munden, 2013). Overexploited fish species include Atlantic cod (*Gadus morhua*), Atlantic halibut (*Hippoglossus hippoglossus*), redfish (*Sebastes mentella*), Atlantic wolfish (*Anarhichas lupus*), starry ray (*Rujuradiuta*), and American plaice (*Hippoglossoides platessoides*). Deep-sea fish species are particularly vulnerable to overexploitation as they mature late and have a low fecundity and slow growth rate (Jorgensen *et al.,* 2014). Barkley (2015) reports two key datasets to develop sustainable harvest levels for Greenland halibut (*Reinhardtius hippoglossoides*) in the Canadian Arctic and understanding the stock connectivity between inshore and offshore environments as well as examining capture induced stress metabolites in Greenland halibut caught in a trawl and Greenland sharks (*Somniosus microcephalus*) caught as bycatch on bottom longlines.

Mortality of non-target species, or bycatch, is a fisheries management problem that can be solved with innovative fishing gear and practices. Traditional fishing gears, like trawls, not only contribute to bycatch, but can greatly modify marine habitat. FAO (2016c) reports that 35% of landings are bycatch with at least 8% being thrown back into the sea. In Newfoundland, Munden (2013) found that impacts of bycatch and habitat alteration can be mitigated through gear modification. She found that a modified shrimp trawl can reduce contact area by 39% while increasing shrimp harvesting by 23%. A change in the type of gangions can lead to a significant reduction in shark bycatch without negatively impacting commercial catches of turbot (*Scophthalmus maximus*). In Davis Strait, West Greenland, one of the world's largest cold-water shrimp fisheries, with an annual catch of about 80,000 tons, bottom trawls have excessively modified bottom habitats and community structures (Pedersen *et al.,* 2004). Jorgensen *et al.* (2014) studied nine bycatch species from bottom-trawl surveys of Greenland halibut over a 24-year period and found that four populations showed a significant reduction in mean weight of individuals that was significantly correlated with increases in fishing effort.

species groups throughout Mesoamerica include crabs, sea-spiders, and shrimp, as well as various demersal fish (croakers, snappers, groupers) that form a large portion of the bycatch from shrimp fisheries (FAO, 2011). The vaquita have also become overfished to endangerment in recent years after becoming entangled bycatch in gillnets set for the totoaba, a large white fish (Morell, 2017). The overharvesting of sea turtles continues to be a problem as all seven species of sea turtles are threatened by the sale of meat, jewelry, and leather products. Their eggs are sold on a thriving Central American market as a male aphrodisiac. Heavy exploitation of sea turtles in the Mexican and Caribbean regions began in the 15th century. In the 1970s, sea turtles were added to Appendix I of CITES, banning commercial trade between member states. Despite CITES and U.S. Endangered Species Act listing, sea turtles are still declining. Turtles also die in huge numbers entangled in the nets of fishers. Another species threatened by trade and illegal harvest is Mexico's totoaba, an endangered fish endemic to the Gulf of California. Totoaba are valued for its swim bladders, used to make a specialty soup, and individual fish can be sold for $10,000 to $20,000 apiece in the Asian market (Neme, 2016). Sea cucumbers also remain overexploited throughout Mesoamerica, driven by lucrative export markets to Asian countries (Purcell *et al.,* 2013). Effective fisheries management regulations and capacity are lacking in many parts of Mesoamerica. In cases where management systems do exist, they are often jeopardized by data deficiencies, a lack of enforcement and monitoring, and corruption. Lack of effective management has led to *de facto* open access and overfishing.

Caribbean

Marine

According to the FAO, the Caribbean Sea (FAO area 31) has the highest proportion of overfished stocks in the world, about 54% in 2009 (FAO, 2011). Long-term catch data suggest that fish catches in the Caribbean increased by about 800% since 1950, and have been declining since 2001. Conclusions about the recent declines in fish landings as indicators of the status of fish stocks can only be made with very low certainty as the fish landings data comprise multiple fish species across many trophic levels, data sources have changed over the years, and landings from artisanal fishers are thought to be unreported. However, it is likely that the declining trend in fish landings indicate decreases in the size of fish stocks across the region (Agard *et al.,* 2007).

Overfishing is affecting virtually all Atlantic coral reefs and particularly in the Caribbean, with almost 70% of reefs at medium or high risk (Burke *et al.,* 2011). Atlantic reefs have some of the lowest recorded fish biomass measures within reef habitats in the world – largely from overfishing (Burke *et al.,* 2011; Jackson *et al.,* 2014). While the Caribbean only supplies a small percentage of the global trade in marine ornamental species, the environmental and biological impacts of the industry are well recognized. At least 16 Caribbean countries have export markets for ornamental reef fish, with the biggest markets being the USA, the European Union, and Japan. The impacts of the ornamental reef fish industry include the overharvesting of key species, coral reef degradation associated with gear impacts and from use of cyanide and other poisons, changes in the ecology of the reefs due to focused collection of specific trophic groups like herbivores, and loss of biodiversity due to removal of rare species (Bruckner, 2005). While less than 1% of the stony corals that have been reported to CITES database originate from the western Atlantic reefs, the USA and most Caribbean nations have prohibited the trade of stony corals. Hundreds of other genera of invertebrates, including echinoderms, sponges, molluscs and crustaceans are also collected and exported from the western Atlantic, primarily for the aquarium trade (Bruckner, 2005). An additional case study on queen conch in the wider Caribbean is explained in **Box 4.15**.

South America

Terrestrial

South America is home to a multitude of species that are highly prized for the pet trade, bush meat, and traditional medicines. Many of these species are harvested by indigenous peoples and sold to traffickers. The wildlife trade affects endangered and valuable birds, mammals, reptiles and amphibians, fish, and rare trees and plants. Some bird species, like the blue-throated macaw (*Ara glaucogularis*) are prized for their brilliant color and command a high dollar price on the illegal pet trade. Estimates of annual bushmeat consumption for the Brazilian Amazon are estimated at 89,000 tons (Peres, 2000 in Ripple *et al.,* 2015). In remote forest areas, eating bushmeat may be a matter of survival, being often the main (or only) source of animal protein available. When wild fish is available the role of bushmeat in people's diets may drop, thereby their consumption seems to be closely linked to both availability and/or prices (e.g. Rushton *et al.,* 2005 in Peru; Nasi *et al.,* 2011). As a cascade effect, a decline in one wild resource may drive up an unsustainable exploitation of the other (Brashares *et al.,* 2004; Nasi *et al.,* 2008 in Nasi *et al.,* 2011). Nevertheless, for richer sectors of society, bushmeat is harvested for sports hunters and as a novelty food for tourist in high-end restaurants in the region.

Freshwater

Manatees (*Trichechus inunguis*) and giant otters (*Pteronura brasiliensis*) are the most demanded aquatic species of mammals found in wetlands with very high demand as food and leather, respectively. Caimans (black giant caiman, *Melanosuchus niger*, and spectacled caiman, *Caiman* spp.), the Orinoco crocodile (*Crocodylu sintermedius*) and river turtles (mainly the Amazon giant turtle – *Podocnemis expansa*) are under strong harvesting pressure in the wetlands. While caimans are still found in healthy and very abundant populations in more remote areas, clear of human interference, river turtles struggle to resist to very high harvest rates (Seijas *et al.,* 2010; Rhodin *et al.,* 2011). In the Amazon and Pantanal, the overexploitation of large frugivorous fish may affect the dispersal of seeds within wetlands covering 15% of South America by area (Correa *et al.,* 2015). Ornamental fish are caught in large numbers in the Amazon, and there is evidence of overharvest of species like the cardinal tetra (*Paracheirodon axelrodi*) (Begossi, 2010).

Box 4 15 Overharvesting of queen conch in the wider Caribbean.

With a life span of up to 40 years, the queen conch (*Strombus gigas*) is a unique marine mollusc found in tropical waters throughout the wider Caribbean, Bermuda and the Gulf of Mexico. Its shell is emblematic of the oceans it inhabits with many cultures referring to conch shells as a "megaphone" for hearing the ocean's sound. In addition to the ornamental use of its shell, conch shells are used in jewelry making.The meat is consumed throughout the Caribbean and exported as a seafood product to the USA, France and other countries. Live queen conch are also sold in the aquarium trade. Because of its slow growth and density requirements to reproduce, queen conch are easily overharvested and the Americas have plenty of cases where this overharvesting is evident (Appeldoorn *et al.,* 2011).

In the USA, Florida's queen conch fishery collapsed in the 1970s and today both recreational and commercial harvests of queen conch are prohibited in the State. Demand for queen conch however remains high. Since the 1980s, commercial catch has increased in response to international market demand, especially from the USA, which imported approximately 80% of the annual queen conch catch in 2004 (Paris *et al.,* 2008). Regulatory measures to manage queen conch stocks in the region vary considerably throughout the Caribbean (Berg & Olsen, 1989; Chakalall & Cochrane, 1997). Some countries have minimum size restrictions on harvested conchs; others have closed seasons, harvest quotas, gear restrictions, spatial closures, or a combination of these; however in management response at all levels, from regional to local, has been slow in tackling overexploitation (Appeldoorn *et al.,* 2011). In 1992, queen conch became the first large-scale fisheries product regulated under Appendix II of CITES. Appendix II includes species that are not necessarily threatened with extinction, but unless trade is strictly controlled, may become extinct. Despite CITES listing, conservation actions and management policies, few countries report substantial recovery of queen conch populations, which may be due to reduced densities that limit reproduction (Stoner & Ray, 1996; Stoner, 1997; Paris *et al.,* 2008). More science, monitoring and management action will be required to put conch on the path to recovery and it will take time, resources and political will to achieve sustainability of this emblematic species.

Even though Amazonian wetland forests are the most diverse in the world (Wittmann *et al.,* 2006) and exploited for timber for many decades (Castello *et al.,* 2013), quite a small number (N=14) of tree species were considered especially vulnerable (Ribeiro, 2007). Forest products for manufacturing and construction include timber, rattan and bamboo for furniture, plant oils and gums, dyes, resins and latex (Shirey *et al.,* 2013). Some species, like mahogany (*Swietenia macrophylla*), are highly valued commercially for its beauty, durability, and color. It is estimated that approximately 58 million hectares (21%) of mahogany's historic range had been lost to forest conversion by 2001 (Grogan *et al.,* 2010). Commercial exploitation has sometimes led to traditional medicines becoming unavailable to the indigenous peoples that have relied on them for centuries or millennia. The fate of remaining mahogany stocks in South America will depend on transforming current forest management practices into sustainable production systems. Given the potential costs and benefits associated with trade, the challenges suggest that a collaborative approach between agencies, nurseries, and plant collectors is needed to regulate the trade of listed plants. There is a substantial international trade and demand for products like Brazil nuts, palm hearts, pine nuts, mushrooms and spices (Shirey *et al.,* 2013). In regulating commercial trade, policymakers and conservation biologists may want to consider potential risks and benefits of private efforts to recover species (Shirey *et al.,* 2013). More details on overharvesting in Amazonian wetlands are presented in **Box 4.16**.

Marine

While just over 27% of assessed fish stocks on the Pacific coast of South America are considered overexploited, roughly 69% of assessed fish stocks are overfished on the Atlantic coast. Conversely, 59% of unassessed stocks on the Pacific coast of South America are estimated to be overexploited, while 53% of assessed fish stocks are estimated to be overfished on the Atlantic coast (FAO, 2011; Hilborn & Ovando, 2014).

The Humboldt Current moves cold Antarctic waters along the western coast of South America and drives upwelling of nutrient-rich water, making the coastal shelf one of the most productive marine environments in the world. Large environmental variations are known to cause large year-to-year fluctuations as well as longer-term changes in fish abundance and total production of the main exploited species (FAO, 2011). The world's largest fishery by volume, the anchoveta, is targeted mainly by Peru and Chile. Overfishing played a major role in the collapse of the anchoveta fishery in 1973, 1983, and again in 1998, however it is also recognized that environmental conditions also significantly influenced the decline (FAO, 2016). More recently, the adoption of an individual quota system for the industrial sector of the fleet and other management measures have contributed to reducing the excess industrial fishing capacity for anchoveta. The small and medium scale sector still need reforms, but the fishery is considered by fisheries scientists to be managed within sustainable limits.

Additionally, local populations of sea urchins, clams, scallops, and other shellfishes have been overexploited in some areas (FAO, 2011). As coastal stocks decline, commercial fishers continue to move further offshore in search of higher trophic-level species that are more valuable. Lack of effective fisheries management has also led to illegal, unreported, and unregulated fishing, and exploitation by foreign fleets. The bycatch of seabirds, marine mammals, and sea turtles is thought to be significant in both southwest Atlantic and southeast Pacific for gillnet and driftnet fishing gears, although there are large data gaps in the existing

Box 4 16 Amazonian wetlands.

In general, overexploitation of Amazonian wetland species has two types: timber species and fish species. Main reasons include strong market pressures from an increasing affluent urban population, unregulation of markets, and adoption of unsustainable techniques of extraction and/or production of resources, reduction of stocks, depletion and even extinctions. The Amazon human population is very dependent on local fisheries for their animal protein intake. Fish consumption is among the highest in the world. And almost 50% of the fished species exploited (and more than 60% of the biomass estimate of 450,000 tons produced annually) is directly related to the Amazonian wetlands, where they use either as spawning grounds or as nurseries to larval stages. As a very selective activity, this fishery exploits only a small fraction of the local fish diversity. Consequently, many stocks of the larger species exploited are already overfished, mainly in the more populated areas of the Amazonian wetlands (Junk *et al.,* 2007). Although almost two hundred species of fish are of commercial value, fish yields are dominated by 18 to 20 species only. There was a reduction in the mean maximum body length of the main species harvested in 1895 (circa 206 cm) to the main species harvested in 2007 (circa 79 cm). From the group of species harvested in the early 19th century, three are now endangered. From the 18 species dominating yields nowadays, one is endangered and four were found to be overexploited in at least one region of the Amazon basin (Castello *et al.,* 2013). Modern technologies allow fishermen to explore more distant places, to travel longer and further, and to catch and store a higher amount of fish biomass.

knowledge its extent and contribution to the overexploitation of marine species (Wiedenfeld *et al.,* 2015).

In the Americas, incorporation of traditional values, knowledge, and social taboos within indigenous communities is increasingly being recognized as a fundamental part of effective resource management (Colding & Folke, 2001; Heyman *et al.,* 2001; Moller *et al.,* 2004; Fraser *et al.,* 2006; Herrmann, 2006). Trends are towards participatory, inclusive, community-based approaches to conservation (Berkes, 2007) that provides a sense of ownership and promotes self-management. Traditional ecological knowledge within indigenous communities accumulates across multiple generations and is learned through years of observations in nature (Drew, 2005). Invaluable local insight provides a deep understanding of the critical balance to maintain ecological integrity within an environment and it fosters shared responsibilities between locals and the science community. Moller *et al.* (2004) suggest that by combination traditional ecological knowledge and science, insight can be gained into prey population dynamics as well sustainable wildlife harvests. By doing so, partnerships and community buy in is garnered and indigenous users develop their own adaptive management actions which are often more effective since they have greater investment in having a sustainable resource.

4.5 DIRECT NATURAL DRIVERS

Nature of the driver, its recent status, and trends and what influences its intensity

Direct natural drivers of biodiversity loss include large environmental disturbances. Effects of disturbance on biodiversity have been studied in many ecosystems (Dornelas, 2010; Vega-Rodriguez *et al.,* 2015). The types of disturbance include everything from single tree-falls (Brokaw, 1985) to ecological catastrophes (Hughes, 1994).

Natural disturbances are caused by natural climatic, geologic, and biological fluctuations. Large, severe disturbances are often considered natural disasters, because they can threaten human life and have striking short-term effects on plant and animal populations (Lindenmayer *et al.,* 2009). They are often event-triggered by natural hazards that overwhelm local response capacity and seriously affect the social and economic development of a region (United Nations & The World Bank, 2010).

Globally, natural hazards are classified as: geophysical (e.g. earthquake, volcano, mass movement; meteorological (short-lived/small to meso scale atmospheric processes, e.g. storms); hydrological (e.g. flood, wet mass movement, climatological (long-lived/meso to macro scale processes, e.g. extreme temperature, drought, wildfire), or biological (e.g. epidemic, insect infestation, animal stampede) (Guha *et al.,* 2014). Biological disasters are not included in this assessment.

Sources of risk are both natural and man-made. Ecosystem structure can ameliorate "natural" hazards and disruptive natural events. For example, vegetative structure can reduce potentially catastrophic effects of storms, floods, and droughts through its storage capacity and surface resistance whilecoral reefs can reduce wave energy and protect adjacent coastlines from storm damage (de Groot *et al.,* 2002). Forests and riparian wetlands or coastal ecosystems like vegetated dunes, mangroves, coral reefs and sea-grass, reduce exposure to natural hazards by acting as natural buffers and protective barriers that, reducing the impacts of extreme natural events like landslides, tidal waves or tsunamis (Welle *et al.,* 2012; Rodil *et al.,* 2015). Consequently, environmental degradation directly magnifies the risk natural hazards by destroying natural barriers, leaveing human settlements and socioeconomic activities more vulnerable.

Climate change is predicted to increase the frequency of high-intensity storms in selected ocean basins depending on the climate model. The majority of tropical hurricanes damage from climate change tends to be concentrated in North America and the Caribbean–Central American region (Mendelsohn *et al.,* 2012). Increasing water temperatures along the Pacific coast through strong El Niño conditions and global warming can increase hurricane intensity. Although rare, more subtropical hurricanes have developed in the South Atlantic Ocean near Brazil. Changes in global atmospheric circulation patterns accompaning La Niña are responsible for weather extremes in parts of the world that are typically opposite to the El Niño changes.

The Americas suffered from 74 natural disasters in 2013 (Guha-Sapir *et al.,* 2014). Hydrological disasters (43.2%) and meteorological disasters (31.1%) occurred most often, followed by climatological (20.3%) and geophysical (5.4%) disasters. Globally, the Americas (22.2%) was only second after Asia (40.7%) in experiencing natural disasters in 2013. The nature of the risk, however, is different for different subregions of the Americas as presented below.

North America

North America has a vast range of natural disasters per year with hurricanes being one of the most common. The prevailing winds in the tropical latitudes of the Northern Hemisphere, where tropical hurricanes typically form, blow from east to west directing hurricanes to the eastern and southern coasts of the USA the islands of the Caribbean, Central America, and Mexico (see next sections). Hurricanes on eastern coasts can venture much further north due to the influence of warm waters of the Gulf stream. The west coast of Central America and Mexico are often affected by severe topical storms in the Pacific Ocean, or storms that cross from the Atlantic to the Pacific Ocean. Hurricanes, tornadoes, and other ecological disturbances alter structure and create periodic forest clearings. Hurricane Katrina (a category 5 storm) was the second costliest disaster, with total losses of $140 billion (in US 2010 values) (Wirtz *et al.,* 2014). The aftermath resulted in an estimated loss of 320 million trees in Louisiana and Mississippi in 2005 (Hanson *et al.,* 2010). Florida, in particular, is one of the most hurricane-prone areas in the USA (Leatherman & Defraene, 2006). Delphin *et al.* (2013) project major hurricane-related losses in two key ecosystem services over time: aboveground carbon storage and timber volume. Other ecosystem services that are at risk due to impacts of severe storms include storm protection from coral reef and mangroves, and other benefits obtained from low-lying coastal habitats. In the west coast of the USA, major landslides have been associated with El Nino events, especially in California State, mainly from intense rainfall (Godt *et al.,* 1999).

Earthquake and volcanic events occur along plate boundaries in the west coast. Volcanic eruptions are active in the hot spot zone of Hawaii and in the North Pacific region including volcanoes in Alaska, the Aleutian Islands, and the Kamchatkan Peninsula.

Severe forest fires occur in western North America where conditions are drier. Fires are a natural and important disturbance in many temperate forests, but natural fire regime can be changed by poor forestry management, invasive species, encroachment, and by humans. In North America, fire suppression in some areas, has contributed to the decline of grizzly bear (*Ursus arctos horribilis*) numbers (Contreras *et al.,* 1986). Fires promote and maintain many important berry-producing shrubs and forbs, which are important food source for bears, as well as providing habitat for insects and, in some cases, carrion. Some of the largest fires in the world occur in boreal forests. Fire return times in natural forests vary greatly, from 40 years in some Jack pine (*Pinus banksiana*) ecosystems in central Canada, to 300 years, depending on climate (van Wagner, 1978). Most boreal conifers and broad-leaved deciduous trees suffer high mortality even at low fire intensities, owing to canopy architecture, low foliar moisture, and thin bark (Johnson, 1992). Generally, the ability of post-fire boreal forest to regenerate is high, but frequent high intensity fires can offset this balance. Weather and climate are determinants for behavior and severity of wildfires, along with fuel properties, topography (Pyne *et al.,* 1996), and the effects of climate variability which are apparent as summer temperatures increase and many regions experience long-term droughts. Under warm and dry conditions, a fire season becomes longer, and fires are easier to ignite and spread. In addition, the spread of annual invasive grasses has led to much larger, more frequent fires in dryland regions (e.g. Brooks & Minnich, 2006). La Niña favors slighty higher than normal temperatures in a broad area covering the southern Rockies and Great Plains, the Ohio valley, the southeast, and the mid-Atlantic States.

Mesoamerica

Mesoamerica also faces a variety of natural disasters, with 31% caused by floods, 26% by wind storms, 19% by earthquakes and 8% by volcanoes (Charveriat *et al.,* 2000). Rainfall-induced disasters rank first among all natural disasters in Central America. In Central America and the Caribbean, storms that develop along the intertropical convergence zone and the subtropical high-pressure zone, dominate the weather. In Mesoamerica, it is common for two or more countries to be struck by the same rainfall event. For example, Hurricane Mitch in 1998 affected the entire region, killing more than 18,000 people (Guinea Barrientos *et al.,* 2015). In tropical semi-deciduous forest on the Yucatan Peninsula, Mexico, species richness of bees declined after hurricane Hurricane Dean (2007), with a loss of 40% of the species present beforehand, however the native bee community returned to previous species diversity levels just two months after the hurricane, probably due to the rapid recovery of the vegetation (Ramírez *et al.,* 2016).

El Niño years are associated with intense droughts and an increase in wildfires. In Mexico, during El Niño of 1998 near to 849,632 hectares were affected for 14,445 fires (Delgadillo, 1999). While the El Niño of 2005 registered 9,709 fires in Mexico that affected 276,089 hectares (Villers & Hernández, 2007).

There is also a great deal of seismic activity in the region due to the presence of several active geologic faults[22] within the Central America Volcanic Arc[23]. Volcanic eruptions and earthquakes occur frequently that have resulted in the loss of lives and property and impacted natural ecosystems.

Caribbean

In the Caribbean, windstorms constitute more than half of disasters while flooding is the second most common disaster. Floods are a function of climate, hydrology, and soil characteristics and are usually associated with hurricanes and other tropical storms which generate heavy rainfall. Small Island Developing States of the Caribbean

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[22] https://en.wikipedia.org/wiki/Active_fault

[23] https://en.wikipedia.org/wiki/Central_America_Volcanic_Arc

are particularly vulnerable. The region experiences regular annual losses due to natural hazard events in the order of $3 billion (Collymore, 2011). In Haiti, a devastating earthquake struck the island in 2010, killing more than 300,000 people. The human impact of the earthquake was immense primarily because it occurred in a large urban area with many poorly-constructed buildings (Zephyr, 2011). Geology and climate contribute to the prevalence of landslides in the Caribbean. Weather patterns, deforestation in some places, and increasing population density are among the major causes of landslides in the region (Holcombe *et al.,* 2012). Droughts have also negatively affected the economic and social sectors of several Caribbean states and are often related to the El Niño Southern Oscillation. Some countries in the region, like Guyana in 1997 and Cuba between 2004-2006 and 2015-2017, experience severe droughts that direct influence biodiversity and ecosystem services. The Caribbean and eastern Central America are also prone to disturbance due to tsunamis, which have historically caused substantial loss of life and property in many countries of the region (Henson *et al.,* 2006).

Huge and very rare catastrophes affecting entire regions are likely to remain imprinted in the structure of local biological communities for millennia (Brooks & Smith, 2001). The increasing frequency and range of natural disasters which, when coupled with the intensified vulnerability in the Caribbean, demonstrates the need for sustained regional efforts to reduce vulnerability to climatic and environmental hazards there. Given that the Caribbean coastal zones are at the heart of the tourism industry in the region, the economy and well-being of many countries is immensely vulnerable to natural disasters.

South America

In South America, between 1904 and 2011, 966 natural disasters were recorded, 735 of which of hydrometeorological nature. The most common events were floods and earthquakes corresponding to more than 55% of the calamitous occurrences in South America, however droughts and floods affected the largest number of people in the period (Nunes, 2011). El Niño events have resulted in higher rainfall in Peru, Ecuador, Argentina, Paraguay and Southern Brazil. The hydrological system in the region also contributes to flooding risk. The major drainage divide is far to the west along the crest of the Andes. West from this divide, in the mountainous regions, slopes of the riverbeds are very steep, which, in the event of storms, increases risk of flash flooding, the most dangerous types of flooding.

Landslides are also common in the region due to the nature of soils and steep topography and usually occur in connection with earthquakes, volcanoes, wildfires, and floods. Andean soils are relatively young and are subject to great erosion by water and winds because of the steep gradients of much of the land. Along the Andean mountain chain, landslides produce serious damage with widespread environmental and economical effects for Andean countries (Lozano *et al.,* 2006). Landslides may have severe and long-lasting negative effects on natural and human-dominated ecosystems, but they may also influence ecosystems in positive ways. For example, landslides play a key role in the dynamics of mountainscapes and creating suitable habitat patches for some species (Restrepo *et al.,* 2009).

With a current total of 204, South America has more active volcanoes than any other region of the world. The volcanic eruption of Puyehue-Cordón Caulle volcanic complex in Chile in 2011 dispersed about 100 million tons of pyroclastic materials. Impacts included changes in the reproduction and the body condition of a population of a lizard population (Boretto *et al.,* 2014), increased mortality of honeybees (*Apis mellifera*) (Martínez *et al.* 2013), and reduced availability of forage by 90% to 100% (Siffredi *et al.,* 2011).

Seismic activity is significant along the South American portion of the Ring of Fire. Jaramillo *et al.* (2012) provided the first quantification of earthquake and tsunami effects on sandy beach ecosystems after Chile's 2010 Mw 8.8 earthquake which indicated that ecological responses of beach ecosystems were strongly affected by the magnitude of land-level change.

Seasonal drought occurs in climates that have well-defined annual rainy and dry seasons. However, there are important and severe drought and precipitation changes that are not seasonal and can last months to years. The arid (northeast Brazil, Mexico) and cold (south Chile) climate zones in the region have a higher propensity to drought episodes. Forest fires are associated with the dry season and drought conditions.

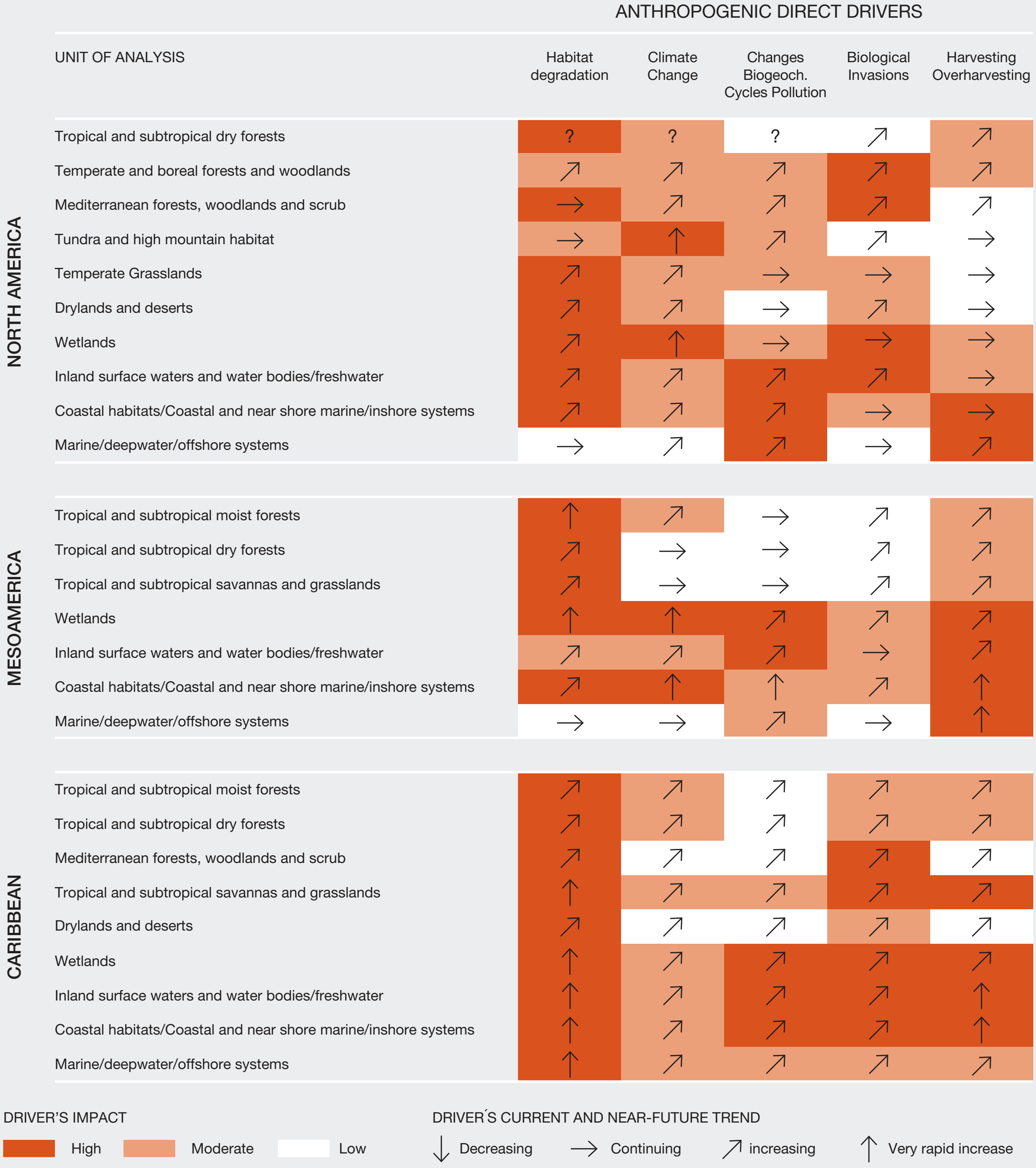
4.6 INTERACTIONS BETWEEN DIRECT DRIVERS

Although biodiversity may also change due to natural causes (section 4.5), anthropogenic drivers dominate current change in the Americas. As presented in **Figure 4.12** in all four subregions of the Americas, multiple drivers such as habitat loss and fragmentation, changes in biogeochemical cycles and pollution, climate change, overexploitation and invasive species increasingly threaten biodiversity, ecosystem services, and their benefits to society.

The analysis of status and trends of the different drivers indicates that habitat degradation has been the largest threat to freshwater, marine, and terrestrial biodiversity in the Americas. The net change in local diversity (for both species richness and total abundance) caused by land use and related pressures by 2005 is highlighted in **Figure 4.13** (Newbold *et al.,* 2015). All four subregions showed critical areas with significant loss of biodiversity in association to habitat degradation. As presented in section 4.4.1 and further discussed in the following section, indirect

Figure 4 12 **Relative importance of, and trends in, the impact of direct drivers on biodiversity and ecosystem services for the Americas (divided in North America, Mesoamerica, the Caribbean and South America).**

Cell color indicates the impact to date of each driver on extent and condition of the units of analysis.The arrows indicate the current and near-future trend in the impact of the driver on extent and condition of the units of analysis. Change in both impacts or trends can be positive or negative. This figure is based on information synthesized from the present chapter and expert opinion. This figure presents unit of analysis-wide impacts and trends, and so may be different from those in specific sub-habitats. Source: own representation



ANTHROPOGENIC DIRECT DRIVERS

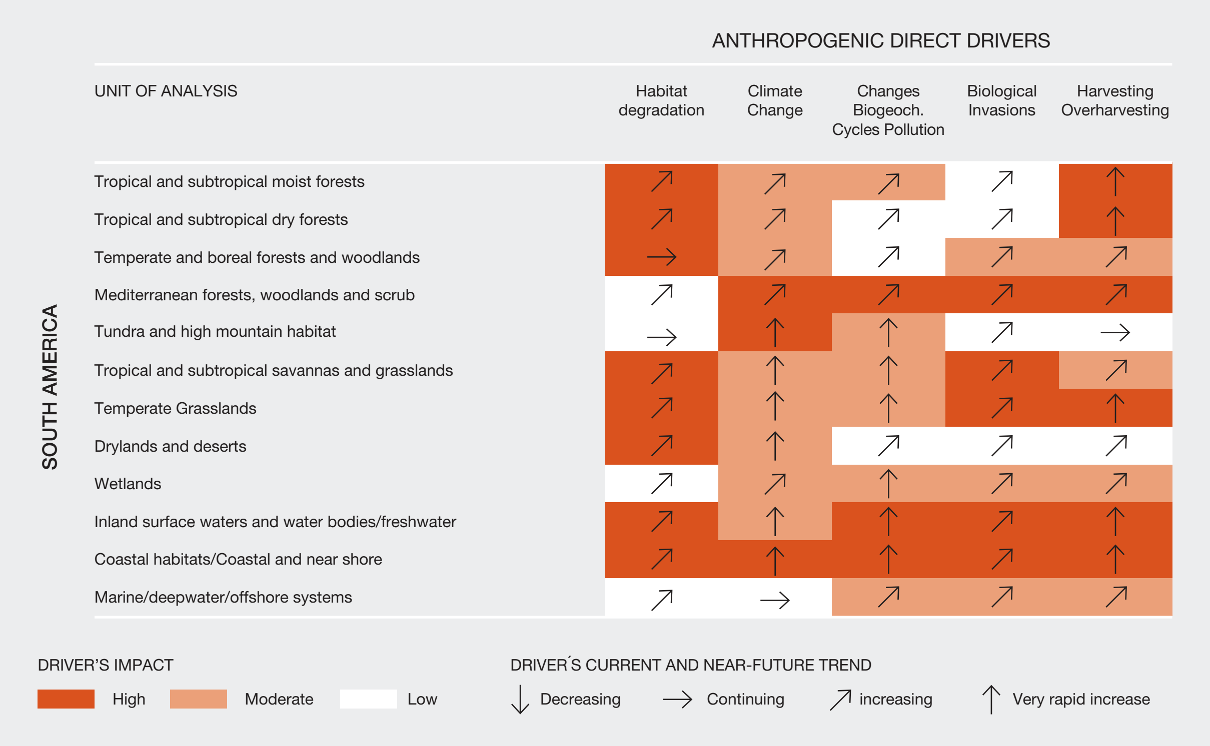
	UNIT OF ANALYSIS	Habitat degradation	Climate Change	Changes Biogeoch. Cycles Pollution	Biological Invasions	Harvesting Overharvesting
NORTH AMERICA	Tropical and subtropical dry forests	🟥 ?	🟧 ?	⬜ ?	⬜ ↗	🟧 ↗
	Temperate and boreal forests and woodlands	🟧 ↗	🟧 ↗	🟧 ↗	🟥 ↗	🟧 ↗
	Mediterranean forests, woodlands and scrub	🟥 →	🟧 ↗	🟧 ↗	🟥 ↗	⬜ ↗
	Tundra and high mountain habitat	🟧 →	🟥 ↑	🟧 ↗	⬜ ↗	⬜ →
	Temperate Grasslands	🟥 ↗	🟧 ↗	🟧 →	🟧 →	⬜ →
	Drylands and deserts	🟥 ↗	🟧 ↗	⬜ →	🟧 ↗	⬜ →
	Wetlands	🟥 ↗	🟥 ↑	🟧 →	🟥 →	🟧 →
	Inland surface waters and water bodies/freshwater	🟥 ↗	🟧 ↗	🟥 ↗	🟥 ↗	🟧 →
	Coastal habitats/Coastal and near shore marine/inshore systems	🟥 ↗	🟧 ↗	🟥 ↗	🟧 →	🟥 →
	Marine/deepwater/offshore systems	⬜ →	⬜ ↗	🟥 ↗	⬜ →	🟥 ↗
MESOAMERICA	Tropical and subtropical moist forests	🟥 ↑	🟧 ↗	⬜ →	⬜ ↗	🟧 ↗
	Tropical and subtropical dry forests	🟥 ↗	⬜ →	⬜ →	⬜ ↗	🟧 ↗
	Tropical and subtropical savannas and grasslands	🟥 ↗	⬜ →	⬜ →	⬜ ↗	🟧 ↗
	Wetlands	🟥 ↑	🟥 ↑	🟥 ↗	🟧 ↗	🟥 ↗
	Inland surface waters and water bodies/freshwater	🟧 ↗	🟧 ↗	🟥 ↗	🟧 →	🟥 ↗
	Coastal habitats/Coastal and near shore marine/inshore systems	🟥 ↗	🟥 ↑	🟧 ↑	🟧 ↗	🟥 ↑
	Marine/deepwater/offshore systems	⬜ →	⬜ →	🟧 ↗	⬜ →	🟥 ↑
CARIBBEAN	Tropical and subtropical moist forests	🟥 ↗	🟧 ↗	⬜ ↗	🟧 ↗	🟧 ↗
	Tropical and subtropical dry forests	🟥 ↗	🟧 ↗	⬜ ↗	🟧 ↗	🟧 ↗
	Mediterranean forests, woodlands and scrub	🟥 ↗	⬜ ↗	⬜ ↗	🟥 ↗	⬜ ↗
	Tropical and subtropical savannas and grasslands	🟥 ↑	🟧 ↗	🟧 ↗	🟥 ↗	🟥 ↗
	Drylands and deserts	🟥 ↗	⬜ ↗	⬜ ↗	🟧 ↗	⬜ ↗
	Wetlands	🟥 ↑	🟧 ↗	🟥 ↗	🟥 ↗	🟥 ↗
	Inland surface waters and water bodies/freshwater	🟥 ↑	🟧 ↗	🟥 ↗	🟥 ↗	🟥 ↑
	Coastal habitats/Coastal and near shore marine/inshore systems	🟥 ↑	🟧 ↗	🟥 ↗	🟥 ↗	🟥 ↑
	Marine/deepwater/offshore systems	🟥 ↑	🟧 ↗	🟧 ↗	🟧 ↗	🟧 ↗

DRIVER'S IMPACT

🟥 High 🟧 Moderate ⬜ Low

DRIVER´S CURRENT AND NEAR-FUTURE TREND

↓ Decreasing → Continuing ↗ increasing ↑ Very rapid increase



SOUTH AMERICA

	ANTHROPOGENIC DIRECT DRIVERS				
UNIT OF ANALYSIS	Habitat degradation	Climate Change	Changes Biogeoch. Cycles Pollution	Biological Invasions	Harvesting Overharvesting
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 habitat	⬜ →	🟥 ↑	🟧 ↑	⬜ ↗	⬜ →
Tropical and subtropical savannas and grasslands	🟥 ↗	🟧 ↑	🟧 ↑	🟥 ↗	🟧 ↗
Temperate Grasslands	🟥 ↗	🟧 ↑	🟧 ↑	🟥 ↗	🟥 ↑
Drylands and deserts	🟥 ↗	🟧 ↑	⬜ ↗	⬜ ↗	⬜ ↗
Wetlands	⬜ ↗	🟧 ↗	🟧 ↑	🟧 ↗	🟧 ↗
Inland surface waters and water bodies/freshwater	🟥 ↗	🟧 ↑	🟥 ↑	🟥 ↗	🟥 ↑
Coastal habitats/Coastal and near shore	🟥 ↗	🟥 ↑	🟥 ↑	🟥 ↗	🟥 ↑
Marine/deepwater/offshore systems	⬜ ↗	⬜ →	🟧 ↗	🟧 ↗	🟧 ↗

DRIVER'S IMPACT

🟥 High 🟧 Moderate ⬜ Low

DRIVER´S CURRENT AND NEAR-FUTURE TREND

↓ Decreasing → Continuing ↗ increasing ↑ Very rapid increase

drivers such as agriculture expansion, energy demand, and urbanization are linked to extensive changes in natural landscapes.

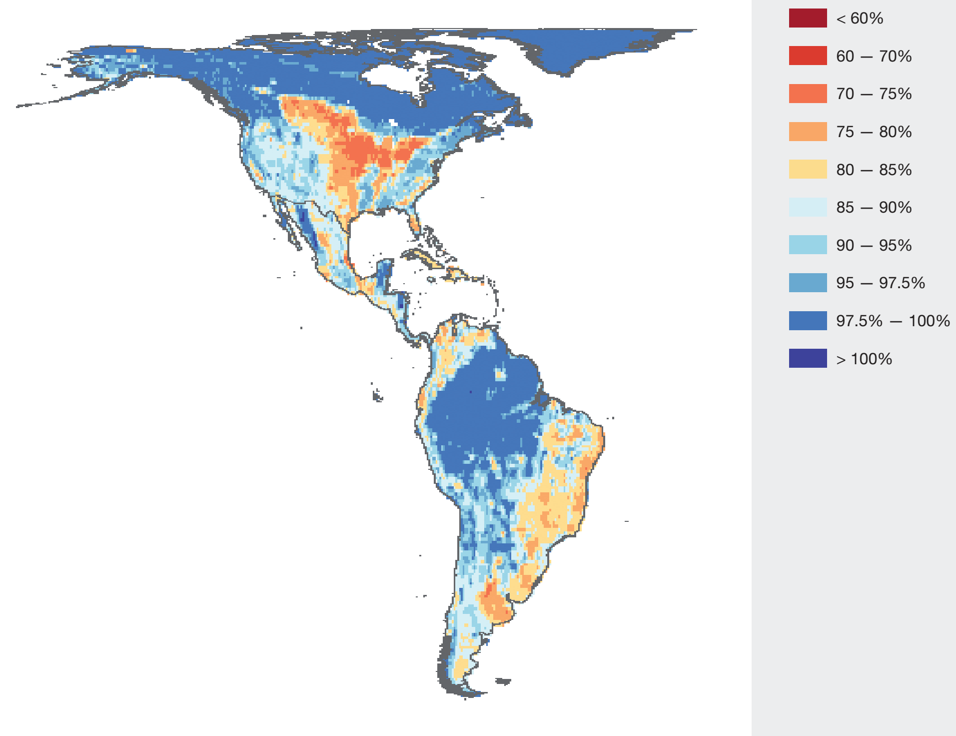
Over time, however, it is expected that the relative importance of direct drivers will change and the effects of climate change are expected to significantly increase (Alkemade *et al.,* 2009). The importance of the drivers of biodiversity change differs across realms, with land-use change being a dominant driver in terrestrial systems, and overexploitation in marine systems, while climate change is ubiquitous across all realms (Pereira *et al.,* 2010). A meta-analysis of 1,319 studies that quantified the effects of habitat loss on biological populations (different taxa, landscapes, land-uses, geographic locations and climate) pointed out the magnitude of these effects depends on current climatic conditions and historical rates of climate change (Mantyka-Pringle *et al.,* 2012). Current maximum temperature was the most important determinant of habitat loss and fragmentation effects with mean precipitation change over the last 100 years of secondary importance (Mantyka-Pringle *et al.,* 2012).

Climate change will have far-reaching impacts on biodiversity, including increasing extinction rates. Besides exposure to climate change, there are biological differences between species that may significantly increase or reduce their vulnerability. Species that are both highly vulnerable and threatened by climate change, and the regions in which they are concentrated, deserve particular conservation attention to reduce both threats and climate change adaptation interventions (Foden *et al.,* 2013). For example, the Amazon and Mesoamerica emerge as regions of high climate change vulnerability for both birds and amphibians, due to the large overall numbers and proportions of these groups that exist there (Foden *et al.,* 2013).

Future impacts of climate change are also related to different mitigation strategies, especially those related to land-based carbon sequestration **Figure 4.4** shows historical and future estimates of net change in local diversity from 1500-2095, based on estimates of land-use intensity and human population density from the four IPCC RCP scenarios, which correspond to different intensities of global climate change (Newbold *et al.,* 2015). Studies that addressed the interactions between land use and climate change (e.g. Oliver & Morecroft, 2014; Jantz *et al.,* 2015) indicate the loss of natural vegetation cover generally decrease as mitigation efforts increase (RCP scenarios). The worst biodiversity outcomes arise from the scenario with the most dramatic climate change (MESSAGE 8.5) **Figure 4.14** in which rapid human population growth drives widespread agricultural expansion, even though the projections omit direct climate effects on local assemblages. Recent trends in greenhouse gasses emissions most closely match this scenario (Newbold *et al.,* 2015).

In addition, concurrent effects of climate and land use changes can further increase the already dramatic rates of biological invasions. Projections using multiple species distribution models, several global climate models, and

Figure 4 13 **Species richness relative to an uninhabited baseline, for the year 2000. Source: based on Leadley *et al.* (2014).**



land cover change scenarios, evaluated the vulnerability of biomes to 100 of the world's worst invasive species and highlighted the need to consider both climate and land use change when focusing on biological invasions (Bellard *et al.,* 2013). Analysis of the future vulnerability of various biome types to these invasive alien species indicated northeastern North America as one of the three global future hotspots of invasion. Southern Brazil could be affected at a lower rate (20–40 invasive alien species) (Bellard *et al.,* 2013).

The recognition of the interactions between direct drivers and conservation efforts implies that not only strategies focusing on a single driver might be inadequate, but also there are opportunities to align biodiversity conservation and mitigation. The cumulative and synergistic effects of drivers reinforces the need of effective adaptation strategies and policies to better safeguard protected areas under multiple drivers of change, especially since land use changes, invasives, and climate are expected to impact ecosystem function and biodiversity significantly (Hansen *et al.,* 2014). Future trends and scenarios are developed in Chapter 5 and governance and policy options in Chapter 6.

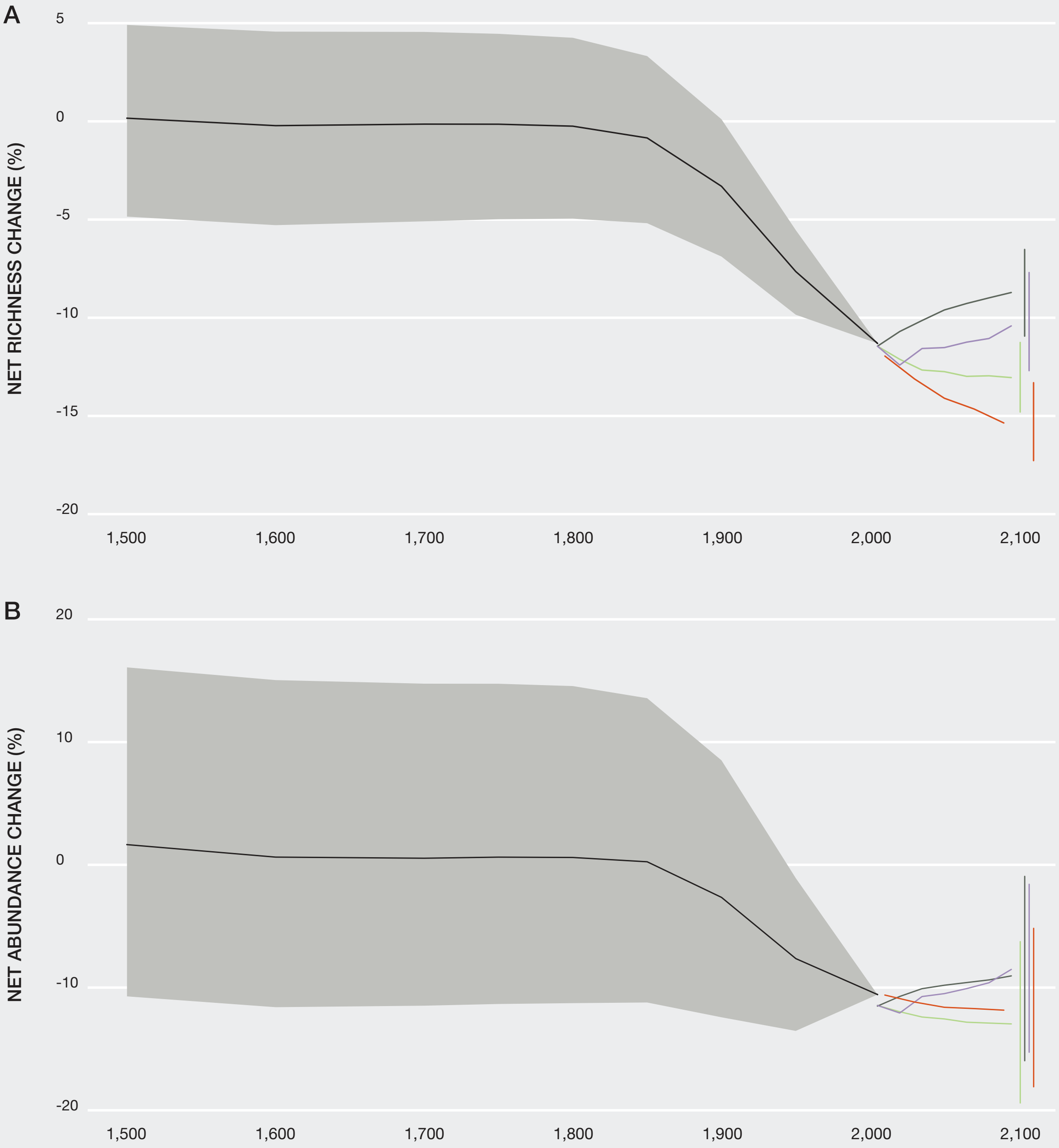
4.7 EFFECTS OF INDIRECT DRIVERS ON DIRECT DRIVERS

Changes in the behaviour and values of individuals, institutions and organizations are a prerequisite for sustainable development which is a means to reduce environmental degradation and improve the quality of life within generations as well as between generations. Therefore, the identification of drivers of change, especially indirect drivers, would contribute to discerning the characteristics that need to be targeted in order to achieve sustainable development.

In the Americas, the usage and exploitation of available natural resources are expected to intensify. The indirect drivers behind this are demographic, economic, socio-political, cultural, scientific, and technological advances among others (section 4.3.). The understanding of causal dependencies between human activities and their various impacts on ecosystems is a major challenge for science and requires integration of knowledge across different ecosystem components, linking physical, chemical and biological aspects with existing and emerging anthropogenic stressors. Likewise, an effective response to these interacting

Figure ④ ⑭ **Historical and future estimates of net change in local diversity from 1500–2095, based on estimates of land-use, land-use intensity and human population density from the four RCP scenarios.**

Net changes in richness Ⓐ, total abundance Ⓑ are shown. Historical (shading) and future (error bars) uncertainty shown as 95% confidence intervals. Source: based on Newbold *et al.* (2015).



threats involves a better understanding of governance systems (section 4.3) and ecological processes that affect the resilience of the biota and ecosystems including the identification of early warnings of change, tipping points and the characteristics of species, communities and ecosystems that underpin ecological resilience.

The cumulative effects of multiple stressors may not be additive but may be magnified by their interactions (synergy) and can lead to critical thresholds and transitions of ecological systems (Cotê *et al.,* 2016). Synergistic interactions are caused by amplifying feedbacks and can provoke unpredictable "ecological surprises" that can accelerate biodiversity loss and impair the functioning of ecosystems. The conservation implications of synergies are that cascading impacts of co-occurring stressors will degrade ecosystems faster and more severely. For example, the unforeseen crash of the Peruvian anchovy populations is proposed to have resulted from the interaction between El Niño driven warming and reduced productivity, in combination with overfishing (Jackson *et al.,* 2001).

The Americas, and in particular South American, has a major role in the global trade of products where cultivation involves deforestation and vegetation clearing in the producing countries. These products are referred to as forest and biodiversity risk commodities (Henders *et al.,* 2015), such as beef, soybeans, maize, cotton, cocoa, coffee and timber products. There is a large potential to increase South America's role in the trade of a number of others like palm oil and biofuels. The Americas account for the vast majority of global soy exports, for about two thirds of global maize exports and for about one third of bovine meat exports (**Table 4.14** and Chapter 2, section 2.2.1). This reliance on land-based export commodities, paired with the relative abundance of arable land currently sustaining natural vegetation, clearly poses a threat to the preservation of the remaining natural areas. It has been hypothesized that in order to increase food security globally more trade liberalization is crucial, but that it would also lead to more environmental pressures in some regions across Latin America (Flachsbarth *et al.,* 2015). The global trade network has increased enormously since 1950s in terms of the total value of exchanged goods. The technological development of means of transportations (e.g. large-scale transport of goods by airplanes, transcontinental containerships) has decreased the time necessary for transport, greatly expanded the type and value of goods transported. Increases in trading activity will cause substantial increases in invasion levels within a few decades, particularly in emerging economies (Seebens *et al.,* 2013). These countries show most pronounced growth of naturalized plant numbers compared to countries with similar trade value increases (Seebens *et al.,* 2013) and most of these economies coincide with regions of megadiversity (Brooks *et al.,* 2006), rich in endemic and rare species.

The Americas experienced an early and intense urbanization process. While urbanization rates will be highest in China and India, it is in Central and South America where the largest number of species will be affected (Seto *et al.,* 2012). Urban land-cover change threatens biodiversity and affects ecosystem productivity through loss of habitat, biomass, and carbon storage. Even relatively small decreases in habitat can cause extinction rates to rise disproportionately in already diminished and severely fragmented habitats, like the Atlantic forest hotspots in South America (Seto *et al.,* 2012). Coastal regions and islands are particularly under pressure to increase their urban footprint. The projected urban expansion in the Caribbean islands is relatively small in total area, but they are home to a significant proportion of endemic plants and invertebrates (Chapter 3).

Energy production and agriculture are related to pollution and changes in biogeochemical cycles of major nutrients (nitrogen, carbon, phosphorus, sulfur). Atmospheric ozone occurs where emissions from fossil fuel combustion (energy utilities, industry, motor vehicle exhaust) or biomass burning interact with vapors from solvents, gasoline or vegetation. Emissions from motor vehicles and other fossil fuel combustion are also large contributors to atmospheric fine particulate matter a human health hazard. The geographic distribution of atmospheric nitrogen deposition is related to fossil fuel combustion for utilities, industry and transportation. The levels of nutrients in rivers are expected to increase in the Americas, particularly as per capita GDP, food crop, meat and milk production increase. Widespread trends in pesticide concentrations, some downward and some upward, occur in response to shifts in use patterns primarily driven by regulatory changes and introductions of new pesticides or crops, but the use of pesticides is projected to increase. Urban systems, via runoff and treated and untreated sewage, add more nutrients, sediment and organic matter to aquatic systems.

Even places with low human density are subjected to pollution from human activities. Pollution from past mining and smelting exposes wildlife to toxic metal contamination across the Americas. Lead contamination has also reached the Arctic from coal combustion and Amazonian countries are among the largest sources of mercury emissions from artisanal gold mining in the Americas. Major sources of atmospheric mercury also include fossil fuel, non-ferrous metal manufacturing, cement production, waste disposal and caustic soda production and emissions from soils, sediment, water, and biomass burning, which include re-emissions from sites that have legacy contamination issues. Toxic releases from these sites may continue due to weak environmental laws or enforcement, poor public understanding of the continuing environmental effects of these sites, and a lack of public or private funds.

The interactions between drivers presented in this chapter can be further examined using freshwater and wetland ecosystems throughout the Americas as case studies. These units of analysis appear particularly threatened in the qualitative approach presented in **Figure 4.12** and their analyses can provide a means for understanding the interactions of multiple drivers with greater clarity.

Freshwater and wetland ecosystems as examples of interactions

Freshwater is an essential resource for human life and for many natural systems that support human well-being. Human alteration of rivers, lakes and wetlands has followed economic development (Revenga *et al.,* 2005). Most freshwaters have been altered in multiple ways, and changes in any particular freshwater system usually have multiple causes. Water management is also a vast subject embracing such diverse topics as water markets, political conflict over water, connections between water and social development (Carpenter *et al.,* 2011).

A global assessment of patterns of freshwater species diversity, threat and endemism (Collen *et al.,* 2014), indicated that three processes predominantly threatened freshwater species: habitat loss/degradation, water pollution and over-exploitation. Of these, habitat loss/degradation was the most prevalent, affecting more than 80% of threatened species. The main indirect drivers of habitat loss and degradation were conversion to agriculture, logging, urbanization, and infrastructure development (particularly the building of dams). Dams disrupt the ecological connectivity of rivers, whereas water storage in reservoirs and release patterns affect quantity, quality, and timing of downstream flows. Consequences are influenced by interactions between different threat processes (for example, water pollution can be caused by chemical run-off from intensive agriculture or manufacturing, sedimentation by logged riparian habitat, and domestic waste water by urban expansion). On the top of these drivers climate change affects will cause impacts on freshwater and wetland ecosystems due to sea level rise, changes in precipitation, air temperature, and river discharges.

The Americas are particularly rich in terms of freshwater resources. In South America, about 30% of the planet's freshwaters flow through the Amazon, the Parana-Río de la Plata and the Orinoco watershed. In North America, the Great Lakes shared by the USA and Canada span more than 1,200 kilometers from west to east and represent 84% of North America's surface freshwater and about 21% of the world's supply of surface freshwater. The Americas have also significant areas of wetlands. In South America, the exact size of the wetland area is not known but may comprise as much as 20% of the sub-continent, with river floodplains and intermittent interfluvial wetlands as the most prominent types (Junk, 2013). North and Central America has a combined total of 2.5 million km^2 of wetlands, with 51% in Canada, 46% in the USA, and the remainder in subtropical and tropical Mexico and Central America (Mitsch & Hernandez, 2013). Along the Caribbean coast and in addition to coral reefs, saltwater wetlands such as mangroves and seagrass beds are the dominant ecosystems.

Because streams, rivers, and groundwater integrate the landscape, providing a conduit for the transfer of energy and material from terrestrial habitats into freshwater systems and ultimately to the oceans, they are particularly vulnerable to environmental impacts from land use change. Wetlands are also not isolated, but are connected to their surroundings as they are often located at the transition zone between upland and open water, wetlands can be affected by activities and conditions in both terrestrial and aquatic areas. Land use influences sediment, hydrologic, and nutrient regimes, which in turn influence aquatic biota and ecological processes in freshwaters. Land use change occurs largely through human actions affected by economic incentives and regulation. These changes can have both direct and indirect effects on freshwater ecosystems - the former have immediate ecological impacts (e.g. destruction of wildlife habitats), while the latter have impacts that are transmitted via altered flow or sediment transport patterns (e.g. lower productivity due to increasing turbidity) (Palmer *et al.,* 2002). Conversely, on many major rivers the need for hydroelectric power, flood control, and water for irrigation has led to the building of large dams that reduced the amount of sediment carried by those rivers.

North America – The Mississippi Basin

The Mississippi River watershed is the fourth largest in the world and the largest in North America at 3.2 million km^2 and includes all or parts of 31 USA states and two Canadian Provinces. Communities up and down the river use the Mississippi to obtain freshwater and to discharge their industrial and municipal waste. The Missouri River, one of the major tributaries of the basin, has had a long history of anthropogenic modification with considerable impacts on river and riparian ecology, form, and function (Skalak *et al.,* 2013). During the 20th century, several large dam-building efforts in the basin served the needs for irrigation, flood control, navigation, and the generation of hydroelectric power. Agriculture has been the dominant land use for nearly 200 years in the Mississippi basin, and has altered the hydrologic cycle and energy budget of the region. The basin produces 92% of the USA agricultural exports, 78% of the world's exports in feed grains and soybeans, and most of the livestock and hogs produced nationally. Sixty percent of all grain exported from the USA is shipped on the Mississippi River through the Port of New Orleans and other ports in southern Louisiana.

Changes in the watershed and management practices impact the wetlands of Mississippi Delta and the Gulf of Mexico. As the Mississippi River reaches the last phase of its journey to the Gulf of Mexico in southeastern Louisiana, it enters one of the most wetland-rich regions of the world. The total amount of freshwater and saltwater wetlands has been decreasing at a rapid rate in coastal Louisiana, amounting to a total wetland loss of between 66 and 90 km^2 per year and has been attributed to both natural and artificial causes (Dunbar *et al.,* 1992). The Mississippi River Basin accounts for 90% of the freshwater inflow to the Gulf of Mexico (Rabalais *et al.,* 1996). Nitrate–nitrogen concentrations and fluxes from the Mississippi River Basin increased dramatically in the 20th century, particularly in the decades after 1950, when nitrogen fertilizer came into increasing use. Artificial drainage and other hydrologic changes to the landscape, atmospheric deposition of nitrates, runoff and domestic wastewater discharges from cities and suburbs, and point discharges from feedlots and other sites of intensive agricultural activity are also contributing factors to the input of nutrients into the Gulf.

South America – Río de la Plata Basin

The La Plata River Basin is one of the most important river basins of the world. Draining approximately one-fifth of the South American continent, extending over some 3.1 million km^2, and conveys water from central portions of the continent to the south-western Atlantic Ocean. The La Plata River system is recognized as among those watersheds of the world having the highest numbers of endemic fishes and birds but also the highest numbers of major dams. The La Plata Basin represents an important concentration of economic development in southern and central South America (Tucci & Clarke, 1998). Thirty-one large dams and fifty-seven large cities, each with populations in excess of 100,000 including the capital cities of Argentina, Brazil, Paraguay, and Uruguay, are to be found within this Basin. The rivers of the La Plata River Basin are subject to pressures that have modified, and can further modify the quantity and quality of their waters (Cuya *et al.,* 2013). The consequences of these pressures are not restricted to specific countries, but are of a transboundary character. Before 1960, the Plata River Basin was almost undeveloped. The regulation of the Paraná (a large tributary of the La Plata in Brazil) for hydroelectricity has been increasing since the early 1970s. Water in reservoirs of the upper Paraná Basin currently comprises more than 70% of the mean annual discharge at its confluence with the Paraguay River. The expansion of hydroelectric generation in the upper basin brought with it an increase in industry, agriculture, transport and settlements. These in turn increased deforestation, soil erosion, degraded water quality and reduced fisheries opportunities in both the upper and lower basins (FAO, 2016). These pressures are expected to increase in the future as the Basin countries continue to enlarge their agricultural and industrial bases, and provision of services, to improve the living standards of their increasing populations (Cuya *et al.,* 2013). The basin has the second greatest number of planned dams in the world: 27 large dams, of which 6 are under construction. The national governments of the basin are planning a massive navigation and hydroelectric dam project (*Hidrovia*) to facilitate expansion of the export of soybean, timber, iron ore and other commodities during the dry season.

Central America and the Caribbean

Tropical rivers of Central America are highly heterogeneous systems, ranging from fast-flowing mountain torrents in areas of high relief to slow-moving rivers that meander through lowland environments. Relative to rivers in neighboring North and South America, the narrowness of the isthmus means that Central American rivers are shorter in length, carry a substantially lower volume of water as they drain smaller basins, and generally are closely connected to marine environments. Central American rivers contain hundreds of species of fishes and shrimp, including many migratory species that depend on a natural flow regime and upstream-downstream connectivity for survival. Human populations derive most water for consumptive uses from surface waters. Rivers provide a source of food, income, and building materials, serve as transportation routes, and have strong linkages to the cultural identity of rural people. Regionally, hydropower accounts for approximately 50% of net electricity generation and 42% of total installed generation capacity (Anderson, 2013). Central America has experienced a proliferation of hydropower dams in recent years, a trend that began with the construction of a few large dams in the 1980s (e.g. Arenal dam in Costa Rica, El Cajón in Honduras, and Chixoy in Guatemala), that accelerated with the privatization of electricity generation in the 1990s, and that has continued into the 21st century.

Population growth, an increase in rural electrification, and rising electricity consumption (estimated at 4.2% regionally in 2011) and reduced availability of domestic fossil fuel sources are important drivers of hydropower development in Central America. Expansion plans for the period 2012–2027 include many new hydropower developments in Central America, including large dams as well as small and medium-sized dams. Although a critical source of electricity, existing dams in Central America have been linked to declines in migratory and sensitive fish species, compromising other ecosystem services, and having negative impacts on population health and well-being. In the Caribbean, erosion, sedimentation, pollution, water nutrient enrichment, saltwater intrusion, and loss of biodiversity have been identified as the most significant factors affecting wetlands. The causes of these impacts include deforestation, tourism, urban development, industry, agriculture, damming and diversion of rivers, and dredging for navigation. In addition, natural and human enhanced phenomena such as tropical storms and hurricanes, sea level rise, and global warming also threaten these valuable ecosystems.

The challenge of matching scales: drivers, ecological and social responses

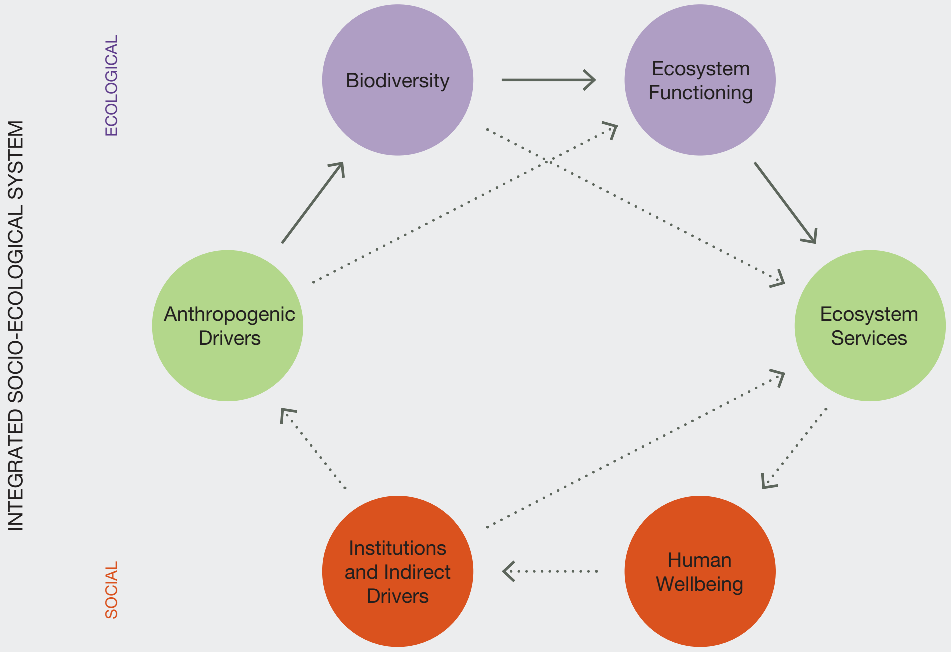
Systematic conservation planning must also ensure that not only biodiversity but also the supporting ecological processes are protected at a relevant and appropriate scale (Possingham & Wilson, 2005). Drivers interact across spatial, temporal, and organizational scales. Studies indicate that different drivers of biodiversity-ecosystem function relationships occur at small plot scales (species identities, composition) and large landscape scales (biomass, species richness) as well as in short and long temporal scales. These results imply that not all relationships and findings obtained by studies at small spatial and short temporal scales can necessarily be translated to larger or longer scales that have relevance for political decisions and conservation biology (Brose & Hillebrand, 2016). Global trends (e.g. climate change or globalization) can influence regional contexts and local ecosystem management while changes in national regulations might influence responses of different stakeholders to global change (Nelson *et al.,* 2006). Changes in ecosystem services also feed back to the drivers of change (e.g. altered ecosystems create new opportunities and constraints on land use) (Nelson *et al.,* 2006).

Some effects of drivers emerge in the short-term (e.g. land use, deforestation), while others mainly in the long-term (e.g. climate change, changes in biogeochemical cycles). Long-term impacts of anthropogenic drivers of environmental change on ecosystem functioning can strongly depend on how such drivers gradually decrease biodiversity and restructure communities (Isbell *et al.,* 2013). Current models do not account for potentially important indirect effects of habitat destruction on ecosystem services resulting from changes in biodiversity that occur within nearby remaining ecosystem fragments, even though many species could be lost from such fragments (Isbell *et al.,* 2015).

Socio-ecological systems are characterized by causal relationships between their different components (Fischer & Christopher, 2007) **Figure 4.15a** and environmental problems can originate from the relationships between stakeholders, from the inefficiency of institutional arrangements in implementing regulation, from social inequality or from the inadequacy of policy actions for a given social context (Maxim *et al.,* 2009). In addition, uncertainty is intrinsic to complex biological and social systems (Maxim *et al.,* 2009). In the case of the Americas, reducing uncertainties through the improvement of integrated monitoring networks will enhance the ability to respond to environmental changes in the different subregions and improve the understanding of potential interactions of multiple drivers and scales and how the interactive effects of change drivers might impact (positively or negatively) ecosystem in the future.

Figure 4 15 A **The influence and dependence of people on biodiversity.**

People influence biodiversity directly by changing land-use, climate and biogeochemical cycles, as well as by introducing species — actions known collectively as anthropogenic drivers. At the global scale, these activities are driving the sixth mass extinction in the history of life on Earth. At the local scale, species losses decrease ecosystem functioning (for example, ecosystem productivity and resource uptake) and stability (the invariability of ecosystem productivity across a period of years). At the intermediate scales such as landscapes or regions, changes in ecosystem functioning can alter the supply of ecosystem services, including the production of wood in forests, livestock forage in grasslands and fish in aquatic ecosystems. It is important to build multiscale knowledge at the intersections of the numerous components of the system. Various system components are positioned in a gradient that spans the social (orange) to ecological (purple) ends of a socio–ecological continuum. Dashed arrows indicate other important relationships that are beyond the scope of this review. Source: Isbell *et al.* (2017).



The **Figure 4.15b** represents the mismatches in the spatial and temporal scales at which the relationships between anthropogenic drivers, biodiversity, and ecosystem functions and services (Isbell *et al.,* 2017). These mismatches pose a challenge to link the cascading effects of human activities on biodiversity, ecosystems and ecosystem services. Furthermore, the scales at which knowledge is available for some of the relationships do not yet align with the scales at which policies and other decisions are often made.

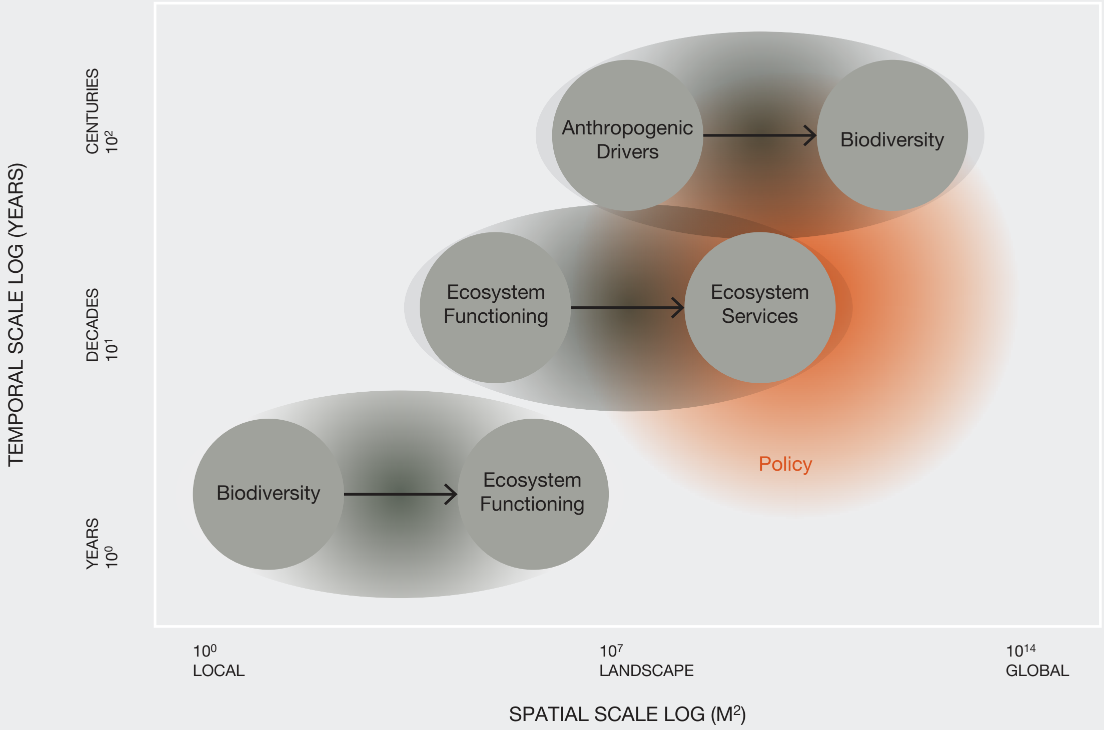
The Aichi 2020 targets, under the CBD, endeavor to halt the loss of biodiversity by 2020, in order to ensure that ecosystems continue to provide essential services. The present evaluation of the status and trends of the multiple drivers of change for the different units of analysis in the Americas shows that most of the Aichi targets will be not achieved without significant policy interventions. This analysis is in accordance with a study at the global scale of the many impediments for the accomplishment of the Aichi targets that indicated 15 of the Aichi targets as unlikely to be delivered; three likely to be delivered in part; and two in full (Hill *et al.,* 2015).

Understanding and managing ecosystem-service delivery is of key importance for human wellbeing (Chapter 2). Development, poverty eradication, and biodiversity conservation are key areas of focus of the United Nations SDG. The initiative adopted in 2015 by more then 150 world leaders set targets to be achieved by 2030 as part of a new sustainable development agenda and reinforces the demand for integrated analyses of indirect and direct drivers of biodiversity and ecosystem changes. This agenda is particularly relevant to Mesoamerica and South America whose countries still show social inequality allied to economies highly dependent on the export of natural resources and agricultural commodities.

The rapidly increasing dependency on biodiversity-risk commodities, which are expanding mostly at the expense of existing natural vegetation, is currently not

Figure ④ ⑮ Ⓑ **The influence and dependence of people on biodiversity.**

People influence biodiversity directly by changing land-use, climate and biogeochemical cycles, as well as by introducing species — actions known collectively as anthropogenic drivers. At the global scale, these activities are driving the sixth mass extinction in the history of life on Earth. At the local scale, species losses decrease ecosystem functioning (for example, ecosystem productivity and resource uptake) and stability (the invariability of ecosystem productivity across a period of years). At the intermediate scales such as landscapes or regions, changes in ecosystem functioning can alter the supply of ecosystem services, including the production of wood in forests, livestock forage in grasslands and fish in aquatic ecosystems. It is important to build multiscale knowledge at the intersections of the numerous components of the system. Various system components are positioned in a gradient that spans the social (orange) to ecological (purple) ends of a socio–ecological continuum. Dashed arrows indicate other important relationships that are beyond the scope of this review. Source: Isbell *et al.* (2017).



accompanied by comprehensive governance policies and land planning (Lemos & Agrawal, 2006). Efforts to revise this situation face a variety of challenges. The increased globalization of the world economy has catalyzed rapid growth and the complexity of international trade, leading to a disconnection and physical separation of the places of production, transformation and consumption of land-based products. This disconnectedness strongly hampers socio-environmental governance and the implementation of regulatory frameworks, beyond the intrinsic difficulties to govern sectors already in rapid transition driven by increasing global demand for food, fuel, feed and fiber **Figure 4.16**. As a result, natural resource use policies often come in place only when fundamental shifts in the land-use system are already underway and interventions become costly and have limited influence. Furthermore, while benefits from trade of agricultural commodities are easily measured and perceived by those in the supply chain and production countries as a whole, the associated externalities have so far been poorly understood and/or poorly translated into economic costs in future years.

The application of the knowledge of ecological and socio-ecological processes to the sustainable management of natural systems is the foundation to build resilience to future environmental change. In the different units of analysis, increasing and diverse exploitation of natural resources demands the development of different regional and national legislative initiatives aimed at protection and restoration of biodiversity and ecosystems and further adequate and sustainable management of nature (see Chapter 6). Policies and strategies could reduce the anthropogenic impacts on biodiversity by modifying the trends of drivers and underlying causes. The integration of biodiversity protection into other sectoral policies might enhance the chances for effective political action. Planning of measures to prevent and mitigate biodiversity loss, like habitat preservation, restoring degraded landscapes, maintaining or creating connectivity, avoiding overharvest, reducing fire risk and control of greenhouse gasses emissions, should consider the need to manage multiple drivers simultaneously over longer terms (Brook *et al.,* 2008). Usually, conservation plans are developed for regions that encompass only one environmental realm (terrestrial, freshwater or marine) because of logistical, institutional and political constraints (Beger *et al.,* 2010). However, as shown above for freshwater and wetland ecosystems, these realms often interact through processes that form, utilize and maintain interfaces or connections, which are essential for the persistence of some species and ecosystem functions. These linkages must be also considered in policy framing processes as well as the analysis of values and human behavior that induce, are affected by or respnd to the changes in environmental conditions.

Figure ④ ⑯ **Direct and indirect drivers of NCP in the Americas and their interdependencies. Source: own representation.**

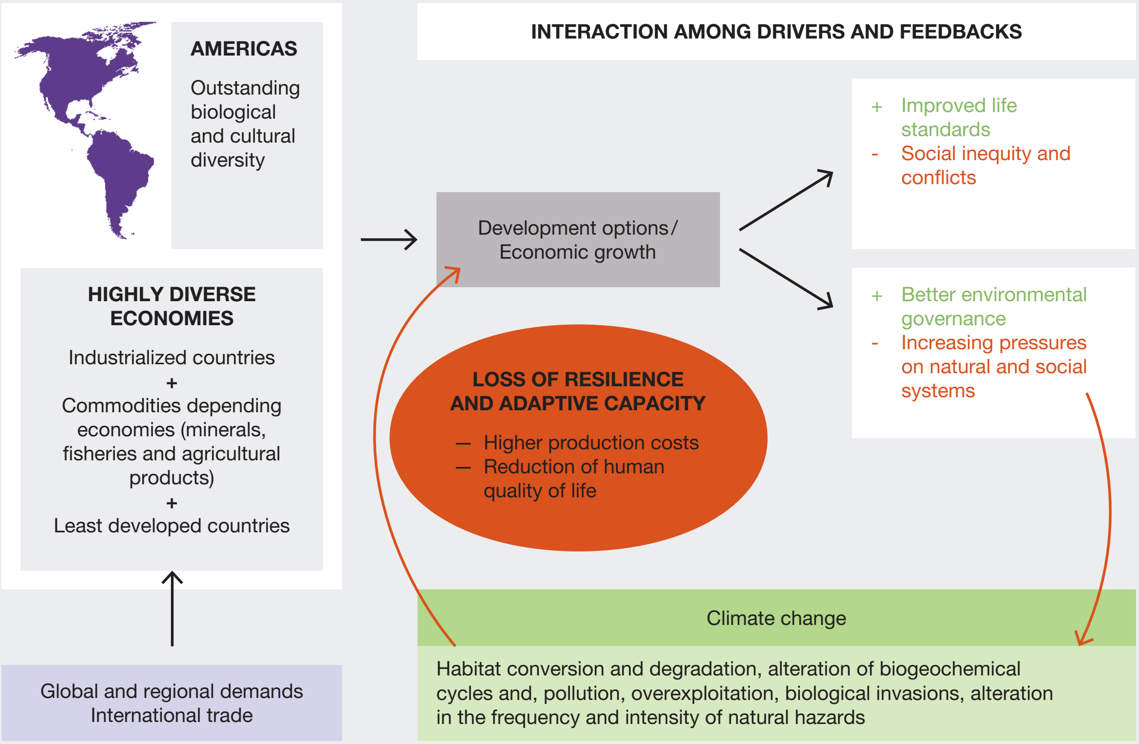


Table 4 14 Weight of the Americas in the global exports of key bodiversity-risk commodities (as percentage of global exports), 2015.

COUNTRY	Agricult. products*	Total merchandise trade*	Soy beans**	Soy oil**	Soy meal**	Meat of bovine animals; fresh or chilled**	Meat of bovine animals, frozen**
Argentina	2,87	0,43	8,87	44,03	39,66	1,52	1,41
Aruba	0,01	0,00	0,00	0,00	0,00	0,00	0,00
Belize	0,01	0,00	0,00	0,00	0,00	0,00	0,00
Bolivia (Plurinational State of)	0,14	0,06	0,01	3,04	2,31	0,02	0,02
Brazil	6,01	1,29	41,37	13,13	22,09	3,22	17,79
Canada	3,22	2,44	3,23	1,21	0,34	5,90	1,13
Chile	0,83	0,41	0,00	0,00	0,00	0,07	0,11
Colombia	0,46	0,31	0,00	0,00	0,00	0,05	0,15
Costa Rica	0,28	0,06	0,00	0,07	0,00	0,16	0,18
Dominican Rep.	0,12	0,05	0,00	0,05	0,00	0,00	0,01
Ecuador	0,35	0,13	0,00	0,00	0,00	0,00	0,00
El Salvador	0,08	0,03	0,00	0,01	0,00	0,00	0,00
Guatemala	0,34	0,05	0,00	0,03	0,00	0,01	0,02
Honduras	0,14	0,04	na	na	na	na	na
Jamaica	0,02	0,01	0,00	0,00	0,00	0,00	0,00
Mexico	1,67	2,02	0,00	0,02	0,02	3,78	0,45
Nicaragua	0,12	0,01	0,00	0,15	0,00	0,84	1,15
Panama	0,02	0,00	0,00	0,00	0,00	0,03	0,06
Paraguay	0,42	0,05	3,48	5,49	3,72	2,57	3,30
Peru	0,30	0,22	0,00	0,00	0,00	0,00	0,00
Uruguay	0,44	0,05	2,31	0,00	0,02	1,10	4,06
USA	10,57	8,39	36,73	7,53	13,92	8,56	7,57
Venezuela (Bolivarian Republic of)	0,00	0,47	na	na	na	na	na
TOTAL VS WORLD	**28,45**	**16,54**	**96,01**	**74,78**	**82,09**	**27,81**	**37,41**

*FAOSTAT (2013), % of USA Dollars value versus world, **COMTRADE (2015), % of weight versus world, ***COMTRADE (2015), % of USA Dollars value versus world, ****COMTRADE (2015), % of m^3 volume versus world

Maize**	Maize flour**	Cocoa beans**	Cocoa butter, fat and oil**	Cocoa paste**	Cotton***	Wood in the rough or roughly squared****	Wood sawn or chipped lengthwise****
11,48	0,18	0,00	0,00	0,00	0,02	0,00	0,10
0,00	0,00	0,00	0,00	0,00	0,00	0,00	0,00
0,00	0,02	0,01	0,00	0,00	0,00	0,00	0,01
0,07	0,00	0,01	0,01	0,00	0,00	0,01	0,04
19,84	17,78	0,27	3,66	1,15	0,54	0,13	1,53
0,41	0,10	0,05	0,20	0,51	0,01	5,06	0,00
0,01	0,00	0,00	0,00	0,00	0,01	0,07	3,22
0,00	1,71	0,55	0,36	0,29	0,01	0,05	0,00
0,00	0,24	0,01	0,01	0,00	0,01	0,07	0,22
0,00	2,12	3,17	0,17	0,02	0,07	0,01	0,00
0,00	0,00	9,41	0,70	1,47	0,01	0,18	0,07
0,00	6,76	0,00	0,00	0,00	0,02	0,01	0,00
0,01	1,53	0,00	0,00	0,00	0,02	0,01	0,05
na	na	na	na	na	na	na	na
0,00	0,00	0,01	0,00	0,00	0,00	0,00	0,00
0,51	13,20	0,01	0,65	0,01	0,10	0,07	0,03
0,00	0,00	0,16	0,00	0,00	0,00	0,00	0,02
0,00	0,00	0,02	0,01	0,00	0,00	0,14	0,05
2,25	0,00	0,00	0,00	0,00	0,00	0,01	0,03
0,01	0,02	2,36	0,94	0,31	0,03	0,00	0,10
0,00	0,00	0,00	0,00	0,00	0,00	6,29	0,22
30,63	16,71	0,62	3,35	3,08	2,19	16,37	0,00
na	na	na	na	na	na	na	na
65,23	**60,37**	**16,65**	**10,05**	**6,86**	**3,04**	**28,49**	**5,69**

4.8 GAPS IN KNOWLEDGE AND DATA

Relevant information on indirect drivers is extremely limited at environmental scales (e.g. habitats, ecosystems, biomes), which in many cases may be more relevant than institutional scales (e.g. administrative, municipalities, provinces, countries) for IPBES assessments. In addition, internationally comparable data on indirect drivers are not always available for all countries and regions of the Americas being particularly limited for small economies.

The mechanisms by which direct drivers interact are poorly understood. The mechanisms include interactions between demographic parameters, evolutionary trade-offs and synergies and threshold effects of population size and patch occupancy on population persistence. Understanding how multiple drivers of global change interact to impact biodiversity and ecosystem services requires a multiscale approach as drivers act at from global to local scales, and their interactions have emergent properties (i.e. change with the scale). The lack of appropriate research is partially due to limited data availability and analytical issues in addressing interaction effects.

In the case of the Americas, for some regions, there is still substantial uncertainty associated to spatial and temporal magnitude of the drivers (e.g. area and spatial distribution of the different land-use classes and infrastructure maps, measurements and model forecasts for climate and nitrogen deposition, distribution of invasive species). For example, studies that quantify the impacts of invasive species on biodiversity and ecosystems are still very scarse, especially outside North America. In addition, there is very little information on the effects of nitrogen deposition on tropical forests, woodlands, savannas and grasslands (Bobbink *et al.,* 2010). Likewise, in contrast to North America, no systematic surveys exist for pollutants, including agricultural chemicals, persistent organic pollutants and mercury, in South America, the Caribbean and Mesoamerica. Another major difficult to assess the effects of pesticides on biodiversity and ecosystem services is just knowing what pesticides are used, when and how much as well as having little information on the environmental occurrence of these same pesticides. Regarding climate change, the degree to which climate change in tundra and boreal ecosystems will promote fires and droughts is not well-documented considering that these disturbances have major consequences for species productivity and dynamics in this region (Abbott *et al.,* 2016; Pastick *et al.,* 2017).

For some ecosystems, lack of consistent information on drivers of change is observed in all subregions of the Americas. Trends in land condition, and drivers of those trends, remain unstudied or understudied in most dryland areas across the Americas. Coastal aquatic and pelagic ocean biodiversity also remains poorly characterized throughout the Americas. Understanding how sensitive areas change in relation to regional- to global-scale processes, a mechanism to communicate the needs of people making decisions about local resources to scientists, and pathways to deliver scientific knowledge to decision makers remain prioritary needs for the region. At this time it is not possible to make a generalized statement of impacts of global changes in physical ocean dynamics and atmospheric carbon dioxide concentrations on coastal ecology. Another major unknown is the fate of plastic pollution in coastal regions of the Americas, as the amount of plastic pollution on the ocean surface is much less than the amount that is released to oceans, yet we know that many plastics can take hundreds of years to degrade (Clark *et al.,* 2016).

A major limitation in the study and management of coastal zones around the world has been the lack of a capacity to collect, handle, and process repeated, frequent observations of aquatic and nearby wetland resources in an integrated manner to enable the detection of changes in the chemistry and in the diversity of wetland and aquatic organisms.

Regarding American mangroves, more data on consequences of nitrogen and phosphorus enrichment to nutrient cycling rates, fluxes and stocks, sediment microbial communities structure and functioning, and the resulting primary productivity in the different types of mangroves are needed, especially in underrepresented areas like South America (Reis *et al.,* 2017). Information about oil contamination effects on sediment microbial communities and the effects of bioremediation techniques on microbial diversity in mangroves are also needed (Santos *et al.,* 2011; Machado & Lacerda, 2004).

Improved management for overharvested species requires inventories, baselines, and monitoring knowledge of targeted species. Managers need to know population densities, sizes and trends, breeding and migration patterns, and ecological conditions they require. Understanding the threats that are causing their decline (e.g. trade markets) as well as traditional values and knowledge will assist both management and enforcement.

There are active efforts to organize partnerships and collaborations to observe biodiversity and ecosystem characteristics in the Americas. Specifically, a series of Biodiversity Observation Network efforts are being organized under the Group on Earth Observations with some of these are at the country level. Networks of regional observation systems that collaborate and share information, and that work jointly to understand biodiversity and ecosystems could provide support to existing national programs and contribute to address United Nations SDG.

4.9 SUPPLEMENTARY MATERIAL

Box 4 17 Nutrient pollution in the Mississippi River and Gulf of Mexico.

Run-off from fields used for food and fiber production, point sources of municipal waste (from human waste and manufacturing), as well asand urban run-off, can transport nutrients and sediment to rivers and streams. This can increase nutrient (phosphorus, nitrogen, and carbon) concentrations and promote algal and aquatic vegetation growth causing eutrophication.

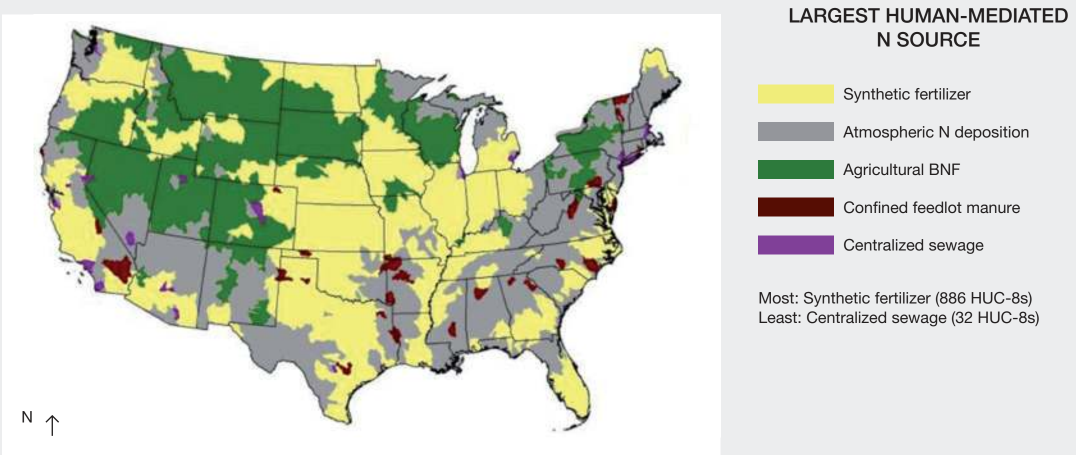
Over the last 30 years a hypoxic zone in the northern Gulf of Mexico has been measured each summer. This is an area along the Louisiana-Texas coast in which water near the bottom of the Gulf contains less than two parts per million of dissolved oxygen. Hypoxia can cause fish to leave the area disrupting fisheries and can cause stress or death to bottom dwelling organisms that can't move out of the hypoxic zone. Hypoxia is believed to be caused primarily by excess nutrients delivered from the Mississippi river in combination with seasonal stratification of Gulf waters. Excess nutrients promote algal and attendant zooplankton growth. The associated organic matter sinks to the bottom where it decomposes, consuming available oxygen. Stratification of fresh and saline waters prevents oxygen replenishment by mixing of oxygen-rich surface water with oxygen-depleted bottom water. Despite scientific concern, serious debate and billions of dollars used to ameliorate the offsite movement of nutrients in the Mississippi river basin over the past 20 years, the amount of nutrients being discharged from the Mississippi river into the Gulf of Mexico has not decreased (Sprague *et al.,* 2011).

Shorebirds like the interior least tern and piping plover preferred habitat is sparsely vegetated sandbars along rivers or lakes and reservoir shorelines. The interior least tern was put on the Endangered Species List in the USA in 1985 and it was widely believed that river engineering threatened the species continued existence especially in the lower Mississippi river. In 2013, a Government report recommended that the interior least tern be removed from the list of plants and animals protected by the Endangered Species Act. Much of the credit for this has been given to two Federal agencies, The Fish and Wildlife Service and the Army Corps of Engineers who have specific differing responsibilities in manging the Mississippi river basin, but decided to cooperate in order to achieve objectives of flood control, navigation, and biodiversity (Nielsen, 2014).

One of the major improvements to interior least tern habitat came from a slight modification to the many engineered dikes along the lower Mississippi river which are used to focus the current into the main channel. Many of this dikes had notches built into them that allow some water through and creates backwater for fish habitat and keeps the interior least tern sand bars, isolated from shore and away from mammalian predators. Now, as Paul Hartfield, from Jackson, Mississippi, says "the interior least tern is one of the most abundant shorebirds in the lower Mississippi river" (Nielsen, 2014).

Nutrient and organic matter pollution from human sewage, urban runoff and agriculture are also a major concern in Central and South America and the Caribbean. Most municipal wastewater in South America is not treated, and rivers and estuaries draining lands with large urban areas or extensive agriculture, like the Río de la Plata, exhibit relatively high concentrations of dissolved nitrogen and organic matter (Bustamante *et al.,* 2015; Mekonnen *et al.,* 2015; Venturini *et al.,* 2015). Eutrophic zones are also found in the Amazon river basin.

Figure 4 17 Dominant sources of nitrogen to USA watershed units. Watershed units are hydrologic unit code level 8. Source: Sobota *et al.* (2013).



Box 4 18 Organochlorine contaminant effects on bald Eagles in the Laurentian Great Lakes.

Bald eagles (*Haliaeetus leucocephalus*) have been treated as bioindicator species in the recovery of the Laurentian Great Lakes from organochlorine contamination. As studies documented in early studies (Mitchell *et al.,* 1953; Wurster *et al.,* 1965; Wurster & Wingate, 1968) in addition to acute toxicity to songbirds, offspring of certain bird populations suffered from eggshell thinning when adults were exposed to commercial DDT. Commercial DDT is a mixture of compounds including dichlorodiphenyldichloroethylene, a much more potent toxicant towards avian populations than DDT itself. Migration surveys showed drastic declines of bald eagles from the 1940s-1960s. The species almost became extinct (Farmer *et al.,* 2008), but populations have shown recovery since the 1970s.

The recovery of the bald eagle population in the Great Lakes was not uniform, however (Bowerman *et al.,* 1995). Bald eagles nesting along the shores of the lakes and rivers open to spawning runs of anadromous fishes from the Great Lakes continued to exhibit impaired reproduction due to continued exposure to contaminants through consumption of contaminated fish. Total polychlorinated biphenyls, dichlorodiphenyldichloroethylene and also 2,3,7,8-tetrachlordibenzo-dioxin equivalents (TCDD-EQ; http://www.dioxinfacts.org/tri_dioxin_data/sitedata/test3/def.html) in fishes were shown to represent a significant hazard to bald eagles living along these shorelines or near the rivers. Bowerman *et al.* (1998) attributed the recovery of the bald eagle population along the Great Lakes to immigration of healthy individuals from interior regions. This conclusion was supported by findings that the reproduction rate of bald eagles nesting along Lake Superior's shore was significantly less than that in neighboring inland regions in Wisconsin and other inland Great Lakes sites (Dykstra *et al.,* 1998). It was concluded that the low productivity of Lake Superior eagles was at least partly attributable to low food availability, but another factor, possibly polychlorinated biphenyls, could also have contributed to low productivity. Dykstra *et al.* (2001) further showed that bald eagle populations nesting on the shores of Green Bay, Lake Michigan, where concentrations of polychlorinated biphenyls are high, due to the historical presence of numerous pulp and paper mills, had reproductive rates significantly lower than those of neighboring eagles nesting inland (0.55 versus 1.1 young per occupied territory). It was concluded that organochlorine contaminants caused all or most of the depression in reproductive rates of Green Bay bald eagles.

More recently bald eagle populations have recovered. Although other contaminants, including methylmercury (Depew *et al.,* 2013), may have sublethal or lethal effects, Dykstra *et al.* (2005) found that concentrations of polychlorinated biphenyls and dichlorodiphenyldichloroethylene decreased significantly in bald eagle nestling blood plasma from Lake Superior from 1989-2001. Mean concentrations were near or below threshold concentrations for reproduction impairment, and reproductive rate and contaminant concentrations were not correlated, suggesting that polychlorinated biphenyls and dichlorodiphenyldichloroethylene no longer limited Lake Superior eagle population reproduction.

Figure 4 18 **Bald eagle (*Haliaeetus leucocephalus*) Photo Credit: Ron Holmes / U.S. Fish and Wildlife Service.**

Box 4 19 Pollution in Greenland.

Mining within Greenland is limited but related issues with pollution can occur. For example, the Black Angel mine in Maarnorilik, West Greenland, one of the richest zinc mines in the world, operated from 1973 to 1990 and restarted in 2009, has contaminated nearby waters with heavy metals especially zinc, lead, and mercury, plus others. But 30 km from the mine heavy metals are not elevated (Perner *et al.,* 2010).

In 2004 - 2005 air samples were collected from a site in Nuuk, in Southwestern Greenland and analyzed for a suite of persistent organic pollutants. The results from the study indicate that a number of persistent organic pollutants were detected in the air in significant quantities; these included alpha and gamma hexacholorhexane, cis- and trans- chlordane, dieldrin, and degradants of DDT.

There were several studies, in two locations in Greenland that examined the long-term trends in persistent organic pollutants in biota including ringed seal, seabirds, and fish. In Greenland, there were no upwards trends in concentrations for any persistent organic pollutants, most had decreasing concentrations, although not all were statistically significant (Hung *et al.,* 2005; Rigét *et al.,* 2010).

In another study that examined 17 whitetail eagles found dead in Western Greenland from 1997 to 2009 all had detectable levels of persistent organic pollutants and methoxylated polybrominated diphenyl ethers in different tissues. The majority of the chemicals were found in muscle tissue and the largest portion of sum of the chemicals was polybrominated biphenyl ethers with over 50% of the totals, followed by components of DDT. Collectively the concentrations in the birds did not reach known toxic levels, but some individual birds did have levels that would be considered toxic (Jaspers *et al.,* 2013)

In Greenland, pregnant Inuit women, women of child-bearing age and infants have high mercury and persistent organic pollutants levels in maternal blood and hair; maternal blood mercury levels exceed guidelines and are much greater compared with most Europeans; and mercury levels increase with increasing marine mammal consumption (Bjerregaard & Hansen, 2000; Dietz *et al.,* 2013; Visnjevec *et al.,* 2014; Weihe *et al.,* 2002). The combined evidence suggests mercury exposure is causing subtle neurobehavioral deficits in children (Weihe *et al.,* 2002). In the Faroe Islands, which are also in the north Atlantic, modeling suggests that mercury inputs would have to decline by ~50% to achieve safe Inuit exposure levels (Booth & Zeller, 2005), which is about the portion of the global environmental mercury burden that has man-made origins (Bergan *et al.,* 1999). Polar bears in Greenland also have mercury levels in tissues that are high enough to be toxic. As in other Arctic biota, Greenland birds of prey have been exposed to steadily increasing levels of mercury, beginning with the industrial revolution and through the 10th century, as indicated by feather mercury levels. A few samples from the late 20th century suggest recent declines in mercury (Dietz *et al.,* 2006).

Box 4 20 Pollution of South American mangroves.

South American mangroves are threatened by human-induced alterations in the nitrogen and phosphorus cycles. Increased nitrogen availability originating from agriculture and mining activities, sewage pollution, and also from shrimp farming and direct solid waste disposal that take place in South American mangroves (Lacerda *et al.,* 2002; Castellanos-Gallindo *et al.,* 2014; Rodríguez-Rodríguez *et al.,* 2016) can lead to intensification of nitrogen cycling in mangrove sediment with direct effects on ecosystem functioning and also potential indirect effects on ecosystem structure and biodiversity. As a consequence of anthropogenic nitrogen enrichment, mangroves may increase nitrous oxyde fluxes to the atmosphere, also contributing to global warming (Reis *et al.,* 2017). Phosphorous enrichment may also extensive affect nutrient cycling in mangrove sediment by modifying physical and chemical conditions and phosphorus fractionation, and by increasing microbial activity and organic matter decomposition in sediment (Nóbrega *et al.,* 2014). Other pollutants affecting mangroves in South America are oil spills (Lacerda & Kjerfve, 1999; Lacerda *et al.,* 2002) and toxic metals (Machado & Lacerda, 2004). In general, consequences of oil spills to mangroves include trees defoliation and leaf deformation, mortality of seedlings and trees, bioaccumulation of toxic compounds, and reduction in faunal density, which can persist over many years after the spill (Lacerda *et al.,* 2002). Oil spills were also reported to affect the structure and biodiversity of microbial and fungal communities in mangrove sediment (e.g. Taketani *et al.,* 2010; Fasanella *et al.,* 2012). Enhanced trace metal availability due to engineering works at watersheds and input of waste from urban and industrial centers and aquaculture and agriculture areas has favored trace metals trapping and storage in mangrove sediment (e.g. Machado & Lacerda, 2004; Lacerda *et al.,* 2011; Costa *et al.,* 2013). While the retention of such elements within mangrove sediments may contribute to the reduction of metal transfer to surrounding coastal areas, it may also cause negative effects on mangrove plants and animals, with special concerns on transfer within food chains, and transfer to man through fisheries (Machado & Lacerda, 2004).

Box ④ ㉑ Case study: *Pterois volitans* (Linnaeus 1758) and *P. miles* (Bennett 1828) Family Scorpaenidae.

The Indo-Pacific lionfish is the first nonnative marine fish to establish in the western north Atlantic and Caribbean Sea. The lionfish invasion is predicted to be the most ecologically impacting marine invasion ever recorded (Albins & Hixon, 2011). Invasive lionfish prey on a wide range of native fish species (Côté *et al.,* 2013) due to a suite of predatory characteristics and behaviors that have no parallel in the Atlantic (Albins & Lyons, 2012; Albins & Hixon, 2013). Field experiments have demonstrated that lionfish reduced recruitment of native species in coral reef patches, including important functional groups like parrotfishes (Albins & Hixon, 2008; Green *et al.,* 2012). The reduction in the abundance of native fishes caused by lionfish in controlled experiments was 2.5 times greater than the one caused by a similarly sized native predator (Albins, 2013), suggesting that lionfish can outcompete native predators. The first confirmed record of lionfish occurrence in the USA was a specimen taken 1985 (Morris & Akins, 2009). Whitfield *et al.* (2002) documented the presence and likely establishment of the Indo-Pacific lionfish *Pterois volitans* in the western Atlantic. They postulated that the source of the introduction was the marine aquarium trade. Lionfish specimens are now found along the USA east coast from Cape Hatteras, North Carolina, to Florida, and in Bermuda, The Bahamas, and the Caribbean throughout, treats including the Turks and Caicos, Haiti, Cuba, Dominican Republic, Puerto Rico, St. Croix, Belize, and Mexico (Schofield, 2009; Schofield, 2010; Betancur *et al.,* 2011). In less than 30 years, lionfish have dramatically expanded their non-native distribution range to an area of roughly 7.3 million km^2, encompassing the eastern coast of the USA, Bermuda, the entire Caribbean region and the Gulf of Mexico (Schofield, 2010). Because of euryhaline and eurythermal features of this species, its expansion was not constrained by the Amazon-Orinoco plume (Luiz *et al.,* 2013) and it was recently reported almost in the southeastern coast of Brazil expanding its distribution range to the Atlantic coast of South America (Ferrerira *et al.,* 2015)

Box ④ ㉒ Impacts of invasive alien species *Clarias* sp. on populations of freshwater fish in the biosphere Reserve Cienaga de Zapata, Cuba.

Biosphere Reserve Cienaga de Zapata, is the largest wetland in the Caribbean islands and is home to high biodiversity in the presence of many local endemic. As 75% of the territory is flooded, water regime is the main ecological factor that determines the characteristics of its complex ecosystems (ACC-ICGC, 1993).The physical, geographical and hydrological characteristics, together with the periodic floods that occur in rainy periods, and the incidence of major hurricanes, have influenced the introduction and rapid increase of two exotic and invasive species of the genus *Clarias* (*Clarias macrocephalus* and *Clarias gariepinus*), being more abundant *C. gariepinus*. This is an omnivorous species with high fertility, rapid growth and high resistance to diseases, and stress management, justifying its rapid distribution in the natural environment.

Studies for more than a decade (2003-2014) on the impact of the species on wetland biodiversity are based on the results of the analysis of stomach contents. These results showed that *C. gariepinus* feeding was mainly composed of fish in the first two years of sampling, predominantly the endemic, biajaca criolla (*Nandopsis tetracantus*) accounted for 12.5% of the diet. This species was not found in the stomach contents in the later years. Simultaneously, the analysis of the variation in the composition of catching fish companions showed that in less than two years, fish populations with some degree of endemism began to decline drastically and only introduced species maintain their populations. Importantly, from 2002, specimens of the genus *Clarias* were the most abundant in catches.

Today, populations of biajaca criolla have declined substantially in the wetland, proving to be rare in the lakes and rivers. Studies by Perez & Duarte in 1990 linked the decline in populations of biajaca criolla in Cuba with the introduction of other exotic species such as trout (*Micropterus salmoides*) and sunfish (*Lepomis macrochirus*). However, in 1979 the biajaca criolla represented 46.7% of the population of fish in Laguna del Tesoro, while 24.3% and 20.6% were trout and sunfish, respectively. It is with the arrival of specimens of the genus *Clarias* that the effects on this Cuban endemic species of freshwater fish (meat is of great commercial value), belonging the family Cichlidae became stronger (Howell Rivero & Rivas, 1940; Vales *et al.,* 1998).

Box 4 ㉓ More than an invasive ecosystem engineer: introduced beavers in southern Patagonia as a social-ecological system.

In the 1940s and 1950s, government and private initiatives brought various exotic species to Patagonia, including Canadian beavers (*Castor canadensis*), American mink (*Neovison vison*), muskrats (*Ondatra zibethicus*), red deer (*Cervus elaphus*) and European rabbits (*Oryctolagus cuniculus*) (Ballari *et al.,*2016). The re-construction of this ecological landscape was largely driven by a cultural "mindscape" that valued Northern Hemisphere species over local ones, conceiving these introductions as a way to "enhance" the fauna, "develop" the region or bring "progress" to a remote area (e.g. Sucesos Argentinos) (Anonymous,1946).

Since the late 1990s, ecological research has mostly quantified the negative impacts of introduced invasive species and focused on emblematic or problematic cases like the beaver (Anderson & Valenzuela, 2014). For example, the biological invasion by beavers has been shown to be a significant transformation of sub-Antarctic forests in the Holocene. As an invasive ecosystem engineer, the beaver creates novel ecosystems conformed by meadows and ponds that reorganize biotic communities and facilitate the spread of other exotic flora and fauna, but they also provide habitat for native waterfowl and fish (Anderson *et al.,* 2014). However, unlike the northern hemisphere, southern Patagonian forests in particular are not resilient to beaver impacts, and therefore, they require active restoration measures to ameliorate beaver impacts (Wallem *et al.,* 2010). This ecological information motivated Argentine and Chilean decision-makers to agree to eradicate beavers and restore degraded ecosystems. However, it quickly became apparent that achieving these goals required understanding not only ecological dimensions, but also social aspects of this system. Although global images of Patagonia tend to project it as an unsullied wilderness, but it has a long history of human habitation and a modern social context that is quite complex (Moss, 2008). In the case of beavers, an eradication program must recognize that the Tierra del Fuego Archipelago is one biogeographic unit, but it is administered by two nations with different political-administrative systems. Furthermore, different social groups within each country understand their relationship with beavers differently. For example, while environmental managers in southern Patagonia rank invasive species as a primary threat to ecosystems, the 98% of residents who live in cities do not perceive them as a priority problem (Zagarola *et al.,* 2014). Indeed, the novel social context of beavers includes the fact that they have become a symbol for various tourism enterprises and companies, particularly in Argentina. This social system includes not only two nation-States, but diverse stakeholders and social groups that have multi-relationships and perspectives with this multi-natural ecosystem (Santo *et al.,* 2015). Incorporating this complexity of human and environmental factors means reconceiving biological invasions and restoration ecology as social-ecological systems for both research and management, but achieving this recognition has literally taken decades. By recognizing the social-ecological dimensions of invasive exotic species, not just their «biological invasion», ecologists would be better positioned to effectively and efficiently address these and other problems in association with not only other academic disciplines, but other social actors that are part of the study and management of environmental issues.

Box 4 ㉔ Case study: *Limnoperna fortunei* (Dunker, 1857).

This mussel species, commonly known as the golden mussel, is native to the freshwater systems southeast China. Because of the ecological effects caused in aquatic ecosystems and expenses incurred in industrial infrastructure concerned is considered as aquatic invasive species and environmental issues at regional level (Darrigran, 2002). It was accidentally introduced to the region of the Río de la Plata basin in 1991 through ballast water and first reported on the coast of Río de la Plata, Buenos Aires (Pastorino *et al.,* 1993, Darrigran & Pastorino, 1995). Currently, it has a rapid ascent up the Río de la Plata basin (feed rates of 250 km per year), invading major rivers (Río de la Plata, Uruguay, Parana, Paraguay, Tiete) and smaller water systems in basins Guaíba, Tramandaí (south east Brazil), Laguna de los Patos-Mirim (Brazil-Uruguay), Mar Chiquita (Argentina-central) or Laguna del Sauce (east coast Uruguay) (de Oliveria *et al.,* 2015). It is currently in aquatic environments from five countries in South America: Argentina, Brazil, Bolivia, Paraguay and Uruguay, identified as the main vector of invasion commercial navigation on the waterway of the Río de la Plata basin (Karatayev *et al.,* 2006). Since its arrival to the region, it was found associated with a variety of natural and artificial substrates consolidated, increasing its population abundances, causing changes in the benthic communities and in the eating habits of native fish. It generates further problems macrofouling (settlement and colonization of organisms greater than 50 micrometre on artificial substrates) in hydraulic systems of companies and industries that use different branches water resources in their production cycles (Boltovskoy & Correa, 2014). Among the effects caused are clogging of filters, disablement of hydraulic sensors, damages to pumps or decreased uptake diameter line pipe for cooling water, irrigation, or water purification. These effects cause overhead in major water purification water plants, nuclear, hydroelectric plants, refineries, steel mills and agro-industrial plants (aquaculture, forestry, food), due to maintenance, structural modifications, as well as management plans and population control (Brugnoli *et al.,* 2006; Boltovskoy & Correa, 2014; Boltovskoy *et al.,* 2015).

Box 4 ㉕ Case study: *Rapana venosa* (Valenciennes, 1846).

The snail rapana is native to the Sea of Japan, Yellow Sea, Bohai Sea and the Sea of China to Taiwan (Mann *et al.,* 2004). In 1947, it was described for the first time outside of its original range in the Black Sea and then subsequently reported in the Azov, Aegean, Adriatic Seas and North America (Pastorino *et al.,* 2000, Mann *et al.,* 2004, Kerckhof *et al.,* 2006). It is a predator of molluscs subtidal, usually feeding on bivalves of economic interest such as oysters, mussels and clams (Harding & Mann, 1999; Savini & Occhipinti-Ambrogi, 2006; Giberto *et al.,* 2011; Lanfranconi *et al.,* 2013).

It was first recorded in South America in 1999 in the Río de la Plata, Argentinian coast (Bay Samborombón) (Pastorino *et al.,* 2000). A decade after its first records outside Samborombón Bay, the species expanded its distribution to all muddy bottoms of the subtidal mixohaline zone of the Río de la Plata (Giberto *et al.,* 2006). For the Uruguayan coast of the Río de la Plata, Scarabino *et al.* (1999) reported on the coast of Maldonado; meanwhile, Carranza *et al.* (2007) describe its distribution in the outer area of the Río de la Plata. Currently, it presents its limit of this distribution in the Bay of Maldonado-Punta del Este (Lanfranconi *et al.,* 2009; Carranza *et al.,* 2010).

Perception of local communities: conducting a study with a multidisciplinary approach involving biologists, sociologists and consultation of fisherfolk (mussel) in the south east of Uruguay coast, allowed to highlight the importance of considering local knowledge with stakeholders involved daily with the impact of invasive species on fishery resources (Brugnoli *et al.,* 2014). The «empirical» knowledge, largely consolidates existing scientific knowledge concerning *R. venosa* and, in certain cases, brings new questions for future research. Both approaches (scientific-community local) agree on the dates of the first observations of the snail to the area as well as observation of mucous trail left by its movement. This empirical knowledge as well as information collected in the field by local people, is sometimes prescinded by the academy. However, it could play an important role in monitoring programs that include early warning, monitoring of abundance and distribution, as well as the identification of direct or indirect effects on the native fauna caused by invading organisms like *R. venosa*

REFERENCES

Abbott, B. W., Jones, J., Schuur, E., Chapin III, F. S., Bowden, W., Bret-Harte, M., Epstein, H., Flannigan, M., Harms, T., Hollingworth, T., Mack, M., McGuire, A. D., Natali, S., Rocha, A., Tank, S., Turetsky, M., Vonk, J. E., Wickland, K. P., Aiken, G. R., Alexander, H., Amon, R. M. W., Welker, J., & Zimov, S. (2016). Biomass offsets little or none of permafrost carbon release from soils, streams, and wildfire: an expert assessment. Environmental Research Letters, 11(3), 34014. https://doi.org/10.1088/1748-9326/11/3/034014

Abreu, R. C. R. de, & Durigan, G. (2011). Changes in the plant community of a Brazilian grassland savannah after 22 years of invasion by Pinus elliottii Engelm. *Plant Ecology & Diversity*, *4*(2–3), 269–278. https://doi.org/10.1080/17550874.2011.594101

ACC-ICGC. (1993). *Estudio Geográfico Integral. Ciénaga de Zapata*. La Habana: Publicaciones del Servicio de Información y Traducciones.

Achard, F., Eva, H. D., Stibig, H. J., Mayaux, P., Gallego, J., Richards, T., & Malingreau, J. P. (2002). Determination of deforestation rates of the world's humid tropical forests. *Science, 297*(5583), 999-1002.

Achard, F., Eva, H., Glinni, A., Mayaux, P., Richards, T., & Stibig, H. J. (1998). Identification of deforestation hot spot areas in the humid tropics. *Trees Publication Series B*, *4*, 1–81.

Adkins, Collette. (2016). The US Endangered Species Act: a powerful tool to protect biodiversity (if we use it). *Biodiversity*, *17*(3), 101-103.

Affonso, A G., Barbosa, C., & Novo, E. M. L. M. (2011). Water quality changes in floodplain lakes due to the Amazon River flood pulse: Lago Grande de Curuaí (Pará). *Brazilian journal of biology (Revista brasleira de biologia), 71*(3), 601–10. http://www.ncbi.nlm.nih.gov/pubmed/21881783

Agard, J., Cropper, A., Aquing, P., Attzs, M., Arias, F., Beltran, J., Bennett, E., Carnegie, R., Clauzel, S., Corredor, J., & Creary, M. (2007). Caribbean Sea ecosystem assessment (CARSEA). *Caribbean Mar Stud*, 8, 1-85.

Agencia EFE. (1998). *Anuario Iberoamericano*. Madrid: Agencia EFE / ANUARIO.

Agnew, D. J., Gutiérrez, N. L., Stern-Pirlot, A., & Hoggarth, D. D. (2014). The MSC experience: Developing an operational certification standard and a market incentive to improve fishery sustainability. *ICES Journal of Marine Science, 71*(2), 216–225.

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. (2015). The dominant role of semi-arid ecosystems in the trend and variability of the land CO$_2$ sink. *Science, 348*(6237), 895-899. DOI: http://science.sciencemag.org/content/348/6237/895

Aide, T. M., Clark, M. L., Grau, H. R., Lopez-Carr, D., Levy, M. A., Redo, D., Bonilla-Moheno, M., Riner, G., Andrade-Nuñez, M. J., & Muñiz, M. (2013). Deforestation and reforestation of Latin America and the Caribbean (2001 – 2010). *Biotropica*, *45*(2), 262–271. https://doi.org/10.1111/j.1744-7429.2012.00908.x

Ajanovic, A. (2011). Biofuels versus food production: does biofuels production increase food prices? *Energy, 36*(4), 2070–2076.

Albert, D. (2000). *Borne of the wind: an introduction to the ecology of Michigan sand dunes.* Michigan Natural Features Inventory. Michigan. https://mnfi.anr.msu.edu/pub/dunes/borneofthewind.pdf

Albins, M. A. & Hixon, M. A. (2013). Worst-case scenario: potential long-term effects of invasive predatory lionfish (*Pterois volitans*) on Atlantic and Caribbean coral-reef communities. *Environmental Biology of Fishes, 96*(10-11), 1151-1157.

Albins, M. A. (2013). Effects of invasive Pacific red lionfish *Pterois volitans* versus a native predator on Bahamian coral-reef fish communities. *Biological Invasions, 15*(1), 29–43.

Albins, M. A. & Hixon, M. A. (2008). Invasive Indo-Pacific lionfish *Pterois volitans* reduce recruitment of Atlantic coral-reef fishes. *Marine Ecology Progress Series,* 367, 233–238.

Albins, M. A. & Hixon, M. A. (2011). Worst case scenario: Potential long-term effects of invasive predatory lionfish (*Pterois volitans*) on Atlantic and Caribbean coralreef communities. *Environmental Biology of Fishes*, *96*(10–11), 1151–1157.

Albins, M. A. & Lyons, P. J. (2012). Invasive red lionfish *Pterois volitans* blow directed jets of water at prey fish. *Marine Ecology Progress Series*, 448, 1–5.

Albright, R., & Langdon, C. (2011). Ocean acidification impacts multiple early life history processes of the Caribbean coral *Porites astreoides*. *Global Change Biology, 17*(7), 2478–2487.

Alcamo, J., Vuuren, D. Van, Cramer, W., Alder, J., Bennett, E., Carpenter, S., Christensen, V., Foley, J., Masui, T., Morita, T., Neill, B. O., Peterson, G., Ringler, C., Schulze, K., Bouwman, L., Eickhout, B., Floerke, M., Lal, R., Takahashi, K., Editors, R., Tan, B., Hammond, A., & Field, C. (2005). Changes in Ecosystem Services and Their Drivers across the Scenarios. *Ecosystems and Human Well-Being: Scenarios* 2(2), 297–373.

Alencar, A. A., Brando, P. M., Asner, G. P., & Putz, F. E. (2015). Landscape fragmentation, severe drought, and the new Amazon forest fire regime. *Ecological applications, 25*(6), 1493-1505.

Alencar, A., Nepstad, D., & Vera Diaz, M. C. (2006). Forest understory fire in the Brazilian Amazon in ENSO and non-ENSO years: area burned and committed carbon emissions. *Earth Interactions 10*(6), 1–17. https://doi.org/10.1175/EI150.1

Alexander, C., Bynum, N., Johnson, E., King, U., Mustonen, T., Neofotis, P., Oettlé, N., Rosenzweig, C.,

Sakakibara, C., Shadrin, V., Vicarelli, M., Waterhouse, J., & Weeks, B. (2011). Linking Indigenous and scientific knowledge of climate change. *BioScience, 61*(6), 477–484. https://doi.org/10.1525/bio.2011.61.6.10

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., Reis, R., & Aquino, P. (2015). Amazonian freshwater habitats experiencing environmental and socioeconomic threats affecting subsistence fisheries. *Ambio, 44*(5), 412–425. https://doi.org/10.1007/s13280-014-0610-z

Alig, R. J., Kline, J. D., & Lichtenstein, M. (2004). Urbanization on the US landscape: looking ahead in the 21st century. *Landscape and urban planning, 69*(2), 219-234.

Alkemade, R., van Oorschot, M., Miles, L., Nellemann, C., Bakkenes, M., & Ten Brink, B. (2009). GLOBIO3: a framework to investigate options for reducing global terrestrial biodiversity loss. *Ecosystems*, *12*(3), 374–390. https://doi.org/10.1007/s10021-009-9229-5

Allan, J. D., McIntyre, P. B., Smith, S. D. P., Halpern, B. S., Boyer, G. L., Buchsbaum, A., Burton, G. A., Campbell, L. M., Chadderton, W. L., Ciborowski, J. J. H., Doran, P. J., Eder, T., Infante, D. M., Johnson, L. B., Joseph, C. A., Marino, A. L., Prusevich, A., Read, J. G., Rose, J. B., Rutherford, E. S., Sowa, S. P., & Steinman, A. D. (2013). Joint analysis of stressors and ecosystem services to enhance restoration effectiveness. *Proceedings of the National Academy of Sciences of the United States of America*, *110*(1), 372–377. https://doi.org/10.1073/pnas.1213841110

Allan, J. D., Abell, R. Hogan, Z., Revenga, C., Taylor, B. W., Welcomme, R. L., & Winemiller, K. (2005). Overfishing of Inland Waters. *BioScience 55*(12), 1041–1051.

Almeida-Neto, M., Prado, P. I., & Lewinsohn, T. M. (2011). Phytophagous insect fauna tracks host plant responses to exotic grass invasion. *Oecologia*, *165*(4), 1051–1062. https://doi.org/10.1007/s00442-010-1783-1

Alongi, D. M. (2008). Mangrove forests: resilience, protection from tsunamis, and responses to global climate change. *Estuarine, Coastal and Shelf Science, 76*(1), 1-13.

Alongi, D. M. (2015). The impact of climate change on mangrove forests. *Current Climate Change Reports, 1*(1), 30-39. https://doi.org/10.1007/s40641-015-0002-x

Altieri, M.A., & Funes-Monzote, F. R. (2012). The paradox of Cuban agriculture. *Monthly Review*, 63(8), 23-33.

Altieri, M.A., Funes-Monzote, F.R., & Petersen, P. (2012). Agroecologically efficient agricultural systems for smallholder farmers: contributions to food sovereignty. *Agronomy for Sustainable Development, 32*(1), 1-13.

Álvarez-Berríos, N. L., Redo, D. J., Aide, T. M., Clark, M. L., & Grau, R. (2013). Land Change in the Greater Antilles between 2001 and 2010. *Land*, *2*(2), 81-107.

Alvarez-Filip, L., Dulvy, N. K., Gill, J. A., Côté, I. M., & Watkinson, A. R. (2009). Flattening of Caribbean coral reefs: region-wide declines in architectural complexity. *Proceedings. Biological sciences / The Royal Society* 276*(1669),* 3019–25. https://doi.org/10.1098/rspb.2009.0339

Anderson, C. B., Griffith, C. R., Rosemond, A. D., Rozzi, R., & Dollenz, O. (2006). The effects of invasive North American beavers on riparian plant communities in Cape Horn, Chile: Do exotic beavers engineer differently in sub-Antarctic ecosystems? *Biological Conservation*, *128*(4), 467–474. https://doi.org/10.1016/j.biocon.2005.10.011

Anderson, C. B. & Valenzuela, A. E. J. (2014). Do what I say, not what I do. Are we linking research and decision-making about invasive species in Patagonia? *Ecología Austral, 24*(2), 193-202.

Anderson, C.B., Lencinas, M.V., Valenzuela, A.E.J., Simononok, M.P., Wallem, P.K., & Martinez P., G. (2014). Ecosystem engineering by an invasive species, the beaver, increases landscape-level ecosystem function but does not affect biodiversity in Tierra del Fuego's freshwater systems. *Diversity and Distributions, 20*(2), 214–222.

Anderson, E. P. (2013). *Hydropower development and ecosystem services in Central America*. Inter-American Development Bank. Technical note No. IDB-TN-513.

Anderson, O. R. J., Small, C. J., Croxall, J. P., Dunn, E. K., Sullivan, B. J., Yates, O., & Black, A. (2011). Global seabird bycatch in longline fisheries. *Endangered Species Research, 14*(2), 91–106. https://doi.org/10.3354/esr00347

Anderson, P. J., & Piatt, J. F. (1999). Community reorganization in the Gulf of Alaska following ocean climate regime shift. *Marine Ecology Progress Series*, 189, 117-123.

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, 177–181.

Angel, J. R., & Huff, F. A. (1997). Changes in heavy rainfall in midwestern U.S. *Journal of Water Resources Planning and Management*, *123*(4), 246–249. https://doi.org/10.1061/(ASCE)0733-9496(1997)123:4(246)

Angel, J. R., & Kunkel, K. E. (2010). The response of Great Lakes water levels to future climate scenarios with an emphasis on Lake Michigan-Huron. *Journal of Great Lakes Research, 36* (spl 2*)*, 51–58. http://dx.doi.org/10.1016/j.jglr.2009.09.006

Angelelli, P., & Saffache, P. (2013). Some remarks on mangroves in the Lesser Antilles. *Revista de Gestão Costeira Integrada (Journal of Integrated Coastal Zone Management)*, *13*(4), 473-489

Anonymous. (1946). Viaje al Sur. *Sucesos Argentinos,* Buenos Aires: Government Newsreel.

Anthony, K. R. N., Maynard, J. A., Diaz-Pulido, G., Mumby, P. J., Marshall, P. A., Cao, L., & Hoegh-Guldberg, O. (2011). Ocean acidification and warming will lower coral reef resilience. *Global Change Biology,* *17*(5), 1798–1808. https://doi.org/10.1111/j.1365-2486.2010.02364.x

Appeldoorn, R. S., Gonzalez, E. C., Glazer, R., & Prada, M. (2011). Applying EBM to queen conch fisheries in the Caribbean. In Fanning, L., R. Maho, & P. McConney, (Eds.). *Towards marine ecosystem-based management in the Caribbean*. (pp.177–186)Centre for Maritime Research, Amsterdam: Amsterdam University Press.

Apte, J. S., Marshall, J. D., Cohen, A. J., & Brauer, M. (2015). Addressing global mortality from ambient PM2. 5. *Environmental Science & Technology, 49*(13), 8057-8066.

Archer, S., Schimel, D. S., & Holland, E. A. (1995). Mechanisms of shrubland expansion: land use, climate or CO_2? *Climatic Change*, *29*(1), 91–99. https://doi.org/10.1007/BF01091640

Armendáriz-Villegas, E. J., Covarrubias-García, M. D. L. A., Troyo-Diéguez, E., Lagunes, E., Arreola-Lizárraga, A., Nieto-Garibay, A., Beltrán-Morales, L.F., & Ortega-Rubio, A. (2015). Metal mining and natural protected areas in Mexico: Geographic overlaps and environmental implications. *Environmental Science and Policy*, 48, 9-19. and vegetation in Colombia. *Agricultural and Forest Meteorology, 151*(3), 279-289.

Armenteras-Pascual, D., Retana-Alumbreros, J., Molowny-Horas, R., Roman-Cuesta, R. M., Gonzalez-Alonso, F., & Morales-Rivas, M. (2011). Characterising fire spatial pattern interactions with climate and vegetation in Colombia. *Agricultural and Forest Meteorology, 151*(3), 279-289. http://doi.org/10.1016/j.agrformet.2010.11.002

Armitage, D., De Loë, R., & Plummer, R. (2012). Environmental governance and its implications for conservation practice. *Conservation Letters, 5*(4), 245–255. http://doi.org/10.1111/j.1755-263X.2012.00238.x

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., Kuhn, I., MacGregor-Fors, I., McDonnell, M., Mortberg, U., Pysek, P., Siebert, S., Sushinsky, J., 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), 20133330. https://doi.org/10.1098/rspb.2013.3330

Arriaga, L. (2000). Types and causes of tree mortality in a tropical montane cloud forest of Tamaulipas, Mexico. *Journal of Tropical Ecology*, 16(5), 623-636.

Arriaga, L., Castellanos, A. E., Moreno, E., & Alarcon, 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. https://doi.org/10.1111/j.1523-1739.2004.00166.x

Arsel, M., Hogenboom, B., & Pellegrini, L. (2016). The extractive imperative and the boom in environmental conflicts at the end of the progressive cycle in Latin America. *Extractive Industries and Society, 3*(4), 877–879. https://dare.uva.nl/search?identifier=8d0bd84f-002b-4fe0-bb92-98741f98c79b

Astor, Y. M., Lorenzoni, L.B., Thunell, R., Varela, R., Muller-Karger, F., Troccoli, L., Taylor, G. T., Scranton, M. I., Tappa, E., & Rueda, D. (2013). Interannual variability in sea surface temperature and CO_2 changes in the Cariaco Basin. *Deep-Sea Research II, special issue of Ocean Biogeochemistry Time Series Research*, 93, 33-43. https://doi.org/10.1016/j.dsr2.2013.01.002

Auch, R.F. (2016). *Middle Atlantic Coastal Plain*. U.S. Geological Survey.

Avcı, D., & Fernández-Salvador, C. (2016). Territorial dynamics and local resistance: Two mining conflicts in Ecuador compared. *Extractive Industries and Society, 3*(4), 912–921. http://doi.org/10.1016/j.exis.2016.10.007

Ávila, S., Muñoz, C., Jaramillo, L., Martínez, A. (2005). *Un análisis del subsidio a la tarifa 09 Gaceta Ecológica, 75, abril-junio*. (pp. 65-76). México: Secretaría de Medio Ambiente y Recursos Naturales Distrito Federal.

Baker, D. B., Richards, R. P., Loftus, T. T., & Kramer. J. W. (2004). A new flashiness index: Characteristics and applications to midwestern rivers and streams. *Journal of the American Water Resources Association, 40*(2), 503–522.

Baldi, G., & Paruelo, J. (2008). Land-use and land cover dynamics in South American temperate grasslands. *Ecology and Society*, 13(2).

Ballari, S.A., Anderson, C. B. & Valenzuela, A. E. J. (2016). Understanding trends in biological invasions by introduced mammals in southern South America: a review of research and management. Mammal Review 46: 229–240.

Balvanera, P., Cotler, H., Aburto, O., Aguilar, A., Aguilera, M., Aluja, M., Andrade, A., Arroyo, I., Ashworth, L., Astier, L., Ávila, P., Bitrán, D., Camargo, T., Campo, J., Cárdenas, B., Casas, A., Díaz-Fleischer, F., Etchevers, J., Ghillardi, A., González-Padilla, E., Guevara, A., Lazos, E., López, C., López, R., Martínez, J., Masera, O., Mazari, M., Nadal, A., Pérez-Salicrup, D., Pérez-Gil, R., Quesada, M., Ramos-Elorduy, J., Robles, A., Rodríguez, H., Rull, J. Suzán, G., Vergara, C., Xolalpa, S., Zambrano, L., & Zarco, A. (2009). Estado y tendencias de los servicios ecosistémicos. *Capital natural de México*, 2, 185-245. Retrieved from http://www.biodiversidad.gob.mx/pais/pdf/CapNatMex/Vol II/II04_EdoTendenciasServiciosEcosistemicos.pdf

Barbarán, F. R., Rojas, L., & Arias, H. M. (2015). Sostenibilidad institucional y social de la expansión de la frontera agropecuaria. *Revibec: revista de la Red Iberoamericana de Economia Ecológica*, 24, 21-37. http://hdl.handle.net/11336/8829

Barbaran, F. (2015). Biodiversity values and payment for ecosystem services in Argentina: Who pays? IUCN Commission on Environmental, Economic and Social Policy (CEESP). *CEESP Newsletter*, 1515. http://iucn.org/union/commissions/ceesp/?21652

Barbour, A. B., Montgomery, M. L., Adamson, A. A., Díaz-Ferguson, E., & Silliman, B. R. (2010). Mangrove use by the invasive lionfish *Pterois volitans*. *Marine Ecology Progress Series*, 401, 291–294. https://doi.org/10.3354/meps08373

Barkley, A. N. (2015). Implications of developing deep-sea Arctic fisheries for Greenland Halibut (*Reinhardtius hippoglossoides*): Inshore stock connectivity and capture induced stress of ecologically important fish species. *Electronic Theses and Dissertations.* Paper 5429. University of Windsor. https://scholar.uwindsor.ca/etd/5429

Barona, E., Ramankutty, N., Hyman, G., & Coomes, O. T. (2010). The role of pasture and soybean in deforestation of the Brazilian Amazon. *Environmental Research Letters*, *5*(024002). Doi:10.1088/1748-9326/5/2/024002.

Barral, M. P., Rey Benayas, J. M., Meli, P. & Maceira, N. O. (2015). Quantifying the impacts of ecological restoration on biodiversity and ecosystem services in agroecosystems: A global meta-analysis. *Agriculture, Ecosystems and Environment,* 202, 223–23. https://doi.org/10.1016/j.agee.2015.01.009

Barros, V. R., Boninsegna, J. A. Camilloni, I. A., Chidiak, M., Magrín, G. O., & Rusticucci, M. (2015). Climate change in Argentina: Trends, projections, impacts and adaptation. Wiley Interdisciplinary Reviews. *Climate Change*, *6*(2), 151–169. DOI: 10.1002/wcc.316.

Bartrons, M., Catalan, J. & Penuelas, J. (2016). Spatial and temporal trends of organic pollutants in vegetation from remote and rural areas. *Scientific reports*, 6. 25446; https://doi.org/10.1038/srep25446

Basili, G. D., & Temple, S. A. (1999). Dickcissels and crop damage in Venezuela: defining the problem with ecological models. *Ecological applications, 9*(2), 732-739. https://doi.org/10.1890/1051-0761(1999)009%5B0732:DACDIV%5D2.0.CO;2

Bates, N., Astor, Y., Church, M., Currie, K., Dore, J., Gonaález-Dávila, M., Lorenzoni, L., Muller-Karger, F., Olafsson, J., & Santa-Casiano, M. (2014). A time-series view of changing ocean chemistry due to ocean uptake of anthropogenic CO_2 and ocean acidification. *Oceanography*, *27*(1), 126–141. https://doi.org/10.5670/oceanog.2014.16

Battaglin, W. A., Meyer, M. T., Kuivila, K. M. & Dietze, J. E. (2014). Glyphosate and its degradation product AMPA occur frequently and widely in US soils, surface water, groundwater, and precipitation. *JAWRA Journal of the American Water Resources Association, 50*(2), 275-290. https://doi.org/10.1111/jawr.12159

Batzer, D. P., & Baldwin, A. H. (2012). Wetland Habitats of North America: Ecology and Conservation Concerns. *Wetland Habitats of North America: Ecology and Conservation Issues*, 405. https://doi.org/10.1899/32.1.BR.359.1

Baud, M., Castro, F. de, & Hogenboom, B. (2011). Environmental governance in Latin America: Towards an integrative research agenda. *European Review of Latin American and Caribbean Studies*, *90*, 79–88. http://doi.org/10.18352/erlacs.9749

Bauhardt, C. (2014). Solutions to the crisis? The green new deal, degrowth, and the solidarity economy: Alternatives to the capitalist growth economy from an ecofeminist economics perspective. *Ecological Economics*, *102*, 60–68. http://doi.org/10.1016/j.ecolecon.2014.03.015

Beck, M. W., Brumbaugh, R. D., Airoldi, L., Carranza, A., Coen, L. D., Crawford, C., Defeo, O., Edgar, G. J., Hancock, B., Kay, M. C., Lenihan, H. S., Luckenbach, M. W., Toropova, C. L., Zhang, G. & Guo, X. (2011). Oyster reefs at risk and recommendations for conservation, restoration, and management. *BioScience, 61*(2), 107–116. https://doi.org/10.1525/bio.2011.61.2.5

Bednaršek, N., Harvey, C. J., Kaplan, I. C., Feely, R. A., & Možina, J. (2016). Pteropods on the edge: Cumulative effects of ocean acidification, warming, and deoxygenation. *Progress in Oceanography*, 145, 1–24. https://doi.org/10.1016/j.pocean.2016.04.002

Beger, M., Grantham, H. S., Pressey, R. L., Wilson, K. A., Peterson, E. L., Dorfman, D., Mumby, P. J., Lourival, R., Brumbaugh, D. R., & Possingham, H. P. (2010). Conservation planning for connectivity across marine, freshwater, and terrestrial realms. *Biological Conservation, 143*(3), 565–575. https://doi.org/10.1016/j.biocon.2009.11.006

Begossi, A. (2010. Small-scale fisheries in Latin America. *Management models and challenges*, *9*(2), 7–31.

Begotti, R., & Landesmann, L. (2008). Predação de ninhos por um grupo híbrido de sagüis (*Callithrix Jacchus*/Penicillata) Introduzidos em área urbana: implicações para a estrutura da comunidade. *Neotropical, 15*(1), 28–29. https://doi.org/10.1896/044.015.0107

Bellard, C., Leclerc, C., & Courchamp, F. (2014). Impact of sea level rise on the 10 insular biodiversity hotspots. *Global Ecology and Biogeography*, *23*(2), 203-212. https://doi.org/10.1111/geb.12093

Bellard, C., Thuiller, W., Leroy, B., Genovesi, P., Bakkenes, M., & Courchamp, F. (2013). Will climate change promote future invasions? *Global Change Biology*, *19*(12), 3740–3748. https://doi.org/10.1111/gcb.12344

Belnap J., & Büdel B. (2016) Biological soil crusts as soil stabilizers. In Weber, B., B. Büdel, & J. Belnap. (Eds.) Biological soil crusts: an organizing principle in drylands. *Ecological Studies (Analysis and Synthesis), 226*, 305–320. Springer, Cham. http://link.springer.com/10.1007/978-3-319-30214-0_16 Accessed 5 Jun 2016.

Benayas, J. M. R., 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://www.ncbi.nlm.nih.gov/pubmed/19644076

Benson, A., Raikow, D., Larson, J., Fusaro, A., & Bogdanoff, A. (2017). *Dreissena polymorpha* (Pallas, 1771) *USGS nonindigenous aquatic species database*. Gainesville, Florida.

Berg, Jr C. J., & Olsen, D. A. (1989). Conservation and management of queen conch (*Strombus gigas*) fisheries in the Caribbean. In J.F. Caddy (Ed.), *Marine invertebrate fisheries: Their assessment and management* (pp. 421-442). New York: John Wiley

Bergan, T., Gallardo, L., & Rodhe, H. (1999). Mercury in the global troposphere: A three-dimensional model study. *Atmospheric Environment*, *33*(10), 1575–1585. https://doi.org/10.1016/S1352-2310(98)00370-7

Berkes, F. (2007). Community-based conservation in a globalized world. *Proceedings of the National Academy of Sciences*, *104*(39), 15188-15193. https://doi.org/10.1073/pnas.0702098104

Betancur-R., R., Hines, A., Acero P., A., Ortí, G., Wilbur, A. E., & Freshwater, D. W. (2011). Reconstructing the lionfish invasion: insights into Greater Caribbean biogeography. *Journal of Biogeography*, *38*(7), 1281–1293. https://doi.org/10.1111/j.1365-2699.2011.02496.x

Beyer, W.N., Franson, J.C., French, J.B., May, T., Rattner, B.A., Shearn-Bochsler, V.I., Warner, S.E., Weber, J., & Mosby, D. (2013). Toxic exposure of songbirds to lead in the Southeast Missouri lead mining district. *Archives of environmental contamination and toxicology*, 65(3), 598-610. https://doi.org/10.1007/s00244-013-9923-3

Bhattacharya, A., Romani, M., & Stern, N. (2012). Infrastructure for development: meeting the challenge. In *Centre for Climate Change Economics and Policy*,15. Retrieved from http://www.cccep.ac.uk/Publications/Policy/docs/PP-infrastructure-for-development-meeting-the-challenge.pdf

Bigatti G., & Penchaszadeh P.E. (2008). Invertebrados del Mar Patagónico, diagnóstico de la problemática actual y potencial de su conservación y manejo. In *Estado de Conservación del Mar Patagónico y áreas de influencia*. (pp. 105–133). Argentina: Edición del Foro. Retrieved from http://www.marpatagonico.org/libro/articulo.php?id=bigatti-penchaszdeh-invertebrados

Bilbao, B. A., Leal, A. V., & Méndez, C. L. (2010). Indigenous use of fire and forest loss in Canaima National Park, Venezuela. Assessment of and tools for alternative strategies of fire management in Pemón indigenous lands. *Human Ecology*, *38*(5), 663-673.

Bingham, H., Fitzsimons, J. A., Redford, K. H., Brent, A., Bezaury-creel, J., & Cumming, T. L. (2017). Privately protected areas: advances and challenges in guidance, policy and documentation. *Parks*, 23(1).

Bjerregaard, P., & Hansen, J. C. (2000). Organochlorines and heavy metals in pregnant women from the Disko Bay area in Greenland. *Science of The Total Environment*, *245*(1), 195-202.

Blackburn, T. M., Essl, F., Evans, T., Hulme, P. E., Jeschke, J. M., Kühn, I., Kumschick, S., Marková, Z., Mrugała, A., Nentwig, W., Pergl, J., Pyšek, P., Rabitsch, W., Ricciardi, A., Richardson, D. M., Sendek, A., Vilà, M., Wilson, J. R. U., Winter, M., Genovesi, P., & Bacher, S. (2014). A unified classification of alien species based on the magnitude of their environmental impacts. *PLoS Biology*, *12*(5), e1001850. https://doi.org/10.1371/journal.pbio.1001850

Blackwell, B.D., & Driscoll, C.T. (2015). Deposition of mercury in forests along a montane elevation gradient. *Environmental Science & Technology*, *49*(9), 5363-5370. https://doi.org/10.1021/es505928w

Blaser, J., & Zabel, A. (2015). Forest crime in the tropics. In Pacel, L., & M. Köhl. *Tropical forestry handbook*. Berlin, Heidelberg: Springer.

Bobbink, R., Hicks, K., Galloway, J., Spranger, T., Alkemade, R., Ashmore, M., Bustamante, M., Cinderby, S., Davidson, E., Dentener, F., & Emmett, B. (2010). Global assessment of nitrogen deposition effects on terrestrial plant diversity: a synthesis. *Ecological applications*, *20*(1), 30-59. https://doi.org/10.1890/08-1140.1

Bohlen, P. P. J., Scheu, S., Hale, C. M., McLean, M. A., Migge, S., Groffman, P. M., & Parkinson, D. (2004). Non-native invasive earthworms as agents of change in northern temperate forests. *Frontiers in Ecology and the Environment, 2*(8), 427–435. https://doi.org/10.1890/1540-9295(2004)002%5B0427:NIEAAO%5D2.0.CO;2

Boisier, J. P., Ciais, P., Ducharne, A., & Guimberteau, M. (2015). Projected strengthening of Amazonian dry season by constrained climate model simulations. *Nature Climate Change* *5*(7), 656–660. http://www.scopus.com/inward/record.url?eid=2-s2.0-84932110744&partnerID=tZOtx3y1

Boltovskoty, D., & N. Correa. (2014). Ecosystem impacts of the invasive bivalve *Limnoperna fortunei* (golden mussel) in South America. *Hydrobiologia*, *746*(1), 81-95. DOI 10.1007/s10750-014-1882-9.

Boltovskoy, D. (ed.). (2015). Larval Development of *Limnoperna fortunei*. In Boltovskoy D. (Ed.), *Limnoperna fortunei*: *The ecology, distribution and control of a swiftly spreading invasive fouling mussel*. (pp. 43–53). Cham: Springer International Publishing.

Boltovskoy, D. (Ed.). (2015). *Limnoperna fortunei*, Invading nature - Springer series in invasion ecology.

Bond, N. A., Cronin, M. F., Freeland, H., & Mantua, N. (2015). Causes and impacts of the 2014 warm anomaly in the NE Pacific. *Geophysical Research Letters*, *42*(9), 3414-3420.

Bond, W. J. (2016). Ancient grasslands at risk. *Science*, *351*(6269), (pp.120-122).

Booth, S., & Zeller, D. (2005). Mercury, food webs, and marine mammals: implications of diet and climate change for human health. *Environmental Health Perspectives*, *113*(5), 521-526. https://doi.org/10.1289/ehp.7603

Boretto, J. G., Pacher, N., Giunta, D., Gallucci, G. L., Alfie, V., & De Carli, P. (2014). Comparative clinical study of locking screws versus smooth locking pegs in volar plating of distal radius fractures. *Journal of Hand Surgery* (European Volume), *39*(7), 755-760. https://doi.org/10.1177/1753193413517806

Borrvall, C., & Ebenman, B. (2006). Early onset of secondary extinctions in ecological communities following the loss of top predators. *Ecology Letters, 9*(4), 435–442. https://doi.org/10.1111/j.1461-0248.2006.00893.x

Bowerman, W. W., Best, D. A., Grubb, T. G., Zimmerman, G. M., & Giesy, J. P. (1998). Trends of contaminants and effects in bald eagles of the Great Lakes Basin. *Environmental Monitoring and Assessment, 53*(1), 197–212. https://doi.org/10.1023/A:1006068330050

Bowerman, W. W., Giesy, J. R., Best, D. A., & Kramer, V. J. (1995). A review of factors affecting productivity of bald eagles in the great lakes region: implications for recovery. *Environmental health perspectives*, *103*(spl 4), 51–59.

Bramanti, L., Movilla, J., Guron, M., Calvo, E., Gori, A., Dominguez-Carrió, C., Grinyó, J., Lopez-Sanz, A., Martinez-Quintana, A., Pelejero, C., Ziveri, P., & Rossi, S. (2013). Detrimental effects of ocean acidification on the economically important Mediterranean red coral (*Corallium rubrum*). *Global Change Biology, 19*(6), 1897–1908. doi: 10.1111/gcb.12171.

Brancalion, P. H. S., Viani, R. A. G., Rodrigues, R. R., & Gandolfi, S. (2012). Avaliação e monitoramento de áreas em processo de restauração. In S. V. Martins. *Restauração ecológica de ecossistemas degradados.* (pp.262-293). First edition. Viçosa: Editora UFV.

Brandeis, T. J., Helmer, E. H., Marcano-Vega, H., & Lugo, A. E. (2009). Climate shapes the novel plant communities that form after deforestation in Puerto Rico and the U.S. Virgin Islands. *Forest Ecology and Management*, *258*(7), 1704–1718. https://doi.org/10.1016/j.foreco.2009.07.030

Brashares, J. S., Arcese, P., Sam, M. K., Coppolillo, P. B., Sinclair, A. R. E., & Balmford, A. (2004). Bushmeat hunting, wildlife declines, and fish supply in West Africa. *Science*, *306*(5699), 1180–1183. https://doi.org/10.1126/science.1102425

Brauman, K. A., Richter, B. D., Postel, S., Malsy, M., & Flörke, M. (2016). Water depletion: An improved metric for incorporating seasonal and dry-year water scarcity into water risk assessments. *Elem Sci Anth*, *4,* art. 83, 1–12.

Braune, B. M., Outridge, P. M., Fisk, A., Muir, D. C. G., Helm, P. A., Hobbs, K., Hoekstra, P. F., Kuzyk, Z.A., Kwan, M., Letcher, R. J., & Lockhart, W.L. (2005). Persistent organic pollutants and mercury in marine biota of the Canadian Arctic: an overview of spatial and temporal trends. *Science of the Total Environment*, *351*, 4-56. https://doi.org/10.1016/j.scitotenv.2004.10.034

Brennan, L. A., & Kuvlesky Jr, W. P. (2005). North American grassland birds: an unfolding conservation crisis? *Journal of Wildlife Management*, *69*(1), 1-13.

Brett, F, & Côté, I. M. (2015). Do by-catch reduction devices in longline fisheries reduce capture of sharks and rays? A global meta-analysis. *Fish and Fisheries*, *16*(2), 300-309. https://doi.org/10.1111/faf.12055

Broad, S., Mulliken, T., & Roe, D. (2003). The nature and extent of legal and illegal trade in wildlife. In S. Oldfield (Ed.). *The trade in wildlife: regulation for conservation*. (pp. 3–22). London: Earthscan.

Brokaw, N. V. (1985). Gap-phase regeneration in a tropical forest. *Ecology*, *66*(3), 682-687.

Brook, B.W., Sodhi, N.S. & Bradshaw, C.J. (2008). Synergies among extinction drivers under global change. *Trends in ecology & evolution*, *23*(8), 453-460. https://doi.org/10.1016/j.tree.2008.03.011

Brookes, G. & Barfoot, P. (2007). Global impact of biotech crops: Socio-economic and environmental effects in the first ten years of commercial use. *The journal of agrobiotechnology management & economics*, 9(3), art 4.

Brooks & Smith (2001) Caribbean catastrophes. *Science*, *294*(5546), 1469–1470. https://doi.org/10.1126/science.1066927

Brooks, M. L., & Chambers, J. C. (2011). Resistance to invasion and resilience to fire in desert shrublands of North America. *Rangeland Ecology & Management*, *64*(5), 431–438. https://doi.org/10.2111/REM-D-09-00165.1

Brooks, M. L., Minnich, R. A., & Agee, J. K. (2006). Southeastern Deserts Bioregion. In Sugihara, N.G., J. W.Van Wagtendonk, K. E. Shaffer, J. Fites-Kaufman, & A.E. Thoe. *Fire in California's Ecosystems.* (pp. 391–414) First edition. University of California Press. http://www.jstor.org/stable/10.1525/j.ctt1pnb25.21

Brooks, T. M., Mittermeier, R. A., da Fonseca, G. A. B., Gerlach, J., Hoffmann, M., Lamoreux, J. F., Mittermeier, C. G., Pilgrim, J. D., & Rodrigues, A. S. L. (2006). Global Biodiversity Conservation Priorities. *Science*, *313*(5783), 58–61. https://doi.org/10.1126/science.1127609

Brose, U., & Hillebrand, H. (2016). Biodiversity and ecosystem functioning in dynamic landscapes. *Philosophical Transactions of the Royal Society of London. Series B, Biological Sciences*, *371*(1694): 20150267. http://doi.org/10.1098/rstb.2015.0267

Brown, D. G., Polsky, C. Bolstad, P., Brody, S. D., Hulse, D., Kroh, R. Loveland, T. R., & Thomson, A. (2014): Ch. 13: Land use and land cover change. Climate change impacts in the United States. In Melillo, J. M.,Richmond, T.T.C, & Yohe, G. W. (Eds.) *The Third National Climate Assessment*. U.S. Global Change Research Program, (pp. 318-332). http://doi.org/10.7930/J05Q4T1Q

Brown, K. A., Scatena, F. N., & Gurevitch, J. (2006). Effects of an invasive tree on community structure and diversity in a tropical forest in Puerto Rico. *Forest Ecology and Management, 226*(1–3), 145–152. https://doi.org/10.1016/j.foreco.2006.01.031

Bruckner, A. W. (2005). The importance of ornamental reef fish trade in the wider Caribbean. *International Journal of Tropical Biology and Conservation* 53, 127–138.

Brugnoli, E., Clemente, J., Riestra, G., Boccardi, L. & Borthagaray, A. (2006). Especies acuáticas exóticas en Uruguay: situación, problemática y gestión. In Menafra, R., L. Rodríguez, F. Scarabino, & D. Conde, (Eds.). Bases para la conservación y manejo de la costa uruguaya. *Vida Silvestre Uruguay*. (pp. 351-362).

Brugnoli, E., Giberto, D. A., Lanfranconi, A., Schiariti, A., Aguilera, F., Bremec, C. S., Barrero, G., & Muniz, P. (2013). El gastópodo invasor *Rapana venosa* (Valenciennes 1846) y sus posibles efectos en el sistema costero estuarial del Río de la Plata. In *Problemática de los ambientes costeros* (pp. 1–19).

Brugnoli, E., Giberto, D., Lanfranconi, A., Schiariti, A., Aguilera, F., Bremec, C. S., Barrero, G., & Muniz, P. (2014). El gasterópodo invasor *Rapana venosa* (Valenciennes 1846) y sus posibles efectos en el ecosistema costero estuarial del Río de la Plata. In Gosso, C (Compilador y Revisor). *Nuevas miradas a la problemática de los ambientes costeros. Sur de Brasil, Uruguay y Argentina*. (pp. 211-228). DIRAC, Facultad de Ciencias, Montevideo.

Bucher, E. H. (1992). The causes of extinction of the passenger pigeon. *Current Ornithology, 9,* 1-36. Springer US.

Bulte, E. H., Damania, R., & López, R. (2007). On the gains of committing to inefficiency: Corruption, deforestation and low land productivity in Latin America. *Journal of Environmental Economics and Management*, *54*(3), 277–295. http://doi.org/10.1016/j.jeem.2007.05.002

Bunn, A. G., & Goetz, S. J. (2006). Trends in satellite-observed circumpolar photosynthetic activity from 1982 to 2003 : The influence of seasonality, cover type , and vegetation density. *Earth Interactions*, 10(12), 1-19.

Burger, J., Gochfeld, M., Jeitner, C., Burke, S., Stamm, T., Snigaroff, R., Snigaroff, D., Patrick, R. and Weston, J. (2007). Mercury levels and potential risk from subsistence foods from the Aleutians. *Science of the Total Environment*, 384(1), 93-1059.

Burke, L., Reytar, K., Spalding, M., & Perry, A. (2011). *Reefs at risk revisited*. Washington, D.C. http://www.wri.org

Burnett, A. W., Kirby, M. E., Mullins, H. T., & Patterson, W. P. (2003). Increasing Great Lake-effect snowfall during the twentieth century: A regional response to global warming? *Journal of Climate*, *16*(21), 3535–3542.

Bush, M. B., Silman, M. R., & Urrego, D. H. (2004). 48,000 years of climate and forest change in a biodiversity hot spot. *Science*, *303*(5659), 827-829.

Bush, S. R., Toonen, H., Oosterveer, P., & Mol, A. P. J. (2013). The "devil's triangle" of MSC certification: Balancing credibility, accessibility and continuous improvement. *Marine Policy, 37*(1), 288–293. http://doi.org/10.1016/j.marpol.2012.05.011

Bushaw-Newton, K. I., & Sellner, K. G. (1999). *Harmful algal blooms. NOAA´s state of the coast report*. Silver Spring. MD: National Oceanic and Atmospheric Administration.

Bustamante, M. M. C., Martinelli, L. A., Pérez, T., Rasse, R., Ometto, J. P. H. B., Siqueira P., F., Machado L., S. R., & Marquina, S. (2015). Nitrogen management challenges in major watersheds of South America. *Environmental Research Letters 10*(6), 065007. IOP Publishing. http://stacks.iop.org/1748-9326/10/i=6/a=065007?key=crossref.879056f1355a6f61ce40d257c1205aed

Bustamante, M. M. C., Nardoto, G. B., Pinto, A. S., Resende, J. C. F., Takahashi, F. S. C., & Vieira, L. C. G. (2012). Potential impacts of climate change on biogeochemical functioning of Cerrado ecosystems. *Brazilian Journal of Biology*, *72*(3), 655-671.

Butt, N., Beyer, H.L., Bennett, J.R., Biggs, D., Maggini, R., Mills, M., Renwick, A.R., Seabrook, L.M., & Possingham, H.P. (2013). Biodiversity risks from fossil fuel extraction. *Science*, 342(6157), 425-426.

Bykova, O., Laursen, A., Bostan, V., Bautista, J., & McCarthy, L. (2006). Do zebra mussels (*Dreissena polymorpha*) alter lake water chemistry in a way that favours Microcystis growth? *Science of the Total Environment*, *371*(1–3), 362–372. https://doi.org/10.1016/j.scitotenv.2006.08.022

Byrne, S., Miller, P., Waghiyi, V., Buck, C.L., von Hippel, F.A., & Carpenter, D.O. (2015). Persistent organochlorine pesticide exposure related to a formerly used defense site on St. Lawrence Island, Alaska: data from sentinel fish and human sera. *Journal of Toxicology and Environmental Health*, *Part A, 78*(15), 976-992.

Cabré, M. F., Solman, S., & Núñez, M. (2016). Regional climate change scenarios over southern South America for future climate (2080-2099) using the MM5 Model. Mean, interannual variability and uncertainties. *Atmósfera*, *29*(1), 35-60.

Cabrera, A.L., & Willink, A. (1980) *Biogeografía de América Latina*. Serie Biología, Monografía 13. OEA, Washington, DC.

Callaghan, T. V, Björn, L.O., Iii, F.S.C., Chernov, Y., Christensen, T.R., Huntley, B., Ims, R., Johansson, M., Riedlinger, D.J., Jonasson, S., Matveyeva, N., Oechel, W., Panikov, N., & Shaver, G. (2005). Arctic tundra and polar desert ecosystems. In *Arctic Climate Impact Assessment*. Cambridge University Press, Cambridge.

Calle, Z., & Murgueitio, E. (2015) Ganaderos aliados de la biodiversidad en el Magdalena Medio. *Carta Fedegan*, *149*, 80-85.

Calle, Z., Murgueitio, E., & Chará J. (2012) Integrating forestry, cattle-ranching and landscape restoration. *Unasylva*, *239*(63), 31-40.

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.

Cameron, R.D., Smith, W.T., White, R.G., & Griffith, D.B. (2005). Central Arctic caribou and petroleum development: Distributional, nutritional, and reproductive implications. *Arctic*, 58(1), 1-9.

Campuzano, F. J., Mateus, M. D., Leitão, P. C., Leitão, P. C., Marín, V. H., Delgado, L. E., Tironi, A., Pierini, J. O., Sampaio, A. F. P., Almeida, P., & Neves, R. J. (2013). Integrated coastal zone management in South America: A look at three contrasting systems. *Ocean and Coastal Management*, *72*, 22–35. https://doi.org/10.1016/j.ocecoaman.2011.08.002

Camus, P., Castro, S., & Jaksic, F. (2008). El conejo europeo en chile: historia de una invasión biológica. *Historia (Santiago), 41*(2) 305–339.

Canning-Clode, J. (Ed.) (2015). *Biological invasions in changing ecosystems vectors, ecological impacts, management and predictions*. Walter de Gruyter GmbH & Co KG.

Carabias, J., Sarukhán, J., de la Maza, J., & Galindo, C. (2010). *Patrimonio natural de México. Cien casos de éxito*. *Comisión Nacional para el Conocimiento y Uso de la Biodiversidad*. México.

Cardozo, M. (2011). Economic displacement and local attitude towards protected area establishment in the Peruvian Amazon. *Geoforum*, *42*(5), 603–614. http://doi.org/10.1016/j.geoforum.2011.04.008

Carlson, D.L., Vault, D.S.D., & Swackhamer, D.L. (2010). On the rate of decline of persistent organic contaminants in lake trout (*Salvelinus namaycush*) from the Great Lakes, 1970– 2003. *Environmental science & technology*, 44(6),2004-2010.

Carothers, C. (2015). Fisheries privatization, social transitions, and well-being in Kodiak, Alaska. *Marine Policy*, *61*, 313–322. http://doi.org/10.1016/j.marpol.2014.11.019

Carpenter, K. E., Abrar, M., Aeby, G., Aronson, R. B., Banks, S., Bruckner, A., Chiriboga, A., Cortés, J., Delbeek, J. C., DeVantier, L., Edgar, G. J., Edwards, A. J., Fenner, D., Guzmán, H. M., Hoeksema, B. W., Hodgson, G., Johan, O., Licuanan, W. Y., Livingstone, S. R., Lovell, E. R., Moore, J. A., Obura, D. O., Ochavillo, D., Polidoro, B. A., Precht, W. F., Quibilan, M. C., Reboton, C., Richards, Z. T., Rogers, A. D., Sanciangco, J., Sheppard, A., Sheppard, C., Smith, J., Stuart, S., Turak, E., Veron, J. E. N., Wallace, C., Weil, E., & Wood, E. (2008). One-third of reef-building corals face elevated extinction risk from climate change and local impacts. *Science*, *321*(5888), 560–563.

Carpenter, S. R., Bennett, E. M., & Peterson, G. D. (2006). Scenarios for ecosystem services: An overview. *Ecology and Society*, *11*(1), art 29.

Carpenter, S. R., Stanley, E. H., & Vander Zanden, M. J. (2011). State of the world's freshwater ecosystems: physical, chemical, and biological changes. *Annual review of Environment and Resources*, *36*, 75-99.

Carranza, A., de Mello, C., Ligrone, A., González, S., Píriz, P., & Scarabino, F. (2010). Observations on the invading gastropod *Rapana venosa* in Punta del Este, Maldonado Bay, Uruguay. *Biological Invasions*, *12*(5), 995-998.

Carranza, A., Scarabino, F., & Ortega, L. (2007). Distribution of large benthic gastropods in the Uruguayan continental shelf and Río de la Plata estuary. *Journal of Coastal Research*, 24(sp1), 161-168.

Casas, G., Scrosati, R., & Luz Piriz, M. (2004). The invasive kelp *Undaria pinnatifida* (Phaeophyceae, Laminariales) reduces native seaweed diversity in Nuevo Gulf (Patagonia, Argentina). *Biological Invasions*, *6*(4), 411–416.

Castellanos-Gallindo, G. A., Cantera, J. R., Saint-Paul, U., & Ferrol-Schulte, D. (2014). Threats to mangrove social-ecological systems in the most luxuriant coastal forests of the Neotropics. *Biodiversersity and Conservation, 24*(3), 701-704.

Castello, L., Arantes, C. C., Mcgrath, D. G., Stewart, D. J., & De Sousa, F. S. (2015). Understanding fishing-induced extinctions in the Amazon. *Aquatic Conservation: Marine and Freshwater Ecosystems, 25*(5). https://doi.org/10.1002/aqc.2491

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. (2013a). The vulnerability of Amazon freshwater ecosystems. *Conservation Letters, 6*(4), 217–229.

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. (2013b). The vulnerability of Amazon freshwater ecosystems. *Conservation Letters*, *0* (2013) 1–13.

Castro, S. A., Figueroa, J. A., Muñoz-Schick, M., & Jaksic, F. M. (2005). Minimum residence time, biogeographical origin, and life cycle as determinants of the geographical extent of naturalized plants in continental Chile. *Diversity and Distributions*, *11*(3), 183–191. https://doi.org/10.1111/j.1366-9516.2005.00145.x

Cavole, L. M., Demko, A.M., Diner, R. E., Giddings, A., Koester, I., Pagniello, C. M. L. S., Paulsen, M. -L., Ramirez-Valdez, A., Schwenck, S. M., Yen, N.K., Zill, M. E., & Franks, P. J. S. (2016). Biological impacts of the 2013–2015 warm-water anomaly in the Northeast Pacific: Winners, losers, and the future. *Oceanography*, *29*(2), 273–285. http://dx.doi.org/10.5670/oceanog.2016.32

Cella-Ribeiro, A., Assakawa, L. F., Torrente-Vilara, G., Zuanon, J., Leite, R. G., Doria, C., & Duponchelle, F. (2015). Temporal and spatial distribution of young *Brachyplatystoma* spp. (Siluriformes: Pimelodidae) along the rapids stretch of the Madeira River (Brazil) before the construction of two hydroelectric dams. *Journal of Fish Biology 86*(4), 1429–1437.

CEPAL (2014). *Panorama Social de América Latina.* Comisión Económica para América Latina y el Caribe. Santiago, Chile.

CEPAL (2015). *Balance Preliminar de las Economías de América Latina y el Caribe*. Comisión Económica para América Latina y el Caribe. Santiago, Chile.

CEPAL (2016). *CEPAL apoya visibilidad estadística y participación de los pueblos indígenas en la Agenda 2030*. Santiago, Chile.

CEPAL (2016). *Balance preliminar de las economías de América Latina y el Caribe*. Santiago, Chile.

CEPAL (2017). *Balance preliminar de las economías de América Latina y el Caribe*. Santiago, Chile.

Cesar, H., Burke, L., & Pet-soede, L. (2003). *The economics of worldwide coral reef degradation*. Cesar environmental economics consulting, Arnhem, and WWF-Netherlands 14:23.

Chacon, C. (2005). Fostering conservation of key priority sites and rural development in Central America: the role of private protected areas. *Parks*, 39-47.

Chai, S. L., Tanner, E., & McLaren, K. (2009). High rates of forest clearance and fragmentation pre-and post-National Park establishment: The case of a Jamaican montane rainforest. *Biological Conservation*, *142*(11), 2484-2492.

Chakalall, B., & Cochrane, K. L. (1997). The queen conch fishery in the Caribbean - an approach to responsible fisheries management. In: *Proceedings of the 49th Gulf and Caribbean Fisheries Institute*. (pp. 531–554)

Chapin, F. S. I., Trainor, S. F., Cochran, P., Huntington, H., Markon, C., McCammon, M., McGuire, A. D., & Serreze. M. (2014). Ch. 22: Alaska. climate change impacts in the U.S. In The third national climate assessment. (pp. 514–536). http://nca2014.globalchange.gov/report/regions/alaska

Charlotte, D. R., Yolande, B. N., Cordonnier, S., & Claude, B. (2016). The invasive lionfish, *Pterois volitans*, used as a sentinel species to assess the organochlorine pollution by chlordecone in Guadeloupe (Lesser Antilles). *Marine pollution bulletin, 107*(1), 102-106.

Charvériat, C. (2000). *Natural disasters in Latin America and the Caribbean: An overview of risk*. Inter-Americas Development Bank. Working paper No. 434.

Cheung, W. W. L., Lam, V. W. Y., Sarmiento, J. L., Kearney, K., Watson, R., Zeller, D., & Pauly, D. (2010). Large-scale redistribution of maximum fisheries catch potential in the global ocean under climate change. *Global Change Biology, 16*(1), 24–35.

Chinea, J. D., & Agosto, R. (2007). Forests surrounding the Joyuda Lagoon, Puerto Rico: 67 years of change. *Caribbean Journal of Science*, *43*(1), 142-147.

Chinea, J. D., & Helmer, E. H. (2003). Diversity and composition of tropical secondary forests recovering from large-scale clearing: Results from the 1990 inventory in Puerto Rico. *Forest Ecology and Management*, *180*(1–3), 227–240. https://doi.org/10.1016/S0378-1127(02)00565-0

Christensen, J. H., & Christensen, O. B. (2007). A summary of the PRUDENCE model projections of changes in European climate by the end of this century. *Climatic change*, *81*(1), 7-30

Cibils, A. F., & Borrelli, P. R. (2005). Grasslands of Patagonia. In Suttie, J.M, S.G. Reynolds, & C. Batello (Eds.) *Grasslands of the World'*. (pp. 121-170).

Clark, J. R., Cole, M., Lindeque, P.K., Fileman, E., Blackford, J., Lewis, C., Lenton, T.M., & Galloway, T.S. (2016). Marine microplastic debris: a targeted plan for understanding and quantifying interactions with marine life. *Frontiers in Ecology and the Environment*, *14*(6), 317-324.

Clark, M. L., Aide, T. M., Grau, H. R., & Riner, G. (2010). A scalable approach to mapping annual land cover at 250 m using MODIS time series data: A case study in the Dry Chaco ecoregion of South America. *Remote Sensing of Environment*, *114*(11), 2816-2832.

Clarke, S. C., Harley, S. J., Hoyle, S. D., & Rice, J. S. (2013). Population Trends in Pacific Oceanic Sharks and the Utility of Regulations on Shark Finning. *Conservation Biology 27*(1), 197–209.

Clements, E. A., & Fernandes B. M. (2013). Land grabbing, agribusiness and the peasantry in Brazil and Mozambique. *Agrarian South: Journal of Political Economy 2*(1), 41-69.

Clements, W. H., Carlisle, D. M., Lazorchak, J. M., & Johnson, P. C. (2000). Heavy metals structure benthic communities in Colorado mountain streams. *Ecological Applications*, *10*(2), 626–638.

Clements, W. H., Cherry, D. S., & Cairns, J. (1988). Structural alterations in aquatic insect communities exposed to copper in laboratory streams. *Environmental Toxicology and Chemistry*, *7*(9), 715–722. http://dx.doi.org/10.1897/1552-8618(1988)7%5B715:SAIAIC%5D2.0.CO;2

Clements, W. H., Vieira, N. K. M., & Sonderegger, D. L. (2010b). Use of ecological thresholds to assess recovery in lotic ecosystems. *Journal of the North American Benthological Society*, *29*(3), 1017–1023.

Clements, W. H., Vieira, N. K. M., & Church, S. E. (2010a). Quantifying restoration success and recovery in a metal-polluted stream: A 17-year assessment of physicochemical and biological responses. *Journal of Applied Ecology*, *47*(4), 899–910.

Clements, W. H., Cadmus, P., & Brinkman. S. F. (2013). Responses of aquatic insects to Cu and Zn in stream microcosms: Understanding differences between single species tests and field responses. *Environmental Science and Technology*, *47*(13), 7506–7513.

Cline, T. J., Bennington, V, & Kitchell, J. F. (2013). Climate change expands the spatial extent and duration of preferred thermal habitat for Lake Superior fishes. *PLoS ONE*, *8*(4), e62279 https://doi.org/10.1371/journal.pone.0062279

Coat, S., Bocquené, G., & Godard, E. (2006). Contamination of some aquatic species with the organochlorine pesticide chlordecone in Martinique. *Aquatic Living Resources*, *19*(2), 181-187.

Coat, S., Monti, D., Legendre, P., Bouchon, C., Massat, F., & Lepoint, G. (2011). Organochlorine pollution in tropical rivers (Guadeloupe): role of ecological factors in food web bioaccumulation. *Environmental Pollution*, *159*(6), 1692-1701.

Coates, D. (1995). Inland capture fisheries and enhancement: Status, constraints, and prospects for food security. Paper presented at the Government of Japan/FAO International Conference on Sustainable Contribution of Fisheries to Food Security, Kyoto, Japan 4–9 December 19.

Cóbar-Carranza, A. J., García, R. A., Pauchard, A., & Peña, E. (2014). Effect of Pinus contorta invasion on forest fuel properties and its potential implications on the fire regime of Araucaria araucana and Nothofagus antarctica forests. *Biological Invasions*, *16*(11), 2273–2291. https://doi.org/10.1007/s10530-014-0663-8

Colding, J., & Folke, C. (2001). Social taboos:"invisible" systems of local resource management and biological conservation. *Ecological applications*, *11*(2), 584-600.

Cole, D. C., Kearney, J., Sanin, L. H., Leblanc, A., & Weber, J. P. (2004). Blood mercury levels among Ontario anglers and sport-fish eaters. *Environmental Research, 95*(3), 305-314.

Collen, B., Whitton, F., Dyer, E. E., Baillie, J. E. M., Cumberlidge, N., Darwall, W. R. T., Pollock, C., Richman, N. I., Soulsby, A. M., & Böhm, M. (2014). Global patterns of freshwater species diversity, threat and endemism. *Global Ecology and Biogeography*, *23*(1), 40–51. https://doi.org/10.1111/geb.12096

Collymore, J. (2011) Disaster management in the Caribbean: Perspectives on institutional capacity reform and development, *Environmental Hazards*, *10*(1), 6-22.

Compton, J. E., Harrison, J. A., Dennis, R. L., Greaver, T. L., Hill, B. H., Jordan, S. J., Walker, H., & Campbell, H. V. (2011). Ecosystem services altered b y human changes in the nitrogen cycle: a new perspective for US decision making. *Ecology Letters, 14*(8), 804–815.

CONABIO. (2014). Sistema de información sobre especies invasoras en México. Comisión Nacional para el Conocimiento y Uso de la Biodiversidad. Retrieved from http://www.biodiversidad.gob.mx/invasoras, accessed on February 23, 2016.

CONAGUA -Comisión Nacional del Agua. (2012). Atlas del agua en México

Concostrina-Zubiri, L., Pescador, D. S., Martínez, I., & Escudero, A. (2014). Climate and small scale factors determine functional diversity shifts of biological soil crusts in Iberian drylands. *Biodiversity and conservation*, *23*(7), 1757-1770.

Condit, R. (1998). Ecological implications of changes in drought patterns: Shifts in forest composition in Panama. *Climatic change 39*(2-3), 413–427.

Contreras, L. C. (1986). Bioenergetics and distribution of fossorial Spalacopus cyanus (Rodentia): thermal stress, or cost of burrowing. *Physiological Zoology*, *59*(1), 20-28.

Cook, B. I., & Seager, R. (2013). The response of the North American Monsoon to increased greenhouse gas forcing. *Journal of Geophysical Research: Atmospheres 118*(4), 1690-1699.

Cook, E. R., Woodhouse, C. A., Eakin, C. M., Meko, D. M., & Stahle, D. W. (2004). Long-term aridity changes in the western United States. *Science*, 306(5698), 1015-1018.

Cooke, S. J., & Cowx, I. G. (2004). *Review of the state of the world fishery resources: inland fisheries*.

Cooper, T. F., De'ath, G., Fabricius, K. E., & Lough, J. M. (2008). Declining coral calcification in massive Porites in two nearshore regions of the northern Great Barrier Reef. *Global Change Biology*, *14*(3), 529–538.

Cooper, O.R., Parrish, D.D., Stohl, A., Trainer, M., Nédélec, P., Thouret, V., Cammas, J.P., Oltmans, S.J., Johnson, B.J., Tarasick, D., Leblanc, T., McDermid, I.S., Jaffe, D., Gao, R., Stith, J., Ryerson, T., Aikin, K., Campos, T., Weinheimer, A., & Avery, M.A. (2010). Increasing springtime ozone mixing ratios in the free troposphere over western North America. *Nature*, *463*(7279), 344-348. http://doi.org/10.1038/nature08708

Cordy, P., Veiga, M. M., Salih, I., Al-Saadi, S., Console, S., Garcia, O., Mesa, L. A., Velásquez-López, P. C., & Roeser, M. (2011). Mercury contamination from artisanal gold mining in Antioquia, Colombia: The world's highest per capita mercury pollution. *Science of the Total Environment, 410,* 154–160. http://dx.doi.org/10.1016/j.scitotenv.2011.09.006.

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

Coscieme, L., Pulselli, F. M., Niccolucci, V., Patrizi, N., & Sutton, P. C. (2016). Accounting for "land-grabbing" from a biocapacity viewpoint. *Science of the Total Environment*, *539*, 551–559. http://doi.org/10.1016/j.scitotenv.2015.09.021

Costa, B. G. B, Soares, T. M., Torres, R. F., & Lacerda, L. D. (2013). Mercury distribution in a mangrove tidal creek affected by intensive shrimp farming. *Bulletin of Environmental Contamination and Toxicology*, *90*(5), 537-541.

Costello, C., Gaines, S. D., & Lynham, J. (2008). Can catch shares prevent fisheries collapse? *Science, 321*(5896), 1678-1681.

Côté, I. M., Darling, E. S., & Brown, C. J. (2016). Interactions among ecosystem stressors and their importance in conservation. *Proceedings of the Royal Society of London. Series B*, *Biological Sciences*, *283*(1824), 20152592).

Côté, I.M, Gree, S.J., Morris, J.A. Jr., Akins, J.L., & Steinke, D. (2013). Diet richness of invasive Indo-Pacific lionfish revealed by DNA barcoding. *Marine Ecology Progress Series, 472*, 249–256.

Coupe, R. H., & Capel, P. D. (2016). Trends in pesticide use on soybean, corn and cotton since the introduction of major genetically modified crops in the United States. *Pest Management Science, 72*(5), 1013–1022.

Coupe, R. H., Barlow, J. R. B., & Capel, P. D. (2012). Complexity of human and ecosystem interactions in an agricultural landscape. *Environmental Development,* 4, 88–104.

Courtney, T., Westfield, I., & Ries, J. B. (2013). CO 2 -induced ocean acidification impairs calcification in the tropical urchin *Echinometra viridis*. *Journal of Experimental Marine Biology and Ecology,* 440, 169–175.

Cox, J. R., Martin-R, M. H., Ibarra-F, F. A., Fourie, J. H., Rethman, N. F. G., & Wilcox, D. G. (1988). The influence of climate and soils on the distribution of four African grasses. *Journal of Range Management*, *41*, 127–139.https://doi.org/10.2307/3898948

Cox, O. N., & Clements, W. H. (2013). An integrated assessment of polycyclic aromatic hydrocarbons (PAHs) and benthic macroinvertebrate communities in Isle Royale National Park. *Journal of Great Lakes Research*, *39*(1), 74-82.

Crabbé, A., & Leroy, P. (2008). *The handbook of environmental policy evaluation*. London: Earthscan.

Crema, L. C., Biudes, J. F. V., & Camargo, A. F. M. (2011). Effect of Urucu oil (Brazilian Amazon) on the biomass of the aquatic macrophyte *Eichhornia crassipes* (Mart.) Solms (Pontederiaceae). *Acta Limnologica Brasiliensia, 23*(4), 406–411.

Cross, F.A., Evans, D.W., & Barber, R.T. (2015). Decadal declines of mercury in adult bluefish (1972–2011) from the mid-Atlantic coast of the USA. *Environmental science & technology*, *49*(15), 9064-9072.

Cubbage, F., Diaz, D., Yapura, P., & Dube, F. (2010). Impacts of forest management certification in Argentina and Chile. *Forest Policy and Economics*, *12*(7), 497–504. http://doi.org/10.1016/j.forpol.2010.06.004

Cunha, D. D. A., & Ferreira, L. V. (2012). Impacts of the Belo Monte hydroelectric dam construction on pioneer vegetation formations along the Xingu River, Pará State, Brazil. *Brazilian Journal of Botany*, 35(2), 159-167.

Cusack, D.F., Lee, J.K., McCleery, T.L., & LeCroy, C.S. (2015). Exotic grasses and nitrate enrichment alter soil carbon cycling along an urban–rural tropical forest gradient. *Global change biology*, *21*(12), 4481-4496.

Cyr, D., Gauthier, S., Bergeron, Y., & Carcaillet, C. (2009). Forest management is driving the eastern North American boreal forest outside its natural range of variability. *Frontiers in Ecology and the Environment*, *7(*10), 519-524.

D'Antonio, C. M., & Vitousek, P. M. (1992). Biological invasions by exotic grasses, the grass/fire cycle, and global change. *Annual Review of Ecology and Systematics*, *23*(1), 63–87. https://doi.org/10.1146/annurev.es.23.110192.000431

Dai, A., Qian, T., Trenberth, K. E., & Milliman, J. D. (2009). Changes in continental freshwater discharge from 1948 to 2004. *Journal of Climate*, *22*(10), 2773–2792.

Dai, Z., Johnson, K. D., Birdsey, R. A., Hernandez-Stefanoni, J. L., & Dupuy, J. M. (2015). Assessing the effect of climate change on carbon sequestration in a Mexican dry forest in the Yucatan Peninsula. *Ecological Complexity, 24*, 46–56. Elsevier B.V. http://linkinghub.elsevier.com/retrieve/pii/S1476945X15001002

Dai, A., Trenberth, K. E., & Qian, T. (2004). A global dataset of palmer drought severity index for 1870–2002: relationship with soil moisture and effects of surface warming. *Journal of Hydrometeorology*, *5*(6), 1117–1130.

Daly, G.L., Lei, Y.D., Teixeira, C., Muir, D.C., Castillo, L.E., Jantunen, L.M., & Wania, F. (2007). Organochlorine pesticides in the soils and atmosphere of Costa Rica. *Environmental science & technology*, *41*(4), 1124-1130.

Dallaire, R., Muckle, G., Rouget, F., Kadhel, P., Bataille, H., Guldner, L., Seurin, S., Chajès, V., Monfort, C, Boucher, O., Pierre Thomé, J., Jacobson, S. W., Multigner, L., & Cordier S. (2012). Cognitive, visual, and motor development of 7-month-old Guadeloupean infants exposed to chlordecone. *Environmental Research*, *118*, 79–85.

Darrigran, G. & Pastorino, G. (1995). The recent introduction of Asiatic bivalve, *Limnoperna fortunei* (Mytilidae) in to South America. *The Veliger, 38*(2), 183-187.

Darrigran, G. (2002). Potential impact of filter-feeding invaders on temperate inland freshwater environments. *Biological Invasions, 4*(1-2),145-156.

Davidson, L.N., 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.

Davis, M., Douglas, C., Calcote, R., Cole, K. L., Green Winkler, M., & Flakne, R. (2000). Holocene climate in the Western Great Lakes national park and lakeshores: implications for future climate change. *Conservation Biology*, *14*(4), 968–983.

Day. O. (2009). *The impacts of climate change on biodiversity in Caribbean islands: what we know, what we need to know, and building capacity for effective adaptation*. CANARI Technical Report, 386. http://www.canari.org/macarthurclimatechange.html

De Castro, F., Hagenboom, B., & Baud, M. (2016). *Gobernanza ambiental en América Latina*. Buenos Aires: CLASCO.

De Gouw, J. A., Parrish, D. D., Frost, G. J., & Trainer, M. (2014). Reduced emissions of CO_2, NOx, and SO2 from US power plants owing to switch from coal to natural gas with combined cycle technology. *Earth's Future*, *2*(2), 75-82.

De Groot, R. S., Wilson, M. A., & Boumans, R. M. (2002). A typology for the classification, description and valuation of ecosystem functions, goods and services. *Ecological economics*, *41*(3), 393-408.

de la Barrera, E., & Castellanos, A. (2007). High temperature effects on gas exchange for the invasive buffel grass (*Pennisetum ciliare* [L.] Link). *Weed Biology and Management*, 7(2), 128–131. https://doi.org/10.1111/j.1445-6664.2007.00248.x

De la Casa, A. C., & Ovando, G. G. (2014). Climate change and its impact on agricultural potential in the central region of Argentina between 1941 and 2010. *Agricultural and forest meteorology*, *195*, 1-11.

de Moura, Y. M., Hilker, T., Lyapustin, A. I., Galvão, L. S., dos Santos, J. R., Anderson, L. O., de Sousa, C. H. R., & Arai, E. (2015). Seasonality and drought effects of Amazonian forests observed from multi-angle satellite data. *Remote Sensing of Environment, 171*, 278–290.

de Oliveria, M. D., Campos, M. C. S., Paolucci, E. M., Mansur, M. C., Hamilton, S. K. (2015). Colonization and spread of *Limnoperna fortunei* in South America. In Boltovskoy, D. (Ed.). *Limnoperna fortunei*, invading nature. Springer series in invasion ecology 10 (pp 333-355). Springer International Publishing Switzerland. DOI https://doi.org/10.1007/978-3-319-13494-9_19

Defenders of Wildlife. (2016). *Trends in wildlife imports from latin america deined entry into the United States. Fact Sheet*, Washington DC.

Deines, J. M., Williams, D., Hamlin, Q., & McLachlan, J. S. (2016). Changes in forest composition in Ohio between Euro-American settlement and the present. *The American Midland Naturalist*, *176*(2), 247-271.

Delgadillo M. J., Aguilar O. T., & Rodríguez V. D. (1999). Los aspectos económicos y sociales de El Niño. In: Magaña Rueda, V. O. (Ed.) *Los impactos de El Niño en México. Dirección de protección civil*. (181-210 pp). Secretaria de Gobernación, México.

Delmas, M. A., & Young O.R. (Eds). (2009). Governance for the environment. New York: Cambridge University Press.

Delphin, S., Escobedo, F. J., Abd-Elrahman, A., & Cropper, W. (2013). Mapping potential carbon and timber losses from hurricanes using a decision tree and ecosystem services driver model. *Journal of environmental management*, *129*, 599-607.

Denevan, W. M. (2004). Semi-intensive pre-European cultivation and the origins of anthropogenic dark earths in Amazonia. In: Amazonian dark earths: explorations in space and time (pp. 135-143). Springer Berlin, Heidelberg.

Denslow, J. S., & DeWalt, S. J. (2008). Exotic plant invasions in tropical forests: patterns and hypotheses. In Carson, W.P & S.A. Schnitzer (Eds). *Tropical forest community ecology*. (pp. 409–426). University of Chicago.

Depew, D. C., Burgess, N. M., & Campbell, L. M. (2013). Spatial patterns of methylmercury risks to common loons and piscivorous fish in Canada. *Environmental Science and Technology*, *47*(22), 13093–13103.

Després, C., Beuter, A., Richer, F., Poitras, K., Veilleux, A., Ayotte, P., Dewailly, E., Saint-Amour, D., & Muckle, G. (2005). Neuromotor functions in Inuit preschool children exposed to Pb, PCBs, and Hg. *Neurotoxicology and Teratology, 27*(2), 245-257.

Di Minin, E., & Toivonen, T. (2015). Global protected area expansion: creating more than paper parks. *BioScience*, *65*(7), 637–638. http://doi.org/10.1093/biosci/biv064

Diaz, R.J., & Rosenberg, R. (2008). Spreading dead zones and consequences for marine ecosystems. *Science, 321*(5891), 926-929.

Didham, R. K., Tylianakis, J. M., Gemmell, N. J., Rand, T. A., & Ewers, R. M. (2007). Interactive effects of habitat modification and species invasion on native species decline. *Trends in Ecology and Evolution*, 22(9), (pp. 489–496).

Dietz, R., Outridge, P. M., & Hobson, K. A. (2009). Anthropogenic contributions to mercury levels in present-day Arctic animals—a review. *Science of the Total Environment*. *407*(24), 6120-6131.

Dietz, R., Riget, F.F., Boertmann, D., Sonne, C., Olsen, M.T., Fjeldså, J., Falk, K., Kirkegaard, M., Egevang, C., Asmund, G., &Wille, F. (2006). Time trends of mercury in feathers of West Greenland birds of prey during 1851– 2003. *Environmental science & technology*, 40(19), 5911-5916.

Dietz, R., Sonne, C., Basu, N., Braune, B., O'Hara, T., Letcher, R.J., Scheuhammer, T., Andersen, M., Andreasen, C., Andriashek, D., & Asmund, G. (2013). What are the toxicological effects of mercury in Arctic biota? *Science of the Total Environment*, *443*, 775-790.

Dodds, W.K., Clements, W.H., Gido, K., Hilderbrand, R.H., King, R.S. (2010). Thresholds, breakpoints, and nonlinearity in freshwaters as related to management. *Journal of the North American Benthological Society*, 29(3), 988-997.

dos Santos, C. A. C. (2013). Recent changes in temperature and precipitation extremes in an ecological reserve in Federal District, Brazil. *Revista Brasileira de Meteorologia*, *29*(1).

dos Santos Sales, I., Ruiz-Miranda, C. R., & de Paula Santos, C. (2010). Helminths found in marmosets (Callithrix penicillata and Callithrix jacchus) introduced to the region of occurrence of golden lion tamarins (Leontopithecus rosalia) in Brazil. *Veterinary parasitology*, *171*(1-2), 123-129.

Donat, M. G., Alexander, L. V., Yang, H., Durre, I., Vose, R., Dunn, R. J. H., Willett, K. M., Aguilar, E., Brunet, M., Caesar, J. , Hewitson, B., Jack, C. , Klein Tank, A. M. G., Kruger, A. C., Marengo, J., Peterson, T. C., Renom, M., Oria Rojas, C., Rusticucci, M., Salinger, J., Elrayah, A. S., Sekele, S. S., Srivastava, A. K., Trewin, B., Villarroel, C., Vincent, L. A., Zhai, P., Zhang, X., & Kitching, S. (2013). Updated analyses of temperature and precipitation extreme indices since the beginning of the twentieth century: The HadEX2 dataset. *Journal of Geophysical Research Atmospheres*, *118*(5), 2098–2118.

Dorcas, M. E., Willson, J. D., Reed, R. N., Snow, R. W., Rochford, M. R., Miller, M. A., Meshaka, W. E., Andreadis, P. T., Mazzotti, F. J., Romagosa, C. M., & Hart, K. M. (2012). Severe mammal declines coincide with proliferation of invasive Burmese pythons in Everglades National Park. *Proceedings of the National Academy of Sciences*, *109*(7), 2418–2422. https://doi.org/10.1073/pnas.1115226109

Dornelas, M. (2010). Disturbance and change in biodiversity. *Philosophical Transactions of the Royal Society of London B: Biological Sciences, 365*(1558), 3719-3727.

Drew, J. A. (2005). Use of traditional ecological knowledge in marine conservation. *Conservation Biology*, *19*(4), 1286-1293.

Driscoll, C. T., Han, Y. J., Chen, C. Y., Evers, D. C., Lambert, K. F., Holsen, T. M., Kamman, N. C., & Munson, R. K. (2007). Mercury contamination in forest and freshwater ecosystems in the northeastern United States. *Bioscience*, *57*(1), 17–28.

Drummond, M. A., & Loveland, T. R. (2010). Land-use pressure and a transition to forest-cover loss in the eastern United States. *BioScience*, *60*(4), 286-29).

Dudgeon, D., Arthington, A. H., Gessner, M. O., Kawabata, Z., Knowler, D. J., Lévêque, C., Naiman, R. J., Prieur-Richard, A. H., Soto, D., Stiassny, M. L. J., & Sullivan, C. A. (2006). Freshwater biodiversity: importance, threats, status and conservation challenges. *Biological Reviews of the Cambridge Philosophical Society, 81*(2), 163–82. http://www.ncbi.nlm.nih.gov/pubmed/16336747

Dudka, S., & Adriano, D.C. (1997). Environmental Impacts of Metal Ore Mining and Processing: A Review. *Journal of Environmental Quality*, *26*(3), 590-602. https://doi.org/10.2134/jeq1997.00472425002600030003x

Dudley, N., Jeanrenaud, J.P., & Sullivan, F. (2014). *Bad harvest: The timber trade and the degradation of global forests*. Routledge.

Duke, N. C. (2016). Oil spill impacts on mangroves: recommendations for operational planning and action based on a global review. *Marine Pollution Bulletin, 109*(2), 700–715.

Duke, N. C., Meynecke, J. 0., Dittmann, A. M., Ellison, A. M., Anger, K., Berger, U., Cannicci, S., Diele, K., Ewel, K. C., Field, C. D., Koedam, N., Lee, S.Y., Marchand, C., Nordhaus, I., & Dahdouh-Guebas, F. (2007). A world without mangroves? *Science*, *317*(5834), 41-42.

Dunbar, J.B., Britsch, L.D., & Kemp, III, E.B. (1992). *Land Loss Rates: Report 3, Louisiana Coastal Plain*. Technical Report GL-90-2, U.S. Army Corps of Engineers District, New Orleans, LA.

Durnford, D., Dastoor, A., Figueras-Nieto, D., & Ryjkov, A. (2010). Long range transport of mercury to the Arctic and across Canada. *Atmospheric Chemistry and Physics*, *10*(13), 6063-6086.

Dutfield, G. (2003). *Intellectual Property Rights and the Life Science Industries: A Twentieth Century History*. Burlington, Vermont: Ashgate.

Dykstra, C. R., Meyer, M. W., Warnke, D. K., Karasov, W. H., Andersen, D. E., Bowerman, W. W., & Giesy, J. P. (1998). Low reproductive rates of Lake Superior Bald Eagles: low food delivery rates or environmental contaminants? *Journal of Great Lakes Research, 24*(1), 32–44.

Dykstra, C. R., Meyer, M. W., Stromborg, K. L., Warnke, D. K., Bowerman, IV W. W., & Best, D. A. (2001). Association of low reproductive rates and high contaminant levels in bald eagles on Green Bay, Lake Michigan. *Journal of Great Lakes Research, 27*(2), 239–251.

Dykstra, C. R., Meyer, M. W., Rasmussen, P. W., & Warnke, D. K.

(2005). Contaminant concentrations and reproductive rate of Lake Superior bald eagles, 1989-2001. *Journal of Great Lakes Research, 31*(2), 227–235.

Eakin, C. M., Nim C. J., Brainard, R. E., Aubrecht, C. E., Gledhill, D. K., Muller-Karger, F., Mumby, P. J., Skirving, W. J., Strong, A. E., Wang, M., Weeks, S., W., Wentz, F., & Ziskin, D. (2010). Monitoring coral reefs from space. *Monitoring coral reefs from space. Oceanography, 23(4), 118-133*

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*, 12485. https://doi.org/10.1038/ncomms12485

Ebisu, K., & Bell, M.L. (2012). Airborne PM2. 5 chemical components and low birth weight in the northeastern and mid-Atlantic regions of the United States. *Environmental health perspectives*, *120*(12), 1746-1752.

Echeverría, C., Smith-Ramírez, C., Aronson, J., & Barrera-Cataño, J. I. (2015). Good news from Latin America and the Caribbean: national and international restoration networks are moving ahead. *Restoration Ecology*, *23*(1), 1-3.

Economic Commission for Latin America and the Caribbean (ECLAC). (2014). Compacts for equality towards a sustainable future. Thirty-fifth Session of ECLAC, Lima.

Elliott, J. E., Kirk, D. A., Elliott, K. H., Dorzinsky, J., Lee, S., Inzunza, E. R., Cheng, K. M., Scheuhammer, T., & Shaw, P. (2015). Mercury in forage fish from Mexico and Central America: implications for fish-eating birds. *Archives of environmental contamination and toxicology*, 69(4), 375-389.

Ellis, E. C., Goldewijk, K. K., Siebert, S., Lightman, D., & Ramankutty, N. (2010). Anthropogenic transformation of the biomes, 1700 to 2000. *Global Ecology and Biogeography*, *19*(5), 589–606.

Ellison, C., Bank, M. S., Clinton, B. D., Colburn, E. A., Elliott, K., Ford, C. R., & Foster, D. R. (2005). Loss of foundation species. Consequences for the structure and dynamics of forested ecosystems. *Fronteers in Ecology and the Environment, 3*(9). 479-486. https://doi.org/10.2307/3868635

Ellison, J. (2015). Vulnerability assessment of mangroves to climate change and sea-level rise impacts. *Wetlands Ecology and Management, 23*(2), 115-137.

Elser, J. J., Andersen, T., Baron, J. S., Bergström, A. K., Jansson, M., Kyle, M., Nydick, K. R., Steger, L., & Hessen, D. O. (2009). Shifts in lake N: P stoichiometry and nutrient limitation driven by atmospheric nitrogen deposition. *Science*, *326*(5954), 835-837.

Emery, S. M., & Rudgers, J. A. (2009). Evaluating dune restorations in the Great Lakes region. In The 94th ESA annual meeting. (pp. 73-168)

Engstrom, D., & Swain, E. (1997). Recent decline in atmospheric mercury deposition in the upper Midwest. *Environmental Science & Technology, 31*(4), 960–967.

Erickson, H. E., Helmer, E. H., Brandeis, T. J., & Lugo, A. E. (2014). Controls on fallen leaf chemistry and forest floor element masses in native and novel forests across a tropical island. *Ecosphere, 5*(4), 1-28.

Erisman, J.W., Sutton, M.A., Galloway, J., Klimont, Z., & Winiwarter, W. (2008). How a century of ammonia synthesis changed the world. *Nature Geoscience*, *1*(10), 636-639.

Essl, F., Mang, T., & Moser, D. (2011). Ancient and recent alien species in temperate forests: steady state and time lags. *Biological Invasions*, *14*(7), 1331–1342.

ETC Group (2008) Who owns nature? Corporate power and the final frontier in the commodification of life, Issue No. 100, November 2008, https://www.panna.org/sites/default/files/etc_WhoOwnsNature.pdf

Evers, D.C., Han, Y.J., Driscoll, C.T., Kamman, N.C., Goodale, M.W., Lambert, K.F., Holsen, T.M., Chen, C.Y., Clair, T.A., & Butler, T. (2007). Biological mercury hotspots in the northeastern United States and southeastern Canada. *AIBS Bulletin*, *57*(1), 29-43.

Evers, D.C., Savoy, L.J., DeSorbo, C.R., Yates, D.E., Hanson, W., Taylor, K.M., Siegel, L.S., Cooley, J.H., Bank, M.S., Major, A., & Munney, K. (2008). Adverse effects from environmental mercury loads on breeding common loons. *Ecotoxicology*, *17*(2), 69-81.

Fabricius, K. E. (2005). Effects of terrestrial runoff on the ecology of corals and coral reefs: Review and synthesis. *Marine Pollution Bulletin, 50*(2), 125–146.

Fabry, V. J., Seibel, B. A., Feely, R. A., & Orr, J. C. (2008). Impacts of ocean acidification on marine fauna and ecosystem processes. *ICES Journal of Marine Sciences, 65*(3), 414–432.

Fadini, P. S., & Jardim, W. F. (2001). Is the Negro River basin (Amazon) impacted by naturally occurring mercury? *Science of the Total Environment*, 275(1-3), 71–82.

Fajardo, L., González, V., Nassar, J. M., Lacabana, P., Portillo Q, C. A., Carrasquel, F., & Rodríguez, J. P. (2005). Tropical dry forests of Venezuela: Characterization and current conservation status. *Biotropica*, *37*(4), 531–546. https://doi.org/10.1111/j.1744-7429.2005.00071.x

FAO. (1996). *Enseñanzas de la revolución verde: hacia una nueva revolución verde. Documentos ténicos de referencia*. Rome: http://www.fao.org/docrep/003/w2612s/w2612s06.htm

FAO. (2011). Review of the state of world marine fishery resources. FAO Fisheries and Aquaculture Technical Paper.

FAO. (2011) Why invest in sustainable mountain development? Rome, Italy: FAO Publications.

FAO, GEF & TerrAfrica (2011) Transboundary agro-ecosystem management project for the Kagera river basin. Rome, Italy: FAO Publications.

FAO. (2013a) FAO Statistical Yearbook: World Food and Agriculture: FAO.

FAO. (2013b). FAO Success Stories on Climate Smart Agriculture. Rome, Italy: FAO Publications.

FAO Committee on Fisheries. (2014). Inland Fisheries: Issues, Developments and Needs.

FAO. (2016a). FAOSTAT Database. Food and Agriculture Organization of the United Nations. Cited 14 December 2016. http://faostat3.fao.org/home/E

FAO (2016b). FishStatJ - software for fishery statistical time series. Cited 14 December 2016. http://www.fao.org/fishery/

FAO (2016c). The State of World Fisheries and Aquaculture. Food and Agriculture Organization of the United Nations.

Farmer, C. J., Goodrich, L. J., Inzunza, E. R. & Smith, J. P. (2008). Conservation status of North America's birds of prey. State of North America's Birds of Prey, *Series in Ornithology*, *3*, 303-420.

Fasanella, C. C., Dias, A. C. F., Rigonato, J., Fiore, M. F., Soares Jr, F. L., Melo, I. S., Pizzirani-Kleiner, A. A., van Elsas, J. D., & Andreote, F. D.(2012). The selection exerted by oil contamination on mangrove fungal communities. *Water, Air, & Soil Pollution, 223*(7), 4233-4243.

Fearnside, P. M. (2005). Brazil's Samuel dam: Lessons for hydroelectric development policy and the environment in Amazonia. *Environmental Management, 35*(1), 1–19.

Fearnside, P. M. (2013). Carbon credit for hydroelectric dams as a source of greenhouse-gas emissions: The example of Brazil's Teles Pires Dam. *Mitigation and Adaptation Strategies for Global Change 18*(5), 691–699.

Fearnside, P. M. (2015). Amazon dams and waterways: Brazils Tapajós Basin plans. *Ambio*, *44*(5), 426–439.

Feely, R.A., Sabine, C.L., Byrne, R.H., Millero, F.J., Dickson, A.G., Wanninkhof, R., Murata, A., Miller, L.A., & Greeley, D. (2012). Decadal changes in the aragonite and calcite saturation state of the Pacific Ocean. *Global Biogeochemical Cycles*, *26*(3).

Fernández, R. J., & Reynolds, J. F. (2000). Potential growth and drought tolerance of eight desert grasses: lack of a trade-off? *Oecologia*, *123*(1), 90-98.

Ferraro, P. J., Hanauer, M. M., Miteva, D. A., Nelson, J. L., & Pattanayak, S. K. (2015). Estimating the impacts of conservation on ecosystem services and poverty by integrating modeling and evaluation. *Proceedings of the National Academy of Science of the United States of America*, *112*(24). http://doi.org/10.1073/pnas.1406487112

Ferreira, C. E. L., Luiz, O. J., Floeter, S. R., Lucena, M. B., Barbosa, M. C., Rocha, C. R., & Rocha, L. A. (2015). First record of invasive lionfish (*Pterois volitans*) for the Brazilian coast. *PLoS ONE*, *10*(4), 1–5.

Ferreira, L. V., Cunha, D. A., Chaves, P. P., Matos, D. C. L, & Parolin, P. (2013). Impacts of hydroelectric dams on alluvial riparian plant communities in eastern Brazilian Amazonian. *Anais da Academia Brasileira de Ciencias, 85*(3), 1013–1023.

Ferriter, A. (Ed.). (1997). Brazilian pepper management plan for Florida: a report from The Florida exotic pest plant council's Brazilian pepper task force. http://www.fleppc.org/Manage_Plans/schinus.pdf

Field, C.B., Barros, V.R., Dokken, D.J., Mach, K.J., Mastrandrea, M.D., Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B., Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., & White, L.L. (eds.). (2014). Summary for policymakers. In: Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. *Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. (pp. 1-32). Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press

Field, J. P., Breshears, D. D., & Whicker, J. J. (2009). Toward a more holistic perspective of soil erosion: why aeolian research needs to explicitly consider fluvial processes and interactions. *Aeolian Research*, *1*(1), 9-17.

Filip, O., Janda, K., Kristoufek, L., & Zilberman, D. (2017). Food versus fuel: An updated and expanded evidence. *Energy Economics*. CAMA Working Paper 73. https://cama.crawford.anu.edu.au/sites/default/files/publication/cama_crawford_anu_edu_au/2017-11/73_2017_filip_janda_kristoufek_zilberman.pdf

Finer, M., & Jenkins, C. N. (2012). Proliferation of hydroelectric dams in the Andean Amazon and implications for Andes-Amazon connectivity. PLoS ONE, *7*(4), e35126.

Finer, M., Jenkins, C.N., Pimm, S.L., Keane, B., & Ross, C. (2008). Oil and gas projects in the western Amazon: threats to wilderness, biodiversity, and indigenous peoples. *PLoS ONE*, *3*(8), p.e2932. https://doi.org/10.1371/journal.pone.0002932

Fischer, J., Abson, D. J., Butsic, V., Chappell, M. J., Ekroos, J., Hanspach, J., Kuemmerle, T., Smith, H. G., & Wehrden, H. (2014). Land sparing versus land sharing: moving forward. *Conservation Letters, 7*(3), 149-157.

Fisher, B., & Christopher, T. (2007). Poverty and biodiversity: measuring the overlap of human poverty and the biodiversity hotspots. *Ecological Economics, 62*(1), 93-101.

Fitzgerald, W. F., Engstrom, D. R., Mason, R. P., & Nater, E. A. (1998). The case for atmospheric mercury contamination in remote areas. *Environmental science & technology, 32*(1), 1-7.

Flachsbarth, I., Willaarts, B., Xie, H., Pitois, G., Mueller, N. D., Ringler, C., & Garrido, A. (2015). The role of Latin America's land and water resources for global food security: environmental trade-offs of future food production pathways. *PloS one*, 10(1), e0116733.

Fleeger, J. W., Carman, K. R., & Nisbet, R. M. (2003). Indirect effects of contaminants in aquatic ecosystems. *Science of the Total Environment*, *317* (1-3), 207–233.

Flinn, K. M., & Vellend, M. (2005). Recovery of forest plant communities in post-agricultural landscapes. *Frontiers in Ecology and the Environment*, *3*(5), 243-250.

Foden, W. B., Butchart, S. H. M., Stuart, S. N., Vié, J.-C., Akçakaya, H. R., Angulo, A., DeVantier, L. M., Gutsche, A., Turak, E., Cao, L., Donner, S. D., Katariya, V., Bernard, R., Holland, R. A., Hughes, A. F., O'Hanlon, S. E., Garnett, S. T., Şekercioğlu, Ç. H., & Mace, G. M. (2013). Identifying the world's most climate change vulnerable species: a systematic trait-based assessment of all birds, amphibians and corals. *PLoS ONE*, *8*(6), e65427. https://doi.org/10.1371/journal.pone.0065427

Foley, J. A., Asner, G. P., Costa, M. H., Coe, M. T., DeFries, R., Gibbs, H. K., Howard, E. A., Olson, S., Patz, J., Ramankutty, N., & Snyder, P. (2007). Amazonia revealed: Forest degradation and loss of ecosystem goods and services in the Amazon Basin. *Frontiers in Ecology and the Environment*, *5*(1), 25–32. https://doi.org/10.1890/1540-9295(2007)5%5B25:ARFDAL%5D2.0.CO;2

Foley, J. A., Ramankutty, N., Brauman, K. A., Cassidy, E. S., Gerber, J. S., Johnston, M., Mueller, N. D., O'Connell, C., Ray, D. K., West, P. C., Balzer, C., Bennett, E. M., Carpenter, S. R., Hill, J., Monfreda, C., Polasky, S., Rockström, J., Sheehan, J., Siebert, S., Tilman, D., & Zaks, D. P. (2011). Solutions for a cultivated planet. *Nature, 478*(7369), 337–342.

Ford, J. D., Cameron, L., Rubis, J., Maillet, M., Nakashima, D., Willox, A. C., & Pearce, T. (2016). Including indigenous knowledge and experience in IPCC assessment reports. *Nature Climate Change*, *6*(4), 349.

Foster, P. (2001). The potential negative impacts of global climate change on tropical montane cloud forests. *Earth-Science Reviews*, *55*(1), 73-106.

Foure, J., Benassy-Quere, A., Fontagne, L. (2012). The great shift: macroeconomic projections for the world economy at the 2050 horizon, G-MonD, Paris School of Economics. *Working Paper No. 23*, 70-72.

Fowler, C. (1994). *Unnatural selection: Technology, politics and plant evolution. International Studies in Global Change.* Yverdon, Switzerland and Langhorne, Pa., U.S.A.: Gordon and Breach

Frank, K. T., Petrie, B., Choi, J. S., & Leggett, W. C. (2005). Trophic cascades in a formerly cod-dominated ecosystem. *Science, 308*(5728), 1621-1623.

Franklin, K. A., Lyons, K., Nagler, P. L., Lampkin, D., Glenn, E. P., Molina-Freaner, F., Markow, T., & Huete, A. R. (2006). Buffelgrass (*Pennisetum ciliare*) land conversion and productivity in the plains of Sonora, Mexico. *Biological Conservation*, *127*(1), 62–71. https://doi.org/10.1016/j.biocon.2005.07.018

Fraser, B. (2016). Oil in the forest. *Science*, *353*(6300), 641-643.

Fraser, D., Coon, T., Prince, M., Dion, R., & Bernatchez, L. (2006). Integrating traditional and evolutionary knowledge in biodiversity conservation: a population level case study. *Ecology and Society*, *11*(2). Retrieved from http://www.ecologyandsociety.org/vol11/iss2/art4/

Freedman, J.A., Butler, S. E. & Wahl, D. H. (2012). *Impacts of invasive Asian carps on native food webs. Final project report – Illinois-Indiana Sea Grant*. Kaskaskia biological station, Illinois natural history survey. University of Illinois at Urbana-Champaign.

Freitas, M. A. B., Vieira, I. C. G., Albernaz, A. L. K. M., Magalhães, J. L. L., & Lees, A. C. (2015). Floristic impoverishment of Amazonian floodplain forests managed for açaí fruit production. *Forest Ecology and Management*, *351*, 20–27. http://www.sciencedirect.com/science/article/pii/S0378112715002777

Friis, C., Nielsen, J. Ø., Otero, I., Haberl, H., Niewöhner, J., & Hostert, P. (2016). From teleconnection to telecoupling: taking stock of an emerging framework in land system science. *Journal of Land Use Science*, *11*(2), 131-153.

Frommel, A. Y., Maneja, R., Lowe, D., Malzahn, A. M., Geffen, A. J., Folkvord, A., Piatkowski, U., Reusch, T. B., & Clemmesen, C. (2012). Severe tissue damage in Atlantic cod larvae under increasing ocean acidification. *Nature Climate Change*, 2(1), 42-46.

Fuentes, N., Pauchard, A., Sánchez, P., Esquivel, J., & Marticorena, A. (2013). A new comprehensive database of alien plant species in Chile based on herbarium records. *Biological Invasions*, 15(4), 847–858. https://doi.org/10.1007/s10530-012-0334-6

Fuentes-Franco, R., Giorgi, F., Coppola, E., Pavia, E., & Graef, F. (2015). Inter-annual variability of precipitation over Southern Mexico and Central America and its relationship to sea surface temperature from RegCM4 CORDEX projections. *Climate Dynamics*, *45*(1-2), 425–440. Springer Berlin Heidelberg. http://dx.doi.org/10.1007/s00382-014-2258-6

Fuentes-Ramírez, A., Pauchard, A., Cavieres, L. A., & García, R. A. (2011). Survival and growth of *Acacia dealbata* vs. native trees across an invasion front in south-central Chile. *Forest Ecology and Management*, *261*(6),1003–1009. https://doi.org/10.1016/j.foreco.2010.12.018

Fujimura, M., Matsuyama, A., Harvard, J.P., Bourdineaud, J.P., & Nakamura, K. (2012). Mercury contamination in humans in Upper Maroni, French Guiana between 2004 and 2009. *Bulletin of environmental contamination and toxicology*, 88(2), 135-139.

Galetti, M., Bovendorp, R. S., Fadini, R. F., Gussoni, C. O. a., Rodrigues, M., Alvarez, A. D., Guimarães Jr, P. R., & Alves, K. (2009). Hyper abundant mesopredators and bird extinction in an Atlantic forest island. *Zoologia*, *26*(2), 288–298. https://doi.org/10.1590/S1984-46702009000200011

Gall, S. C., & Thompson, R. C. (2015), The impact of debris on marine life. Mar. *Pollut. Bull., 92*(1-2), 170–179. http://doi.org/10.1016/j.marpolbul.2014.12.041

García, J., Ventura, M. I., Requena, M., Hernández, A. F., Parrón, T., & Alarcón, R. (2017). Association of reproductive disorders and male congenital anomalies with environmental exposure to endocrine active pesticides. *Reproductive Toxicology*, *71*, 95-100.

García-Frapolli, E., Ramos-Fernández, G., Galicia, E., & Serrano, A. (2009). The complex reality of biodiversity conservation through Natural Protected Area policy: Three cases from the Yucatan Peninsula, Mexico. *Land Use Policy, 26*(3), 715–722. http://doi.org/10.1016/j.landusepol.2008.09.008

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.

Garmo, Ø.A., Skjelkvåle, B.L., de Wit, H.A., Colombo, L., Curtis, C., Fölster, J., Hoffmann, A., Hruška, J., Høgåsen, T., Jeffries, D.S., & Keller, W.B. (2014). Trends in surface water chemistry in acidified areas in Europe and North America from 1990 to 2008. *Water, Air, & Soil Pollution*, *225*(33), 1880.

Gauthier, P. T., Norwood, W. P., Prepas, E. E., & Pyle, G. G. (2014). Metal-PAH mixtures in the aquatic environment: A review of co-toxic mechanisms leading to more-than-additive outcomes. *Aquatic Toxicology, 154*, 253–269.

Gavilán-García, I., Santos-Santos, E., Tovar-Gálvez, L.R., Gavilán-García, A., Suárez, S., & Olmos, J. (2008). Mercury speciation in contaminated soils from old mining activities in Mexico using a chemical selective extraction. *Journal of the Mexican Chemical Society*, *52*(4), 263-271.

Gellis, A. C. (2013). Factors influencing storm-generated suspended-sediment concentrations and loads in four basins of contrasting land use, humid-tropical Puerto Rico. *Catena*, *104*, 39-57.

Giberto, D. A., Bremec, C. S., Schejter, L., Schiariti, A., Mianzan, H., & Acha, E. M. (2006). The invasive Rapa Whelk Rapana venosa (Valenciennes 1846): status and potential ecological impacts in the Río de la Plata estuary, Argentina-Uruguay. *Journal of Shellfish Research*, *25*(3), 919-924.

Giberto, D. A., Schiariti, A., & Bremec, C. S. (2011). Diet and daily consumption rates of *Rapana venosa* (Valenciennes, 1846) (Gastropoda: Muricidae) from the Río de la Plata (Argentina-Uruguay). *Journal of Shellfish Research* (2), 349-358.

Gilliom, R. J., Barbash, J. E., Crawford, C. G., Hamilton, P. A., Martin, J. D., Nakagaki, N., Nowell, L. H., Scott, J. C., Stackelberg, P. E., Thelin, G. P., & Wolock, D. M. (2006). Pesticides in the Nation's streams and groundwater, 1992–2001: *U.S Geological Survey*. Circular 1291. http://pubs.usgs.gov/circ/2005/1291/pdf/circ1291.pdf

Giorgi, F., & N. Diffenbaugh. (2008). Developing regional climate change scenarios for use in assessment of effects on human health and disease. *Climate Research*, *36*(2), 141-151.

Giri, C., Ochieng, E., Tieszen, L. L., Zhu, Z., Singh, A., Loveland, T., Masek, J., & Duke, D. (2011). Status and distribution of mangrove forests of the world using earth observation satellite data. *Global Ecology and Biogeography, 20*(1),154–159.

Gledhill, D. K., Wanninkhof, R., Millero, F. K., & Eakin, M. (2008). Ocean acidification of the greater Caribbean region 1996-2006. *Journal of Geophysical Research: Oceans,* 113 (10), 1–11.

Glibert, P. M. (2017). Eutrophication, harmful algae and biodiversity—Challenging paradigms in a world of complex nutrient changes. *Marine Pollution Bulletin*, *124*(2), 591-606.

Glibert, P.M., Harrison, J., Heil, C., & Seitzinger, S. (2006). Escalating worldwide use of urea–a global change contributing to coastal eutrophication. *Biogeochemistry*, 77(3), 441-463.

Global Biodiversity Information Facility. (2011). *GBIF position paper on data hosting infrastructure for primary biodiversity data. Version 1.0.*

Global Environmental Outlook GEO 4, UNEP. (2007). United Nations Environmental Programme.

Global Footprint Network (GFN). (2017) http://data.footprintnetwork.org/countryMetrics.html?cn=all&yr=. Date accessed: June 10, 2017.

Gloor, M., Barichivich, J. Ziv, G., Brienen, R., Schöngart, J., & Peylin, P., Barcante L. C., B., Feldpausch, T., Phillips, O., & Baker, J. (2015). Recent Amazon climate as background for possible ongoing and future changes of Amazon humid forests. *Global Biogeochemical Cycles, 29* (9). https://doi.org/10.1002/2014GB005080

Godar, J., Suavet, C., Gardner, T. A., Dawkins, E., & Meyfroidt, P. (2016). Balancing detail and scale in assessing transparency to improve the governance of agricultural commodity supply chains. *Environmental Research Letters*, *11*(3), 035015.

Godt, J. W., Arnal, C. H., Baum, R. L., Brien, D., Coe, J. A., De Mouthe, J., Ellis, W., Graymer, R. W., Harp, E. L., Hillhouse, J. W., Houdre, N., Howell, D. G., Jayko, A. S., Lajoie, K. R., Morrissey, M. M., Ramsey, D. W., Savage, W. Z., Schuster, R. L., Wieczorek, G.F., & Wilson, R. C. (1999). *Maps showing locations of damaging landslides caused by El Niño rainstorms, winter season 1997-98, San Francisco Bay region, California*. Retrieved from https://pubs.usgs.gov/mf/1999/mf-2325/

Golicher, D. J., Cayuela, L., & Newton, A. C. (2012). Effects of climate change on the potential species richness of Mesoamerican forests. *Biotropica*, 44(3), 284-293.

Gomez-Salazar, C., Trujillo, F., Portocarrero-Aya, M., Whitehead, H. (2012). Population, density estimates, and conservation of river dolphins (Inia and Sotalia) in the Amazon and Orinoco river basins. *Marine Mammal Science, 28*(1), 124–153.

Goosem, S. P., & Tucker, N. I. (2013). *Repairing the rainforest* (second edition). Wet Tropics Management Authority and Biotropica Australia Pty. Ltd. Cairns.

Goreau, T. J. (1992). Bleaching and reef community change in Jamaica: 1951–1991. *American Zoologist*, 32(6), 683-695.

Gorenflo, L. J., Romaine, S., Mittermeier, R. A., & Walker-Painemilla, K. (2012). Co-occurrence of linguistic and biological diversity in biodiversity hotspots and high biodiversity wilderness areas. *Proceedings of the National Academy of Sciences*, 109(21), 8032-8037.

Gorokhovich, Y., Voros, A., Reid, M., & Mignone, E. (2003). Prioritizing abandoned coal mine reclamation projects within the contiguous United States Using geographic information system extrapolation. *Environmental Management*, *32*(4), 527-534, https://doi.org/10.1007/s00267-003-3043-1

Graesser, J., Aide, T. M., Grau, H. R., & Ramankutty, N. (2015). Cropland/pastureland dynamics and the slowdown of deforestation in Latin America. *Environmental Research Letters*, 10(3), 034017.

Grau, H. R., & Aide, M. (2008). Globalization and Land-Use Transitions in Latin America. *Ecology and Society, 13*(2), art 16. Retrieved from http://www.ecologyandsociety.org/vol13/iss2/art16/

Green, S., Akins, J., Maljkovic, A., & Côté, I. (2012). Invasive lionfish drive Atlantic coral reed fish declines. *PLoS ONE*, *7*(3), e32596. https://doi.org/10.1371/journal.pone.0032596

Griffiths, R. W., Schloesser, D. W., Leach, J. H., & Kovalak, W. P. (1991). Distribution and dispersal of the zebra mussel (*Dreissena polymorpha*) in the Great Lakes region. *Canadian Journal of Fisheries and Aquatic Sciences*, *48*(8), 1381–1388. https://doi.org/10.1139/f91-165

Grogan, J., Blundell, A. G., Landis, R. M., Youatt, A., Gullison, R. E., Martinez, M., Kometter, R., Lentini, M., & Rice, R. E. (2010). Over-harvesting driven by consumer demand leads to population decline: big-leaf mahogany in South America. *Conservation Letters, 3*(1), 12–20.

Guallar, E., Sanz-Gallardo, M. I., Veer, P. V. T., Bode, P., Aro, A., Gómez-Aracena, J., Kark, J.D., Riemersma, R. A., Martín-Moreno, J. M., & Kok, F. J. (2002). Mercury, fish oils, and the risk of myocardial infarction. *New England Journal of Medicine*, 347(22), 1747-1754.

Guatemala Ramsar National Report (2015). *Informe nacional sobre la aplicación de la convención de ramsar sobre los humedales*. http://www.ramsar.org/sites/default/files/documents/library/cop12nrfguatemala_20140903final.pdf

Gudynas, E. (2011). Buen Vivir: Today's tomorrow. *Development, 54*(4), 441–447. http://doi.org/10.1057/dev.2011.86

Guha-Sapir D., Hoyois Ph., & Below. R. (2014) *Annual disaster statistical review 2013:the numbers and trends*. Brussels. Centre for Research on the Epidemiology of Disasters.

Guha, S., & Bhattacharya, S. (2014). Non-parametric Non-stationary modeling of spatio-temporal data through state space approach. *arXiv preprint arXiv*:1405.6531.

Guinea B., H. E., Swain, A., Wallin, M. B., & Nyberg, L. (2015). Disaster Management Cooperation in Central America: The case of rainfall-induced natural disasters. *Geografiska Annaler: Series A, Physical Geography, 97*(1), 85-96.

Gutiérrez-Galindo, E. A., Casas-Beltrán, D. A., Muñoz-Barbosa, A., Macías-Zamora, J.V., Segovia-Zavala, J.A., Orozco-Borbon, M.V., & Daessle, L.W. (2007). Spatial distribution and enrichment of mercury in surface sediments off the northwest coast of Baja California, Mexico. *Ciencias Marinas*, *33*(4), 473-482.

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., & J. R. Townshend. (2015). Habitat Fragmentation and its Lasting Impact on Earth's Ecosystems. *Science Advances, 1*(2), 1–9. http://advances.sciencemag.org/content/1/2/e1500052.abstract

Hall-Spencer, J. M., Rodolfo-Metalpa, R., Martin, S., Ransome, E., Fine, M., Turner, S. M., Rowley, S. J., Tedesco, D., & Buia, M. C. (2008). Volcanic carbon dioxide vents show ecosystem effects of ocean acidification. *Nature, 454*(7200), 96-99.

Halpern, B. S., Selkoe, K. A., Micheli, F., & Kappel, C. V. (2007). Evaluating and ranking the vulnerability of global marine ecosystems to anthropogenic threats. *Conservation Biology*, *21*(5), 1301-1315.

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., Spalding, 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

Hansen, A. J., Piekielek, N., Davis, C., Haas, J., Theobald, D. M., Gross, J. E., Monahan, W. B., Olliff, T., & Running, S. W. (2014). Exposure of U.S. National Parks to land use and climate change 1900-2100. *Ecological Applications : A Publication of the Ecological Society of America, 24*(3), 484–502. Retrieved from http://www.ncbi.nlm.nih.gov/pubmed/24834735

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

Hanski, I., Zurita, G. a, Bellocq, M. I., & Rybicki, J. (2013). Species-fragmented area relationship. *PNAS*, *110*(31), 12715–12720. https://doi.org/10.1073/pnas.1311491110

Hanson, C., Yonavjak, L., Clarke, C., Minnemeyer, S., Boisrobert, L., Leach, A., & Schleeweis, K. (2010). *Southern Forests for the Future*. Washington, DC.

Hanson, C., Buckingham, K. DeWitt, S., & Laestadius, L. (2015) *The Restoration Diagnostic: a method for developing forest landscape restoration strategies by rapidly assessing the status of key success factors*. WRI, IUCN, Washington, DC, USA.

Hao, Y., Strosnider, H., Balluz, L., & Qualters, J.R. (2016). Geographic variation in the association between ambient fine particulate matter (PM2. 5) and term low birth weight in the United States. *Environmental health perspectives*, *124*(2), 250-255.

Hardell, S., Tilander, H., Welfinger-Smith, G., Burger, J., & Carpenter, D.O. (2010). Levels of polychlorinated biphenyls (PCBs) and three organochlorine pesticides in fish from the Aleutian Islands of Alaska. *PloS one*, *5*(8), p.e12396.

Hardin, G. (1968). The tragedy of the commons. *Science*, *162*(3859), 1243-1248 DOI: 10.1126/science.162.3859.1243.

Harding, J. M., & Mann, R. (1999). Observations on the biology of the Veined Rapa Whelk, *Rapana venosa* (Valenciennes, 1846) in the Chesapeake Bay. *Journal of Shellfish Research*, *18*(1), 9-18.

Hare, J. A., Alexander, M. A., Fogarty, M. J., Williams, E. H., & Scott, J. D. (2010). Forecasting the dynamics of a coastal fishery species using a coupled climate–population model. *Ecological Applications, 20*(2), 452–464. https://doi.org/10.1890/08-1863.1

Harmon, D., & Loh, J. (2010). The index of linguistic diversity: a new quantitative measure of trends in the status of the world's languages. *Language Documentation & Conservation*, *4*, 97-151.

Hayhoe, K., VanDorn, J., Croley, T., Schlegal, N., & Wuebbles, D. (2010). Regional climate change projections for Chicago and the US Great Lakes. *Journal of Great Lakes Research*, *36*, 7–21. https://doi.org/10.1016/j.jglr.2010.03.012

Hebblewhite, M. (2017). Billion-dollar boreal woodland caribou and the biodiversity impacts of the global oil and gas industry. *Biological Conservation*, *206*, 102–111.

Heithaus, M. R., Frid, A., Wirsing, A. J., & Worm, B. (2008). Predicting ecological consequences of marine top predator declines. *Trends in Ecology & Evolution, 23*(2), 202–210.

Helmer, E. H., Kennaway, T. A., Pedreros, D. H.,. Clark, M., Marcano-Vega, H. , Tieszen, L. L., Ruzycki, T. R., Schill, S. R., & Carrington, C. M. S. (2008). Land cover and forest formation distributions for St. Kitts, Nevis, St. Eustatius, Grenada and Barbados from decision tree classification of cloud-cleared satellite imagery. *Caribbean Journal of Science*, *44*(2), 175–198.

Helmer, E. H., Brandeis, T. J., Lugo, A. E., & Kennaway, T. (2008). Factors influencing spatial pattern in tropical forest clearance and stand age: Implications for carbon storage and species diversity. *Journal of Geophysical Research: Biogeosciences*, *113*(2). https://doi.org/10.1029/2007JG000568

Helmer, E. H., Ruzycki, T. S., Benner, J., Voggesser, S. M., Scobie, B. P., Park, C., Fanning, D. W., & Ramnarine, S. (2012). Detailed maps of tropical forest types are within reach: Forest tree communities for Trinidad and Tobago mapped with multiseason Landsat and multiseason fine-resolution imagery. *Forest Ecology and Management*, *279*, 147–166. https://doi.org/10.1016/j.foreco.2012.05.016

Henders, S., Persson, U. M., & Kastner, T. (2015). Trading forests: land-use change and carbon emissions embodied in production and exports of forest-risk commodities. *Environmental Research Letters*, *10*(12), 125012.

Henson, J. I., Muller-Karger, F., Wilson, D., Morey, S. L., Maul, G. A., Luther, M., & Kranenburg, C. (2006). Strategic geographic positioning of sea level gauges to aid in early detection of tsunamis in the Intra-Americas Sea. *Science of Tsunami Hazards 25*(3), 173-207.

Hernández, D. L., Vallano, D. M., Zavaleta, E. S., Tzankova, Z., Pasari, J. R., Weiss, S., Selmants, P. C., & Morozumi, C. (2016). Nitrogen pollution is linked to US listed species declines. *BioScience*, *66*(3), 213–222.

Hernández, G., Lahmann, E., & Pérez-Gil, R. (2002) Invasores en Mesoamérica y el Caribe (Invasives in Mesoamerica and the Caribbean). 1.ed. San José, C.R.: *UICN*. http://www.issg.org/pdf/publications/GISP/Resources/Mesoamerica.pdf

Herrmann, T. M. (2006). Indigenous knowledge and management of Araucaria araucana forest in the Chilean Andes: implications for native forest conservation. *Biodiversity and Conservation, 15*(2), 647-662.

Heyman, W. D., Graham, R. T., Kjerfve, B., & Johannes, R. E. (2001). Whale sharks Rhincodon typus aggregate to feed on fish spawn in Belize. *Marine Ecology Progress Series*, 215, 275-282.

Hietz, P., Turner, B.L., Wanek, W., Richter, A., Nock, C.A., & Wright, S.J. (2011). Long-term change in the nitrogen cycle of tropical forests. *Science, 334*(6056), 664-666.

Hilborn, R., & Ovando, D. (2014). Reflections on the success of traditional fisheries management. *ICES Journal of Marine Science*, *71*(5), 1040-1046.

Hill, R., Dyer, G. A., Lozada-Ellison, L. M., Gimona, A., Martín-Ortega, J., Munoz-Rojas, J., & Gordon, I. J. (2015). A social–ecological systems analysis of impediments to delivery of the Aichi 2020 Targets and potentially more effective pathways to the conservation of biodiversity. *Global Environmental Change, 34*, 22-34.

Hillstrom, M. L., & Lindroth, R. L. (2008). Elevated atmospheric carbon dioxide and ozone alter forest insect abundance and community composition. *Insect Conservation and Diversity*, *1*(4), 233–241. https://doi.org/10.1111/j.1752-4598.2008.00031.x

Hinsley, A., Verissimo, D., & Roberts, D. L. (2015). Heterogeneity in consumer preferences for orchids in international trade and the potential for the use of market research methods to study demand for wildlife. *Biological Conservation*, *190*, 80–86.

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

Hixon, M. A., Green, S. J., Albins, M. A., Akins, J. L., & Morris, J. A. (2016). Lionfish: A major marine invasion. *Marine Ecology Progress Series, 558*, 161–165.

Hobbs, R. J. (2000). Land-use changes and invasions. In Mooney H. A., & R. J. Hobbs (Eds.) *Invasive species in a changing world*. (pp. 55–64)Washington, DC: Island Press.

Hodgson, D. L. (2002). Introduction: Comparative perspectives on the indigenous rights movement in Africa and the Americas. *American Anthropologist, 104*(4), 1037-1049.

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.* Molnar, J.L.(Ed.). Berkeley: University of California Press.

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.

Holcombe, E., Smith, S., Wright, E., & Anderson, M. G. (2012). An integrated approach for evaluating the effectiveness of landslide risk reduction in unplanned communities in the Caribbean. *Natural Hazards, 61*(2), 351–385. https://doi.org/10.1007/s11069-011-9920-7

Holway, D. A., Lach, L., Suarez, A. V., Tsutsui, N. D., & Case, T. J. (2002). The causes and consequences of ant invasions. *Annual Review of Ecology and Systematics*, *33*(1), 181–233.

Honduras Ramsar National Report (2015). Informe nacional sobre la aplicación de la convención de ramsar sobre los humedales. Informes Nacionales que se presentarán a la 12ª Reunión de la Conferencia de las Partes Contratantes. http://www.ramsar.org/sites/default/files/documents/library/cop12_nr_honduras.pdf

Hooper, M.J., Mineau, P., Zaccagnini, M. E., & Woodbridge, B. (2002). Pesticides and International Migratory Bird Conservation. Chapter 25, (pp. 737-753), In Hoffman, D. J., Rattnes, B. A., Burton, G. A., & Cairns Jr., J. (Eds). *Handbook of Ecotoxicology*. Boca Raton. Florida: Lewis Publishers, CRC Press.

Hoover, E., Cook, K., Plain, R., Sanchez, K., Waghiyi, V., Miller, P., Dufault, R., Sislin, C., & Carpenter, D.O. (2012). Indigenous peoples of North America: environmental exposures and reproductive justice. *Environmental Health Perspectives, 120*(12), 1645-49. DOI:10.1289/ehp.1205422.

Horowitz, H. M., Jacob, D. J., Amos, H. M., Streets, D. G., & Sunderland, E.M. (2014). Historical mercury releases from commercial products: Global environmental implications. *Environmental science & technology*, *48*(17), 10242-10250.

Howarth, R.W. (2014). A bridge to nowhere: methane emissions and the greenhouse gas footprint of natural gas. *Energy Science & Engineering, 2*(2), 47-60.

Howell Rivero, L., & Rivas, L. R. (1940). Algunas consideraciones sobre los ciclidos de Cuba. *Mem Soc. Cub. Hist. Nat*, 14(4), 373-395

Huffman, M. R. (2013). The many elements of traditional fire knowledge: Synthesis, classification, and aids to cross-cultural problem solving in firedependent systems around the world. *Ecology and Society*, *18*(4), art3. https://doi.org/10.5751/ES-05843-180403

Hufkens, K., Keenan, T. F., Flanagan, L. B., Scott, R. L., Bernacchi, C. J., Joo, E., Brunsell, N. A., Verfaillie, J., & Richardson, A. D. (2016). Productivity of North American grasslands is increased under future climate scenarios despite rising aridity. *Nature Climate Change, 6*(7), 710–714. https://doi.org/10.1038/nclimate2942

Hughes, R. M., Amezcua, F., Chambers, D. M., Daniel, W. M., Franks, J. S., Franzin, W., MacDonald, D., Merriam, E., Neall, G., dos Santos Pompeu, P., Reynolds, L., & Woody, C. A. (2016). AFS Position Paper and Policy on Mining and Fossil Fuel Extraction. *Fisheries, 41*(1), 12–15. https://doi.org/10.1080/03632415.2016.1121742

Hughes, T. P. (1994). Catastrophes, phase shifts, and large-scale degradation of a Caribbean coral reef. *Science*, *265*(5178), 1547–1551. https://doi.org/10.1126/science.265.5178.1547

Hull Sieg, C. (1987). Small mammals: pests or vital components of the ecosystem. Great Plains Wildlife Damage Control Workshop 97:88–92.

Hung, H., Blanchard, P., Halsall, C.J., Bidleman, T.F., Stern, G.A., Fellin, P., Muir, D.C.G., Barrie, L.A., Jantunen, L.M., Helm, P.A., Ma, J., & Konoplev, A. (2005). Temporal and spatial variabilities of atmospheric polychlorinated biphenyls (PCBs), organochlorine (OC) pesticides and polycyclic aromatic hydrocarbons (PAHs) in the Canadian Arctic: Results from a decade of monitoring. *Science of the Total Environment, 342*(1), 119-144.

IAWG, U. (2013). *Technical support document: Technical update of the social cost of carbon for regulatory impact analysis under executive order 12866*. Interagency Working Group on Social Cost of Carbon. United States Government, Washington, DC.

Ibanez, M., & Blackman, A. (2016). Is Eco-Certification a Win–Win for Developing Country Agriculture? Organic Coffee Certification in Colombia. *World Development*, *82*, 14–27. https://doi.org/10.1016/J.WORLDDEV.2016.01.004

IEA. (2015). *Key World Energy Statistics*. International Energy Agency, Paris: OECD/IEA.

IEA. (2016). *Key World Energy Statistics*. International Energy Agency, Paris: OECD/IEA.

International Monetary Fund (IMF). (2014). *World Economic Outlook: Legacies, Clouds, Uncertainties*. Washington (October). http://www.imf.org

International Monetary Fund (IMF). (2015). *World Economic Outlook: Adjusting to Lower Commodity Prices*. Washington (October). http://www.imf.org

International Monetary Fund (IMF). (2016). *World Economic Outlook: Subdued Demand: Symptoms and Remedies*. Washington, October. http://www.imf.org

International Monetary Fund (IMF). (2017). *Seeking Sustainable Growth: Short-Term Recovery, Long-Term Challenges.* Washington, DC, October. http://www.imf.org

International Tropical Timber Organization (ITTO). (2011). *25 Success stories - Illustrating ITTO's 25-year quest to sustain tropical forests*. Yokohama, Japan.

Inoue, C. Y. A., & Moreira, P. F. (2016). Many worlds, many nature(s), one planet: indigenous knowledge in the Anthropocene. *Revista Brasileira de Política Internacional, 59*(2), 1–19.

Intergovernmental Panel on Climate Change. (2014). *Climate Change 2014 Synthesis Report Summary Chapter for Policymakers*. IPCC.

IPBES. (2016). *The methodological assessment report on scenarios and models of biodiversity and ecosystem services*. Ferrier, S., Ninan, K. N., Leadley, P., Alkemade, R., Acosta, L. A., Akçakaya, H. R., Brotons, L., Cheung, W. W. L., Christensen, V., Harhash, K. A., Kabubo-Mariara, J., Lundquist, C., Obersteiner, M., Pereira, H. M., Peterson, G., Pichs-Madruga, R., Ravindranath, N., Rondinini C., & Wintle, B.A. (eds.). Secretariat of the Intergovernmental. Science-Policy Platform on Biodiversity and Ecosystem Services, Bonn, Germany.

IPCC. (2014a). *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects*. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change. Field, C. B., Barros, V. R., Dokken, D. J., Mach, K. J., Mastrandrea, M. D., Bilir, T. E, Chatterjee, M., Ebi, K. L., Estrada, Y. O., Genova, R. C., Girma, B., Kissel, E. S., Levy, A. N., MacCracken, S., Mastrandrea, P. R., & White, L. L. (Eds.). United Kingdom and New York, NY, USA: Cambridge University Press, Cambridge.

IPCC. (2014b). Summary for policymakers. In Field, C.B., V.R. Barros, D.J. Dokken, K.J. Mach, M.D. Mastrandrea, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, & L.L.White (Eds.). *Climate Change 2014: Impacts,Adaptation, and Vulnerability. Part A: Global and Sectoral Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. (pp. 1-32). United Kingdom and New York, NY, USA: Cambridge University Press, Cambridge.

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.

Irigoyen, A. J., Trobbiani, G., Sgarlatta, M. P., & Raffo, M. P. (2011). Effects of the alien algae *Undaria pinnatifida* (Phaeophyceae, Laminariales) on the diversity and abundance of benthic macrofauna in Golfo Nuevo (Patagonia, Argentina): Potential implications for local food webs. *Biological Invasions, 13*(7), 1521–1532.

Isbell, F., Tilman, D., Polasky, S., & Loreau, M. (2015). The biodiversity-dependent ecosystem service debt. *Ecology Letters, 18*(2), 119-134 https://doi.org/10.1111/ele.12393

Isbell, F., Gonzalez, A., Loreau, M., Cowles, J., Díaz, S., Hector, A., Mace, G. M., Wardle, D. A., O'Connor, M. I., Duffy, J. E., Turnbull, L. A., Thompson, P. L., & Larigauderie, A. (2017). Linking the influence and dependence of people on biodiversity across scales. *Nature, 546*(7656), 65–72. https://doi.org/10.1038/nature22899

Isbell, F., Reich, P. B., Tilman, D., Hobbie, S. E., Polasky, S., & Binder, S. (2013). Nutrient enrichment, biodiversity loss, and consequent declines in ecosystem productivity. *Proceedings of the National Academy of Sciences, 110*(29), 11911-11916.

IUCN. (2016). *The IUCN Red List of Threatened Species*. Version 2016. http://www.iucnredlist.org. Downloaded on 12 May 2016.

Jackson, J. B. C., Donovan, M. K., Cramer, K. L., & Lam, V. V. (2014). *Status and trends of Caribbean coral reefs.* Gland: global coral reef monitoring network, IUCN. Available at http://cmsdata.iucn.org/downloads/caribbean_coral_reefs___status_report_1970_2012. pdf

Jackson, J. B. C., Kirby, M. X., Berger, W. H., Bjorndal, K. A., Botsford, L. W., Bourque, B. J., Bradbury, R. H., Cooke, R., Erlandson, J., Estes, J. A., Hughes, T. P., Kidwell, S., Lange, C. B., Lenihan, H. S., Pandolfi, J. M., Peterson, C. H., Steneck, R. S., Tegner, M. J., & Warner, R. R. (2001). Historical overfishing and the recent collapse of coastal ecosystems. *Science, 293*(5530), 629-637. Retrieved from http://science.sciencemag.org/content/293/5530/629.abstract

Jacobs, D. F., Dalgleish, H. J., & Nelson, C. D. (2013). A conceptual framework for restoration of threatened plants: The effective model of American chestnut (Castanea *dentata*) reintroduction. *New Phytologist, 197*(2), 378–393.

Jambeck, J. R., Geyer, R., Wilcox, C., Siegler, T. R., Perryman, M., Andrady, A., Narayan, R., & Law, K. L. (2015). Plastic waste inputs from land into the ocean. *Science, 347*(6223), 768–771. https://doi.org/10.1126/science.1260352

Jantz, S. M., Barker, B., Brooks, T. M., Chini, L. P., Huang, Q., Moore, R. M., Noel, J., & Hurtt, G. C. (2015). Future habitat loss and extinctions driven by land-use change in biodiversity hotspots under four scenarios of climate-change mitigation. *Conservation Biology, 29*(4), 1122–1131. https://doi.org/10.1111/cobi.12549

Jaramillo, A., Osman, D., Caputo, L. & L. Cardenas. (2015). Molecular evidence of a *Didymosphenia geminata* (Bacillariophyceae) invasion in Chilean freshwater systems. *Harmful Algae, 49,* 117–123

Jaramillo, E., Dugan, J. E., Hubbard, D. M., Melnick, D., Manzano, M., Duarte, C., Campos, C., & Sanchez, R. (2012). Ecological implications of extreme events: Footprints of the 2010 earthquake along the Chilean coast. *PLoS ONE, 7*(5), e35348. https://doi.org/10.1371/journal.pone.0035348

Jaspers, V.L.B., Sonne, C., Soler-Rodriguez, F., Boertmann, D., Dietz, R., Eens, M., Rasmussen, L.M., & Covaci, A. (2013). Persistent organic pollutants and methoxylated polybrominated diphenyl ethers in different tissues of white-tailed eagles (*Haliaeetus albicilla*) from West Greenland. *Environmental pollution, 175*, 137-146.

Jenkins, C. N., Pimm, S. L., & Joppa, L. N. (2013). Global patterns of terrestrial vertebrate diversity and conservation. *Proceedings of the National Academy of Sciences, 110*(28), 2602–2610.

Jeschke, J. M., & Strayer, D. L. (2005). Invasion success of vertebrates in Europe and North America. *Proceedings of the National Academy of Sciences of the United States of America, 102*(20), 7198-7202.

Jetoo, S., Thorn, A., Friedman, K., Gosman, S., & Krantzberg, G. (2015). Governance and geopolitics as drivers of change in the Great Lakes—St. Lawrence basin. *Journal of Great Lakes Research, 41*(1), 108-118.

Jia, Y., Yu, G., Gao, Y., He, N., Wang, Q., Jiao, C., & Zuo, Y. (2016). Global inorganic nitrogen dry deposition inferred from ground-and space-based measurements. *Scientific reports, 6*, art 19810.

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.

Johnson, J. W., Oelkers, E. H., & Helgeson, H. C. (1992). SUPCRT92: A software package for calculating the standard molal thermodynamic properties of minerals, gases, aqueous species, and reactions from 1 to 5000 bar and 0 to 1000 C. *Computers & Geosciences, 18*(7), 899-947.

Johnson, L. E., & Padilla, D. K. (1996). Geogaphic spread of exotic species. Ecological lessons and opportunities from the invasion of the zebra mussel *Dreissena polymorpha*. *Biological Conservation, 78*(96), 23–33.

Johnson, W. C., & Poiani, K. A. (2016). Climate Change Effects on Prairie Pothole Wetlands: Findings from a Twenty-five Year Numerical Modeling Project. *Wetlands, 36*(2), 273–285. https://doi.org/10.1007/s13157-016-0790-3

Johnstone, J. F., Hollingsworth, T. N., Chapin, F. S., & Mack, M. C. (2010). Changes in fire regime break the legacy lock on successional trajectories in Alaskan boreal forest. *Global Change Biology, 16*(4), 1281-1295.

Jones, H. P., & Schmitz, O. J. (2009). Rapid recovery of damaged ecosystems. *PloS one*, *4*(5), e5653.

Jones, N., McGinlay, J., & Dimitrakopoulos, P. G. (2017). Improving social impact assessment of protected areas: A review of the literature and directions for future research. *Environmental Impact Assessment Review, 64*, 1–7. http://doi.org/10.1016/j.eiar.2016.12.007

Jørgensen, O. A., Bastardie, F., & Eigaard, O. R. (2014). Impact of deep-sea fishery for Greenland halibut (*Reinhardtius hippoglossoides*) on noncommercial fish species off West Greenland. *ICES. Journal of Marine Science, 71*(2), 845–852.

Joyce, L. A., Running, D., Breshears, V., Dale, R., Malmsheimer, R. W., Sampson, B., Sohngen, B., & Woodall, C. W. (2014). In Melillo, J. M., Richmond, T. C., Yohe, G. W. (Eds). *Climate change impacts in the United States: The Third National Climate Assessment.* (pp. 175-194). Washington, DC: US Global Change Research Program. http://nca2014.globalchange.gov/report/our-changing-climate/introduction

Juliano, S. A., & Philip Lounibos, L. (2005). Ecology of invasive mosquitoes: Effects on resident species and on human health. *Ecology Letters, 8*(5), 558–574.

Junk, W. J. (2013). Current state of knowledge regarding South America wetlands and their future under global climate change. *Aquatic Sciences, 75*(1), 113-131.

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.

Junk, W. J., Piedade, M. T. F., Schöngart, J., & Wittmann, F. (2012). A classification of major natural habitats of Amazonian white-water river floodplains (várzeas). *Wetlands Ecology and Management*, *20*(6), 461-475.

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

Junqueira, A. B., Shepard, G. H., & Clement, C. R. (2010). Secondary forests on anthropogenic soils in Brazilian Amazonia conserve agrobiodiversity. *Biodiversity and Conservation, 19*(7), 1933-1961.

Kairo, M., Ali, B., Cheesman, O., Haysom, K., & Murphy, S. (2003). Invasive species threats in the Caribbean region. *Report to the Nature Conservancy Arlington*.

Kaiser Family Foundation. (2013). Global health facts – Urban population (Percent of total population living in urban areas). http://kff.org/global-indicator/urban-population/. Accessed 10 July 2013.

Kaplanis, N. J., Harris, J. L., & Smith, J. E. (2016). Distribution patterns of the non-native seaweeds *Sargassum horneri* (Turner) *C. Agardh* and *Undaria pinnatifida* (Harvey) Suringar on the San Diego and Pacific coast of North America. *Aquatic Invasions, 11*(2), 111–124. https://doi.org/10.3391/ai.2016.11.2.01

Karatayev, A. Y., Padilla, D. K., Minchin, D., Boltovskoy, D., & Burlakova, L. E. (2007). Changes in global economies and trade: the potential spread of exotic freshwater bivalves. *Biological Invasions*, *9*(2), 161-180.

Karmalkar, A. V., Bradley, R. S., & Diaz, H. F. (2008). Climate change scenario for Costa Rican montane forests. *Geophysical Research Letters, 35*(11).

Karmalkar, A. V., Bradley, R. S., & Diaz, H. F. (2011). Climate change in Central America and Mexico: regional climate model validation and climate change projections. *Climate dynamics*, *37*(3-4), 605- 629.

Kastner, T., Erb, K. H., & Haberl, H. (2015). Global human appropriation of net primary production for biomass consumption in the European Union, 1986–2007. *Journal of Industrial Ecology, 19*(5), 825-836.

Kauffman, J. B., Trejo, H. H., Garcia, M. D. C. J., Heider, C., & Contreras, W. M. (2016). Carbon stocks of mangroves and losses arising from their conversion to cattle pastures in the Pantanos de Centla, Mexico. *Wetlands Ecology and Management, 24*(2), 203-216.

Kaufmann, D., Kraay, A., & Mastruzzi, M. (2007). *The worldwide governance indicators project: answering the critics*. The World Bank Policy Research Working Paper, 4149. Available at SSRN: https://ssrn.com/abstract=965077

Kaufmann, D., Kraay, A., & Mastruzzi, M. (2010). The worldwide governance indicators: Methodology and analytical issues. World Bank Policy Research Woking Paper No. 5430.

Kawaguchi, S., Ishida, A., King, R., Raymond, B., Waller, N., Constable, A., Nicol, S., Wakita, M., & Ishimatsu, A. (2013). Risk maps for Antarctic krill under projected Southern Ocean acidification. *Nature Climate Change,* 3(9), p.843.

Keegan, K. M., Albert, M. R., McConnell, J. R., & Baker, I. (2014). Climate change and forest fires synergistically drive widespread melt events of the Greenland Ice Sheet. *Proceedings of the National Academy of Sciences*, *111*(22), 7964-7967.

Keeler, B. L., Gourevitch, J. D., Polasky, S., Isbell, F., Tessum, C. W., Hill, J. D., & Marshall, J. D. (2016). The social costs of nitrogen. *Science advances*, *2*(10), e1600219.

Keenan, R. J., Reams, G. A., Achard, F., de Freitas, J. V., 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.

Kerckhof, F., Vink, R. J., Nieweg, D. C., & Post, J. J. N. (2006). The veined whelk Rapana venosa has reached the North Sea. *Aquatic Invasions*, (1), 35-37.

Kerr, L. A., Secor, D. H., & Piccoli, P. M. (2009). Partial migration of fishes as exemplified by the estuarine-dependent white perch. *Fisheries, 34*(3), 114-123.

Kershaw, P., Katsuhiko, S., Lee, S., & Woodring, D. (2011). *Plastic debris in the ocean*. United Nations Environment Programme.

King, R. S., & Baker, M. E. (2010). Considerations for analyzing ecological community thresholds in response to anthropogenic environmental gradients. *Journal of the North American Benthological Society*, *29*(3), 998-1008.

Kirby, K. R., Laurance, W. F., Albernaz, A. K., Schroth, G., Fearnside, P. M., Bergen, S., Venticinque, E. M., & da Costa, C. (2006). The future of deforestation in the Brazilian Amazon. *Futures, 38*(4), 432–453. https://doi.org/10.1016/j.futures.2005.07.011

Kirchner, M., Faus-Kessler, T., Jakobi, G., Levy, W., Henkelmann, B., Bernhöft, S., Kotalik, J., Zsolnay, A., Bassan, R., Belis, C., & Kräuchi, N. (2009). Vertical distribution of organochlorine pesticides in humus along Alpine altitudinal profiles in relation to ambiental parameters. *Environmental pollution, 157*(12), 3238-3247.

Kirwan, M. L., & Megonigal, J. P. (2013). Tidal wetland stability in the face of human impacts and sea-level rise. *Nature, 504*(7478), 53-60.

Klein E., Cardenas, J.J., Esclasans, D. (2009). *Prioridades de conservación de la biodiversidad marina del Frente Atlántico y Golfo de Paria*.

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.

Knick, S. T., Dobkin, D. S., Rotenberry, J. T., Schroeder, M. A., & Vander Haegen, W. M. (2003). Teetering on the edge or too late? Conservation and research issues for avifauna of sagebrush habitats. *The Condor, 105*(4), 611-634.

Kocman, D., Horvat, M., Pirrone, N., & Cinnirella, S. (2013). Contribution of contaminated sites to the global mercury budget. *Environmental research, 125*, 160-170.

Koizumi, T. (2015). Biofuels and food security. *Renewable and Sustainable Energy Reviews, 52*, 829-841.

Kolar, C. S., Chapman, D.C., Courtenay, W. R. Jr, Housel, Ch. M., Williams, J.D., & Jennings, D. P. (2007). *Bigheaded carps: a biological synopsis and environmental risk assessment.* American Fisheries Society Special Publication 33. Bethesda, Maryland.

Konikow, L. F. (2013). Groundwater depletion in the United States (1900 – 2008). *Scientific Investigations Report, 2013 – 5079* 75.

Kornis, M. S., Sharma, S., & Jake Vander Zanden, M. (2013). Invasion success and impact of an invasive fish, round goby, in Great Lakes tributaries. *Diversity and Distributions, 19*(2), 184–198. https://doi.org/10.1111/ddi.12001

Krause-Jensen, D., & Duarte, C. M. (2016). Substantial role of macroalgae in marine carbon sequestration. *Nature Geoscience, 9*(10), 737-742.

Kreuter, U. P., Iwaasa, A. D., Theodori, G. L., Ansley, R. J., Jackson, R. B., Fraser, L. H., Naeth, M. A., McGillivray, S., & Moya, E. G. (2016). State of knowledge about energy development impacts on North American rangelands: An integrative approach. Journal of Environmental Management. *Journal of environmental management*, *180*, 1-9. https://doi.org/10.1016/j.jenvman.2016.05.007

Kunkel, K. E., Andsager, K., & Easterling, D. R. (1999). Long-term trends in extreme precipitation events over the conterminous United States and Canada. *Journal of climate*, *12*(8), 2515-2527.

Kunkel, K. E., Palecki, M., Ensor, L., Hubbard, K. G., Robinson, D., Redmond, K., & Easterling, D. (2009). Trends in twentieth-century US snowfall using a quality-controlled dataset. *Journal of Atmospheric and Oceanic Technology*, *26*(1), 33-44.

Kurtz, C. M., & Hansen, M. H. (2014). An assessment of garlic mustard in northern US forests. *Res. Note NRS-199. Newtown Square, PA: US Department of Agriculture, Forest Service, Northern Research Station. 5 p.*, *199*, 1-5.

Lacerda, L. D., & Kjerfve, B. (1999). Conservation and management of Latin American mangroves. In Salomons, W. & R. K. Turner, L. D. Lacerda, & S. Ramachandran. (Eds.). *Perspectives on integrated coastal zone management.* (pp. 183-194).

Lacerda, L. D., Conde, J. E., Kjerfve, B., Alvarez-León, R., Alarcón, C., & Polanía, J. (2002). American mangroves. In. Lacerda, L. D. (Ed.). *Mangrove ecosystems: function and management*. (pp. 1-62). Springer.

Lacerda, L. D., Soares, T. M., Costa, B. G. B., & Godoy, M. D. P. (2011). Mercury emission factors from intensive shrimp aquaculture and their relative importance to the Jaguaribe River Estuary, NE Brazil. *Bulletin of environmental contamination and toxicology*, *87*(6), 657-661.

Lacy, R. C. (2000). Considering threats to the viability of small populations using individual-based models. *Ecological Bulletins, 48,* 39–51.

Lammam, Ch. & MacIntyre, H. (2016). An Introduction to the state of poverty in Canada. Fraser Institute. fraserinstitute.org

Lammers, P., Richter, T., Lux, M., Ratsimbazafy, J., & Mantilla-Contreras, J. (2017). The challenges of community-based conservation in developing countries—A case study from Lake Alaotra, Madagascar. *Journal for Nature Conservation,40*(2), 100-112. https://doi.org/10.1016/j.jnc.2017.08.003

Lanfranconi, A., Hutton, M., Brugnoli, E., & Muniz, P. (2009). New record of the alien mollusc *Rapana venosa* (Valenciennes 1846) in the Uruguayan coastal zone of Río de la Plata. *Pan-American Journal of Aquatic Sciences, 4*(2), 216-221.

Lanfranconi, A., Brugnoli, E., & Muniz, P. (2013). Preliminary estimates of consumption rates of *Rapana venosa* (Gastropoda,Muricidae); a new threat to mollusk biodiversity in the Río de la Plata. *Aquatic Invasions 8*(4): 437-442.

Langbein, L., & Knack, S. (2010). The Worldwide governance indicators: six, one, or none? *Journal of Development Studies, 46*(2), 350–370. http://doi.org/10.1080/00220380902952399

Langdon, B., Pauchard, A., & Aguayo, M. (2010). Pinus contorta invasion in the Chilean Patagonia: Local patterns in a global context. *Biological Invasions, 12*(12), 3961–3971. https://doi.org/10.1007/s10530-010-9817-5

Lange, D. (1998). Europe's medicinal and aromatic plants: their use, trade and conservation. Traffic International.

Lapola, D.M., Martinelli, L.A., Peres, C.A., Ometto, J.P., Ferreira, M.E., Nobre, C.A., Aguiar, A.P.D., Bustamante, M.M., Cardoso, M.F., Costa, M.H., Joly, C.A, Leite, C.C., Moutinho, P., Sampaio, G., Strassburg, B.B.N., & Vieira, I.C.G. (2014). Pervasive transition of the Brazilian land-use system. *Nature climate change, 4*(1), 27-35

Larsen, J.N., Anisimov, O.A., Constable, A., Hollowed, A.B., Maynard, N., Prestrud, P., Prowse, T.D., & Stone, J.M.R. (2014). Polar regions. In Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi, Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, & L.L. White (Eds.)]. *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. (pp. 1567-1612) Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press

Laufer, G., Gobel, N., Borteiro, C., Soutullo, A., Martínez-Debat, C., & de Sá, R. O. (2018). Current status of American bullfrog, *Lithobates catesbeianus*, invasion in Uruguay and exploration of chytrid infection. *Biological Invasions*, *20*(2), 285-291.

Laurance, W. F. (2004). The perils of payoff: corruption as a threat to global biodiversity. *Trends in Ecology and Evolution, 19*(8), 0–2.

Laurance, W.F., Clements, G.R., Sloan, S., O'connell, C.S., Mueller, N.D., Goosem, M., Venter, O., Edwards, D.P., Phalan, B., Balmford, A. and Van Der Ree, R., & Arrea, I. B. (2014). A global strategy for road building. *Nature*, *513*(7517), 229–232.

Lauvset, S.K., Gruber, N., Landschützer, P., Olsen, A. & Tjiputra, J. (2015). Trends and drivers in global surface ocean pH over the past 3 decades. *Biogeosciences, 12*(5), p.1285.

Law, K. (2010). Atmospheric chemistry: More ozone over North America. *Nature, 463*(7279), 307-308.

Le Maitre, D. C., Gaertner, M., Marchante, E., Ens, E. J., Holmes, P. M., Pauchard, A., O'Farrell, P. J., Rogers, A. M., Blanchard, R., Blignaut, J., & Richardson, D. M. (2011). Impacts of invasive Australian acacias: Implications for management and restoration. *Diversity and Distributions, 17*(5), 1015–1029. https://doi.org/10.1111/j.1472-4642.2011.00816.x

Leach, M. K., & Givnish, T. J. (1996). Ecological determinants of species loss in remnant prairies. *Science*, *273*(5281), 1555-1558.

Leadley, P. W., Krug, C. B., Alkemade, R., Pereira, H. M., Sumaila U.R., Walpole, M., Marques, A., Newbold, T., Teh, L. S. L, van Kolck, J., Bellard, C., Januchowski-Hartley, S. R., & Mumby, P. J. (2014). *Progress towards the Aichi Biodiversity Targets: An assessment of biodiversity trends, policy scenarios and key actions*. Secretariat of the Convention on Biological Diversity. Montreal, Canada. Technical Series 78.

Leatherman, S., & Defraene, N. (2006). 10 Most Hurricane Vulnerable Areas. Miami, Florida.

LeBauer, D. S., & Treseder, K. K. (2008). Nitrogen limitation of net primary productivity in terrestrial ecosystems is globally distributed. *Ecology*, *89*(2), 371-379.

Lee, P. K., Brook, J. R., Dabek-Zlotorzynska, E., & Mabury, S. A. (2003). Identification of the major sources contributing to PM2. 5 observed in Toronto. *Environmental science & technology*, *37*(21), 4831-4840.

Lembrechts, J. J., Milbau, A., & Nijs, I. (2015). Trade-off between competition and facilitation defines gap colonization in mountains. *AoB Plants, 7* 1–13. https://doi.org/10.1093/aobpla/plv128

Lemos, M. C., & Agrawal, A. (2006). Environmental governance. *Annual Review of Environment and Resources 31*, 297–325.

Lenoir, S., Beaugrand, G., & Lecuyer, É. (2011). Modelled spatial distribution of marine fish and projected modifications in the North Atlantic Ocean. *Global Change Biology, 17*(1), 115–129. https://doi.org/10.1111/j.1365-2486.2010.02229.x

Lenton. T. M. (2011). Early warning of climate tipping points. Review article. Nature Climate Change 1(4), 201-209. https://doi.org/10.1038/NCLIMATE1143

Lepš, J., & Rejmánek, M. (1991). Convergence or divergence: what should we expect from vegetation succession?. *Oikos*, 261-264.

Lesica, P. (2014) Arctic-alpine plants decline over two decades in Glacier National Park, Montana, U.S.A. *Arctic Antarctic and Alpine Research, 46*(2), 327-332.

Lewis, J., Hoover, J., & MacKenzie, D. (2017). Mining and Environmental Health Disparities in Native American Communities. Current Environmental Health Reports, 4(2) 130-141. https://doi.org/10.1007/s40572-017-0140-5

Lewison, R. L., Freeman, S. A., & Crowder, L. B. (2004). Quantifying the effects of fisheries on threatened species: the impact of pelagic longlines on loggerhead and leatherback sea turtles. *Ecology letters*, *7*(3), 221-231.

Li, C., Balluz, L. S., Vaidyanathan, A., Wen, X. J., Hao, Y., & Qualters, J. R. (2016). Long-term exposure to ozone and life expectancy in the United States, 2002 to 2008. *Medicine*, *95*(7).

Liebezeit, J. R., Kendall, S. J., Brown, S., Johnson, C. B., Martin, P., McDonald, T. L., Payer, D. C., Rea, C. L., Streever, B., Wildman, A.M., & Zack, S. (2009). Influence of human development and predators on nest survival of tundra birds, Arctic Coastal Plain, Alaska. *Ecological Applications*, *19*(6), 1628-1644

Lima, L. S., Coe, M. T., Soares Filho, B. S., Cuadra, S. V., Dias, L. C. P., Costa, M. H., Lima, L. S., & Rodrigues, H. O. (2014). Feedbacks between deforestation, climate, and hydrology in the Southwestern Amazon: Implications for the provision of ecosystem services. *Landscape Ecology, 29*(2), 261–274. https://doi.org/10.1007/s10980-013-9962-1

Lindenmayer, D. B., & Likens, G. E. (2009). Adaptive monitoring: a new paradigm for long-term research and monitoring. *Trends in Ecology & Evolution, 24*(9), 482-486.

Lindsey. R. (2016). *Global impacts of El Niño and La Niña*. Retrieved from https://www.climate.gov/news-features/featured-images/global-impacts-el-niño-and-la-niña

Liu, J., Hull, V., Batistella, M., DeFries, R., Dietz, T., Fu, F., Hertel, T. W., Izaurralde, R. C., Lambin, E. F., Li, S., Martinelli, L. A., McConnell, W. J., Moran, E. F., Naylor, R., Ouyang, Z., Polenske, K. R., Reenberg, A., de Miranda Rocha, G., Simmons, C. S., Verburg, P. H., Vitousek, P. M., Zhang, F., & Zhu, C. (2013). Framing Sustainability in a Telecoupled World. *Ecology and Society, 18*(2), art26. https://doi.org/10.5751/ES-05873-180226

Liu, J., You, L., Amini, M., Obersteiner, M., Herrero, M., Zehnder, A. J., & Yang, H. (2010). A high-resolution assessment on global nitrogen flows in cropland. *Proceedings of the National Academy of Sciences, 107*(17), 8035-8040.

Liu, Y., Lee, S. K., Enfield, D. B., Muhling, B. A., Lamkin, J. T., Muller-Karger, F. E., & Roffer, M. A. (2015). Potential impact of climate change on the Intra-Americas Sea: Part-1. A dynamic downscaling of the CMIP5 model projections. *Journal of Marine Systems*, *148*, 56-69. https://doi.org/10.1016/j.jmarsys.2015.01.007

Liu, Y., Lee, S.-K., Enfield, D. B., Muhling, B. A., Lamkin, J. T., Muller-Karger, F. E., & Roffer, M. A. (2016). Past and future climate variability in the Intra-Americas Sea and its impact on the marine ecosystem and fisheries, *US CLIVAR Variations, 14*(1), 27-32.

Liverman, D. M., & Vilas, S. (2006). Neoliberalism and the Environment in Latin America. *Annual Review of Environment and Resources, 31*(1), 327–363. http://doi.org/10.1146/annurev.energy.29.102403.140729

Lôbo, D., Leão, T., Melo, F. P. L., Santos, A. M. M., & Tabarelli, M. (2011). Forest fragmentation drives Atlantic forest of northeastern Brazil to biotic homogenization. *Diversity and Distributions, 17*(2), 287–296. https://doi.org/10.1111/j.1472-4642.2010.00739.x

Loh, T. L., McMurray, S. E., Henkel, T. P., Vicente, J., & Pawlik, J. R. (2015). Indirect effects of overfishing on Caribbean reefs: sponges overgrow reef-building corals. *PeerJ*, *3*, e901. http://www.pubmedcentral.nih.gov/articlerender.fcgi?artid=4419544&tool=pmcentrez&rendertype=abstract

Lopes, A., & Piedade, M. T. F. (2014). Experimental study on the survival of the water hyacinth (Eichhornia crassipes (Mart.) Solms—Pontederiaceae) under different oil doses and times of exposure. *Environmental Science and Pollution Research*, *21*(23), 13503-13511

Lopes, R. M., Coradin, L., Beck, V., & Rimoldi, D. (2009*). Informe sobre as espécies exóticas invasoras marinhas no Brasil*. Série Biodiversidade, 33. Brasília, Ministério do Meio Ambiente.

López-Lanús, B., Roesler, I., Blanco, D. E., Petracci, P. F., Serra, M., & Zaccagnini, M. E. (2007). Bobolink (*Dolichonyx oryzivorus*) numbers and non breeding ecology in the rice fields of San Javier, Santa Fe province, Argentina. *Ornitologia Neotropical*, *18*, 493-502.

Lopez, C.B., Jewett, E.B., Dortch, Q., Walton, B.T., Hundell, H. K. (2008). *Scientific assessment of marine harmful algal blooms*. Interagency working group on harmful algal blooms, hypoxia, and human health of the joint sucommittee on ocean science and technology.

Lovingood, T., Parker, B., Smith, T. N., Canes, H., Fennell, F., Cofer, D., & Reilly, T. (2004). *Nationwide identification of hardrock mining sites*. Office of Inspector General (OIG) of the US Environmental Protection Agency, Washington, DC. Retrieved from http://www.epa.gov/oig/reports/2004/20040331-2004-p-00005.pdf, accessed on Nov. 1, 2012.

Lozano, P., Bussmann, R. W., & Küppers, M. (2006). Landslides as ecosystem disturbance-their implications and importance in Southern Ecuador. *Lyonia, 9*, 75-81.

Lugo, A. E. (1998). Mangrove forests: A tough system to invade but an easy one to rehabilitate. *Marine Pollution Bulletin 37*:427–430.

Luiz, O. J., Floeter, S. R., Rocha, L. A., & Ferreira, C. E. L. (2013). Perspectives for the lionfish invasion in the South Atlantic: Are Brazilian reefs protected by the currents? *Marine Ecology Progress Series, 485*, 1–7.

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.

Lyon, J. S., Hilliard, T. J., & Bethell, T. N. (1993). Burden of gilt: the legacy of environmental damage from abandoned mines, and what America should do about it. Mineral Policy Center.

Lyra-Neves, R. M. De, Oliveira, M. A. B., Telino-Júnior, W. R., & Santos, E. M. Dos. (2007). Comportamentos interespecíficos entre *Callithrix jacchus* (Linnaeus) (Primates, Callitrichidae) e algumas aves de Mata Atlântica, Pernambuco, Brasil. *Revista Brasileira de Zoologia, 24*(3), 709–716.

Macdonald, R. W., Barrie, L. A., Bidleman, T. F., Diamond, M. L., Gregor, D. J., Semkin, R. G., Strachan, W. M. J., Li, Y. F., Wania, F., Alaee, M., & Alexeeva, L.B. (2000). Contaminants in the Canadian Arctic: 5 years of progress in understanding sources, occurrence and pathways. *Science of the Total Environment, 254*(2), 93-234.

MacDougall, A. S., & Turkington, R. (2005). Are invasive species the drivers or passengers of change in degraded ecosystems? *Ecology, 86*(1), 42–55.

Machado, W., & Lacerda, L. D. (2004). Overview of the biogeochemical controls and concerns with trace metal accumulation in mangrove sediments. In. Lacerda, L. D., R. E. Santelli, E. Duursma, & J. J. Abrao (Eds). (pp. 319-334). *Environmental geochemistry in tropical and subtropical environments*. Springer, Berlin, Heidelberg.

Mack, R. N., & Lonsdale, W. M. (2001). Humans as global plant dispersers: getting more than we bargained for: current introductions of species for aesthetic purposes present the largest single challenge for predicting which plant immigrants will become future pests. *BioScience, 51*(2), 95-102. https://doi.org/10.1641/0006-3568(2001)051%5B0095:HAGPDG%5D2.0.CO;2

Mack, R. N., & Thompson, J. N. (1982). Evolution in steppe with few large, hooved mammals. *The American Naturalist, 119*(6), 757-773.

Mack, R., Simberloff, D., Lonsdale, W., Evans, H., Clout, M., & Bazzaz, F. (2000). Biotic invasions: causes, epidemiology, global consequences, and control. *Ecological Applications, 10*(3), 689–710.

Madrigal-González, J., Cea, A. P., Sánchez-Fernández, L. A., Martínez-Tillería, K. P., Calderón, J. E., & Gutiérrez, J. R. (2013). Facilitation of the non-native annual plant Mesembryanthemum crystallinum (Aizoaceae) by the endemic cactus *Eulychnia acida* (Cactaceae) in the Atacama Desert. *Biological Invasions, 15*(7), 1439–1447. https://doi.org/10.1007/s10530-012-0382-y

Maffi, L. (2005). Linguistic, cultural, and biological diversity. *Annu. Rev. Anthropol.*, *34*, 599-617.

Magalhães, J. L. L., Lopes, M. A., & de Queiroz, H. L. (2015). Development of a Flooded Forest Anthropization Index (FFAI) applied to Amazonian areas under pressure from different human activities. *Ecological indicators*, *48*, 440-447.

Magrin, G. O., Marengo, J. A., Boulanger, J. -P., Buckeridge, M. S., Castellanos, E., Poveda, G., Scarano, F. R., & Vicuña, S. (2014). Central and South America. In Barros, V. R., C. B. Field, D. J. Dokken, M. D. Mastrandrea, K. J. Mach, T. E. Bilir, M. Chatterjee, K. L. Ebi, Y. O. Estrada, R. C. Genova, B. Girma, E. S. Kissel, A. N. Levy, S. MacCracken, P. R. Mastrandrea, & L. L. White (Eds.). *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change* (pp. 1499-1566). Cambridge, United Kingdom and New York, NY, USA: Cambridge University Press.

Magris, R. A., & Barreto, R. (2010). Mapping and assessment of protection of mangrove habitats in Brazil. *Pan-American Journal of Aquatic Sciences*, *5*(4), 546-556.

Mahaffey, K. R., & Mergler, D. (1998). Blood levels of total and organic mercury in residents of the upper St. Lawrence River basin, Quebec: association with age, gender, and fish consumption. *Environmental Research*, *77*(2), 104-114.

Maki, A. W. (1992). Of measured risks: The environmental impacts of the Prudhoe Bay, Alaska, oil field. *Environmental Toxicology and Chemistry*, *11*(12), 1691-1707.

Malcolm, J. R., Liu, C., Neilson, R. P., Hansen, L., & Hannah, L. E. E. (2006). Global warming and extinctions of endemic species from biodiversity hotspots. *Conservation biology*, *20*(2), 538-548.

Malhi, Y., & Wright, J. (2004). Spatial patterns and recent trends in the climate of tropical rainforest regions. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, *359*(1443), 311-329.

Malm, O. (1998). Gold mining as a source of mercury exposure in the Brazilian Amazon. *Environmental Research*, *77*(2), 73-78.

Mann, R., Occhipinti, A., & Harding, J. M. (Eds.). (2004). *Alien species alert: Rapana venosa (veined whelk)*. International Council for the Exploration of the Sea. Cooperative Research Report, 264.

Mantyka-Pringle, C. S., Martin, T. G., & Rhodes, J. R. (2012). Interactions between climate and habitat loss effects on biodiversity: a systematic review and meta-analysis. *Global Change Biology, 18*(4), 1239-1252.

Manuschevich, D. (2016). Neoliberalization of forestry discourses in Chile. *Forest Policy and Economics*, *69*, 21-30. http://doi.org/10.1016/j.forpol.2016.03.006

Manzello, D. P. (2010). Ocean acidification hotspots: Spatiotemporal dynamics of the seawater CO_2 system of eastern Pacific coral reefs. *Limnology and Oceanography*, *55*(1), 239-248.

Marchese, C. (2015). Biodiversity hotspots: A shortcut for a more complicated concept. *Global Ecology and Conservation, 3*, 297-309.

Marengo, J. A. (2004). Interdecadal variability and trends of rainfall across the Amazon basin. *Theoretical and applied climatology*, *78*(1-3), 79-96. http://link.springer.com/10.1007/s00704-004-0045-8. Accessed 6 Jun 2016.

Marengo, J. A., & Espinoza, J. C. (2016). Extreme seasonal droughts and floods in Amazonia: causes, trends and impacts. *International Journal of Climatology*, *36*(3), 1033-1050.

Marengo, J. A., Jones, R., Alves, L. M., & Valverde, M. C. (2009). Future change of temperature and precipitation extremes in South America as derived from the PRECIS regional climate modeling system. *International Journal of Climatology: A Journal of the Royal Meteorological Society*, *29*(15), 2241-2255.

Marengo, J. A., Pabón, J. D., Díaz, A., Rosas, G., Montealegre, E., Villacis, M., Solman, & Rojas, M. (2011). Climate Change : evidence and future scenarios for the Andean region. In Herzog, S. K., R. Martinez, P. M. Jørgensen, & H. Tiessen (Eds.). *Climate Change and biodiversity in the tropical Andes*. (pp.110–127). Inter-American Institute for Global Change Research (IAI) and Scientific Committee on Problems of the Environment (SCOPE).

Marengo, J. A., Chou, S. C., Kay, G., Alves, L. M., Pesquero, J. F., Soares, W. R., Santos, D. C. Lyra, A. A., Sueiro, G., Betts, R., Chagas, D. J., Gomes, J. L., Bustamante, J. F., & Tavares, P. (2012). Development of regional future climate change scenarios in South America using the Eta CPTEC/HadCM3 climate change projections: climatology and regional analyses for the Amazon, São Francisco and the Paraná River basins. *Climate Dynamics*, *38*(9-10), 1829-1848. https://doi.org/10.1007/s00382-011-1155-5

Marengo, J. A., Nunes, L. H., Souza, C. R. G., Harari, J., Muller-Karger, F., Greco, R., Hosokawa, E. K., Tabuchi, E. K., Merrill, S. B., Reynolds, C. J., Pelling, M., Alves, L. M., Aragão, L. E., Chou, S. C., Moreira, F., Paterson, S., Lockman, J. T., & Gray, A. G. (2017). A globally deployable strategy for co-development of adaptation preferences to sea-level rise: the public participation case of Santos, Brazil. *Natural Hazards, 88*(1), 39–53. https://doi.org/10.1007/s11069-017-2855-x

Maret T.R. & MacCoy D.E. (2002). Fish assemblages and environmental variables associated with hard rock mining in the Coeur d'Alene River Basin, Idaho. *Transactions of the American Fisheries Society, 131*, 865-884, https://doi.org/10.1577/1548-8659(2002)131<0865:FAAEVA>2.0.CO;2

Maret, T. R., Cain, D. J., MacCoy, D. E., & Short, T.M. (2003). Response of benthic invertebrate assemblages to metal exposure and bioaccumulation associated with hard-rock mining in northwestern streams, USA. *Journal of the North American Benthological Society, 22*, 598-620.

Marshall, V. M., Lewis, M. M., & Ostendorf, B. (2012). Buffel grass (Cenchrus ciliaris) as an invader and threat to biodiversity in arid environments: A review. Journal of Arid Environments, 78, 1–12. https://doi.org/10.1016/j.jaridenv.2011.11.005

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.

Martin, P. H., Canham, C. D., & Marks, P. L. (2009). Why forests appear resistant to exotic plant invasions: Intentional introductions, stand dynamics, and the role of shade tolerance. *Frontiers in Ecology and the Environment, 7*(3), 142–149. https://doi.org/10.1890/070096

Martínez, A. S., Masciocchi, M., Villacide, J. M., Huerta, G., Daneri, L., Bruchhausen, A., ... & Corley, J. C. (2013). Ashes in the air: the effects of volcanic ash emissions on plant–pollinator relationships and possible consequences for apiculture. *Apidologie*, *44*(3), 268-277.

Martínez, M.O., Napolitano, D.A., MacLennan, G.J., O'Callaghan, C., Ciborowski, S., & Fabregas, X. (2007). Impacts of petroleum activities for the Achuar people of the Peruvian Amazon: summary of existing evidence and research gaps. *Environmental Research Letters, 2*(4), p.045006.

Martínez-Alier, J. (2014). The environmentalism of the poor. *Geoforum,* 54, 239–241. http://doi.org/10.1016/j.geoforum.2013.04.019

Martínez, V., & Castillo, O. L. (2016). The political ecology of hydropower: Social justice and conflict in Colombian hydroelectricity development. *Energy Research and Social Science, 22*, 69–78.

Martínez-López, E., Espín, S., Barbar, F., Lambertucci, S.A., Gómez-Ramírez, P., & García-Fernández, A.J. (2015). Contaminants in the southern tip of South America: Analysis of organochlorine compounds in feathers of avian scavengers from Argentinean Patagonia. *Ecotoxicology and environmental safety, 115*, 83-92.

Martins, C. R. (2006). Caracterização e manejo da gramínea Melinis minutiflora P. Beauv.(capim-gordura): uma espécie invasora do cerrado. *Brasília, Universidade de Brasília. Tese.*

Martins, C.C., Bícego, M.C., Mahiques, M.M., Figueira, R.C., Tessler, M.G., & Montone, R.C. (2011). Polycyclic aromatic hydrocarbons (PAHs) in a large South American industrial coastal area (Santos Estuary, Southeastern Brazil): sources and depositional history. *Marine pollution bulletin, 63*(5), 452-458.

Marubini, F., & Davies, P.S. (1996). Nitrate increases zooxanthellae population density and reduces skeletogenesis in corals. *Marine Biology, 127*(2), pp.319-328.

Mascia, M. B., Pailler, S., Krithivasan, R., Roshchanka, V., Burns, D., Mlotha, M. J., Murray, D. R., & Peng, N. (2014). Protected area downgrading, downsizing, and degazettement (PADDD) in Africa, Asia, and Latin America and the Caribbean, 1900-2010. *Biological Conservation, 169*, 355–361. https://doi.org/10.1016/j.biocon.2013.11.021

Masek, J. G., Cohen, W. B., Leckie, D., Wulder, M. A., Vargas, R., de Jong, B., Healey, S., Law, B., Birdsey, R., Houghton, R. A., Mildrexler, D., Goward, S., & Smith, W. B. (2011). Recent rates of forest harvest and conversion in North America. *Journal of Geophysical Research, 116*(G4), G00K03. https://doi.org/10.1029/2010JG001471

Masi, E., Pino, F. A., Santos, M. das G. S., Genehr, L., Albuquerque, J. O. M., Bancher, A. M., & Alves, J. C. M. (2010). Socioeconomic and environmental risk factors for urban rodent infestation in Sao Paulo, Brazil. *Journal of Pest Science, 83*(3), 231–241. https://doi.org/10.1007/s10340-010-0290-9

Mathis, J.T., Cross, J.N., Evans, W., & Doney, S.C. (2015). Ocean acidification in the surface waters of the Pacific-Arctic boundary regions. *Oceanography, 28*(2), 122-135.

Maxim, L., Spangenberg, J. H., & O'Connor, M. (2009). An analysis of risks for biodiversity under the DPSIR framework. *Ecological Economics, 69*(1), 12-23.

McClintock, J. B., Angus, R. A., Mcdonald, M. R., Amsler, C. D., Catledge, S. A., & Vohra, Y. K. (2009). Rapid dissolution of shells of weakly calcified Antarctic benthic macroorganisms indicates high vulnerability to ocean acidification. *Antarctic Science*, *21*(5), 449-456.

McConnell, J. R., & Edwards, R. (2008). Coal burning leaves toxic heavy metal legacy in the Arctic. *Proceedings of the National Academy of Sciences*, *105*(34), 12140-12144.

McDonald, R.I., Weber, K., Padowski, J., Flörke, M., Schneider, C., Green, P.A., Gleeson, T., Eckman, S., Lehner, B., Balk, D., & Boucher, T. (2014). Water on an urban planet: Urbanization and the reach of urban water infrastructure. *Global Environmental Change, 27*, 96-105.

McEwan, R. W., Dyer, J. M., & Pederson, N. (2011). Multiple interacting ecosystem drivers: toward an encompassing hypothesis of oak forest dynamics across eastern North America. *Ecography, 34*(2), 244-256.

McGraw, J. B., Souther, S., & Lubbers, A. E. (2010). Rates of harvest and compliance with regulations in natural populations of American ginseng (Panax quinquefolius L.). *Natural Areas Journal*, *30*(2), 202-210.

McGuire, A. D., Chapin Iii, F. S., Walsh, J. E., & Wirth, C. (2006). Integrated regional changes in arctic climate feedbacks: implications for the global climate system. *Annu. Rev. Environ. Resour.*, *31*, 61-91.

McLinden, C.A., Fioletov, V., Krotkov, N.A., Li, C., Boersma, K.F. & Adams, C. (2015). A decade of change in NO2 and SO2 over the Canadian oil sands as seen from space. *Environmental science & technology, 50*(1), pp.331-337.

McNeil, B. I., & Matear, R. J. (2008). Southern Ocean acidification: A tipping point at 450-ppm atmospheric CO_2. *Proceedings of the National Academy of Sciences*, *105*(48), 18860-18864. https://doi.org/10.1073/pnas.0806318105

McNeish, J. A. (2013). Extraction, protest and indigeneity in Bolivia: the TIPNIS effect. *Latin American and Caribbean Ethnic Studies*, *8*(2), 221-242.

McPhearson T., Auch R., & Alberti M. (2013) Regional assessment of North America: urbanization trends, biodiversity patterns, and ecosystem services. In: Elmqvist, Th., M. Fragkias, J. Goodness, , B. Güneralp, P.J. Marcotullio, R.I. McDonald, S. Parnell, M. Schewenius, M. Sendstad, M., & K.C. Seto.(Eds.) Urbanization, Biodiversity and Ecosystem Services: Challenges and Opportunities. (PP. 279-286). Springer, Dordrecht

Meissner, K. J., Lippmann, T., & Gupta, A. S. (2012). Large-scale stress factors affecting coral reefs: open ocean sea surface temperature and surface seawater aragonite saturation over the next 400 years. *Coral Reefs, 31*(2), 309-319.

Mekonnen, M. M., Pahlow, M., Aldaya, M. M., Zarate, E., & Hoekstra, A. Y. (2015). Sustainability, efficiency and equitability of water consumption and pollution in Latin America and the Caribbean. *Sustainability*, *7*(2), 2086-2112.

Melillo, J. M., Richmond T. (T.C.), & Yohe, G. W. (Eds.). (2014). *Climate Change Impacts in the United States: The Third National Climate Assessment*. U.S. Global Change Research Program

Mendelsohn, R., Emanuel, K., Chonabayashi, S., & Bakkensen, L. (2012). The impact of climate change on global tropical cyclone damage. *Nature climate change, 2*(3), 205-209.

Mendes, D., & Marengo, J. A. (2010). Temporal downscaling: a comparison between artificial neural network and autocorrelation techniques over the Amazon Basin in present and future climate change scenarios. *Theoretical and Applied Climatology*, *100*(3-4), 413-421.

Méndez-Lázaro, P., Muller-Karger, F. E., Otis, D., McCarthy, M. J., & Rodríguez, E. (2018). A heat vulnerability index to improve urban public health management in San Juan, Puerto Rico. *International journal of biometeorology*, *62*(5), 709-722. https://doi.org/10.1007/s00484-017-1319-z

Méndez-Lázaro, P., Nieves-Santiango, A., & Miranda-Bermúdez, J. (2014). Trends in total rainfall, heavy rain events, and number of dry days in San Juan, Puerto Rico, 1955-2009. *Ecology and Society, 19*(2). https://www.jstor.org/stable/26269555

Mendonça, A. F., Armond, T., Camargo, A. C. L., Camargo, N. F., Ribeiro, J. F., Zangrandi, P.L., & Vieira, E.M. (2015). Effects of an extensive fire on arboreal small mammal populations in a neotropical savanna woodland. *ournal of Mammalogy*, *96*(2), 368-379.

Merritt, D. M., & Leroy Poff, N. (2010). Shifting dominance of riparian Populus and Tamarix along gradients of flow alteration in western North American rivers. *Ecological Applications, 20*(1), 135–152. https://doi.org/10.1890/08-2251.1

Meserve, P. L., Kelt, D. A., Gutiérrez, J. R., Previtali, M. A., & Milstead, W. B. (2016). Biotic interactions and community dynamics in the semiarid thorn scrub of Bosque Fray Jorge National Park, north-central Chile: A paradigm revisited. *Journal of Arid Environments, 126*, 81–88. https://doi.org/10.1016/j.jaridenv.2015.08.016

Meyfroidt, P., Carlson, K. M., Fagan, M. E., Gutiérrez-Vélez, V. H., Macedo, M. N., Curran, L. M., DeFries, R. S., Dyer, G. A., Gibbs, H. K., Lambin, E. F., Morton, D. C., & Robiglio, V. (2014). Multiple pathways of commodity crop expansion in tropical forest landscapes. *Environmental Research Letters, 9*(7), 74012. https://doi.org/10.1088/1748-9326/9/7/074012

Michalak, A. M., Anderson, E. J., Beletsky, D., Boland, S., Bosch, N. S., Bridgeman, T. B., Chaffin, J. D., Cho, K., Confesor, R., Daloğlu, I., & DePinto, J.V. (2013). Record-setting algal bloom in Lake Erie caused by agricultural and meteorological trends consistent with expected future conditions. *Proceedings of the National Academy of Sciences, 110*(16), 6448-6452.

Michener, W. K., Blood, E. R., Bildstein, K. L., Brinson, M. M., & Gardner, L. R. (1997). Climate change, hurricanes and tropical storms, and rising sea level in coastal wetlands. *Ecological Applications*, *7*(3), 770-801.

Middeldorp, N., Morales, C., & van der Haar, G. (2016). Social mobilisation and violence at the mining frontier: The case of Honduras. *Extractive Industries and Society,* 3(4), 930–938. http://doi.org/10.1016/j.exis.2016.10.008

Miglioranza, K.S., Gonzalez, M., Ondarza, P.M., Shimabukuro, V.M., Isla, F.I., Fillmann, G., Aizpún, J.E., & Moreno, V.J. (2013). Assessment of Argentinean Patagonia pollution: PBDEs, OCPs and PCBs in different matrices from the Río Negro basin. *Science of the Total Environment, 452*, 275-285.

Millar, C. I., & Stephenson, N. L. (2015). Temperate forest health in an era of emerging megadisturbance. *Science, 349*(6250), 823-826.

Millennium Ecosystem Assessment (MEA). (2005). Ecosystems and human well-being: synthesis. Island Press, Washington, DC. World Resources Institute.

Miller, K. A., Aguilar-Rosas, L. E., & Pedroche, F. F. (2011). A review of non-native seaweeds from California, USA and Baja California, Mexico = Reseña de algas marinas no nativas de California, EUA y Baja California, México. *Hidrobiológica, 21*(3), 365–379. Retrieved from http://www.vliz.be/en/imis?module=ref&refid=218235

Miller, M. J. (2011). Persistent illegal logging in Costa Rica: the role of corruption among forestry regulators. *The Journal of Environment & Development*, *20*(1), 50-68. http://doi.org/10.1177/1070496510394319

Mills, D. J., Westlund, L., de Graaf, G., Kura, Y., Willman, R., & Kelleher, K. (2011). Under-reported and undervalued: small-scale fisheries in the developing world. *Small-scale fisheries management: frameworks and approaches for the developing world*, 1-15.

Miloslavich, P., Martín, A., Klein, E., Díaz, Y., Lasso, C. A., Cárdenas, J. J., & Lasso-Alcalá, O. M. (2011). Biodiversity and conservation of the estuarine and marine ecosystems of the Venezuelan Orinoco Delta. In Ecosystems Biodiversity. InTech. Retrieved from http://cdn.intechweb.org/pdfs/25323.pdf

Mistry, J., Berardi, A., Andrade, V., Krahô, T., Krahô, P., & Leonardos, O. (2005). Indigenous fire management in the cerrado of Brazil: the case of the Krahô of Tocantíns. *Human ecology*, *33*(3), 365-386.

Mitchell, R. T., Blagbrough, H. P., & VanEtten, R. C. (1953). The effects of DDT upon the survival and growth of nestling songbirds. *The Journal of Wildlife Management*, *17*(1), 45-54.

Mitsch, W. J., & Hernandez, M. E. (2013). Landscape and climate change threats to wetlands of North and Central America. *Aquatic sciences, 75*(1), 133-149.

Modernel, P., Rossing, W. A., Corbeels, M., Dogliotti, S., Picasso, V., & Tittonell, P. (2016). Land use change and ecosystem service provision in Pampas and Campos grasslands of southern South America. *Environmental Research Letters, 11*(11), 113002.

Mol, J. H., Ramlal, J. S., Lietar, C., & Verloo, M. (2001). Mercury contamination in freshwater, estuarine, and marine fishes in relation to small-scale gold mining in Suriname, South America. *Environmental Research*, *86*(2), 183-197.

Molinos-Senante, M., & Sala-Garrido, R. (2015). The impact of privatization approaches on the productivity growth of the water industry: A case study of Chile. Environmental *Science & Policy, 50*, 166–179. http://dx.doi.org/10.1016/j.envsci.2015.02.015

Moller, H., Berkes, F., Lyver, P. O. B., & Kislalioglu, M. (2004). Combining science and traditional ecological knowledge: monitoring populations for co-management. *Ecology and society*, 9(3).

Morales-Hidalgo, D., Oswalt, S. N., & Somanathan, E. (2015). Status and trends in global primary forest, protected areas, and areas designated for conservation of biodiversity from the Global Forest Resources Assessment 2015. *Forest Ecology and Management*, *352*, 68-77.

Morang, A., Rosati, J. D., & King, D. B. (2013). Regional sediment processes, sediment supply, and their impact on the Louisiana coast. *Journal of Coastal Research*, *63*(sp1), 141-165.

Morell, V. (2017). World's most endangered marine mammal down to 30. *Science, 355*(6325): 558-559. hdl.handle.net/10.1126/science.355.6325.558

Moreno, T. A., & Huber-Sannwald, E. (2011). Impacts of drought on agriculture in Northern Mexico. In *Coping with global environmental change, disasters and security* (pp. 875-891). Springer, Berlin, Heidelberg.

Moreno-Mateos, D., Barbier, E. B., Jones, P. C., Jones, H. P., Aronson, J., López-López, J. A., McCrackin, M. L., Meli, P., Montoya, D., & Rey Benayas, J. M. (2017). Anthropogenic ecosystem disturbance and the recovery debt. *Nature Communications, 8,* 14163. https://doi.org/10.1038/ncomms14163

Morris, J. A., & Akins, J. L. (2009). Feeding ecology of invasive lionfish (*Pterois volitans*) in the Bahamian archipelago. *Environmental Biology of Fishes*, *86*(3), 389.

Morton, D. C., Le Page, Y., DeFries, R., Collatz, G. J., & Hurtt, G. C. (2013). Understorey fire frequency and the fate of burned forests in southern Amazonia. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, *368*(1619), 20120163.

Moss, C. (2008). Patagonia. A cultural history. Oxford: Signal Books.

Mueller, N. D., Gerber, J. S., Johnston, M., Ray, D. K., Ramankutty, N., & Foley, J. A. (2012). Closing yield gaps through nutrient and water management. *Nature*, *490*(7419), 254. http://dx.doi.org/10.1038/nature11420

Muhling, B. A., Lamkin, J. T., Alemany, F., García, A., Farley, J., Ingram, G. W., Berastegui, D. A., Reglero, P., & Carrion, R. L. (2017). Reproduction and larval biology in tunas, and the importance of restricted area spawning grounds. *Reviews in Fish Biology and Fisheries, 27*(4), 697–732. https://doi.org/10.1007/s11160-017-9471-4

Muhling, B. A., Liu, Y., Lee, S. K., Lamkin, J. T., Roffer, M. A., Muller-Karger, F., & Walter, J. F. (2015). Potential impact of climate change on the Intra-Americas Sea: Part 2. Implications for Atlantic bluefin tuna and skipjack tuna adult and larval habitats. *Journal of Marine Systems, 148,* 1–13. https://doi.org/10.1016/j.jmarsys.2015.01.010

Mueller, S. A., Anderson, J. E., & Wallington, T. J. (2011). Impact of biofuel production and other supply and demand factors on food price increases in 2008. *Biomass and Bioenergy, 35*(5), 1623-1632.

Muller-Karger, F. E., Rueda-Roa, D., Chavez, F. P., Kavanaugh, M. T., & Roffer, M. A. (2017). Megaregions among the large marine ecosystems of the Americas. *Environmental development*, *22*, 52-62. http://dx.doi.org/10.1016/j.envdev.2017.01.005

Muller-Karger, F. E., Smith, J. P., Werner, S., Chen, R., Roffer, M., Liu, Y., Muhling, B., Lindo-Atichati, D., Lamkin, J., Cerdeira-Estrada, S. & Enfield, D.B. (2015). Natural variability of surface oceanographic conditions in the offshore Gulf of Mexico. *Progress in Oceanography*, *134*, 54-76. https://doi.org/10.1016/j.pocean.2014.12.007

Munden, J. G. (2013). *Reducing negative ecological impacts of capture fisheries through gear modification.* Masters Thesis Biology Department, Faculty of Science, Memorial University of Newfoundland, St. Johns, Newfoundland and Labrador

Muñoz, A. A., Celedon-Neghme, C., Cavieres, L. A., & Arroyo, M. T. K. (2005). Bottom-up effects of nutrient availability on flower production, pollinator visitation, and seed output in a high-Andean shrub. *Oecologia, 143*(1), 126–135. https://doi.org/10.1007/s00442-004-1780-3

Munson, S. M., Belnap, J., & Okin, G. S. (2011). Responses of wind erosion to climate-induced vegetation changes on the Colorado Plateau. *Proceedings of the National Academy of Sciences, 108*(10), 3854-3859.

Munthe, J., Bodaly, R. A., Branfireun, B. A., Driscoll, C. T., Gilmour, C. C., Harris, R., Horvat, M., Lucotte, M., & Malm, O. (2007). Recovery of mercury-contaminated fisheries. *AMBIO: A Journal of the Human Environment*, *36*(1), 33-44. https://doi.org/10.1579/0044-7447(2007)36[33:ROMF]2.0.CO;2

Murcia, C., & Guariguata, M. R. (2014). *La restauración ecológica en Colombia: tendencias, necesidades y oportunidades* (Vol. 107). CIFOR.

Murphy, G. E., & Romanuk, T. N. (2014). A meta-analysis of declines in local species richness from human disturbances. *Ecology and evolution*, *4*(1), 91-103. https://doi.org/10.1002/ece3.909

Myers-Smith, I. H., Forbes, B. C., Wilmking, M., Hallinger, M., Lantz, T., Blok, D., ... & Boudreau, S. (2011). Shrub expansion in tundra ecosystems: dynamics, impacts and research priorities. *Environmental Research Letters*, *6*(4), 045509. Doi:10.1088/1748-9326/6/4/045509

Myers, N., & Kent, J. (2003). New consumers: the influence of affluence on the environment. *Proceedings of the National Academy of Sciences*, *100*(8), 4963-4968. https://www.ncbi.nlm.nih.gov/pmc/articles/PMC153663/

Nascimento Jr, W. R., Souza-Filho, P. W. M., Proisy, C., Lucas, R. M., & Rosenqvist, A. (2013). Mapping changes in the largest continuous Amazonian mangrove belt using object-based classification of multisensor satellite imagery. *Estuarine, Coastal and Shelf Science*, *117*, 83-93.

Nasi, R., Taber, A., & Van Vliet, N. (2011). Empty forests, empty stomachs? Bushmeat and livelihoods in the Congo and Amazon Basins. *International Forestry Review*, *13*(3), 355-368.

Nasi, R., Brown, D., Wilkie, D., Bennett, E., Tutin, C., Van Tol, G., & Christophersen, T. (2008). Conservation and use of wildlife-based resources: the bushmeat crisis. Secretariat of the Convention on Biological Diversity, Montreal, and Center for International Forestry Research (CIFOR), Bogor. *Technical Series*, *33*. http://www.cbd.int/doc/publications/cbd-ts-33-en.pdf

Nations, F. A. O. (2013). Current world fertilizer trends and outlook to 2015; Rome.

Nations, F. A. O. (2017). World fertilizer trends and outlook to 2020. Summary report; Rome.

Nations, U. (2010). *Natural hazards, unnatural disasters: the economics of effective prevention*. The World Bank.

Neff, J. C., A. P. Ballantyne, G. L. Farmer, N. M. Mahowald, J. L. Conroy, C. C. Landry, J. T. Overpeck, T. H. Painter, C. R. Lawrence, and R. L. Reynolds. (2008). Increasing eolian dust deposition in the western United States linked to human activity. *Nature Geoscience,* 1, 189–195.

Nellemann, C., & Corcoran, E. (2010). *Dead planet, living planet: biodiversity and ecosystem restoration for sustainable development*. United Nations Environment Programme (UNEP).

Nelson, E., G. Mendoza, J. Regetz, S. Polasky, H. Tallis, D. R. Cameron, K. M. A. Chan, G. C. Daily, J. Goldstein, P. M. Kareiva, E. Lonsdorf, R. Naidoo, T. H. Ricketts, & M. R. Shaw. (2009). Modeling multiple ecosystem services, biodiversity conservation, commodity production, and tradeoffs at landscape scales. *Frontiers in Ecology and the Environment,* 7, 4–11.

Nelson, G. C., Dobermann, A., Nakicenovic, N., & O'Neill, B. C. (2006). Anthropogenic drivers of ecosystem change: an overview. *Ecology and Society*, 11(2).

Neme, L. (2016). In Latin America, Illegal Wildlife Trade Explodes. *Huffington Post*. Retrieved from http://www.huffingtonpost.com/laurel-neme/in-latin-america-illegal-_b_8492020.html

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

Newbold, T., L. N. Hudson, S. L. Hill, S. Contu, I. Lysenko, R. A. Senior, L. Börger, D. J. Bennett, A. Choimes, B. Collen, J. Day, A. De Palma, S. Dıáz, S. Echeverria-Londoño, M. J. Edgar, A. Feldman, M. Garon, M. L. K. Harrison, T. Alhusseini, D. J. Ingram, Y. Itescu, J. Kattge, V. Kemp, L. Kirkpatrick, M. Kleyer, D. Laginha Pinto Correia, C. D. Martin, S. Meiri, M. Novosolov, Y. Pan, H. R. P. Phillips, D. W. Purves, A. Robinson, J. Simpson, S. L. Tuck, E. Weiher, H. J. White, R. M. Ewers, G. M. Mace, J. P. Scharlemann, and A. Purvis. (2015). Global effects of land use on local terrestrial biodiversity. *Nature*, *520*(7545), 45.

Newman, M. E., McLaren, K. P., & Wilson, B. S. (2014a.) Long-term socio-economic and spatial pattern drivers of land cover change in a Caribbean tropical moist forest, the Cockpit Country, Jamaica. Agriculture, *Ecosystems & Environment*, 186, 185-200.

Newman, M.E., McLaren, K.P., & Wilson, B.S. (2014b.) Assessing deforestation and fragmentation in a tropical moist forest over 68 years; the impact of roads and legal protection in the Cockpit Country, Jamaica. *Forest Ecology and Management*, 315, pp.138-152.

Newman, S. P., Meesters, E. H., Dryden, C. S., Williams, S. M., Sanchez, C., Mumby, P. J., & Polunin, N. V. (2015). Reef flattening effects on total richness and species responses in the Caribbean. *Journal of Animal Ecology*, *84*(6), 1678-1689.

Nielsen, J. L. (2014). Recovering the Interior Least Tern: A Fresh Approach to De-listing a Species. *Bird Conservation*, 6–11.

NOAA. (2016). *Status of Stocks 2015: Annual Report to Congress on the Status of U.S. Fisheries.* National Oceanic and Atmospheric Administration. Retrieved from https://digital.library.unt.edu/ark:/67531/metadc948829/

Nóbrega, G. N., Otero, X. L., Macías, F., & Ferreira, T. O. (2014). Phosphorus geochemistry in a Brazilian semiarid mangrove soil affected by shrimp farm effluents. *Environmental Monitoring and Assessment,* 186, 5749-5762.

Nori, J., Urbina-Cardona, J. N., Loyola, R. D., Lescano, J. N., & Leynaud, G. C. (2011). Climate change and American Bullfrog invasion: What could we expect in South America? *PLoS ONE*, 6(10), 1–8. https://doi.org/10.1371/journal.pone.0025718

Notaro, M., Mauss, A., & Williams, J. W. (2012). Projected vegetation changes for the American Southwest : combined dynamic modeling and bioclimatic-envelope approach. *Ecological Applications,* 22, 1365–1388.

Notaro, M., Bennington, V., & Lofgren, B. (2015a.) Dynamical downscaling-based projections of Great Lakes' water levels. *Journal of Climate*, *28*(24), 9721-9745.

Notaro, M., Bennington, V., & Vavrus, S. (2015b). Dynamically downscaled projections of lake-effect snow in the Great Lakes basin. *Journal of Climate,* 28, 1661–1684.

Nunes, L. (2011). An Overview of Recent Natural Disasters in South America. Bulletin des séances- *Académie royale des sciences d'outre-mer,* 57, 409-425.

Nungesser, M., C. Saunders, C. Coronado-Molina, J. Obeysekera, J. Johnson, C. McVoy, & B. Benscoter. (2015). Potential Effects of Climate Change on Florida's Everglades. *Environmental Management*, 55, 824–835.

Nurse, L.A., R.F. McLean, J. Agard, L.P. Briguglio, V. Duvat-Magnan, N. Pelesikoti, E. Tompkins, & A. Webb. (2014). Small islands. In: Barros, V.R., C.B. Field, D.J. Dokken, M.D. Mastrandrea, K.J. Mach, T.E. Bilir, M. Chatterjee, K.L. Ebi,Y.O. Estrada, R.C. Genova, B. Girma, E.S. Kissel, A.N. Levy, S. MacCracken, P.R. Mastrandrea, & L.L. White (Eds.) *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects*. *Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. (pp. 1613-165). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA

Oken, E., Wright, R.O., Kleinman, K.P., Bellinger, D., Amarasiriwardena, C.J., Hu, H., Rich-Edwards, J.W., & Gillman, M.W. (2005). Maternal fish consumption, hair mercury, and infant cognition in a US cohort. *Environmental health perspectives*, *113*(10), 1376.

Olivares, I., Svenning, J. C., van Bodegom, P. M., & Balslev, H. (2015). Effects of warming and drought on the vegetation and plant diversity in the Amazon basin. *The Botanical Review*, *81*(1), 42-69.

Oliver, T. H., & Morecroft, M. D. (2014). Interactions between climate change and land use change on biodiversity: attribution problems, risks, and opportunities. *Wiley Interdisciplinary Reviews: Climate Change*, 5(3), 317-335.

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., Píriz, M.L., Scarabino, F., Spivak, E., & 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.

Orta-Martínez, M, Napolitano, D. A., MacLennan, G. J., O'Callaghan, C., Ciborowski, S., & Fabregas, X. (2007). Impacts of petroleum activities for the Achuar people of the Peruvian Amazon: summary of existing evidence and research gaps. *Environmental Research Letters*, 2(4), 045006.

Orta-Martínez, M., & Finer, M. (2010). Oil frontiers and indigenous resistance in the Peruvian Amazon. *Ecological Economics,* 70, 207–218.

Paavola, J. (2007). Institutions and environmental governance: A reconceptualization. *Ecological Economics*, 63(1), 93–103. http://doi.org/10.1016/j.ecolecon.2006.09.026

Pacyna, E. G., Pacyna, J. M., Steenhuisen, F., & Wilson, S. (2006). Global anthropogenic mercury emission inventory for 2000. *Atmospheric environment*, *40*(22), 4048-4063.

Padoch, C., Brondizio, E., Costa, S., Pinedo-Vasquez, M., Sears, R. R., & Siqueira, A. (2008). Urban forest and rural cities: multi-sited households, consumption patterns, and forest resources in Amazonia. *Ecology and Society*, *13*(2).

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. *Proceedings of the National Academy of Sciences*, *107*(40), 17125-17130.

Palmer, M. A., Moglen, G. E., Bockstael, N. E., Brooks, S., Pizzuto, J. E., Wiegand, C., & VanNess, K. (2002). The ecological consequences of changing land use for running waters, with a case study of urbanizing watersheds in Maryland. *Science*, *287*, 1170-1774.

Pandolfi, J. M., Connolly, S. R., Marshall, D. J., & Cohen, A. L. (2011). Projecting coral reef futures under global warming and ocean acidification. *Science*, *333*(6041), 418-422.

Pardo, L. H., Fenn, M. E., Goodale, C. L., Geiser, L. H., Driscoll, C. T., Allen, E. B., Baron, J. S., Bobbink, R., Bowman, W. D., Clark, C. M., Emmett, B., Gilliam, F. S., Greaver, T. L., Hall, S. J., Lilleskov, E. A., Liu, L., Lynch, J. A., Nadelhoffer, K. J., Perakis, S. S., Robin-Abbott, M. J., Stoddard, J. L., Weathers, K. C., & Dennis, R. L. (2011). Effects of nitrogen deposition and empirical nitrogen critical loads for ecoregions of the United States. *Ecological Applications*, *21*(8), 3049-3082.

Paredes, M. (2016). The glocalization of mining conflict: Cases from Peru. *Extractive Industries and Society*, 3(4), 1046–1057. http://doi.org/10.1016/j.exis.2016.08.007

Paris, C. B., Aldana-Aranda, D., Pérez-Pérez, M., & Kool, J. (2008, July). Connectivity of Queen conch, Strombus gigas, populations from Mexico. In *Proceedings of the Coral Reef Symposium* (Vol. 11, pp. 439-443).

Parrotta, J. A. (1992). The role of plantation forests in rehabilitating degraded tropical ecosystems. *Agriculture, Ecosystems & Environment*, 41(2), 115-133.

Parrotta, J. A., Turnbull, J. W., & Jones, N. (1997). Catalyzing native forest regeneration on degraded tropical lands. *Forest Ecology and Management*, 99(1), 1-7.

Paruelo, J. M., Jobbágy, E. G., Oesterheld, M., Golluscio, R. A., & Aguiar, M. R. (2007). *The grasslands and steppes of Patagonia and the Rio de la Plata plains* (pp. 232-248). Oxford University Press: Oxford, England.

Passos, C. J., & Mergler, D. (2008). Human mercury exposure and adverse health effects in the Amazon: a review. *Cadernos de Saúde Pública*, *24*, s503-s520.

Pastick, N. J., Duffy, P., Genet, H., Rupp, T. S., Wylie, B. K., Johnson, K. D., Jorgenson, M.T., Bliss, N., McGuire, A.D., Jafarov, E.E., & Knight, J. F. (2017). Historical and projected trends in landscape drivers affecting carbon dynamics in Alaska. *Ecological Applications*. https://doi.org/10.1002/eap.1538

Pastorino, G., Penchaszadeh, P. E., Schejter, L., & Bremec, C. S. (2000). Rapana venosa (Valenciennes, 1846) (Mollusca: Muricidae): A new gastropod in South Atlantic waters.[Rapana venosa (Valenciennes, 1846) (Mollusca: Muricidae): un nuevo gasterópodo en aguas del Atlántico sudoccidental]. *Journal of Shellfish Research*, *19*(2), 897-899.

Pastorino, G., Darrigran, G. A., Lunaschi, L., & Martín, S. M. (1993). *Limnoperna fortunei (Dunker, 1857) (Mytilidae), nuevo bivalvo invasor en aguas del Río de la Plata* (No. DOC 0065).

Patterson, K. L., Porter, J. W., Ritchie, K. B, Polson, S. W., Mueller, E., Peters, E. C., Santavy, D. L., & Smith, G. W. (2002). The etiology of white pox, a lethal disease of the Caribbean elkhorn coral, Acropora palmata. *Proceedings of the National Academy of Sciences*, *99*(13), 8725-8730. http://www.pnas.org/content/99/13/8725.short

Pauchard, A., García, R., Zalba, S., Sarasola, M., Zenni, R., Ziller, S., & Nuñez, M. A. (2015). Pine invasions in South America: reducing their ecological impacts through active management. *Biological invasions in changing ecosystems. De Gruyter Open Ltd, Berlin*, 318-342.

Pauchard, A., & Barbosa, O. (2013). Regional assessment of Latin America: rapid urban development and social economic inequity threaten biodiversity hotspots. In *Urbanization, biodiversity and ecosystem services: Challenges and opportunities* (pp. 589-608). Springer, Dordrecht.

Pauchard, A., Kueffer, C., Dietz, H., Daehler, C. C., Alexander, J., Edwards, P. J., Arévalo, J. R., 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

Pauchard, A., Milbau, A., Albihn, A., Alexander, J., Burgess, T., Daehler, C., Englund, G., Essl, F., Evengard, B., Greenwood, G. B., Haider, S., Lenoir, J., McDougall, K., Muths, E., Nuñez, M. A., Olofsson, J., Pellissier, L., Rabitsch, W., Rew, L. J., Robertson, M., Sanders, N., & Kueffer, C. (2016). Non-native and native organisms moving into high elevation and high latitude ecosystems in an era of climate change: new challenges for ecology and conservation. *Biological Invasions*, 18(2), 345–353. https://doi.org/10.1007/s10530-015-1025-x

Pauchard, A., Nuñez, M. A., Raffaele, E., Bustamante, R., Ledgard, N. J., Relva, M. A., & Simberloff, D. (2010). Introduced conifer invasion in South America: an update. *The Scientific Magazine of the International Biogeography Society*, 2(2), 34–36.

Pauly, D., & Palomares, M. L. (2005). Fishing down marine food web: it is far more pervasive than we thought. *Bulletin of Marine Science*, *76*(2), 197-212. https://doi.org/10.1126/science.279.5352.860

Pawlak, A. R., Mack, R. N., Busch, J. W., & Novak, S. J. (2014). Invasion of *Bromus tectorum* (L.) into California and the American Southwest: rapid, multi-directional and genetically diverse. *Biological Invasions*, 17(1), 287–306.

Pedersen, S., Madsen, J., & Dyhr-Nielsen, M. (2004). *Global international waters assessment. arctic greenland, east Greenland Shelf, West Greenland shelf, GIWA regional assessment 1b, 15, 16*. United Nations Environment Programme.

Pelc, R., Max, L., Norden, W., Roberts, S., Silverstein, R., & Wilding, S. (2015). Further action on bycatch could boost United States fisheries performance. *Marine Policy*, 56, 56–60. https://doi.org/10.1016/J.MARPOL.2015.02.002

Penchaszadeh, P. E., Boltovskoy, D., Borges, M., Cataldo, D., Damborenea, C., Darrigran, G. & Silvestre, F. (2005). Invasores: Invertebrados exóticos en el Río de la Plata y región marina aledaña. *Eudeba, Buenos Aires*, *384*.

Pereira, H. M., Leadley, P., Proença, V., Alkemade, R., Scharlemann, J. P. W., Fernandez-Manjarrés, J. F., Araújo, M. B., Balvanera, P., Biggs, R., Cheung, W. W. L., Chini, L., Cooper, H. D., Gilman, E. L., Guénette, S., Hurtt, G. C., Huntington, H. P., Mace, G. M., Oberdorff, T., Revenga, C., Rodrigues, P., Scholes, R. J., Sumaila, U. R., & Walpole, M. (2010). Scenarios for global biodiversity in the 21st century. *Science*, 330(6010), 1496–1501. https://doi.org/10.1126/science.1196624

Perelman, S. B., Chaneton, E. J., Batista, W. B., Burkart, S. E., & León, R. J. C. (2007). Habitat stress, species pool size and biotic resistance influence exotic plant richness in the Flooding Pampa grasslands. *Journal of Ecology*, 95(4), 662–673. https://doi.org/10.1111/j.1365-2745.2007.01255.x

Pérez-Ramírez, M., Castrejón, M., Gutiérrez, N. L., & Defeo, O. (2016). The Marine Stewardship Council certification in Latin America and the Caribbean: A review of experiences, potentials and pitfalls. *Fisheries Research*, 182, 50–58. http://doi.org/10.1016/j.fishres.2015.11.007

Perez, C. (2004). Technological revolutions, paradigm shifts and socio-institutional change. *Globalization, economic development and inequality: An alternative perspective*, 217-242.

Perner, K., Leipe, T., Dellwig, O., Kuijpers, A., Mikkelsen, N., Andersen, T.J. & Harff, J. (2010). Contamination of arctic Fjord sediments by Pb–Zn mining at Maarmorilik in central West Greenland. *Marine pollution bulletin*, 60(7), pp.1065-1073.

Peterson, T. C., Heim, R. R., Hirsch, R., Kaiser, D. P., Brooks, H., Diffenbaugh, N. S., Dole, R. M., Giovannettone, J. P., Guirguis, K., Karl, T. R., Katz, R. W., Kunkel, K., Lettenmaier, D., McCabe, G. J., Paciorek, C. J., Ryberg, K. R., Schubert, S., Silva, V. B. S., Stewart, B. C., Vecchia, A. V., Villarini, G., Vose, R. S., Walsh, J., Wehner, M., Wolock, D., Wolter, K., Woodhouse, C. A., & Wuebbles, D. (2013). Monitoring and understanding changes in heat waves, cold waves, floods, and droughts in the United States: State of knowledge. *Bulletin of the American Meteorological Society*, *94*(6), 821–834. https://doi.org/10.1175/BAMS-D-12-00066.1

Phillips, O.L., Aragão, L.E., Lewis, S.L., Fisher, J.B., Lloyd, J., López-González, G., Malhi ,Y., Monteagudo, A., Peacock, J., Quesada, C.A., van der Heijden, G., Almeida, S., Amaral, I., Arroyo, L., Aymard, G., Baker, T.R., Bánki, O., Blanc, L., Bonal, D., Brando, P., Chave, J., de Oliveira, A.C., Cardozo, N.D., Czimczik, C.I., Feldpausch, T.R., Freitas, M.A., Gloor, E., Higuchi, N., Jiménez, E., Lloyd, G., Meir, P., Mendoza, C., Morel, A., Neill, D.A., Nepstad, D., Patiño, S., Peñuela, M.C., Prieto, A., Ramírez, F., Schwarz, M., Silva, J., Silveira, M., Thomas, A.S., Steege, H.T., Stropp, J., Vásquez, R., Zelazowski, P., Alvarez Dávila, E., Andelman, S., Andrade, A., Chao, K.J., Erwin, T., Di Fiore, A., Honorio, C.E., Keeling, H., Killeen, T.J., Laurance, W.F., Peña Cruz, A., Pitman, N.C., Núñez Vargas, P., Ramírez-Angulo, H., Rudas, A., Salamão, R., Silva, N., Terborgh, J., Torres-Lezama, A. (2009). Drought sensitivity of the Amazon rainforest. *Science,* 323(5919), 1344-1347.

Philpot, S., Hipel, K., & Johnson, P. (2016). Strategic analysis of a water rights conflict in the Southwestern United States. *Journal of Environmental Management*, 180, 247–256. http://doi.org/10.1016/j.jenvman.2016.05.027

Pichs, R. (2008). Cambio Climático, globalización y subdesarrollo. *Editorial Científico-Técnica. La Habana*.

Pichs, R. (2012). Recursos naturales, economía mundial y crisis ambiental. *Científico Técnica–RUTH Casa Editorial*.

Pimentel, D., Lach, L., Zuniga, R., & Morrison, D. (2000). Environmental and economic costs of nonindigenous species in the United States. *Bio Science*, 50(1), 53–64.

Pinkerton, E., & Davis, R. (2015). Neoliberalism and the politics of enclosure in North American small-scale fisheries. *Marine Policy*, 61, 303–312. http://doi.org/10.1016/j.marpol.2015.03.025

Pinto, L. F. G., & McDermott, C. (2013). Equity and forest certification - A case study in Brazil. *Forest Policy and Economics*, 30, 23–29.

Pirard, R., & Belna, K. (2012). Agriculture and Deforestation: Is REDD+ Rooted in Evidence? *Forest Policy and Economics*, 21, 62–70. http://doi.org/10.1016/j.forpol.2012.01.012

Pirrone, N., Cinnirella, S., Feng, X., Finkelman, R.B., Friedli, H.R., Leaner, J., Mason, R., Mukherjee, A.B., Stracher, G.B., Streets, D.G., & Telmer, K. (2010). Global mercury emissions to the atmosphere from anthropogenic and natural sources. *Atmospheric Chemistry and Physics*, 10(13), 5951-5964.

Pivello, V. R. (2014). *Invasões Biológicas no Cerrado Brasileiro: Efeitos da Introdução de Espécies Exóticas sobre a Biodiversidade.* Retrieved from http://www.ecologia.info/cerrado.htm

Pivello, V. R. (2011). The use of fire in the cerrado and Amazonian rainforests of Brazil: Past and present. *Fire Ecology,* 7, 24–39.

Poff, N. L., Allan, J. D., Bain, M. B., Karr, J. R., Prestegaard, K. L., Richter, B. D., Sparks, R. E., & Stromberg, J. C. (1997). The natural flow regime: A paradigm for river conservation and restoration. *BioScience*, *47*(11), 769-784.

Pokorny, B., Pacheco, P., Cerutti, P.O., van Solinge, T.B., Kissinger, G., & Tacconi, L. (2016). Drivers of Illegal and Destructive Forest Use. In: Kleinschmit, D., Mansourian, S., Wildburger, C., Purret, A. (Eds.) *Illegal logging and related timber trade : dimensions, drivers, impacts and responses : a global scientific rapid response assessment report.* (pp. 61-80). International Union of Forest Research Organizations (IUFRO), World Series no. 35. Vienna, Austria. Retrieved from http://www.cifor.org/library/6312/drivers-of-illegal-and-destructive-forest-use/

Pollock, H.S., Cheviron, Z.A., Agin, T.J., & Brawn, J.D. (2015). Absence of microclimate selectivity in insectivorous birds of the Neotropical forest understory. *Biological Conservation,* 188, 116-125. http://dx.doi.org/10.1016/j.biocon.2014.11.013

Ponce-Reyes, R., Nicholson, E., Baxter, P. W., Fuller, R. A., & Possingham, H. (2013). Extinction risk in cloud forest fragments under climate change and habitat loss. *Diversity and Distributions*, 19(5-6), 518-529.

Poore, J. A. (2016). Call for conservation: Abandoned pasture. *Science*, 351(6269), 132-132.

Poorter, L., Bongers, F., Aide, T.M., Zambrano, A.M.A., Balvanera, P., Becknell, J.M., Boukili, V., Brancalion, P.H.S., Broadbent, E.N., Chazdon, R.L., Craven, D., de Almeida-Cortez, J.S., Cabral, G.A.L., de Jong, B.H.J., Denslow, J.S., Dent, D.H., DeWalt, S.J., Dupuy, J.M., Durán, S.M., Espírito-Santo, M.M., Fandino, M.C., César, R.G., Hall, J.S., Hernandez-Stefanoni, J.L., Jakovac, C.C., Junqueira, A.B., Kennard, D., Letcher, S.G., Licona, J.C., Lohbeck, M., Marín-Spiotta, E., Martínez-Ramos, M., Massoca, P., Meave, J.A., Mesquita, R., Mora, F., Muñoz, R., Muscarella, R., Nunes, Y.R.F., Ochoa-Gaona, S., de Oliveira, A.A., Orihuela-Belmonte, E., Peña-Claros, M., Pérez-García, E.A., Piotto, D., Powers, J.S., Rodríguez-Velázquez, J., Romero-Pérez, I.E., Ruíz, J., Saldarriaga, J.G., Sanchez-Azofeifa, A., Schwartz, N.B., Steininger, M.K., Swenson, N.G., Toledo, M., Uriarte, M., van Breugel, M., van der Wal, H., Veloso, M.D.M., Vester, H.F.M., Vicentini, A., Vieira, I.C.G., Vizcarra Bentos, T., Williamson, G.B., & Rozendaal, D.M.A. (2016). Biomass resilience of Neotropical secondary forests. *Nature*, *530*(7589), 211. http://www.nature.com/doifinder/10.1038/nature16512

Pope III, C. A., Ezzati, M., & Dockery, D. W. (2009). Fine-particulate air pollution and life expectancy in the United States. *New England Journal of Medicine*, *360*(4), 376-386.

Porter, E. M., Bowman, W. D., Clark, C. M., Compton, J. E., Pardo, L. H., & Soong, J. L. (2013). Interactive effects of anthropogenic nitrogen enrichment and climate change on terrestrial and aquatic biodiversity. *Biogeochemistry*, *114*(1-3), 93-120.

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.

Possingham, H. P., & Wilson, K. A. (2005). Biodiversity: Turning up the heat on hotspots. *Nature*, 436(7053), 919-920.

Pozo, K., Urrutia, R., Barra, R., Mariottini, M., Treutler, H. C., Araneda, A., & Focardi, S. (2007). Records of polychlorinated biphenyls (PCBs) in sediments of four remote Chilean Andean Lakes. *Chemosphere*, *66*(10), 1911-1921.

Pozo, K., Urrutia, R., Mariottini, M., Rudolph, A., Banguera, J., Pozo, K., Parra, O. and Focardi, S. (2014). Levels of persistent organic pollutants (POPs) in sediments from Lenga estuary, central Chile. *Marine pollution bulletin*, *79*(1-2), 338-341.

Price, D.T.; Alfaro, R.I.; Brown, K.J.; Flannigan, M.D.; Fleming, R.A.; Hogg, E.H.; Girardin, M.P.; Lakusta, T.; Johnston, M.; McKenney, D.W.; Pedlar, J.H.; Stratton, T.; Sturrock, R.N.; Thompson, I.D.; Trofymow, J.A.; Venier, L.A. (2013). Anticipating the consequences of climate change for Canada's boreal forest ecosystems. *Environmental Reviews*, *21*(4), 322-365. http://www.nrcresearchpress.com/doi/full/10.1139/er-2013-0042

Prieto-Torres, D. A., Navarro-Sigüenza, A. G., Santiago-Alarcon, D., & Rojas-Soto, O. R. (2016). Response of the endangered tropical dry forests to climate change and the role of Mexican Protected Areas for their conservation. *Global change biology*, *22*(1), 364-379.

Purcell, S. W., Mercier, A., Conand, C., Hamel, J. F., Tora-Granda, M. V., Lovatelli, A., & Uthicke, S. (2013). Sea cucumber fisheries: global analysis of stocks, management measures and drivers of overfishing. *Fish and fisheries*, *14*(1), 34-59.

Pyne, S. J., Andrews, P. L., Laven, R. D., & Cheney, N. P. (1998). Introduction to wildland fire. *Forestry*, *71*(1), 82-82.

Pyšek, P., Jarošík, V., Pergl, J., Randall, R., Chytrý, M., Kühn, I., Tichý, L., Danihelka, J., Chrtek Jun, J., & Sádlo, J. (2009). The global invasion success of Central European plants is related to distribution characteristics in their native range and species traits. *Diversity and Distributions*, 15(5), 891–903. https://doi.org/10.1111/j.1472-4642.2009.00602.x

Rabalais, N. N., Turner, R. E., Justić, D., Dortch, Q., Wiseman, W. J., & Gupta, B. K. S. (1996). Nutrient changes in the Mississippi River and system responses on the adjacent continental shelf. *Estuaries*, 19(2), 386-407.

Rabalais, N.N., Cai, W.J., Carstensen, J., Conley, D.J., Fry, B., Hu, X., Quinones-Rivera, Z., Rosenberg, R., Slomp, C.P., Turner, R.E., & Voss, M. (2014). Eutrophication-driven deoxygenation in the coastal ocean. *Oceanography*, 27(1), 172-183.

Rajotte, T. (2012). The negotiations web: complex connections. In *The Future Control of Food* (pp. 163-190). Routledge.

Ramirez-Villegas, J., Cuesta, F., Devenish, C., Peralvo, M., Jarvis, A., & Arnillas, C. A. (2014). Using species distributions models for designing conservation strategies of Tropical Andean biodiversity under climate change. *Journal for Nature Conservation*, *22*(5), 391-404. http://dx.doi.org/10.1016/j.jnc.2014.03.007

Ramírez, V. M., Ayala, R., & González, H. D. (2016). Temporal variation in native bee diversity in the tropical sub-deciduous forest of the Yucatan Peninsula, Mexico. *Tropical Conservation Science*, 9(2), 718-734.

Ramos-Scharrón, C. E., Torres-Pulliza, D., & Hernández-Delgado, E. A. (2015). Watershed-and island wide-scale land cover changes in Puerto Rico (1930s–2004) and their potential effects on coral reef ecosystems. *Science of the total environment*, *506*, 241-251.

Rankin, D. (2002) Freshwater Ecosystems and Human Populations: Great Lakes Case Study. *Yale School of Forestry & Environmental Studies Bulletin Series,* 107.

Rathmann, R., Szklo, A., & Schaeffer, R. (2010). Land use competition for production of food and liquid biofuels: An analysis of the arguments in the current debate. *Renewable Energy*, 35(1), 14-22.

Reátegui-Zirena, E. G., Stewart, P. M., Whatley, A., Chu-Koo, F., Sotero-Solis, V. E., Merino-Zegarra, C., & Vela-Paima, E. (2014). Polycyclic aromatic hydrocarbon concentrations, mutagenicity, and Microtox® acute toxicity testing of Peruvian crude oil and oil-contaminated water and sediment. *Environmental monitoring and assessment*, *186*(4), 2171-2184.

Rebellato, L., Woods, W. I., & Neves, E. G. (2009). Pre-Columbian settlement dynamics in the Central Amazon. In *Amazonian dark earths: Wim Sombroek's vision* (pp. 15-31). Springer, Dordrecht.

Redmore, L., Stronza, A., Songhurst, A., & McCulloch, G. (2013). Which Way Forward? Past and New Perspectives on Community-Based Conservation in the Anthropocene.

Reenberg, A., Langanke, T., Kristensen, S. B. P., & Colding, T. S. (2010). Globalization of agricultural landscapes: a land systems approach. In *Globalisation and Agricultural Landscapes* (pp. 31-56). Cambridge University Press.

Reis, A., Bechara, F. C., Espíndola, M. D., Vieira, N. K., & Souza, L. D. (2003). Restauração de áreas degradadas: a nucleação como base para incrementar os processos sucessionais. *Natureza & Conservação*, *1*(1), 28-36.

Reis, C. R. G., Nardoto, G. B., & Oliveira, R. S. (2017). Global overview on nitrogen dynamics in mangroves and consequences of increasing nitrogen availability for these systems. *Plant and soil*, *410*(1-2), 1-19.

Restrepo, J. C., Gruber, C. G., & Machuca, C. M. (2009, June). Energy profile aware routing. In *Communications Workshops, 2009. ICC Workshops 2009. IEEE International Conference on* (pp. 1-5). IEEE.

Revenga, C., Campbell, I., Abell, R., De Villiers, P., & Bryer, M. (2005). Prospects for monitoring freshwater ecosystems towards the 2010 targets. *Philosophical Transactions of the Royal Society of London B: Biological Sciences*, *360*(1454), 397-413.

Reyer, C. P. O., Adams, S., Albrecht, T., Baarsch, F., Boit, A., Trujillo, N. C., Cartsburg, M., Coumou, D., Eden, A., Fernandes, E., Langerwisch, F., Marcus, R., Mengel, M., Mira-Salama, D., Perette, M., Pereznieto, P., Ramirez-Villegas, J., Reinhardt, J., Robinson, A., Rocha, M., Sakschewski, B., Schaeffer, M., Schleussner, C.-F., Serdeczny, O., & Thonicke, K. (2017). Climate change impacts in Latin America and the Caribbean and their implications for development. Regional environmental change, 17(6), 1601-1621.

Rhodin, A., Walde, A., Horne, B., Van Dijk, P., Blanck, T., & Hudson, R. (2011). Turtles in trouble: the world's 25+ most endangered tortoises and freshwater turtles—2011. IUCN/SSC Tortoise and Freshwater Turtle Specialist Group, Turtle Conservation Fund, Turtle Survival Alliance, Turtle Conservancy, Chelonian Research Foundation, Conservation International, Wildlife Conservation Society, and San Diego Zoo Globa. Lunenburg, MA.

Ribeiro, K. H., Favaretto, N., Dieckow, J., Souza, L. C. D. P., Minella, J. P. G., Almeida, L. D., & Ramos, M. R. (2014). Quality of surface water related to land use: a case study in a catchment with small farms and intensive vegetable crop production in southern Brazil. *Revista Brasileira de Ciência do Solo*, *38*(2), 656-668.

Ricaurte, L. F., Wantzen, K. M., Agudelo, E., Betancourt, B., & Jokela, J. (2014). Participatory rural appraisal of ecosystem services of wetlands in the Amazonian Piedmont of Colombia: elements for a sustainable management concept. *Wetlands ecology and management*, *22*(4), 343-361.

Richards, M., Wells, A., Del Gatto, F., Contreras-Hermosilla, A., & Pommier, D. (2003). Impacts of illegality and barriers to legality: a diagnostic analysis of illegal logging in Honduras and Nicaragua. *International Forestry Review*, *5*(3), 282-292.

Richards, P. D., Myers, R. J., Swinton, S. M., & Walker, R. T. (2012). Exchange rates, soybean supply response, and deforestation in South America. Global environmental change, 22(2), 454-462.

Rigét, F., Bignert, A., Braune, B., Stow, J., & Wilson, S. (2010). Temporal trends of legacy POPs in Arctic biota, an update. *Science of the Total Environment*, *408*(15), 2874-2884.

Rignot, E., Velicogna, I., van den Broeke, M. R., Monaghan, A., & Lenaerts, J. T. (2011). Acceleration of the contribution of the Greenland and Antarctic ice sheets to sea level rise. *Geophysical Research Letters*, *38*(5).

Ríos, A. F., Resplandy, L., García-Ibáñez, M. I., Fajar, N. M., Velo, A., Padin, X. A., Wanninkhof, R., Steinfeldt, R., Rosón, G., & Pérez, F. F. (2015). Decadal acidification in the water masses of the Atlantic Ocean. *Proceedings of the National Academy of Sciences*, *112*(32), 9950-9955.

Ríos, M., Zaldúa, N., & Cupeiro, S. (2010). Evaluación participativa de plaguicidas en el sitio RAMSAR, Parque Nacional Esteros de Farrapos e Islas del Río Uruguay. *Montevideo: Vida silvestre Uruguay. ISBN*, 978-9974.

Ripple, W. J., Newsome, T. M., Wolf, C., Dirzo, R., Everatt, K. T., Galetti, M., Hayward, M. W., Kerley, G. I. H., Levi, T., Lindsey, P. A., Macdonald, D. W., Malhi, Y., Painter, L. E., Sandom, C., Terborgh, J., & Van Valkenburgh, B. (2015). Collapse of the world's largest herbivores. Science Advances 1: e1400103.

Rizzo, D. M., & Garbelotto, M. (2003). Sudden oak death: endangering California and Oregon forest ecosystems. *Frontiers in Ecology and the Environment*, *1*(4), 197-204.

Robeson, S. M. (2002). Increasing growing-season length in Illinois during the 20th century. Climatic Change, 52(1-2), 219-238.

Rockström, J., Steffen, W., Noone, K., Persson, Å., Chapin, F.S., Lambin, E.F., Lenton, T.M., Scheffer, M., Folke, C., Schellnhuber, H.J. & Nykvist, B. (2009). A safe operating space for humanity. Nature, 461(7263), 472.

Rodil, I. F., Lucena-Moya, P., Olabarria, C., & Arenas, F. (2015). Alteration of macroalgal subsidies by climate-associated stressors affects behavior of wrack-reliant beach consumers. *Ecosystems*, *18*(3), 428-440.

Rodrigues Capitulo, A., Gómez, N., Giorgi, A., & Feijoó, C. (2010). Global changes in pampean lowland streams (Argentina): implications for biodiversity and functioning. In *Global Change and River Ecosystems—Implications for Structure, Function and Ecosystem Services* (pp. 53-70). Springer, Dordrecht.

Rodrigues, R. R., Gandolfi, S., Nave, A. G., Aronson, J., Barreto, T. E., Vidal, C. Y., & Brancalion, P. H. (2011). Large-scale ecological restoration of high-diversity tropical forests in SE Brazil. *Forest Ecology and Management*, *261*(10), 1605-1613.

Rodrigues, R. R., Lima, R. A., Gandolfi, S., & Nave, A. G. (2009). On the restoration of high diversity forests: 30 years of experience in the Brazilian Atlantic Forest. *Biological conservation*, *142*(6), 1242-1251.

Rodríguez-Rodríguez, J. A., Sierra-Correa, P. C., Gómez-Cubillos, M. C., & Villanueva, L. V. L. (2016). Mangrove ecosystems (Colombia). In The Wetland Book (pp. 1-10). Finlayson, C. M., Everard, M., Irvine, K., McInnes, R. J., Middleton, B., van Dam, A., & Davidson, N. C. (Eds.) Springer.

Rojas-Sandoval, J., & Acevedo-Rodríguez, P. (2015). Naturalization and invasion of alien plants in Puerto Rico and the Virgin Islands. *Biological Invasions*, *17*(1), 149-163.

Romero-Lankao, P., Smith, J.B., Davidson, D.J., Diffenbaugh, N.S., Kinney, P.L., Kirshen, P., Kovacs, P., & Villers Ruiz, L. (2014). North America. In Barros, V.R., Field, C.B., Dokken, D.J., Mastrandrea, M.D., Mach, K.J., Bilir, T.E., Chatterjee, M., Ebi, K.L., Estrada, Y.O., Genova, R.C., Girma, B., Kissel, E.S., Levy, A.N., MacCracken, S., Mastrandrea, P.R., & White L.L. (Eds.) *Climate Change 2014: Impacts, Adaptation, and Vulnerability. Part B: Regional Aspects. Contribution of Working Group II to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, United

Kingdom and New York, NY, USA: Cambridge University Press.

Ropelewski, C. F., & Halpert, M. S. (1987). Global and regional scale precipitation patterns associated with the El Niño/Southern Oscillation. *Monthly weather review*, *115*(8), 1606-1626.

Rossi, R. D., Martins, C. R., Viana, P. L., Rodrigues, E. L., & Figueira, J. E. C. (2014). Impact of invasion by molasses grass (Melinis minutifloraP. Beauv.) on native species and on fires in areas of campo-cerrado in Brazil. *Acta Botanica Brasilica*, *28*(4), 631-637.

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.

Rudel, T. K., Defries, R., Asner, G. P., & Laurance, W. F. (2009). Changing drivers of deforestation and new opportunities for conservation. *Conservation Biology*, *23*(6), 1396-1405.

Ruiz, G. M., Rawlings, T. K., Dobbs, F. C., Drake, L. A., Mullady, T., Huq, A., & Colwell, R. R. (2000). Global spread of microorganisms by ships. *Nature*, *408*(6808), 49.

Ruiz-Toledo, J., Castro, R., Rivero-Pérez, N., Bello-Mendoza, R., & Sánchez, D. (2014). Occurrence of glyphosate in water bodies derived from intensive agriculture in a tropical region of southern Mexico. *Bulletin of environmental contamination and toxicology*, *93*(3), 289-293.

Runde, D.F., & Magpile, J. (2014). Science, technology, and innovation as drivers of development: http://csis.org/publication/science-technology-and-innovation-drivers-development

Rupp, T. S., P. Duffy, M. Leonawicz, M. Lindgren, A. Breen, T. Kurkowski, A. Floyd, A. Bennett, & L. Krutikov. (2016). Chapter 2. Climate Simulations, Land Cover, and Wildfire. Pages 17–52 in Zhu, Z., & McGuire, A. D. (Eds.). *Baseline and projected future carbon storage and greenhouse-gas fluxes in ecosystems of Alaska*. US Department of the Interior, US Geological Survey, Reston.

Rushton, J., Viscarra, R., Viscarra, C., Basset, F., Baptista, R., & Brown, D. (2005). How important is bushmeat consumption in South America: now and in the future. *Odi wildlife policy Briefing*, 11:1–4.

Ruviaro, C. F., da Costa, J. S., Florindo, T. J., Rodrigues, W., de Medeiros, G. I. B., & Vasconcelos, P. S. (2016). Economic and environmental feasibility of beef production in different feed management systems in the Pampa biome, southern Brazil. *Ecological indicators*, *60*, 930-939.

Ruyle L.E. (2017). The Impacts of Conflict on Biodiversity in the Anthropocene. Reference Module in Earth Systems and Environmental Sciences. Published on line. doi.org/10.1016/B978-0-12-409548-9.09849-3

Ryan, K. C., Knapp, E. E., & Varner, J. M. (2013). Prescribed fire in North American forests and woodlands: history, current practice, and challenges. *Frontiers in Ecology and the Environment*, *11*(s1), e15-e24.

Saatchi, S., Asefi-Najafabady, S., Malhi, Y., Aragão, L. E., Anderson, L. O., Myneni, R. B., & Nemani, R. (2013). Persistent effects of a severe drought on Amazonian forest canopy. *Proceedings of the National Academy of Sciences*, *110*(2), 565-570.

Sala, O. E., & Paruelo, J. M. (1997). Ecosystem services in grasslands.:'Nature's Services: Societal Dependence on Natural Ecosystems'.(Ed. G. Daily.) pp. 237–252.

Sala, O. E., Chapin III, F. S., Armesto, J. J., Berlow, E., Bloomfield, J., Dirzo, R., Huber-Sanwald, E., Huenneke, L. F., Jackson, R. B., Kinzig, A., Leemans, R., Lodge, D. M., Mooney, H. A., Oesterheld, M., Poff, N. L., Skykes, M. T., Walker, B. H., Walker, M., & Wall, D. H. (2000). Global biodiversity scenarios for the year 2100. *science*, *287*(5459), 1770-1774.

San Sebastián, M., & Karin Hurtig, A. (2004). Oil exploitation in the Amazon basin of Ecuador: a public health emergency. *Revista panamericana de salud pública*, *15*, 205-211.

San Sebastián, M., Armstrong, B., & Stephens, C. (2002). Outcomes of pregnancy among women living in the proximity of oil fields in the Amazon basin of Ecuador. *International journal of occupational and environmental health*, *8*(4), 312-319.

Sánchez-Azofeifa, A., Powers, S. J., Fernandez, G.W., Quesada, Q. (2014). *Tropical Dry Forests in the Americas: Ecology, Conservation, and Management.* CRC Press, Taylor & Francis Group, Boca Raton, USA

Sánchez-Azofeifa, G. A., Castro-Esau, K. L., Kurz, W. A., & Joyce, A. (2009). Monitoring carbon stocks in the tropics and the remote sensing operational limitations: from local to regional projects. *Ecological Applications*, *19*(2), 480-494.

Sánchez, A., Ortiz-Hernández, M. C., Talavera-Sáenz, A., & Aguíñiga-García, S. (2013). Stable nitrogen isotopes in the turtle grass Thalassia testudinum from the Mexican Caribbean: Implications of anthropogenic development. *Estuarine, Coastal and Shelf Science*, *135*, 86-93.

Sandoval-Herrera, N. I., Vargas-Soto, J. S., Espinoza, M., Clarke, T. M., Fisk, A. T., & Wehrtmann, I. S. (2016). Mercury levels in muscle tissue of four common elasmobranch species from the Pacific coast of Costa Rica, Central America. *Regional Studies in Marine Science*, *3*, 254-261.

Santo, A. R., Sorice, M. G., Donlan, C. J., Franck, C. T., & Anderson, C. B. (2015). A human-centered approach to designing invasive species eradication programs on human-inhabited islands. *Global Environmental Change*, *35*, 289-298.

Santos-Santos, E., Yarto-Ramírez, M., Gavilán-García, I., Castro-Díaz, J., Gavilán-García, A., Rosiles, R., Suárez, S. & López-Villegas, T. (2006). Analysis of arsenic, lead and mercury in farming areas with mining contaminated soils at Zacatecas, Mexico. *Journal of the Mexican Chemical Society*, *50*(2), 57-63.

Santos, H. F., Carmo, F. L., Paes, J. E., Rosado, A. S., & Peixoto, R. S. (2011). Bioremediation of mangroves impacted by petroleum. *Water, Air, & Soil Pollution*, *216*(1-4), 329-350.

Sarukhán, J., Urquiza-Haas, T., Koleff, P., Carabias, J., Dirzo, R., Ezcurra, E., Cerdeira-Estrada, S., & Soberón, J. (2014). Strategic actions to value, conserve, and restore the natural capital of megadiversity countries: the case of Mexico. *BioScience*, *65*(2), 164-173.

Satyamurty, P., de Castro, A. A., Tota, J., da Silva Gularte, L. E., & Manzi, A. O. (2010). Rainfall trends in the Brazilian Amazon Basin in the past eight decades. *Theoretical and Applied Climatology*, *99*(1-2), 139-148.

Savini, D., & Occhipinti-Ambrogi, A. (2006). Consumption rates and prey preference of the invasive gastropod Rapana venosa in the Northern Adriatic Sea. *Helgoland Marine Research*, *60*(2), 153.

Sayer, E. J., Wright, S. J., Tanner, E. V., Yavitt, J. B., Harms, K. E., Powers, J. S., Kaspari, M., Garcia, M.N., & Turner, B. L. (2012). Variable responses of lowland tropical forest nutrient status to fertilization and litter manipulation. *Ecosystems*, *15*(3), 387-400.

Scanlon, B. R., Jolly, I., Sophocleous, M., & Zhang, L. (2007). Global impacts of conversions from natural to agricultural ecosystems on water resources: Quantity versus quality. *Water resources research*, *43*(3).

Scarabino, F., Menafra, R., & Etchegaray, P. (1999). Presencia de Rapana venosa (Valenciennes, 1846) (Gastropoda: Muricidae) en el Río de la Plata. *Boletín de la Sociedad Zoológica del Uruguay (Actas de las V Jornadas de Zoología del Uruguay)*, *11*(Segunda Época), 40.

Scatena, F. N., & Larsen, M. C. (1991). Physical aspects of hurricane Hugo in Puerto Rico. *Biotropica*, 317-323.

Schaeffer-Novelli, Y., Soriano-Serra, E. J., Vale, C. C., Bernini, E., Rovai, A. S., Pinheiro, M. A. A., Schmidt, A. J., Almeida, R., Jr. Coelho C., Menghini, R. P., Martinez, D. I., Abuchahla, G. M. O., Cunha-Lignon, M., Charlier-Sarubo, S., Shirazawa-Freitas, J., & Cintrón-Molero, G. (2016). Climate changes in mangrove forests and salt marshes. *Brazilian Journal of Oceanography*, *64*(spe2), 37-52.

Scheffers, B. R., De Meester, L. Bridge, T. C. L., Hoffmann, A. A., Pandolfi, J. M., Corlett, R. T., Butchart, S. H. M., Pearce-Kelly, P., Kovacs, K. M., Dudgeon, D., Pacifici, M., Rondinini, C., Foden, W. B., Martin, T. G., Mora, C., Bickford, D., & Watson, J. E. M. (2016). The broad footprint of climate change from genes to biomes to people. *Science*, *354*(6313), aaf7671.

Schindler, D. W., Dillon, P. J., & Schreier, H. (2006). A review of anthropogenic sources of nitrogen and their effects on Canadian aquatic ecosystems. In *Nitrogen Cycling in the Americas: Natural and Anthropogenic Influences and Controls* (pp. 25-44). Springer, Dordrecht.

Schleupner, C., & Link, P. M. (2008). Potential impacts on important bird habitats in Eiderstedt (Schleswig-Holstein) caused by agricultural land use changes. *Applied Geography*, *28*(4), 237-247.

Schmidtko, S., Stramma, L., & Visbeck, M. (2017). Decline in global oceanic oxygen content during the past five decades. *Nature*, *542*(7641), 335.

Schneider, L. C. (2006). Invasive species and land-use: the effect of land management practices on bracken fern invasion in the region of Calakmul, Mexico. *Journal of Latin American Geography*, 91-107.

Schneider, L. C., & Fernando, D. N. (2010). An untidy cover: invasion of bracken fern in the shifting cultivation systems of Southern Yucatán, Mexico. *Biotropica*, *42*(1), 41-48.

Schofield, P. J. (2010). Update on geographic spread of invasive lionfishes (Pterois volitans [Linnaeus, 1758] and P. miles [Bennett, 1828]) in the Western North Atlantic Ocean, Caribbean Sea and Gulf of Mexico. *Aquatic Invasions*, *5*(Supplement 1), S117-S122.

Schofield, P. J. (2009). Geographic extent and chronology of the invasion of non-native lionfish (Pterois volitans [Linnaeus 1758] and P. miles [Bennett 1828]) in the Western North Atlantic and Caribbean Sea. *Aquatic Invasions*, *4*(3), 473-479.

Schöngart, J., & Wittmann, F. (2010). Biomass and net primary production of central Amazonian floodplain forests. In *Amazonian Floodplain Forests* (pp. 347-388). Springer, Dordrecht.

Schwarzenbach, R. P., Escher, B. I., Fenner, K., Hofstetter, T. B., Johnson, C. A., Von Gunten, U., & Wehrli, B. (2006). The challenge of micropollutants in aquatic systems. *Science*, *313*(5790), 1072-1077.

Seabloom, E. W., Williams, J. W., Slayback, D., Stoms, D. M., Viers, J. H., & Dobson, A. P. (2006). Human impacts, plant invasion, and imperiled plant species in California. *Ecological Applications*, *16*(4), 1338-1350.

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., Gastner, M. T., Blasius, B., & Courchamp, F. (2013). The risk of marine bioinvasion caused by global shipping. *Ecology letters*, *16*(6), 782-790.

Seijas, A. E., Antelo, R., Thorbjarnarson, J. B., & Ardila Robayo, M. C. (2010). Orinoco Crocodile *Crocodylus intermedius*. In Manolis S. C., & C. Stevenson (Eds). *Crocodiles. Status Survey and Conservation Action Plan*. (pp. 9–65). Third. Crocodile Specialist Group: Darwin

Seipel, T., Kueffer, C., Rew, L. J., Daehler, C. C., Pauchard, A., Naylor, B. J., Alexander, J. M., Edwards, P. J., Parks, C. G., Arevalo, J. R., Cavieres, L. A., Dietz, H., Jakobs, G., Mcdougall, K., Otto, R., & Walsh, N. (2012). Processes at multiple scales affect richness and similarity of non-native plant species in mountains around the world. *Global Ecology and Biogeography*, *21*(2), 236-246.

Sell, S. K. (2009). Private Power, Public Law. The Globalization of Intellectual Property Rights. Cambridge University Press. Online publication date:

September 2009. https://doi.org/10.1017/CBO9780511491665

Serenari, C., Peterson, M. N., Wallace, T., & Stowhas, P. (2017). Indigenous perspectives on private protected areas in Chile. *Natural Areas Journal*, *37*(1), 98-107.

SERNA (Secretaría de Recursos Naturales y Ambiente de Honduras). (2009). Inventario nacional de humedales de las República de Honduras. Retrieved from 2009 https://www.scribd.com/document/98703779/Inventario-de-Humedales-de-Honduras#

Seto, K. C., Fragkias, M., Güneralp, B., & Reilly, M. K. (2011). A meta-analysis of global urban land expansion. *PloS one*, *6*(8), e23777.

Seto, K. C., Güneralp, B., & Hutyra, L. R. (2012). Global forecasts of urban expansion to 2030 and direct impacts on biodiversity and carbon pools. *Proceedings of the National Academy of Sciences*, *109*(40), 16083-16088.

Shelton, P. A., & Morgan, M. J. (2014). Impact of maximum sustainable yield-based fisheries management frameworks on rebuilding North Atlantic cod stocks. *Journal of Northwest Atlantic Fishery Science*, *46*.

Shen, L., Wania, F., Lei, Y. D., Teixeira, C., Muir, D. C., & Xiao, H. (2006). Polychlorinated biphenyls and polybrominated diphenyl ethers in the North American atmosphere. *Environmental pollution*, *144*(2), 434-444.

Sherman, K., & Hamukuaya, H. (2016). Sustainable development of the world's Large Marine Ecosystems. *Environmental Development 17* (Spt 1), 1-6.

Sherman, K., Sissenwine, M., Christensen, V., Duda, A., Hempel, G., Ibe, C., Levin, S., Lluch-Belda, D., Matishov, G., McGlade, J. & O'toole, M. (2005). A global movement toward an ecosystem approach to management of marine resources. *Marine Ecology Progress Series*, *300*, 275-279.

Shirey, P. D., Kunycky, B. N., Chaloner, D. T., Brueseke, M. A., & Lamberti, G. A. (2013). Commercial trade of federally listed threatened and endangered plants in the United States. *Conservation Letters*, *6*(5), 300-316.

Shunthirasingham, C., Gouin, T., Lei, Y. D., Ruepert, C., Castillo, L. E., & Wania, F. (2011). Current-use pesticide transport to Costa Rica's high-altitude tropical cloud forest. *Environmental toxicology and chemistry*, *30*(12), 2709-2717.

Siffredi, G., López, D., Ayesa, J., Bianchi, E., Velasco, V., & Becker, G. (2011). Reducción de la accesibilidad al forraje por caída de cenizas volcánicas. *Revista Presencia*, (57).

Silva, E. (2016). Patagonia, without Dams! Lessons of a David vs. Goliath campaign. *The Extractive Industries and Society*, *3*(4), 947-957.

Silva, J. S. O., da Cunha Bustamante, M. M., Markewitz, D., Krusche, A. V., & Ferreira, L. G. (2011). Effects of land cover on chemical characteristics of streams in the Cerrado region of Brazil. *Biogeochemistry*, *105*(1-3), 75-88.

Silva, J. S. V. D., & Souza, R. C. C. L. D. (2004). Água de lastro e bioinvasão. In *Agua de lastro e bioinvasao*. Interciência.

Silver, W. L., Lugo, A. E., & Keller, M. (1999). Soil oxygen availability and biogeochemistry along rainfall and topographic gradients in upland wet tropical forest soils. *Biogeochemistry*, *44*(3), 301-328.

Silvério, D. V., Brando, P. M., Balch, J. K., Putz, F. E., Nepstad, D. C., Oliveira-Santos, C., & Bustamante, M. M. (2013). Testing the Amazon savannization hypothesis: fire effects on invasion of a neotropical forest by native cerrado and exotic pasture grasses. *Philosophical transactions of the Royal Society of London B: Biological sciences*, *368*(1619), 20120427.

Simberloff, D., Martin, J. L., Genovesi, P., Maris, V., Wardle, D. A., Aronson, J., Courchamp, F., Galil, B., García-Berthou, E., Pascal, M., Pyšek, P., Sousa, R., Tabacchi, E., & Vilà, M. (2013). Impacts of biological invasions: what's what and the way forward. *Trends in ecology & evolution*, *28*(1), 58-66.

Skalak, K. J., Benthem, A. J., Schenk, E. R., Hupp, C. R., Galloway, J. M., Nustad, R. A., & Wiche, G. J. (2013). Large dams and alluvial rivers in the Anthropocene: The impacts of the Garrison and Oahe Dams on the Upper Missouri River. *Anthropocene*, *2*, 51-64.

Smith, R. J., Muir, R. D., Walpole, M. J., Balmford, A., & Leader-Williams, N. (2003). Governance and the loss of biodiversity. *Nature*, *426*(6962), 67.

Soares-Filho, B. S., Nepstad, D. C., Curran, L. M., Cerqueira, G. C., Garcia, R. A., Ramos, C. A., Mcdonald, A., Lefebvre, P., & Schlesinger, P. (2006). Modelling conservation in the Amazon Basin. Nature 440: 520-523. *Geophysical Research Letters*, *33*, L12704.

Sobota, D. J., Compton, J. E., & Harrison, J. A. (2013). Reactive nitrogen inputs to US lands and waterways: how certain are we about sources and fluxes? *Frontiers in Ecology and the Environment*, *11*(2), 82-90.

Somers, B., Asner, G. P., Martin, R. E., Anderson, C. B., Knapp, D. E., Wright, S. J., & Van De Kerchove, R. (2015). Mesoscale assessment of changes in tropical tree species richness across a bioclimatic gradient in Panama using airborne imaging spectroscopy. *Remote Sensing of Environment*, *167*, 111-120.

Son, J. Y., Lee, H. J., Koutrakis, P., & Bell, M. L. (2017). Pregnancy and Lifetime Exposure to Fine Particulate Matter and Infant Mortality in Massachusetts, 2001–2007. *American journal of epidemiology*, *186*(11), 1268-1276.

Sorensson, A. A., Menéndez, C. G., Ruscica, R., Alexander, P., Samuelsson, P., & Willén, U. (2010). Projected precipitation changes in South America: a dynamical downscaling within CLARIS. *Meteorologische Zeitschrift*, *19*(4), 347-355.

Speziale, K. L., Lambertucci, S. A., Carrete, M., & Tella, J. L. (2012). Dealing with non-native species: what makes the difference in South America?. *Biological Invasions*, *14*(8), 1609-1621.

Sprague, L. A., Hirsch, R. M., & Aulenbach, B. T. (2011). Nitrate in the Mississippi River and its tributaries, 1980 to 2008: Are we making progress?. *Environmental Science & Technology*, *45*(17), 7209-7216.

Steiner, N. S., Christian, J. R., Six, K. D., Yamamoto, A., & Yamamoto-Kawai, M. (2014). Future ocean acidification in the Canada Basin and surrounding Arctic Ocean from CMIP5 earth system models. *Journal of Geophysical Research: Oceans*, *119*(1), 332-347.

Sténs, A., Bjärstig, T., Nordström, E. M., Sandström, C., Fries, C., & Johansson, J. (2016). In the eye of the stakeholder: The challenges of governing social forest values. *Ambio*, *45*(2), 87-99.

Stephenson, P. J., Bowles-Newark, N., Regan, E., Stanwell-Smith, D., Diagana, M., Höft, R., Abarchi, H., Abrehamse, T., Akello, C., Allison, H., Banki, O., Batieno, B., Dieme, S., Domingos, A., Galt, R., Githaiga, C., Bine, A., Hafashimana, D., & Thiombiano, A. (2017). Unblocking the flow of biodiversity data for decision-making in Africa. *Biological Conservation*, *213*, 335-340.

Stocker, T. F., Qin, D., Plattner, G. -K., Tignor, M., Allen, S. K., Boschung, J., Nauels, A., Xia, Y., Bex V., & Midgley P.M. (Eds.). (2013.) Summary for Policymakers. In *Climate Change 2013: The Physical Science Basis. Contribution of Working Group I to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change*. Cambridge, United Kingdom and New York, NY: Cambridge University Press, USA.

Stohlgren, T. J., Jarnevich, C., Chong, G. W., & Evangelista, P. H. (2006). Scale and plant invasions: a theory of biotic acceptance. *Preslia*, *78*(4), 405-426.

Stohlgren, T. J., Schell, L. D., & Heuvel, B. V. (1999). How grazing and soil quality affect native and exotic plant diversity in Rocky Mountain grasslands. *Ecological Applications*, *9*(1), 45-64.

Stolton, S., Redford, K. H., & Dudley, N. (2014). The futures of privately protected areas. Gland, Switzerland: IUCN.

Stone, W. W., Gilliom, R. J., & Ryberg, K. R. (2014). Pesticides in US streams and rivers: occurrence and trends during 1992–2011. *Environmental Science & Technology*, 48(19), 11025-30.

Stoner, A. W. (1997). Shell middens as indicators of long-term distributional pattern in Strombus gigas, a heavily exploited marine gastropod. *Bulletin of marine science*, *61*(3), 559-570.

Stoner, A. W., & Ray, M. (1996). Queen conch, Strombus gigas, in fished and unfished locations of the Bahamas: effects of a marine fishery reserve on adults, juveniles, and larval production. *Fishery Bulletin*, *94*(3), 551-556.

Strecker, U. (2006). The impact of invasive fish on an endemic Cyprinodon species flock (Teleostei) from Laguna Chichancanab, Yucatan, Mexico. *Ecology of Freshwater Fish*, *15*(4), 408-418.

Suárez. A., Garraway, E., Vilamajo, D., Mujica, L., Gerhartz, J., Capote, R., & Blake N. (2008) *Climate change impacts on terrestrial biodiversity in the insular Caribbean: Report of Working Group III, Climate Change and Biodiversity in the Insular Caribbean*. CANARI Technical Report No.383.

Sunda, W. (2012). Feedback interactions between trace metal nutrients and phytoplankton in the ocean. *Frontiers in Microbiology*, *3*, 204.

Surber, S. J., & Simonton, D. S. (2017). Disparate impacts of coal mining and reclamation concerns for West Virginia and central Appalachia. *Resources Policy*, *54*, 1-8.

Sutherland, K. P., Porter, J. W., & Torres, C. (2004). Disease and immunity in Caribbean and Indo-Pacific zooxanthellate corals. *Marine Ecology Progress Series*, *266*, 273-302.

Sutton, M. A., Oenema, O., Erisman, J. W., Leip, A., van Grinsven, H., & Winiwarter, W. (2011). Too much of a good thing. *Nature*, *472*(7342), 159.

Sutton, R., Sedlak, M. D., Yee, D., Davis, J. A., Crane, D., Grace, R., & Arsem, N. (2014). Declines in polybrominated diphenyl ether contamination of San Francisco Bay following production phase-outs and bans. *Environmental science & technology*, *49*(2), 777-784.

Swain, E. B., Engstrom, D. R., Brigham, M. E., Henning, T. A., & Brezonik, P. L. (1992). Increasing rates of atmospheric mercury deposition in midcontinental North America. *Science*, *257*(5071), 784-787.

Swenson, J. J., Carter, C. E., Domec, J. C., & Delgado, C. I. (2011). Gold mining in the Peruvian Amazon: global prices, deforestation, and mercury imports. *PloS one*, *6*(4), e18875.

Taketani, R. G., Franco, N. O., Rosado, A. S., & van Elsas, J. D. (2010). Microbial community response to a simulated hydrocarbon spill in mangrove sediments. *The Journal of Microbiology*, *48*(1), 7-15.

Telmer, K. H., & Veiga, M. M. (2009). World emissions of mercury from artisanal and small scale gold mining. In *Mercury fate and transport in the global atmosphere* (pp. 131-172). Springer, Boston, MA.

Texeira, M., Oyarzabal, M., Pineiro, G., Baeza, S., & Paruelo, J. M. (2015). Land cover and precipitation controls over long-term trends in carbon gains in the grassland biome of South America. *Ecosphere*, *6*(10), 1-21.

The World Bank Database. (2017): https://data.worldbank.org/indicator/

The World Bank. (2017). Terrestrial and Marine protected areas. Available at: http://data.worldbank.org/indicator/ER.PTD.TOTL.ZS?end=2014&page=5&start=2014&view=bar

The World Bank (2016). Global Economic Prospects (2016). Latin America and the Caribbean, Washington

Thomas, M. A. (2010). What do the worldwide governance indicators measure? *The European Journal of Development Research*, *22*(1), 31-54.

Thompson, J., Charpentier, A., Bouguet, G., Charmasson, F., Roset, S., Buatois, B., Vernet, P., & Gouyon, P. H. (2013). Evolution of a genetic polymorphism with climate change in a Mediterranean landscape. *Proceedings of the National Academy of Sciences*, *110*(8), 2893-2897.

Tijoux, M. E. (2016). *Naturaleza Americana. Extractivismo y geopolítica del capital. Actuel Marx N° 19: Naturaleza Americana. Extractivismo y geopolítica del capital. Actuel Marx N° 19*. LOM Ediciones.

Tilman, D., & Clark, M. (2014). Global diets link environmental sustainability and human health. *Nature*, *515*(7528), 518.

Tilman, D., Balzer, C., Hill, J., & Befort, B. L. (2011). Global food demand and the sustainable intensification of agriculture. *Proceedings of the National Academy of Sciences*, *108*(50), 20260-20264.

Tilman, D., Cassman, K. G., Matson, P. A., Naylor, R., & Polasky, S. (2002). Agricultural sustainability and intensive production practices. *Nature*, *418*(6898), 671.

Tlaiye, Laura E.; Aryal, Dinesh. (2013). Latin America and Caribbean Region Environment and Water Resources occasional paper series. Washington DC; World Bank. http://documents.worldbank.org/curated/en/620071468054545005/Ampliando-el-financiamiento-para-la-conservacion-de-la-biodiversidad-las-experiencias-de-America-Latina-y-el-Caribe

Tófoli, R. M., Dias, R. M., Alves, G. H. Z., Hoeinghaus, D. J., Gomes, L. C., Baumgartner, M. T., & Agostinho, A. A. (2017). Gold at what cost? Another megaproject threatens biodiversity in the Amazon. *Perspectives in Ecology and Conservation*, *15*(2), 129-131.

Tognetti, P. M., & Chaneton, E. J. (2015). Community disassembly and invasion of remnant native grasslands under fluctuating resource supply. *Journal of applied ecology*, *52*(1), 119-128.

Tognetti, P. M., Chaneton, E. J., Omacini, M., Trebino, H. J., & León, R. J. (2010). Exotic vs. native plant dominance over 20 years of old-field succession on set-aside farmland in Argentina. *Biological Conservation*, *143*(11), 2494-2503.

Tovar, C., Arnillas, C. A., Cuesta, F., & Buytaert, W. (2013). Diverging responses of tropical Andean biomes under future climate conditions. *PloS One, 8*(5), e63634.

Townsend, J. M., Rimmer, C. C., Driscoll, C. T., McFarland, K. P., & Inigo-Elias, E. (2013). Mercury concentrations in tropical resident and migrant songbirds on Hispaniola. *Ecotoxicology, 22*(1), 86-93.

Trace, S. (2016). *Rethink, Retool, Reboot: Technology as if people and planet mattered*: Practical Action Publishing Rugby, UK.

Twiss, M. R., Rattan, K. J., Sherrell, R. M., & McKay, R. M. L. (2004). Sensitivity of phytoplankton to copper in Lake Superior. *Journal of Great Lakes Research, 30*, 245-255.

U.S. Department of the Interior Fish and Wildlife Service. (2014). Notice of Intent to Include Four Native U.S. Freshwater Turtle Species in Appendix III of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES).

UN Habitat. (2016). (2016). Urbanization and Development Emerging Futures. World Cities Report: Nairobi, Kenya: United Nations Human Settlement Programme (UN-Habitat).

UNDP. (2014). Human development report 2014. United Nations Development Programme. http://hdr.undp.org/en/media/HDR_2013_EN_complete.pdf

UNDP. (2016). Human development report 2016. United Nations Development Programme. http://hdr.undp.org/sites/default/files/2016_human_development_report.pdf

UNEP. (2016). *Regional Cooperation for Environmental Sustainability in the Latin American and Caribbean Region*. XX Meeting of the Forum of Ministers of the Environment of Latin America and the Caribbean. Cartagena, Colombia.

UNEP. (2010). Latin America and the Caribbean: Environmental outlook. Relations between Environmental Changes and Human Well Being in Latin America and the Caribbean. United Nations Environment Programme.

UNEP. (2014). The Importance of Mangroves to People: A Call to Action. J. van Bochove, E. Sullivan, and T. Nakamura, editors. United Nations Environment Programme, World Conservation Monitoring Centre, Cambridge. 128 pp.

UNEP. (2009). Persistent Organic Pollutants Along Environmental Transects in Costa Rica, Chile, Nepal, and Botswana. United Nations Environment Programme.

UNESCO. (2006). UNESCO Science Report. Towards 2030. UNESCO Publishing. Second revised edition. Paris.

United Nations Conference on Trade and Development (UNCTAD) (2016). UNCTAD Handbook of Statistics, New York: www.unctad.org

United Nations. (2016). Report of the Special Rapporteur on the rights of indigenous peoples on her visit to Honduras. Human Rights Council. Thirty-third session. Agenda item 3. Promotion and protection of all human rights, civil, political, economic, social and cultural rights, including the right to development 21 July 2016. United Nations, A/HRC/33/42/Add.2

United Nations, Department of Economic and Social Affairs, Population Division (2014). World Urbanization Prospects: The 2014 Revision, Highlights (ST/ESA/SER.A/352).

Uribe B., E. (2015). Estudios del cambio climático en América Latina: El cambio climático y sus efectos en la biodiversidad en América Latina. CEPAL. https://repositorio.cepal.org/handle/11362/39855

Valdez-Moreno, M., Quintal-Lizama, C., Gómez-Lozano, R., & García-Rivas, M. d. C. (2012). Monitoring an Alien Invasion: DNA Barcoding and the Identification of Lionfish and Their Prey on Coral Reefs of the Mexican Caribbean. *PloS One, 7*(6), e36636. doi:10.1371/journal.pone.0036636

Vales García, M. A., Álvarez de Zayas, A., Montes, L., & Ávila, A. (1998). Estudio nacional sobre la diversidad biológica en la República de Cuba. In: Centro Nacional de Biodiversidad.

Valiela, I., Bowen, J. L., & York, J. K. (2001). Mangrove Forests: One of the World's Threatened Major Tropical Environments: At least 35% of the area of mangrove forests has been lost in the past two decades, losses that exceed those for tropical rain forests and coral reefs, two other well-known threatened environments. *AIBS Bulletin, 51*(10), 807-815.

Valliere, J. M., Irvine, I. C., Santiago, L., & Allen, E. B. (2017). High N, dry: experimental nitrogen deposition exacerbates native shrub loss and nonnative plant invasion during extreme drought. *Global Change Biology, 23*(10), 4333-4345.

Van Aardenne, J., Dentener, F., Olivier, J., Goldewijk, C. K., & Lelieveld, J. (2001). A 1× 1 resolution data set of historical anthropogenic trace gas emissions for the period 1890–1990. *Global Biogeochemical Cycles, 15*(4), 909-928.

van Andel, T., van der Hoorn, B., Stech, M., Arostegui, S. B., & Miller, J. (2016). A quantitative assessment of the vegetation types on the island of St. Eustatius, Dutch Caribbean. *Global Ecology and Conservation, 7*, 59-69.

Van Beusekom, A. E., González, G., & Rivera, M. M. (2015). Short-term precipitation and temperature trends along an elevation gradient in northeastern Puerto Rico. *Earth Interactions, 19*(3), 1-33.

Van der Burg, W., De Freitas, J., Debrot, A., & Lotz, L. (2012). *Naturalised and invasive alien plant species in the Caribbean Netherlands: status, distribution, threats, priorities and recommendations: report of a joint Imares/Carmabi/PRI project financed by the Dutch Ministry of Economic Affairs, Agriculture & Innovation*. Retrieved from

Van Hooidonk, R., Maynard, J. A., Liu, Y., & Lee, S. K. (2015). Downscaled projections of Caribbean coral bleaching that can inform conservation planning. *Global Change Biology, 21*(9), 3389-3401.

Van Kleunen, M., Dawson, W., Essl, F., Pergl, J., Winter, M., Weber, E., Kreft, H., Weigelt, P., Kartesz, J., & Nishino, M. (2015). Global exchange and accumulation of non-native plants. *Nature, 525*(7567), 100.

Van Lexmond, M. B., Bonmatin, J.-M., Goulson, D., & Noome, D. A. (2015). Worldwide integrated assessment on systemic pesticides. In: Springer.

van Ruijven, B. J., Daenzer, K., Fisher-Vanden, K., Kober, T., Paltsev, S., Beach, R. H., Calderon, S. L., Calvin, K., Labriet, M., & Kitous, A. (2016). Baseline projections for Latin America: base-year assumptions, key drivers and greenhouse emissions. *Energy Economics, 56*, 499-512.

Van Wagner, V. (1978). Age-class distribution and the forest fire cycle. Canadian Journal of Forest Research, 8(2), 220-227.

Vanhulst, J., & Beling, A. E. (2014). Buen vivir: Emergent discourse within or beyond sustainable development? *Ecological Economics, 101*, 54-63.

Vaquer-Sunyer, R., & Duarte, C. M. (2008). Thresholds of hypoxia for marine biodiversity. *Proceedings of the National Academy of Sciences, 105*(40), 15452-15457.

Vargas, C. A., Aguilera, V. M., San Martín, V., Manríquez, P. H., Navarro, J. M., Duarte, C., Torres, R., Lardies, M. A., & Lagos, N. A. (2015). CO_2-driven ocean acidification disrupts the filter feeding behavior in Chilean gastropod and bivalve species from different geographic localities. *Estuaries and coasts, 38*(4), 1163-1177.

Vargas, C.A., Contreras, P.Y., Pérez, C.A., Sobarzo, M., Saldías, G.S., & Salisbury, J. (2016). Influences of riverine and upwelling waters on the coastal carbonate system off Central Chile and their ocean acidification implications. *Journal of Geophysical Research: Biogeosciences*, 121(6), pp.1468-1483.

Vega-Rodriguez, M., Müller-Karger, F., Hallock, P., Quiles-Perez, G., Eakin, C., Colella, M., Jones, D., Li, J., Soto, I., & Guild, L. (2015). Influence of water-temperature variability on stony coral diversity in Florida Keys patch reefs. *Marine Ecology Progress Series, 528*, 173-186.

Veldman, J. W., & Putz, F. E. (2010). Long-distance dispersal of invasive grasses by logging vehicles in a tropical dry forest. *Biotropica, 42*(6), 697-703.

Vellend, M., Harmon, L. J., Lockwood, J. L., Mayfield, M. M., Hughes, A. R., Wares, J. P., & Sax, D. F. (2007). Effects of exotic species on evolutionary diversification. *Trends in ecology & evolution, 22*(9), 481-488.

Venier, M., & Hites, R. A. (2010). Time trend analysis of atmospheric POPs concentrations in the Great Lakes region since 1990. *Environmental science & technology, 44*(21), 8050-8055.

Venturini, N., Bícego, M. C., Taniguchi, S., Sasaki, S. T., García-Rodríguez, F., Brugnoli, E., & Muniz, P. (2015). A multi-molecular marker assessment of organic pollution in shore sediments from the Río de la Plata Estuary, SW Atlantic. *Marine pollution bulletin, 91*(2), 461-475.

Veuthey, S., & Gerber, J.-F. (2012). Accumulation by dispossession in coastal Ecuador: Shrimp farming, local resistance and the gender structure of mobilizations. *Global Environmental Change, 22*(3), 611-622.

Victoria, R., Martinelli, L., Moraes, J., Ballester, M., Krusche, A., Pellegrino, G., Almeida, R., & Richey, J. (1998). Surface air temperature variations in the Amazon region and its borders during this century. *Journal of Climate, 11*(5), 1105-1110.

Vignola, R., Locatelli, B., Martinez, C., & Imbach, P. (2009). Ecosystem-based adaptation to climate change: what role for policy-makers, society and scientists? *Mitigation and adaptation strategies for global change, 14*(8), 691.

Vila, M., & Ibáñez, I. (2011). Plant invasions in the landscape. *Landscape ecology, 26*(4), 461-472.

Villalba-Eguiluz, C. U., & Etxano, I. (2017). Buen Vivir vs development (II): the limits of (Neo-) Extractivism. *Ecological Economics, 138*, 1-11.

Villarini, G., Smith, J. A., Baeck, M. L., Vitolo, R., Stephenson, D. B., & Krajewski, W. F. (2011). On the frequency of heavy rainfall for the Midwest of the United States. *Journal of Hydrology, 400*(1-2), 103-120.

Villers-Ruiz, L., & Hernández-Lozano, J. (2007). *Incendios forestales y el fenómeno de El Niño en México.* Paper presented at the IV Conferencia Internacional sobre Incendios Forestales, Sevilla, España, May.

Vincent, L. A., Peterson, T., Barros, V., Marino, M., Rusticucci, M., Carrasco, G., Ramirez, E., Alves, L., Ambrizzi, T., & Berlato, M. (2005). Observed trends in indices of daily temperature extremes in South America 1960–2000. *Journal of Climate, 18*(23), 5011-5023.

Virtanen, J.K., Voutilainen, S., Rissanen, T.H., Mursu, J., Tuomainen, T.P., Korhonen, M.J., Valkonen, V.P., Seppänen, K., Laukkanen, J.A., & Salonen, J.T. (2005). Mercury, fish oils, and risk of acute coronary events and cardiovascular disease, coronary heart disease, and all-cause mortality in men in eastern Finland. *Arteriosclerosis, thrombosis, and vascular biology,* 25(1), pp.228-233.

Višnjevec, A.M., Kocman, D. and Horvat, M. (2014). Human mercury exposure and effects in

Europe. *Environmental toxicology and chemistry*, 33(6), pp.1259-1270.

Vogt, N. D., M. Pinedo-Vasquez, E. S. Brondízio, O. Almeida, and S. Rivero. (2015). Forest Transitions in Mosaic Landscapes: Smallholder's Flexibility in Land-Resource Use Decisions and Livelihood Strategies From World War II to the Present in the Amazon Estuary. *Society & Natural Resources* 28:1043–1058.

Volante, J., Mosciaro, J., Morales Poclava, M., Vale, L., Castrillo, S., Sawchik, J., Tiscornia, G., Maldonado, I., Vega, A., & Trujillo, R. (2015). Expansión agrícola en Argentina, Bolivia, Paraguay, Uruguay y Chile entre 2000-2010: Caracterización espacial mediante series temporales de índices de vegetación. *RIA. Revista de investigaciones agropecuarias, 41*(2), 179-191.

Volk, M., P. Bungener, F. Contat, M. Montani, & J. Fuhrer. (2006). Grassland yield declined by a quarter in 5 years of free-air ozone fumigation. *Global Change Biology* 12:74–83.

Vos, Vincent A.; Olver Vaca, & Adrián Cruz (2015). "Sistemas Agroforestales en la Amazonía boliviana. Una valoración de sus múltiples funciones", Cuadernos de Investigación 82, La Paz, junio de 2015.

Vranken, L., Avermaete, T., Petalios, D., & Mathijs, E. (2014). Curbing global meat consumption: emerging evidence of a second nutrition transition. *Environmental Science & Policy, 39*, 95-106.

Vuille, M., Franquist, E., Garreaud, R., Casimiro, W. S. L., & Cáceres, B. (2015). Impact of the global warming hiatus on Andean temperature. *Journal of Geophysical Research: Atmospheres, 120*(9), 3745-3757.

Vuille, M., R. S. Bradley, M. Werner, & F. Keimig. (2003). 20Th Century Climate Change in the Tropical Andes: Observations and Model Results. *Climatic Change* 59:75–99.

Wallem, P.K., C.B. Anderson, G. Martínez Pastur & M.V. Lencinas (2010). Community re-assembly by an exotic herbivore, *Castor canadensis*, in subantarctic forests, Chile and Argentina. *Biological Invasions* 12: 325-335. Special edition on invasive herbivores.

Walters, W. *(2016). Border/control. In* An anthology of migration and social transformation (pp. 151-165). Springer International Publishing.

Ward, J. M., & Ricciardi, A. (2007). Impacts of Dreissena invasions on benthic macroinvertebrate communities: A meta-analysis: Biodiversity research. *Diversity and Distributions*, 13(2), 155–165. https://doi.org/10.1111/j.1472-4642.2007.00336.x

Ward, R. D., D. A. Friess, R. H. Day, & R. A. MacKenzie. (2016). Impacts of climate change on mangrove ecosystems: a region by region overview. Ecosystem Health and Sustainability 2:e01211.

Wardle D.A., Bardgett R.D., Callaway R.M. & Van Der Putten W.H. (2011). Terrestrial ecosystem responses to species gains and losses. Science, 332, 1273-1277

Washington, S., & Ababouch, L. (2011). Private standards and certification in fisheries and aquaculture: current practice and emerging issues. FAO Fisheries and Aquaculture Technical Paper. http://doi.org/10.1017/CBO9781107415324.004

Weber, B., Belnap, J., & Büdel, B. (2016). Synthesis on Biological Soil Crust Research. In Biological Soil Crusts: An Organizing Principle in *Drylands* (pp. 527-534). Springer International Publishing.

Webster, N. S., S. Uthicke, E. S. Botté, F. Flores, , & A. P. Negri. 2013. Ocean acidification reduces induction of coral settlement by crustose coralline algae. *Global Change Biology* 19:303–315.

Weihe, P., Hansen, J.C., Murata, K., Debes, F., Jørgensen, P.J., Steuerwald, U., White, R.F., & Grandjean, P. (2002). Neurobehavioral performance of Inuit children with increased prenatal exposure to methylmercury. International Journal of Circumpolar Health, 61(1), pp.41-49.

Weinhold, D., Killick, E., & Reis, E. J. (2013). Soybeans, poverty and inequality in the Brazilian Amazon. *World Development, 52*, 132-143.

Weiser, E. L., & Powell, A. N. (2010). Does garbage in the diet improve reproductive output of Glaucous Gulls? *The Condor, 112*(3), 530-538.

Weiss-Penzias, P. S., Gay, D. A., Brigham, M. E., Parsons, M. T., Gustin, M. S., & ter Schure, A. (2016). Trends in mercury wet deposition and mercury air concentrations across the US and Canada. *Science of the Total Environment, 568*, 546-556.

Weiss, J. L., & Overpeck, J. T. (2005). Is the Sonoran Desert losing its cool? *Global Change Biology, 11*(12), 2065-2077.

Welcomme, R. (1999). A review of a model for qualitative evaluation of exploitation levels in multi-species fisheries. *Fisheries Management and Ecology, 6*(1), 1-19.

Welcomme, R. L., Cowx, I. G., Coates, D., Béné, C., Funge-Smith, S., Halls, A., & Lorenzen, K. (2010). Inland capture fisheries. *Philosophical Transactions of the Royal Society of London B: Biological Sciences, 365*(1554), 2881-2896.

Welle, T., Birkmann, J., Rhyner, J., Witting, M., & Wolfertz, J. (2012). World risk index 2012: Concept, updating and results. *World risk report*, 11-26.

Wertin, T. M., Reed, S. C., & Belnap, J. (2015). C 3 and C 4 plant responses to increased temperatures and altered monsoonal precipitation in a cool desert on the Colorado Plateau, USA. *Oecologia, 177*(4), 997-1013.

Whitfield, P. E., Gardner, T., Vives, S. P., Gilligan, M. R., Courtenay Jr, W. R., Ray, G. C., & Hare, J. A. (2002). Biological invasion of the Indo-Pacific lionfish Pterois volitans along the Atlantic coast of North America. *Marine Ecology Progress Series, 235*, 289-297.

Wiebusch, R. K., & Lant, C. L. (2017). Policy Drivers of US Wetland Conversion Rates, 1955–2009. *Society & natural resources, 30*(1), 16-30.

Wiedenfeld, D., Crawford, R., & Pott, C. (2015). Results of a Workshop on.

Wik, M., Pingali, P., & Brocai, S. (2008). Global agricultural performance: past trends and future prospects.

Wilcove, D. S., Rothstein, D., Dubow, J., Phillips, A., & Losos, E. (1998). Quantifying threats to imperiled species in the United States. *BioScience, 48*(8), 607-615.

Wilcox, C., Mallos, N. J., Leonard, G. H., Rodriguez, A., & Hardesty, B. D. (2016). Using expert elicitation to estimate the impacts of plastic pollution on marine wildlife. *Marine Policy, 65*, 107-114.

Wilkie, D. S., & Godoy, R. A. (2001). Income and price elasticities of bushmeat demand in lowland Amerindian societies. *Conservation Biology, 15*(3), 761-769.

Wilkinson, C. R., & Souter, D. N. (2008). *Status of Caribbean coral reefs after bleaching and hurricanes in 2005* (Vol. 148): Global Coral Reef Monitoring Network.

Willer, H., & Lernoud, J. (2016). *The world of organic agriculture. Statistics and emerging trends 2016*: Research Institute of Organic Agriculture FiBL and IFOAM Organics International.

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.

Wirtz, D., Sorensen, D. C., & Haasdonk, B. (2014). A posteriori error estimation for DEIM reduced nonlinear dynamical systems. *SIAM Journal on Scientific Computing, 36*(2), A311-A338.

Wittmann, F., Schöngart, J., Montero, J. C., Motzer, T., Junk, W. J., Piedade, M. T., Queiroz, H. L., & Worbes, M. (2006). Tree species composition and diversity gradients in white-water forests across the Amazon Basin. *Journal of biogeography, 33*(8), 1334-1347.

Wolfe, B. T., & Van Bloem, S. J. (2012). Subtropical dry forest regeneration in grass-invaded areas of Puerto Rico: understanding why Leucaena leucocephala dominates and native species fail. *Forest Ecology and Management, 267*, 253-261.

Wolfe, S. A., Griffith, B., & Wolfe, C. A. G. (2000). Response of reindeer and caribou to human activities. *Polar Research, 19*(1), 63-73.

Woodley, S., Bertzky, B., Crawhall, N., Dudley, N., Londoño, J. M., MacKinnon, K., Redford, K., & Sandwith, T. (2012). Meeting Aichi Target 11: what does success look like for protected area systems. *Parks, 18*(1), 23-36.

Woodruff, T. J., Grillo, J., & Schoendorf, K. C. (1997). The relationship between selected causes of postneonatal infant mortality and particulate air pollution in the United States. *Environmental health perspectives, 105*(6), 608.

Woody, C. A., Hughes, R. M., Wagner, E. J., Quinn, T. P., Roulson, L. H., Martin, L. M., & Griswold, K. (2010). The Mining Law of 1872: Change is Overdue. Fisheries, 35(7), 321–331. https://doi.org/10.1577/1548-8446-35.7.321

Wooldridge, S. A., & Done, T. J. (2009). Improved water quality can ameliorate effects of climate change on corals. *Ecological Applications, 19*(6), 1492-1499.

Worldometers (2017). Accessed 2 May 2017, and 3 September 2017 at: http://www.worldometers.info/world-population/population-by-region/

Worm, B., Lotze, H. K., Jubinville, I., Wilcox, C., & Jambeck, J. (2017). Plastic as a persistent marine pollutant. *Annual Review of Environment and Resources, 42*.

Wortley, L., Hero, J. M., & Howes, M. (2013). Evaluating ecological restoration success: a review of the literature. *Restoration Ecology, 21*(5), 537-543.

Wright, C. K., & Wimberly, M. C. (2013). Recent land use change in the Western Corn Belt threatens grasslands and wetlands. *Proceedings of the National Academy of Sciences, 110*(10), 4134-4139.

Wu, Y., Wang, S., Streets, D. G., Hao, J., Chan, M., & Jiang, J. (2006). Trends in anthropogenic mercury emissions in China from 1995 to 2003. *Environmental science & technology, 40*(17), 5312-5318.

Wurster, C. F., & Wingate, D. B. (1968). DDT residues and declining reproduction in the Bermuda petrel. *Science, 159*(3818), 979-981.

Wurster, D. H., Wurster Jr, C. F., & Strickland, W. N. (1965). Bird mortality following DDT spray for Dutch elm disease. *Ecology, 46*(4), 488-499.

WWF. (2014). Living Planet Report 2014: Species and spaces, people and places. Research accounting. World Wide Fund for Nature.

WWF. (2016). *Living Planet: Report 2016: Risk and Resilience in a New Era*: World Wide Fund for Nature.

Yang, Z., Wang, T., Leung, R., Hibbard, K., Janetos, T., Kraucunas, I., Rice, J., Preston, B., & Wilbanks, T. (2014). A modeling study of coastal inundation induced by storm surge, sea-level rise, and subsidence in the Gulf of Mexico. *Natural hazards, 71*(3), 1771-1794.

Yee, S. H., Santavy, D. L., & Barron, M. G. (2008). Comparing environmental influences on coral bleaching across and within species using clustered binomial regression. *Ecological Modelling, 218*(1-2), 162-174.

Zagarola, J.-P. A., Anderson, C. B., & Veteto, J. R. (2014). Perceiving Patagonia: an assessment of social values and perspectives regarding watershed ecosystem services and management in southern South America. *Environmental Management, 53*(4), 769-782.

Zambrano, L., Valiente, E., & Vander Zanden, M. J. (2010). Food web overlap among native axolotl (Ambystoma mexicanum) and two exotic fishes: carp (Cyprinus carpio) and tilapia (Oreochromis niloticus) in Xochimilco, Mexico City. *Biological Invasions, 12*(9), 3061-3069.

Zenni, R. D., & Ziller, S. R. (2011). An overview of invasive plants in Brazil. *Brazilian Journal of Botany, 34*(3), 431-446.

Zenni, R. D., Ziller, S. R., Pauchard, A., Rodriguez-Cabal, M., & Nuñez, M. A. (2017). Invasion science in the developing world: A response to Ricciardi *et al. Trends in ecology & evolution, 32*(11), 807-808.

Zéphyr, P. M. D., Guillén, A. C., Salgado, H., & Seligson, M. A. (2011). *Haiti in distress: The impact of the 2010 earthquake on citizen lives and perceptions*: LAPOP.

Zhou, Y., Michalak, A. M., Beletsky, D., Rao, Y. R., & Richards, R. P. (2015). Record-breaking Lake Erie hypoxia during 2012 drought. *Environmental science & technology, 49*(2), 800-807.

Zilberman, D., Hochman, G., Rajagopal, D., Sexton, S., & Timilsina, G. (2012). The impact of biofuels on commodity food prices: Assessment of findings. *American Journal of Agricultural Economics, 95*(2), 275-281.

Ziller, S. R., Reaser, J. K., Neville, L. E., & Brand, K. (2005). Invasive alien species in South America: national reports & directory of resources. Prevention and Management of Invasive Alien Species: Forging Cooperation throughout South America, 114 p. Retrieved from www.gisp.org

Zimmerle, D. J., Williams, L. L., Vaughn, T. L., Quinn, C., Subramanian, R., Duggan, G. P., Willson, B., Opsomer, J. D., Marchese, A. J., & Martinez, D. M. (2015). Methane emissions from the natural gas transmission and storage system in the United States. *Environmental science & technology, 49*(15), 9374-9383.

Zulkafli, Z., Buytaert, W., Manz, B., Rosas, C. V., Willems, P., Lavado-Casimiro, W., Guyot, J.-L., & Santini, W. (2016). Projected increases in the annual flood pulse of the Western Amazon. *Environmental Research Letters, 11*(1), 014013.