

CHAPTER 13

Conservation in human-modified landscapes

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In the previous two chapters, we learn about the importance and difficulties of prioritizing areas for conservation (Chapter 11), and the management of endangered species in these habitats (Chapter 12). In this chapter, we discuss the challenges of conserving biodiversity in degraded and modified landscapes with a focus on the tropical terrestrial biome, which is undergoing rapid deforestation and habitat degradation (Chapter 4) and contains an untold diversity of rare and endemic species that are in urgent need of conservation attention. We first highlight the extent to which human activities have modified natural ecosystems, and how these changes are fundamental in defining ongoing conservation efforts around the world. We then outline opportunities for conserving biodiversity within the dominant types of human land-use, including logged forests, agroforestry systems, monoculture plantations, agricultural lands, urban areas, and regenerating land. We also highlight the highly dynamic nature of modified landscapes and the need to recognize important human development benefits that can be derived from conservation action in these areas.

13.1 A history of human modification and the concept of “wild nature”

Efforts to improve human welfare have led to landscapes and ecosystems worldwide being domesticated to enhance food supplies and reduce exposure to natural dangers (Kareiva *et al.* 2007). As a consequence there are few places left on earth that have escaped some form of obvious

human impact (see Chapter 4) that can have negative effects on biodiversity. This is especially so because human beings have released toxic synthetic organic chemicals, many of which are endocrine disruptors (Box 13.1), that are now distributed from pole to pole.

Although few data are available on changes to the extent and condition of many habitats, regions and ecosystems, what we do know is that, with few exceptions, changes that are currently underway are negative, anthropogenic in origin, ominously large and often accelerating (Balmford and Bond 2005). For example, the conversion of forests to agricultural land continues at a rate of approximately 13 million hectares per year, and the last global assessment classified a full two-thirds of the world’s forests as having been modified by human impacts (FAO 2006).

Some ecologists have gone so far as to consider that the traditional concept of an intact ecosystem is obsolete, and instead propose a classification system based on global patterns of human interaction with ecosystems, demonstrating that much of the world currently exists in the form of different “anthropogenic biomes” (Figure 13.1 and Plate 15; Ellis and Ramankutty 2008). For many types of ecosystems, large areas of intact vegetation simply no longer exist, as is the case of the Atlantic forest hotspot of Brazil which has been reduced, except for a few conservation units, to a fragmented network of very small remnants (< 100 ha), mainly composed of secondary forest, and immersed in agricultural or urban matrices (Ribeiro *et al.* 2009).

Even when we turn to areas that at first appear to be undisturbed by human impact, the boundaries between “pristine” and “degraded” can

Box 13.1 Endocrine disruption and biological diversity

J. P. Myers

Since the beginning of the Industrial Revolution, over 80 000 new chemicals have entered commerce and hence the biosphere. These are compounds for which no organism has any evolutionary history and hence no opportunity to evolve over generations any metabolic protections against potential harm.

Depending upon how they are used and upon their chemical characteristics, they have dispersed widely, many globally. For example, whales feeding hundreds of feet beneath the surface of the mid-Atlantic accumulate brominated flame retardants from their prey. Bark of mature trees from virtually any forest in the world contains pesticides and industrial pollutants, even though they may be thousands of miles from the source. Penguins in the Antarctic store persistent organic pollutants that have been carried to the Antarctic by atmospheric transport and stored for decades in glacial snow but that are now being liberated by global warming. Seemingly pristine cloud forest in Costa Rica is more contaminated by the pesticides used on lowland banana plantations than forest adjacent to the bananas, because the pesticides volatilize in the lowland but are carried downwind and upward into the mountains, where they condense because of lower temperatures.

Decades of toxicological research focused on the effects of high exposures, which unquestionably can be serious, indeed directly lethal. Over the past 20 years, however, research has emerged revealing that this approach to toxicology was blind to serious effects that stem from the ability of some contaminants to interfere with hormones, altering gene expression, even at extremely low doses. These effects, deemed 'endocrine disruption' have forced toxicologists to rethink how they assess risk and have raised a wide array of questions about how contaminants may be affecting the biosphere in unexpected ways, since hormones regulate a wide array of biological functions in both plants and animals. Moreover, the signaling systems used by the endocrine system are highly conserved evolutionarily, operating in essentially the same ways in fish and mammals despite 300 million years of evolutionary separation. Hence

the sudden and unprecedented arrival of hundreds, if not thousands, of chemicals capable of disrupting hormone action and novel to body chemistry is a source of concern.

Three key discoveries lie at the center of this revolution in toxicology. First, hormones – and contaminants that behave like hormones – can cause completely different effects at different levels of exposure. This is because the suite of genes up- or down-regulated by a hormone can vary dramatically as the concentration of the hormone varies. And at high levels, the hormone (or a hormone-like contaminant) can be overtly toxic, shutting down gene expression altogether. Hence all of the tests that toxicologists have run that assume high dose testing will catch low dose effects are invalid. Compounds judged to be safe based on data from high dose testing may not be. Some, widely used in commerce, clearly are not.

Second, changes in gene expression as an organism is developing—in the womb, as an egg, as a larvae or a tadpole, etc—can have lifelong consequences, affecting virtually every system of the body, including altering fertility, immune system function, neurological competency (and thus behavior), etc. Frogs in suburban Florida are less likely to be feminized than frogs in agricultural Florida, where endocrine-disrupting agricultural chemicals are used. Frogs exposed as tadpoles to a mixture of pesticides die from bacterial meningitis when adult, from a common bacteria easily resisted by control animals.

Third, individuals vary significantly in their capacity to metabolize these compounds and resist their effects. Specific variants of genes are more, or less, effective at safely metabolizing a contaminant and rendering it harmless. In people, for example, there is at least a 40-fold difference in capacity to metabolize organophosphate pesticides.

This is the stuff of Darwin... heritable differences among individuals that alter reproductive success... but it is happening to people and biodiversity at a pace that may be unprecedented in the history of most, if not all species. Hundreds, if not more, of compounds capable of altering gene expression at low levels

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Box 13.1 (Continued)

of exposure have been introduced into the biosphere in fewer than 200 years. They alter fertility, cognition, immune and cardiovascular function, and more. The inescapable prediction, clearly speculative but highly plausible, is that this past 200 years has been a period of remarkable, if not unprecedented speed in the molecular evolution of life on earth.

Documented effects extend to interactions among species as well. For example, several environmental estrogens decrease the efficacy of communication between *Rhizobium* bacteria and their leguminaceous hosts, reducing nitrogen fixation. One widely used herbicide, atrazine, both increases the likelihood that ponds will contain large numbers of trematode parasites, which cause limb deformities in frogs, it also undermines the frog's immune defenses against trematode infections.

These emerging discoveries have come as surprises to traditional toxicology, because they raise questions about many chemicals in common use that based on traditional approaches had been deemed safe. For conservation biologists, they offer competing hypotheses to test against other interpretations. For example, is the disappearance of the golden toad (*Bufo perigrinus*) from Costa Rica a result of global warming? Or have the pesticides now known to be present in significant concentrations in Costa Rican cloud forests undermined their viability? What is the role of contaminant-reduced immune system function in fungal-caused deaths in frogs, clearly an important factor in amphibian extinctions? Is the chytrid fungus new? Or are frogs less able to withstand infestation? Was the lake trout extinction in the Great Lakes the result of lampreys and over-fishing, or because dioxin sediment loads became so heavy that 100% of fry died? Have impairments by endocrine disruptors in the ability of young salmon to switch their osmoregulation from fresh water to salt water when they reach the ocean in their first downstream migration contributed to salmon population declines along the Pacific coast? Are declines in Chesapeake Bay oysters and crabs a result of invertebrate vulnerability to endocrine-disrupting contaminants? Is the relationship between coral and their symbiotic algae disrupted by contamination? Does this contribute to coral bleaching?

In the most elegant experimental field test to date of population-level effects of endocrine disruptors, Kidd *et al.* (2007) contaminated a lake in western Ontario with an active ingredient of birth control pills (17alpha-ethynylestradiol), maintaining the contaminant's concentration at 5–6 parts per trillion for two years. This concentration is just above levels typically found in sewage effluent and also in surface waters. The treatment led initially to delayed sexual development of fathead minnows in the lake. By the second year they observed that some males had eggs in their testes (ova-testis). And by the end of the seventh year, long after the treatments were halted, very few individuals were left. The population had crashed. There are many reports of ova-testis in fresh water fish populations from around the world.

How large a role endocrine disruption plays in biodiversity declines isn't yet clear, because few conservation biologists have included these mechanisms in the suite of hypotheses their studies are designed to test. The solutions to biodiversity declines caused by endocrine disruption will contrast sharply with those from more conventional forces. No harvest zones and artificial reefs, for example, will prove futile if shellfish declines are caused by chemical contamination. Hence in the search for tools to maintain biodiversity, it is imperative that conservation biologists' science widens to incorporate these effects.

Relevant website

- Synopses of new studies on endocrine disruption: <http://tinyurl.com/a6puq7>.

REFERENCE AND SUGGESTED READING

- Colborn, T., Dumanoski, D., and Myers, J. P. (1996). *Our stolen future*. Dutton, New York, NY.
- Cook, P. M., Robbins, J. A., Endicott, D. D., *et al.* (2003). Effects of aryl hydrocarbon receptor-mediated early life stage toxicity on Lake Trout populations in Lake Ontario during the 20th century. *Environmental Science and Technology*, **37**, 3864–3877.
- Dally, G. L., Lei, Y. D., Teixeira, C., *et al.* (2007). Accumulation of current-use pesticides in Neotropical montane

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Box 13.1 (Continued)

forests. *Environmental Science and Technology*, **41**, 1118–1123.

Gore, A. C. (2007). Introduction to endocrine-disrupting chemicals. In A.C. Gore, ed. *Endocrine-disrupting chemicals: From basic research to clinical practice*, pp. 3–8. Humana Press, New Jersey.

Kidd, K. A., Blanchfield, P. J., Mills, K. H., et al. (2007). Collapse of a fish population after exposure to a synthetic estrogen. *Proceedings of the National Academy of Science of the United States of America*, **104**, 8897–8901.

Welshons, W. V., Nagel, S. C., and vom Saal, F. S. (2006). Large effects from small exposures: III. Endocrine mechanisms mediating effects of bisphenol A at levels of human exposure. *Endocrinology*, **147**, S56–S69.

quickly become blurred on closer inspection. Archaeological and paleoecological studies over the last two decades suggest that many contemporary pristine habitats have in fact undergone some form of human disturbance in the past (Figure 13.2 and Plate 16; Willis *et al.* 2005; Willis and Birks 2006; see Chapter 14).

For example, the Upper Xingu region of Brazil comprises one of the largest contiguous tracts of tropical rainforest in the Amazon today. Emerging archaeological evidence suggests that parts of this region had been densely populated with pre-European human settlements (circa ~1250 to ~1600 A.D.), and that extensive forests underwent large-scale transformation to agricultural areas

and urbanized centres (Heckenberger *et al.* 2003; Willis *et al.* 2004). Much of the lowland rainforests of the Congo basin had similarly experienced extensive human habitation, forest clearance, and agricultural activities between ~3000 and ~1600 years ago, as evidenced by extensive finds of stone tools, oil palm nuts, charcoal horizons (subsoil layers of charcoal), banana phytoliths (silica bodies found in plants preserved in sediments), and pottery fragments (Mbida *et al.* 2000; White 2001). Many further examples of extensive pre-European disturbance have been found in areas that conservationists today frequently describe as “pristine” or “intact”, including Southeast Asia, Papua New Guinea and Central America (Willis *et al.* 2004).

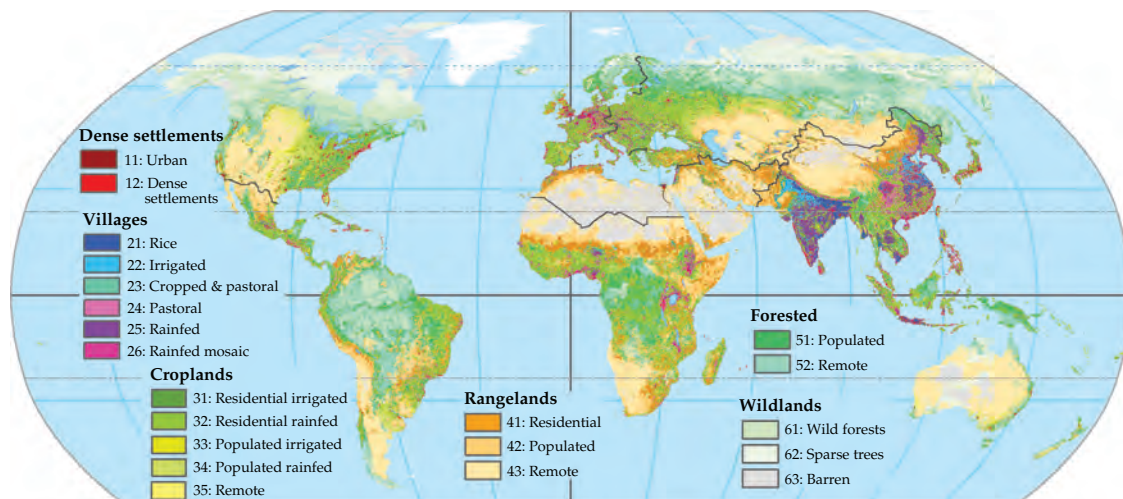


Figure 13.1 Anthropogenic biomes. Global land-cover analysis reveals that less than a quarter of the Earth’s ice-free land can still be considered as wild. Biomes displayed on the map are organized into groups and are ranked according to human population density. Reprinted from Ellis and Ramankutty (2008).

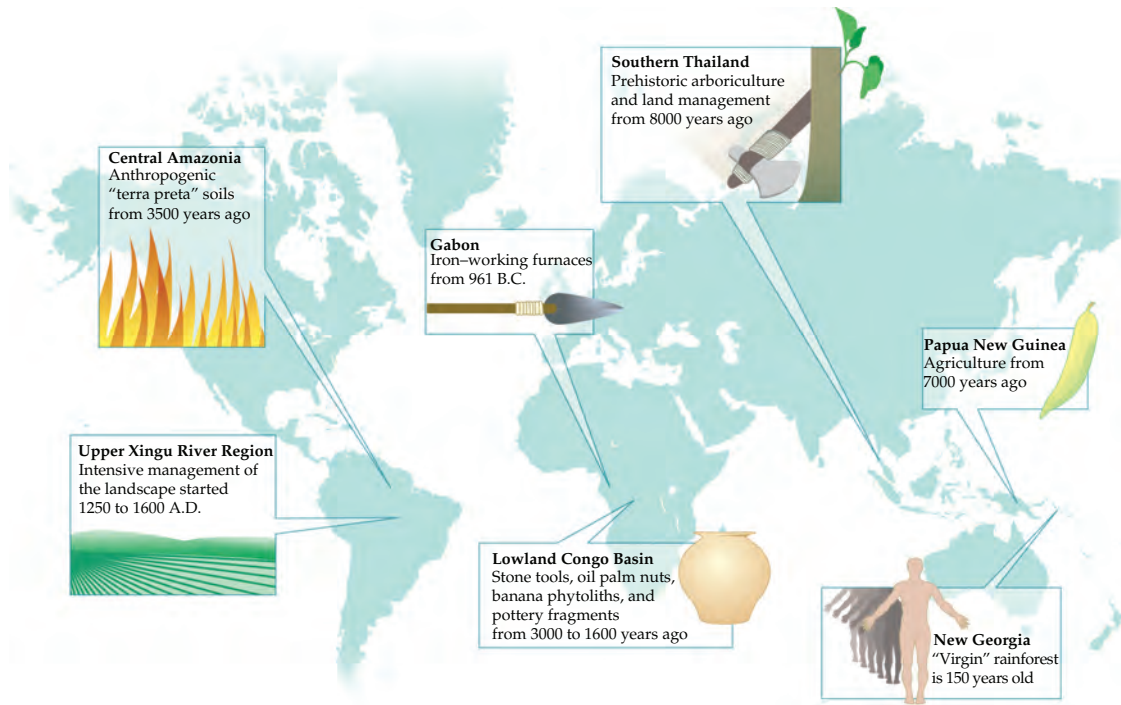


Figure 13.2 Evidence of human modification of “pristine” tropical rainforest. Archaeological and paleoecological studies suggest that rainforests in the Amazon basin, the Congo basin, and Southeast Asia have regenerated from disturbance by prehistoric human settlements. Reprinted from Willis *et al.* (2004) with permission from AAAS (American Association for the Advancement of Science).

In most of these cases, forest regeneration followed the abandonment of human settlements and agricultural activities resulting in the old-growth stands that are regarded as pristine today.

13.2 Conservation in a human-modified world

How does all this evidence of historical and ongoing human modification of the natural world relate to efforts to conserve biological diversity today? There are at least two very profound implications.

First, the sheer extent to which we have dominated the biosphere (terrestrial, freshwater, and marine) (Ehrlich and Ehrlich 2008) means that we have no choice but to integrate conservation efforts with other human activities. It is broadly accepted that strictly protected areas provide a necessary yet grossly inadequate component of a

broader strategy to safeguard the future of the world’s biota. Gap analyses show that approximately one quarter of the world’s threatened species live outside protected areas (Rodrigues *et al.* 2004; Chapter 11), and that most of the world’s terrestrial ecoregions fall significantly short of the 10% protection target proposed by the IUCN (Figure 13.3 and Plate 17; Schmitt *et al.* 2009). Even where they exist, the integrity of protected areas is often threatened by encroachment and illegal extraction in areas that are undergoing widespread deforestation (Pedlowski *et al.* 2005), and management of neighboring areas is vital to ensuring their long-term viability (Wittemyer *et al.* 2008; Sodhi *et al.* 2008).

Second, evidence of historical recovery in areas that once hosted high levels of human activity illustrates that while long-time scales are often involved, the biotic impacts of many types of disturbance might not be completely irreversible.

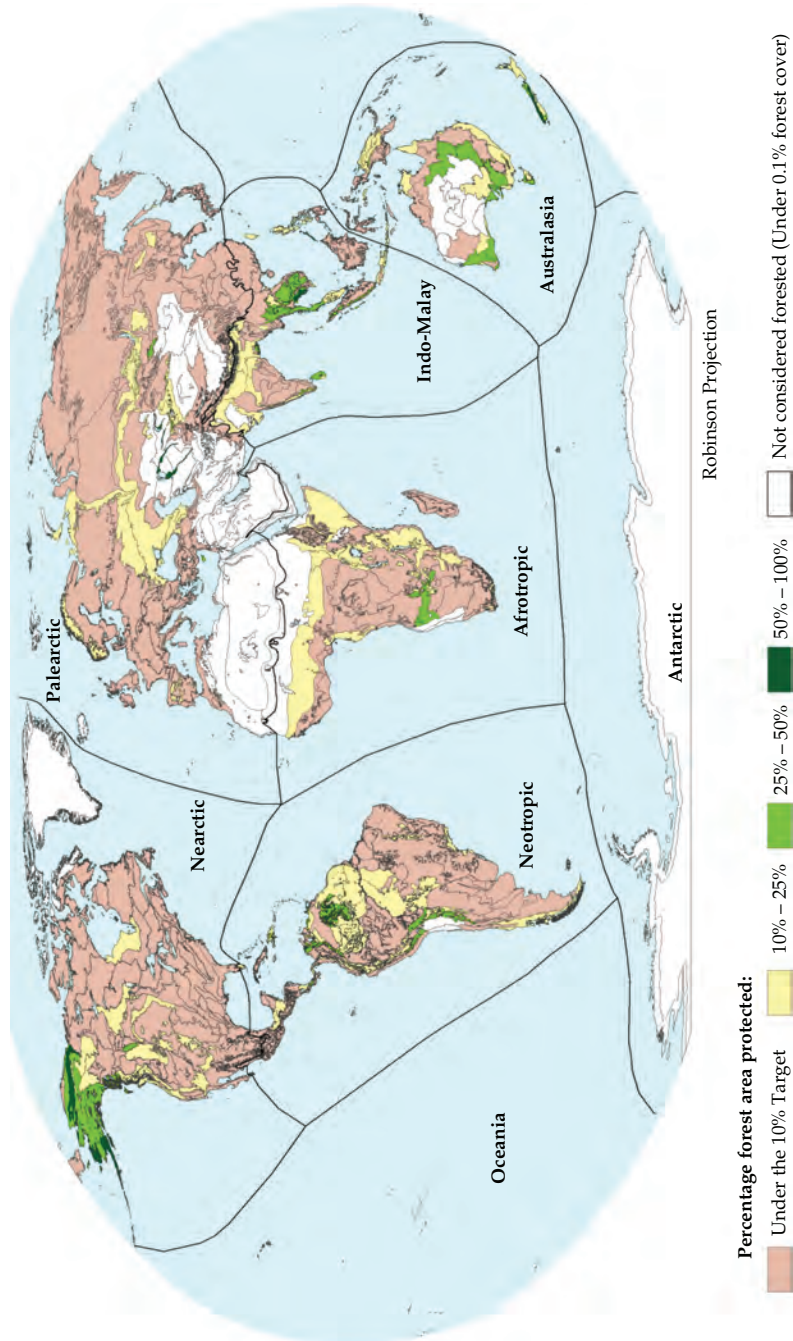


Figure 13.3 Distribution of the percentage of protected forest area within WWF ecoregions. The highest levels of protection can be seen in parts of Australia, the Amazon, Southeast Asia, and Alaska. Notable areas of low protection include the Congo Basin in Central Africa and Northern Boreal forests. Black lines indicate biogeographic realms. White areas indicate no forest cover. Reprinted from Schmitt *et al.* (2009).

It is clear therefore, that partially modified landscapes are an important and valuable asset for biodiversity conservation, and should not be overlooked by biologists and conservationists, and abandoned to yet further levels of intensification.

Against this backdrop of necessity and hope, it is self-evident that the future of much of the world's biodiversity depends on the effective management of human-modified systems (Daily 2001; Lindenmayer and Franklin 2002; Bawa *et al.* 2004). To face up to this challenge conservation biology needs to adopt a research perspective that incorporates human activities as integral components of ecosystems, and place a strong emphasis on understanding the coupled social-ecological dynamics of modified lands (Palmer *et al.* 2004; Sayer and Maginnis 2005).

Ultimately conservation biologists need to improve their understanding of how different types of human land-use may confer different benefits for conservation. To what extent can modified land-uses support viable populations of native species, and help ensure the long-term viability of isolated remnants of undisturbed vegetation? Understanding which native species can maintain viable populations in modified landscapes, and under what management regimes, is one of the greatest challenges currently facing conservation biologists (Fischer and Lindenmayer 2007; Sekercioglu *et al.*, 2007; Sodhi 2008; Chazdon *et al.* 2009a). While it is generally accepted that the conversion of primary habitat for intensive agriculture inevitably leads to dramatic losses in biodiversity (Donald 2004; Sodhi *et al.* 2009), more information is certainly needed. Conservation biologists are particularly uncertain of the extent to which more structurally and floristically complex land-uses such as secondary and agroforests can conserve native biotas (e.g. Dunn 2004; Gardner *et al.* 2007), although mixed agricultural landscapes can be more hospitable to forest birds than once suspected (Daily *et al.* 2001; Ranganathan *et al.* 2008). In the rest of this chapter we briefly outline the biodiversity prospects that exist within different land-use systems, focusing in particular on forested landscapes in the tropics.

13.3 Selectively logged forests

As of 2005, approximately one third of the world's forests—a total of 1.3 billion hectares—were designated primarily for timber production (FAO 2006). In 2006, member nations of the International Tropical Timber Organization (ITTO) exported over 13 million cubic meters of tropical non-coniferous logs worth US\$2.1 billion, making a substantial contribution to the economies of these nations (ITTO 2007). Logging activity on this massive scale has resulted in huge areas of forest being degraded following the selective removal of high-value trees, and the collateral damage associated with tree felling and extraction. Asner *et al.* (2005) estimated that in the Brazilian Amazon between 1999 and 2002 the area of rainforest annually degraded by logging is approximately the same as that which is clear-felled for agriculture (between 12 and 19 million hectares).

Although all logging activity has a negative impact on the structure and composition of the forest, the severity of this impact depends on the logging intensity, including the number of trees removed per ha, length of the rotation time, and site management practices. The density of felled trees varies among regions and management regimes from as few as one tree every several ha (e.g. mahogany, *Swietenia macrophylla* in South America) to more than 15/ha in lowland dipterocarp forests of Southeast Asia (Fimbel *et al.* 2001). In the last few decades Reduced Impact Logging (RIL) techniques have been developed that involve careful planning and controlled harvesting (e.g. preliminary inventories, road planning, directional felling) to greatly minimize deleterious impacts (Fimbel *et al.* 2001; Putz *et al.* 2008).

Differences in how forests are managed determine the extent to which logging negatively affects wildlife, with impacts felt through changes to the structure and composition of the forest environment, including alterations in tree size structure, a shift towards early successional vegetation, changes in composition of fruiting trees, fragmentation of the canopy, soil compaction, and alteration of aquatic environments. In general, broad patterns of wildlife response can be

explained by differences in the intensity of logging activity as well as the amount of recovery time elapsed before a study was conducted (Putz *et al.* 2001).

While there is no available evidence of any species having been driven extinct by selective logging there are abundant data showing marked population declines and local extinctions in a wide range of species groups (Fimbel *et al.* 2001; Meijaard and Sheil 2008). Arboreal vertebrates appear to be particularly badly affected through the loss of nesting and food resources. Both Thiollay (1995) and Sekercioglu (2002) reported losses of approximately 30% of forest dependent birds from logged areas in Sumatra and Uganda, respectively. Felton *et al.* (2003) reported depleted numbers of adult orangutans (*Pongo borneo*) in selectively logged peat forest in Kalimantan, Borneo, compared to neighboring intact sites. Bats also appear to be especially sensitive to even low levels of logging as changes in canopy cover and understory foliage density have knock-on effects on foraging and echolocation strategies (e.g. Peters *et al.* 2006).

Nevertheless, for many taxa the impacts of selective logging are far less severe, even under conventional management regimes. For example, Lewis (2001) found that logging at a density of six stems per hectare had little effect on the diversity and structure of butterfly assemblages in Belize, while Meijaard and Sheil (2008) concluded that only a few terrestrial mammal species have shown marked population declines following logging in Borneo. These studies suggest that different species groups exhibit significantly different responses to logging impacts depending on their life-history strategies and resource requirements. Within any one group it is invariably the forest dependent and specialist species that decline, while generalist and omnivorous species are unaffected or even increase in abundance and diversity.

For most of the world we lack detailed information on the extent to which specific management practices can enhance levels of biodiversity in managed natural forests. Nevertheless, many best practice general guidelines do exist, which, if implemented more broadly, could greatly improve the value of logged forests for wildlife

(Fimbel *et al.* 2001; Lindenmayer *et al.* 2006; Meijaard and Sheil 2008). These guidelines include stand-level practices such as the retention of structural complexity (including dead wood), long-rotation times, maintenance of canopy cover, and fire control and timber removal techniques. In addition many landscape scale measures can greatly improve the value of logged forests for conservation, including the designation of no-take areas, careful road design and maintenance of landscape connectivity with intact corridors and riparian buffers (Gillies and St Clair 2008).

More work is urgently needed to prescribe strategies for effective biodiversity conservation in managed forests. Despite receiving criticism from conservation biologists on the adequacy of criteria to support conservation, timber certification authorities such as the Forest Stewardship Council (www.fsc.org) offer a promising approach to improving the responsibility of forest management standards.

13.4 Agroforestry systems

Agroforestry is a summary term for practices that involve the integration of trees and other woody perennials into crop farming systems through the conservation of existing trees, their active planting and tending, or the tolerance of natural regeneration in fallow areas (Schroth *et al.* 2004). Its main purpose is to diversify production for increased social, economic and environmental benefits, and has attracted increasing attention from scientists working at the interface between integrated natural resource management and biodiversity conservation, especially in tropical countries (Schroth *et al.* 2004; Scherr and McNeely 2007). Farmers in many traditional agricultural systems have maintained or actively included trees as parts of the landscape for thousands of years to provide benefits such as shade, shelter, animal and human food (McNeely 2004).

Although many different definitions exist to define different agroforestry systems, here we highlight two broad categories; complex agroforestry and home-gardens (Scales and Marsden

2008). Complex agroforestry is an extension of the swidden agriculture system where tree seedlings are co-planted with annual crops and left in fallow (e.g. rattan), or maintained in an annual-perennial association (e.g. damar-coffee). After 25–50 years the trees are felled and the cycle is repeated. Home-gardens are small areas of agricultural land located near to houses that are cultivated with a mixture of annuals and perennials, including trees and shrubs. They are semi-permanent and typically more intensively managed than complex agroforests. Because of their high levels of floristic diversity and complex vegetation, agroforests represent a mid-point in forest structural integrity between monoculture plantations and primary forest (Figure 13.4; Schroth and Harvey 2007).

Agroforestry can benefit biodiversity conservation in three ways; the provision of suitable habitat for forest species in areas that have suffered

significant historical deforestation, the provision of a landscape matrix that permits the movement of species among forest remnants, and the provision of livelihoods for local people which may in turn relieve pressure on remaining areas of primary forest (see also Chapter 14). In areas of the tropics that have lost the majority of old-growth forest the dominant near-forest vegetation is frequently comprised of some form of agro-forestry, highlighting the importance of these systems for conservation in some regions, including shade-coffee in Central America, shade-cacao in the Atlantic Forest of Brazil, jungle rubber in the Sumatran lowlands, and home-gardens in countries across the world.

The majority of studies that have examined the biodiversity value of agroforestry systems have found that although some species are invariably lost following conversion of native habitat, a large proportion of the original fauna and flora is maintained when compared to more intensified agricultural land-uses (Ranganathan *et al.* 2008). In reviewing the results of 36 studies Bhagwat *et al.* (2008) found that agroforestry systems consistently hosted more than two-thirds of the species found in reserves, while patterns of similarity in species composition between agroforest plots and areas of native forest ranged from 25% (herbaceous plants) to 65% (mammals). Although existing studies have not revealed any clear pattern regarding which groups of species are unlikely to be conserved within agroforestry systems, it appears that rare and range-restricted species are often those that suffer the greatest declines following forest conversion, while those that increase in abundance are often open-habitat and generalist taxa (Scales and Marsden 2008). However, even species that are usually only found in areas of native vegetation may use agroforests to move between forest remnants, as is the case for two species of sloth in Costa Rica that frequently use shade-cacao plantations as a source of food and resting sites (Vaughan *et al.* 2007).

Differences in the amount of biodiversity that is retained in different agroforestry systems can often be explained by differences in the intensity of past and present management regimes (Bhagwat *et al.* 2008). For example, the effect of



Figure 13.4 Shade-coffee plantation in the Western Ghats, India. Photograph by M. O. Anand.

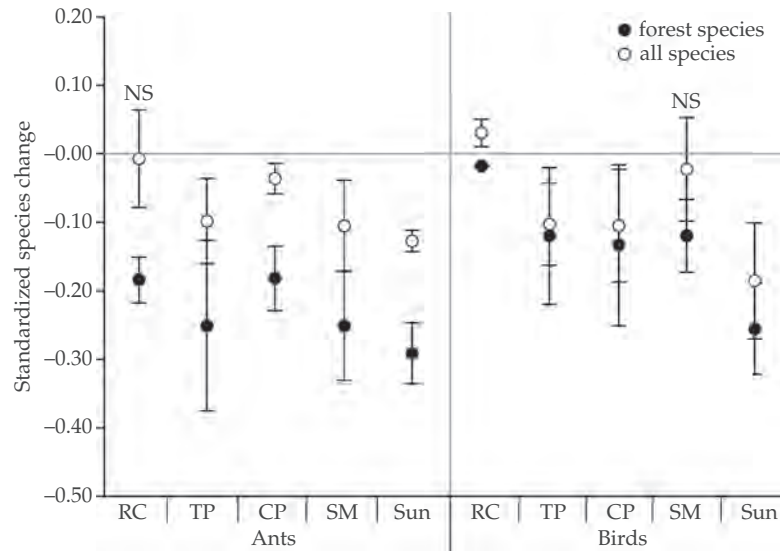


Figure 13.5 Standardized change in species richness for ants and birds in coffee sites compared with nearby forests from 18 datasets in the Neotropics. Error bars are bootstrapped 95% CIs (Confidence Intervals). Points below zero show species loss relative to forests, and points above zero show significant increases in species richness compared with forests. Error bars that do not overlap zero show significantly higher or lower richness in coffee habitats compared with forests (NS, points not significantly different from zero). Habitat abbreviations: RC, rustic coffee; TP, traditional polyculture coffee; CP, commercial polyculture coffee; SM, shade monoculture coffee; Sun, sun coffee. Reprinted from Philpott *et al.* (2008).

management intensification on biodiversity is clearly demonstrated by the marked loss of forest species following the simplification of shade-coffee plantations and a decrease in the density and diversity of shade trees (Figure 13.5; reviewed by Philpott *et al.* 2008).

Despite the potential value of agroforestry systems for biodiversity, it is important to recognize key limitations in their contribution towards long-term conservation strategies. First, the ability of agroforestry systems to maintain a significant proportion of the regional biota depends on the maintenance of sufficient areas of natural habitat, both to support highly sensitive species (Schroth and Harvey 2007) and to provide source populations (Anand *et al.* 2008). By encompassing sufficient areas of native forest within an agroforestry landscape it is possible to ensure the persistence of a large number of species for very long time periods, as recently demonstrated by Ranganathan *et al.* (2008) who reported the presence of more than 90% of the regional forest avifauna in arecanut (*Areca catechu*) production systems that have been cultivated for more than 2000

years in the Western Ghats, India. Second, appropriate regulations on hunting and resource extraction are vital to ensure that keystone vertebrate and plant species are not depleted from otherwise diverse systems. Finally, and most importantly, agroforestry systems can only survive with the support of market incentives and favorable land-use policies that maintain viable livelihoods of local people, and prevent conversion to more intensified land-uses (Steffan-Dewenter *et al.* 2007).

13.5 Tree plantations

As for agroforestry systems, tree plantations have the potential to make an important contribution to biodiversity conservation for two key reasons: (i) they may more closely reflect the structural complexity of native forest than many more intensive production land-uses; and (ii) they occupy a large area of once-forested land in many parts of the world. The total area of the plantation forest estate in 2005 was about 109

million hectares, and is continuing to increase by approximately 2.5 million hectares per year (FAO 2006). In the tropics alone, the total coverage of plantation forestry increased from approximately 17 million hectares in 1980 to 70 million hectares in 2000 (FAO 2006). As demands for timber and wood fiber continue to increase around the world, it is highly likely that these upward trends will persist or even accelerate.

Many tree plantations have been traditionally labeled as “green deserts”, and are presumed or found to be hostile to native species and largely devoid of wildlife (Kanowski *et al.* 2005; Sodhi *et al.* 2009). However, closer inspection of available data indicates that while it is certainly true that some intensively managed plantation monocultures offer very little value to biodiversity (e.g. oil palm in Southeast Asia; Koh and Wilcove 2007, 2008, 2009; Koh 2008a, b), other plantation systems may provide valuable species habitat, even for some threatened and endangered taxa (Hartley 2002; Carnus *et al.* 2006). This apparent contradiction is explained in part by marked differences in the levels of biodiversity that can be supported by different types of plantation. For example there is a stark contrast in the conservation value of industrial monocultures of exotic species that often have little or no intrinsic value for native forest species, compared with complex multi-species plantations that encompass remnants of native vegetation and are managed as a mosaic of differently aged stands (Hartley 2002; Lindenmayer and Hobbs 2004; Kanowski *et al.* 2005). However, a second reason why many plantations are incorrectly presumed to be biological deserts is that human perceptions of habitat quality are often distinct from how native species themselves perceive the landscape (Lindenmayer *et al.* 2003). Although few comprehensive and robust field studies have been conducted to examine the conservation value of plantations, those that exist suggest that under certain conditions the numbers of species inhabiting these areas may be greater than expected. For example, a very thorough study in north-east Brazilian Amazonia found that *Eucalyptus* plantations contained nearly half of the regional forest fauna, although it is very unlikely that all of

these taxa could maintain viable populations in the absence of large areas of neighboring primary forest (Barlow *et al.* 2007; see Box 13.2).

The value of a given plantation forest for conservation is partly determined by how it is managed. For example, at the stand level, many studies have found that faunal diversity in tree plantations is strongly influenced by the maintenance of structural attributes such as snags and dead wood, and the tolerance of succession by native plant species in the understory (Hartley 2002). More floristically and structurally complex plantations provide more resources for many forest species (e.g. fruit feeding butterflies; Barlow *et al.* 2008). At the landscape scale, spatial heterogeneity in stand management and age has been shown to be a key factor in determining the overall level of diversity within a given plantation forest (Lindenmayer and Hobbs 2004; Lindenmayer *et al.* 2006).

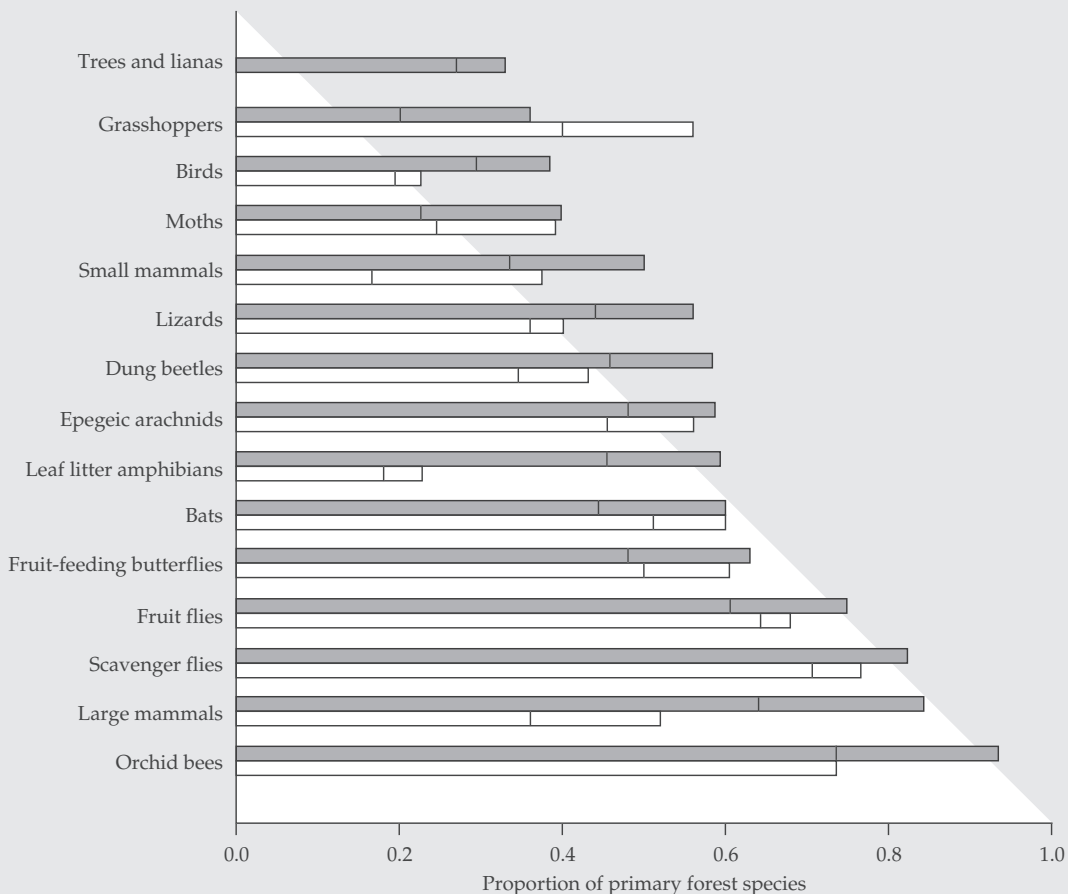
However, the true conservation value of a plantation depends upon the comparison with alternative land-uses that may otherwise exist in its place (Kanowski *et al.* 2005; Brockerhoff *et al.* 2008). Clearly there is a net loss of biodiversity if plantations replace native forest. There is also a net loss of regional biodiversity if plantations are grown on areas of natural grassland, as seen in many areas of southern Africa. However, if plantations represent the “lesser evil” and prevent land from being converted to croplands or pasture, or have been grown on areas of degraded land, then their importance for biodiversity may be significant. In areas where very little native vegetation remains plantation forests may provide the last refuge for endemic species, such as the case of the critically endangered ground beetle (*Holcaspis brevicula*) in New Zealand which is only known from *Pinus* plantations (Brockerhoff *et al.* 2005).

Ultimately, the extent to which plantations can be managed to enhance biodiversity depends upon the level of economic cost incurred by responsible management strategies, and the availability of market incentives to offset such costs. Some minor improvements in management technique may generate some conservation benefits with little loss in productivity (Hartley 2002) but our knowledge of the economic-conservation trade-offs implicit in major changes to stand and

Box 13.2 Quantifying the biodiversity value of tropical secondary forests and exotic tree plantations
Jos Barlow

Ecologists and conservation scientists have found it difficult to make an accurate assessment of the conservation value of secondary and plantation forests in the tropics. Many studies have been conducted in small forest blocks, and may be influenced by the presence of transient species moving between patches of adjacent old-growth forest. Furthermore, studies are often beset by a variety of methodological shortcomings. As a result, there is little consistency in their results, and studies may systematically overestimate the conservation value of non-primary forests (Gardner *et al.* 2007).

Many of these potential methodological shortcomings were addressed by a recent comprehensive study that utilized a quasi-experimental landscape mosaic that resulted from a large-scale attempt to implement fast-growing tree monocultures in the Brazilian Amazon in the 1970s. In 2004, a large international team of researchers attempted to quantify the biodiversity that persists in primary forests, 4–5 year old *Eucalyptus* plantations and 14–19 year old native second-growth (Barlow *et al.* 2007). They sampled 15 different groups of biodiversity, including most of the terrestrial vertebrates, a wide range of invertebrates, and



Box 13.2 Figure The proportion of primary forest species that were also recorded in 14–19 year old native second growth (grey bars) and 4–6 year old *Eucalyptus* plantations (white bars) in the Brazilian Amazon. The bars are split by a line that indicates the decrease in the proportion of primary forest species when occasional species (those that were recorded only once in each of the non-primary forests) are removed from the comparison.

continues

Box 13.2 (Continued)

the trees and lianas (see Box 13.2 Figure). The researchers spent >18 200 person hours collecting specimens in the field and identifying them in the laboratory, and recorded 61 325 individuals and identified 1442 species.

Their results provide a clear message regarding the unique value of primary or old-growth forests. Averaging across all taxa, secondary forests and Eucalyptus plantations held only 59% and 47% of the species that were recorded in the old-growth forests, respectively. These results should be interpreted as a best case scenario, as the wider landscape was dominated by old-growth forests, maximizing recolonization opportunities for primary forest species. Furthermore, many primary forest species were recorded just once within the non-primary habitats, and the presence of single individuals is unlikely to represent a species ability to persist in these regenerating forests. Removing these occasional species from the results reduces the estimated value of non-primary habitats for most taxa (Box 13.2 Figure) to an average of 46% of species for second-growth and 39% for plantations.

This research was unique as it allows us to make a robust comparison between the responses of different taxa across the same land-use gradient. This shows that the estimated value of non-primary forests is much higher for highly mobile taxa such as orchid bees, large mammals, and bats (see Box 13.2 Figure), which include many mobile species that

fly tens of kilometers each day, and perceive landscape and habitat quality at a very large-scale. There was also a marked difference among taxa in the kinds of species that come to dominate these non-primary forests. For example, more than 60% of the species of birds, grasshoppers and moths that were recorded in secondary forests were never recorded in old-growth forests. These taxa contrast with the orchid bees, fruit flies and large mammals, for which most of the species recorded in secondary forests (more than 75%) were also recorded in primary forests. These data illustrate an important point about the consequences of land-use change; the species persisting in anthropogenic habitats can be either composed of a subset of the species found in primary forests, or like birds, they may be wide-ranging generalists that have invaded from open habitats, riparian vegetation, and even urban areas.

REFERENCES AND SUGGESTED READING

- Barlow, J., Gardner, T. A., Araujo, I. S., *et al.* (2007). Quantifying the biodiversity value of tropical primary, secondary and plantation forests. *Proceedings of the National Academy of Science of the United States of America*, **104**, 18555–18560.
- Gardner, T. A., Barlow, J., Parry, L. T. W., and Peres, C. A. (2007). Predicting the Uncertain Future of Tropical Forest Species in a Data Vacuum. *Biotropica*, **39**, 25–30.

landscape management regimes is poor. An alternative to more ecologically sensitive management within individual plantations which also deserves further research attention is to adopt a land sparing approach, where intensified silviculture in one area generates sufficient revenue to “spare” other lands for conservation (e.g. Cyranoski 2007; see next section).

13.6 Agricultural land

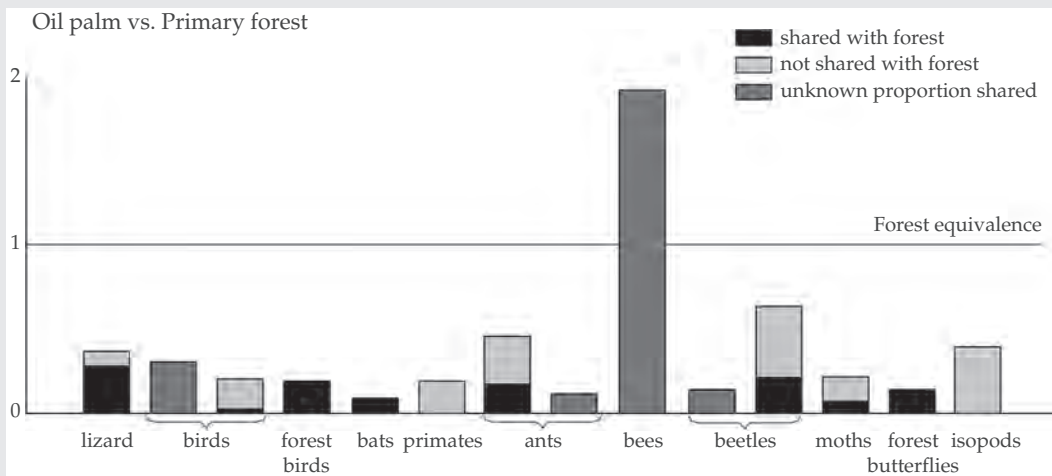
The human population is expected to increase from 6 billion today to 8–10 billion by 2050

(Cohen 2003). Global demand for agricultural products is predicted to grow even faster due to rising demand for food and higher quality food (e.g. meat), as well as for bioenergy crops used in biofuel production (UN 2005; Scherr and McNeely 2008). It has been estimated that feeding a population of 9 billion people would require the conversion of another billion hectares of natural habitats to croplands (Tilman *et al.* 2001), which will almost certainly increase the risks of extinction already faced by numerous species worldwide (see Boxes 13.3 and Introduction Box 1).

Box 13.3 Conservation in the face of oil palm expansion
Matthew Struebig, Ben Phalan, and Emily Fitzherbert

The African oil palm (*Elaeis guineensis*) is one of the world's most rapidly expanding crops, and has the highest yields and largest market share of all oil crops. While cultivation has historically focused in Malaysia and Indonesia, oil palm is increasingly grown across the lowlands of other countries in Southeast Asia, Latin America and Central Africa. Expansion is driven by large companies and smallholders responding to global demand for vegetable oil (mainly from Indonesia, India and China), and the growing biofuel markets of the European Union. With high demand, and strong overlap between areas suitable for oil palm and those of endemic-rich tropical forests, expansion poses an increasing threat to biodiversity.

The few studies available show that oil palm is a poor substitute habitat for the majority of tropical forest species, particularly those of conservation concern. On average only 15% of species recorded in primary forest are found in oil palm plantations (Box 13.3 Figure), even fewer than in most other tree crops. Plantation assemblages are typically dominated by a few abundant generalists (e.g. macaques), alien invasives (e.g. crazy ants), pests (e.g. rats), and their predators (e.g. pythons). Oil palm is a major contributor to deforestation in a few countries, although its role is sometimes obscured by ambiguous land-tenure laws and its links with other enterprises (e.g. timber profits are used to offset plantation establishment costs in Indonesia).



Box 13.3 Figure The biodiversity impact of converting forests to plantations is shown by comparing species richness in oil palm relative to primary forest. The species richness of oil palm is presented as a proportion of forest richness such that equal species richness is 1. Each column contains a study of one taxon and shows the proportion of oil palm species shared and those not shared with forest. One study of bees found fewer species in forests than oil palm, but might have underestimated forest species richness because the canopy was not sampled.

In response to consumer concerns about deforestation, the Roundtable for Sustainable Palm Oil (RSPO) was formed from industry-NGO (non-governmental organization)

collaboration in 2003. Under this scheme members commit to environmental and social standards for responsible palm oil production, including an assurance that no forests of High

continues

Box 13.3 (Continued)

Conservation Value (<http://www.hcvnetwork.org/>) will be cleared for plantations. However, certification is not yet a panacea. Unless land planning is expanded to regional assessments, biodiversity losses outside of RSPO-member plantations will continue; certification risks remaining a niche market, with mainly older plantations exporting to responsible buyers, while demand from others is filled by newer plantations pushing into forests.

Conservation science is needed to inform oil palm policies, but it is not enough to understand only the biodiversity impacts of plantations. The real challenge is for

conservation scientists to translate their findings into better land planning and forest protection strategies, whilst accounting for social, economic and political realities.

SUGGESTED READING

- Fitzherbert, E. B., Struebig, M. J., Morel, A., *et al.* (2008). How will oil palm expansion affect biodiversity? *Trends in Ecology and Evolution*, **23**, 538–545.
- Koh, L. P. and Wilcove, D. S. (2008). Is oil palm agriculture really destroying tropical biodiversity? *Conservation Letters*, **1**, 60–64.

What can conservation biologists do to mitigate the threat from agricultural expansion? This problem has traditionally been framed as a zero-sum game—agricultural production will take away land that would otherwise be used for biodiversity conservation, and vice versa. More recently however, researchers have suggested that “countryside biogeography” (also known as “win-win ecology” or “reconciliation ecology”) should be a key consideration in practical conservation (Dale *et al.* 2000; Daily *et al.* 2001; Miller and Hobbs 2002; Daily 2003; Rosenzweig 2003).

Proponents of countryside biogeography argue that because a large proportion of the planet is already dominated by humans and what little remains of pristine habitats will not be sufficient for the long-term survival of many species, conservation planning should include mitigation measures that enable human activities to proceed with minimum displacement of native species (Rosenzweig 2003, see Box 13.4).

In the context of agricultural expansion, it is often the case that after natural habitats have been converted, what remains is an agricultural mosaic—forest fragments in a matrix of production systems (Vandermeer and Perfecto 2007). Both theoretical and empirical ecological research over the past decade has shown that species survival in such fragmented landscapes depends on the size and isolation of fragments,

as well as the permeability of the intervening matrix to the movement of organisms (Hanski 1999; Stratford and Stouffer 1999; Vandermeer and Carvajal 2001; Perfecto and Vandermeer 2002; Chapter 5).

To enhance the survivability of native species in an agricultural mosaic, two approaches may be pursued. The first approach is to intensify agricultural production to increase overall yield while avoiding further cropland expansion and deforestation (Balmford *et al.* 2005; Green *et al.* 2005). This “land sparing” approach, though conceptually straightforward, remains controversial among the conservation community. Critics have argued that the ecological impacts of intensive farming often extend over a wider area than the land so farmed (Matson and Vitousek 2006). Intensive farming would require more irrigation, and fertilizer and pesticide inputs, which would divert water away from downstream ecosystems and species, and result in greater pollution. Furthermore, intensifying agricultural production could lead to extensive land use by displacing people to other forested areas or by providing the economic incentives for migration into the area (Matson and Vitousek 2006).

A second approach is to focus on improving the quality of the matrix to make it more hospitable for habitat generalist species that are able to utilize it, and be less of a barrier to the migration

Box 13.4 Countryside biogeography: harmonizing biodiversity and agriculture
Jai Ranganathan and Gretchen C. Daily

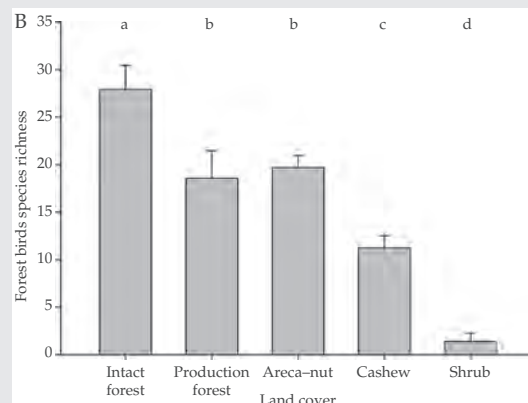
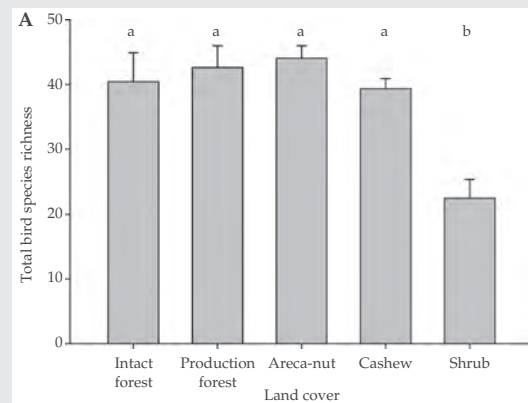
With human impacts expected to intensify rapidly (e.g. Tilman *et al.* 2001), the future of biodiversity cannot be separated from the future of people. Although protected areas are central to conservation strategy, they alone are unlikely to ensure survival of more than a tiny fraction of Earth’s biodiversity (e.g. Rosenweig 2003). Here we discuss the scope for expanding conservation strategy to include the countryside: active and fallow agricultural plots, gardens and pasture, plantation or managed forest, and remnants of native vegetation in landscapes otherwise devoted primarily to human activities (Daily *et al.* 2001). Little is known about the capacity of the countryside to support native species, particularly in the tropics, where the majority of the Earth’s species are found (Wilson and Peter 1988).

We summarize information on the best-studied groups—birds, mammals, and insects—in well-studied systems in Mesoamerica. On the question of what fraction of native species can survive in countryside, the answer appears to be about 50% or more, though abundance of many species is low (Estrada *et al.* 1997; Daily *et al.* 2001; Daily *et al.* 2003; Horner-Devine *et al.* 2003). Three landscape characteristics stand out as important in conferring a survival advantage to native species in the countryside. First, species richness is considerably higher in the vicinity of large remnants of relatively intact forest, suggesting that many species that occur in the countryside can persist only in the nearby presence of that native forest (Estrada *et al.* 1997; Ricketts *et al.* 2001; Perfecto and Vandermeer 2002; Sekercioglu *et al.* 2007). Second, the presence of native vegetation in human-dominated habitat (in the form of living fences, windbreaks, and remnant trees) facilitates persistence (Estrada *et al.* 1994; Estrada *et al.* 2000; Hughes *et al.* 2002; Harvey *et al.* 2004). Third, the intensity of agriculture in a landscape is negatively correlated with that landscape’s conservation potential (Bignal and McCracken 1996; Green *et al.* 2005).

The question of which attributes of native species confer an advantage in the countryside has perhaps been best studied in birds, where a high population growth rate and the ability to disperse through open habitat greatly increases the chance of occurrence in the countryside (Sekercioglu *et al.* 2002; Pereira *et al.* 2004). Additionally, the conversion of forest to

agriculture severely impacts forest-interior bird species; in one case the cause seemed to be a decrease in available nesting habitat (Lindell and Smith 2003).

It is uncertain if high levels of native diversity can be maintained over the long term (centuries to millennia), as almost all of the countryside under study has been under cultivation for less than a century (at least in recent centuries). A possible indication of the long-term prospects can be found within the Western Ghats mountain range, India, where high levels of bird diversity have been maintained in a low-intensity agricultural landscape, despite >2000 years of continuous agricultural use (Ranganathan *et al.* 2008, see Box 13.4 Figures 1 and 2). Though tentative, these results show that conservation investments in countryside may pay off for biodiversity in the long term.



Box 13.4 Figure 1 Patterns of bird species richness within an agricultural landscape on the fringes of the Western Ghats, India, where land use patterns help to maintain avian diversity (reprinted from Ranganathan *et al.* 2008). There are five major land covers in the landscape: forest (itself divided into relatively pristine “intact

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Box 13.4 (Continued)

forest" and "production forest", within which the extraction of non-timber forest products is permitted), arecanut plantations, cashew plantations, shrubland, and rice/peanut farms. The last land cover was omitted from analysis due to the fact that they are seasonally devoid of vegetation and, thus, wildlife. With the exception of the depauperate shrublands, the land covers contained a similar richness of birds (A). However, when just birds associated with forest habitat ("forest species") are examined, much larger differences can be seen, with production forest and arecanut plantations second only to intact forest in richness (B). Thus, it can be seen that arecanut plantations are important for maintaining forest species across the landscape. Their importance is all the greater because the production forests serve primarily as a source of agricultural inputs for the arecanut plantations, thereby providing a powerful economic incentive to maintain those areas as forest. © National Academy of Sciences, USA.



Box 13.4 Figure 2 Biodiversity of birds, and likely other taxa, is especially rich in the low-intensity agricultural landscapes on the fringes of the Western Ghats, India. Photograph by J. Ranganathan.

The time is ripe for developing and promoting best management practices for farmers—and, similarly, best conservation practices for conservation organizations—that integrate biodiversity and human well-being in meaningful, effective ways globally.

REFERENCES

- Bignal, E. M., and McCracken, D. I. (1996). Low-intensity farming systems in the conservation of the countryside. *Journal of Applied Ecology*, **33**, 413–424.
- Daily, G. C., Ehrlich, P. R., and Sanchez-Azofeifa, G. A. (2001). Countryside biogeography: Use of human-dominated habitats by the avifauna of southern Costa Rica. *Ecological Applications*, **11**, 1–13.
- Daily, G. C., Ceballos, G., Pacheco, J. Suzan, G., and Sanchez-Azofeifa, A. (2003). Countryside biogeography of neotropical mammals: Conservation opportunities in agricultural landscapes of Costa Rica. *Conservation Biology*, **17**, 1814–1826.
- Estrada, A., Coates-Estrada, R., and Meritt, D. (1994). Non-flying Mammals and Landscape Changes in the Tropical Rain-Forest Region of Los-Tuxtlas, Mexico. *Ecography*, **17**, 229–241.
- Estrada, A., Coates-Estrada, R., and Meritt, D. A. (1997). Anthropogenic landscape changes and avian diversity at Los Tuxtlas, Mexico. *Biodiversity and Conservation*, **6**, 19–43.
- Estrada, A., Cammarano, P., and Coates-Estrada, R. (2000). Bird species richness in vegetation fences and in strips of residual rain forest vegetation at Los Tuxtlas, Mexico. *Biodiversity and Conservation*, **9**, 1399–1416.
- Green, R. E., Cornell, S. J., Scharlemann, J. P. W., and Balmford, A. (2005). Farming and the fate of wild nature. *Science*, **307**, 550–555.
- Harvey, C. A., Tucker, N. I. J., and Estrada, A. (2004). Live fences, isolated trees, and windbreaks: tools for conserving biodiversity in fragmented tropical landscapes. In G. Schroth, G. A. B. da Fonseca, C. A. Harvey, C. Gascon, H. L. Vasconcelos, and A.-M. N. Izac, eds *Agroforestry and biodiversity conservation in tropical landscapes*, pp. 261–289. Island Press, Washington, DC.
- Horner-Devine, M. C., Daily, G. C., Ehrlich, P. R., and Boggs, C. L. (2003). Countryside biogeography of tropical butterflies. *Conservation Biology*, **17**, 168–177.
- Hughes, J. B., Daily, G. C., and Ehrlich, P. R. (2002). Conservation of tropical forest birds in countryside habitats. *Ecology Letters*, **5**, 121–129.
- Lindell, C. and Smith, M. (2003). Nesting bird species in sun coffee, pasture, and understory forest in southern Costa Rica. *Biodiversity and Conservation*, **12**, 423–440.
- Pereira, H. M., Daily, G. C., and Roughgarden, J. (2004). A framework for assessing the relative vulnerability of species to land-use change. *Ecological Applications*, **14**, 730–742.
- Perfecto, I. and Vandermeer, J. (2002). Quality of agro-ecological matrix in a tropical montane landscape: Ants in coffee plantations in southern Mexico. *Conservation Biology*, **16**, 174–182.
- Ranganathan, J., Daniels, R. J. R., Chandran, M. D. S., Ehrlich, P. R., and Daily, G. C. (2008). How biodiversity can live with agriculture. *Proceedings of the National*

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Box 13.4 (Continued)

Academy of Sciences of the United States of America, **105**, 17852–17854.

Ricketts, T. H., Daily, G. C., Ehrlich, P. R., and Fay, J. P. (2001). Countryside biogeography of moths in a fragmented landscape: Biodiversity in native and agricultural habitats. *Conservation Biology*, **15**, 378–388.

Rosenzweig, M. L. (2003). *Win-win ecology: how the earth's species can survive in the midst of human enterprise*. Oxford University Press, Oxford, UK.

Sekercioglu, C. H., Ehrlich, P. R., Daily, G. C., Aygen, D., Goehring, D., and Sandi, R. F. (2002). Disap-

pearance of insectivorous birds from tropical forest fragments. *Proceedings of the National Academy of Sciences of the United States of America*, **99**, 263–267.

Sekercioglu, C. H., Loarie, S. R., Brenes, F. O., Ehrlich, P. R., and Daily, G. C. (2007). Persistence of forest birds in the Costa Rican agricultural countryside. *Conservation Biology*, **21**, 482–494.

Tilman, D., Fargione, J., Wolff, B., et al. (2001). Forecasting agriculturally driven global environmental change. *Science*, **292**, 281–284.

Wilson, E. O. and Peter, F. M., eds (1988). *Biodiversity*. National Academy Press, Washington, DC.

of forest specialist species between forest fragments (Vandermeer and Perfecto 2007). The goal is to increase the permeability of the matrix, which is critical for the long-term persistence of metapopulations and metacommunities (Hanski 1999; Stratford and Stouffer 1999; Vandermeer and Carvajal 2001; Chapter 5).

13.7 Urban areas

Urban areas represent an extreme case in the spectrum of human-modified land uses. Unlike the other forms of habitat modification discussed above, urbanization often irreversibly replaces natural habitats with persistent artificial ones, resulting in long-term impacts on many native species (Stein *et al.* 2000). Despite the rapid rate at which urban sprawl is occurring worldwide, urban ecology has received relatively little attention from conservation biologists (Miller and Hobbs 2002). This can be attributed to the traditional focus of conservation research on “natural” ecosystems such as old-growth forests (Fazey *et al.* 2005).

As the trend of rapid economic growth continues in the tropics, urban areas will likely be increasingly ubiquitous in the tropics. An obvious research agenda, therefore, is to understand the response of tropical species to urbanization

and to develop effective measures for their conservation. We ideally would want to be able to excise a tropical country, allow it to fulfill its economic potential and experience the associated landscape changes within a greatly accelerated time frame, and use this natural laboratory to study what species survive, where they persist and how they are able to do so. The island nation of Singapore in tropical Southeast Asia represents just such an ecological worst case scenario (Sodhi *et al.* 2004).

Koh and Sodhi (2004) studied butterfly diversity in Singapore, and found that forest reserves had higher species richness than secondary forest fragments and urban manmade parks (Figure 13.6). They attributed this to the larger areas of forest reserves and greater floristic complexity (compared to the other habitats they studied), which can sustain larger populations of species with lower risks of extinction, and contain greater diversities of microhabitats with myriad ecological niches that can support more species (MacArthur and Wilson 1963, 1967; Simberloff 1974; Laurance *et al.* 2002). Koh and Sodhi further explained that the last remaining tracts of old-growth vegetation in forest reserves can provide the unique microclimatic conditions such as a closed canopy, and specific larval host plants vital to the persistence of specialist butterfly species.



Figure 13.6 Urban manmade park in Singapore. Photograph by Lian Pin Koh.

A second important finding of Koh and Sodhi's study was that urban parks adjoining forests were more diverse than secondary forest fragments. This was likely due to the prevalence of numerous ornamental flowering plants cultivated in these urban parks, which can support resident butterfly species adapted to an open canopy, as well as species from adjacent forests that forage in these parks. Indeed, the authors reported that both the number of potential larval host-plant species and the amount of surrounding forest cover were statistically significant predictors of butterfly species richness in urban parks.

Koh and Sodhi's study has two key conservation lessons: first, in highly urbanized tropical landscapes the least human-disturbed land uses are likely also most valuable for preserving the native biodiversity, and should therefore be given the highest conservation priority; second, with a good understanding of the biology of organisms, it is possible to enhance the conservation value of manmade habitats within human-modified landscapes. Although urban landscapes represent the worst case scenario in ecosystem management we are increasingly faced with the task of conserving species in such "unnatural" environments. Therefore, it is crucial that more research be focused on developing viable strategies for the effective conservation of biodiversity in urban landscapes.

13.8 Regenerating forests on degraded land

In most areas of the world, secondary forests regenerate naturally on abandoned agricultural land if human disturbance declines. Following centuries of human disturbance, the total area of regenerating forest is now enormous (millions of hectares). Indeed, for parts of the world that have suffered widespread historical deforestation secondary forests comprise the majority of remaining forest area (e.g. east coast of the USA, much of Western Europe, and areas of high human population density like Singapore). In the tropics secondary regrowth together with degraded old-growth forests (e.g. through logging, fire, fragmentation) comprise roughly half of the world's remaining tropical forest area (ITTO 2002).

Understanding the potential importance of these large areas of secondary forest for conservation has attracted much research attention from ecologists and conservation biologists, as well as considerable controversy. For example, Wright and Muller-Landau (2006) recently proposed that the regeneration of secondary forests in degraded tropical landscapes is likely to avert the widely anticipated mass extinction of native forest species. However, other researchers have highlighted serious inadequacies in the quantity

and quality of species data that underpin this claim, casting doubt on the potential for secondary forest to serve as a “safety net” for tropical biodiversity (Brook *et al.* 2006; Gardner *et al.* 2007).

In perhaps the only quantitative summary of biodiversity responses to forest regeneration to date, Dunn (2004) analyzed data from 39 tropical data sets and concluded that species richness of some faunal assemblages can recover to levels similar to mature forest within 20–40 years, but that recovery of species composition is likely to take substantially longer. The recovery of biodiversity in secondary forests varies strongly between different species groups depending on their life histories with species responses generally falling into three categories (Bowen *et al.* 2007): (i) species that decline in abundance or are absent from regrowth due to specialist habitat requirements; (ii) old-growth forest species that benefit from altered conditions in regenerating forest and increase in abundance or distribution; and (iii) open-area species that invade regenerating areas to exploit newly available resources. These conclusions are mirrored by the results the comprehensive Jari study in north-east Brazil that found that 41% of old-growth vertebrate and invertebrate species were lacking from secondary forests of 12–18 years of age, and that species responses varied strongly among and within taxonomic groups (see Box 13.2).

The general lack of data and the context dependent nature of existing studies on biodiversity recovery in secondary forests severely limit our ability to make general predictions about the potential for species conservation in tropical secondary forests (Chazdon *et al.* 2009b). However, we can conclude that secondary forests are likely to be more diverse the more closely they reflect the structural, functional, and compositional properties of mature forest and are set within a favorable landscape context (Chazdon 2003; Bowen *et al.* 2007). In particular, the conservation of old-growth species in secondary forests will be maximized in areas where extensive tracts of old-growth forest remain within the wider region, older secondary forests have persisted, post-conversion land-use was of limited duration

and low intensity, post-abandonment anthropogenic disturbance is relatively low, seed dispersing fauna are protected, and old-growth forests are close to abandoned sites (Chazdon *et al.* 2009b).

The conservation value of a secondary forest should increase over time as old-growth species accumulate during forest recovery, but older secondary forests are poorly studied and long-term datasets are lacking. Existing chronosequence studies of regenerating forests demonstrate that biotic recovery occurs over considerably longer time scales than structural recovery, and that re-establishment of certain species and functional group composition can take centuries or millennia (de Walt *et al.* 2003; Liebsch *et al.* 2008). However, for much of the world, secondary forests exist in highly dynamic landscape mosaics and are invariably clear-felled within one or two decades, thereby greatly limiting the opportunity for these forests to develop into older successional stands that are of higher value for conservation (Chazdon *et al.* 2009b).

Despite this uncertainty, regeneration represents the only remaining conservation option for many regions of the world that have suffered severe historical deforestation. An estimated 350 million hectares of the tropics are classified as degraded due to poor management (Maginnis and Jackson 2005). While the natural recovery of this land is not inevitable there is encouraging evidence that judicious approaches to reforestation can greatly facilitate the regeneration process and enhance the prospects of biodiversity in modified landscapes (Chazdon 2008).

13.9 Conservation and human livelihoods in modified landscapes

Modified and degraded landscapes around the world are not only of vital importance for biodiversity conservation, but are also home to millions of the world’s poorest people. This is especially true in tropical countries where areas of high species richness and endemism frequently overlap with centers of human population density (e.g. sub-Saharan Africa; Balmford *et al.*

2001). It is estimated that the livelihoods of at least 300 million rural poor in tropical countries depend upon degraded or secondary forests (ITTO 2002). For impoverished communities biodiversity is about the basic human needs of eating, staying healthy, and finding shelter (see Chapter 14). Furthermore, it is local people that ultimately decide the fate of their local environments, even if the decisions they make fall within a wider political, social and economic context (Sodhi *et al.* 2006, 2008; Ghazoul 2007; Chapter 14).

These facts make it clear that human livelihoods and poverty concerns need to receive

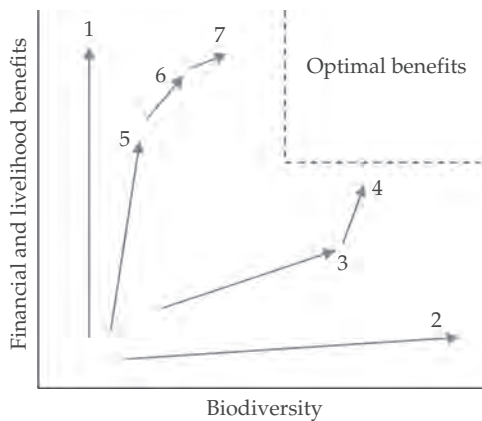


Figure 13.7 Balancing trade-offs between human livelihoods and biodiversity conservation in reforestation projects. Arrows represent alternative reforestation methods. Traditional monoculture plantations of exotic species (arrow 1) mostly generate just financial benefits, whereas restoration using methods that maximize diversity and enhance biodiversity (arrow 2) yields few direct financial benefits to landowners, at least in the short term. Protecting forest regrowth (arrow 3) generates improvements in both biodiversity and livelihoods, although the magnitude of the benefits depends on the population density of commercially or socially important species; these can be increased by enrichment of secondary forest with commercially attractive species (arrow 4). Restoration in landscapes where poverty is common necessitates attempting both objectives simultaneously. But, in many situations, it may be necessary to give initial priority to forms of reforestation that improve financial benefits, such as woodlots and agroforestry systems (arrow 5). In subsequent rotations, this balance might change over time (moving to arrow 6 and later to arrow 7 by using a greater variety of species). There may be greater scope for achieving multiple objectives by using several of these options at different locations across the landscape. Reproduced with permission from Lamb *et al.* (2005).

high priority in the conservation agenda if we are to develop management strategies for agricultural and modified landscapes that are not only viable into the long term, but are also socially just (Perfecto and Vandermeer 2008; Chapter 14). Recognition of this broader challenge has led to calls for a “pro-poor” approach to conservation (Kaimowitz and Sheil 2007). However, developing such an approach and successfully reconciling the interdependent objectives of poverty alleviation and biodiversity conservation is far from trivial. Opportunities for much-sought after “win-win” solutions (Rosenzweig 2003) are often hard if not impossible to achieve when faced by real-world trade-offs between economic and conservation goals, especially in the short-term. However, with careful planning and good science there is significant potential for synergies in achieving development and biodiversity benefits in the management of modified landscapes (Figure 13.7; Lamb *et al.* 2005).

The greatest difficulty in developing a pro-poor approach to biodiversity conservation lies in the fact that the structure and dynamics of human communities, and their interactions with the local environment, varies significantly across different parts of the world. There are no silver bullet, “off the shelf” solutions that can be successfully applied to any situation. Instead individual management strategies for individual landscapes need to be developed with explicit recognition of the socioeconomic, political, and ecological context within which they are embedded (Osrom 2007). Furthermore, it is not enough to accommodate development considerations that do no more than secure livelihood levels at subsistence levels. Local guardians of modified landscapes have the right to develop management strategies that generate higher economic returns that can raise them out of poverty (Ghazoul 2007).

13.10 Conclusion

The challenge of safeguarding the future of tropical forest species is daunting. Spatial and temporal patterns of biodiversity in modified

landscapes are the product of interacting human and ecological processes that vary strongly between different land-use systems and among regions, and have effects that may take years to become fully manifest (Gardner *et al.* 2009). Conservation biologists have little option but to tackle this challenge head-on as very few, if any tropical forest species exist in isolation from human interference. Perhaps the most important conclusion to emerge from biodiversity research in modified landscapes is that different human land-uses can have enormously different implications for conservation. In this chapter we have shown that a broad gradient of structural complexity and species diversity exists from lightly logged production forests at one end to intensive arable and pastoral systems and semi-urban landscapes at the other. We have also highlighted how responsible management strategies at local and landscape scales can greatly enhance opportunities for biodiversity conservation in these systems. Throughout we have drawn attention to some of the real world economic and social considerations that will determine the success of any attempts to implement improved conservation strategies in the real world.

To truly understand the prospects for conservation in modified landscapes, we need to increase our emphasis on the study of biodiversity in managed land-use systems (Chazdon *et al.* 2009a). Key knowledge gaps remain in our understanding of the long-term viability of native species in different land-uses (Sodhi 2008), and how patterns of species persistence are influenced by differences in the composition and configuration of entire landscapes. Increasingly severe levels of environmental degradation in modified landscapes across the world means that the costs and benefits of ecological restoration are deserving of particular research attention. There is also an urgent need for an improved understanding of the interaction between people and their local environment in human-modified systems, including the importance of ecosystem services (see Chapter 3) and opportunities for generating livelihood benefits from conservation activities.

If it is to be successful, the conservation research agenda in modified landscapes needs to be effective at incorporating new tools and approaches, both conceptual and analytical, that have the potential to bridge the divide between theory and practice and translate policies into effective field implementation (Chazdon *et al.* 2009a; Gardner *et al.* 2009). Key to achieving success and developing sustainable management strategies is the ability to build participatory and multidisciplinary approaches to research and management that involve not only conservation biologists, but also agroecologists, agronomists, farmers, indigenous peoples, rural social movements, foresters, social scientists, and land managers (see Chapter 14).

Summary

- Given that approximately one quarter of the world's threatened species live outside protected areas, and that the integrity of protected areas where they exist is often threatened, we need to integrate conservation efforts with other human activities.
- Recent studies demonstrate there are important opportunities for conserving biodiversity within the dominant types of human land-use, including logged forests, agroforestry systems, monoculture plantations, agricultural lands, urban areas, and regenerating land.
- It is the local people that ultimately decide the fate of their local environments, even if the decisions they make fall within a wider political, social, and economic context.
- Key to achieving success and developing sustainable management strategies is the ability to build participatory and multidisciplinary approaches to research and management that involve not only conservation biologists, but also agroecologists, agronomists, farmers, indigenous peoples, rural social movements, foresters, social scientists, and land managers.

Suggested reading

Barlow, J., Gardner, T. A., Araujo, I. S., *et al.* (2007). Quantifying the biodiversity value of tropical primary,

secondary and plantation forests. *Proceedings of the National Academy of Sciences of the United States of America*, **104**, 18555–18560.

Dunn, R. R. (2004). Recovery of faunal communities during tropical forest regeneration. *Conservation Biology*, **18**, 302–309.

Gardner, T., Barlow, J., Chazdon, R., *et al.* (2009). Prospects for tropical forest biodiversity in a human-modified world. *Ecology Letters*, **12**, 561–582.

Koh, L. P. (2008). Can oil palm plantations be made more hospitable for forest butterflies and birds? *Journal of Applied Ecology*, **45**, 1002–1009.

Relevant website

- Mongabay: <http://www.mongabay.com>

REFERENCES

- Anand, M., Krishnaswamy, J., and Das, A. (2008). Proximity to forests drives bird conservation value of coffee plantations: implications for certification. *Ecological Applications*, **18**, 1754–1763.
- Asner, G. P., Knapp, D. E., Broadbent, E. N., Oliveira, P. J. C., Keller, M., and Silva, J. N. (2005). Selective logging in the Brazilian Amazon. *Science*, **310**, 480–482.
- Balmford, A. and Bond, W. (2005). Trends in the state of nature and their implications for human well-being. *Ecology Letters*, **8**, 1218–1234.
- Balmford, A., Moore, J. L., Brooks, T. *et al.* (2001). Conservation conflicts across Africa. *Science*, **291**, 2616–2619.
- Balmford, A., Green, R. E., and Scharlemann, J. P. W. (2005). Sparing land for nature: exploring the potential impact of changes in agricultural yield on the area needed for crop production. *Global Change Biology*, **11**, 1594–1605.
- Barlow, J., Gardner, T. A., Araujo, I. S., *et al.* (2007). Quantifying the biodiversity value of tropical primary, secondary and plantation forests. *Proceedings of the National Academy of Sciences of the United States of America*, **104**, 18555–18560.
- Bawa, K. S., Kress, W. J., Nadkarni, N. M., *et al.* (2004). Tropical ecosystems into the 21st century. *Science*, **306**, 227–228.
- Bhagwat, S. A., Willis, K. J., Birks, H. J. B., and Whittaker, R. J. (2008). Agroforestry: a refuge for tropical biodiversity? *Trends in Ecology and Evolution*, **23**, 261–267.
- Bowen, M. E., McAlpine, C. A., House, A. P. N., and Smith, G. C. (2007). Regrowth forests on abandoned agricultural land: A review of their habitat values for recovering forest fauna. *Biological Conservation*, **140**, 273–296.
- Brockerhoff, E. G., Berndt, L. A., and Jactel, H. (2005). Role of exotic pine forests in the conservation of the critically endangered New Zealand ground beetle *Holcaspis brevicula* (Coleoptera: Carabidae). *New Zealand Journal of Ecology*, **29**, 37–43.
- Brockerhoff, E. G., Jactel, H., Parrotta, J. A., Quine, C. P., and Sayer, J. (2008). Plantation forests and biodiversity: oxymoron or opportunity? *Biodiversity and Conservation*, **17**, 925–951.
- Brook, B. W., Bradshaw, C. J. A., Koh, L. P., and Sodhi, N. S. (2006). Momentum drives the crash: mass extinction in the tropics. *Biotropica*, **38**, 302–305.
- Carnus, J. M., Parrotta, J., Brockerhoff, E., *et al.* (2006). Planted forests and biodiversity. *Journal of Forestry*, **104**, 65–77.
- Chazdon, R. L. (2003). Tropical forest recovery: legacies of human impact and natural disturbances. *Perspectives in Plant Ecology Evolution and Systematics*, **6**, 51–71.
- Chazdon, R. L. (2008). Beyond Deforestation: Restoring Forests and Ecosystem Services on Degraded Lands. *Science*, **320**, 1458–1460.
- Chazdon, R. L., Harvey, C. A., Oliver, K., *et al.* (2009a). Beyond Reserves: A Research Agenda for Conserving Biodiversity in Human-modified Tropical Landscapes. *Biotropica*, **41**, 142–153.
- Chazdon, R. L., Peres, C. A., Dent, A. (2009b). Where are the wild things? Assessing the potential for species conservation in tropical secondary forests *Conservation Biology* (in Press).
- Cohen, J. E. (2003). Human population: the next half century. *Science*, **302**, 1172–1175.
- Cyranoski, D. (2007). Logging: the new conservation. *Nature*, **446**, 608–610.
- Daily, G. C. (2001). Ecological forecasts. *Nature*, **411**, 245–245.
- Daily, G. C. (2003). Time to rethink conservation strategy. *Science*, **300**, 1508–1509.
- Daily, G. C., Ehrlich, P. R., and Sanchez-Azofeifa, A. (2001). Countryside biogeography: Utilization of human-dominated habitats by the avifauna of southern Costa Rica. *Ecological Applications*, **11**, 1–13.
- Dale, V. H., Brown, S., Haeuber, R. A., *et al.* (2000). Ecological principles and guidelines for managing the use of land. *Ecological Applications*, **10**, 639–670.
- DeWalt, S. J., Maliakal, S. K., and Denslow, J. S. (2003). Changes in vegetation structure and composition along a tropical forest chronosequence: implications for wildlife. *Forest Ecology and Management*, **182**, 139–151.
- Donald, P. F. (2004). Biodiversity impacts of some agricultural commodity production systems. *Conservation Biology*, **18**, 17–37.

- Dunn, R. R. (2004). Recovery of faunal communities during tropical forest regeneration. *Conservation Biology*, **18**, 302–309.
- Ehrlich, P. R. and Ehrlich, A. H. (2008). *The Dominant Animal: Human Evolution and the Environment*. Island Press, Washington, DC.
- Ellis, E. C. and Ramankutty, N. (2008). Putting people in the map: anthropogenic biomes of the world. *Frontiers in Ecology and the Environment*, **6**, 439–447.
- FAO (Food and Agriculture Organization of the United Nations) (2006). *Global forest resources assessment 2005: progress towards sustainable forest management*. FAO, Rome, Italy.
- Fazey, I., Fischer, J., and Lindenmayer, D. B., (2005). What do conservation biologists publish? *Biological Conservation*, **124**, 63–73.
- Felton, A. M., Engstrom, L. M., Felton, A., and Knott, C. D. (2003). Orangutan population density, forest structure and fruit availability in hand-logged and unlogged peat swamp forests in West Kalimantan, Indonesia. *Biological Conservation*, **114**, 91–101.
- Fimbel, R. A., Grajal, A., and Robinson, J. G., eds (2001). *The cutting edge. Conserving wildlife in logged tropical forests*. Colombia University, New York, NY.
- Fischer, J. and Lindenmayer, B. D. 2007. Landscape modification and habitat fragmentation: a synthesis. *Global Ecology and Biogeography*, **16**, 265–280.
- Gardner, T. A., Barlow, J., Parry, L. T. W., and Peres, C. A. (2007). Predicting the uncertain future of tropical forest species in a data vacuum. *Biotropica*, **39**, 25–30.
- Gardner, T. A., Barlow, J., Chazdon, R. L., et al. (2009). Prospects for tropical forest biodiversity in a human-modified world. *Ecology Letters*, **12**, 561–582.
- Ghazoul, J. (2007). Placing humans at the heart of conservation. *Biotropica*, **39**, 565–566.
- Gillies, C. S. and St Clair, C. C. (2008). Riparian corridors enhance movement of a forest specialist bird in fragmented tropical forest. *Proceedings of the National Academy of Sciences of the United States of America*, **105**, 19774–19779.
- Green, R. E., Cornell, S. J., Scharlemann, J. P. W., and Balmford, A. (2005). Farming and the fate of wild nature. *Science*, **307**, 550–555.
- Hanski, I. (1999). *Metapopulation ecology*. Oxford University Press, Oxford, UK.
- Hartley, M. J. (2002). Rationale and methods for conserving biodiversity in plantation forests. *Forest Ecology and Management*, **155**, 81–95.
- Heckenberger, M. J., Kuikuro, A., Kuikuro, U. T., et al. (2003). Amazonia 1492: Pristine forest or cultural parkland? *Science*, **301**, 1710–1714.
- ITTO (International Tropical Timber Association) (2002). *ITTO guidelines for the restoration, management and rehabilitation of degraded and secondary tropical forests*. ITTO Policy Development Series No. 13, International Tropical Timber Organization.
- ITTO (International Tropical Timber Association) (2007). *Annual Review and Assessment of the World Timber Situation 2007*. ITTO, Yokohama, Japan.
- Kaimowitz, D. and Sheil, D. (2007). Conserving what and for whom? Why conservation should help meet basic human needs in the tropics. *Biotropica*, **39**, 567–574.
- Kanowski, J., Catterall, C. P., and Wardell-Johnson, G. W. (2005). Consequences of broadscale timber plantations for biodiversity in cleared rainforest landscapes of tropical and subtropical Australia. *Forest Ecology and Management*, **208**, 359–372.
- Kareiva, P., Watts, S., McDonald, R., and Boucher, T. (2007). Domesticated nature: shaping landscapes and ecosystems for human welfare. *Science*, **316**, 1866–1869.
- Koh, L. P. (2008a). Birds defend oil palms from herbivorous insects. *Ecological Applications*, **18**, 821–825.
- Koh, L. P. (2008b). Can oil palm plantations be made more hospitable for forest butterflies and birds? *Journal of Applied Ecology*, **45**, 1002–1009.
- Koh, L. P., and Sodhi, N. S. (2004). Importance of reserves, fragments and parks for butterfly conservation in a tropical urban landscape. *Ecological Applications*, **14**, 1695–1708.
- Koh, L. P. and Wilcove, D. S. (2007). Cashing in palm oil for conservation. *Nature*, **448**, 993–994.
- Koh, L. P. and Wilcove, D. S. (2008). Is oil palm agriculture really destroying tropical biodiversity? *Conservation Letters*, **1**, 60–64.
- Koh, L. P. and Wilcove, D. S. (2009). Oil palm: disinformation enables deforestation. *Trends in Ecology and Evolution*, **24**, 67–68.
- Lamb, D., Erskine, P. D., and Parrotta, J. A. (2005). Restoration of degraded tropical forest landscapes. *Science*, **310**, 1628–1632.
- Laurance, W. F., Lovejoy, T. E., Vasconcelos, H. L., et al. (2002). Ecosystem decay of Amazonian forest fragments: a 22-year investigation. *Conservation Biology*, **16**, 605–618.
- Lewis, O. T. (2001). Effect of experimental selective logging on tropical butterflies. *Conservation Biology*, **15**, 389–400.
- Liebsch, D., Marques, M. C. M., and Goldenberg, R. (2008). How long does the Atlantic Rain Forest take to recover after a disturbance? Changes in species composition and ecological features during secondary succession. *Biological Conservation*, **141**, 1717–1725.
- Lindenmayer, D. B. and Franklin, J. F. (2002). *Conserving biodiversity: a comprehensive multiscaled approach*. Island Press, Washington, DC.
- Lindenmayer, D. B. and Hobbs, R. J. (2004). Fauna conservation in Australian plantation forests – a review. *Biological Conservation*, **119**, 151–168.

- Lindenmayer, D. B., McIntyre, S., and Fischer, J. (2003). Birds in eucalypt and pine forests: landscape alteration and its implications for research models of faunal habitat use. *Biological Conservation*, **110**, 45–53.
- Lindenmayer, D. B., Franklin, J. F., and Fischer, J. (2006). General management principles and a checklist of strategies to guide forest biodiversity conservation. *Biological Conservation*, **131**, 433–445.
- MacArthur, R. H. and Wilson, E. O. (1963). An equilibrium theory of insular zoogeography. *Evolution*, **17**, 373–387.
- MacArthur, R. H. and Wilson, E. O. (1967). *The theory of island biogeography*. Princeton University Press, Princeton, New Jersey.
- Maginnis, S. and Jackson, W. (2005). *Balancing restoration and development*. ITTO Tropical Forest Update, International Tropical Timber Organization. 15/2, 4–6.
- Matson, P. A. and Vitousek, P. M. (2006). Agricultural intensification: will land spared from farming be land spared for nature? *Conservation Biology*, **20**, 709–710.
- Mbida, C. M., Van Neer, W., Doutrelepont, H., and Vrydaghs, L. (2000). Evidence for banana cultivation and animal husbandry during the First Millennium BC in the forest of Southern Cameroon. *Journal of Archaeological Science*, **27**, 151–162.
- McNeely, J. A. (2004). Nature vs. nurture: managing relationships between forests, agroforestry and wild biodiversity. *Agroforestry Systems*, **61**, 155–165.
- Meijaard, E. and Sheil, D. (2008). The persistence and conservation of Borneo's mammals in lowland rain forests managed for timber: observations, overviews and opportunities. *Ecological Restoration*, **23**, 21–34.
- Miller, J. R. and Hobbs, R. J. (2002). Conservation where people live and work. *Conservation Biology*, **16**, 330–337.
- Ostrom, E. (2007). A diagnostic approach for going beyond panaceas. *Proceedings of the National Academy of Sciences of the United States of America*, **104**, 15181–15187.
- Palmer, M., Bernhardt, E., Chornesky, E., et al. (2004). Ecology for a crowded planet. *Science*, **304**, 1251–1252.
- Pedlowski, M. A., Matricardi, E. A. T., Skole, D., et al. (2005). Conservation units: a new deforestation frontier in the Amazonian state of Rondonia, Brazil. *Environmental Conservation*, **32**, 149–155.
- Perfecto, I. and Vandermeer, J. (2002). The quality of agroecological matrix in a tropical montane landscape: ants in coffee plantations in southern Mexico. *Conservation Biology*, **16**, 174–182.
- Perfecto, I. and Vandermeer, J. (2008). Biodiversity conservation in tropical agroecosystems. *Annals of the New York Academy of Science*, **1134**, 173–200.
- Peters, S. L., Malcolm, J. R., and Zimmerman, B. L. (2006). Effects of selective logging on bat communities in the southeastern Amazon. *Conservation Biology*, **20**, 1410–1421.
- Philpott, S. M., Arendt, W. J., Armbrecht, I., et al. (2008). Biodiversity loss in Latin American coffee landscapes: review of the evidence on ants, birds, and trees. *Conservation Biology*, **22**, 1093–1105.
- Putz, F. E., Blate, G. M., Redford, K. H. Fimbel, R., and Robinson, J. (2001). Tropical forest management and conservation of biodiversity: an overview. *Conservation Biology*, **15**, 7–20.
- Putz, F. E., Sisk, P., Fredericksen, T., and Dykstra, D. (2008). Reduced-impact logging: Challenges and opportunities. *Forest Ecology and Management*, **256**, 1427–1433.
- Ranganathan, J., Daniels, R. J. R., Chandran, M. D. S., Ehrlich, P., and Daily, G. C. (2008). Sustaining biodiversity in ancient tropical countryside. *Proceedings of the National Academy of Sciences of the United States of America*, **105**, 17852–17854.
- Ribeiro, M. C., Metzger, J. P., Martensen, A. C., Ponzoni, F., and Hirota, M. M. (2009). Brazilian Atlantic forest: how much is left and how is the remaining forest distributed? Implications for conservation. *Biological Conservation*, **142**, 1141–1153.
- Rodrigues, A. S. L., Andelman, S. J., Bakarr, M. I., et al. (2004). Effectiveness of the global protected area network in representing species diversity. *Nature*, **428**, 640–643.
- Rosenzweig, M. L. (2003). *Win-win ecology: how the earth's species can survive in the midst of human enterprise*. Oxford University Press, New York.
- Sayer, J. and Maginnis, S., eds (2005). *Forests in landscapes: ecosystem approaches to sustainability*. Earthscan, London.
- Scales, B. R. and Marsden, S. J. (2008). Biodiversity in small-scale tropical agroforests: a review of species richness and abundance shifts and the factors influencing them. *Environmental Conservation*, **35**, 160–172.
- Scherr, S. J. and McNeely, J. A., eds (2007). *Farming with nature: the science and practice of ecoagriculture*. Island Press, Washington, DC.
- Scherr, S. J. and McNeely, J. A. (2008). Biodiversity conservation and agricultural sustainability: towards a new paradigm of 'ecoagriculture' landscapes. *Philosophical Transactions of the Royal Society of London B*, **363**, 477–494.
- Schmitt, C. B., Belokurov, A., Besançon, C., et al. (2009). *Global Ecological Forest Classification and Forest Protected Area Gap Analysis. Analyses and recommendations in view of the 10% target for forest protection under the Convention on Biological Diversity (CBD)*. 2nd revised edn Freiburg University Press, Freiburg, Germany.
- Schroth, G. and Harvey, C. A. (2007). Biodiversity conservation in cocoa production landscapes: an overview. *Biodiversity and Conservation*, **16**, 2237–2244.
- Schroth, G., da Fonseca, G. A. B., Harvey, C. A., Gascon, C., Vasconcelos, and Izac, A.-M. N., eds (2004).

- Agroforestry and biodiversity conservation in tropical landscapes*. Island Press, Washington, DC.
- Sekercioglu, C. H. (2002). Effects of forestry practices on vegetation structure and bird community of Kibale National Park, Uganda. *Biological Conservation*, **107**, 229–240.
- Sekercioglu, C. H., Loarie, S. R., Ruiz-Gutierrez, V., Oviedo Brenes, F., Daily, G. C., and Ehrlich, P. R. (2007). Persistence of forest birds in tropical countryside. *Conservation Biology* **21**, 482–494.
- Simberloff, D. S. (1974). Equilibrium theory of island biogeography and ecology. *Annual Review of Ecology and Systematics*, **5**, 161–182.
- Sodhi, N. S. (2008). Tropical biodiversity loss and people – a brief review. *Basic and Applied Ecology*, **9**, 93–99.
- Sodhi, N. S., Koh, L. P., Brook, B. W., and Ng, P. K. L. (2004). Southeast Asian biodiversity: an impending disaster. *Trends in Ecology and Evolution*, **19**, 654–660.
- Sodhi, N. S., Brooks, T. M., Koh, L. P. Koh *et al.* (2006). Biodiversity and human livelihood crises in the Malay Archipelago. *Conservation Biology*, **20**, 1811–1813.
- Sodhi, N. S., Acciaioli, G., Erb, M., and Tan, A. K.-J., eds (2008). *Biodiversity and human livelihoods in protected areas: case studies from the Malay Archipelago*. Cambridge University Press, Cambridge, UK.
- Sodhi, N. S., Lee, T. M., Koh, L. P., and Brook, B. W. (2009). A meta-analysis of the impact of anthropogenic forest disturbance on Southeast Asia's biotas. *Biotropica*, **41**, 103–109.
- Steffan-Dewenter, I., Kessler, M., Barkmann, J., *et al.* (2007). Tradeoffs between income, biodiversity, and ecosystem functioning during tropical rainforest conversion and agroforestry intensification. *Proceedings of the National Academy of Sciences of the United States of America*, **104**, 4973–4978.
- Stein, B.A., Kutner, L. S., and Adams, J. S. (2000). *Precious heritage*. Oxford University Press, Oxford, UK.
- Stratford, J. A. and Stouffer, P. C. (1999). Local extinctions of terrestrial insectivorous birds in a fragmented landscape near Manaus, Brazil. *Conservation Biology*, **13**, 1416–1423.
- Thiollay, J. M. (1995). The role of traditional agroforests in the conservation of rain-forest bird diversity in Sumatra. *Conservation Biology*, **9**, 335–353.
- Tilman, D., Fargione, J., and Wolff, B., *et al.* (2001). Forecasting agriculturally driven global environmental change. *Science*, **292**, 281–284.
- UN (United Nations) (2005). *Halving hunger: it can be done*. Earthscan, London, UK.
- Vandermeer, J. and Carvajal, R. (2001). Metapopulation dynamics and the quality of the matrix. *The American Naturalist*, **158**, 211–220.
- Vandermeer, J. and Perfecto, I. (2007). The agricultural matrix and a future paradigm for conservation. *Conservation Biology*, **21**, 274–277.
- Vaughan, C., Ramirez, O., Herrera, G., and Guries, R. (2007). Spatial ecology and conservation of two sloth species in a cacao landscape in limon, Costa Rica. *Biodiversity and Conservation*, **16**, 2293–2310.
- White, L. J. T. (2001). The African rain forest: climate and vegetation. In W. Weber, L. J. T. White, A. Vedder, and L. Naughton-Treves, eds *African rain forest ecology and conservation: an interdisciplinary perspective*, pp. 3–29. Yale University Press, New Haven, CT.
- Willis, K. J. and Birks, H. J. B. (2006). What is natural? The need for a long-term perspective in biodiversity conservation. *Science*, **314**, 1261–1265.
- Willis, K. J., Gillson, L., and Brncic, T. M. (2004). How “virgin” is virgin rainforest? *Science*, **304**, 402–403.
- Willis, K. J., Gillson, L., Brncic, T. M., and Figueroa-Rangel, B. L. (2005). Providing baselines for biodiversity measurement. *Trends in Ecology and Evolution*, **20**, 107–108.
- Wittemyer, G., Elsen, P., Bean, W. T., Burton, C. O., and Brashares, J. S. (2008). Accelerated human population growth at protected area edges. *Science*, **321**, 123–126.
- Wright, S. J. and Muller-Landau, H. C. (2006). The future of tropical forest species. *Biotropica*, **38**, 207–301.