

Chapter 10

Biodiversity across Scenarios

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Main Messages

This chapter discusses the consequences of the four scenarios developed by the Millennium Ecosystem Assessment for biodiversity and focuses on two different aspects of biodiversity—losses of local populations and global species extinctions. In this assessment, local extinctions occur from a reduction in habitat availability. On a longer time scale, global extinctions may occur when species reach equilibrium with the altered habitat. Global and local extinctions occur on a time scale that we cannot accurately anticipate.

Habitat loss in terrestrial environments will lead, with high certainty, to a sharp decline in local diversity and the ecosystem services it provides (very certain) in all four MA scenarios during the 2000–50 time period. Scenario analysis demonstrated a decline of habitat availability by 2050 that ranged from 20% in Order from Strength to 13% in TechnoGarden (*medium certainty*) relative to habitat availability in the year 1970.

Habitat loss in terrestrial environments will lead (with high certainty) in all four MA scenarios to global species extinctions and associated losses of ecosystem services (such as the development of new drugs). Analyses using the well-established species-area relationship and a state-of-the-art model of land use change indicate that 12–16% (*low certainty*) of species will potentially be lost at ecological equilibrium with the altered habitat. Also, significant loss of ecosystem services will occur long before a species becomes globally extinct.

Order from Strength is the scenario (with high certainty) with the largest losses of habitat and local plant populations, whereas TechnoGarden and Adapting Mosaic had the smallest losses. The Order from Strength scenario showed the highest expansion of cropland resulting from slow rates of yield improvement and higher population growth. The Adapting Mosaic scenario showed a relatively low rate of habitat losses in part because of the slow development rate in developing countries, which reduced the demand for food and the change in land use. Scenarios that showed the largest losses of habitat also put the largest number of species in trajectories that may lead to global extinctions (*medium certainty*), although the time lags from habitat reduction to extinction are unknown.

Scenario results showed that the different terrestrial biomes of Earth will lose habitat and local plant species populations at different rates (high certainty) during the 2000–50 period. The biomes with the higher rates of habitat and local species diversity losses are warm mixed forests, savannas, scrub, tropical forests, and tropical woodlands. Biomes that lose species at the lowest rate include those with low human impact as well as those where land use changes and human intervention have already occurred.

It is unlikely that the Convention on Biological Diversity target for reducing the rate of biodiversity loss by 2010 will be met for terrestrial ecosystems under the explored scenarios. The two scenarios that take a more proactive approach to the environment (TechnoGarden and Adapting Mosaic) have more success in reducing loss rates of terrestrial biodiversity than the two that take a reactive approach (Global Orchestration and Order from Strength) (*medium certainty*).

For the three drivers tested globally across scenarios, land use change was the dominant driver of biodiversity change in terrestrial ecosystems, followed by changes in climate and nitrogen deposition (medium certainty). Some individual biomes showed different patterns. For example, climate change was the dominant driver of biodiversity change in tundra, boreal and cool conifer forest, savanna, and deserts. Nitrogen deposition was found to be a particularly important driver in warm mixed forests and temperate

deciduous forest (ecosystems that are sensitive to deposition and relatively close to densely populated areas). In addition, the impact of other drivers, such as invasive species, could not be assessed as fully and may therefore be underestimated.

Under all scenarios, 70% of the world's rivers, especially those at higher latitudes, will increase in water availability, raising the potential for production of fishes adapted to higher flow habitats, which would likely be nonindigenous species (low certainty). No quantitative models exist that allow estimation of any additional consequences of increased discharge on biodiversity.

Under all scenarios, 30% of the modeled river basins will decrease in water availability from the combined effects of climate change and water withdrawal. Based on established but incomplete scientific understanding, this will result in eventual losses (at equilibrium) of 1–55% (by 2050; 1–65% by 2100) of fish species from these basins (*low certainty*). Climate change rather than water withdrawal was the major driver for the species losses from most (~80%) basins, with projected losses from climate change alone of about 1–30% (by 2050; 1–65% by 2100). Differences among scenarios were minor relative to the average magnitude of projected losses of freshwater biodiversity.

Losses of biodiversity of fishes predicted only on the basis of drying are underestimates. Drivers other than loss of water availability will cause additional losses that are likely to be greater than losses from declining water. Many of the rivers and lakes in drying regions will also experience increased temperatures, eutrophication, acidification, and increased invasions by nonindigenous species. These factors all increased losses of native biodiversity in rivers and lakes that are drying and caused losses of fishes and other freshwater taxa in other rivers and lakes. No algorithms exist for estimating the numbers of riverine and lake species lost from these drivers, but recent experience suggests that they cause losses greater than those caused by climate change and water withdrawal.

Rivers that are forecast to lose fish species are concentrated in poor tropical and sub-tropical countries, where the needs for human adaptation are most likely to exceed governmental and societal capacity to cope. The current average GDP in drying countries is about 20% lower than that in countries whose rivers are not drying.

Diversity of marine biomass was quite sensitive to changes in regional policy. Scenarios with policies that focused on maintaining or increasing the value of fisheries resulted in declining biomass diversity, while the scenarios with policy that focused on maintaining the ecosystem responded with increasing biomass diversity. However, rebuilding selected stocks did not necessarily increase biomass diversity as effectively as an ecosystem-focused policy.

Diversity of commercial fisheries showed large differences among scenarios until 2030, but all scenarios converge into a common value by 2050. Policy changes after 2030 generally included increasing the value of the fisheries by lowering costs, focusing on high-value species, substituting technology for ecosystem services, or a combination of the three approaches. However, no approach was optimal, since the approaches used in the scenarios reduced biomass diversity to a common level in each ecosystem.

As global trade increases, the numbers of intentional and unintentional introductions will increase in terrestrial, freshwater, and marine biomes. Unless greater management steps are taken to prevent harmful introductions that accompany increased trade, invasive species will cause increased ecologi-

cal changes and losses of ecosystem services in all scenarios. Because of differences among scenarios in economic growth and openness to foreign trade, invasive species increase most in Global Orchestration, followed in order by TechnoGarden, Adapting Mosaic, and Order from Strength.

Lag times in species extinctions provide a window of opportunity for humans to deploy aggressive restoration practices that may rescue species that otherwise may be lost. Many actions that can be taken by policymakers (such as habitat restoration and establishment of protected areas) may change the fate of a species that would otherwise become extinct in a few generations.

Ecosystem services provided mostly by species in the upper trophic levels, such as biological control, tend to be lost first with increasing habitat loss (low certainty). Ecosystem services provided by species in the lower trophic levels, such as provisioning of food, fiber, and clean water, tend to be lost only after severe habitat loss has occurred. The relationship between habitat loss, biodiversity loss, and the provisioning ecosystem services depends on the notion that all species in an ecosystem do not have the same probability of extinction and all the ecosystem services are not provided by the same type of species. Increasing habitat loss leads first to the loss of species in the higher trophic levels (top predators), while only extreme losses of habitat result in the extinction of species in the lower trophic levels (plants and microorganisms).

10.1 Introduction

The four scenarios developed by the Millennium Ecosystem Assessment explore a broad set of possible socioeconomic trajectories for human society. Each scenario will have different consequences for biodiversity and the provisioning of ecosystem services. In this chapter, we consider the future of biodiversity under each scenario.

Since biodiversity forms the basis for ecosystem services, the current decline of global biodiversity is of great concern. Despite the ongoing conservation efforts of the international community, biodiversity loss continues to occur at an unprecedented rate of up to 100–10,000 times the background rate in the fossil record of the Cenozoic (Reid 1992; Barbault and Sastrapradja 1995; May et al. 1995; Pimm et al. 1995; Foote and Raup 1996). Changes in land use are expected to be the major driver of biodiversity change in this century, followed in importance by changes in climate, nitrogen deposition, biotic exchange (accidental or deliberate introduction of a species into an ecosystem), and atmospheric CO₂ levels (Sala et al. 2000). Depending on the assumptions that are made, the precise ranking of some drivers may vary; for instance, Thomas et al. (2004) have suggested that climate change may be as important as land use change in driving biodiversity loss over the next 50 years. In any case, it is clear that all these drivers will have major impacts on biodiversity.

Biodiversity is a composite measure of the number of species (species richness) and the number of individuals of different species (relative abundance). Most ecosystem services, such as the provisioning of food or clean water, depend on the presence of sufficient numbers of individuals of each species. These services will decline locally with the local extirpation or reduction of populations, long before global extinctions take place. For other ecosystem services, and in particular those that rely on genetic diversity, the

central issue is species richness. For example, the provisioning of new pharmaceutical drugs to cure current and future diseases and the maintenance of genetic resources to improve current crop varieties are not directly related to the abundance of individuals within a species. In these instances, the provision of services only ceases after global extinction.

For terrestrial ecosystems, we considered changes in both local and global biodiversity. Local losses of biodiversity are important because they may anticipate global losses and because they affect local people who benefited from the services provided by the species that became extinct. In addition, local extinctions affect the global provisioning of ecosystem services that depend on the abundance of individuals, as noted. Global biodiversity changes are important because they are irreversible; species that go extinct globally will never reappear. Losses of global biodiversity affect the provisioning of both types of ecosystem services—those that depend on abundance and those that depend on the maintenance of unique genetic combinations.

The freshwater biodiversity exercise focused on local extinctions because freshwater communities are organized around watersheds, which means that extinctions are watershed-specific. The marine biodiversity assessment focused on the diversity of commercial fish species, both because these species are directly relevant to humans and because more comprehensive data sets were unavailable.

In all three cases, the assessment focused on species diversity because of the availability of published information. We note that diversity within species (genetic diversity) could be equally affected by human activity, with potentially large consequences for the provisioning of ecosystem services.

We used different approaches to assess changes in biodiversity in terrestrial, freshwater, and marine environments because the drivers of biodiversity change and our level of scientific understanding, as reflected in the available models, are different in the three different environments. We used the species-area relationship to assess the global impact of land use change on terrestrial ecosystems. The species-area relationship has been documented in more than 150 articles for many taxa and many systems, ranging from oceanic islands to isolated habitat patches in terrestrial landscapes. We estimated the area of habitat lost (and the ensuing local loss of species diversity) as a function of local changes in land use, as assessed by the IMAGE model. (See Chapters 6 and 9.)

The temporal aspect of the species-area relationship must be considered carefully, because extinctions do not occur immediately after a reduction in the area of available habitat (Tilman et al. 1994; Magsalay et al. 1995; Brooks and Balmford 1996). The scenarios for future biodiversity based on the species-area relationship in this chapter refer to the number of species that would be expected to go extinct when populations relax to an equilibrium in a reduced area of habitat. In our models, habitat reductions result from either land use or climate change. Determining the relaxation times for entire communities is particularly difficult because it requires tracking the species composition of habi-

remnants through time. Furthermore, the time lag will depend on the life history of the species concerned; relaxation to an equilibrium may occur faster in species with shorter generation times (Brook et al. 2003).

Recent studies have placed some bounds on relaxation times. Brooks et al. (1999) fitted exponential decay curves to estimated bird species losses in forest fragments in Kenya (100–10,000 hectares) and found half-lives (the time to lose 50% of species predicted to go extinct at equilibrium) to be in a range of 23–80 years. Ferraz et al. (2003) used bird occurrence data taken during 14 years in Amazon forest fragments to show that half-lives were shorter for smaller fragments, with fragments in the size range 10–100 hectares having half-lives of about a decade. Finally, Leach and Givnish (1996) studied Wisconsin prairie remnants (0.2–6 hectares) and found that 8–60% of the original plant species had gone locally extinct over 32–52 years.

Overall, these results suggest that about half of the species losses predicted in this chapter may occur over a period of decades to 100 years. Our assessment yields estimates of the number of vascular plant species that are expected to go extinct when populations reach equilibrium with the reduced habitat. From a policy perspective, time lags between habitat reduction and species extinction provide a precious opportunity for humans to deploy aggressive restoration practices to rescue those species that would otherwise become extinct, although habitat restoration measures will not save the most sensitive species that go extinct soon after habitat loss. The time lags between habitat reduction and extinction can also mask serious problems; for example, long-lived tree species that have lost their pollinators may linger for hundreds of years before extinction finally occurs.

In addition to land use change, we considered climate change and nitrogen deposition as major influences on terrestrial biodiversity. Several other factors, including elevated CO₂, species invasions, and patterns of habitat fragmentation, are potentially important but were not included in this assessment because of a lack of appropriate data and models at the global scale. We used three complementary, published approaches to explore the effects of climate change on terrestrial biodiversity. These included analysis of changes in the locations of the boundaries of entire biomes, in potential biodiversity as a response to climate, and in tick diversity in Africa based on the summed predictions of models for individual species. We assessed the potential impacts of nitrogen deposition by estimating nitrogen loads in different regions and applying the concept of a critical load below which no damage occurs.

Freshwater ecosystems are among the most threatened on Earth. Consequently, understanding the relationships between aquatic species diversity and environmental drivers is of critical importance. Compared with terrestrial ecosystems, however, the patterns and determinants of biodiversity in freshwater ecosystems are poorly known. Quantitative information on species richness patterns and responses to anthropogenic environmental changes is largely lacking for freshwater taxa. This lack of information is particularly acute at large spatial and temporal scales. Freshwater taxa occupy the first four places in the IUCN

list of the proportion of U.S. species at risk of extinction: freshwater mussels, crayfishes, amphibians, and freshwater fish. Globally, the best evidence suggests that freshwater biodiversity is more threatened than terrestrial taxa are by global changes (Ricciardi and Rasmussen 1999). This is partly because humans are drawn to riparian habitats, leading to a concentration of anthropogenic impacts near coastal and freshwater habitats. Furthermore, human consumption of water is reducing available habitat for freshwater organisms (see *MA Current State and Trends*, Chapter 8; Lodge 2001; Poff et al. 2001).

It is difficult to make quantitative predictions for how freshwater and marine biodiversity will be affected by future global changes. In addition to the rarity of freshwater biodiversity data (see *MA Current State and Trends*, Chapter 5), many of the statistical and conceptual tools available for use in conservation planning for terrestrial biodiversity are not readily transferable to analyses of freshwater biodiversity. Species-area curves, for example, cannot realistically be used to predict species loss in lakes. While dramatic examples of loss of lake habitat area exist (such as the drying of the Aral Sea from irrigation withdrawals), the biodiversity of lakes is in general more affected by a reduction in the quality than the quantity of water.

We examine four of the five most globally important, proximate drivers of biodiversity loss in lakes and rivers (Sala et al. 2000): eutrophication/land use change, acidification, climate change, and water withdrawal. The impact of climate change and water withdrawal on riverine fauna is addressed quantitatively using previously published species-river discharge relationships (Oberdorff et al. 1995). We focus on fishes and river discharge because those are the only previously published data and models that exist with suitably global coverage. (See *MA Current State and Trends*, Chapter 5.) The results for fish are of general importance because fish are an important controller of aquatic food webs and are often the taxa providing the most direct ecosystem goods to humans.

The biodiversity of marine systems is not as well described as that of terrestrial systems for a number of reasons. (See *MA Current State and Trends*, Chapter 18.) While there is a solid understanding of biodiversity changes in commercial fisheries, other areas such as the deep sea, the mid-water column, seamounts, and thermal vents are poorly described. We used a quantitative modeling approach to investigate how the diversity of fisheries and the biomass of different species might change under different MA scenarios in the three regions of the world for which we had good modeling tools.

This chapter describes the outcomes of the four MA scenarios for biodiversity in each of the three environments—terrestrial, freshwater, and marine. In the case of terrestrial ecosystems, we also assess the feasibility of achieving the Convention on Biodiversity target of significantly reducing the rate of biodiversity loss by the year 2010. Finally, we explore the uncertainties associated with our analysis and the regional differences in biodiversity changes across scenarios.

10.2 Terrestrial Biodiversity

10.2.1 The Approach

Our assessment of global changes in terrestrial biodiversity is based on the implementation of the MA scenarios in the IMAGE model, describing changes in native habitat cover, climate change, and nitrogen deposition over time. This section explains the calculation of native habitat cover using the IMAGE model, how the species-area relationship links changes in habitat area to global species extinctions, the different approaches used to estimate the effect of climate change, and how we estimated the effect of nitrogen deposition.

10.2.1.1 The IMAGE 2.2 Modeling Framework

The IMAGE 2.2 integrated assessment modeling framework consists of a set of linked and integrated models that together describe important elements of the cause-response chain of global environmental change. The framework and its submodels have been described in detail in several publications (Alcamo et al. 1998; IMAGE-team, 2001). Important elements include the description of emissions of greenhouse gases and regional air pollutants, climate change, and land use change. In the model, socioeconomic processes are mostly modeled at the level of 17 world regions, while climate, land use, and several environmental parameters are modeled at a 0.5x0.5 degree resolution.

The land cover model of IMAGE simulates the change in land use and land cover in each region driven by demands for food, forage, grass, timber, and biofuels and by changes in climate. It also includes a modified version of the BIOME model of Prentice et al. (1992) that is used to compute (changes in) potential vegetation. The potential vegetation is the equilibrium vegetation that should eventually develop under a given climate. The shifts in vegetation zones, however, do not occur instantaneously. In IMAGE 2.2, such dynamic adaptation is modeled explicitly according to the algorithms developed by Van Minnen and Ihle (2000).

10.2.1.2 The Species-Area Relationship Approach and Limitations

The relationship between species numbers and area is ubiquitous in nature (Rosenzweig 1995; Lomolino and Weiser 2001): the larger the area sampled, the larger the number of species found. The SAR is well described by the power law $S = cA^z$, where c is species local density and depends on the taxon and region being studied and z is the slope of the relationship and depends primarily on the type of SAR (oceanic islands, nested areas in a region, or different biological provinces). The value of z is also influenced by other factors, such as the scale of sampling (Crawley and Harral 2001). The strengths and weaknesses of using the SAR to forecast biodiversity loss are discussed in Chapter 4. Here, we briefly review those strengths and weaknesses and our theoretical understanding of the SAR.

Several factors contribute to the increase in species with area (Rosenzweig 1995). First, larger areas have a larger

number of habitats, and therefore they will contain more specialized species. Second, when comparing among isolated units such as islands, habitat fragments, or biogeographic provinces, larger units will have lower extinction rates and, to a lesser extent, higher immigration rates. Third, at a geological time scale, larger units will have higher speciation rates. Fourth, there is a sampling issue; larger areas have a larger number of individuals and a higher probability of including rare species than smaller units. The second type of explanation, extinction versus immigration, is the basis for the theory of biogeography (Mac Arthur and Wilson 1967), which explains the variation of species diversity among islands of an oceanic archipelago.

Decrease in area of a habitat will lead to biodiversity loss through all four mechanisms just mentioned, but the loss of specialized species and the increase in extinction rates will be the first impacts felt (Rosenzweig 2001). The precise shape of the relationship describing the loss of species from an original habitat as a function of the remaining habitat area after conversion to agriculture is still an open question. There are three associated issues. First, many species are not restricted to their native habitat and can live in the agricultural landscape (Pereira et al. 2004). Second, the slope of the species-area relationship used for the loss of total area of a biome is still uncertain. Third, it is not clear how the SAR can account for the effect of habitat fragmentation. Although there is high certainty of the overall shape of the species-area relationship, there is uncertainty in the z parameter determining the exact slope of the relationship. We established bounds for the slope of the species-area relationship (z -value) based on an extensive literature search. The distribution of the values reported in the literature for the slope of the species-area relationship was the basis for a statistical analysis that provided confidence intervals for some of the estimates.

It is also uncertain whether the biodiversity loss associated with the decrease in biome area caused by climate change is well described by the SAR approach. Thomas et al. (2004) used this approach to forecast the impact of climate change for animal and plant species in six regions of the world. Here, we applied the SAR approach to assess the effect on biodiversity of biome-area decreases caused both by land use change and by climate change.

As discussed in the introduction, it is important to note that the extinctions predicted by SAR do not occur instantaneously because there is a time lag between habitat loss and species extinctions (Brooks et al. 1999; Tilman et al. 2002). We do not know precisely how long this is, and it will vary according to the life history of individual species. A few studies have suggested that many species extinctions would occur during the first 100 years after habitat reduction (Wilson 1992). In this chapter we estimate, for each scenario, the number of species that would go extinct when populations reach equilibrium with a reduced habitat resulting from either land use or climate change.

The number of endemic species is another source of uncertainty associated with the use of SARs to calculate species losses based on changes in habitat area. Species extinctions due to reductions in area represent global ex-

tinctions only for species that are endemic and do not exist outside this area. The proportion of endemic species is related to the level of disaggregation of vegetation units. If the vegetation of the world were lumped in a small number of large units, these would be quite different from each other in environmental conditions and species composition, and they would have very few species in common. In this case, the abundance of endemic species in each unit would be large. But if the vegetation of Earth were partitioned in a large number of small units, these would be similar to each other in environmental space and they may contain larger numbers of species in common and fewer endemic species. A large degree of disaggregation, consequently, would result in an overestimation of species losses estimated with the species-area approach. Here, we minimized this error by using large vegetation units.

Our units of analysis for the terrestrial biodiversity loss are the intersection of the 17 IMAGE biomes and the six biogeographic realms of Olson et al. (2001). These units were chosen to ensure that units were not too large as to miss regional patterns but at the same time not too small as to have a low percentage of endemics in each unit.

10.2.1.3 Estimating Original Biodiversity of Vascular Plants

Ideally, we would like to have species counts of vascular plants for each realm-biome unit, but with the exception of North America (Kartesz and Meacham 1999) these data are not available at the regional scale. Therefore, we used an indirect estimate of the diversity of vascular plants by scaling up the species-area relationship from local diversity of vascular plants to the realm-biome units. Using the SAR to estimate regional species counts based on local data is a common practice (e.g., Groombridge and Jenkins 2002), but it has limitations (Crawley and Harral 2001). In order to perform this scaling, we did an assessment of the species local density and z-values to be used for each realm-biome unit.

Data on the local species density, the c value of the SAR (which represents the intrinsic diversity of each system for a unity size, in our case 10,000 square kilometers), were obtained for each set of realm-biome combination by comparing the 1995 IMAGE land cover map to a map of the local diversity of vascular plants (Barthlott et al. 1999). Because each realm-biome combination spanned a range of classes of local diversities of vascular plants (the c-values), mean values of the local diversity in each realm-biome were calculated using a Geographic Information System. While in general each realm-biome unit is only moderately heterogeneous in diversity; this averaging may underestimate the diversity of species in cases where both species-rich and species-poor areas are found within a unit.

The slope of the SAR, the z-value, depends on the type (Rosenzweig 1995) and scale (Crawley and Harral 2001) of the sampling, and it has been hypothesized to depend on other variables as well, such as latitude/biome and taxon (Preston 1962; Connor and McCoy 1979). We compiled 82 values of z reported in the literature for species-area relationship of vascular plants. For each study, we recorded the author and the original data source, the location, the biome,

the z-value, the c-value, the scale of the study (minimum and maximum area sampled), and the type of sampling. We considered three types of sampling units: continental, islands, and provinces. The continental SAR was obtained by sampling nested areas within a region. The island SAR was obtained by sampling the number of species of each island of an archipelago, where islands and archipelago included not only oceanic islands but also mountaintops. The provincial SARs were obtained from sampling the number of species in different biogeographic regions.

In our database, the most important variable in determining the z-value was the type of sampling. (See Figure 10.1, ANOVA $p < 0.001$, $r^2 = 0.52$.) Continental SARs produced the lowest z-values, intermediate z-values were obtained for islands SARs, and the highest z-values came from provincial SARs. After accounting for the effect of the SAR type, the minimum area and the maximum area sampled did not have significant effects on the z-value (ANCOVA, $p > 0.05$; see also Table 10.1).

To calculate the number of actual species in each biome/region, we needed to scale up from the unit area of 10,000 square kilometers of Barthlott's map (1999) to the total area of the biome-region. To compute future loss of biodiversity, we scaled down from the original biome area to the smaller biome area remaining after habitat conversion. It could be argued that the continental SAR is the most appropriate to scale up, whereas the island SAR is the most

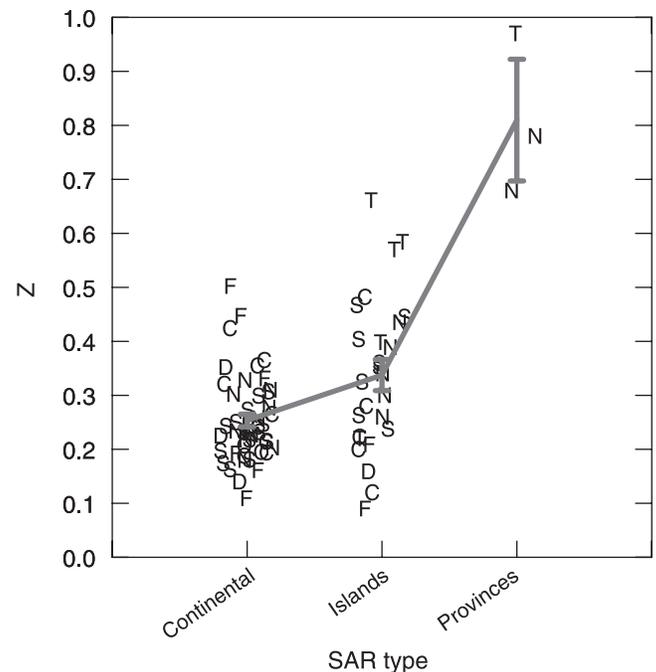


Figure 10.1. Z-Values Reported in Studies of the Species-Area Relationship in Vascular Plants. Each point corresponds to a study and is labeled with the biome category of the studied area: T-tropical forest and tropical woodland; F-temperate deciduous forest; C-boreal forest, coniferous forest, wooded tundra, and tundra; S-warm mixed forest, scrubland, and savanna; D-grassland and desert; N-no specific biome. The line joins the means in each type, and the error bars are standard errors.

Table 10.1. Mean and Standard Deviation of Z-Values and of the Natural Logarithm of Z-Values in Studies of Species-Area Relationships in Vascular Plants. Values for a dataset restricted to studies where the minimum area sampled was greater than 1 square kilometer are in parentheses.

Area	N	Mean	Standard Deviation	Natural Logarithm of z-Values	
		(μ_z)	(σ_z)	($\mu_{\ln(z)}$)	($\sigma_{\ln(z)}$)
Continental	47 (5)	0.253 (0.265)	0.079	-1.417	0.299
Islands	26 (7)	0.338 (0.308)	0.144	-1.186	0.480
Provinces	3	0.810	0.148	-0.222	0.180

appropriate to scale down. Several studies have used the island SAR to predict biodiversity loss (May et al. 1995; Pimm et al. 1995; Brooks et al. 1999), based on the argument that habitat conversion results in islands of native habitat in a sea of human-modified habitat. However, Rosenzweig (2001) has suggested the use of the provincial SAR, arguing that in the long term each native habitat fragment will behave as an isolated province. At the other extreme, it could be argued that in the short term the species that go extinct are the ones endemic to the area of lost habitat, and that is best described by the continental SAR (Kinzig et al. 2001; Rosenzweig 2001). In order to give the full range of possibilities, we made our calculations using the three types of SAR.

Finally, we found that tropical forest and tropical woodland had higher z-values than other biomes. Therefore, for those biomes and for each type of SAR we used z-values 20% higher than the means reported in Table 10.1. At the other extreme, for the tundra biome, we used z-values 20% lower than the mean.

10.2.1.4 Estimating Global Species Losses

We estimated changes in global biodiversity by calculating the change in area as a result of both habitat loss (due to agricultural expansion) and climate change. We assumed that the SAR could be applied independently for each realm-biome combination—thus assuming that the overlap in numbers of species was minimal (relative to the number of species that are endemic to each realm-biome combination). Furthermore, we assumed that diversity loss would occur as a result of the transformation of natural vegetation into a human-dominated land cover unit and that human-dominated vegetation had a diversity of zero endemic native species.

In our calculations, we did not assume any extinction rate with time, but simply assumed that at some point in time, the number of species will reach the level as indicated by the SAR. This means that our results should not be interpreted in terms of an immediate loss of number of species but in terms of species that may go extinct when populations eventually reach equilibrium with the reduced habitat.

We also applied SAR in only one direction: habitat loss leads to extinctions of species, but subsequent increase in area of a habitat would not lead to a similar increase. We

did so because the processes of extinction and speciation occur in different time scales, with losses occurring much faster than the evolution of new species.

Finally, in IMAGE 2.2 fast climate change can lead to a difference between actual and potential vegetation. Areas where the colonization of the potential vegetation has not yet occurred can be assumed to have a significantly reduced biodiversity (Leemans and Eickhout 2004). Here, we assumed a loss of 50% of the species in the grid cells where adaptation of the vegetation lags. Our estimates of species losses at equilibrium have low certainty because they are based on a series of models that are linked sequentially, with the output of one being the input to the next. Moreover, each of them has its own assumptions and uncertainties.

10.2.1.5 Estimating Local Species Losses

We estimated local losses of biodiversity as a direct function of habitat loss. Species inhabiting a patch of native vegetation go locally extinct as the patch is converted into habitats such as agricultural land or urban patches in which they cannot survive. Losses of species are directly proportional to losses in native habitat. Moreover, in contrast to the SAR calculations for global biodiversity, this indicator is assumed to be fully reversible under the time scales considered. This difference from the global losses calculated in the previous section is based on the fact that species that go locally extinct in one patch may survive in other patches.

Local and global losses of biodiversity are important for different reasons. Global extinctions are particularly important for humans because they are irreversible and they eliminate some ecosystem services, such as the maintenance of the genetic library. This type of ecosystem service depends on the existence of unique genetic combinations that can be used to develop new pharmaceutical drugs as well as new varieties of plants and animals that may cope with new diseases or climate change. Local extinctions are important because they affect local human populations and global provisioning of ecosystem services that depend on the abundance of individual species.

Local species losses were estimated as a function of the transformation of native habitat into another category such as several agricultural categories and urban patches. We report changes in native habitat availability for 2050 relative to habitat availability for 1970 for the four MA scenarios and disaggregated by biome and biological realm.

10.2.2 Uncertainties in Extinction Predictions

As explained, one important source of uncertainty in our predictions is the slope of the species-area relationship. This results both from not knowing which type of SAR is more appropriate to describe biodiversity loss and also from the wide range of the z-values for a given type of SAR. To quantify the uncertainty associated with the type of SAR used, we present results for the three types. To quantify the uncertainty associated with the wide range of z-values, we use Monte Carlo simulations based on the distribution of z-values in the literature.

The z-values for the continental SAR follow a lognormal distribution. (See Figure 10.2; Kolmogorov-Smirnov, $p = 0.764$.) A similar pattern is observed for the island SAR (Kolmogorov-Smirnov, $p = 0.994$). We used Monte Carlo simulations to convert this probability distribution of the z-values into confidence intervals for the extinctions. In each Monte Carlo simulation, a random z-value was drawn from the lognormal distribution and used for all region/biome combinations (except tropical forest/woodland, with a value 20% higher than the drawn value, and tundra, with a value 20% lower). Then, using IMAGE, we calculated the number of extinctions per biome and region and also the total number of extinctions. Five hundred Monte Carlo simulations were performed. The mean, the standard deviations, and the range of the predicted extinctions are reported.

Our biodiversity loss estimates assume a low overlap in the species composition of the different biomes. In order to examine this assumption, we studied the distribution of North American plants, using the database of Kartesz and Meacham (1999), which lists for each state in the United States and for each Canadian province the composition of vascular plants. We selected for each biome a set of states that were covered only by that biome, based on the 1995 IMAGE land cover map. (See Table 10.2.)

The matrix of vascular plant species overlap between the different biomes is shown in Table 10.3. The matrix is asymmetric around the diagonal. For instance, although 73% of the plant species in tundra and ice are also present in the boreal forest and cool conifer biomes, only about 12% of the latter biomes' species are present in tundra and ice. This asymmetry is caused by forest and cool conifer biomes being more spacious than tundra and ice.

Overlaps vary widely, but the general pattern is that the bigger the distance between the biomes, the smaller the species overlap. On average, the maximum overlap with a

Table 10.2. States or Provinces Selected as Representatives of Each Biome in Order to Analyze Species Overlap

Biome	State or Province
Tundra and ice	Franklin
Warm mixed forest	South Carolina, Mississippi, Louisiana, Alabama
Temperate deciduous forest	Virginia, Kentucky, West Virginia
Temperate mixed forest	Nova Scotia, Maine, Vermont, New York, Michigan, Iowa, Wisconsin
Scrubland, grassland, and hot desert	California, Arizona, New Mexico
Tropical forest	Puerto Rico, Virgin Islands
Boreal forest, cool conifer	Newfoundland, Manitoba

neighbor biome is about 60%. Thus, assuming the extreme and unlikely case of one biome disappearing and the other remaining intact, we would predict a little more than twice the extinctions that would in fact occur. One possible approach to avoid this problem would be to consider only the species endemic to each biome. However, it has been found (Borges personal communication) that restricted range species may follow a SAR with a higher z-value than broadly distributed species. Therefore the problem of not having a very large proportion of endemics to each biome may be smaller than what Table 10.3 could suggest.

Our approach focused primarily on vascular plants because of the numerous studies of SARs and the availability of global data sets of species density. How representative are scenarios based on vascular plants of the patterns expected in other taxa? Can we extrapolate results obtained with vascular plants to other taxa? The answers to these questions are related to another one. How related are richness patterns among taxa? In order to address these questions, we undertook an assessment of the literature within the MA framework.

The review on richness correlations among taxa was largely based on the literature published over the last decade, but some earlier works have also been included. Of the more than 100 publications reviewed, only 48 were appropriate for extracting data. The attention paid to richness correlations is strongly different for different groups, with mammals (10 publications), vascular plants (13), beetles (12), butterflies (19), and birds (26) being the best-represented taxa.

Most of the authors of the literature focused on a certain continent or country or even on smaller regions. Some claimed that certain groups such as tiger beetles and large moths can serve as reliable predictors of the richness of other taxa at the global level, but these conclusions seem to be driven by the wish to make generalizations from a few charismatic groups rather than by detailed analyses of complex assemblies. The diversity of trees and shrubs as well as that of butterflies, birds, and mammals showed a comparatively high share of correlations with other taxa that were significantly positive. More than 50% of the total amount

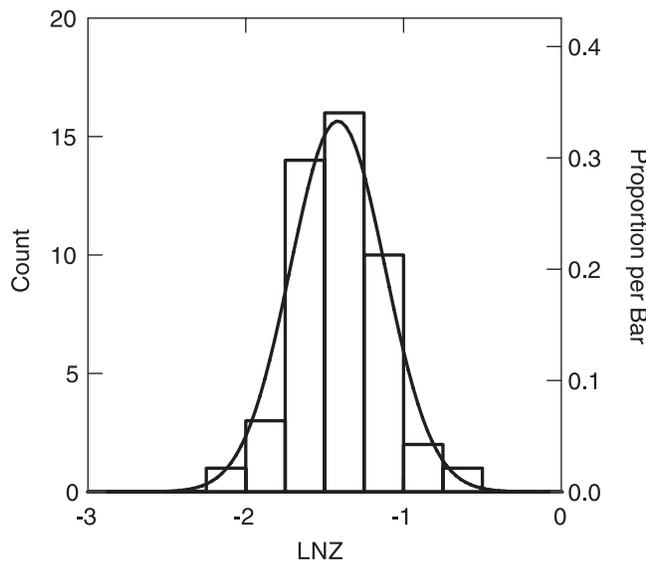


Figure 10.2. Histogram of Z-Values of the Species-Area Relationship in Continental Studies of Vascular Plants, Fitted with a Log-normal Distribution

Table 10.3. Overlap in Vascular Plant Composition between Different Biomes in North America and the Caribbean

Biome	Tundra and Ice	Warm Mixed Forest	Temperate Deciduous Forest	Temperate Mixed Forest	Scrubland, Grassland, and Hot Desert	Boreal Forest and Cool Conifer	Tropical Forest
			(percent)				
Tundra and ice		4.6	11.8	40.2	30.0	73.2	0.8
Warm mixed forest	0.3		63.6	50.1	28.9	17.3	14.1
Temperate deciduous forest	1.1	76.6		73.8	32.9	29.6	9.1
Temperate mixed forest	3.2	52.5	64.1		37.8	40.1	6.0
Scrubland, grassland, and hot desert	1.2	15.2	14.3	19.0		12.4	4.1
Boreal forest, cool conifer	12.1	37.9	53.6	83.7	51.5		3.9
Tropical forest	0.1	21.2	11.3	8.5	11.5	2.7	

of the significant positive correlations reported focused on birds, mammals, and vascular plants. Significant negative correlations, in contrast, have been recorded with species richness of lichens, ants, and beetles (only one case available for each group). Unfortunately, most authors do not provide the complete results of regression analyses but only mention correlation coefficients. Thus, the list of regression equations that could be used for modeling exercises only includes four studies covering four pairs of taxa.

Correlation between identical taxa can be different in grids of different size. Significant positive correlations for butterflies, birds, and mammals prevail at the low-resolution scales, while correlation results for vascular plants and beetles are more randomly distributed across scales. For vascular plants, resolution scale distribution is much more scattered than for all other groups. In the size class used in the MA model (100–999 square kilometers), six correlations were discovered but no promising results on correlations of vascular plants with other groups could be found. Similar checks could not be made for large moths and tiger beetles (the groups assumed to be good indicators for global biodiversity) (Pearsson and Cassola 1992).

The current state of knowledge generally indicates that we cannot generalize spatially explicit estimates of changes in global biodiversity based on diversity changes of particular focal taxa. Prendergast et al. (1993) showed that coincidence of biodiversity hotspots for several taxa is rare, suggesting the difficulties of extrapolating data from one taxa to others.

10.2.3 Terrestrial Biodiversity Change across Scenarios

10.2.3.1 Local Loss of Species through Loss of Habitat

Conversion of a patch of native vegetation into a cropland or part of the urban landscape results in the immediate extirpation of populations. In the case of the shift to agriculture, a simple community made up of one or a few cultivated species and a reduced number of cosmopolitan weeds replaces a diverse ecosystem. Conversion may involve log-

ging of the forest, burning remnants with no commercial value, plowing the soil, and planting of a monospecific crop. The losses of the local populations with the conversion of a unit of native vegetation into a human-dominated patch are rapid and directly proportional to habitat loss.

Local losses are very important because without adequate conservation efforts they can lead to global losses of species. Furthermore, local losses affect the provisioning of ecosystem services derived from biodiversity to local people and after some time affect the global services provided by ecosystems.

Habitat losses by 2050 relative to habitat availability in the year 1970, and the corresponding extirpation of population and local losses of species, increase dramatically in all scenarios, ranging between 13 and 20%. (See Figure 10.3.) Order from Strength had the largest losses (20%) as a result of the relatively large increase in food demand due to the high population growth rate and the relatively slow increase in yield. The latter resulted from the relatively low transfer of technology to regions far from the centers of technology development, where the highest population growth is also expected. This means that under this scenario, increasing

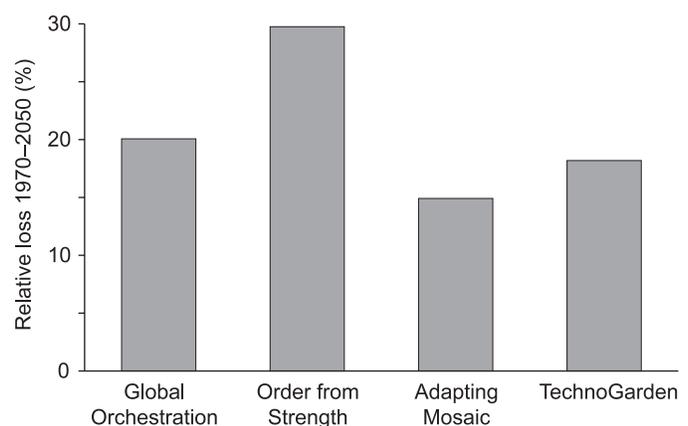


Figure 10.3. Losses of Habitat in 2050 Relative to Habitat Availability in 1970 for the MA Scenarios

the crop area was the necessary response to meet the increasing food demand. Finally, the increase in agricultural area occurred at the expense of a reduction of native habitat, with the consequent extirpation of local population and the local losses of species.

TechnoGarden was the scenario with the lowest loss of habitat (13%), mostly as a result of optimistic estimates in the increase in crop yield resulting from fast technological progress. Adapting Mosaic and Global Orchestration yielded intermediate results resulting from the relatively slow increase in food demand in the former and relatively low human population growth in the latter. It should be noted that in these calculations, losses have been assumed to be fully reversible.

Habitat loss was not uniformly distributed across the different biomes of the world. (See Table 10.4 and Figure 10.4.) Warm mixed forests and savannas in the Order from Strength scenario were the biomes with the highest losses. These biomes are located in areas such as Africa with the highest growth in human population and corresponding growth in food demand. Order from Strength constrained the trade of food and consequently most of the food demand had to be met locally, resulting in a large-scale transformation of native habitat into agricultural land and large losses of local populations. Adapting Mosaic showed the largest losses of habitat for warm mixed forests in addition to large losses in temperate deciduous forests due to similar mechanisms as those described for Order from Strength.

Habitat availability increased slightly in some biomes, such as the temperate mixed forest, mostly as a result of abandonment of pastures. A large fraction of the temperate mixed forest is located in the industrial world, where most of the land use changes have already occurred. In addition, in the Order from Strength and Adapting Mosaic scenarios, which emphasize regionalization (as opposed to globalization), demands for food and agricultural land in the temperate mixed forest region are stable.

Local and global losses of biodiversity differ in reversibility. Local losses could be reversed as a result of abandonment of agricultural land or active conservation practices. Populations can invade from adjacent patches naturally or with human intervention. Ecosystem services derived from local diversity can therefore increase or decrease as a result of gains and losses of habitat. Habitat and population losses can occur very rapidly, but gains in habitat from abandonment take longer periods of time, depending on the ecosystem of interest.

10.2.3.2 Global Loss of Vascular Plant Species through Loss of Habitat

The loss of vascular plant species that would occur when they reach equilibrium in 2050 differs among the four MA scenarios because of differences in expansion of agricultural area. (See Figure 10.5.) Worldwide, the changes in habitat availability experienced during the 1970–2050 period may result in a decrease of 12–16% in biodiversity at equilibrium. By far the strongest decrease occurred for the Order from Strength scenario (which had the largest expansion of cropland due to slow yield improvement and high population growth). In fact, by 2050 the changes in habitat and consequent species losses of this scenario had hardly slowed down compared with historical rates.

The TechnoGarden and Adapting Mosaic scenarios, in contrast, showed the slowest losses—on the order of 12% of species lost at equilibrium compared with current biodiversity. In TechnoGarden, the lower species decline rate was mainly due to much higher yield improvements in developing countries and a stabilizing population. Consistent with the storyline of the scenario, the assumptions of this scenario are relatively optimistic. In Adapting Mosaic, slower development rates in developing countries slowed down increases in food demand. Global Orchestration fell in between the other three scenarios, with a 13% loss of species at equilibrium with 2050 land use changes.

Table 10.4. Change in Land Cover in 2050 in Four Scenarios

Category of Land	2000	GO	OS	AM	TG	GO	OS	AM	TG	
		(million hectares)					(percent change)			
Agricultural land	3,357	3,646	4,162	3,580	3,660	109	124	107	109	
Extensive grassland	1,711	1,700	1,704	1,704	1,707	99	100	100	100	
Regrowth forests	446	630	523	550	462	141	117	123	103	
Ice	231	224	225	222	221	97	97	96	96	
Tundra	768	727	727	726	724	95	95	95	94	
Wooded tundra	106	84	83	86	89	79	78	81	83	
Boreal forest	1,509	1,554	1,551	1,556	1,553	103	103	103	103	
Cool conifer	168	196	188	192	194	117	112	114	116	
Temperate mixed forest	201	262	236	250	287	130	117	124	143	
Temperate deciduous forest	145	133	110	119	155	91	76	82	107	
Warm mixed forest	95	79	62	76	109	83	65	80	115	
Steppe	804	750	692	749	730	93	86	93	91	
Desert	1,678	1,643	1,637	1,660	1,665	98	98	99	99	
Scrubland	207	170	122	183	182	82	59	88	88	
Savanna	705	404	316	511	450	57	45	73	64	
Tropical woodland	483	517	426	524	503	107	88	109	104	
Tropical forest	670	568	520	597	594	85	78	89	89	

Key: GO = Global Orchestration; OS = Order from Strength; AM = Adapting Mosaic; TG = TechnoGarden

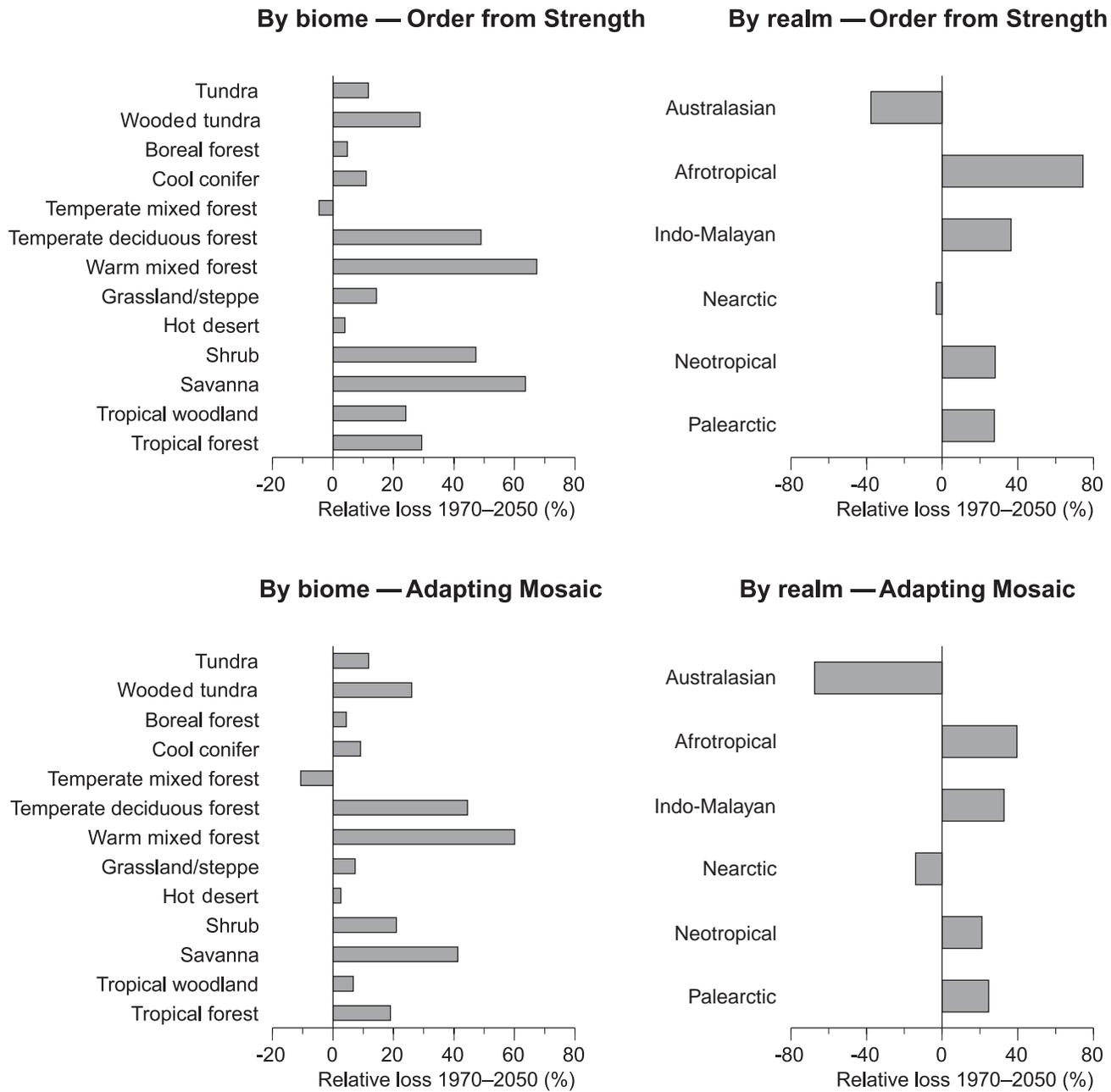


Figure 10.4. Losses of Habitat in 2050 Relative to 1970 for Different Biomes and Realms for Two Scenarios (Negative values indicate net increase.)

There are major differences in the species loss trends among the different biomes. Figure 10.6 shows the relative losses by major biome type and ecological realm for Order from Strength and Adapting Mosaic. The results indicate that warm mixed forests, savanna, scrub, tropical forests, woodlands, and temperate deciduous forests seem to suffer most from biodiversity losses through loss of habitat. In particular, tropical forest, tropical woodland, savanna, and warm mixed forest account for 80% of all species lost (in total, nearly 30,000 species).

While all biomes show lower habitat and species losses under Adapting Mosaic than under Order from Strength, this is particularly so for the tropical biomes. When comparing both scenarios, there are also differences in the time frames in which the different biomes suffer habitat losses. In Order

from Strength, it can be seen that for the temperate biomes and warm mixed forests almost all habitat losses occur before 2020, while for the tropical habitats the 2020–50 period sees almost similar habitat losses as the 2000–20 period. In contrast, under Adapting Mosaic a considerably different time dynamic is seen, with most of the habitat losses in warm mixed forest occurring in the second period.

The largest relative habitat and species losses occur in the Afrotropical region, which has the largest expansion of agricultural land (driven jointly by a rapidly increasing population and strong increases in per capita food consumption) under all scenarios. The second most important region in terms of relative losses is the Indo-Malayan region. The Palaearctic region, in contrast, sees the lowest losses in biodiversity through loss of habitat.

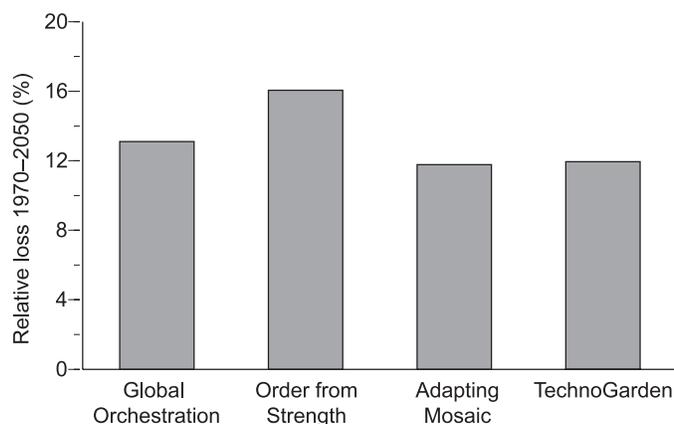


Figure 10.5. Relative Losses of Global Vascular Plant Biodiversity when Populations Reach Equilibrium with Reduced Habitat for MA Scenarios

A major uncertainty in the analysis is represented by the z value used in the SAR equation. To analyze the uncertainty of z , two experiments were conducted: a Monte Carlo analysis sampling z 's from the distribution indicated earlier and replacement of the z 's from the studies on the island scale by the provincial scale. The analysis indicates that a 16% loss of vascular plant species for the Order from Strength scenario falls within a range of plus and minus the standard deviation of 10–20%. (See Figure 10.7.) The highest and lowest runs (within the total set of 500 runs) show a 6% and 29% loss, respectively.

Replacing the z 's from the island scale to those established for provinces resulted in an increase of forecast losses. For Order from Strength, losses increased from 16% to 26%. On the other hand, using the z values as established for the continental scale decreased losses to 13%. The relative results and the changes over time, however, remain the same. The left-hand side of Figure 10.7 shows the results of a Monte Carlo analysis sampling from the log-normal distribution of island scale z -values for vascular plants as shown in Figure 10.5 (from a set of 500 runs). The right-hand side of Figure 10.7 compares the results of using z values from studies at the continent, island, and provincial scale for Adapting Mosaic and Order from Strength.

The level of aggregation (in terms of the total number of biomes in which the terrestrial vegetation was divided) represents an uncertainty within the analysis. In our analysis, we assumed that regions have a high proportion of endemism. Increasing the number of regions under this assumption will increase the global number of species lost. Having too few regions would underestimate species loss, since it would not take into account different ecological regions. Alternatively, having too many regions would overestimate species loss, as it would double count too many species when aggregating from local to global extinction.

The influence of the regional definitions was analyzed by varying the number of regions in the analysis from 4 to 75 (versus 65 in standard run). (See Figure 10.8.) The highest aggregated regional definition used in this analysis corresponded to the highest aggregation level in Bailey's map

of global ecoregions (four domains: arctic, humid temperate, dry tropic, and humid tropical). The next level corresponds to the 14 biomes recognized in IMAGE at the global level (loosely corresponding to the level of divisions in the Bailey set). The third level combines the four domains of the Bailey set with the realms of the WWF ecoregion map, creating a total set of 24 regions. The fourth level corresponded to the standard regional definitions used in this analysis, while the fifth level adds an additional 10 regions by assuming that East Asia and Japan can be identified as separate realms (which can be concluded from the province level map of the Bailey ecoregion definitions). The results indicate that indeed our results do vary for these different definitions, with a larger species loss with increasing disaggregation of vegetation units. The number of species lost increased with the number of units in which Earth vegetation was partitioned.

10.2.3.3 Convention on Biological Diversity Target

The target of the Convention on Biological Diversity is to bring the rate of loss of biodiversity by 2010 significantly lower than the current rate. To test this, we assumed that the current rate should be interpreted as the historic average of the last two decades as calculated using the SAR approach on the basis of the IMAGE global change data. First, we calculated the habitat loss that occurred during the last two decades and estimated its effect on species diversity when populations reach equilibrium with the reduced habitat. Second, we calculated the average rate of change in habitat loss and the consequent change in the number of equilibrium species for each of the four scenarios for the period 2000–20, centered in the year 2010. In order to estimate the probability that each scenario meets the CBD target, we plotted the loss of equilibrium species relative to historic rates. (See Figure 10.9.)

It is unlikely that the CBD target will be met for terrestrial ecosystems under the scenarios explored by the MA. Order from Strength and Global Orchestration would probably not meet the target because the estimated rates of habitat loss and the consequent losses of species at equilibrium exceeded those of the previous 20 years. Order from Strength presented a rate of loss that was considerably higher than the historic rate, mostly as a result of the relatively slow improvement of agricultural efficiency in combination with a sharp increase in food demand. Global Orchestration also showed relative rates of loss that were somewhat higher than the historic rates, resulting from an improvement of the historic rate of food consumption (and therefore agricultural expansion). TechnoGarden and Adapting Mosaic have more success at reducing the loss rates of terrestrial biodiversity relative to the historical rates. However, we expect with *high certainty* that our analysis is underestimating losses of the different scenarios because it does not take into account other pressures on biodiversity such as climate change and nitrogen loading, which are expected to increase.

10.2.3.4 Loss of Terrestrial Biodiversity through Climate Change

Climate change will certainly influence several aspects of ecosystems. Grabherr et al. (1994) were among the first to

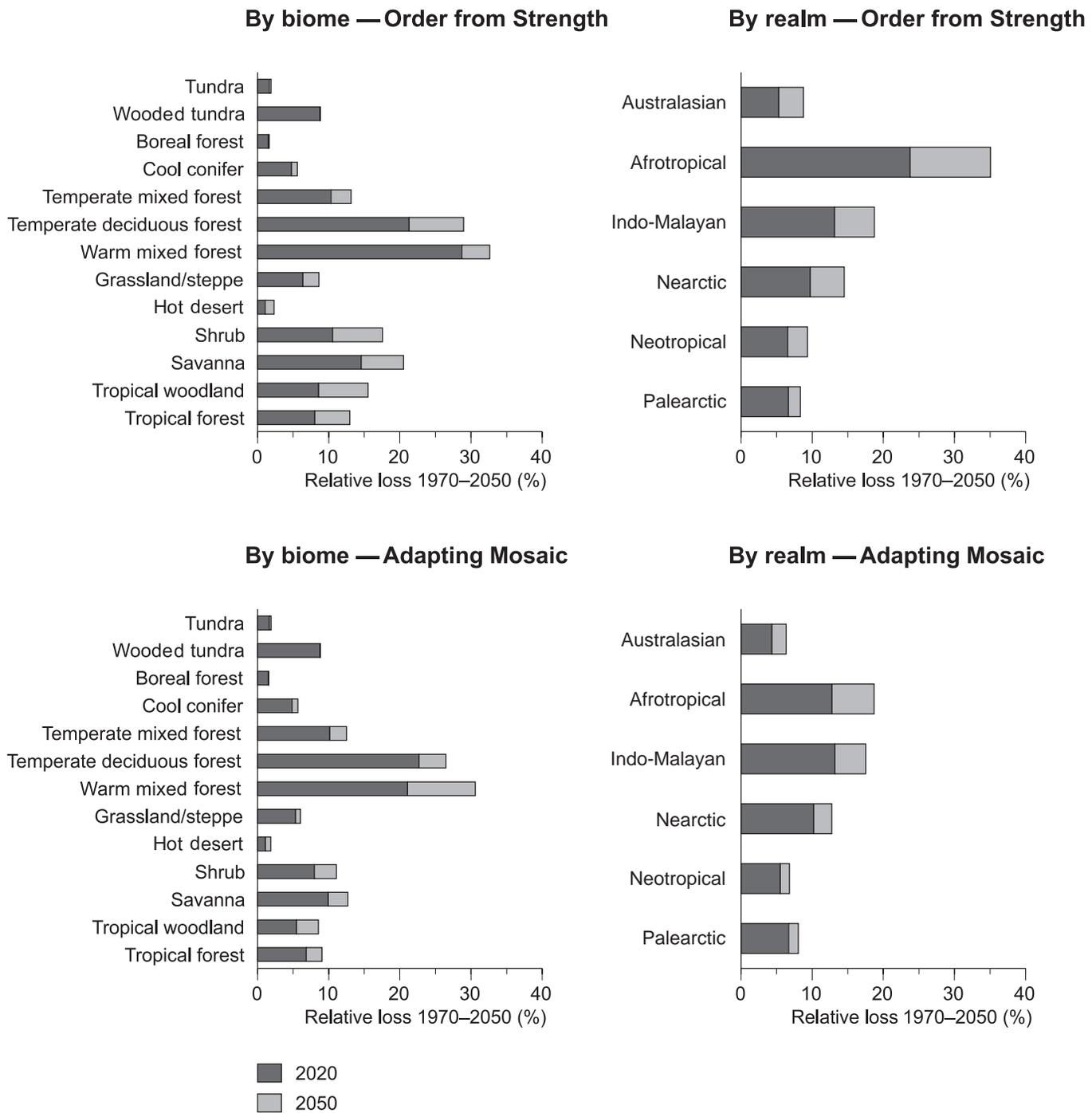


Figure 10.6. Relative Losses of Biodiversity of Vascular Plants through Habitat Loss for Different Biomes and Realms for Two Scenarios. Losses would occur when populations reach equilibrium with habitat available in 2050 and are relative to 1970 values.

report that ecosystems are already changing as a result of climate change. They used long-term observations from alpine vegetation and demonstrated that the distributions of many species had increased in altitude. Parmesan and Yohe (2003) analyzed the response of more than 1,560 plant and animal species in both marine and terrestrial environments and reported a clear effect of climate change on their distribution. Their analysis documented an average range shift of 6 kilometers per decade toward the poles or meters per decade upward. Other similar observations have been made of the impacts of climate change on the distribution of several

plant and animal species (Both and Visser 2001; Root et al. 2003).

One of the most common ways to study the impact of climate change on the distribution of ecosystems is to describe their climatic envelope and compare them against climate-change scenarios provided by global circulation models (e.g., Prentice et al. 1992; Cramer and Leemans 1993; Malcolm and Markham 2000). Van Minnen and Ihle (2000) have attempted to add some form of transient impacts to the “climate envelope” approach by modeling the migration process of total biomes as a function of distance,

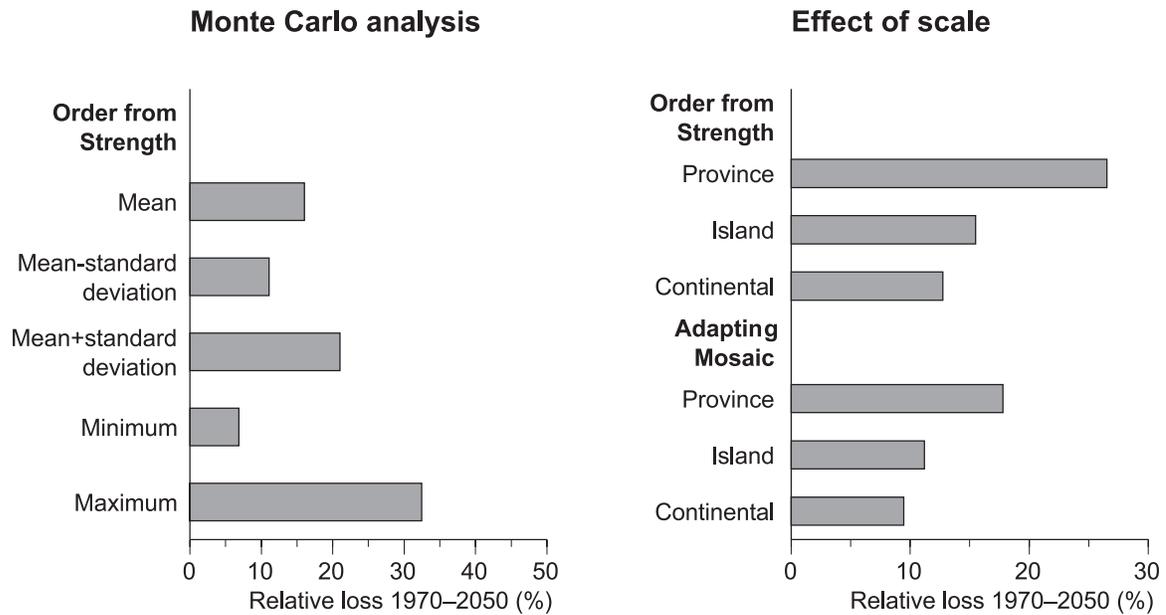


Figure 10.7. Uncertainty Analysis for Different Z Estimates. Influence of using different estimates of z calculated using a Monte Carlo technique on relative species loss for the Order from Strength Scenario shown left. Influence of using z-values for continents, islands, and provinces on the estimate of biodiversity change for two Scenarios shown right.

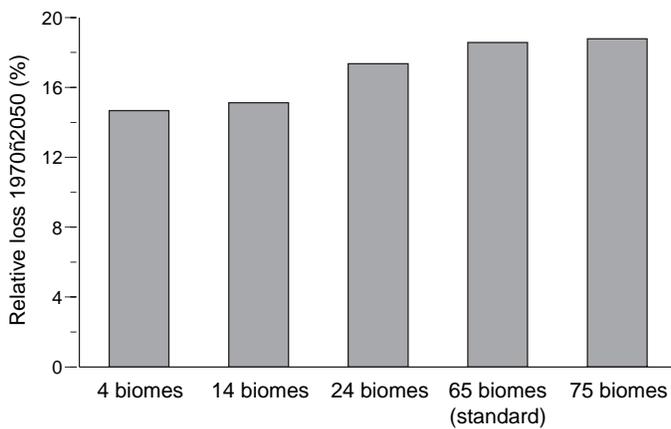


Figure 10.8. Influence of Using Different Regional Definition on Loss of Vascular Plant Estimates at Equilibrium for Order from Strength Scenario

migration rates, and original and new vegetation types. Real changes could be much more complex because individual species are the units that will respond to climate change. Solomon and Leemans (1990) concluded that future climate change could lead to large-scale synchronization of disturbance regimes, leading to the emergence of early-phase succession vegetation, with opportunistic generalist species dominating over large areas.

The decline of individual plant species due to climate change results from either competitive exclusion or the direct effect of climate change through increased drought frequency. Species changes resulting from competitive exclusion occur much more slowly than the response due to increased drought occurrence. Therefore, the fastest impacts might be expected in areas that show increased

drought as a result of climate change. A comparison of global circulation models identified Central America, Southern Africa, Southern Europe, and Northern Australia as areas where all models agree that water availability will decline.

To test the impacts of climate change on biodiversity under the MA scenarios, we used three different approaches. First, we used a process-based model developed by Kleidon and Mooney (2000) that simulates the response of randomly chosen parameter combinations (“species”) to climate processes. The model mimics the current distribution of biodiversity under current climate, and modeled “species” can be grouped into categories that closely match currently recognized biomes. Second, we used a model of African tick species diversity to show possible changes in species ranges as result of climate change. Finally, we con-

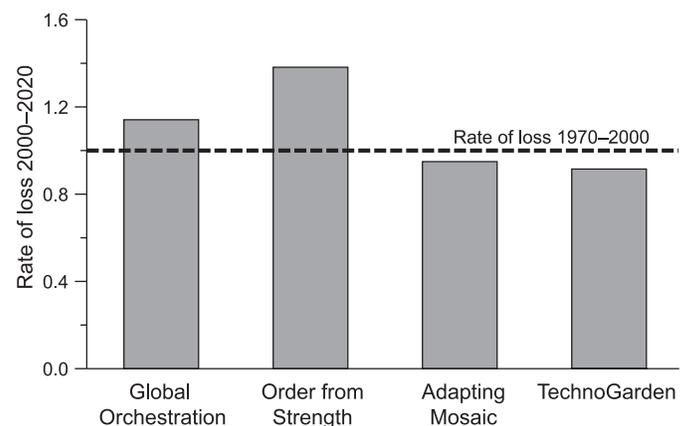


Figure 10.9. Potential Species Loss for MA Scenarios Compared with Historic Rates

sidered the possible impact of climate change at the level of biomes as calculated by IMAGE.

10.2.3.4.1 *Climate change and potential biodiversity change*

A warmer climate with altered hydrological regimes will affect plant functioning and the distribution of plant species. The impact of these changes on the potential distribution of plant diversity within the context of the MA scenarios was estimated by using the simulation modeling approach of Kleidon and Mooney (2000). Using the monthly mean temperature and precipitation anomalies of the Global Orchestration scenario, we altered the climatic forcing of the model. Light use efficiency was increased by 23%, consistent with the simulated increase of NPP by the IMAGE model. Modeling results indicated (*with low certainty*) that global environmental changes would lead to an increased environmental capacity for plant diversity in most regions of the world. (See Figure 10.10 in Appendix A.) These higher levels are mainly attributable to the increase in light use efficiency. Since the evolution of new species is unlikely to happen in this time frame, the increased capacity might increase the vulnerability of ecosystems to invasions.

Note that the magnitude of change is especially sensitive to the assumed increase in light use efficiency. If, for instance, the increase of productivity with elevated CO₂ were less, the increase would be less or counteracted by the decrease in precipitation. In this sense, Figure 10.10 mainly illustrates the point that biodiversity—the *c* in the species-area curves—is likely to be affected by global climatic change.

10.2.3.4.2 *Climate change and tick diversity in Africa*

We used a well-documented set of tick distribution data (Cumming 1998, 1999, 2000, 2002) to develop multivariate logistic regression models of the pan-African occurrences of individual tick species from climate data at 0.5-degree resolution. We used ticks as a model organism for invertebrates as their distributions are tightly linked to climate. Ticks are also significant to the public because they can be vectors of diseases. Maps for 73 species (from a fauna of about 240 species) were stacked to produce species-richness estimates under current (see Figure 10.11 in Appendix A) and projected conditions. The most severe impacts of climate change are likely to be on species that are highly specialized, including those with small species ranges and limited physiological tolerances. Although some African ticks probably fall into this category, insufficient data were available to model the distributions of rare ticks or those with limited species ranges. Consequently, the analysis should be interpreted as giving insight into how hardy, long-lived invertebrates with relatively high dispersal ability may respond to climate change.

Although the analysis predicted no extinctions among the 73 tick species considered in this analysis, changes in local tick species richness as a consequence of expansion and contraction of species ranges are likely to occur in all scenarios. (See Figure 10.12 in Appendix A.) The differences among scenarios are subtle and reflect the spatial nature of changes in weather patterns. Local biodiversity

increased sharply in some areas and decreased sharply in others. TechnoGarden is the scenario in which the least expansion of tick species ranges occurs; the most expansion occurs in Order from Strength, although tick species richness in this scenario is reduced in areas of Angola and Tanzania. The dominant trend across all scenarios is an increase in local tick species richness (probably accompanied by increased tick burdens on livestock), with reductions in a smaller subset of locations.

10.2.3.4.3 *Climate change and biome shifts*

In many cases, biomes will shift geographically along with changes in climate. In general, the edges of current ecological zones are affected the most. It should be noted that each species would respond independently because each of them has different environmental requirements and a different capacity to adapt. However, our current understanding and models do not allow us to model shifts at the species level.

This exercise has modeled shifts with climate of vegetation types, which may have some structural similarity with current vegetation types but likely will not have the same species composition. The loss of species belonging to vegetation types that lost area as a result of climate change was calculated using the SAR approach described earlier. Increases in area of vegetation types did not result in increases in diversity. Vegetation types that would not have enough time to shift along the rate of climate change will result in degradation of remaining systems. In this assessment, we assumed that areas that shifted to new potential vegetation have lost all the endemic species of the original vegetation type but that those areas where species did not have time to adjust to the new climatic conditions lost only 50% of the original species. Figure 10.13 in Appendix A indicates these different categories according to IMAGE calculations.

The number of affected ecosystems increases with time, given increasing climate change. Only in the TechnoGarden scenario, after 2050, does the number of ecosystems without adaptation decrease because of the stabilization of GHG concentrations and the lower rate of temperature change. In the four scenarios, about 5–20% of the ecosystems will be seriously affected by climate change—without possible adaptation, the worst would be Global Orchestration. Focusing on protected areas only reveals similar numbers. Under Global Orchestration, in 20% of the protected areas the originally protected ecosystem will have been either replaced or severely damaged as a result of climate change. In the case of protected areas, any change (with and without adaptation) can be assumed to be negative, as in most cases they were selected because of the uniqueness of the ecosystems.

Figure 10.14 summarizes the results by biome for the two extreme climate change scenarios, Order from Strength and TechnoGarden. It shows that climate change is going to have an impact on biodiversity under both scenarios. The impacts, however, are more severe in Order from Strength. The Figure shows that most heavily affected biomes (in terms of percent change) include cool conifer forests, tundra, shrubland, savanna, and boreal forest.

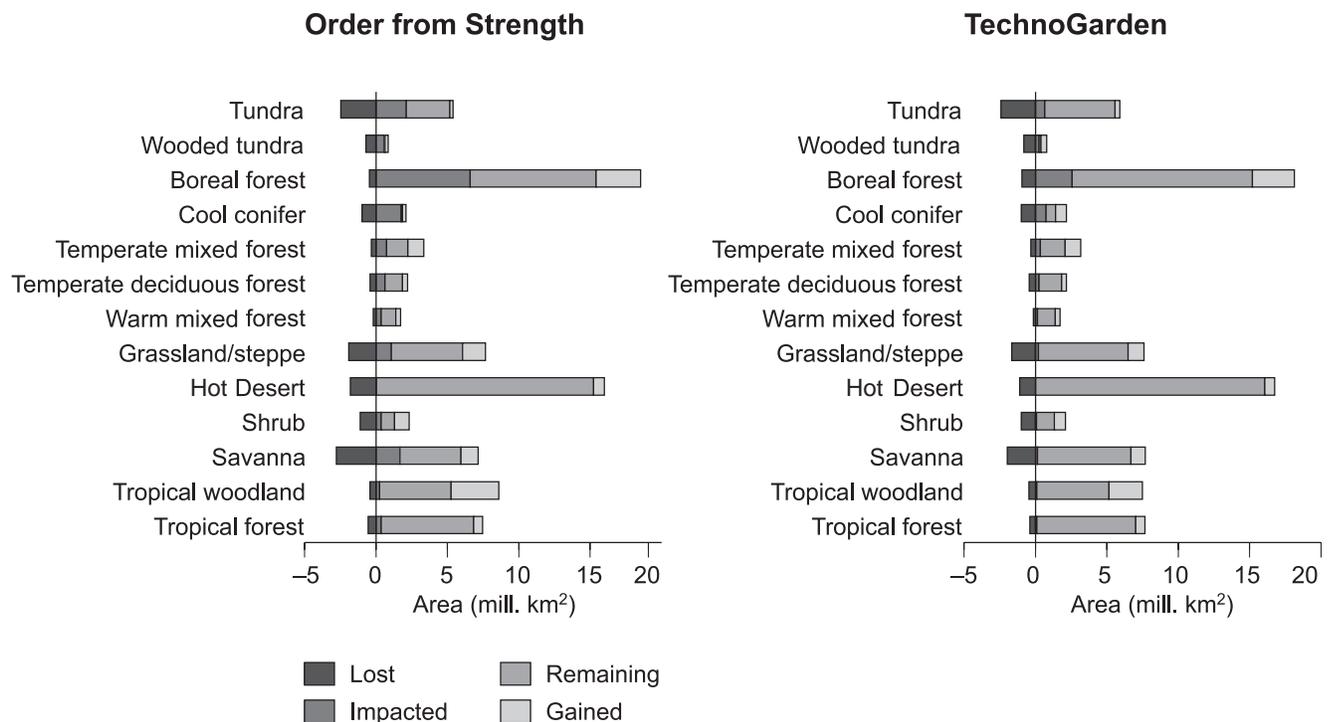


Figure 10.14. Impact of Climate Change in 2100 on Area of Different Biomes in Order from Strength and TechnoGarden Scenarios

Thomas et al. (2004) performed a similar analysis of the potential effect of climate change by 2050 on global extinctions for selected regions of the world, which accounted for 20% of the area of terrestrial ecosystems. The authors evaluated three rates of climate change and two hypothetical cases with or without dispersal limitations. Results of this analysis encompass the results of the four MA scenarios reported in Figure 10.14. Maximum extinction due to climate change, as reported by Thomas et al. (2004), was 100% of the plant species in Amazonia, assuming no dispersal capabilities and maximum rate of climate change; there was a minimum value of 3% for Europe, assuming no dispersal constraints and minimum climate change. On average, across regions and taxa, the Thomas et al. (2004) exercise yielded losses of species at equilibrium with reduced habitat that ranged between 15% and 37%.

10.2.3.5 Loss of Terrestrial Biodiversity through Changes in Atmospheric Deposition

Atmospheric deposition of nitrogen can lead to change of ecosystems as a result of nitrogen excess (also called terrestrial eutrophication). Nitrogen excess can be an important cause of ecosystem degradation as it is the primary nutrient limiting plant production in many terrestrial environments. Increases in nitrogen input can therefore alter these ecosystems and lead to shifts in species composition, increased productivity, decreased species diversity, and altered tolerance to stress conditions (Pitcairn 1994). The most important anthropogenic sources of nitrogen emissions are fossil fuel burning and industrial and agriculture activities. Excess nitrogen deposition has been a prominent environmental problem in North America and Europe since 1970.

Bouwman et al. (2002) made a global assessment of acidification and excess nitrogen deposition effects on natural ecosystems by overlaying current and future deposition maps of sulfur and nitrogen with sensitivity maps for both acidification and nitrogen deposition (at 0.5x0.5 degrees). These sensitivity maps are expressed in so-called critical load values below which no damage is assumed to be negligible. They are calculated on the basis of soil, ecosystem, and climate data and on soil dust deposition. For future emissions, they used deposition maps calculated by the STOCHEM environmental chemistry and transport model based on the IPCC IS92a scenario.

To estimate excess nitrogen deposition risks under the MA scenarios, we followed the approach of Bouwman et al. (2002) and scaled their deposition map on the basis of changes in nitrogen emissions in each of the 17 global regions of the IMAGE 2.2 model (on their turn, based on changes in energy use, agriculture, and environmental policy). At a 0.5x0.5 degree map, we calculated the deposition of nitrogen and compared it to critical loads of these grid cells. The ratio between deposition and critical load was used as an indication of risks of nitrogen deposition. A ratio below 1 implies limited risks (at least based on the average grid cell; obviously within the grid cell many ecosystems will occur that are more sensitive than the average, and this approach will therefore result in an underestimation of the actual risk); a high ratio indicates a very high-risk level of disturbance.

Bobbink (2004) reported biodiversity losses for different levels of nitrogen loading for a large number of different ecosystems, and these results were summarized in a large number of ecosystem-specific relationships. Wedin (1996) (see also Tilman et al. 1996; Haddad et al. 2000) showed

changes in plant biodiversity for grasslands for different levels of nitrogen loading. (See Figure 10.15.) These studies indicated a 25% reduction of diversity for a ratio of three times critical load, a 50% reduction for eight times the critical load, and a 60% reduction for 25 times the critical load. We used those numbers to calculate our overall threat indicators—and aggregated our results to the level of the IMAGE 2.2 biomes (by WWF region). The numbers show the average reduction in diversity by biome. (See Figure 10.16 in Appendix A.)

10.2.3.6 Integrating Different Environmental Pressures on Biodiversity

On the basis of the indicators calculated above, it is possible to compare the impact of different drivers of biodiversity loss. (See Figure 10.17.) For habitat loss, we used the SAR approach described earlier, but accounting only for changes due to agricultural expansion and timber production. For climate change and excess nitrogen, we also used the SAR approach to aggregate grid-level effects at the level of complete biomes. Figure 10.18 shows the impacts at the global level of the combination of habitat loss, climate change, and nitrogen deposition under the four MA scenarios in 2050.

Earlier, Sala et al. (2000) performed an exercise in which they developed the same kind of graphs on the basis of expert judgment, supported by selected modeling results. In general, the current calculations confirmed the findings of Sala et al. (2000), with some exceptions. As in the earlier study, habitat loss was found to be the most important driver of future biodiversity loss. However, in some biomes climate change was identified as the major cause of biodiversity loss, including tundra and deserts and to some degree boreal forests. The overall impacts of climate change (and other drivers) on boreal forests was assessed to be higher by Sala et al. (2000), which might be due to the limitations of the present method (focusing mainly on the total size of the different biomes) but also to the assessment year (2050 ver-

sus 2100). Deposition of nitrogen has been identified as a major driver of species loss in temperate forests, warm mixed forest (particularly in Asia), and, to a lesser degree, savanna. This is consistent with the earlier assessment. Habitat loss, finally, was found to be particularly important for species loss in temperate forests, warm mixed forests, savanna, and tropical forest.

The differences among the scenarios are relatively small due to delays within the system and counteracting assumptions. The 50-year modeling window chosen for this exercise may not be enough for the different climate change scenarios to unfold fully. In addition to delays in the drivers, there are important delays in the response of biodiversity. Losses of species at the global level do not occur immediately after the loss of habitat or alteration of the environment. For example, reduction of habitat lowers the number of individuals in a population and puts this species on an extinction trajectory. However, the species extinction would not effectively occur for quite some time, depending on the life cycle of the species and the characteristics of the ecosystem.

The four MA scenarios have assumptions about effects on biodiversity that compensate each other. The compensatory nature of the different assumptions reduces the differences in biodiversity effects among scenarios. For example, one scenario assumes lower food demand and other assumes higher demand but also higher yield. The end result is that the differences in land use change and biodiversity loss between these two scenarios were relatively small. Similarly, in different scenarios the increase in pastureland is compensated by the decrease in cropland and vice versa. In general, TechnoGarden results in the least amount of pressure on biodiversity—although the difference with Adapting Mosaic is small, and mainly due to lower pressure from climate change. In contrast, the highest pressures were found for Order from Strength—in particular, for land use change and deposition.

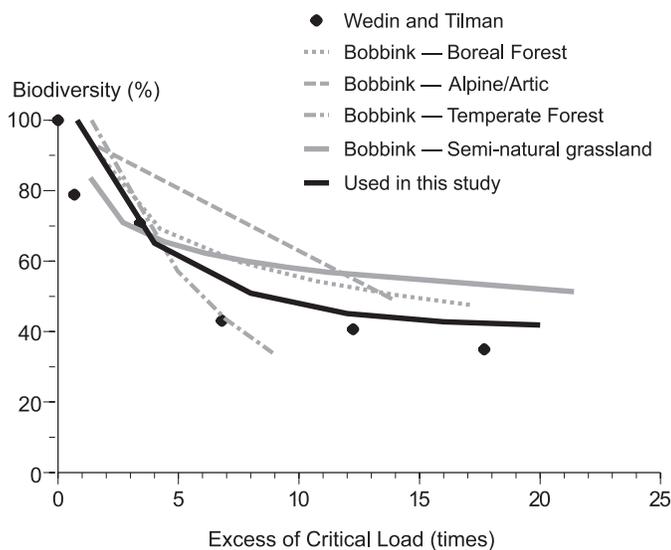


Figure 10.15. Effect of Ratio of Nitrogen Deposition to the Critical Load on Plant-Species Diversity

10.3 Freshwater Biodiversity

10.3.1 The Approach

Species-discharge curves are similar to species-area curves for terrestrial biota in the sense that richness numbers increase logarithmically with discharge (Oberdorff et al. 1995). They are subject to some of the same hypotheses to explain their occurrence, including the theory of island biogeography and the dependence of species immigration and extinction on river size and the theory on the increase types of resources and habitats, certainly including more open-water, floodplains, backwater, and high-flow habitats that only high discharges provide.

Although a positive relationship exists between riverine species richness and catchment size for fish, mussels, and aquatic invertebrates (Sepkoski and Rex 1974; Welcomme 1979; Livingstone et al. 1982; Strayer 1983; Brönmark et al. 1984; Eadie et al. 1986; Angermeier and Schlosser 1989; Hugueny 1989; Oberdorff et al. 1993), discharge and other indices of habitat volume are better predictors of species

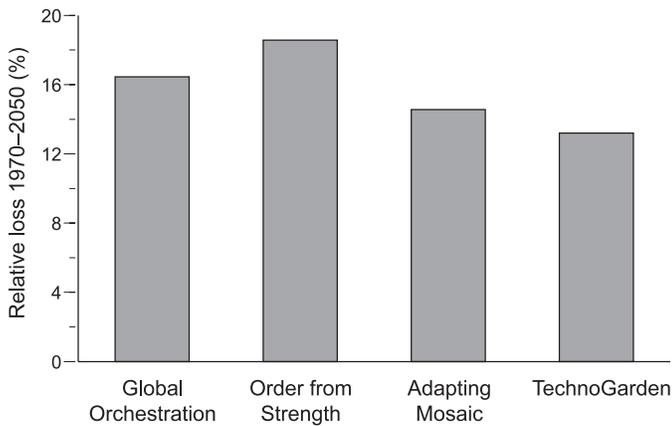


Figure 10.18. Relative Losses of Global Vascular Plant Biodiversity when Populations Reach Equilibrium with Reduced Habitat, Taking into Account Agricultural Expansion, Climate Change, and Nitrogen Deposition for Four Scenarios

richness than area (Livingstone et al. 1982; Angermeier and Schlosser 1989). Thus we address the effects of reductions in water discharge (an index of habitat type and amount) on fish species number. We address only fishes because no regional or global data sets exist for other taxa. Reductions in discharge result not only from climate change, but also from consumption of water for agriculture and other human uses.

To forecast the impact of drivers in addition to climate and water withdrawal (eutrophication/land use, acidification) on rivers and for all drivers on lakes, we rely exclusively on semi-quantitative or qualitative algorithms, such as previously published experimental or regional studies summarized for lakes by Lodge (2001) and for rivers by Poff et al. (2001). The only top driver (Sala et al. 2000; Brönmark and Hansson 2002) that we are largely unable to address is nonindigenous species; we address these only in the most qualitative way because no rigorous algorithms exist to forecast changes in the occurrence or impact of such species.

10.3.1.1 Quantitative Algorithms for Forecasting Biodiversity Loss

We used a species-discharge regression published in Oberdorff et al. (1995) to forecast loss of riverine fish. We obtained fish species numbers for 344 global rivers from Oberdorff et al. (1995) and from FishBase (www.fishbase.org). This quantitative approach should be regarded as speculative because this is the first application to forecasting of existing fish species-discharge relationships. For many rivers, fish species numbers include native and established nonindigenous species because most data sets (e.g., Oberdorff et al. 1995) did not distinguish between them. For rivers where it was possible to distinguish native from nonindigenous species, the percentage of nonindigenous species was low (< 5%) and simply added a minor amount of uncertainty to the species-discharge model. Although some human influence may be incorporated in the data (such as dams and nonindigenous species), we assume that such ef-

fects are minimal and that our species-discharge model reflects evolutionary and ecological outcomes roughly in equilibrium with natural discharge.

We used the Water Global Assessment and Prognosis model (Alcamo et al. 2003) to compute current and future discharge. The effects of climate change and water consumption are calculated separately in WaterGAP (global hydrology model and global water use model), thus allowing us to assess their independent impacts on discharge and hence on fish biodiversity. Briefly, in the global hydrology, model river discharge is computed by performing a grid-based water balance of the vegetation canopy and soil, driven by precipitation and other climate data. Water withdrawal and consumptive water use are computed in the global water use model using national estimates of domestic and industrial use in addition to estimates of irrigated areas and livestock. Additional details are available in Alcamo et al. (2003) and in Chapter 9.

We calibrated the discharge values used in the published model (Oberdorff et al. 1995) against the baseline discharge values used by WaterGAP in order to obtain a relationship that we could use to forecast future biodiversity with the WaterGAP output for the scenarios. Discharges from Oberdorff et al. (1995) and WaterGAP were highly correlated ($r^2 = 0.84$). The final species discharge regression we used was constructed for rivers located between 42° N and 42° S, the latitudinal band within which reduced discharge is predicted to occur under the scenarios (see Figure 10.19):

$$\log \text{ fish species number} = 0.4 \cdot (\log \text{ WaterGAP discharge, m}^3 \text{ s}^{-1}) + 0.6242$$

Discharge values (annual means) were obtained at the river mouth and represent discharge during the baseline climate period. This regression explained slightly more of the variance in fish species number ($r^2 = 0.57$) than the regression originally published by Oberdorff et al. (1995) ($\log \text{ fish species number} = 0.3311 \cdot (\log \text{ discharge, m}^3 \text{ s}^{-1}) + 0.83$, $r^2 = 0.52$). We assessed the relative contribution to fish biodiversity of loss of climate and water withdrawal by comparing fish species numbers based first on discharge with

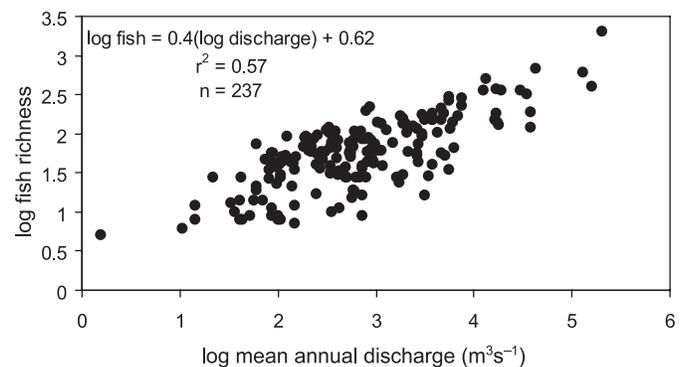


Figure 10.19 Fish Species Discharge Curve Used to Build Scenarios of Fish Loss. The regression was modeled with rivers found between 42° N and 42° S, where reduced discharge is predicted to occur.

only climate change in the scenario and then on discharge affected by both climate and human withdrawals of water. We calculated fish loss in this order because humans are more likely to manage water than climate; the incentives for a country or region to manage water withdrawal are stronger than the ones to reduce greenhouse gases. In other words, calculating fish loss in this way treats climate change as a given (because it is beyond the control of individual countries or regions) and water withdrawal as a driver that is more likely to be managed effectively.

As a consequence of this order of calculations, the forecast impact of water withdrawal on fish loss is maximized because of the nonlinear relationship of species with discharge (that is, the loss of species per unit discharge reduction increases as discharge is reduced). Forecasts of fish loss from water withdrawal include the effects of current and past water use in addition to any additional consumption that occurs in the scenarios. Confidence intervals (95%) were calculated from the critical values of the *t* distribution using the standard deviation of the predicted fish species number to which we added the uncertainty (slope * standard deviation of predicted discharge) generated from WaterGAP for the future discharge. (See Chapter 9.)

We selected for analysis the two scenarios for 2100 that produced the most fish losses (Global Orchestration) and the least (TechnoGarden), plus one intermediate scenario (Order from Strength); we also conducted fish loss analyses for 2050 on Global Orchestration and Order from Strength.

Because change in the magnitude of extreme discharge events could have strong biological consequences (Poff et al. 2001), we also tested whether the WaterGAP index of the magnitude of low flow (Q_{90} = the discharge exceeded by 90% of monthly averages) changed from the 1995 baseline. Q_{90} correlated strongly and positively with 2100 annual discharge by river basin for all scenarios ($r^2 = 0.94$). Furthermore, the slope of this relationship for each scenario did not differ from the slope for 1995. Thus, this index of flow did not change from baseline conditions in any scenario. However, other features of the hydrograph important to fishes and other aquatic biota (such as the timing and duration of low or high flows) may change under future climate and other drivers. We could not, however, assess those potential impacts in this analysis.

Even in the absence of any mitigation or conservation measures, the forecast loss of species would not, of course, occur instantaneously; rather, the expectation would be that these species would be likely to become extinct on a schedule that we cannot accurately anticipate (Minckley et al. 2003; see also *MA Current State and Trends* for a discussion about the difficulty of accurately calculating extinctions). The slow pace of many extinctions would provide time to plan and implement measures to prevent some losses in biodiversity. In North America, spacious river basins are also rich in endemics, and this pattern likely holds for other continents (Oberdorff et al. 1999). However, endemic species lists are unavailable for most rivers, especially those in the parts of the world that dry in the scenarios (see description of results that follows). Therefore, our fish richness data in-

clude both species endemic to each basin and those that occur in multiple basins.

Potential fish losses that we calculate here are thus a combination of extinctions at equilibrium for that river basin (local biodiversity losses) and global losses. In terms of ecosystem services provided by fisheries, local extinctions are a more relevant metric than global extinctions.

We believe that all plausible biases in this species-discharge approach are likely to underestimate long-term species losses at equilibrium, because this method does not account for interactions between the effects of discharge and the effects of other habitat features that will no doubt be affected simultaneously by decreases in discharge and by other drivers (eutrophication, acidification, dams, and so on), which we can only address qualitatively (see following description).

Increases in discharge (which occur for 60–70% of the world's river basins in all four scenarios) would not necessarily lead to increases in fish richness on the time scale of the scenarios, even on a local scale, because species migration from other river basins might happen only slowly or at least at rates that we cannot model in this context. At the global scale, species richness would not increase appreciably in 100 years because evolution typically happens more slowly. Thus we assumed no discharge-related change in fish species richness for river basins that experience increases in discharge, although many of these river basins will be strongly affected by other drivers (anthropogenic introductions of nonindigenous fish species). While increases in discharge may increase the production of fishes, nonindigenous species would probably be favored as a result of the habitat changes that come with increased discharge.

Overall, then, we consider these regression analyses to be a speculative guide to plausible outcomes for biodiversity of fishes. They provide a conservative index of river system-specific extirpation of fish species as a function of the drivers that affect discharge in the WaterGAP model (climate and water withdrawals), assuming steps to prevent such extinctions are not taken.

10.3.1.2 Qualitative Approaches to Forecasting Biodiversity Change

To supplement the quantitative fish species-discharge approach to riverine fish species number, we also provide qualitative analyses of the potential impact of other important drivers. We rely on qualitative scenarios of these other drivers because no quantitative algorithms exist to relate them to biodiversity. Our scenarios of impact on biodiversity, community structure, and ecosystem productivity are thus qualitative.

10.3.1.2.1 Eutrophication

We used WaterGAP's return flow (see Chapter 9) as an index of poor water quality derived from human water use. We interpreted this as an index of eutrophication in rivers, lakes, and wetlands, because human use is likely to result in increases in the nitrogen and phosphorus content of waters, especially in countries where water treatment capacities are poor or lacking. In more industrialized countries, where

water treatment exists, return flow may be a poor indicator of eutrophication, and we put less emphasis on this for those areas. In some regions, return flow is likely to be laden with industrial pollutants, but this will depend on what types of local industry exist, which is beyond the scope of the present analysis. As an index of aerial deposition of N, we use the same approach as described earlier for terrestrial ecosystems: we combine estimates of N deposition from IMAGE with spatially explicit estimates of sensitivity to N based on soils (Bouwman et al. 2002). Abundant literature (at least for selected regions) on the biological impacts of eutrophication provides a basis for assessing potential biological impacts. Because the productivity of many freshwater ecosystems is limited more by P than by N (Elser et al. 1990; see also MA *Current State and Trends*), our conclusions about eutrophication of freshwaters remain highly speculative.

10.3.1.2.2 Acidification

As an index of SO_x aerial deposition, we use the same approach as for aerial deposition of N: we combine estimates of S deposition from IMAGE with spatially explicit estimates of sensitivity to acidification based on soils (Bouwman et al. 2002; see also Chapter 9). Because acidification sensitivity as estimated globally by Bouwman et al. (2002) is consistent with more local studies on the biological impacts of acidification (e.g., Schindler et al. 1985; Brezonik et al. 1993b; Frost et al. 1995; Vinebrooke et al. 2003), these data provide a basis for assessing potential biological impacts in different scenarios.

10.3.1.2.3 Temperature

The direct effects of temperature per se on biodiversity are difficult to assess. Many previous studies, especially of fishes in lakes and streams, have illustrated that relative distributions and abundance of fish species are likely to change within a basin or within a lake as a result of temperature increases; likewise, the edges of geographic ranges of fishes will move toward higher latitudes (e.g., Lodge (2001); Rahel (2002)). More specific scenarios of the impact of temperature increases at a global scale are currently impossible and we do not attempt to construct any here.

10.3.2 Quantitative Results for Fish Biodiversity Based on River Discharge

The major patterns of discharge changes from baseline conditions are very similar across scenarios. Under all scenarios, approximately 70% of the world's rivers have increased discharge. Fish production may increase and could benefit humans. There is little basis on which to forecast consequences of increasing discharge for freshwater biodiversity. Some native species would no doubt decline as conditions change. If nonindigenous species are introduced, they would have an increased probability of success as new habitats appear to which native species are not adapted. Because of the highly uncertain consequences of increased discharge on fish and other aquatic biota, we do not consider these river basins further.

In contrast, under all scenarios, approximately 30% of the world's rivers have decreased discharge and decreased fish species diversity, resulting largely from climate change and, to a lesser extent, from increasing water withdrawal by humans. Basin-specific reductions in fish species numbers differ much more widely between basins than between scenarios. By 2050, for the 110 modeled river basins that are drying, the basin-specific percentage of fish species likely to face extinction ranges from about 1% to about 60%; for 2100, analogous values range from about 1% to about 65%. (See Figure 10.20 here and Figure 10.21 in Appendix A for 23 representative rivers.) Water withdrawal contributed little (generally an additional 1–5%) to potential fish species loss in most (~80%) rivers. In some regions, however, including India, Australia, and parts of Eastern Europe, water withdrawal was a substantial driver. In the Middle East and India especially, water withdrawal caused most of the extinctions. Considering both climate and water withdrawal, Global Orchestration resulted in the highest fish species losses overall, Order from Strength marginally lower losses, and TechnoGarden produced the fewest species losses. (Adapting Mosaic was not modeled.)

Losses of fish biodiversity were concentrated in southern Africa, northern Africa, eastern Europe and the Middle East, India, Australia, south-central South America, northern South America, and southern Central America. In many of these countries, fishes are an important indigenous source of protein, and governments and society have less capacity to cope with losses of such ecosystem services than in countries experiencing lower losses of fish biodiversity (see MA *Current State and Trends*). As documented by the IPCC, areas predicted to experience drying differ under different global climate models. However, most of the rivers predicted by WaterGAP to lose discharge are in areas predicted to dry by most general circulation models (see Figure 3.3 in the Third IPCC Assessment). Thus these patterns are robust for most areas of the world.

10.3.3 Qualitative Results for Fish Biodiversity

10.3.3.1 Eutrophication

Increases in return flows (estimated by WaterGAP) differ across scenarios (see Chapter 9), with the greatest increases over baseline conditions in Global Orchestration, followed by Order from Strength and TechnoGarden. Almost all areas with large increases in return flows are also areas with decreased discharge. This is especially true for central and southern Africa, the Middle East, India and neighboring states, Central America, and eastern and southern South America. For freshwater taxa in these areas, habitat quality will be declining simultaneously with habitat volume, as pollution by nutrients and other chemicals increases. Under Global Orchestration and Order from Strength (but less for TechnoGarden), most of these same areas experience increased atmospheric deposition of nitrogen, further increasing the potential for eutrophication in water bodies that are N-limited. Many other regions also experience increased atmospheric deposition of N. Thus even in areas of steady or increasing freshwater habitat, nitrogen enrichment is

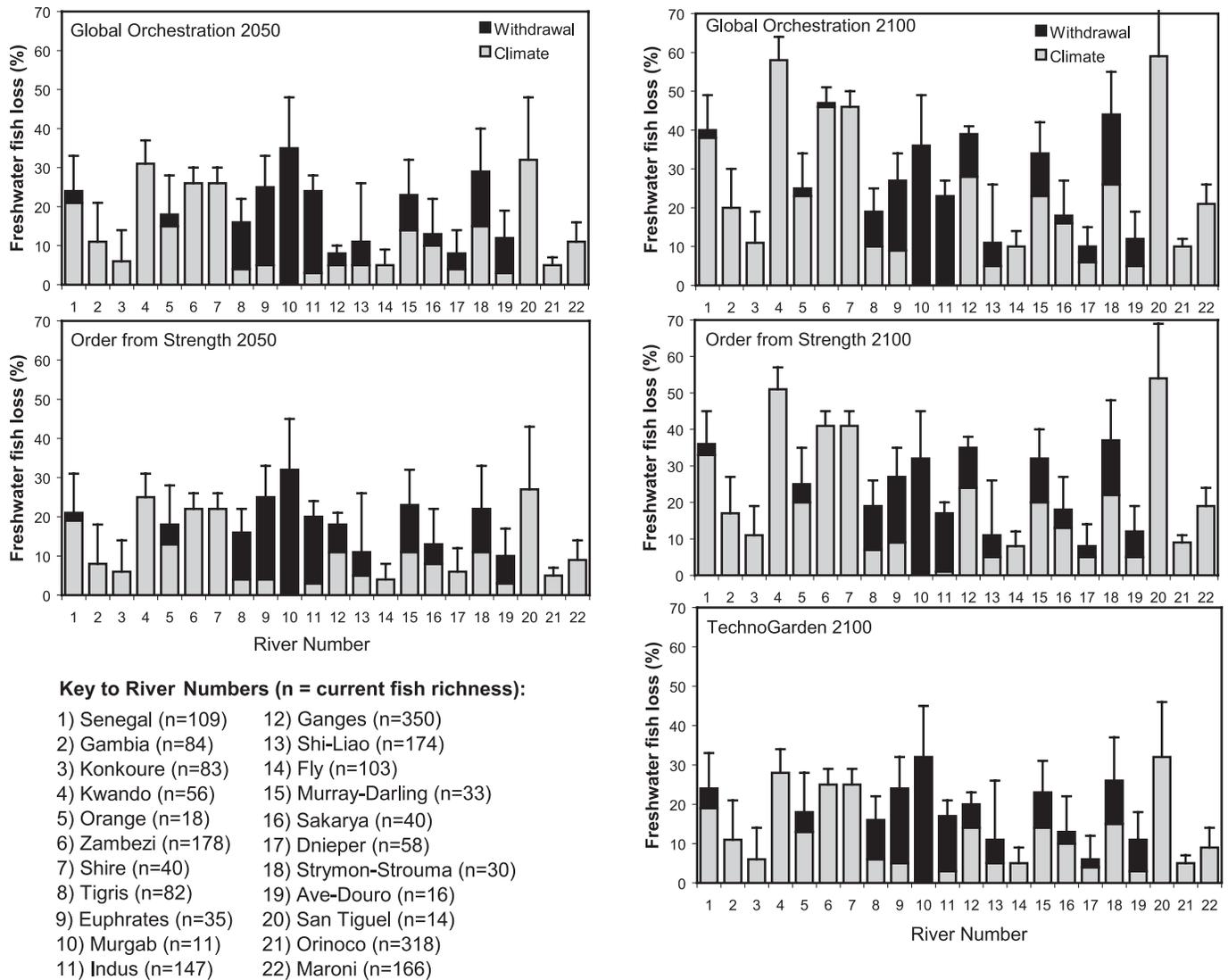


Figure 10.20. Percentage Losses of Fish Species Predicted from Decreases of River Discharge Resulting from Climate Change and Water Withdrawal by Humans for Two Scenarios in 2050 and Three Scenarios in 2100. The 22 rivers depicted are representative of the 110 modeled rivers that experience losses of discharge under the scenarios. Percentage loss is \pm 95% confidence intervals. Gray indicates discharge resulting from climate change; black indicates discharge resulting from water withdrawals by humans.

likely to reduce freshwater biodiversity and change species composition of freshwater taxa. Of the three scenarios we examined closely, only under TechnoGarden were there regions of steady or declining N deposition.

In general, for the majority of the world under both Global Orchestration and Order from Strength, the symptoms of eutrophication (including both P and N enrichment) can potentially be strong. These include increased concentration of noxious algal blooms (while decreasing total species richness of all taxa) (Schindler 1977), decreases in oxygen, water quality, and aesthetic value, and severe reduction of fish populations and species (see *MA Current States and Trends*).

10.3.3.2 Acidification

Acidification increased in some parts of the world in all three scenarios considered and was especially severe in

Global Orchestration and Order from Strength. Some regions affected by water loss and eutrophication also experienced increasing acid deposition—for instance, the Middle East in all three scenarios. Likely consequences of acidification are well established from many previous observations and experiments. We know from the acidification of many waterways in North America and Europe that substantial ecological and biological changes have occurred. Entire food webs have been affected, with most species disappearing while others increased (Vinebrooke et al. 2003). Fish are particularly vulnerable to decreases in pH and can disappear completely from acidified systems (Schindler et al. 1985; Brezonik et al. 1993a; Frost et al. 1995).

10.3.3.3 Hotspots of Freshwater Biodiversity and Ramsar Sites

According to Groombridge and Jenkins (2002), the major global hotspots for fishes are the Amazon basin and neigh-

boring parts of South America and the basins of central Africa. Although the cores of these areas do not experience drying, some of the edges of these large hotspots do, especially in northern South America. In these scenarios, the global fish hotspots did not experience large increases in eutrophication and acidification. Other freshwater taxa suffered more from one or more drivers. For example, one of the two major hotspots for crayfish, southeastern Australia, will suffer extreme drying. In contrast, two of the major hotspots of freshwater crab diversity occur in Central America/northern South America and India, where drying and water pollution increase greatly in Global Orchestration (both regions) and Order from Strength (Central America/northern South America). The major global hotspot for fairy shrimp, which inhabit wetlands, is southern Africa (Groombridge and Jenkins 2002), where drying, eutrophication, and acidification are all increasing.

Because wetland ecosystems are by definition low-volume aquatic ecosystems and are often seasonally absent under current conditions, they are particularly vulnerable to changing conditions of climate, human water use, and pollution (Revenga and Kura 2003). These scenarios present severe threats to wetlands. For example, in many drying river basins of the world, a large proportion of the basin area is currently wetland: Orinoco 15%, Parana 11%, Ganges 18%, Fly 42%, Sepik 33%, Krishna 16%, and Brahmaputra 21%. Multiple Ramsar wetlands occur in the following river basins that dry and suffer strongly from other drivers in the scenarios: Senegal (4 sites), Parana (7), Indus (10), Ganges (4), and Murray-Darling (10) (Revenga et al. 1998).

10.3.3.4 Multiple Drivers and Interactions

Data on the global distribution of freshwater biodiversity are fragmentary at best, but it is clear from existing information that some taxonomic groups are likely to continue to experience very high extinction rates as a result of combinations of drivers (Jenkins 2003). Other anthropogenic influences are likely to increase freshwater species loss above what is reflected in the species-discharge model. Increased water temperatures would further exacerbate the stress experienced by fish and other taxa (Matthews and Zimmerman 1990; Casselman 2002). Secondary infections of fishes may increase in areas of low water flow (Steedman 1991; Chappel 1995; Janovy et al. 1997).

Overfishing, particularly in poor developing countries, will continue to reduce fish populations (see MA *Current State and Trends*; Bradford and Irvine 2000; Odada et al. 2004). Xenobiotics (human-made organic chemicals) that may alter survival, reproduction, and growth for aquatic biota are forecast to increase (Brönmark and Hansson 2002). Salinity can increase in highly irrigated rivers and lakes, with subsequent negative effects on many freshwater taxa (Williams 2001). Dams may significantly reduce populations of migratory fish (see MA *Current State and Trends*) and negatively affect species richness in general (Cumming in press). Additional dams are planned for some rivers that are forecast to lose discharge and thus potentially lose fish, which may further reduce fishes. For example, there are six new dams planned in the Tigris and Euphrates basins, seven

in the Ganges Basin, and two in the Orange River (Revenga et al. 1998). These river basins all have high numbers of endemic fish (see MA *Current State and Trends*).

All these changes and their interactions are likely to decrease habitat for native species and favor the survival of any nonindigenous species that are introduced (Kolar and Lodge 2000). And the likelihood of introduction of nonindigenous species will increase where human population and trade increases (Levine and D'Antonio 2003). Trade differs among scenarios (see later section), but history suggests in general that we should expect increases in freshwater nonindigenous species in all scenarios in the absence of new prevention and control efforts. The fish fauna of the United States, for example, has been largely homogenized over the last decades (Rahel 2000), while dams and other anthropogenic drivers increase the occurrence and impact of nonindigenous species (Marchetti and Moyle 2001). Thus without major efforts to prevent the introduction of nonindigenous species, and without additional conservation efforts to reduce the impact of other drivers, much greater declines in freshwater biodiversity than those implied by our quantitative and qualitative models are likely to result from multiple, interactive, cumulative, and long-term effects in rivers that experience decreasing or increasing discharge.

10.3.3.5 Aggregated Effects of Drivers and Human Well-being

In all the scenarios, but especially in Global Orchestration and Order from Strength, freshwater ecosystems in some parts of the world changed in major ways. In many parts of the world, declining water quantity and quality occurred simultaneously in all scenarios. While this caused human hardship directly, it also caused large losses of ecosystem services in the form of harvest of freshwater fishes, fiber from freshwater wetlands, and other freshwater taxa. The most negative combinations of drivers coincide with geopolitical regions where the capacity of governments and society to cope with the loss of biodiversity and ecosystem services is low. For example, GDP per capita (CIA 2003) in drying countries is about 20% lower than in countries that do not get dryer in the scenarios.

10.4 Marine Biodiversity

10.4.1 The Approach

A global ecological model of marine systems does not exist yet, but there are more than 100 ecosystem models of various ecosystems throughout the world based on the Ecopath with Ecosim software, which is described in Chapter 6. Ecopath with Ecosim uses a combination of trophic levels (functional groups) and species to describe the ecosystems rather than complex webs of individual species, which limits how well it can describe "biodiversity" changes. However, changes in the diversity of the system based on the various functional groups can be described using Kempton's Q for the biomass of groups with a trophic level of 3 or more. Three models—the Gulf of Thailand (shallow coastal shelf system), Benguela Current (coastal upwelling system),

and the Central North Pacific (pelagic system)—were modeled using the four MA scenarios. These three systems are described in detail in Chapter 6, while changes in their biomass diversity are described here.

Kempton's Q is a relative index of biomass diversity and is based on a modified version of Kempton's Q75 index originally developed for expressing species diversity (Kempton 2002). The index is estimated as $Q75 = S / (2 \log(N0.25 \cdot S / N0.75 \cdot S))$, where S is the number of species (here functional groups) and N times $i \cdot S$ is the number of individuals (here biomass) in the sample of the ($i \cdot S$) most common species (or of a weighted average of the species closest if $i \cdot S$ is not an integer). The Q75 index thus describes the slope of the cumulative species-abundance curve between the lowest and highest quartiles. A sample with high diversity will have a low slope, so an increase in diversity will manifest itself through a lower Q75 index. To reverse this relationship, and to make the Q75 index relative to the baseline run in the ECOPATH with ECOSIM simulations, we expressed the biodiversity index as $(2 - Q_{run} / Q_{baseline})$, truncating the index at zero in the unlikely case that the Q75 index should more than double.

The Q75 index and the inverse diversity index are sensitive to the number of species (functional groups), and have merit mainly for expressing relative changes for a given model or for models with the same group structure. To focus the index on the exploited part of the ecosystem—that is, the part for which there is the most information and where human impact is most likely to be seen—the analysis is limited to groups with a trophic level of 3 or more. This excluded from the index the primary producers and groups that are primarily herbivores or detritivores (such as zooplankton and most benthos groups). In the analysis here, a high biomass diversity index refers to an ecosystem of greater evenness (that is, even distribution of biomass among a number of species), and a low index value refers to an ecosystem where one or two species are much more abundant among a small number of species.

The number of studies focused on the future of marine biodiversity is limited compared with ones on terrestrial systems. Field et al. (2002) examined the future of the world's oceans to 2020, but biodiversity was not a significant focus of their study. Culotta (1994) noted that studies of marine biodiversity were in their infancy. Progress has been made since then, albeit slowly, especially in less accessible environments and in assessing future changes in ecosystems such as the deep sea (Glover and Smith 2003), polar seas (Clark and Harris 2003), and vents and seamounts (Koslow et al. 2001). Potential climate change impacts on coastal marine biodiversity have also been the focus of recent studies (e.g., Kennedy et al. 2002).

However, none of these studies has quantitatively examined changes in biodiversity; they have all been qualitative, based on projections of current trends. The lack of quantitative methods for examining changes in marine biodiversity at the ecosystem scale is limited by methods and robust, broad-scale information. The species-area curves and other species-specific methods are of limited use in marine ecosystems because species extinctions are rarely observed and

because information at the species level is not available for many systems.

10.4.2 Marine Biodiversity Change across Scenarios

10.4.2.1 The Gulf of Thailand Model

The results of the modeling of the Gulf of Thailand using the four scenarios are illustrated in Figure 10.22. By 2050, all four scenarios approached a similar Kempton Q index (biomass diversity index) that was less than the value for 2000. Initially, in the TechnoGarden scenario the biomass diversity index declined slightly as the Gulf of Thailand ecosystem was managed so that the profits from the high-value fisheries were increased. This resulted in the number of species and the biomass of some species declining more

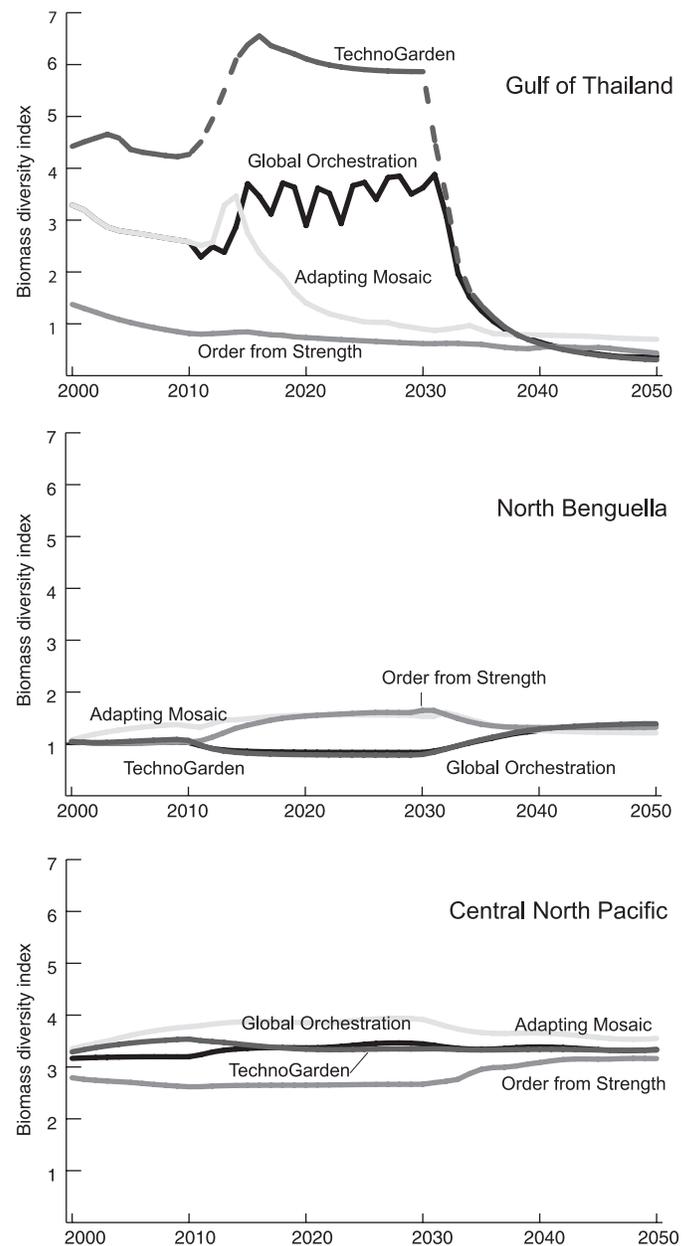


Figure 10.22. Changes in Biomass Diversity by 2050 in Four MA Scenarios in Three Specific Regions. Diversity is for groups with a trophic level equal to or higher than three.

relative to other species. In 2010, the policy shifted to rebuilding the ecosystem, which accounted for the increasing biomass diversity index as the number of species increased and the biomass was more evenly distributed among these species. This resulted in a more stable ecosystem structure. As technology developed and was able to provide some ecosystem services, the policy focus turned to providing fish that were used to produce fishmeal for the aquaculture sector, which had taken over the role of food provisioning. This resulted in the biomass diversity index declining rapidly when the ecosystem changed from an even distribution of species and associated biomass to an ecosystem dominated by a few functional groups.

The Gulf of Thailand responded to the Global Orchestration scenario in a similar manner to the TechnoGarden, but the changes in the biomass diversity index (number of species and abundance) were relatively lower. The Adapting Mosaic scenario also commenced with a focus on increasing the profits of high-value fisheries and as a consequence the biomass diversity index declined slowly, with some species and species abundance within functional groups declining. In 2010, attempts to rebuild the ecosystem experiencing climate change within a policy that still had increasing profits as a priority were of limited success since the biomass diversity increase was much less than in the TechnoGarden and Global Orchestration scenarios. In this case, changes in the number of species and abundance were small relative to the above scenarios. Despite policies that included rebuilding selected high-value stocks, the biomass diversity index declined rapidly to a system with a few functional groups dominating the ecosystems. The Order from Strength scenario, which had a focus on increasing the value of the fisheries throughout the 50 years of the scenario, resulted in a steady decline in the biomass diversity index.

The results of the four scenarios suggest that it is (moderately) likely that the Gulf of Thailand would lose biomass diversity—that is, the number of species will decline. For those species that remain, a few of them will dominate (in abundance) the ecosystem.

10.4.2.2 *Benguela Current Model*

The results of the modeling of the North Benguela Current system using the four scenarios are also presented in Figure 10.22. The biomass diversity index for the four scenarios changed very little over the 50 years of modeled results. In Adapting Mosaic, the biomass diversity index increased consistently, as the management policy was a mix of increasing profits from food fisheries and maintaining employment opportunities. This could be achieved if a number of fisheries were maintained and fisheries that employed a number of fishers were also maintained. In the North Benguela ecosystem, this resulted in a diversity of fisheries, with the abundance of species more evenly distributed as well as the number of species, but not necessarily increasing (biomass diversity index relatively stable).

In the Order from Strength scenario, the focus was also on maintaining profits and employment opportunities; however, in this scenario high-value fisheries were targeted initially and therefore there was little change in the biomass

diversity index in the first 10 years. In 2010, ecosystem rebuilding commenced, and in response the biomass diversity index also increased to a level similar to the Adapting Mosaic scenario. This was not surprising, since this phase of the scenario also tried to maintain employment opportunities and value. In this case, it was likely the species that were maintained were not identical to those in the Adapting Mosaic scenario, but biomass abundance was similar.

The TechnoGarden and Global Orchestration scenarios behaved similarly in this ecosystem: both scenarios had food security as a component as well as maintaining or increasing the value of the fisheries. Neither scenario had a focus on ecosystems rebuilding except Global Orchestration after 2030, which accounted for the increasing biomass diversity index. The increasing biomass diversity index in the TechnoGarden scenario was due to a policy shift to maintain fisheries for fishmeal, which encompassed many of the species found in the Benguela system.

The differences between scenarios for the North Benguela ecosystem were not as dramatic as in the Gulf of Thailand, possibly due to different ecosystems and different states. The Benguela and Central North Pacific ecosystems were not as disturbed as the Gulf of Thailand.

10.4.2.3 *The Central North Pacific Model*

In the North Pacific ecosystem, current diversity was maintained (*low to moderate certainty*) under the four scenarios (see Figure 10.22). However, if increasing value or employment opportunities were the policy imperatives, biomass diversity could not be sustained. In the Central North Pacific, the Order from Strength scenario did not maintain biomass diversity compared with the other three scenarios. The biomass diversity index in the Order through Strength scenario declined for the first 20 years as the value of the fisheries increased. The slight decline in evenness reflected the slight lowering of biomass abundance for some species as distant water fleets continued to fish in the area. After 2010, drift net fishing resumed as companies continued to improve the value of the fisheries. This resulted in a lowered biomass diversity index until around 2040, when drift net fishing was banned again. This was reflected in an increasing biomass diversity index as more fisheries are sustained.

The Adapting Mosaic scenario, in the Central North Pacific, governments closed the turtle fisheries and focused on increasing the value of the tuna fisheries, which resulted in a slight increase in the biomass diversity index as species abundance increased in evenness until 2030. After 2030, the focus was on rebuilding bigeye tuna stocks, one of the most valuable species, and therefore increasing the value of the fisheries. This resulted in declining biomass diversity index as species abundance for some species decreased. The biomass diversity index was maintained in the TechnoGarden and Global Orchestration scenarios despite the initial divergence between them.

The biomass diversity index in TechnoGarden increased initially as the status quo was maintained and the biomass abundance was distributed more evenly among the species. After 2010, fishing costs declined as technology overcame

several cost issues associated with catching the fish. Consequently, the biomass diversity index declined, with some species declining in abundance. Ocean ranching continued to develop in the open sea, resulting in younger tuna being caught for this sector. This resulted in a minimal change in the biodiversity diversity index until around 2030, when the technology for hatchery production of valuable species such as tuna was developed. However, this change in target species did not affect the biomass diversity index since the species abundance did not change overall but instead was redistributed in different functional groups.

The Adapting Mosaic scenario also maintained the Central North Pacific's diversity with little change over the 50 years of modeled results. The focus throughout this scenario was the continued optimizing of the value of the fisheries, which was dominated by large pelagic species. This modeling suggested that waiting until 2040 to rebuild bigeye stocks would have had no impact on the biomass diversity.

It was likely that biomass diversity could have been maintained in the Central North Pacific across all four scenarios; however, diversity was not improved unless the management imperatives for increasing the value of fisheries were substantially reduced.

10.5 Invasive Species across Biomes and Scenarios

Human-caused invasions by non-native species are a major cause of reductions in native biodiversity and consequent changes in ecosystem structure, ecosystem services, and human well-being (Mack et al. 2000; see also *MA Current State and Trends*, Chapters 4, 5, and 23). Although quantitative evidence for invasion-caused extinctions is poor for some ecosystems (Gurevitch and Padilla 2004), the dramatic impact of invasions on ecosystem functioning and ecosystem goods and services is well established (Mack et al. 2000). This section briefly explains the scale of the current problem, the major impacts of anthropogenic species introductions, and the worrying outlook for the impacts of non-native species on ecosystems.

While most non-native species do not cause harm, a small proportion do (harmful non-native species are termed "invasive") (Mack et al. 2000). Nonnative species are pervasive; the number of species introductions being discovered is accelerating over time (Cohen and Newman 1991; Ricciardi 2001). Anthropogenic species introductions with severe effects on ecosystems have occurred in terrestrial, freshwater, and marine systems (Mack 2000). Approximately half of the currently threatened or endangered species in the United States are affected by invasive species (Wilcove and Chen 1998), and some of the largest vertebrate extinctions in recent times have been driven in large part by invasives (Witte et al. 2000). A conservative estimate of the annual cost of invasive species to the U.S. economy is \$137 billion (Pimentel et al. 2000). While intended introductions in agriculture, aquaculture, and other sectors can have great net benefits to society (for example, most crop plants in most parts of the world are non-native), the

number of unintended and harmful introductions is increasing as trade increases (Levine and D'Antonio 2003; Drake and Lodge 2004).

Species are transported intentionally and unintentionally into every country by every conceivable conveyance. Most nations have few safeguards to prevent the escape of non-native species into natural environments (Invasive Species Advisory Committee 2003). Virtually no economic incentive structure exists for limiting species invasions because the unintended costs of harmful invasions are usually borne by all citizens while the benefits of importation of species are concentrated in commercial interests (Perrings et al. 2002).

Unless steps are taken to reduce the unintended transport of species, the numbers of non-native species established in most countries are expected to continue to increase with increasing trade (Levine and D'Antonio 2003; Drake and Lodge 2004). For example, because of extensive trade between northern Europe and the North American Great Lakes region over the last two to three centuries, a high proportion of the roughly 100 non-native species in the Baltic Sea are native to the Great Lakes, and 75% of the recent arrivals of the about 170 non-native species in the Great Lakes derive from the Baltic Sea (some by way of earlier trade between the Ponto-Caspian region and the Baltic region) (Ricciardi and MacIsaac 2000). In recent decades, this species exchange has been driven largely by the unintentional release of organisms from the ballast tanks of ships.

Nonnative species are also introduced deliberately into many ecosystems to enhance food production, provide aesthetic services, or reduce disease (Mack and Lonsdale 2001). Such introductions may involve either the direct release of a non-native species into the wild or its secondary release from a captive environment. For example, the Nile perch was introduced into Lake Victoria as a food species; African grasses have been introduced to many parts of the United States and Latin America as forage for cattle; pigs and goats introduced for food and milk have a history of causing erosion and ecological degradation on islands; rabbits and cats introduced for aesthetic purposes and pest control, respectively, have reduced biodiversity in Australia; black wattle trees (*Acacia mearnsii*) were originally introduced to southern Africa for use in the leather industry and have now become a target for biocontrol using gall-forming wasps introduced from Australia; and releases of unwanted aquarium specimens have been the source of numerous harmful introductions of invasive species into freshwater systems, including water hyacinth (Padilla and Williams 2004).

Introductions of species that are perceived as economically valuable and ecologically benign are likely to increase with increasing globalization—for example, as farmers become more aware of food production methods in other countries and consumer demand for a greater variety of produce increases. Such introductions carry a substantial risk. Although many non-native species introductions are not harmful, the impacts of the small percentage of non-native species that become invasive may be severe in both

environmental and financial terms (Leung et al. 2002; Pimentel 2002).

With even the most conservative forecast of the relationship between trade and species introductions, it is likely that between 2000 and 2020 some 115 new insect species and five new plant pathogen species will become established in the United States (Levine and D'Antonio 2003). Experiences in North America with the introduction and spread of chestnut blight, Dutch elm disease, and, more recently, sudden oak death (*Phytophthora ramorum*) illustrate how ecologically and economically damaging single-plant pathogen species can be. Early in the twentieth century, for example, the American chestnut was the dominant overstory tree in the deciduous forests of the North American Appalachian Mountains. Not only was it ecologically important, but it provided many large ecosystem services, especially nuts and lumber. Since the mid-twentieth century, those ecosystem services completely disappeared and were replaced only in part, if at all. (See MA *Current State and Trends*, Chapter 4.) It is now possible that sudden oak death could have a similar effect on many oak species in North America.

As commerce develops or grows between countries not previously linked strongly by trade, especially those with similar terrestrial or aquatic climates, whole new sets of species will become established. For example, imports from China into the United States have increased about sixfold over the last decade; over this period a subset of the Chinese species discovered in the United States have become very damaging, including snakehead fishes (*Channa spp.*), the Asian longhorned beetle (*Anoplophora glabripennis*), and the emerald ash borer beetle (*Agrilus planipennis*). Thus, scenarios about the impact of changes in trade need to consider not just changing volumes of trade, but also which countries are linked by new trading patterns.

While the MA scenarios do not specify trading partners or quantify levels of trade, they do allow the assessment of differences in overall trade volume based on calculations of GDP and openness to trade in the scenario storylines. (See Table 10.5.) Global Orchestration and TechnoGarden both assume relatively rapid income growth and further globalization leading to a strong increase in global trade. This increase will be greatest under Global Orchestration. The TechnoGarden scenario also emphasizes increased use of

GMOs and other forms of human intervention in ecosystems, which could increase the risks of escape of intentionally introduced species into natural ecosystems. On the other hand, the higher awareness of ecosystem functioning under TechnoGarden might be reflected in tighter management of non-native species.

According to the storylines, global trade will be much smaller under Order from Strength because of lower growth rates and protective trade policies. In Adapting Mosaic, trade levels will be lower than in Global Orchestration and TechnoGarden, but probably higher than in Order from Strength. Nevertheless, global trade levels in both cases could still be similar to those of today as a consequence of net expansion of the global economy. Assuming that the positive relationship between imports and non-native species that exists for the United States (Levine and D'Antonio 2003) applies globally, invasive species would cause the severest ecological changes and losses of ecosystem services in Global Orchestration, followed in order by TechnoGarden, Adapting Mosaic, and Order from Strength. Effects of invasive species could, in fact, play a much larger role in environmental changes than other drivers (such as climate, land use, or water consumption) that we have been able to assess more quantitatively. They will certainly interact strongly with other drivers (Mooney and Hobbs 2000).

10.6 Opportunities for Intervention

This assessment focused on losses of biodiversity at local and global scales. These two scales represent different opportunities for intervention. Local species losses can be reverted by a series of active management actions that range from abandonment and natural colonization to artificially increasing immigration rates. Ecosystem services provided by species in the original ecosystem would be restored with different delays, depending on the ecosystem service and the ecosystem.

Global extinctions are irreversible, and no human action can reverse this loss for future generations. However, major time lags occur between a reduction in habitat availability (as described in the Introduction) and the global extinction of species. Changes in habitat availability beyond a certain threshold may reduce the size of populations to a point that in a number of generations they would not be able to sustain themselves. These lags provide a wonderful opportunity for policy-makers to react and deploy actions that may reverse the trend that could have led to the global extinction of species. Many actions may change the trajectory of a species that was bound to become extinct in a few generations, including the location of protected areas, establishing corridors connecting small patches, and other steps that fall within the realm of the discipline of habitat restoration.

10.7 Ecosystem Services Derived from Biodiversity

The study of the effects of biodiversity on the functioning of ecosystems and their ability to provide goods and services has recently attracted a lot of attention from theoreticians

Table 10.5. Scenario Characteristics Relevant to Estimating Relative Magnitude of International Trade

Characteristic	Global Orchestration	Order from Strength	Adapting Mosaic	TechnoGarden
GDP (annual growth rate of real total GDP)	3.5%	2.0%	2.6%	3.0%
Economic openness	++	--	-	++
Rank of magnitude of species transport	1	4	3	2

and experimentalists. Ecologists predict a negative relationship between decreasing biodiversity and functioning of ecosystems and provisioning of services (Chapin et al. 2000). The exact shape of this relationship depends on the ecosystem process and service as well as the order in which species are lost or added (Mikkelsen 1993; Sala et al. 1996; Petchey et al. 1999; Petchey and Gaston 2002).

Two kinds of ecosystem services can be identified, depending on their dependence or not on the abundance of individuals. This classification of ecosystem services has important consequences for the relationship with biodiversity and has broad policy implications. Type-I ecosystem services depend on the abundance of individuals and include provisioning services such as food and fiber production, regulating services such as air quality maintenance and erosion control, and cultural services such as aesthetic values. Biodiversity affects the rate of these ecosystem services at a local scale, and the provisioning depends on the abundance of each species.

Type-I ecosystem services are related to the disappearance of species at the local scale and the extirpation of populations. A decline in habitat availability and in the presence of a species results in a proportional decline in the service. For example, a 50% decline in the abundance of a fruit-tree species determines a proportional decline in the provisioning of that food type. Biodiversity declines affect the provisioning of this kind of ecosystem service before global extinctions occur. Habitat loss and local extinctions provide good estimates of the loss of this type of ecosystem services. Another important characteristic of Type-I ecosystem services is that changes in their availability are reversible. A reduction in the abundance of one species and the services that it provided could be reversed as a result of reduced pressure or active conservation practices.

Type-II ecosystem services are independent of the abundance of individuals of a given species. This service type includes the provisioning of genetic resources, which are the basis for animal and plant breeding and biotechnology. Another example of Type-II ecosystem service is the provisioning of pharmaceuticals that modern medicine depends on heavily. The service is provided by the unique genetic combination resident in native populations and not by the number of copies of this combination. The availability of Type-II services is affected by global extinctions because local extinctions do not affect the availability of the genetic code. Consequently, changes in the provisioning of type-II ecosystem services are completely irreversible. Species that become globally extinct are lost forever. Global losses of species also result in the irreversible loss of the ecosystem service that the species was providing or was going to provide. Global extinctions are the best way of estimating losses in the provisioning of Type-II ecosystem services.

In this assessment, we discussed provisioning of Type-I ecosystem services earlier in the chapter when we analyzed patterns of habitat loss and local extinctions in the different MA scenarios. The loss of this type of ecosystem services was estimated as directly proportional to the loss of habitat. Type-II ecosystem services were covered when we discussed global extinctions in the different scenarios. Global

extinctions are important beyond the Type-II ecosystem services that they affect.

Empirical support of the relationship between biodiversity and ecosystem functioning and provisioning of ecosystem services is currently lagging behind the development of models. The first group of experiments on this topic focused on aboveground primary production and plant-species diversity. Large-scale manipulative experiments using grassland species in different regions of the world showed a similar pattern, with the first species losses resulting in small decreases in primary production while further reductions in species diversity resulted in an accelerated decrease in production (Tilman et al. 1996, 1997; Hector et al. 1999). Empirical evidence of the effects of biodiversity on other services and for other ecosystem types is still not available.

A number of possible functional forms have been suggested for the relationships that couple biological diversity to the rate with which different types of ecosystem processes are undertaken (Sala et al. 1996; Tilman et al. 1996; Kinzig et al. 2001). Central to all of these is the argument that there is some asymptotic maximum rate at which the activity is undertaken that declines to zero as species diversity and abundance are reduced (Mikkelsen 1993; Tilman et al. 1997; Loreau 1998; Crawley et al. 1999; Loreau et al. 2001). (See Figure 10.23.) The shape of this relationship depends on the service under consideration.

The allocation of ecosystem services to these different shapes of the relationship depends on two basic assump-

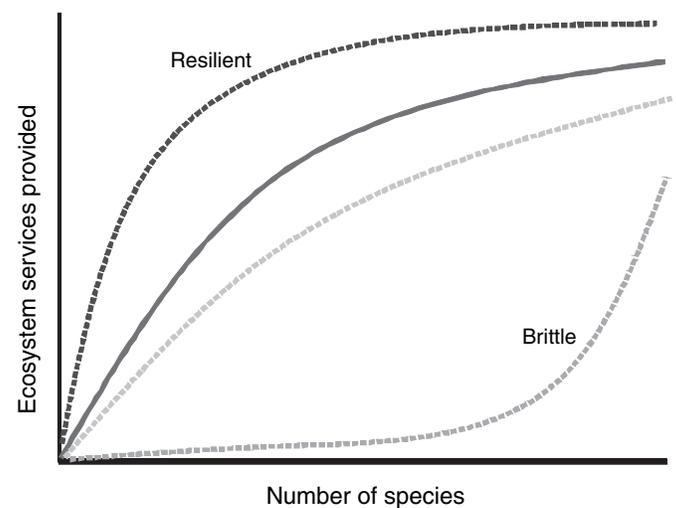


Figure 10.23. Relationship between Species Abundance and Ecosystem Function for Resilient and Brittle Ecosystem Services. In some cases decline may be rapid as the abundance of the species undertaking the activity declines (for example, population regulation of herbivores by top carnivores). In others, there may be considerable redundancy, and the relative efficiency with which any function is undertaken declines only slowly with loss of species diversity and abundance declines. Arguably, this is the case for nutrient cycling and water cleansing, though it is worth noting that the net amount of nutrients and water processed will remain dependent upon the net area (and quality) of land available.

tions: species at different trophic levels perform different ecosystem services, and species at higher trophic levels will be lost more rapidly than those at lower trophic levels. A number of examples of faunal collapse support our contention that species at higher trophic levels are lost more rapidly than those at lower trophic levels. (See Figure 10.24.) The classic studies of John Terborgh and colleagues at Lago Guri illustrate that the loss of top predators first and then mesopredators in fragmented natural systems lead to cascading effects that sequentially disrupt regulatory processes at lower trophic levels (Terborgh et al. 1997). Recent studies of a variety of organisms also suggest that species on higher trophic levels are more sensitive to climate-induced perturbations (Voight et al. 2003).

Each of these cases provides an example of faunal collapse following either a change in habitat quality or in response to exploitation that has removed species from the higher trophic levels. Although there are no studies that explicitly explore the change in trophic structure as a habitat is fragmented and reduced in size, a number of studies suggest that food-chain length is a function of habitat size (Cohen and Newman 1991; Post et al. 2000; Post 2002) and that species at higher trophic levels have steeper slopes in their species-area curves (Holt et al. 1999). All these empirical studies suggest that declines in habitat quality or quantity will lead to decreases in the length of food chains and hence a more rapid loss of services provided by species at higher trophic levels.

Using expert opinion, we assigned the different MA ecosystem services as belonging to one of the many different shapes of the biodiversity-ecosystem service relationship. Higher numbers reflect services that are brittle and are mostly performed by species in upper trophic levels. Lower numbers indicate ecosystem services that are quite resilient and performed by species in lower trophic levels. The central criterion to allocate services to a particular shape was that functions undertaken by species at higher trophic levels are more brittle than those at lower ones. (See Table 10.6.)

Different ecosystem services tend to be undertaken by species at different trophic levels. While top carnivores such as tigers and wolves provide a heightened spiritual quality to ecosystems, species such as nematodes, mites, beetles, fungi, and bacteria undertake many of the processes that cleanse air and water. At intermediate trophic levels, autotrophs (plants) provide not only structure and buffering against erosion but also most of the nutrients that are then passed up the food chain by primary and secondary consumers. Because species at the top of the food chain tend to be lost from declining habitats before those lower in the food web, it is likely that ecosystem services supplied by these species will be lost before those supplied by species at the base of the food chain. We would thus expect to see an initial sequential reduction in economic goods and services as natural systems are degraded, followed by a more rapid sequential collapse of goods and services. This implies that the sequence of ecosystem service loss is likely to be pre-

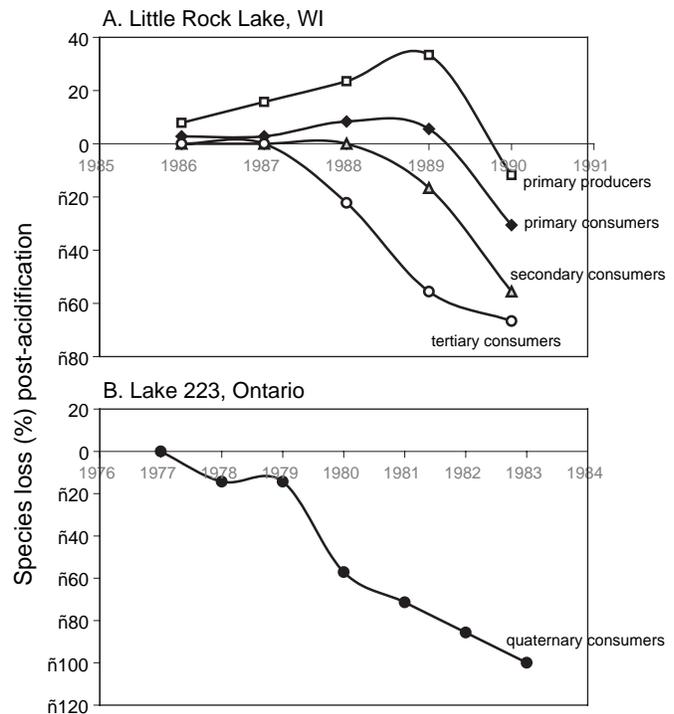


Figure 10.24. Annual Species Loss in Response to Gradual Experimental Acidification in Two North Temperate Lakes. Loss is measured as percent of pre-acidification species number. A) Four lower trophic levels in Little Rock Lake, Wisconsin, United States: primary producers (initial $N = 51$ phytoplankton species); primary consumers (initial $N = 36$ primarily herbivorous zooplankton species); secondary consumers (initial $N = 9$ omnivorous zooplankton species); and tertiary consumers (initial $N = 9$ primary carnivorous zooplankton species). B) Quaternary consumers in Lake 223, Ontario, Canada: (initial $N = 7$ fish species). For A, initial $pH = 5.59$, final $pH = 4.75$; unavailable for B. For B, the cessation of recruitment (absence of young-of-the-year) was treated as species extirpation. Additional experimental details are available. Additional experimental details are available for A) in Brezonik et al. (1993) and B) in Schindler et al. (1985).

dictable; the relative position of the thresholds at which services breakdown requires further attention.

This chapter has distinguished between two types of biodiversity losses: extirpations of local populations (local extinctions) and global extinctions. We concluded with *high certainty* that under all scenarios in the near future there will be important losses of habitat and consequently losses of local populations. Global extinctions will occur at uncertain times because of lags between environmental change and the occurrence of global extinctions. This last section of the chapter links the two types of biodiversity losses with two types of ecosystem services. Furthermore, we suggested a relationship between the rate of decrease of an ecosystem service with species losses and the location in the trophic hierarchy of the species associated with that particular service.

Table 10.6. Response of Different Ecosystem Services in Different Ecosystem Types to Changes in Biodiversity. Responsiveness is described in an arbitrary scale of 1–5, with higher values describing services and ecosystems that are performed by species in upper trophic levels and therefore are brittle in comparison with services and ecosystems with lower values that are performed by species in lower trophic levels and are resilient.

Ecosystem Service	Urban Systems	Cultivated	Drylands	Forest and Woodlands	Coastal	Inland Water Systems	Island	Mountain	Polar	Marine
Provisioning										
Food	1	1	1	5	1	5	1	1	5	5
Biochemicals and pharmaceuticals	0	1	3	3	5	3	3	3	5	5
Genetic resources	0	5	3	3	5	3	3	3	3	3
Fuelwood	1	5	1	1	5	0	1	1	5	5
Fiber	1	1	1	1	1	5	1	1	5	1
Ornamental resources	0	1	5	5	5	5	5	5	3	5
Fresh water	1	5	1	1	0	3	1	1	1	0
Regulating										
Air quality	2	1	1	1	1	1	1	1	1	1
Climate regulation	3	1	1	1	1	1	1	1	1	1
Erosion control	3	1	1	1	5	5	1	1	1	0
Water purification and waste treatment	3	1	2	2	5	1	3	3	1	1
Regulation of human diseases	5	5	2	3	0	4	3	3	1	1
Biological control	4	5	4	4	5	5	3	3	1	5
Detoxification	3	1	3	3	5	1	3	3	5	1
Storm protection	1	1	1	3	5	3	1	1	1	0
Cultural										
Cultural diversity and identity	3	1	4	4	3	5	5	5	3	3
Recreation and ecotourism	5	1	4	4	3	5	5	5	3	3
Supporting										
Primary production	1	1	1	1	1	1	1	1	1	1
O ₂ production	1	1	1	1	1	1	1	1	1	1
Pollination	3	5	3	3	1	0	3	3	5	0
Soil formation and retention	1	1	1	1	5	1	1	1	1	0
Nutrient cycling	3	5	3	3	3	1	3	3	5	1
Provision of habitat	4	5	3	3	5	4	3	3	1	5

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