6 Forest Biodiversity and Ecosystem Services: Drivers of Change, Responses and Challenges

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Abstract: We briefly describe four important anthropogenic drivers of change in biodiversity using literature with particular reference to cases from Latin America and the Mediterranean Basin. Conversion of forests into agricultural lands, over-exploitation, air pollution leading to climate change and acid rain, and invasive species, all cause great stress on forest ecosystems. Conscious of the negative effects of human activities, society has responded by increasing the area of protected and well-managed forests, and by incorporating management of trees and forest patches into the management of agricultural landscapes. Still, most natural forests and agricultural landscapes are not well managed and their existence continues to be threatened by these same four drivers. We propose elements of a new vision of biodiversity and ecosystem services management based on our own experiences and the evidence found in the studies we examined. We suggest a positive approach to forest conservation, combining aspects of willingness to conserve with willingness to pay for further conservation; removal of administrative barriers to good forest management and protection; landscape management; inter-sectoral coordination between international, national, and local policies; increased communication among stakeholders; and more research on the interactions between biodiversity and ecosystem services.

Keywords: biodiversity, ecosystem services, anthropogenic drivers, protected areas, forest management, agricultural landscapes, integration

6.1 Introduction

The Convention on Biological Diversity (CBD) defines biological diversity as "the variability among living organisms from all sources, including, *inter alia*, terrestrial, marine and other aquatic ecosystems, and the ecological complexes of which they are part. This includes diversity within species, between species, and of ecosystems" (CBD 1992). Biodiversity is complex and difficult to quantify, but is much more than the number of species in a community (species richness), or that number weighted by species abundance (species diversity). Biodiversity encompasses differences in species composition, their genetics, and even the functional roles that species play within an ecosystem (responses to change and effects on ecosystem processes).

Ecosystem services have been defined as the benefits that people obtain from ecosystems (MEA 2005). The Millennium Ecosystem Assessment (MEA) (Díaz et al. 2005, MEA 2005) proposes four groups of ecosystem services: provisioning services, supporting services, regulating services, and cultural services. Many of these services are directly related to biodiversity.

The relationship between biological diversity and ecosystem services is complex and often poorly understood (Díaz et al 2005, Kremen 2005). In some cases, for example under Costa Rica's 1996 forest law, biological diversity itself is considered to be a service. A more scientific approach, however, considers biodiversity as a mechanism through which services are provided (CBD 2008). Changes in the species richness, abundance, and composition of an ecosystem may lead to parallel changes in the amount or quality of services provided by that ecosystem, including carbon sequestration (Bunker et al. 2005), pollination (Ricketts 2004), or pest control (Phillpott et al. 2009), that are indicative of a linear relationship between biodiversity and ecosystem services (Schwartz et al. 2000).

Biodiversity can also be divided into functional components or groups where species are classified according to their functional response to ecosystem change (e.g., drought tolerance) or by their contribution to ecosystem services (e.g., nitrogen fixing). Many plant and animal communities have shown functional complementarity or redundancy, up to a certain minimum level of species, causing ecosystem functionality to rapidly decline if species numbers drop below that level (Flynn et al. 2009). Kremen (2005) calls this a saturation function. A meta-analysis of 446 measures of biodiversity effects (319 of which involved primary-producer manipulations or measures) conducted by Balvanera et al. (2006) on temperate non-forested ecosystems suggests that there is a critical species number between 10 and 20 beyond which diversity effects on ecosystem services decrease because of increasing redundancy and competition between species, but it remains to be confirmed whether such a threshold value also applies to other ecosystems.

Kremen (2005) and Haines-Young and Potschin (in press) add to these two possible cases a third scenario of accelerating and rapidly declining ecological functionality with species lost, mainly due to the dominance of a few species with high ecological functionality within communities (Kremen 2005, Haines-Young and Potschin in press), and/or intensive interaction among species and between species and processes (Kremen 2005).

While the link between biodiversity and ecosystem services is not always clear or direct, the link between the adaptive capacity of ecosystems to changes or disturbances is better documented (Díaz et al. 2005), particularly in species-poor ecosystems. Balvanera et al. (2006) found that resistance to invasion increased with greater species richness. In contrast, resistance to drought and other disturbances, such as windfall, did not (Balvanera et al. 2006). Acosta et al. (2001) also found that managed and un-managed stands of tropical broadleaf forests were equally affected by hurricane Mitch, but that the former showed greater recovery and diversity in the first years after the storm than did the un-managed stands. DeClerck et al. (2006) found similar results in a 60-year study of conifer forest drought resistance and resilience in the Sierra Nevada of California, although they did find that greater conifer diversity increased the resilience of these forest stands in the sense that they returned to pre-disturbance productivity levels much faster than did species-poor stands. In this sense, biodiversity provides insurance against dramatic ecosystem change and helps maintain stability within systems (Walker 1992).

Biological diversity has been continuously changing for as long as life has existed on Earth as a response to both natural and, more recently, anthropogenic changes in the environment (MEA 2005). There are six mass extinctions recorded in the geological record. Many scientists, including Harvard biologist, E.O. Wilson, argue that we are now in the middle of the seventh mass extinction. This time, however, *Homo sapiens* is the driving cause rather than any geological or astronomical phenomena (Wilson 1994). Major anthropogenic drivers of the current changes being observed include the degradation and loss of ecosystems due to changes in land uses, human impacts on the biogeochemical cycles (e.g., climate change, pollution), invasive species that displace or outcompete endemic species, and poor management or over-exploitation of the natural resource base (Díaz et al. 2005, Kanninen et al. 2007, Fischlin et al. 2009).

In this chapter, we propose to provide an overview of these major anthropogenic drivers of biodiversity change and their impacts on the provision of ecosystem services (section 6.2). In section 6.3, we analyse how society has responded to these changes in order to reduce their negative impacts and/or enhance their positive impacts on human well-being. We will close with the lessons learned from these responses regarding biodiversity management and the provision of ecosystem services (section 6.4).

6.2 Drivers of Change in Biodiversity and Their Importance for Ecosystem Services

Drivers (or direct causes) of change in biodiversity may be natural or human-induced. Many human interventions in ecosystems generate abrupt and large scale changes that trigger loss of biodiversity and make it more difficult for ecosystems to recover from the negative impacts associated with these human interventions. Ecosystem recovery from such human-induced change is not only slow and costly, in some cases, ecosystem changes may be irreversible (Ellatifi 2005). Here we focus on human-induced changes on forest biomes. Since these may differ according to geographic location and actors involved, it is impossible to discuss all possible variations. Ecosystem services provided by forest ecosystems can vary according to biome, geographic location, and socio-economic and cultural contexts (for examples, see Fischlin et al. 2009). We have chosen some clear examples of drivers and describe their effects on forests, with particular reference to locales in North and South America and in the Mediterranean Basin.

Photo 6.1 Recently deforested area in Norte do Mato Grasso, Brazil.

6.2.1 Conversion of Natural Forest Ecosystems to Other Land Covers

Between 2000 and 2005, the world lost about 7.3 million ha of natural forests annually (FAO 2009); before 2000, this amount often exceeded 13 million ha/year (FAO 2006). Agriculture (food crops and livestock) remains the largest direct human-induced driver of biodiversity degradation, species loss, and conversion of natural habitats (Ellatifi 2004, MEA 2005). In addition to agriculture as a direct driver, many other well-known indirect drivers of deforestation also exist (Kanninen et al. 2007), with the subsequent loss of biodiversity. Many of these are related to people's livelihoods, which often make it more attractive to convert forest into agricultural land than to manage the forest. Difficult and slow administrative procedures to obtain permits for forest use, subsidies to farm inputs or farm product exports, policies that recognise deforestation as land improvement are just some of the factors mentioned by Geist and Lambin (2002), Kanninen et al. (2007) and FAO (2009). Particularly in emerging countries these factors are exacerbated by population growth, governance limitations (unequal and unclear access to resources, lack of transparency in decision-making, deficient normative frameworks, insufficient resources for implementation and enforcement of laws and regulations), growing international demand for agricultural products, increasing interest in biofuels, incoherent sector policies, local cultural and demographic factors, climate change, and perverse subsidies.. The complex interactions of these drivers have made it difficult to change the course of deforestation, particularly in developing countries. Apart from differences in deforestation rates between developed and developing countries (MEA 2005; FAO 2006, 2009), there are also clear differences in deforestation rates between forest biomes and within the forest types of each biome. In Central America, for example, dry forests have been much more heavily deforested and degraded than humid forests, possibly due to a combination of factors, such as population density, access, climate, and agricultural potential of the soils (Finegan and Bouroncle 2008).

Immediate effects of such conversions are the loss of biodiversity and changes in composition, structure, and ecological processes of the forest ecosystem. Other changes include changes in the composition and structure of forest edges that increase the proportion of forest edge habitat compared to forest interior habitat that is essential to forest-dependent species. These effects are dependent on, among other things, the level of fragmentation of the remaining forests, the shape and size of the fragments, and the types of other land uses within the landscape (Finegan and Bouroncle 2008).

Deforestation causes loss of biological diversity with a concomitant loss or reduction of ecosystem services (Díaz et al. 2005, Metzger et al. 2006, Flynn et al 2009, Laliberté et al. 2010). Ecosystem services, however, are more influenced by a particular species composition than by the number of species present (Díaz et al. 2005). A number of ecosystem services can be at least partially restored by responsible land use practices oriented toward the conservation and restoration of specific functional groups and communities, rather than simply by increasing species richness (Kremen 2005, Balvanera et al. 2006). Carbon sequestration and storage, soil quality, habitat for specific bird and insect communities that provide regulating and pollination services for agricultural crops, regulation of water quality and runoff, are some of the ecosystem services reported from forest and agroforest systems (Díaz et al. 2005, Kremen 2005, Agbenyega et al. 2009). Some authors (Agbenyega et al. 2009) argue that the same measures that increase non-agricultural crop vegetation – for example roadside verges, live fences, abandoned fields, or companion plants in agricultural fields within the landscape – may also provide ecosystem dis-services, such as habitat for crop pests and pathogens, and competition for resources. Lara et al. (2009), in their discussion of the effect of native forests versus exotic plantations on recreational fishing, provide a good example of such trade-offs: increasing secondary growth in buffer zones along rivers near Valdivia in Chile increased the abundance of exotic trout species, valuable for recreational purposes, at the cost of abundance and diversity of native fish species; and increasing the current economic value of the streams at the risk of increasing the vulnerability of the fish population to future changes in the environment and climate. Forest fragments, however, may be managed or depleted according to the perception of benefits by the owners and neighbours of the fragments, rather than according to measurable social economic benefits and costs.

Biological diversity has a direct link to the capacity of ecosystems to adjust to changes (Naeem and Li 1997, Loreau et al. 2002, DeClerck et al. 2005, Díaz et al. 2005). While additional species may have little influence in species-rich ecosystems, in species-poor ecosystems, an increase in species number may increase ecosystem productivity and resilience (Díaz et al. 2005, Thompson et al. 2009). Of particular concern is that loss of biodiversity through forest conversion will decrease the potential for adaptation and maintenance of ecosystem services in the light of projected climate change scenarios (Flynn et al. 2009, Innes et al. 2009, Laliberté et al. 2010). Laliberté et al. (2010) provide a particularly compelling, though theoretical, example that includes more than 3000 species from forest ecosystems in Australia, China, North America, and Europe. Their study documents that not only is functional diversity lost with deforestation and agricultural intensification, but the species lost often belong to different functional response groups, inhibiting the capacity of these forest ecosystems to respond to a diversity of disturbance regimes, including changes due to climate variability or climate change.

6.2.2 Over-Exploitation

Over-exploitation occurs when the number of individuals that are removed annually from a population exceeds the natural annual increment of that population so that it can no longer sustain itself without intervention, which leads to decline and threatens its existence. While over-exploitation affects many renewable natural resources (e.g., over-harvesting of timber, fuelwood, grazing areas, hunting, fishing), it has been severe in tropical grasslands and savannahs, and in marine ecosystems, while it is increasingly having a negative impact in tropical forests and coastal areas (MEA 2005).

Over-exploitation of species may result in degradation of the ecosystems followed by the loss of genetic diversity and the extinction of species. In Morocco's forests, for example, there is an over-exploitation of fuelwood (three times the forest production) and an over-exploitation of fodder (three times the forest-grazing possibility (Ellatifi 2004, 2005, 2008; Karmouni 2006), affecting a range of species. In many countries, the effects of such over-exploitation are poorly studied. In countries such as Guatemala, Honduras, or Nicaragua, where firewood is still the main energy source for a large part of the population, firewood harvests occur both within and outside forests from trees planted for this purpose, as well as from natural forests or forest patches. While the use of wood for energy in such less-developed countries is several times greater than the use of wood for other purposes (FAO 2010), and may go well beyond the production capacity of the forests, it is often associated either with clearing of forests for agricultural activities – and therefore its effects on the forests are difficult to distinguish from those of forest conversion and fragmentation – or with trees in agricultural lands, for which few data are available.

Over-exploitation in forests for reasons other than fuel and fodder, however, often affects only a few species. In Guyana, for example, ter Steege et al. (2002) found that 75 years of harvesting of greenheart (*Chlorocardium rodiei*) affected the commercial availability of that species but had little effect on overall tree species diversity. Rice et al. (2001) suggested over-exploitation of mahogany (*Swietenia macrophylla*) in Latin America also causes only marginal changes to species diversity if it is not followed by uncontrolled exploitation of other species or conversion of forests to other land uses. Over-exploitation of a single species, however, may become more harmful in areas with a low tree diversity, or when harvests shift from one species to the next, once the most attractive or available species have been seriously depleted. In Latin America, the Mediterranean Basin, and elsewhere, over-exploitation is often linked to illegal activities, opening up the forest to other land uses, and thus forming a first step towards forest conversion with its subsequent effects on biodiversity.

Whereas for tree species, such over-exploitation of a single species has rarely had serious consequences for the forests and their diversity, this may not be the case for over-exploitation of animal species. Since such over-exploitation is largely illegal, few, if any, formal data are available on its extent and consequences. However, Nasi et al. (2008) documented numerous changes in populations of wild species hunted for bushmeat, suggesting that "empty forest syndrome" was becoming common in many tropical regions. Of particular importance for biodiversity is the over-exploitation of those animals and plants that play a key role in the ecological processes of the forests, for example as seed-dispersers s (e.g., rodents, monkeys), as pollinators (e.g., bats), or as important sources of food during periods when few plants provide seeds or fruits (e.g., some Manilkara species).

6.2.3 Changes in Biogeochemical Cycles

Biogeochemical cycles are movements of chemical constituents through biotic and abiotic components of the Earth. The most well-known of these cycles are those of water, carbon, nitrogen, oxygen, phosphorus, and sulphur. More detailed discussions of the importance of forests for the water and carbon cycles can be found elsewhere in this volume. The relationship between climate change and biodiversity is also amply discussed both in this volume and in the report on adaptation of forests and people edited by Seppälä et al. (2009).

Timber harvesting may influence different nutrient cycles, but current sustainable forest management practices allow for their recovery within reasonable periods of time (e.g., Poels 1987 for Suriname). Earlier studies indicate the importance of vegetation regrowth in abandoned agricultural fields for the recovery of soil nutrients (e.g., Nye and Greenland 1960). For detailed discussions of these impacts and functions, we refer readers to the extensive literature on the subject (e.g., Cole 1995, Gerding 2009, González 2009). In this section we will only briefly highlight the importance and impacts on biodiversity from alterations of the carbon, sulphur, and nitrogen cycles.

The use of fossil fuels moves large stocks of carbon that were stored in the ground into the atmosphere. Such mass translocation of carbon may cause changes in regional climate regimes and thus alter terrestrial ecosystem functions and services. The Intergovernmental Panel on Climate Change (IPCC) review of scientific literature concerning climate change indicates that since the Industrial Revolution, about 160 years ago, carbon (mostly as carbon dioxide [CO_2] and methane [CH_4]) and nitrates (N_2O) have accumulated in the atmosphere (Le Treut et al. 2007). Le Treut et al. (2007) attributed the increase in average global temperature beyond that caused by natural phenomena detected for the same period to this accelerated increase in the proportion of these gases in the atmosphere. Although there is still some uncertainty on the actual effects of further increases in greenhouse gas (GHG) emissions on the climate, many scientists use some relationship between GHG emissions and climate to project future climate changes under different emission scenarios (e.g., Meehl et al. 2007).

Besides the effect on climate and its subsequent effects on forest vegetation, the increased concentration of CO_2 in the atmosphere also has had a fertilisation effect on many plant species, in particular in areas where other growth factors, especially moisture and nitrogen, are not limiting. Phillips et al. (2008), for example, attributed the biomass growth and changes in forest composition of the Amazon forests to an increase in atmospheric CO_2 combined with increased temperatures, although the latter may cause the plants' capacities to absorb carbon to decrease once a critical temperature has been reached.

Since climate is an important factor in determining species distributions, climate change may have a great effect over current, local biological diversity, possibly causing an increase in species extinctions (Fischlin et al. 2007). Climate change is also expected to affect the distribution of ecosystems, increasing forest cover in boreal and alpine regions, decreasing cover and changing forest type in temperate regions, and decreasing forest and woodland cover in the tropics (Fischlin et al. 2007).

The expected effects of climate change on the provisioning of ecosystem services differ among the biomes and forest types. In general, climate change is expected to bring about increased frequency and intensity of disturbances, including greater extremes in temperature and precipitation, the effects of which on biodiversity are likely to be enhanced by increased intensity and occurrence of pests and diseases, fires, and cyclic extreme weather events (Fischlin et al. 2009). Climate change is also expected to have effects on the phenology of many plant species, altering leaf bud flush, flowering, and fruiting periods, and negatively affecting reproduction processes (McMullen and Jabbour 2009). Due to the great uncertainty about which of the different emission scenarios might become the reality, and to the low resolution of most climate change models, predicting the actual levels and locations of future changes in biodiversity is difficult. Climate change policies should, therefore, concentrate on reducing emissions, increasing information on changes expected to be caused by climate change, and on reducing the vulnerability of natural and human systems to climate change effects.

In addition to carbon, burning fossil fuels also releases sulphur, which reacts with water and oxygen in the atmosphere to form sulphuric acid. This may cause acid rain that has negative effects on plants, animals (e.g., Dirnböck et al. 2007), and infrastructure. Such effects may also be caused by the sulphur emitted during volcanic eruptions. It may take forests many years to recover from these effects, even once sulphur emissions have been curbed (Dirnböck et al. 2007).

Finally, the nitrogen cycle has also been largely altered by human activities, with the potential to contribute to acid rain, as well as eutrophication of (aquatic and terrestrial) ecosystems. Both phenomena have caused changes in species composition, favouring species with a greater tolerance for acid environments (e.g., slight changes in alpine areas in Austria, Dirnböck et al. 2007) and driving local species extinctions in grasslands (e.g., Tilman et al. 2002). On the other hand, positive fertilisation effects of nutrient-rich rains on productivity have also been observed in some systems, especially in Europe (Fischlin et al. 2007).

6.2.4 Invasive Species

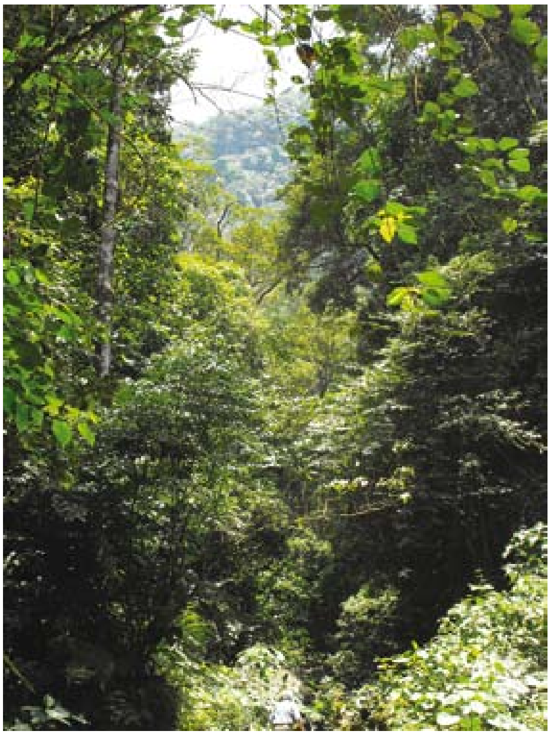
We define an invasive species as any species that successfully invades a forest type (ecosystem) where it was previously unknown, causing biological change and/or ecological or economic harm in that ecosystem (Levine et al. 2002). Invasive alien species (those invasive species that originate from outside the ecosystem) are a major cause of species extinction (Norton 2009). This has especially been studied on islands and in Mediterranean environments owing to the high numbers of invasive species in these ecosystems (Blackburn et al. 2004, Norton 2009). Most introductions of invasive alien species have resulted from human actions, and while many introduced species fail to increase in numbers, the relative few that become successful often cause disproportionate damage (Mack et al. 2000). More recently, however, climate change has increased the success of invasive species by changing the conditions to favour invaders over local species. Further, the success of invasive species is often linked to a chain of events acting additively or synergistically to promote conditions favourable to invasion, including habitat alteration and degradation, community structure alteration, and over-exploitation (Diamond 1989). In this sense, invasive species may be a proximate cause of change following a chain of earlier events and environmental changes.

The mechanisms by which invasive species cause local ecosystem change include: competition with or predation on local species, alteration of ecosystem functioning, and even genetic contamination (e.g., Shea and Chesson 2002). The results of invasions include altered community structure, altered biodiversity (including extinctions), homogenisation of flora and/or fauna, and, ultimately, reduced ecosystem services (Chapin et al. 2000). Once invasive species are established, they may decrease the resistance and resilience of the systems, and, in addition, create an alternative stable state that is exceedingly difficult to eradicate (Hooper et al. 2005, Thompson et al. 2009).

Examples of ecosystem change following species invasions are numerous and include the loss of native flora and fauna on many islands, such as Guam, where invasion by the brown tree snake in the 1950s led to the decimation of most of the island's bird, lizard, and mammal populations (e.g., Mortensen et al. 2008). Loss of biodiversity often results in a loss in the capacity of a system to resist invasion; diverse systems are often better able to resist invasion (Balvanera et al. 2006). Published evidence is consistent with the concept that diversity enhances the stability of ecosystem processes (DeClerck et al 2005, Hooper et al. 2005, Laliberté et al. 2010) and the flow of goods and services. The evidence relating resistance to invasion success is based on the capacity of species in more diverse systems to better use and/or partition resources, compared to in simple systems, where vacant niches are more available (e.g., Post and Pimm 1983, Levine et al. 2002, Hooper et al. 2005). Various factors, both extrinsic and intrinsic to an ecosystem, such as availability of niches, system degradation, and fragmentation, may affect the capacity of alien species to invade.

A number of examples are available of introduced trees invading temperate forest ecosystems (e.g., Richardson 1998), suggesting that many forests are not especially resistant to invasion and that many invading species are superior competitors to many local species and/or that forest plant communities are not saturated. Considerable evidence indicates that disturbed systems are more prone to invasion than undisturbed systems and that diverse tropical ecosystems are less prone to invasion (e.g., Sax 2001, Simberloff et al. 2002, Fridley et al. 2007). Lack of resistance to invasion in temperate forests, may be a long-term result of a reduced number of endemic species following ice ages coupled with loss of species owing to invasive diseases anthropogenic effects, which have resulted in vacant niches (Simberloff et al. 2002).

Many studies compare the number of native plant species in a system to the number of introduced plant species as an indicator of the invasibility of an ecosystem (e.g., Macdonald et al. 1989, Keeley et al. 2003). Such comparisons may be too simple, however, since the effects of invasion are also affected by the level of disturbance in a given ecosystem, the extent of the undisturbed area owing to edge effects, and the scale of measurement. Deriving a general hypothesis for forests is confounded by this complexity.



Matti Nummelin

Photo 6.2 Effectively and efficiently managed protected areas are the best way for ensuring the maintenance of biological diversity (San Martin Colliera Escalera, Peru).

6.3 Biodiversity Management as a Response to Drivers of Change

Biological diversity alone is not a guarantee for provision of abundant goods and services (Díaz et al. 2005). This diversity also needs to contain the right mix of species and structure to provide goods and services in a sustainable manner. Diversity, however, may enhance an ecosystem's capacity to resist or adapt to changes (Balvanera et al. 2006, Fischlin et al. 2009, Laliberté et al. 2010). Possibly the most direct links between diversity and ecosystem services can be established between growth rates and carbon contents of specific species. A greater proportion of fast-growing tree species increases carbon storage (e.g., Bunker et al. 2005), but diversity seems to be more linked to ecosystem stability than with the level of carbon sequestration (Bunker et al. 2005, DeClerck et al. 2005, 2006). Nevertheless, there is considerable evidence that multi-species ecosystems are more productive than simple forest systems, such as monoculture plantations (Thompson et al. 2009).

In this section we highlight some promising experiences of management of biological diversity that were able to maintain or enhance a variety of ecosystem services. Conservation of biological diversity needs an approach that goes beyond relying only on protected areas. This has been well understood by many decision-makers, resulting for example, in the establishment of Biosphere Reserves. These have different levels of management (core areas, multiple use zones, and buffer areas) and use biological corridors that are oriented at improving the connectivity between protected areas, often by promoting sustainable land use options on private lands within agricultural landscapes. Therefore, we distinguish three broad settings for biodiversity management: (1) management of protected areas, (2) management of forest reserves for timber and non-timber production, and (3) management of biodiversity in agricultural landscapes.

6.3.1 Management of Protected Areas

We consider that effectively and efficiently managed protected areas are the best way for ensuring the maintenance of biological diversity. DeFries et al. (2005), for example, found that habitat destruction within 198 protected areas in tropical forests of different continents was lower than in the associated buffer zones, or than outside the protected areas but within the same biomes. However, they also indicated that the increased isolation of these areas was common. In Latin America, this has resulted in an increase of the negative effects of surrounding land uses on the diversity inside protected areas (Finegan and Bouroncle 2008). Other authors have shown that management of forests for timber and non-timber forest products by communities in Guatemala and Mexico may have greater positive effects on forest conservation than poorly managed protected areas (e.g., Carrera and Prins 2002, Bray et al. 2008).

Protected area networks must be managed and extended to provide adequate protection of biodiversity given the new global realities, including climate change and an ever-expanding human population that continues to erode natural capital. Sánchez Azofeifa et al. (2003) report a positive effect on surrounding areas of effective park management in Costa Rica, finding more forest patches on private lands near the parks than away from these parks. This may, however, also be due to government policies oriented at strengthening those parks through prioritisation of Payment for Ecosystem Services (PES) to land owners within biological corridors linking such protected areas. The location of protected areas in respect to large population centres, main highways, or flat land, may influence the degree to which protected areas contributed to avoiding deforestation. In Costa Rican regions with high rates of deforestation, the establishment of parks has had the effect of reducing this rate of deforestation. Parks established in areas with poor accessibility and low potential for

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Photo 6.3 Payments for environmental services have contributed to forest conservation in Costa Rica.

other land uses contributed little to reduce the already low threat of deforestation (Pfaff et al. 2009). In all of these cases, participation of local land owners has been important to achieve the protection goals within protected areas as well as on the neighbouring private lands.

Another positive example in this regard has been in Canada, where, increasingly, protected areas are being seen as one way to help natural systems adapt to climate change by reducing human-caused stresses on the landscape. To reduce the pressure from increased accessibility to resources in northern Canada, due to a rise in temperatures, over 11 400 000 ha of new protected areas have been established. These new protected areas have doubled the amount of land protected over the past 20 years to about 10% of the Canadian land base. Park planning and expansion has involved aboriginal communities, government agencies and representatives, a variety of national and regional public interest groups, non-aligned members of the public, and industry (e.g., forestry, mining, oil and gas industries).

While climate change is one of the drivers behind the establishment of many of Canada's newest parks, ecosystem integrity remains the main goal of protected areas planning in Canada (in revised National Parks Act 2000). Climate change has not yet been fully accounted for during planning processes, and few parks have developed plans related to climate change (Scott and Lemieux 2005). In Latin America, the prevention of deforestation and forest degradation by over-exploitation are major goals for the establishment of protected areas.

Many older protected areas in Canada and elsewhere were established with the specific objectives of protection of landscapes and/or discrete ecosystems, conservation of biodiversity, particularly wildlife, and the provision of public recreation. However the capacity of the protected areas to support these objectives may change as a result of global warming. It will be essential to review the capacity of protected areas to meet both present and future species and ecosystem protection objectives in the light of climate change scenarios. The lack of certainty in climate change projections adds a tremendous challenge to manage for ecosystem adaptation to climate change in protected areas. Adaptation will require careful landscape management in areas surrounding current protected areas, particularly for maintaining or enhancing connectivity between these areas (e.g., Rayfield et al. 2008). In Costa Rica, for example, climate change has prompted authorities to consider the connectivity between protected areas, above all, along altitudinal ranges, promoting the establishment of biological corridors on private lands (Canet Desanti 2007).

Expansion in existing protected areas is one approach to solving the problems associated with climate change uncertainty. However, improved climate change models and landscape planning will be required to adapt protected areas to a changing climate (Scott and Lemieux 2005). Some estimates suggest that a substantial increase in protected areas will still be required if adaptation to climate change is an objective (e.g., Wiersma and Nudds 2009).

Protected areas, if effectively managed, are good instruments for maintaining biodiversity and the ecosystem services that they provide. However, only 11.5% of the world's natural vegetation is currently protected in such areas (Rodrigues et al. 2004). At the same time, at least in Latin America and other developing regions, many of these areas lack adequate management resources to ensure that the ecosystems are well protected. Protected area networks may not provide the degree of protection needed to conserve habitats having an abundance of particularly sensitive endemic species that have narrow geographic ranges (for example, Meso-America, north of Colombia, and Atlantic forests of Brazil, Rodrigues et al. 2004). While in some cases, increasing the coverage of protected areas will be a viable option, in others, such as in areas where forests are intensively used by local people, other forms of management of biodiversity are necessary.

6.3.2 Sustainable Forest Management

In Latin America, about a quarter of the natural forests have been assigned to local communities (Sunderlin et al. 2008), amounting to over 250 million ha. A similar area is under legal protection, and about 200 million ha are privately held (mainly in the Amazon region); the rest is state land given in concession or without clearly assigned management responsibilities. Of these natural forests, about 3–4% are managed according to internationally recognised standards of responsible forest management [1] in community forests (e.g., in Brazil, Guatemala, Mexico), in private forests (e.g., Brazil, Argentina, Chile, Costa Rica), or in concession areas (e.g., Bolivia, Guatemala, Peru). These areas are considered to be well-managed and contribute, through biological diversity conservation, to the maintenance of ecological functions and at least some ecosystem services. In the Mayan Biosphere Reserve in Guatemala, contribution to biological diversity conservation is probably most striking. Satellite images from 2000–2002 clearly show a low frequency of fires and land use change within the community forest concessions, compared to the adjacent reserve buffer zones and park areas, some of which remain heavily affected (Carrera and Prins 2002).

Well-managed areas include those whose managers know their resources well, typically through a series of inventories at different scales and of differing designs and objectives. In many cases, forest managers establish permanent sample plots of varying sizes to monitor forest dynamics over time, to perform baseline studies of mammals and birds, to confirm presence of endangered species, and as a tool for future monitoring. Apart from increasing the information base for management, these managers plan ahead, apply reduced-impact logging (RIL) practices, and implement mechanisms to become good neighbours and good employers. Most of these experiences are relatively new, and it is very early to determine their success. It has been confirmed, however, that certified Latin American forest managers do set aside a considerable part of their management area for conservation purposes, and that RIL practices indeed appear to reduce direct logging damage to about 50% of that incurred by conventional logging practices (Johns et al. 1996, Durrieu de Madron 2009).

In Costa Rica, nearly 20 years of observations in permanent forests plots has shown that the main negative effect of moderate harvests removing 10 m^3/ha was the area occupied by roads (less than 5% of the harvested area). Re-growth in the gaps caused by felling was dominated by forest species. Diversity measurements in these gaps did not show a significant difference between harvested and un-harvested sites, except for plots on and immediately along the roads (Delgado et al. 1997).

These cases show that forest management has the potential to contribute to the conservation of biodiversity. However, the area of well-managed forests is not much greater than that of protected areas (about 300 million ha between two global certification systems: Program for Endorsement of Forest Certification and the Forest Stewardship Council), still leaving about 75% of the world's forests under unclear management conditions. Many factors are considered to contribute to the lack of increase in well managed forest areas. For Latin America, in spite of a general improvement of forest policies and legislation during the last 10 to 15 years, some of the reasons for the lack of adoption of forest management practices have been the slow and little-transparent administrative procedures, high cost of management combined with low timber prices, high opportunity costs, poor control of illegal logging activities, uncertainty about future forest use rights, and social conflicts due to overlapping rights (e.g., Walters et al. 2005, Smith et al. 2006).

[1] These were successfully evaluated by auditors of the Forest Stewardship Council (FSC) forest certification scheme. It should be noted that besides these certified natural forests, also about 4 million ha of plantations have been certified.

6.3.3 Management of Biological Diversity Within Agricultural Landscapes

The historical relationship between conservation of biological diversity and agricultural production is an adversarial one. Agricultural expansion is often cited (Kanninen et al. 2007) as one of the main driving forces behind deforestation. Less frequently, the relationship between the conserved elements of landscapes and the production elements is considered, in spite of increasing evidence that strategic management of biological diversity at landscape scales within agricultural landscapes can play a critical, if not essential, role in the development of a sustainable agriculture (Balvanera et al. 2006). Indeed, conservationists are rapidly recognising that conservation of biodiversity will be impossible unless the conservation role of the agricultural matrix is considered. Farmers are also recognising the functional role that biological diversity plays in sustainable agricultural production. This recognition by stakeholders that traditionally have been considered diametrically opposed to each other, sets the stage for landscape-scale management that combines conservation goals with production goals and livelihood improvement, which is also the basis for successful implementation and management of biological corridors.

Exercises in landscape classification generally begin with the most evident dichotomy, distinguishing between forest and agricultural land uses, followed by further sub-categories. Traditionally, we are trained to consider the distinct differences between these land uses. From an ecological perspective, however, the boundaries between these land uses can be less clear, varying from abrupt to gradual, depending on the species or processes in question. An increasing number of studies considered the impacts of these transition or edge zones on the movement of individuals and processes. Edge effects occur on the boundaries between ecosystems, between forest and agricultural systems, for example, and typically encompass a change in environmental conditions, notably temperature and humidity, in addition to the obvious structural changes between the two land uses. From the point of view of conservation biologists, these edges represent the barriers between forest and disturbed habitat that, for some forest-dependent species, essentially acts as an impenetrable wall.

The effects of these edges are largely a function of the species in question. Some species prefer edges and their impacts on resource distribution. Others find that the edge habitats provide a suitable blend of nesting habitat (forest) and foraging habitat (the agricultural matrix) and is the source of many of the most valuable ecosystem services for agriculture (Balvanera et al. 2006), such as pollinators and insect predators. It is also possible to consider the impacts of this forest habitat from the opposite point of view – that the forest serves as a barrier to the movement of agricultural pests or agricultural run-off. It is largely this second set of effects that warrant greater consideration and that may play an important role in understanding the functional role of conserved forests within the agricultural landscape. Here we quickly review three such functions: 1) forests as source habitat for species of agricultural importance, 2) forests as buffers, and 3) forests as barriers. By no means do we suggest that the functional interaction between forest and managed portions of the landscape is limited to these functions. The biological corridors in Central America are an example of the maintenance of forest or tree covers for a combination of functions at a landscape level, emphasising that for successful implementation of such an integrated approach, other factors may be as important as the ecological functions. Morse et al. (2009) provide some insight into factors that may influence such decisions, concluding that in Costa Rica, national legislation that forbids forest conversion, the implementation of a payment for environmental services (PES) scheme, and the social and economic situation of the forest owners were important factors influencing the decision to maintain forest on their agricultural land.

Forest Patches as Source Habitat for Species of Agricultural Importance

One of the predominant characteristics of agricultural landscapes is the regular and frequent disturbance originating from the cultivation practices associated with annual crops, or the regular application of agrochemicals and pruning regimes of perennial crops. These activities generally favour insect species with short life spans and/or high dispersal rates, traits that we typically associate with agricultural pests. On the other hand, the species that prey on these pests are often associated with longer life spans requiring less frequently disturbed habitats. Extensive agricultural areas, therefore, tend to promote agricultural pests while inhibiting the presence of the natural control agents (e.g., Díaz et al. 2005). Some studies have shown, however, that reducing the distance between the natural and semi-natural habitats where disturbance rates are low, and the agricultural area, increases the capacity of these natural predators to control pest populations. The semi-natural area serves as habitat where the individuals can breed and survive, whereas the agricultural portion serves as a source of food for the pest-predators. Conservation of forest fragments adjacent to agricultural uses can contribute to pest control.

Pest control is not the only function that behaves in this manner. For example, pollinators have been shown to exhibit the same tendencies. This has also been the main ecosystem service evaluated in developing countries, relating them to their respective ecological functions (or support services in MEA 2005). Ricketts (2004), in a classic study, counted the richness and abundance of bee species along a gradient from within a forest extending 800 m or so into a coffee plantation. His results showed that bee richness and abundance dropped dramatically 50 m away from the forest. At the forest edge, 11 bee species were actively pollinating coffee plants, but at 800 m from the forest edge, only 2 species remained, one of which was the introduced European honeybee that is responsible for 98% of the bee pollination at this distance. While coffee is also self-pollinating, reducing the impact of reduced bee diversity on pollination, these results show the potentially negative effects of removing forest fragments in agricultural landscapes: The reliance on a single species for pollination services compromises the resilience of the pollination service to collapse of the bee populations, a scenario that is not unlikely and that has occurred in the south-western United States, where honeybees play a particularly important role in fruit and food production in the absence of native pollinators.

Forest Patches as Buffers

Strategic placement of forests is not only important to creating habitat, but it can also play an important role as a buffer, inhibiting the movement of agricultural waste products to more sensitive areas, such as aquatic systems. This case of riparian buffers has been particularly well demonstrated in the Mississippi River watershed of the United States (US). Strategic placement of narrow strips of forests along rivers can effectively serve to prevent the movement of agrochemicals and sediments into sensitive waterways. The impact of coordinated protection of riparian corridors not only has tremendous promise for increasing the movement of forest-dependent species due to their linear nature, but their filtering ability plays a tremendous functional role in maintaining or improving water quality.

Protection of riparian forests thousands of kilometres away is one of the most practical solutions to preventing downstream pollution by agricultural wastes. Ecologists working in the Midwestern US states have demonstrated that simple 10 m-wide buffers of riparian vegetation can absorb 90% or more of the agricultural run-off before it enters the waterway and storing it in the plant biomass (Schultz et al. 2004). These riparian forests play additional roles in reducing the loss of sediment to erosion, providing corridors for wildlife, and enabling recreation opportunities in the form of hiking, hunting, and fishing (Schultz et al. 2004). In this case, the conservation of forest buffers within the agricultural matrix has impacts that are felt well beyond the boundaries of the agricultural landscape, indeed, extending into adjacent aquatic or marine ecosystems.

Kareiva and Marvier (2007) have also promoted the use of natural area buffers for ecosystem protection. They focused on the Mississippi River delta, an area that was devastated by Hurricane Katrina in 2005. Kareiva and Marvier noted that the delta includes a combination of endangered biodiversity, poor communities that were disproportionately affected by hurricanes, and low-lying areas where the hazard risk is high. Mapping areas of conservation interest, areas at greatest risk of storm surges and flooding, and low-income communities also at greatest risk from storm surges, permitted a visual overview of priority areas where multiple goals could be met simultaneously. For example, by preserving natural ecosystems in flood zones adjacent to these low-income communities, planners are able to protect endangered biodiversity while protecting human lives and infrastructure.

Much of this sentiment was also noted during the Indonesian tsunami of 2004, where coastal areas that included forest buffers and mangroves were less affected than areas where such natural barriers had been removed. Bradsaw et al. (2007) also supported the notion of forests as buffers in their study of the relationship between forest cover and the damage caused by flooding. Their results demonstrated that conservation of upstream areas can indeed mitigate the effects of flooding and that extensive forests areas must be protected, but that the capacity of these forests to mitigate extreme events may be limited. It is important to note that with the predicted increase in natural hazards associated with climate change (such as sea level rise in the two examples given above), the functional role of forests as mitigation agents is likely to become increasingly important, and thus a critical understanding of how forest elements can be strategically located within landscapes to effectively serve as buffers is urgently needed.

Forest Patches as Barriers

Finally, the functional role of forests as barriers can also be considered. We typically think of forests as corridors, connecting patches of forests on fragmented landscapes and ensuring that species of conservation concern maintain the ability to move throughout the landscape. Consider for a moment a landscape that is 20% forested with 80% agriculture. From the conservation perspective, this landscape is heavily fragmented and it would be difficult for a forest-dependent species to disperse through the agricultural matrix. Connectivity is a species-specific phenomenon and our perspective of this landscape changes completely if we consider connectivity from the point of view of an agricultural pest, such as the coffee berry borer, which has an important economic impact on coffee production. For this organism, our theoretical landscape is not fragmented; rather, it is highly connected (Avelino, personal communication). Preliminary studies have shown that the borer considers forest habitat as hostile, rarely penetrating more than 10 m into the forest. This demands the question: If we managed to increase forest cover and connectivity in a landscape dominated by agriculture, are we effectively decreasing the movement of pest species while increasing the movement of species of conservation concern?

Forests have also been used as barriers against pollution and noise, particularly in more developed countries. Again, a good understanding of the characteristics of forests for the proper functioning of such barriers is essential, as is planning for their location at a landscape level.

6.4 Towards a New Vision of Biodiversity and Ecosystem Services

Loss of biodiversity and economic development cannot be seen as separate processes, although their relationship may differ according to geographic, cultural, socio-economic, and political context. The four drivers discussed are very much related to economic development: emission scenarios are derived from socio-economic scenarios (IPCC 2007). The extent and location of deforestation is at least partially explained by economic models (e.g., Hyde et al. 1996), while its rate over time has been presented as the Kuznets curve, showing increasing deforestation with increasing GNP (gross national product) per capita to reach a minimum and then decrease with a further increase in average GNP, also called "forest transition" (Mather 1992, Cropper and Griffiths 1994, Rudel et al. 2005).

Over-exploitation often is a first step towards deforestation, fulfilling the demand for high value species to a relatively limited market and implemented by generally poor people under difficult working conditions, while increased international trade related to globalisation has facilitated the spread of invasive species.

Reducing biodiversity loss, therefore, has to deal with the complex issues of development, but go beyond merely moving a country or region forward along the Kuznets curve to a level where forest cover increases. If biodiversity is not conserved along the way, the new forests may never come near to recovering the diversity nor the functions (and, therefore, the ecosystem services) of the old forests because the forest resilience has been lost. This would mean that the minimum level of forest cover would need to be as high as possible (probably >30%, Andren 1994), be distributed over the widest range of (forest) ecosystems possible and in large patches, and maintain functional connectivity to areas that are likely to recover their forest cover over time.

6.4.1 Maintaining a Minimum Level of Forest Cover

In the above paragraphs, we have argued that local successes have been (or can be) achieved in reducing pressure on forests. Successful approaches were different depending on the circumstances. We referred, for example, to studies in the Mayan Biosphere Reserve in Guatemala, where management responsibilities were assigned to local communities, accompanied with technical and financial assistance, and resulted in avoiding deforestation and instituting adequate forest management practices. In the highlands of Costa Rica and the eastern lowlands of Bolivia, effective park management combined with establishing local guide associations and PES, resulted in maintaining the park's integrity and increasing forest cover (Costa Rica), or reducing deforestation (Bolivia) immediately outside the park areas. PES increased forest and tree cover in some of Costa Rica's agricultural landscapes, although factors other than an increase in income (e.g., national legislation, socio-economic situation of forest owner) may also have been important in the motivation to maintain or enhance tree cover. The results of these studies suggest that the research should not only focus on the drivers of deforestation (a negative effect) but also on the drivers of conservation (a positive effect), such as PES schemes. PES should not be viewed as a panacea, however, and must be further developed to continue its positive impact on conservation and livelihood improvement. The results of these studies may particularly contribute to the design of local implementation mechanisms for Reducing Emissions from Deforestation and forest Degradation (REDD or REDD-plus).

Those activities that reduce land use change and over-exploitation also contribute to a reduction in carbon emissions. Reduced impact logging practices, usually an integral part of good forest management, also contribute to reduced emissions in comparison with conventional logging due to a reduction of 50% in road areas and damaged remnant trees. Unfortunately, these practices are not widespread. The application of both reduced deforestation and RIL activities are hampered by cultural, socio-economic, and political barriers. The removal of these barriers will be essential for a reduction of biodiversity losses and for mechanisms such as REDD to function. The experience in forest conservation and management over the last decades in Latin America, however, indicates that this may be more expensive than estimated by economists such as Stern (2006), may need more than a mere transfer of money from the developed to the developing countries, and may not be determined by economic factors alone.

6.4.2 Maintaining Functional Connectivity and Ecosystem Services Outside Protected Areas

Strategic placement of forests in agricultural landscapes presents multiple opportunities where conservation, production, and livelihood needs are simultaneously promoted. The science of landscape ecology, particularly of strategic arrangements of forest cover within the agricultural matrix, is nascent; however, it shows tremendous potential for increasing the multi-functionality of agricultural landscapes, including built-in adaptability to climate change, hazard risk reduction, and the provisioning of agro-ecosystem functions. An interesting example of this is the promotion of biological corridors in Costa Rica, involving private land owners in enhancing biodiversity and increasing the provision of ecosystem services within their agricultural fields.

6.4.3 Institutional Challenges

The above analysis leads us to identify a number of institutional challenges that need to be addressed in order to successfully counteract the main anthropogenic drivers for changes in biodiversity. Since the drivers are strongly related to factors of economic development, and economic development is influenced by institutions at different levels, biodiversity conservation needs to be mainstreamed at these same levels. Possibly the most progress can be seen at the international level, where the 1992 Earth-Summit in Rio de Janeiro gave rise to a number of international agreements and conventions calling for development policies and strategies that consider a balance between social, economic, and environmental costs and benefits. While these agreements have catalysed the appearance of a large number of examples of well-managed forests for protection or production, they have so far failed to reduce poverty (e.g., Agenda 21), to implement widespread sustainable forest management (forest principles), to halt deforestation (e.g., Framework Convention on Desertification, Convention on Biological Diversity), or to curb greenhouse gas emissions to acceptable levels (Framework Convention on Climate Change).

Why is it that these international agreements and conventions have not had, so far at least, the desired effect at national and local levels? It is beyond this paper to answer this question. Elsewhere in this document, the functioning of institutions in the forest sector is discussed extensively (see Chapter 23). From our experience and the above analysis, however, it appears that one of the major barriers is that the different international agreements and conventions show much overlap but have failed to integrate their actions; instead, they have acted in isolation, often with duplication of efforts and even with competing interests. Related to the previous point, national and local efforts to improve forest management and protection or to reduce deforestation through legislation and policies, have concentrated on regulating activities within the forest sector, while many of the problems arise from pressures outside that sector. In addition, formal forest sector institutions do not seem to have been able to cope with the "ecosystems approach" proposed by the CBD and calling for involvement of a wider group of stakeholders in the management of natural resources within a territory.

This challenge, among others, lies in creating platforms at different levels (international, national and local) where different actors are able to discuss natural resource management openly, where all stakeholder groups are well represented, and that will have the capacity to address the most pressing issues at the corresponding level. The FAO (Food and Agriculture Organisation of the United Nations) -supported national forest programs have been designed to do just that, but in Latin America, few governments have the experience, skill, and willingness to apply them as designed. At the local or landscape level, the experiences with biological corridors in Costa Rica indicate that such platforms work if stakeholders within a landscape recognise a common objective. They may, however, require government support, subsidies, or a type of payment for ecosystem services to ensure that these public services are also provided on private lands. For this to work, ecosystem services will need to be recognised in law and the state will need to be able to put a value on these services or on the opportunity costs of their provision. Even so, PES may pale compared to profitable land-use alternatives, such as timber harvests, plantation forests, or soybeans, in which case the landowners will reject the PES offer (Wunder 2005).

From the ecological and economic points of view, one of the greatest challenges will be to increase knowledge on the effects of biological diversity on the desired ecosystem services and to valuate them properly. Although current efforts to conserve biological diversity within and outside forests show interesting experiences, for most services, it remains difficult to establish quantitative links between specific biodiversity and a specific level of environmental services, making it difficult to incorporate biodiversity into payments for environmental services or market schemes. An additional challenge will be to make sure that such schemes are accessible to those people most in need of the additional income, and not, as in many cases, to those that do not depend on the forest or forest land to make a living.

A third challenge will be to communicate the information to the stakeholders in such a way that they can use it for individual and group decision-making. This may require decision-making tools, such as multi-criteria analysis tools, or tools that allow them to make simple cost-benefit analyses.

6.5 Conclusions

Conversion of forests into agricultural lands, over-exploitation, air pollution leading to climate change and acid rain, and invasive species, all cause great stress on forest ecosystems. Conscious of the negative effects of human activities, society has responded by increasing the area of forest being protected and well-managed, and by incorporating management of trees and forest patches into management of agricultural landscapes. Still, most of natural forests and agricultural landscapes are not well-managed and their existence continues to be threatened by the same drivers. Our analysis of literature suggests that the lack of reduction in the threats to biological diversity is, among other things, due to lack of addressing the subjacent causes of the threats. These are very much linked to level and form of economic development, and are often found outside the forest and environmental sectors.

In order to reduce the threats to biological diversity, we suggest a positive approach to forest conservation, combining aspects of willingness to conserve with willingness to pay for further conservation; removal of administrative barriers to good forests management and protection; landscape management; intersectoral coordination between international, national and local policies; increased communication between stakeholders; and more research on the interactions between biodiversity and ecosystem services.

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