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# Ecosystems
# and Their Goods and Services

HABIBA GITAY (AUSTRALIA), SANDRA BROWN (USA),
WILLIAM EASTERLING (USA), AND BUBU JALLOW (THE GAMBIA)

Lead Authors:
*J. Antle (USA), M. Apps (Canada), R. Beamish (Canada), T. Chapin (USA), W. Cramer (Germany), J. Frangi (Argentina), J. Laine (Finland), Lin Erda (China), J. Magnuson (USA), I. Noble (Australia), J. Price (USA), T. Prowse (Canada), T. Root (USA), E. Schulze (Germany), O. Sirotenko (Russia), B. Sohngen (USA), J. Soussana (France)*

Contributing Authors:
*H. Bugmann (Switzerland), C. Egorov (Russia), M. Finlayson (Australia), R. Fleming (Canada), W. Fraser (USA), L. Hahn (USA), K. Hall (USA), M. Howden (Australia), M. Hutchins (USA), J. Ingram (UK), Ju Hui (China), G. Masters (UK), P. Megonigal (USA), J. Morgan (USA), N. Myers (UK), R. Neilson (USA), S. Page (UK), C. Parmesan (USA), J. Rieley (UK), N. Roulet (Canada), G. Takle (USA), J. van Minnen (The Netherlands), D. Williams (Canada), T. Williamson (Canada), K. Wilson (USA)*

Review Editors:
*A. Fischlin (Switzerland) and S. Diaz (Argentina)*

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# EXECUTIVE SUMMARY

Ecosystems are subject to many pressures (e.g., land-use change, resource demands, population changes); their extent and pattern of distribution is changing, and landscapes are becoming more fragmented. Climate change constitutes an additional pressure that could change or endanger ecosystems and the many goods and services they provide.

There now is a substantial core of observational and experimental studies demonstrating the link between climate and biological or physical processes in ecosystems (e.g., shifting range boundaries, flowering time or migration times, ice break-up on streams and rivers), most evident in high latitudes. Recent modeling studies continue to show the potential for significant disruption of ecosystems under climate change. Further development of simple correlative models that were available at the time of the Second Assessment Report (SAR) point to areas where ecosystem disruption and the potential for ecosystem migration are high. Observational data and newer dynamic vegetation models linked to transient climate models are refining the projections. However, the precise outcomes depend on processes that are too subtle to be fully captured by current models.

At the time of the SAR, the interaction between elevated carbon dioxide ($CO_2$), increasing temperatures, and soil moisture changes suggested a possible increase in plant productivity through increased water-use efficiency (WUE). Recent results suggest that the gains might be small under field conditions and could be further reduced by human management activities. Many ecosystems are sensitive to the frequency of El Niño-Southern Oscillation (ENSO) and other extreme events that result in changes in productivity and disturbance regimes (e.g., fires, pest and disease outbreak).

### Agriculture

Most global and regional economic studies—with and without climate change—indicate that the downward trend in real commodity prices in the 20th century is likely to continue into the 21st century, although confidence in these predictions decreases farther into the future (see Section 5.3.1).

*Impacts*

- Experiments have shown that relative enhancement of productivity caused by elevated $CO_2$ usually is greater when temperature rises but may be less for crop yields at above-optimal temperatures (established but incomplete). Although the beneficial effects of elevated $CO_2$ on the yield of crops are well established for the experimental conditions tested, this knowledge is incomplete for numerous tropical crop species and for crops grown under suboptimal conditions (low nutrients, weeds, pests and diseases). In experimental work, grain and forage quality declines with $CO_2$ enrichment and higher temperatures (high confidence) (see Sections 5.3.3, 5.4.3, and 5.5.3).
- Experimental evidence suggests that relative enhancement of productivity caused by elevated $CO_2$ usually is greater under drought conditions than in wet soil. Nevertheless, a climate change-induced reduction in summer soil moisture (see Table 3-10)—which may occur even in some cases of increased summer precipitation—would have detrimental effects on some of the major crops, especially in drought-prone regions (medium confidence).
- Soil properties and processes—including organic matter decomposition, leaching, and soil water regimes—will be influenced by temperature increase (high confidence). Soil erosion and degradation are likely to aggravate the detrimental effects of a rise in air temperature on crop yields. Climate change may increase erosion in some regions, through heavy rainfall and through increased windspeed (competing explanations) (see Section 5.3.3).
- Model simulations of wheat growth indicate that greater variation in temperature (change in frequency of extremes) under a changing climate reduces average grain yield. Moreover, recent research emphasizes the importance of understanding how variability interacts with changes in climate means in determining yields (established but incomplete) (see Section 5.3.4).
- Crop modeling studies that compare equilibrium scenarios with transient scenarios of climate change report significant yield differences. The few studies that include comparable transient and equilibrium climate change scenarios generally report greater yield loss with equilibrium climate change than with the equivalent transient climate change. Even these few studies are plagued with problems of inconsistency in methodologies, which make comparisons speculative at this time (see Section 5.3.4).

*Adaptation and Vulnerability*

- Prospects for adaptation of plant material to increased air temperature through traditional breeding and genetic modification appear promising (established but incomplete). More research on possible adaptation of crop species to elevated $CO_2$ is needed before more certain results can be presented (see Section 5.3.3).

- Simulations without adaptation suggest more consistent yield losses from climate change in tropical latitudes than temperate latitudes. Agronomic adaptation abates extreme yield losses at all latitudes, but yields tend to remain beneath baseline levels after adaptation more consistently in the tropics than in temperate latitudes (moderate confidence) (see Section 5.3.4).
- The ability of livestock producers to adapt their herds to the physiological stress of climate change is not known conclusively, in part because of a general lack of experimentation and simulations of livestock adaptation to climate change (see Section 5.3.3).
- Crop and livestock farmers who have sufficient access to capital and technologies are expected to adapt their farming systems to climate change (medium to low confidence) (see Section 5.3.4). Substantial shifts in their mix of crops and livestock production may be necessary, however, and considerable costs could be involved in this process—*inter alia*, in learning and gaining experience with different crops or if irrigation becomes necessary. In some cases, a lack of water resulting from climate change might mean that increased irrigation demands cannot be met (see Section 4.7.2). Although this conclusion is speculative because of lack of research, it is intuitive that the costs of adaptation should depend critically on the rate of climate change.
- Impacts of climate change on agriculture after adaptation are estimated to result in small percentage changes in global income; these changes tend to be positive for a moderate global warming, especially when the effects of $CO_2$ fertilization are taken into account (low confidence) (see Section 5.3.5).
- The effectiveness of adaptation in ameliorating the economic impacts of climate change across regions will depend critically on regional resource endowments. It appears that developed countries will fare better in adapting to climate change; developing countries and countries in transition, especially in the tropics and subtropics, will fare worse. This finding has particularly significant implications for the distribution of impacts within developing countries, as well as between more- and less-developed countries. These findings provide evidence to support the hypothesis advanced in the SAR that climate change is likely to have its greatest adverse impacts on areas where resource endowments are poorest and the ability of farmers to respond and adapt is most limited (medium confidence) (see Section 5.3.5).
- Degradation of soil and water resources is one of the major future challenges for global agriculture (see Section 5.3.2). These processes are likely to be intensified by adverse changes in temperature and precipitation. Land use and management have been shown to have a greater impact on soil conditions than the direct effects of climate change; thus, adaptation has the potential to significantly mitigate these impacts (see Section 5.3.4). A critical research need is to assess whether resource degradation will significantly increase the risks faced by vulnerable agricultural and rural populations (see Section 5.3.6).
- It is concluded with low confidence that a global temperature rise of greater than 2.5°C will result in rising commodity prices. Similarly, a global temperature rise of greater than 2.5ºC increases by 80 million the absolute number of people at risk of hunger. It should be noted, however, that these hunger estimates are based on the assumption that food prices will rise with climate change, which is highly uncertain (see Section 5.3.6).

### *Wildlife*

Recent estimates indicate that 25% of the world's mammals and 12% of birds are at significant risk of global extinction. Climate change is only one of a long list of pressures on wildlife. Other pressures include exploitation of animals, pollution and other biochemical poisonings, extreme climatic events, wildlife diseases, collisions with towers and other structures, anthropogenic barriers to dispersal, and war and other civil conflicts. Alone or in combination, these pressures will greatly increase species' vulnerabilities to rarity and extinction (high confidence). Habitat conversion and degradation affect nearly 89% of all threatened birds and 83% of all threatened mammals. About one-fifth of threatened mammals in Australia and the Americas and the world's birds are affected by introduced species.

#### *Impacts, Adaptations, and Vulnerabilities*

- Many animals already may be responding to local climatic changes. The types of changes already observed include poleward and elevational movement of ranges, changes in animal abundance, changes in body size, and shifts in the timing of events, such as earlier breeding in spring. Possible climatically associated shifts in animal ranges and densities have been noted on many continents and within each major taxonomic group of animals (see Table 5-3).
- Laboratory and field studies have demonstrated that climate plays a strong role in limiting species' ranges (high confidence). Even though only a small fraction of all species have been monitored long enough to detect significant trends, changes exhibited over the past few decades in the bulk of these species are consistent with local warming and expected physiological responses (medium confidence). However, possible *specific* changes in wildlife from climate change can be projected only with low confidence for most species because of many possible contributing factors, such as habitat destruction and introduction of exotic species. Some species clearly are responding to global change (see Section 5.4.3), and many more changes probably have gone undetected. Researchers are in the process of coupling these discernible changes with various biological theories regarding climate and species spatial and temporal patterns; through this process, we expect that reliable *general* projections can be and in fact are being made.
- Protecting threatened and endangered species requires measures that, in general, reverse the trend toward rarity. Without management, rapid climate change—in conjunction

with other pressures—is likely to cause many species that currently are classified as critically endangered to become extinct and several labeled endangered or vulnerable to become much rarer, and thereby closer to extinction, in the 21st century (high confidence).

- Concern over species becoming rare or extinct is warranted because of the goods and services provided by ecosystems and the species themselves. Most of the goods and services provided by wildlife (e.g., pollination, natural pest control) are derived from their roles within systems. Other valuable services are provided by species contributing to ecosystem stability or to ecosystem health and productivity. The recreational value (e.g., sport hunting, wildlife viewing) of species is large in market and nonmarket terms. Species loss also could impact the cultural and religious practices of indigenous peoples around the world. Losses of species can lead to changes in the structure and function of affected ecosystems and loss of revenue and aesthetics. Understanding the role each species plays in ecosystem services is necessary to understand the risks and possible surprises associated with species loss. Without this information, the probability of surprises associated with species loss is high (medium confidence).
- Humans may need to adapt not only in terms of wildlife conservation but also to replace lost ecological services normally provided by wildlife. It may be necessary to develop adaptations to losses to natural pest control, pollination, and seed dispersal. Although replacing providers of these services sometimes may be possible, the alternatives may be costly. Finding replacements for other services, such as contributions to nutrient cycling and ecosystem stability/biodiversity, are much harder to imagine. In many cases, such as the values of wildlife associated with subsistence hunting and cultural and religious ceremonies, any attempt at replacement may represent a net loss. In many countries, climate change impacts, such as reductions in wildlife populations, may have the greatest impact on the lowest income groups—those with the least ability to adapt if hunting opportunities decline.

### *Rangelands*

Most rangelands in the world have been affected by human activity, and many are degraded in some way. Desertification tends to be associated with land degradation in rangelands; however, desertification combines many land degradation processes and can be exacerbated by climate change. Many of the rangelands of the world are affected by ENSO events and are sensitive to the frequency of these events, resulting in changes in productivity of these systems.

*Impacts, Adaptations, and Vulnerabilities*

- Based on observations and modeling studies, the effects of elevated $CO_2$ and climate change could result in increased plant productivity and thus an increase in soil carbon sequestration in many rangelands. However, some of the gains in productivity would be offset by increases in temperatures and by human management activities (medium confidence).
- Modeling studies and observations suggest that plant production, species distribution, disturbance regimes (e.g., frequencies of fires, insect/pest outbreaks), grassland boundaries, and nonintensive animal production would be affected by potential changes in climate and land use. The impacts of climate change are likely to be minor compared to those of land degradation (high confidence).
- Irrigation in semi-arid climates is a major cause of secondary salinization. Elevated $CO_2$ may reduce the impacts of secondary salinization, although experimental work shows that any increase in temperature may negate these benefits and may even exacerbate problems of secondary salinity (medium confidence).
- In many parts of the world that are dominated by rangelands, lack of infrastructure and investment in resource management limits available options for adaptation and makes these areas more sensitive and vulnerable to the impacts of climate change (high confidence). Some adaptation options (e.g., integrated land management) could be implemented irrespective of technology and infrastructure. Other adaptation options could be implemented through active involvement of communities in the management of rangelands.

### *Forests and Woodlands*

Loss in forest cover appears to have slowed in recent years relative to 1980–1995. However, fragmentation, nonsustainable logging of mature forests, degradation, and development of infrastructure—all leading to losses of biomass—have occurred over significant areas in developing and developed countries (high confidence). Pressure from disturbances such as fires appears to be increasing around the world. Fire suppression in temperate managed and unmanaged forests with access to infrastructure and human capital has been largely successful, but regions with comparatively less infrastructure have been more susceptible to natural and human-caused fires. Deforestation will continue to be the dominant factor influencing land-use change in tropical regions. Timber harvests near roads and mills in tropical regions are likely to continue to fragment and damage natural forests (high confidence).

Non-wood forest products (NWFP) such as edible mushrooms, nuts, fruits, palm hearts, herbs, spices, gums, aromatic plants, game, fodder, rattan, medicinal and cosmetic products, resins, and the like make important contributions to household income, food security, national economies, and the environmental objectives of conservation of biodiversity.

*Impacts, Adaptations, and Vulnerabilities*

- Forest response to climate change and other pressures will alter future carbon storage in forests, but the global extent and direction of change is unknown.

- Recent experimental evidence suggests that the net balance between net primary productivity (NPP—usually assumed to increase with warming, but challenged by recent studies), heterotrophic respiration (often assumed to increase with warming, but also challenged by recent studies), and disturbance releases (often ignored, but shown to be important in boreal estimates of net biome productivity) is no longer as clear as stated in the SAR.
- Research reported since the SAR confirms the view that the largest and earliest impacts induced by climate change are likely to occur in boreal forests, where changes in weather-related disturbance regimes and nutrient cycling are primary controls on productivity (high confidence). The effect of these changes on NPP and carbon storage is uncertain.
- Since the SAR, free-air $CO_2$ enrichment (FACE) experiments suggest that tree growth rates may increase, litterfall and fine root increment may increase, and total NPP may increase, but these effects are expected to saturate because forest stands tend toward maximum carrying capacity, and plants may become acclimated to increased $CO_2$ levels (medium confidence).
- Questions of saturation of the $CO_2$ response can be addressed through longer term experiments on tree species grown under elevated $CO_2$ in open-top chambers under field conditions over several growing seasons. Results from these experiments show continued and consistent stimulation of photosynthesis and little evidence of long-term loss of sensitivity to $CO_2$; the relative effect on aboveground dry mass was highly variable and greater than indicated by seedling studies, and the annual increase in wood mass per unit of leaf area increased (high confidence).
- Contrary to the SAR, global timber market studies that include adaptation suggest that climate change will increase global timber supply and enhance existing market trends toward rising market share in developing countries. Consumers are likely to benefit from lower timber prices; producers may gain or lose, depending on regional changes in timber productivity and potential dieback effects. Studies that do not consider global market forces, timber prices, or adaptation predict that supply in boreal regions could decline (medium confidence).
- Industrial timber harvests are predicted to increase by 1–2% $yr^{-1}$. The area of industrial timber plantations is likely to continue to expand, and management in second-growth forests in temperate regions is likely to continue to intensify, taking pressure off natural forests for harvests (high confidence).
- At the regional and global scale, the extent and nature of adaptation will depend primarily on wood and non-wood product prices, the relative value of substitutes, the cost of management, and technology. On specific sites, changes in forest growth and productivity will constrain—and could limit—choices of adaptation strategies (high confidence). In markets, prices will mediate adaptation through land and product management. Adaptation in managed forests will include salvaging dead and dying timber and replanting with new species that are better suited to the new climate.

### *Lakes and Rivers*

Capture, culture, and recreational fisheries are reported to land about 23 Mt $yr^{-1}$ of biomass, but the actual numbers probably are twice that. Fish species in freshwater total about 11,800. High levels of endemism are common for many different groups of freshwater organisms. In addition to climatic changes, lakes and rivers are impacted by pressures such as land and water use, pollution, capture and culture fisheries, water extraction, and hydrologic engineering structures such as dams, dykes, and channelization (well established). These pressures interact with pressures from climate change and vary in their impact even at local levels mediated through changes in hydrology (medium confidence).

#### *Impacts, Adaptations, and Vulnerabilities*

- A 150-year trend in 26 lakes and rivers in the northern hemisphere averaging 9 days later freeze and 10 days earlier ice breakup has resulted from a 1.8°C increase in air temperature (very high confidence).
- Empirical and simulation studies show that elevated water temperature increases summer anoxia in deep waters of stratified lakes (high confidence).
- There has been an observed poleward movement of southern and northern boundaries of fish distributions (high confidence).
- There is a loss of habitat for cold- and coolwater fishes and gain in habitat for warmwater fishes (high confidence).
- There are complex relations between warmer temperature, more episodic rainfall, and poleward movement of warm-water zooplanktivorous fishes, which may exacerbate eutrophication of lakes and rivers (medium confidence).
- Human activities to manage water flow may exacerbate the impact on lakes and rivers (medium confidence); for example, the increase in hydrologic engineering structures is likely to result in fewer free-flowing streams (medium confidence). Attempts to manage the poleward movement of fauna and flora, especially in lakes whose isolation is likely to slow down species from moving poleward, is likely to be contentious and produce frequent surprises and unexpected dynamics of freshwater communities (medium confidence).
- New opportunities may be provided for aquaculture of warmer water species in more poleward locations (medium confidence).
- As a class of ecosystems, inland waters are vulnerable to climatic change and other pressures, owing to their small size and position downstream from many human activities (high confidence). The most vulnerable elements include reduction and loss of lake and river ice (very high confidence), loss of habitat for coldwater fish (very high confidence), increases in extinctions and invasions of exotics (high confidence), and potential exacerbation of existing pollution problems such as eutrophication, toxics, acid rain, and ultraviolet-B (UV-B) radiation (medium confidence).

### Inland Wetlands

Wetlands play an important role in maintaining biological diversity by providing a habitat for many plant and animal species, some of which are endemic or endangered. They also have significant scientific value that goes beyond their plant and animal communities. Peat-accumulating wetlands are important for global change because of the large carbon store accumulated over the millennia and the risk that this store would be released to the atmosphere in conditions modified by global change. Besides being carbon sinks, wetlands are sources of methane to the atmosphere.

We can state with high confidence that the pressures of climate change on habitat, biodiversity, and carbon sink on most wetland types will be largely indirect, operating through changes in water level. On peatlands underlain by permafrost, however, direct temperature impacts can be more important. Food and fiber production are ecosystem services that simultaneously form a pressure (land-use change) on other services (habitat, biodiversity, carbon sink).

#### *Impacts, Adaptations, and Vulnerabilities*

- A warmer, drier climate is not necessarily likely to lead to a large loss of stored peat for all peatland types (e.g., boreal peatlands), but there are competing explanations because feedback mechanisms between climate and peatland hydrology and the autogenic nature of peatland development are poorly understood.
- Peatlands underlain by permafrost are likely to become net carbon sources rather than sinks, mainly because of melting of permafrost and lowering of the water table (high confidence). Work in tussock and wet tundra has shown that these systems already may be net annual sources of 0.19 Gt C.
- Extensive seasonally inundated freshwater swamps, which are major biodiversity foci, could be displaced if predicted sea-level rises of 10–30 cm by 2030 occur (high confidence).
- In the southeast Asian region, droughts in recent years related to ENSO events have lowered local water tables and thus increased the severity of fires in tropical peatlands (high confidence).
- The effects of water-level drawdown after drainage for forestry indicate that the shift in species composition from wetland species to forest species only slightly affects plant species richness of individual sites; in regions dominated by forests, however, there would be a clear reduction in regional diversity as landscapes become homogenized after water-level drawdown (medium confidence).
- Most wetland processes are dependent on catchment-level hydrology. Thus, adaptations to projected climate change may be practically impossible. For degrading key habitats, small-scale restoration may be possible if sufficient water is available.
- Arctic and subarctic ombrotrophic bog communities on permafrost, together with more southern depressional wetlands with small catchment areas, are likely to be most vulnerable to climate change. The increasing speed of peatland conversion and drainage in southeast Asia is likely to place these areas at a greatly increased risk of fires and affect the viability of tropical wetlands.

### Arctic and Alpine Ecosystems

Observations show that high-latitude warming has occurred in the Arctic region since the 1960s. Precipitation and surface evaporation have increased at high latitudes. Permafrost temperatures have warmed in western North America by 2–4°C from 1940 and in Siberia by 0.6–0.7°C from 1970 to 1990, whereas permafrost cooled in northeastern Canada. These patterns roughly parallel recent trends in air temperature. Carbon flux measurements in Alaska suggest that the recent warming trend may have converted tundra from a net carbon sink to a source of as much as 0.7 Gt C $yr^{-1}$. During recent decades, the peak-to-trough amplitude in the seasonal cycle of atmospheric $CO_2$ concentrations has increased, and the phase has advanced at arctic and subarctic $CO_2$ observation stations north of 55°N. This change in carbon dynamics in the atmosphere probably reflects some combination of increased uptake during the first half of the growing season—which could explain the observed increase in biomass of some shrubs, increased winter efflux, and increased seasonality of carbon exchange associated with disturbance. This “inverse” approach generally has concluded that mid-northern latitudes were a net carbon sink during the 1980s and early 1990s. At high northern latitudes, these models give a wider range of estimates; some analyses point to a net and others to a sink.

Climatic changes observed in alpine areas generally have paralleled climatic patterns in surrounding regions, with the most pronounced warming at high latitudes, in the Alps, and in Asia and the least pronounced changes in tropical alpine regions. Precipitation generally has increased, with the most pronounced changes in winter, leading to increased snow depth. Regional trends in climate have led to shrinkage of alpine and subpolar glaciers equivalent to 0.25 ± 0.1 mm $yr^{-1}$ of sea-level change—or 16% of the sea-level rise in the past 100 years. Net mass reduction of the alpine glaciers has been most pronounced since 1980, when regional warming was greatest. Climatic warming observed in the Alps has been associated with upward movement of some plant taxa of 1–4 m per decade on mountaintops and the loss of some taxa that formerly were restricted to high elevations. In general, direct human impacts on alpine vegetation from grazing, tourism, and nitrogen deposition are so strong that climatic effects on the goods and services provided by alpine ecosystems are difficult to detect.

#### *Impacts, Adaptations, and Vulnerabilities*

- Projected climatic warming of 4–10°C by the end of the 21st century is likely to cause substantial increases in decomposition, nutrient release, and primary production (high confidence). Many of these changes in productivity may be threshold effects; they also may be mediated by changes in species composition and therefore are likely to lag changes in climate by years, decades, or centuries (medium confidence).

- Important changes in diversity in the Arctic include changes in the abundance of caribou, waterfowl, and other subsistence resources (medium confidence).
- Tundra has a three- to six-fold higher winter albedo than boreal forest, but summer albedo and energy partitioning differ more strongly among ecosystems within either tundra or boreal forest than between these two biomes (high confidence). Changes in albedo and energy absorption during winter are likely to act as a positive feedback to regional warming.
- Disturbances such as fires are likely to change with changes in regional climate and affect biophysical properties (high confidence).
- In alpine areas, warming is likely to create a shortened snowmelt season, with rapid water release creating floods and later growing-season droughts, affecting productivity in these areas (medium confidence).
- Changes in goods and services in alpine ecosystems are likely to be dominated by changes in land use associated with grazing, recreation, and other direct impacts because of their proximity to population centers (high confidence). Many of the alpine zones with greatest biodiversity, such as the Caucasus and Himalayas, are areas where human population pressures may lead to most pronounced land-use change. Overgrazing, trampling, and nutrient/pollution deposition may tend to destabilize vegetation, leading to erosion and loss of soils that are the long-term basis of the productive capacity of alpine ecosystems.
- Opportunities for adapting to expected changes in Arctic and alpine ecosystems are limited because these systems will respond most strongly to globally induced changes in climate (medium confidence). Opportunities for mitigation will include protection of peatlands, yedoma sediments, and other carbon-rich areas from large-scale hydrological change, land use, and pollutant levels. Careful management of wildlife resources could minimize climatic impacts on indigenous peoples. Many Arctic regions depend strongly on one or a few resources, such as timber, oil, reindeer, or wages from fighting fires. Economic diversification would reduce the impacts of large changes in the availability or economic value of particular goods and services.
- The goods and services provided by many Arctic regions depend on the physical integrity of permafrost and therefore are vulnerable to climate change (high confidence). The large carbon stocks in these regions are vulnerable to loss to the atmosphere as $CO_2$ or methane.
- The high levels of endemism in many alpine floras and their inability to migrate upward means that these species are most vulnerable (medium confidence).

## 5.1. Introduction and Scope

Ecosystems provide many products and services that are crucial to human survival (Daily, 1997; UNEP, 1998; WRI, 2000). Ecosystems affect biogeochemical and physical feedbacks to the biosphere and atmosphere, hence are important for the functioning of the Earth's systems. Ecosystems form a landscape and are connected in many ways, often by streams, rivers, and wildlife. Thus, landscape fragmentation, along with other human activities, affects ecosystems' ability to meet human needs and will continue to do so for the future, possibly at a faster rate (UNEP, 1998). Changes in global climate and atmospheric composition are likely to have an impact on most of these goods and services, with significant impacts on socioeconomic systems (Winnett, 1998).

This chapter assesses the impacts of climate change on ecosystem goods and services from sectors such as agriculture, forests, and wetlands. Inland aquatic systems are covered from an ecosystem perspective in this chapter; hydrology and water as a physical resource are covered in Chapter 4. Marine and coastal systems are considered in Chapter 6. When ecosystems are highly managed, as in agriculture and forestry, or their goods and services are traded in markets, the social and economic consequences of climate change that naturally arise are assessed explicitly.

Biomass production, biogeochemical cycling, soil and water relationships, and animal-plant interactions (including biodiversity) are considered to be some of the major functions of ecosystems. Within these functions, various products (goods) and services can be identified, including food, fiber, fuel and energy, fodder, medicines, clean water, clean air, flood/storm control, pollination, seed dispersal, pest and disease control, soil regeneration, biodiversity, and recreation/amenity (UNEP, 1998; WRI, 2000). Society places values on these goods and services, directly or indirectly (Table 5-1). Ecosystems provide many of these goods and services simultaneously. For example, agricultural systems provide much of our food, fiber, and fuel needs and at the same time influence biogeochemical cycling, soil and water quality, and biodiversity. Many services from ecosystems lie outside market systems, making it difficult to price them (Bawa and Gadgil, 1997; Goulder and Kennedy, 1997; National Research Council, 1999). However, these nonmarket values are likely to be larger (as much as 1,000-fold; WRI, 2000) than the value of services provided by markets in total and at many specific sites (Costanza *et al.*, 1997). Although several studies estimating different values for nonmarket services from ecosystems exist (see Table 5-2), they can be applied only with low to medium confidence (Goulder and Kennedy, 1997). Valuation of ecosystem services is complex because many goods and services occur simultaneously. Thus, it is not sufficient to consider, for example, the timber value of

***Table 5-1****: Ecosystems function with links to good/services and possible societal value (modified from Ewel et al., 1998).*

| Function | Goods/Service | Value[a] |
|---|---|---|
| Production | – Food<br>– Fiber (timber and non-wood products)<br>– Fuel<br>– Fodder | Direct |
| Biogeochemical cycling | – Nutrient cycling (especially N and P absorption/deposition)<br>– Carbon sinks | Mostly indirect, although future values have to be considered |
| Soil and water conservation | – Flood and storm control<br>– Erosion control<br>– Clean water<br>– Clean air<br>– Water for irrigation<br>– Organic matter or sediment export<br>– Pollution control<br>– Biodiversity | Mostly indirect, although future values have to be considered |
| Animal-plant interactions | – Pollination<br>– Animal migration<br>– Biodiversity | Mostly indirect, future, bequest, and existence values have to be considered |
| Carrier | – Lansdcape connectivity<br>– Animal migration<br>– Biodiversity<br>– Aesthetic/spiritual/cultural service | Mostly indirect and existence, but bequest may have to be considered |

[a] Value definitions are from Pearce and Moran (1994); see Table 5-2.

***Table 5-2****: Examples of goods and services with possible uses and values (adapted from Pearce and Moran, 1994).*

| **Value** | **Examples of Goods and Services** |
|---|---|
| Direct use | Food, fiber, fuel, fodder, water supply, recreation, non-wood forest products |
| Indirect use | Biodiversity, biogeochemical cycles, tourism, flood and storm control, clean water supply, pollution control |
| Option | Future discoveries (i.e., pharmacological and biotechnological), future recreation |
| Bequest | Intergenerational and sustainable development |
| Existence | Mostly conservation, aesthetic, spiritual |

the forest; we also must consider the soil/water protection that the trees provide, the habitat for pollinators, or the bequest value of the forest (WRI, 2000).

The Earth is being subjected to many human-induced and natural changes, often referred to as global change. These changes include pressures from increased demand for resources driven by economic growth, increased human population, land-use and land-cover change, the accelerated rate of anthropogenic nitrogen production and other air pollutants, and urbanization and industrialization; resulting fossil fuel emissions contribute to a discernible impact on global climate (Naiman *et al.*, 1995a; Vitousek *et al.*, 1997a; IPCC, 1998; UNEP, 1998; Walker *et al.*, 1999). For ecosystems, the impacts of climate change include changes in atmospheric composition and disturbance regimes, such as frequencies of fires, storms, floods, and drought. The impacts of other pressures often lead to increased demand for access to land, water, and wildlife resources. The result is a change in the state of the Earth's land surface, the services humans receive, and the landscapes where humans live at regional and global scales. Governance and equity issues are important (UNEP, 1998) in overcoming some of these; these are covered in detail in TAR WGIII Chapter 1.

Understanding the current status of ecosystems, pressures on them, and their responses is important for assessing the impact of the additional pressure of climate change. The State-Pressure-Response model of the Organisation for Economic Cooperation and Development (OECD) has been used as a framework in structuring many subsequent sections of this chapter (see Figure 5-1). Using this model, the "state" refers to assessments of current status and recent trends of each sector and "pressures" include direct and indirect human pressures. The concept of "response" has been modified to include automatic responses of ecosystems (referred to as automatic adaptation in Chapter 18) to the impacts of climate change (including that of natural climatic variability). Deliberate "adaptation" options to climate change to overcome some of these impacts-responses could alleviate some of the pressures. Systems that are not able to adapt are likely to become vulnerable (see Chapter 18); thus, vulnerability also is assessed.

In many sections, four goods and services have been emphasized because of the availability of literature and space constraints: food, fiber, carbon storage, and biodiversity. Many other topics—for example, related to water cycling and hydrology—could have been considered. Some of these are covered in other chapters of this report (e.g., Chapter 4 and as key regional concerns in Chapters 10–17).

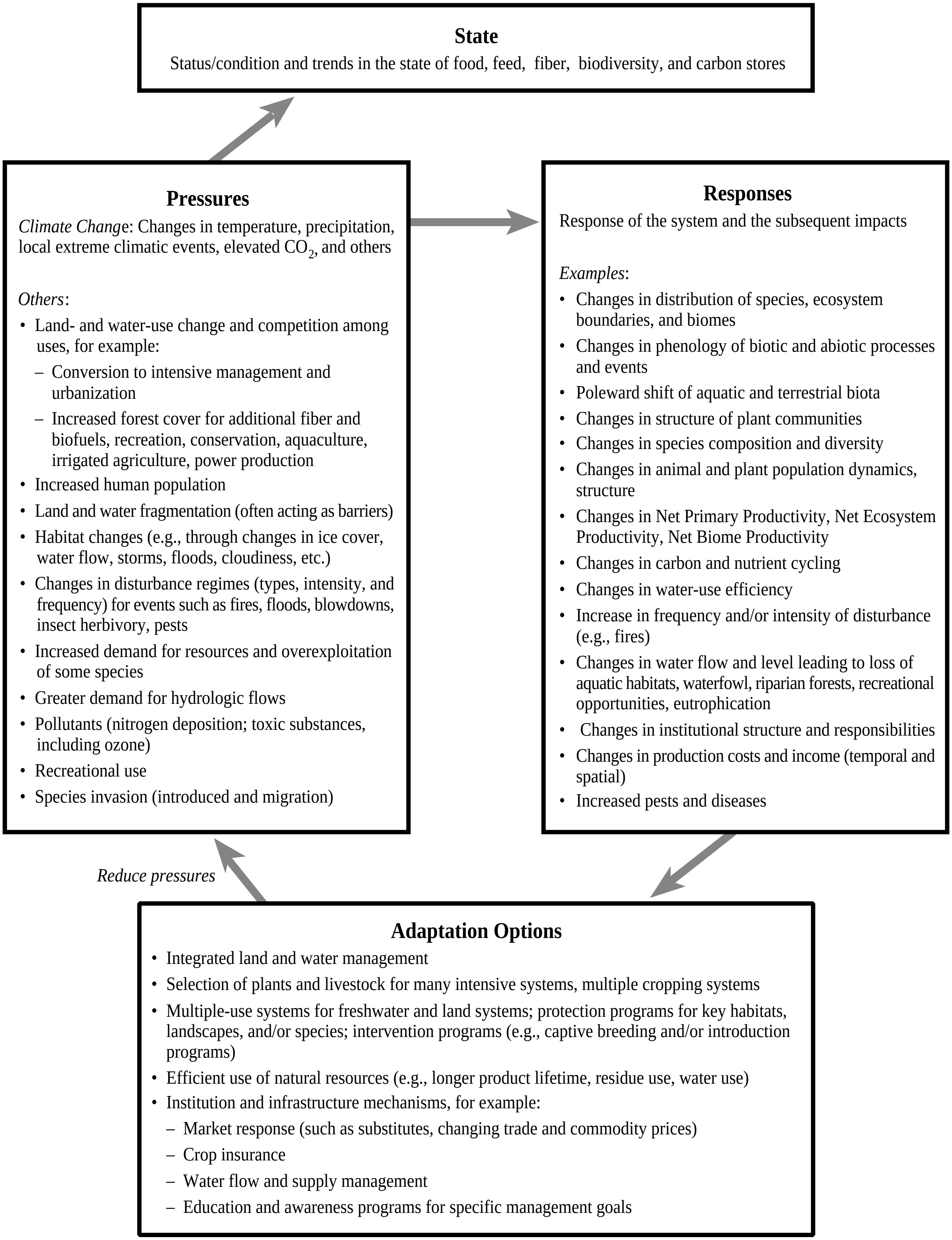
Some impacts of global change apply across many sectors and are considered in Section 5.2. Many of the impacts of climate change are projected by using models, and Section 5.2 includes a critique of several of these models. Some studies in the literature already have identified regional impacts of post-industrial climate change on some ecosystems. These are discussed in the most appropriate sections under "impacts-response" but are dealt with mainly in Sections 5.2 and 5.4.

## 5.2. Effects of Global Change on the Terrestrial Biosphere

Terrestrial ecosystems consist of plants, animals, and soil biota and their environment. The distribution of biota within and across ecosystems is constrained by the physical and chemical conditions of the atmosphere, the availability of nutrients and/or pollutants, and disturbances from natural origin (fire, wind-throw, etc.) or human land use. Global change affects all of these factors, but through widely differing pathways and at different scales. Global climate change, for example, affects local weather and climate in ways that are strongly dependent on location. Increased atmospheric $CO_2$ concentration, on the other hand, is geographically more uniform.

At the global scale, the sum of all ecosystems (including the marine biosphere) exerts a significant role on the balance of carbon and water in the atmosphere. Feedbacks exist between climate-driven changes in biospheric functioning, such as enhanced or reduced primary productivity, and the amount of greenhouse gas (GHG)-related radiative forcing. In addition, changes in land surface characteristics affect the atmosphere by altering the radiation balance at the reflective surface, creating further feedbacks. These biospheric feedbacks are discussed in TAR WGI Chapter 3.

In trying to understand the effects of global change on the biosphere as a whole, scientists often focus on higher level entities such as ecosystems or biomes (the collection of ecosystems within a particular climatic zone with similar structure but differing species—e.g., the temperate forest biome). However, all factors of change act on individual organisms that are part of a complex web of interactions within and between species and their ecosystem and within landscapes that contain



**State**

Status/condition and trends in the state of food, feed, fiber, biodiversity, and carbon stores

**Pressures**

*Climate Change*: Changes in temperature, precipitation, local extreme climatic events, elevated $CO_2$, and others

*Others*:

- Land- and water-use change and competition among uses, for example:
  - Conversion to intensive management and urbanization
  - Increased forest cover for additional fiber and biofuels, recreation, conservation, aquaculture, irrigated agriculture, power production
- Increased human population
- Land and water fragmentation (often acting as barriers)
- Habitat changes (e.g., through changes in ice cover, water flow, storms, floods, cloudiness, etc.)
- Changes in disturbance regimes (types, intensity, and frequency) for events such as fires, floods, blowdowns, insect herbivory, pests
- Increased demand for resources and overexploitation of some species
- Greater demand for hydrologic flows
- Pollutants (nitrogen deposition; toxic substances, including ozone)
- Recreational use
- Species invasion (introduced and migration)

**Responses**

Response of the system and the subsequent impacts

*Examples*:

- Changes in distribution of species, ecosystem boundaries, and biomes
- Changes in phenology of biotic and abiotic processes and events
- Poleward shift of aquatic and terrestrial biota
- Changes in structure of plant communities
- Changes in species composition and diversity
- Changes in animal and plant population dynamics, structure
- Changes in Net Primary Productivity, Net Ecosystem Productivity, Net Biome Productivity
- Changes in carbon and nutrient cycling
- Changes in water-use efficiency
- Increase in frequency and/or intensity of disturbance (e.g., fires)
- Changes in water flow and level leading to loss of aquatic habitats, waterfowl, riparian forests, recreational opportunities, eutrophication
- Changes in institutional structure and responsibilities
- Changes in production costs and income (temporal and spatial)
- Increased pests and diseases

**Adaptation Options**

- Integrated land and water management
- Selection of plants and livestock for many intensive systems, multiple cropping systems
- Multiple-use systems for freshwater and land systems; protection programs for key habitats, landscapes, and/or species; intervention programs (e.g., captive breeding and/or introduction programs)
- Efficient use of natural resources (e.g., longer product lifetime, residue use, water use)
- Institution and infrastructure mechanisms, for example:
  - Market response (such as substitutes, changing trade and commodity prices)
  - Crop insurance
  - Water flow and supply management
  - Education and awareness programs for specific management goals

**Figure 5-1**: Generalized diagram of the state of specific goods and services that ecosystems provide; how these goods and services are affected by the multiple pressures of climate change and human activities; and how the system responds (autonomous adaptation as in Chapter 19), thus affecting the provision of goods and services. Adaptation options reduce the impacts and thus change the vulnerability of the system.

mosaics of different ecosystems. It is not feasible to model the impacts of global change at global, or even regional or landscape, scales at an individual-by-individual level. Thus, most models of global change have dealt with impacts at the ecosystem or biome level. This is in contrast with limitations to observational and experimental studies in which only a selection of individuals and species of a few ecosystems can be included.

### 5.2.1. *Observational Studies*

There is now a substantial number of observational and experimental studies that demonstrate the link between climate and biological or physical processes in ecosystems. The authors of this chapter assembled a database of more than 2,500 studies that address climate and either a physical process (e.g., melting of ice on lakes) or a biological factor (e.g., spring arrival time) of an animal or plant. Most of these studies address experiments that are valuable primarily in helping to understand the biological mechanisms prompting the responses of plants and animals to climate but are not helpful in detecting patterns of change. Many of these studies were conducted over a period of shorter than 10 years. Because at least 10 years of data are needed to show a possible trend, this narrowed the number of studies to approximately 500. Because temperature is the variable that can most reliably be predicted with increasing GHGs, only studies that addressed temperature as the climatic variable were examined, leaving approximately 250 studies. All 250 studies then were examined to determine if they met at least two of the following criteria:

- The authors found a statistically significant correlation between temperature and a species trait (e.g., egg-laying date, location of range boundaries) or physical process.
- The authors found a statistically significant change in the species trait over time.
- The authors found a statistically significant change in temperature over time.

***Table 5-3**: The number of species and processes in each region that were found in each particular study to be significantly associated with regional temperature change. For inclusion in the table, each study had to meet two of the following three criteria: species or processes changing over time; regional temperature changing over time; and significant association between how the temperature and species or processes were changing. The first number indicates the number of species or processes changing in the manner predicted with global warming. The second number is the number of species or processes changing in a manner opposite to that predicted with a warming planet. **When considering those species that have shown a change, 80% are changing in the manner expected with global warming, while 20% are changing in the opposite direction**. Note that about 61 of all species examined did not show a statistically significant change. References for each cell are located below the table and collated by row number and column number (e.g., references for European birds are under **E,5**—row E, column 5). "—" indicates that no studies were found for this region and category.*

| **Region** | **Column 1: Lake and Stream Ice** | | **Column 2: Vegetation** | | **Column 3: Invertebrates** | | **Column 4: Amphibians and Reptiles** | | **Column 5: Birds** | | **Column 6: Mammals** | |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Row A: Africa | — | — | — | — | — | — | — | — | — | — | — | — |
| Row B: Antarctica | — | — | 2 | 0 | — | — | — | — | 2 | 0 | — | — |
| Row C: Asia | 3 | 0 | — | — | — | — | — | — | — | — | — | — |
| Row D: Australia | — | — | — | — | — | — | — | — | — | — | — | — |
| Row E: Europe | 8 | 0 | 13 | 1 | 46 | 1 | 7 | 0 | 258 | 92 | 7 | 0 |
| Row F: North America | 18 | 0 | 32 | 11 | — | — | — | — | 17 | 4 | 3 | 0 |
| Row G: Latin America | — | — | — | — | — | — | 22 | 0 | 15 | 0 | — | — |
| Total | 29 | 0 | 47 | 12 | 46 | 1 | 29 | 0 | 292 | 96 | 10 | 0 |

*Notes*: **B,2**. Smith (1994); **B,5**. Fraser *et al*. (1992), Cunningham and Moors (1994), Smith *et al*. (1999); **C,1**. Magnuson *et al*. (2000); **E,1**. Magnuson *et al*. (2000); **E,2**. Grabherr *et al*. (1994), Ross *et al*. (1994), Hasenauer *et al*. (1999), Menzel and Fabian (1999); **E,3**. Fleming and Tatchell (1995), Zhou *et al*. (1995), de Jong and Brakefield (1998), Rodriguez-Trelles and Rodriguez (1998), Visser *et al*. (1998), Parmesan *et al*. (1999), Parmesan *et al*. (2000); **E,4**. Beebee (1995), Reading and Clarke (1995), Reading (1998), Sparks (1999); **E,5**. Jarvinen (1989, 1994), Gatter (1992), Bezzel and Jetz (1995), Mason (1995), Winkel and Hudde (1996, 1997), Crick *et al*. (1997), Ludwichowski (1997), Forchhammer *et al*. (1998), McCleery and Parrins (1998), Prop *et al*. (1998), Visser *et al*. (1998), Bergmann (1999), Crick and Sparks (1999), Slater (1999), Sparks (1999), Thomas and Lennon (1999); **E,6**. Post and Stenseth (1999); **F,1**. Magnuson *et al*. (2000); **F,2**. Barber *et al*. (2000), Bradley *et al*. (1999); **F,5**. Bradley *et al*. (1999), Brown *et al*. (1999), Dunn and Winkler (1999); **F,6**. Post and Stenseth (1999); **G,4**. Pounds *et al*. (1999); **G,5**. Pounds *et al*. (1999).

In some cases, the criteria were met by two companion papers rather than a single paper. These criteria narrowed the qualifying studies to 60; seven were companion papers. Among the 60 studies, 16 look at physical processes, 10 examine vegetation changes, eight look at invertebrates, six investigate amphibians and reptiles, 26 examine birds, and one addresses mammals. Some of these studies investigate multiple taxa (e.g., bird and insect) in the same paper. A total of 39 physical processes, 117 plants, 65 insects, 63 amphibians and reptiles, 209 birds, and 10 mammal species were examined in the 43 studies (summarized in Table 5-3). Approximately 39% of these species showed no change. Changes in the other 61% included earlier ice-off and later freeze dates in inland lakes and streams, earlier breeding times, shifting to higher elevations or latitudes, and changes in densities, development, morphologies, and genetics.

Several lines of evidence indicate lengthening of the vegetative growing season by 1.2–3.6 days per decade in the Northern Hemisphere, particularly at higher latitudes where temperature rise also has been greatest. This lengthening involved earlier onset of spring and later onset of fall. Summer photosynthetic activity [based on Normalized Differential Vegetation Index (NDVI) estimates from satellite data] increased from 1981 to 1991 (Myneni *et al.,* 1997), concurrent with an advance (by 7 days) and an increase in amplitude of the annual $CO_2$ cycle since the 1960s, most intensely during the 1980s (Keeling *et al.,* 1996). Phenological/climate models for Finland indicate an overall increase in growing season length since 1900 (Carter, 1998). These physical measures are in accord with observations on organisms. In controlled, mixed-species gardens across Europe, a lengthening of the growing season by 10.8 days occurred from 1959 to 1993 (Menzel and Fabian, 1999). Likewise, a study of 36 species in the central United States documented advances in flowering dates by an average of 7.3 days from 1936 to 1998 (Bradley *et al.,* 1999).

Responses to increased atmospheric $CO_2$ have been detected in increased stomatal densities in the leaves of temperate woodland plants (Beerling and Kelly, 1997). Recent changes (over 9- to 30-year periods) in community composition have occurred at protected sites in the lower United States and Alaska, concurrent with local warming trends (Chapin *et al.,* 1995; Brown *et al.,* 1997a; Alward *et al.,* 1999). Results of warming experiments coupled with previous knowledge of species' habitat requirements implicate climate as one factor in these community reorganizations, but additional effects of multiple pressures have led to complex responses that were not always predicted by bioclimatic theory (Schneider and Root, 1996).

Multiple studies of treelines at high latitudes in the northern hemisphere have shown 20th century poleward shifts, often measured as increased growth at northern boundaries and decreased growth at southern boundaries. Interpretation of these trends is not straightforward because most change occurred during the early 20th century warming and the trends have been less pronounced or absent in recent warm decades (Kullman, 1986, 1990; Hamburg and Cogbill, 1988; Innes, 1991; Lescop-Sinclair and Payette, 1995; Jacoby and D'Arrigo, 1995; Briffa *et al.,* 1998). These authors have hypothesized that the general lack of response to recent warming is a result of increases in water stress, severity of insect attack, and UV radiation and trends toward earlier snowmelt or to sunlight becoming a limiting growth factor. In addition, some localities that showed warming and increased growth in the early 20th century have shown cooling and stable growth since the1970s (Kullman, 1991, 1993). In contrast, simple predictions of range shifts have been fulfilled in alpine herbs, which have moved to higher altitudes concurrent with warming in Switzerland (Grabherr *et al.,* 1994), and loss of low-elevation pine forests in Florida as sea-level rise has caused toxic levels of salination near coastal areas (Ross *et al.,* 1994).

### *5.2.2. Current Models of Ecosystem Change*

A large literature is developing on modeling the response of ecosystems to climate and global changes. Most of these models simulate changes in a small patch of land. These models are reviewed as appropriate in other sections of this chapter and elsewhere in this report. The focus here is on modeling changes in ecosystem composition, structure, and function at global or regional scales. There are several reasons for developing such models. One is to estimate carbon fluxes and their contribution to the global carbon cycle. This involves making estimates of NPP, net ecosystem productivity (NEP), and net biome productivity (NBP) (see Box 5-1). Another is to develop models of feedbacks

**Box 5-1. Plant Productivity:**
**Terms and Definitions**

Plants are responsible for the vast majority of carbon uptake by terrestrial ecosystems. Most of this carbon is returned to the atmosphere via a series of processes, including respiration, consumption (followed by animal and microbial respiration), combustion (i.e., fires), and chemical oxidation. Gross primary productivity (GPP) is the total uptake through photosynthesis, whereas net primary productivity (NPP) is the rate of accumulation of carbon after losses from plant respiration and other metabolic processes in maintaining the plant's living systems are taken into account. Consumption of plant material by animals, fungi, and bacteria (heterotrophic respiration) returns carbon to the atmosphere, and the rate of accumulation of carbon over a whole ecosystem and over a whole season (or other period of time) is called net ecosystem production (NEP). In a given ecosystem, NEP is positive in most years and carbon accumulates, even if only slowly. However, major disturbances such as fires or extreme events that cause the death of many components of the biota release greater than usual amounts of carbon. The average accumulation of carbon over large areas and/or long time periods is called net biome productivity (NBP) (see also Box 3-1 in TAR WGI).

between the atmosphere and the land surface (Van Minnen *et al.,* 1995; Foley *et al.,* 1998). Neither of these applications are covered in detail in this chapter (see TAR WGI Chapter 3). Instead, we concentrate on their application in forecasting the impacts of climate change on biodiversity and the provision of other ecosystem goods and services.

### *5.2.2.1. Two Paradigms Describing Ecosystem Responses to Climate Change*

There are two paradigms about the way ecosystems (thus biomes) will respond to global change. The ecosystem movement paradigm assumes that ecosystems will migrate relatively intact to new locations that are closer analogs to their current climate and environment. This paradigm clearly is a gross simplification of what will actually happen, but it has the advantage that the well-demonstrated relationship between ecosystem range and existing climate can be used to project new ecosystem distributions under changed climate scenarios.

Basic ecological knowledge suggests that the ecosystem movement paradigm is most unlikely to occur in reality because of different climatic tolerance of species involved, including intra-species genetic variability (Crawford, 1993); different longevities, including clonal regeneration (e.g., survival over 2,000 years by *Carex curvula*; Steinger *et al.,* 1996); different migration abilities (Pitelka and Plant Migration Workshop Group, 1997); and the effects of invading species (Dukes and Mooney, 1999). It is an idealized working paradigm that is useful for screening scenarios of climate change for potential significant effects.

The alternative paradigm, ecosystem modification, assumes that as climate and other environmental factors change there will be *in situ* changes in species composition and dominance. These changes will occur as some species decline in abundance or become locally extinct (Jackson and Weng, 1999) and others increase in abundance. The longevity of individuals, the age structure of existing populations, and the arrival of invading species will moderate these changes. The outcome will be ecosystem types that may be quite different from those we see today. Paleoecological data indicate that ecosystem types broadly similar to those seen today did exist in the past (Pregitzer *et al.*, 2000), but that there also occurred combinations of dominant species that are not observed today (Davis, 1981; Jablonski and Sepkoski, 1996; Ammann *et al.,* 2000; Prieto, 2000).

Numerous paleoecological studies provide evidence of important species within an ecosystem responding differently to climate change. For example, the postglacial migration pattern for Sierra lodgepole pine (*Pinus contorta* ssp. m*urrayana*) was largely elevational, with little migrational lag, whereas the more widely distributed Rocky Mountain subspecies (*P. contorta* var. *latifolia*) migrated both latitudinally and elevationally (Anderson, 1996). Colinvaux *et al.* (1997) have interpreted pollen data to show that during glacial cooling, Andean vegetation did not move upslope and downslope as belts but that plant associations were reorganized as temperature-sensitive species found different centers of distribution with changing temperature. Similarly, heat-intolerant plants have moved in and out of the Amazonian rainforests during periods of cooling and warming (Colinvaux *et al.*, 2000). Colinvaux *et al.* (2000) also argue that expulsion of heat-intolerant species from the lowland forests in this postglacial warming already is complete and that the forest property of maintaining its own microhabitat will allow high species richness to survive more global warming, provided large enough tracts of forest are conserved. This conclusion is at odds with some modeling studies (e.g., White *et al.*, 1999).

The problem with the ecological modification paradigm is that it is very difficult to use in practical forecasting of possible trends. Thus, most global and regional studies assessing the potential impacts of climate change have had to use the ecosystem paradigm, as illustrated in Box 5-2. They also tend to be limited to projecting changes in vegetation distributions, with the implicit assumption that animal populations will track the vegetation components of an ecosystem. However, observational and experimental studies show many cases in which animals are responding to climate and environmental change well before any significant changes in the vegetation (see Section 5.4).

### *5.2.2.2. Climate or Environment Envelope Models*

Climate envelope models have been used since at least the 1980s (e.g., Box, 1981; Busby, 1988). Usually they are based on describing the climate or environment encompassing the current distribution of a species or ecosystem (the environmental envelope or climatic envelope), then mapping the location of this same envelope under a climate change scenario. Sometimes, other forms of bioclimatic correlations or categorization systems are used, but they are based on the same assumption of a close correlation between climate and ecosystems associated with it. These models are useful as a first screening device to point to potential significant changes in ecosystem composition resulting from climate change (e.g., Huntley, 1995, and Huntley *et al.*, 1995 for European plants and birds; Somaratne and Dhanapala, 1996, for Sri Lankan plants; Brereton *et al.*, 1995, for Australia mammals; Gignac *et al.,* 1998, for Canadian peatlands; Eeley *et al.*, 1999, for South African woody plants; Iverson and Prasad, 1998, Iverson *et al.*, 1999, for eastern U.S. trees).

A basic problem with these models is that every species has a "potential niche"—that is, a location in which it could survive and reproduce under the climatic conditions (Kirschbaum *et al.*, 1996). What is observed and used as the basis for the model, however, is the "realized niche," which is a more limited area in which the species is found given the effects of competitors, predators, and diseases. If climate change results in species remaining within their potential niche and the competitors and predators change, the species may be able to survive *in situ*. In some models, some locations are predicted to have climates that are not encompassed by any of the climate envelopes of any of the vegetation types (Lenihan and Neilson, 1995). Used cautiously, however, the approach can point to

**Box 5-2. Illustration of Use and Limitation of Ecosystem Movement Models**

The study by Malcolm and Markham (2000) is a good example of modeling that uses the ecosystem movement paradigm, but it also demonstrates the inherent weaknesses of the approach.

The study uses two models of existing ecosystem distributions (MAPSS and BIOME3; Neilson *et al.*, 1998) and compares predicted distributions at present $CO_2$ levels with the equilibrium climate associated with doubled $CO_2$ as projected by several general circulation models (GCMs). It avoids the naive assertion that the latter climate constitutes a forecast of the future distribution of ecosystems; instead, it uses the two predictions to calculate the necessary rate of migration (m $yr^{-1}$) for species in the ecosystems to migrate to the new locations within 100 years (other time frames also are explored in a sensitivity analysis). It then maps these required rates to show areas where unusually high rates may be required in the future if a "climatically appropriate" ecosystem is to be established (referred to as "migration-stressed" locations).

The study predicted that about 20% of the Earth's surface will require migration rates ≥1 km $yr^{-1}$, which is equivalent to the highest rates observed in the geological past. The effects of natural barriers (e.g., lakes) or barriers resulting from land-cover modification by humans are globally small but can be regionally significant. Their approach also gives an indication of which regions of the globe may be most likely to be migration-stressed by climate change. It shows that much of the Earth's surface will be "stressed" in at least one of the 14 combinations of vegetation and climate models used. For some regions in the northern boreal zones of Eurasia and North America, most of the models predict such stress.

The study then goes on to deal with locations where the models predict that under climates applying a doubled $CO_2$ scenario, current ecosystems will fall outside their climatic range (referred to as "climate-stressed" locations). One must be careful not to attribute specific impacts or changes to climate-stressed locations. Biomes (or, more correctly, species constituting ecosystems of the biome) may be able to tolerate the new climatic conditions (i.e., new conditions fall within the potential niche) and thus may be relatively little changed. This same proviso should be applied to migration-stressed locations (i.e., existing vegetation may continue to occupy the site; thus, the migratory restriction does not come into play for decades to centuries).

The authors move on to equate climate-stressed locations with habitat loss and conclude that 36% of the land area will be affected. For aforementioned reasons, this must be regarded as an upper bound. The authors then attempt to estimate the reduction in habitat patch size by counting pixels affected by climate-stress in contiguous blocks of the same biome type, then applying a simple species area relationship to estimate species loss (see McDonald and Brown, 1992). Little reliance should be placed on these estimates given the foregoing provisos and the caveats listed by the authors themselves.

priority areas for further study. For example, Lassiter *et al.* (2000) analyzed data for 200 woody plant species in the eastern United States and found the usual result: a northward shift in predicted range. Excluding water limitations (i.e., allowing increased WUE under elevated $CO_2$ to compensate for a drier climate), however, most species survive in their current locations. This implies the need to better understand water-use and water-availability relationships under climate change. If species are able to persist, there may be little spread into new regions and thus little vegetation change for many decades.

In a study of 80 species of trees in the United States, using 100,000 plots to describe current distribution and importance, nearly half of the species assessed showed the potential for ecological optima to shift at least 100 km to the north (Iverson *et al.*, 1999). Whether these species will be able to achieve these potential distributions will depend on their migration rates through fragmented landscapes. When Iverson *et al.* (1999) incorporated a migration model, they found severely limited migration in regions of high forest fragmentation, particularly when the species is low in abundance near the range boundary.

Kirilenko and Solomon (1998) also have developed a bioclimate correlative model that incorporates a migration component. They demonstrate the importance of incorporating migration by showing that in simulations in which tree migration is delayed, the estimated global terrestrial carbon stock decreases by 7–34 Gt C, in contrast to an increase in carbon stock projected under nonlimiting migration (Solomon and Kirilenko, 1997).

Climate envelopes continue to be a useful tool in identifying potential changes in species distributions but are limited by the foregoing problems. Emphasis is shifting to more mechanistically based dynamic models of vegetation change that are linked directly to transient outputs from GCMs.

#### 5.2.2.3. *Toward Dynamic Global and Regional Ecosystem Models*

Since the SAR, there have been significant changes in vegetation models and GCMs. A major modeling comparison project—Vegetation/Ecosystem Modeling and Analysis Project (VEMAP)—compared six equilibrium models of vegetation distribution and biogeochemistry (VEMAPMembers, 1995). In the IPCC *Special Report on the Regional Impacts of Climate Change*, Neilson *et al.* (1998) compared two leading vegetation models (MAPSS and BIOME3) run against transient outputs from recent GCMs.

In MAPPS and BIOME3, potential vegetation is simulated by first calculating carbon flux (plant growth) on the basis of climate and hydrology information derived from the GCM. The models calculate a leaf area index that represents the capacity of the site to support plant canopy. Physiologically based rules are then used to classify the site into a vegetation type (e.g., forest, shrub, or grassland; evergreen or deciduous), leading to allocation to one of 45 vegetation types in MAPSS or 18 in BIOME3. These vegetation models are equilibrium models in that they calculate which vegetation type might be most suited to the climate and do not consider how the existing vegetation type might change to that new type. Details of the model outputs and maps are presented in IPCC (1998) and Neilson *et al.* (1998).

Neilson *et al.* (1998) compared the more recent combination of vegetation and climate models with those being used in the SAR. They found that the newer, transient GCMs produce a cooler climate than earlier versions, mostly because lag effects are simulated and the temperature does not increase to its equilibrium value within the model runs. This means that changes in predicted vegetation tend to be less than those reported in the SAR. Nevertheless, there are significant poleward shifts in cold-limited vegetation types. Vegetation types that are limited by water availability showed more complex changes, depending on the balance between precipitation change, hydrological balance, and physiological adjustment under higher $CO_2$.

The newer modeling combinations continue to predict significant potential changes in the distribution of most ecosystem types and increases in the area of tropical and temperate forests, but it cannot be determined whether this potential will be observed until transient (or dynamic) global vegetation models are developed. These models simulate the change in abundance of important species or "functional groups" of species on a year-by-year (or similar) basis in response to the output of the GCM (Cramer *et al.,* 1999).

Such models are being developed and used for assessments of overall carbon storage potential of the land biosphere (Cramer *et al.*, 2001), but at this stage it is too early to place much reliance on the outputs for specific biomes or ecosystems. The results show the sensitivity of ecosystems to the treatment of water use and especially the balance between changes in water availability resulting from climate change (often decreased availability in a warmer climate) and response to higher $CO_2$ concentrations in the atmosphere (often increased water-use efficiency). This means that model output can vary significantly, depending on the GCM used, because these models have tended to produce different interannual variability in precipitation and thus water availability. Other challenges are to simulate loss of vegetation from disturbances such as fire, blowdown, or pest attacks and migration of species or groups of species to new locations. Van Minnen *et al.* (2000) have shown that by modifying the IMAGE2 model to include unlimited migration, limited migration and no migration result in significantly different patterns of vegetation change, especially in high-latitude regions.

### *5.2.3. Impacts on Biodiversity*

Biodiversity is assessed quantitatively at different levels—notably at the genetic level (i.e., the richness of genetically different types within the total population), the species level (i.e., the richness of species in an area), and the landscape level (i.e., the richness of ecosystem types within a given area). Overall, biodiversity is forecast to decrease in the future as a result of a multitude of pressures, particularly increased land-use intensity and associated destruction of natural or semi-natural habitats (Heywood and Watson, 1996). The most significant processes are habitat loss and fragmentation (or reconnection, in the case of freshwater bodies); introduction of exotic species (invasives); and direct effects on reproduction, dominance, and survival through chemical and mechanical treatments. In a few cases, there might be an increase in local biodiversity, but this usually is a result of species introductions, and the longer term consequences of these changes are hard to foresee.

These pressures on biodiversity are occurring independent of climate change, so the critical question is: How much might climate change enhance or inhibit these losses in biodiversity? There is little evidence to suggest that processes associated with climate change will slow species losses. Palaeoecology data suggest that the global biota should produce an average of three new species per year, with large variation about that mean between geological eras (Sepkoski, 1998). Pulses of speciation sometimes appear to be associated with climate change, although moderate oscillations of climate do not necessarily promote speciation despite forcing changes in species' geographical ranges.

Dukes and Mooney (1999) conclude that increases in nitrogen deposition and atmospheric $CO_2$ concentration favor groups of species that share certain physiological or life history traits that are common among invasive species, allowing them to capitalize on global change. Vitousek *et al.* (1997b) are confident that the doubling of nitrogen input into the terrestrial nitrogen cycle as a result of human activities is leading to accelerated losses of biological diversity among plants adapted to efficient use of nitrogen and animals and microorganisms that depend on them. In a risk assessment of Switzerland alpine flora, Kienast *et al.* (1998) conclude that species diversity could increase or at least remain unchanged, depending on the precise climate change scenario used.

#### *5.2.3.1. Global Models of Biodiversity Change*

Several general principles describe global biodiversity patterns in relation to climate, evolutionary history, isolation, and so forth. These principles continue to be the subject of considerable ecological theory and testing; the Global Biodiversity Assessment

(Heywood and Watson, 1996) and the Encyclopaedia of Biodiversity (Levin, 2000) contain detailed reviews.

Kleidon and Mooney (2000) have developed a process-based model that simulates the response of randomly chosen parameter combinations ("species") to climate processes. They demonstrate that the model mimics the current distribution of biodiversity under current climate and that modeled "species" can be grouped into categories that closely match currently recognized biomes. Sala *et al.* (2000) used expert assessment and a qualitative model to assess biodiversity scenarios for 2100. They conclude that Mediterranean climate and grassland ecosystems are likely to experience the greatest proportional change in biodiversity because of the substantial influence of all drivers of biodiversity change. Northern temperate ecosystems are estimated to experience the least biodiversity change because major land-use change already has occurred.

Modeling to date demonstrates that the global distribution of biodiversity is fundamentally constrained by climate (see Box 5-2). Future development along these lines (e.g., adding competitive relations and migration processes) could provide useful insights into the effect of climate change on biodiversity and the effects of biodiversity on fluxes of carbon and water on a global scale.

### *5.2.4. Challenges*

There has been considerable progress since the SAR on our understanding of effects of global change on the biosphere. Observational and experimental studies of the effects of climate change on biological and physical processes have increased significantly, providing greater insights into the nature of the relationships. Greater biological realism has been incorporated into models of small patches of vegetation (point models), and more realistic biological representations have been incorporated into regional and global change models. The main improvement has been development of dynamic representations of biological processes that respond directly to climate. Nevertheless, several major challenges remain before fully effective models of the interaction between climate and biophysical processes will be available.

#### *5.2.4.1. Landscape Processes*

Most vegetation models still treat patches of vegetation as a matrix of discrete units, with little interaction between each unit. However, modeling studies (Noble and Gitay, 1996; Rupp *et al.*, 2000) have shown that significant errors in predicting vegetation changes can occur if spatial interactions of landscape elements are treated inadequately. For example, the spread of fires is partly determined by the paths of previous fires and subsequent vegetation regrowth. Thus, the fire regime and vegetation dynamics generated by a point model and a landscape model with the same ignition frequencies can be very different. There has been considerable progress in modeling of spatial patterns of disturbances within landscapes (Bradstock *et al.*, 1998; He and Mladenoff, 1999; Keane *et al.*, 1999), but it is not possible to simulate global or regional vegetation change at the landscape scale. Thus, the challenge is to find rules for incorporating landscape phenomena into models with much coarser resolution.

Another challenge is to develop realistic models of plant migration. On the basis of paleoecological, modeling, and observational data, Pitelka and Plant Migration Workshop Group (1997) conclude that dispersal would not be a significant problem for most species in adapting to climate change, provided that the matrix of suitable habitats was not too fragmented. However, in habitats fragmented by human activities that are common over much of the Earth's land surface, opportunities for migration will be limited and restricted to only a portion of the species pool (Björkman, 1999).

#### *5.2.4.2. Will Organisms have to Migrate?*

Section 5.2.2.2 raises the question of whether organisms will need to migrate under climate change or whether many species will be able to survive *in situ* under new climatic conditions (Woodward and Beerling, 1997). Some species do occupy sites that are on the limits of their physiological tolerance, and if climate change takes local climate beyond that threshold, clearly they will not be able to persist at that site (see Section 5.6 for examples). However, there is mounting paleoecological evidence of vegetation types persisting through significant climate changes.

Lavoie and Payette (1996) conclude that the stability of the black spruce (*Picea mariana* [Mill.] BSP.) forest boundary during warm and cold periods of the late Holocene [warm approximately 2,000 years before present (BP), medieval times, and this century; cold approximately 3,000 years BP, 1,300 years BP, and the Little Ice Age] demonstrates that mechanisms that allow forest boundaries to advance or retreat are not easily triggered by climatic change. The black spruce old growth forest persisted for more than 1,500 years through many variations in climate probably because of the buffering effect of the trees on the local environment (Arseneault and Payette, 1997). Only a fire in 1568 AD broke this forest influence on microclimate and local growth conditions and caused the forest vegetation to shift to krummholz. Such studies emphasize the importance of effectively incorporating sufficient biophysical detail to capture climate-ameliorating effects and realistic disturbance regimes.

#### *5.2.4.3. Human Land-Use Issues*

No projection of the future state of the earth's ecosystems can be made without taking into account past, present, and future human land-use patterns. Human land-use will endanger some ecosystems, enhance the survival of others, and greatly affect the ability of organisms to adapt to climate change via migration.

Leemans (1999) used IMAGE2 to forecast possible global shifts in vegetation types and land-use change. Using a mid-level scenario of human responses to global change, he forecasts that nondomesticated land (his proxy for biodiversity) will decrease from 71 to 62% of the land area between 2000 and 2025, then remain approximately stable until the end of the 21st century. Losses in Africa and parts of Asia from 2000 to 2025 may be as much as 20–30% of remaining nondomesticated land.

#### *5.2.4.4. Testing Models of Ecosystem Response*

It is clear that ecosystem change models still contain assumptions that are not fully tested, and most models inevitably work better for the geographical regions and time periods for which they were constructed (Hurtt *et al.,* 1998). This derives from a mixture of fitting parameters to available data and deliberate and subconscious bias in selecting processes to include in a model. However, testing of ecosystem response models is gradually improving. Bugmann and Solomon (1995) tested the behavior of a model developed for European ecosystems by running it for a comparable North American site. The results were broadly in agreement, with useful indicators about where the model could be improved.

Another test is to compare different approaches to modeling for a particular purpose, as Yates *et al.* (2000) did for correlative models such as the Holdridge climatic correlation model and more mechanistic models of vegetation distribution. The strengths, weaknesses, and appropriate areas of application can be determined. Another approach is to conduct sensitivity studies in which the sensitivity of model outputs to changes in the input data and assumptions is evaluated. Hallgren and Pitman (2000) have carried out a sensitivity analysis of BIOME3 (Haxeltine and Prentice, 1996); they conclude that parameters that affect photosynthesis, water use, and NPP change the competitive interactions between specific plant groups (e.g., $C_3$ versus $C_4$ plants). Numerous studies have found that the outputs of ecosystem response models are sensitive to the precise treatment of water availability and water use by vegetation (Gao and Yu, 1998; Churkina *et al*., 1999; Hallgren and Pitman, 2000; Lassiter *et al.,* 2000). This points to areas in which further development is needed.

Beerling *et al.* (1997) used a climate change experiment (CLIMEX) that exposed an entire catchment of boreal vegetation to elevated $CO_2$ and temperature for 3 years to test their Sheffield dynamic global vegetation model. There generally was a good match between observations and predictions, but longer runs of such experiments are needed to test such models thoroughly. As paleoecological data sets improve, there is increasing opportunity to test models against these data. However, this is a multiple test because the validity of paleoclimate models themselves also is under test. For example, Kohfeld and Harrison (2000) tested the ability of GCMs to describe changes in data collected for the environments of the last glacial maximum (21,000 years BP) and/or mid-Holocene (6,000 years BP). They conclude that better land-surface (including vegetation) response models are needed to capture the detail observed in the paleoecological data.

In summary, recent studies show that the potential for significant disruption of ecosystems under climate change remains. Further development of simple correlative models that were available at the time of the SAR points to areas where ecosystem disruption and the potential for ecosystem migration are high. Observational data and newer dynamic vegetation models linked to transient climate models are refining the projections. However, the precise outcomes depend on processes that are too subtle to be fully captured by current models.

## 5.3. Agriculture

Conclusions regarding the consequences of climate change for the agriculture sector in the SAR (Reilly *et al.,* 1996) provide an important benchmark for this section. The focus in this section is on basic mechanisms and processes that regulate the sensitivity of agriculture to climate change, relying mostly on research results since the SAR. Specifically, we ask how the conclusions of the SAR have stood the test of new research. Research advances since the SAR have brought several new issues to light—for example, understanding the adaptation of agriculture to climate change.

The discussion in this section is guided by the State-Pressure-Impact-Response-Adaptation model (see Figure 5-1). The pace of social, economic, and technological change in the agriculture sector will steadily transform the setting in which climate change is likely to interact with sensitive features of the food system. The current *state* of the sector and important trends that would transform it provide a baseline against which to examine the potential consequences of climate change (Section 5.3.1). Multiple *pressures* are being exerted on the agriculture sector, including the need to meet rising demand for food and fiber, resource degradation, and a variety of environmental changes (Section 5.3.2). Agricultural impacts, response, and adaptation are discussed concurrently because they are inseparable parts of the calculus of the vulnerability of agricultural systems to climate change. Hence, we consider the response and adaptive potential of agriculture in each of the succeeding sections. Agriculture is likely to *respond* initially to climate change through a series of automatic mechanisms. Some of these mechanisms are biological; others are routine adjustments by farmers and markets. Note that we equate response with automatic adaptation, as discussed in Chapter 18.

Climate change will *impact* agriculture by causing damage and gain at scales ranging from individual plants or animals to global trade networks. At the plant or field scale, climate change is likely to interact with rising $CO_2$ concentrations and other environmental changes to affect crop and animal physiology (Section 5.3.3). Impacts and adaptation (agronomic and economic) are likely to extend to the farm and surrounding regional scales (Section 5.3.4). Important new work also models agricultural impacts and adaptation in a global economy (Section 5.3.5).

Finally, the vulnerabilities of the agriculture sector, which persist after taking account of adaptation, are assessed (Section 5.3.6).

### *5.3.1. State of the Global Agricultural Sector*

As Reilly *et al.* (1996) argue in the SAR, one of the foremost goals for global agriculture in coming decades will be expansion of the global capacity of food and fiber in step with expansion of global demand. Agriculture in the 20th century accomplished the remarkable achievement of increasing food supply at a faster rate than growth in demand, despite rapidly growing populations and per capita incomes. Key summary indicators of the balance between global demand and supply are world prices for food and feed grains. Johnson (1999) and Antle *et al.* (1999a) show that during the second half of the 20th century, real (inflation-adjusted) prices of wheat and feed corn have declined at an average annual rate of 1–3%. Climate change aside, several recent studies (World Bank, 1993; Alexandratos, 1995; Rosegrant *et al.*, 1995; Antle *et al.*, 1999a; Johnson, 1999) anticipate that aggregate food production is likely to keep pace with demand, so that real food prices will be stable or slowly declining during the first 2 decades of the 21st century.

According to the U.S. Department of Agriculture (1999), food security[1] has improved globally, leading to a decline in the total number of people without access to adequate food. The declining real price of food grains has greatly improved the food security of the majority of the world's poor, who spend a large share of their incomes on these staples. The global number, however, masks variation in food security among regions, countries, and social groups that are vulnerable because of low incomes or a lack of access to food (FAO, 1999a). In lower income countries, political instability and inadequate physical and financial resources are the root causes of the food security problem (see Section 5.3.6). In higher income, developing countries, food insecurity stems from unequal distribution of food that results from wide disparities in purchasing power.

Agricultural production and trade policies also affect global food availability and food security. There is a widespread tendency for high-income countries to maintain policies that effectively subsidize agricultural production, whereas low-income countries generally have policies that tax or discourage agricultural production (Schiff and Valdez, 1996). Many low-income countries also pursue policies that promote food self-sufficiency. Although all of these policies tend to reduce the efficiency of agricultural resource utilization in low- and high-income countries, they have not changed long-run trends in global supply and demand (Antle, 1996a).

Relatively few studies have attempted to predict likely paths for food demand and supply beyond 2020. There are reasons for optimism that growth in food supply is likely to continue apace with demand beyond 2020. For example, population growth rates are projected to decline into the 21st century (Bos *et al.*, 1994; Lutz *et al.*, 1996; United Nations, 1996), and multiple lines of evidence suggest that agricultural productivity potential is likely to continue to increase. Rosegrant and Ringler (1997) project that current and future expected yields will remain below theoretical maximums for the foreseeable future, implying opportunities for further productivity growth.

Other analysts are less optimistic about long-term world food prospects. For example, there is evidence that the Asian rice monoculture may be reaching productivity limits because of adverse impacts on soils and water (Pingali, 1994). Tweeten (1998) argues that extrapolation of the downward trend in real food prices observed in the latter half of the 20th century could be erroneous because the supply of the best arable land is being exhausted and rates of productivity growth are declining. At the same time, demand is likely to continue to grow at reasonably high rates well into the 21st century. Other studies indicate concerns about declining rates of investment in agricultural productivity and their impacts on world food production in some major producing and consuming areas (Hayami and Otsuka, 1994; Rozelle and Huang, 1999). Ruttan (1996) indicates that despite advances in biotechnology, most yield improvements during the first decades of the 21st century are likely to continue to come from conventional plant and animal breeding techniques. These concerns about future productivity growth, if correct, mean that simple extrapolation of yield for impact assessment (e.g., Alexandratos, 1995) may be overoptimistic. The implication is that confidence in predictions of the world food demand and supply balance and price trends beyond the early part of the 21st century is low.

### *5.3.2. Pressures on Agriculture Sector*

#### *5.3.2.1. Degradation of Natural Resources*

Degradation of natural resources—taken here as soils, forests, marine fisheries, air, and water—diminishes agricultural production capacity (Pinstrup-Andersen and Pandya-Lorch, 1998). Soil degradation emerges as one of the major challenges for global agriculture. It is induced via erosion, chemical depletion, water saturation, and solute accumulation. In the post-World War II period, approximately 23% of the world's agricultural land, permanent pastures, forests, and woodland were degraded as defined by the United Nations Environment Programme (UNEP) (Oldeman *et al*., 1991). Various estimates put the annual loss of land at 5–10 Mha $yr^{-1}$ (Scherr and Yadav, 1997). Although irrigated land accounts for only 16% of the world's cropland, it produces 40% of the world's food. There are signs of a slowing in the rate of expansion of irrigation: 10–15% of irrigated land is degraded to some extent by waterlogging and salinization (Alexandratos, 1995). Degradation of natural resources is likely to hinder increases in agricultural productivity and could dim optimistic assessments of the prospects of satisfying growing world food demand at acceptable environmental cost.

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[1] Food security often is defined as "access by all peoples at all times to enough food for an active, healthy life" (Chen and Kates, 1996).

#### 5.3.2.2. Other Global Change Factors

Regional scenarios of seasonal temperature and precipitation change for 32 world regions analyzed in Chapter 3 show the current variability of climate and the range of changes predicted by GCMs for 30-year time periods centered on 2025, 2055, and 2085. This background information is essential to interpret the potential impacts of climate change on crops and livestock production. Equally important background information is provided by agroclimatic indices. Agroclimatic indices are useful in conveying climate variability and change in terms that are meaningful to agriculture. They give a first approximation of the potential effects of climate change on agricultural production and should continue to be used (Sirotenko *et al.*, 1995; Sirotenko and Abashina, 1998; Menzhulin, 1998).

Several other climate-related global environmental changes are likely to affect the agriculture sector in coming years. Reilly *et al.* (1996) reviewed the exposure of crops to tropospheric ozone ($O_3$). Progress in sorting out interactions between $O_3$, $CO_2$, and climate variability is reviewed below.

Climate change is likely to interact with other global changes, including population growth and migration, economic growth, urbanization, and changes in land use and resource degradation. Döös and Shaw (1999) use an accounting system to estimate the sensitivity of agricultural production to various aspects of global change, including loss of cropland from soil degradation and urbanization. Imhoff *et al.* (1997) use remote-sensing techniques and soils data to show that urbanization in the United States has occurred primarily on high-quality agricultural lands.

## 5.3.3. Response of Crops and Livestock and Impacts on Food and Fiber

### 5.3.3.1. Interaction between Rising $CO_2$ Concentrations and Climate Change

Advances in knowledge of $CO_2$ effects on crop and forage plants establish convincingly, although incompletely, that it is no longer useful to examine the impacts of climate change absent their interactions with rising atmospheric $CO_2$ (see Boxes 5-3 and 5-4). Crop and forage plants are likely to be forced to deal with the combined effects of climate change and rising atmospheric $CO_2$ concentrations. In this section, emphasis is placed on understanding basic interactions between plant productivity, climate change, and rising $CO_2$ concentrations. The direct effects of climate change on livestock also are considered.

#### 5.3.3.1.1. Interactive effects of temperature increase and atmospheric $CO_2$ concentration

Because temperature increase enhances photorespiration in $C_3$ species (Long, 1991), the positive effects of $CO_2$ enrichment on photosynthetic productivity usually are greater when temperature rises (Bowes *et al*., 1996; Casella *et al*., 1996). A rise in mean global nighttime temperatures (Horton, 1995)

**Box 5-3. Impacts of Climate Change and Elevated $CO_2$ on Grain and Forage Quality from Experimentation**

The importance of climate change impacts on grain and forage quality emerges from new research. For rice, the amylose content of the grain—a major determinant of cooking quality—is increased under elevated $CO_2$ (Conroy *et al.*, 1994). Cooked rice grain from plants grown in high-$CO_2$ environments would be firmer than that from today's plants. However, concentrations of iron and zinc, which are important for human nutrition, would be lower (Seneweera and Conroy, 1997). Moreover, the protein content of the grain decreases under combined increases of temperature and $CO_2$ (Ziska *et al.*, 1997).

With wheat, elevated $CO_2$ reduces the protein content of grain and flour by 9–13% (Hocking and Meyer, 1991; Conroy *et al.*, 1994; Rogers *et al.*, 1996a). Grain grown at high $CO_2$ produces poorer dough of lower extensibility and decreased loaf volume (Blumentahl *et al*., 1996), but the physiochemical properties of wheat starch during grain fill are not significantly modified (Tester *et al*., 1995). Increases in daily average temperatures above 30°C, even applied for periods of up to 3 days, tend to decrease dough strength (Randall and Moss, 1990). Hence, for breadmaking, the quality of flour produced from wheat grain developed at high temperatures and in elevated $CO_2$ degrades.

With high-quality grass species for ruminants, elevated $CO_2$ and temperature increase have only minor impacts on digestibility and fiber composition of cut material (Akin *et al.*, 1995; Soussana *et al.*, 1997). The large increase in water-soluble carbohydrates in elevated $CO_2$ (Casella and Soussana, 1997) could lead to faster digestion in the rumen, whereas declines in nitrogen concentration occurring mainly with $C_3$ species (Owensby *et al.*, 1994; Soussana *et al.*, 1996; Read *et al*., 1997) reduce the protein value of the forage. The protein-to-energy ratio has been shown to be more critical in tropical climates than in temperate countries (Leng, 1990). Livestock that graze low protein-containing rangeland forage therefore may be more detrimentally affected by increased C:N ratios than energy-limited livestock that graze protein-rich pastures (Gregory *et al.*, 1999). Basically, lowering of the protein-to-energy ratio in forage could reduce the availability of microbial protein to ruminants for growth and production, leading to more inefficient utilization of the feed base and more waste, including emissions of methane.

### Box 5-4. Elevated $CO_2$ Impacts on Crop Productivity: Recent Estimates with Field-Grown Crops under FACE Experimentation

The short-term responses to elevated $CO_2$ of plants grown in artificial conditions are notoriously difficult to extrapolate to crops in the field (Körner, 1995a). Moreover, with field-grown plants, enclosures tend to modify the plant's environment (Kimball *et al.*, 1997). However, even the most realistic free-air $CO_2$ enrichment (FACE) experiments undertaken to date create a modified area (Kimball *et al*., 1993), analogous to a single irrigated field in a dry environment, and impose an abrupt change in $CO_2$ concentration. A cotton crop exposed to FACE increased biomass and harvestable yield by 37 and 48%, respectively, in elevated (550 ppm) $CO_2$. This effect was attributed to increased early leaf area, more profuse flowering, and a longer period of fruit retention (Mauney *et al.*, 1994). At 550 ppm $CO_2$, spring wheat increased grain yields by 8–10% under well-watered conditions (Pinter *et al.*, 1996). More recent studies with optimal nitrogen and irrigation increased final grain yield by 15 and 16% for two growing seasons at elevated $CO_2$ concentration (550 ppm), compared with control treatments (Pinter *et al.*, 1996). If these latter results are linearly extrapolated to the possible effect of a doubling (700 ppm) of the current atmospheric $CO_2$ concentration, yields under ideal conditions would be 28% greater—in agreement with previous statements by Reilly *et al.* (1996). In grass-clover mixtures, the proportion of legume increased significantly under elevated $CO_2$ (Hebeisen *et al.*,1997)—a conclusion also reached by several experimental studies with temperate and fertile managed grasslands (Newton *et al.*, 1996; Soussana and Hartwig, 1996; Stewart and Potvin, 1996).

could enhance carbon losses from crops by stimulating shoot dark respiration (Amthor, 1997). Despite possible short-term effects of elevated $CO_2$ on dark respiration (Amthor, 1997; Drake *et al*., 1997), the long-term ratio of shoot dark respiration to photosynthesis is approximately constant with respect to air temperature and $CO_2$ concentration (Gifford, 1995; Casella and Soussana, 1997). With moderate temperatures, long-term doubling of current ambient $CO_2$ under field-like conditions leads to a 30% enhancement in the seed yield of rice, despite a 5–10% decline in the number of days to heading (Horie *et al*., 2000). The grain yield of $CO_2$-enriched rice shows about a 10% decline for each 1°C rise above 26°C. This decline is caused by a shortening of growth duration and increased spikelet sterility. Similar scenarios have been reported for soybean and wheat (Mitchell *et al*., 1993; Bowes *et al*., 1996). With rice, the effects of elevated $CO_2$ on yield may even become negative at extremely high temperatures (above 36.5 °C) during flowering (Horie *et al*., 2000). However, in some cropping systems with growth in the cooler months, increased rates of phenological development with warm temperatures and/or earlier planting dates may tend to move the grain fill period earlier into the year during the cooler months, offsetting at least part of the deleterious effects of higher temperatures (Howden *et al*., 1999a).

#### *5.3.3.1.2. Interactive effects of water availability and atmospheric $CO_2$ concentration*

Although stomatal conductance is decreased under elevated $CO_2$, the ratio of intercellular to ambient $CO_2$ concentration usually is not modified, and stomata do not appear to limit photosynthesis more in elevated $CO_2$ compared to ambient $CO_2$ (Drake *et al*., 1997). Elevated-$CO_2$ effects on crop evapotranspiration per unit land area (E) have been small with cotton (Dugas *et al*., 1994; Hunsaker *et al*., 1994; Kimball *et al*., 1994) and spring wheat (Kimball *et al*.,1995, 1999) crops supplied with ample nitrogen fertilizer. With rice, under field-like conditions, $CO_2$ enrichment reduced seasonal total E by 15% at 26°C but increased E by 20% at 29.5°C (Horie *et al*., 2000). A larger decline (-22%) in the daily E of a $C_4$-dominated tallgrass prairie was reported by Ham *et al.* (1995), and a strong reduction in water use per plant also was observed for maize (Samarakoon and Gifford, 1996), a $C_4$ plant. The consequences of these direct effects of elevated $CO_2$ concentrations on E are still unclear at the catchment scale (see Section 4.3.3).

Relative enhancement of growth owing to $CO_2$ enrichment might be greater under drought conditions than in wet soil because photosynthesis would be operating in a more $CO_2$-sensitive region of the $CO_2$ response curve (André and Du Cloux, 1993; Samarakoon and Gifford, 1995). In the absence of water deficit, $C_4$ photosynthesis is believed to be $CO_2$ saturated at present atmospheric $CO_2$ concentration (Bowes, 1993; see also Kirschbaum *et al.*, 1996). However, as a result of stomatal closure, it can become $CO_2$-limited under drought. Some of the literature examples in which $C_4$ crop species, such as maize, have responded to elevated $CO_2$ may have involved (possibly unrecognized) minor water deficits (Samarakoon and Gifford, 1996). Therefore, $CO_2$-induced growth enhancement in $C_4$ species (e.g., Poorter, 1993) may be caused primarily by improved water relations and WUE (Samarakoon and Gifford, 1996) and secondarily by direct photosynthetic enhancement and altered source-sink relationships (Ruget *et al.*, 1996; Meinzer and Zhu, 1998). With rice, at the optimal temperature for growth, a doubling of $CO_2$ increases crop WUE by about 50%. However, this increase in WUE declines sharply as temperature increases beyond the optimum (Horie *et al.*, 2000). Although increased productivity from increased WUE is the major response to elevated $CO_2$ in a $C_3$ or $C_4$ crop that is exposed frequently to water stress (Idso and Idso, 1994; Ham *et al.*, 1995; Drake *et al.*, 1997), changes in climatic factors (temperature, rainfall) may interact with elevated $CO_2$ to alter soil water status, which in turn will influence hydrology and nutrient relations. Therefore, to realistically project impacts on crop yields and regional evaporation (see Chapter 4), more research is needed on the interactions of elevated $CO_2$, high temperature, and precipitation.

#### 5.3.3.1.3. Interactive effects of atmospheric chemistry and $CO_2$ concentration

An exposure-response model that linearly relates a change in gas exposure over a time period to log-scale change in biomass increment of a plant during the same period suggests that a decline in recent yields of grain crops caused by an increase in surface ozone concentrations may have reached 20% in some parts of Europe (Semenov *et al*., 1997, 1998, 1999). Recent research has shown that multiple changes in atmospheric chemistry can lead to compensating or synergistic effects on some crops. Heagle *et al*. (1999) used field studies to examine the impact of higher $O_3$ levels on cotton growth under higher $CO_2$ conditions. They found that higher $CO_2$ compensates for growth suppression resulting from elevated $O_3$ levels. With wheat, elevated $CO_2$ fully protects against the detrimental effects of $O_3$ on biomass but not yield (McKee *et al*., 1997). Similar results have been reported with soybean (Fiscus *et al*., 1997) and tomato (Reinert *et al*., 1997). Meyer *et al*. (1997) measured responses of spring wheat to different levels of ozone in chambers at different growth stages and found that photosynthesis and carbohydrate accumulations were strongly affected during anthesis, especially after a period of heat stress.

Mark and Tevini (1997) observed combined effects of UV-B, temperature, and $CO_2$ in growth chambers in seedlings of sunflower and maize. They found that a 4°C rise in daily maximum temperature (from 28 to 32°C), with or without higher $CO_2$**,** compensated for losses from enhanced UV-B. Teramura *et al*. (1990) report that yield increases with elevated $CO_2$ are suppressed by UV-B more in cereals than in soybean; rice also loses its $CO_2$-enhanced WUE. However, Unsworth and Hogsett (1996) assert that many research studies from the preceding decade used unrealistic UV-B exposures, and they conclude that UV-B does not pose a threat to crops alone or in combination with other stressors.

### 5.3.3.2. Interactive Effects of $CO_2$ Concentrations, Climate Change, Soils, and Biotic Factors

#### 5.3.3.2.1. Interactive effects of $CO_2$ concentrations with soils

There is not yet any clear consensus regarding the magnitude and sign of interactions between elevated $CO_2$ and nutrient availability for crop growth. Reviews of available data indicate that, on average, plants grown at high nutrient supply respond more strongly to elevated $CO_2$ than nutrient-stressed plants (Poorter, 1993, 1998). Nevertheless, the current rise in atmospheric $CO_2$ concentration may help plants cope with soil nutritional deficiencies (Idso and Idso, 1994) and especially with low nitrogen availability (Lloyd and Farquhar, 1996; Drake *et al.*, 1997). Several authors emphasize that a strong increase in biomass production under elevated $CO_2$ cannot be sustained in low fertilizer input systems without an appropriate increase in nutrients assimilation (Comins and McMurtrie, 1993; Gifford, 1994; Schimel, 1998). When other nutrients are not strongly limiting, a decline in nitrogen availability could be prevented by an increase in biological $N_2$ fixation under elevated $CO_2$ (Gifford, 1992, 1994). In fertile grasslands, legumes benefit more from elevated $CO_2$ than nonfixing species, resulting in significant increases in symbiotic $N_2$ fixation (Soussana and Hartwig, 1996; Zanetti *et al.*, 1996).

Plants grown under elevated $CO_2$ generally increase the allocation of photosynthates to roots (Rogers *et al*., 1996b; Murray, 1997), which increases the capacity and/or activity of belowground carbon sinks (Rogers *et al.*, 1994; Canadell *et al.*, 1996; Körner, 1996), enhancing root turnover (Pregitzer *et al.*, 1995; Loiseau and Soussana, 1999b), rhizodeposition (Cardon, 1996), and mycorrhizal development (Dhillion *et al.*, 1996) in some but not all systems. Some measurements also have shown an increase in soil N cycling (Hungate *et al.*, 1997a), in response to short-term enrichment in $CO_2$, although other studies have shown either no detectable change (Prior *et al.*, 1997) or even a reduction in soil N mineralization (Loiseau and Soussana, 2001). The relationships between C and N turnover in soils after exposure to elevated $CO_2$ therefore are not fully understood, and it is still a matter of debate whether the availability of soil nitrogen for crop plants is reduced after a step increase in atmospheric $CO_2$ concentration.

Soil organic carbon (SOC) stocks result from the balance between inputs and decomposition of soil organic matter (SOM). Residues of cotton (Torbert, *et al*., 1995), soybean, and sorghum (Henning *et al.*, 1996) display increased C:N ratios from growth under elevated $CO_2$, which may reduce their rate of decomposition in the soil and lead to an increment in ecosystem carbon stocks, similar to that observed in fertile grasslands (Casella and Soussana, 1997; Loiseau and Soussana, 1999a). However, some studies (Newton *et al.*, 1996; Ross *et al.*, 1996, Hungate *et al.*, 1997b) suggest higher carbon turnover rather than a substantial net increase in soil carbon under elevated $CO_2$. Predicted increased air and soil temperatures can be expected to increase the mineralization rate of SOM fractions that are not physically or chemically protected. The degree of protection of SOM varies with several soil-specific factors, including structure, texture, clay mineralogy, and base cation status. This may lead in the long term to negative effects on structural stability, water-holding capacity, and the availability of certain nutrients in the soil (see Reilly *et al*., 1996). Organic matter decomposition tends to be more responsive than NPP to temperature, especially at low temperatures (Kirschbaum, 2000). Within this range, any warming would stimulate organic matter decomposition (carbon loss) more than NPP (carbon gain); the net response would be a loss of soil carbon. Mineralization rates also are influenced by soil water content. For example, lower soil moisture in Mediterranean regions (see Chapter 3) would compensate temperature increase effects on carbon and nitrogen mineralization (Leiros *et al.*,1999).

As a result of these interactions with soil processes, experiments that impose sudden changes in temperature or $CO_2$ and last only a few years are unlikely to predict the magnitude of long-term responses in crop productivity, soil nutrients (Thornley and Cannell, 1997), and carbon sequestration (Luo and Reynolds, 1999). This may imply—in agreement with Walker *et al.* (1999)—that the actual impact of elevated $CO_2$ on crop yields in farmers' fields could be less than in earlier estimates that did not take into account limitations of nutrient availability and plant-soil interactions.

#### *5.3.3.2.2. Interactions between effects of climate change and soil degradation*

Land management will continue to be the principal determinant of SOM content and susceptibility to erosion during the next few decades, but changes in vegetation cover resulting from short-term changes in weather and near-term changes in climate are likely to affect SOM dynamics and erosion, especially in semi-arid regions (Valentin, 1996; Gregory *et al*., 1999).

The severity, frequency, and extent of erosion are likely to be altered by changes (see Table 3-10) in rainfall amount and intensity and changes in wind (Gregory *et al.*, 1999). Models demonstrate that rill erosion is directly related to the amount of precipitation but that wind erosion increases sharply above a threshold windspeed. In the U.S. corn belt, a 20% increase in mean windspeed greatly increases the frequency with which the threshold is exceeded and thus the frequency of erosion events (Gregory *et al.*, 1999). Thus, the frequency and intensity of storms would have substantial effects on the amount of erosion expected from water and wind (Gregory *et al.*, 1999). Different conclusions might be reached for different regions. Thus, before predictions can be made, it is important to evaluate models for erosion and SOM dynamics (Smith *et al.*, 1997). By reducing the water-holding capacity and organic matter contents of soils, erosion tends to increase the magnitude of nutrient and water stress. Hence, in drought-prone and low-nutrient environments such as marginal croplands, soil erosion is likely (high confidence) to aggravate the detrimental effects of a rise in air temperature on crop yields.

#### *5.3.3.2.3. Interactions with weeds, pests, and diseases*

Modest progress has been made in understanding of pest (weeds, insects, pathogens) response to climate change since the SAR. Oerke *et al*. (1995) estimate preharvest losses to pests in major food and cash crops to be 42% of global potential production. Rosenzweig *et al*. (2000) suggest that ranges of several important crop pests in the United States have expanded since the 1970s, including soybean cyst nematode and corn gray leaf blight; these expansions are consistent with enabling climate trends, although there are competing explanations. Promising work linking generic pest damage mechanisms with crop models is reported by Teng *et al*. (1996). For example, Luo *et al.* (1995) linked the BLASTSIM and CERES-RICE models to simulate the effects of climate change on rice leaf blast epidemics. They found that elevated temperature increases maximum blast severity and epidemics in cool subtropical zones; it inhibits blast development in warm humid subtropics. Such model linkages have been used to examine climate change impacts on weed-crop competition (e.g., for rice-weed interactions see Graf *et al*., 1990) and insect pests (Venette and Hutchison, 1999; Sutherst *et al*., 2000). Any direct yield gain caused by increased $CO_2$ could be partly offset by losses caused by phytophagous insects, pathogens, and weeds. Fifteen studies of crop plants showed consistent decreases in tissue nitrogen in high $CO_2$ treatments; the decreases were as much as 30%. This reduction in tissue quality resulted in increased feeding damage by pest species by as much as 80% (Lincoln *et al*., 1984, 1986; Osbrink *et al*., 1987; Coviella and Trumble, 1999). Conversly, seeds and their herbivores appear unaffected (Akey *et al*., 1988). In general, leaf chewers (e.g., lepidoptera) tend to perform poorly (Osbrink *et al.*, 1987; Akey and Kimball, 1989; Tripp *et al.*, 1992; Boutaleb Joutei *et al*., 2000), whereas suckers (e.g., aphids) tend to show large population increases (Heagle *et al*., 1994; Awmack *et al*., 1997a; Bezemer and Jones, 1998)—indicating that pest outbreaks may be less severe for some species but worse for others under high $CO_2$. It is important to consider these biotic constraints in studies on crop yield under climate change. Nearly all previous climate change studies excluded pests (Coakley *et al*., 1999).

### *5.3.3.3. Impacts on Livestock*

Recent research supports the major conclusions of Reilly *et al.* (1996) on animal husbandry. Farm animals are affected by climate directly and indirectly. Direct effects involve heat exchanges between the animal and its environment that are linked to air temperature, humidity, windspeed, and thermal radiation. These linkages influence animal performance (e.g., growth, milk and wool production, reproduction), health, and well-being. Indirect effects include climatic influences on quantity and quality of feedstuffs such as pastures, forages, and grain and the severity and distribution of livestock diseases and parasites. When the magnitudes (intensity and duration) of adverse environmental conditions exceed threshold limits with little or no opportunity for relief (recovery), animal functions can become impaired by the resulting stress, at least in the short term (Hahn and Becker, 1984; Hahn and Morrow-Tesch, 1993; Hahn, 1999). Genetic variation, life stage, and nutritional status also influence the level of vulnerability to potential environmental stresses. These relationships form the basis for developing biological response functions that can be used to estimate performance penalties associated with direct climate factors (Hahn, 1976, 1981, 1995). Earlier work (Hahn *et al.*, 1992; Klinedinst *et al.*, 1993) used such response functions with the Goddard Institute for Space Studies (GISS), Geophysical Fluid Dynamics Laboratory (GFDL), and United Kingdom Meteorological Office (UKMO) scenarios and found substantial reductions in dairy cow performance with climate change. For example, milk production of moderate- to high-producing shaded dairy cows in hot/hot-humid southern regions of the United States might decline an additional 5–14% beyond expected summer reductions. Conception rates of dairy cows were reduced by as much as 36% during the summer season in the southeastern United States. Short-term extreme events (e.g., summer heat waves, winter storms) can result in the death of vulnerable animals (Balling, 1982; Hahn and Mader, 1997; Hahn, 1999), which can have substantial financial impacts on livestock producers.

#### 5.3.3.4. *Response of Plant Crops and Possible Adaptation Options*

Very little work has investigated prospects for natural adaptation of crop species to climate change, and the results of the few studies that do have been inconclusive. However, there appears to be a wide range of resistance to high-temperature stress within and among crop species. For example, moderately large genetic variation in the tolerance to high-temperature induced spikelet sterility has been reported among and between indica- and japonica-type rice genotypes (Matsui *et al.*, 1997). Some rice cultivars have the ability to flower early in the morning, thereby potentially avoiding the damaging effects of higher temperatures later in the day (Imaki *et al.*, 1987).

Prospects for managed genetic modification appear to be more optimistic than for natural adaptation. Intraspecific variation in seed yield of soybean in response to elevated $CO_2$ was observed by Ziska *et al.* (1998). Differences in carbon partitioning among soybean cultivars may influence reproductive capacity and fecundity as atmospheric $CO_2$ increases, with subsequent consequences for future agricultural breeding strategies (Ziska *et al.*, 1998). However, no significant intraspecific variability in responses to elevated $CO_2$ was detected in studies with wheat and temperate forage species (Lüscher and Nösberger, 1997; Batts *et al.*, 1998). To promote adaptation to an environment of high $CO_2$ and high temperature, plant breeders have suggested selection of cultivars that exhibit heat tolerance during reproductive development, high harvest index, small leaves, and low leaf area per unit ground (to reduce heat load) (Hall and Allen, 1993). However, prospects to improve adaptation of crop species to elevated $CO_2$ remain very uncertain, and more research in this direction is required.

### 5.3.4. ***Impacts and Adaptation at Farm to Subnational Regional Scales***

#### 5.3.4.1. *Modeling Crop Yield Impact*

The number of studies that model the yield impacts of climate change (with and without $CO_2$ direct effects and with and without adaptation) across individual sites in regions has continued to grow since the SAR. Of particular note is the expansion of studies that explicitly model the effects of change in climate variability and means simultaneously versus change in climate means only (Southworth *et al*., 1999), use transient climate change scenarios, and report modeling of agronomic and socioeconomic adaptation. A selection of major global and regional model-based studies reported since the SAR is summarized in Table 5-4.

Table 5-4 yields are reported as ranges of percentage change over the climate change scenarios, modeling sites, and crop as noted. Thirteen of the yield ranges—without adaptation—are from studies of tropical crops. Of the 13 ranges, 10 encompass changes that are exclusively lower than current yields. In three ranges, a portion of the range is approximately no different from current yields or slightly above. In the tropics, most crops are at or near theoretical temperature optimums, and any additional warming is deleterious to yields. Thirty ranges of percentage changes in temperate crop yields also appear in Table 5-4. Of these 30 ranges, six encompass changes that are exclusively higher than current yields. In another seven, half or more of the changes were more than current yields. In yet another seven, less than half of the changes extended above current yields. The remaining 10 ranges encompassed changes that were exclusively less than current yields. Hence, in two-thirds of the cases, temperate crop yields benefited at least some of the time from climate change.

New work on climate change scenarios (Mitchell *et al*., 2000) generated with stabilized radiative forcing at 550 and 750 ppm equivalent-$CO_2$ and unstabilized radiative forcing (i.e., unmitigated emissions) in the HadCM2 model simulated major cereal yield response globally in 2080 (Arnell *et al*., 2001). The pattern of yield changes with unstabilized forcing duplicates the pattern described above: Generally positive changes at mid- and high latitudes overshadowed by reductions in yields at low latitudes. Stablization at 550 ppm ameliorates yield reductions everywhere, although substantial reductions persist in many low-latitude countries. Stabilization at 750 ppm produces a pattern of yield response that is intermediate relative to the 550 ppm and unstabilized forcing scenarios, with anomalous yield increases in mid-latitudes relative to 550 ppm as a result of interactions between atmospheric $CO_2$, temperature, and moisture. More studies are needed before confidence levels can be assigned to understanding of the agricultural consequences of stabilization, although this work is an important step.

In all agricultural regions, the effects of natural climate variability are likely to interact with human-induced climate change to determine the magnitude of impacts on agricultural production. Some analyses postulate an increase in weather variability (Mearns *et al.*, 1992, 1995; Rosenzweig *et al.*, 2000); simulations of wheat growth indicate that greater interannual variation of temperature reduces average grain yield (Semenov and Porter, 1995). Hulme *et al*. (1999) simulated natural climate variability in a multi-century control climate for comparison with changed variability in a set of transient climate change simulations. Wheat yields were simulated with control and climate change scenarios. For some regions, the impacts of climate change on wheat yields were undetectable relative to the yield impacts of the natural variability of the control climate (see Table 5-4). Greater efforts to take account of the "noise" of natural climate variability are indicated (Semenov *et al.*, 1996).

The importance of diurnal climate variability has emerged since the SAR (Reilly *et al*., 1996). Cold temperatures presently limit the yield of rice in all temperate rice-growing regions. Jacobs and Pearson (1999) provide new field results on irreversible

***Table 5-4****: Recent agricultural studies: a) studies with explicit global economics and/or global yields; b) studies of yield and production in developed regions, nations, and subnational regions; and c) studies of yield and production in economies-in-transition and developing regions, nations, and subnational regions.*

| Study | Scope | Crops | Climate Scenario[a] | Yield Impact w/o Adaptation[b] | Yield Impact w/ Adaptation[b] | Socioeconomic Impact | Comments |
|---|---|---|---|---|---|---|---|
| *a) Studies with Explicit Global Economics and/or Global Yields* | | | | | | | |
| Parry *et al*. (1999) | Global | Wheat, rice, maize, soybeans | Transient scenarios: 4 HadCM2 ensemble scenarios, 1 HadCM3 (both assume IS92a forcing) | | All cereals by 2080s[c]:<br>NA (-10 to +3%);<br>LA (-10 to +10%);<br>WE (0 to +3%);<br>EE (-10 to +3%);<br>AS (-10 to +5%);<br>AF (-10 to +3%) | By the 2080s: global cereal production (-4 to -2%), cereal prices (+13 to +45%), number of people at risk of hunger (+36 to +50%) | Farm-level adaptations (changes in plant date, varieties, irrigation, fertilizer); economic adjustments (increased investment, reallocation of resources, more land in production); no feedback between economic adjustments and yields; $CO_2$ direct effects included |
| Darwin *et al*. (1995) | Global | 13 commodities | UKMO, GISS | | | Agriculture prices [wheat (-10 to -3%), other grains (-6 to -4%)]; global GDP (+0.3 to +0.4%) | Adaptation through market-induced land-use change; $CO_2$ effect not included |
| Darwin (1999) | Global | Same as Darwin *et al*. (1995) | OSU, GFDL, GISS, UKMO | | | Qualitative impacts: world (positive for temperature change <2°C, negative for temperature change >2°C), regional (positive for high latitudes, negative for tropics) | Same as Darwin *et al*. (1995) |
| Darwin and Kennedy (2000) | Global | Same as Darwin *et al*. (1995) | $CO_2$ effect on yields only, no climate change | Yield changes with full $CO_2$ effect: wheat (7%), rice (19%), soybeans (34%), other crops (25%) | | Previous studies' estimates of economic value of $CO_2$ fertilization effect overstated by 61–166% | Scenarios run for $CO_2$ effect on yields ranging from very low to full effect |
| Adams *et al*. (1998) | USA | Various | +2.5°C, +7% ppt.;<br>+5°C, +0% ppt. | | | Agricultural price changes (-19 to +15%), GDP (+0 to +0.8%) | Includes direct effects of $CO_2$ |

effects and retardation (but recoverable) impacts of cold temperatures on various physiological processes in rice. On the other hand, rice spikelet sterility above 35°C at flowering (usually during daytime) puts rice at risk from increased daily maximum temperatures (Horie *et al.*, 1996). In light of recent observed rises in temperatures that are larger for daily minima

***Table 5-4 (continued)***

| Study | Scope | Crops | Climate Scenario[a] | Yield Impact w/o Adaptation[b] | Yield Impact w/ Adaptation[b] | Socioeconomic Impact | Comments |
|---|---|---|---|---|---|---|---|
| *a) Studies with Explicit Global Economics and/or Global Yields (continued)* | | | | | | | |
| Yates and Strzepek (1998) | Egypt | Wheat, rice, maize, soybean, fruit | GFDL and UKMO $2xCO_2$ equilibrium scenarios, GISS-A transient scenario at $2xCO_2$ | Yield changes: wheat (-51 to -5%), rice (-27 to -5%), maize (-30 to -17%), soybean (-21 to -1%), fruit (-21 to -3%) | Yield changes: wheat (-25 to -3%), rice (-13 to -3%), maize (-15 to -8%), soybean (-10 to 0%), fruit (-10 to -2%) | Change in selected economic indicators[d]: consumer-producer surplus (-3 to +6%), calories per day (-1 to +5%), trade balance (-15 to +36%) | Includes direct effects of $CO_2$; adaptations (shift in plant date, increased fertilizer, new varieties) |
| Rosenzweig and Iglesias (1998) | global (same sites as Parry *et al*., 1999) | Wheat, rice, soybean, maize | Sensitivity analysis (+2, +4°C) | +2°C[e]<br>[+8% (maize) to +16% (soybean)];<br>+4°C[e]<br>[-8% (rice) to -2% (wheat)] | Adaptation more successful at high and mid-latitudes than at low latitudes | | Includes direct effects of $CO_2$; transient yield response highly nonlinear |
| | | | GISS-A transient scenario, GISS $2xCO_2$ scenario | Wheat[f] [2050 (-18 to +25%), $2xCO_2$ (-32 to +27%)]; maize[f] [2050 (-26 to +13%), $2xCO_2$ (-35 to +23%)]; soybean[f] [2050 (+23 to +24%), $2xCO_2$ (+13 to +17%)] | | | |

than daily maxima (Easterling *et al.*, 1997), Dhakhwa *et al.* (1997) and Dhakhwa and Campbell (1998) conclude that, compared to equal day-night warming, differential warming leads to less water loss through evapotranspiration and better WUE. This is likely to lead to enhanced photosynthesis, crop growth, and yield—although at a possible loss of nutritional quality (Murray, 1997). Possible reduction of frost incidence is not normally considered in these studies. On the negative side, higher nighttime temperatures could extend the overwintering range for some insect pests and broaden the range of other temperature-sensitive pathogens.

Substantial progress has been made in development of transient (time-evolving) scenarios of climate change for use in agricultural impact assessment. An important question arises regarding whether 2 years with exactly the same climate, one produced by a transient scenario and the other by an equilibrium scenario, would give different production system responses. Many crop models contain cumulative functions that retain environmental information over several years (e.g., water balance, soil nutrients). This factor alone could account for substantial yield response differences between transient and equilibrium climate change scenarios. Only a few studies deliberately have compared simulated yields with transient and equilibrium climate change scenarios. Using the UKHIV equilibrium scenario with increased interannual variability at Rothamsted, Semenov *et al.* (1996) simulate a loss of wheat yield relative to current with two crop models and no change with a third. With the UKTR transient scenario, all three models show yield increases relative to current climate. The U.S. Country Studies Program (Smith *et al.*, 1996a) used the Clouds and Earth's Radiant Energy System (CERES) model to simulate larger average increases in winter wheat across Kazakhstan with the GFDL transient climate change scenario (for the 10th decade) (+21% winter wheat yield) than the GFDL equilibrium scenario (+17% winter wheat yield). Spring wheat yields decreased with both scenarios; again, however, yields simulated with the transient climate change were not as adversely affected as those simulated with the equilibrium climate change. Rosenzweig and Iglesias (1998) also found that wheat, maize, and soybean yields are less adversely affected by transient climate change than equilibrium climate change. Lack of consistency

***Table 5-4 (continued)***

| Study | Scope | Crops | Climate Scenario[a] | Yield Impact w/o Adaptation[b] | Yield Impact w/ Adaptation[b] | Socioeconomic Impact | Comments |
|---|---|---|---|---|---|---|---|
| *a) Studies with Explicit Global Economics and/or Global Yields (continued)* | | | | | | | |
| Winters *et al*. (1999) | Africa, Asia, Latin America | Maize, rice, wheat, coarse grains, soybean, "cash crops" | GISS, GFDL, UKMO | | *Africa*<br>[maize (-29 to -23%), rice (0%), wheat (-20 to -15%), coarse grains (-30 to -25%), soybean (-2 to +10%), cash crops (-10 to -4%)];<br>*Asia*<br>[maize (-34 to -20%), rice (-12 to -3%), wheat (-54 to -8%), coarse grains (-34 to -22%), soybean (-9 to +10%), cash crops (-13 to +2%)];<br>*Latin America*<br>[maize (-26 to -18%), rice (-26 to -9%), wheat (-34 to -24%), coarse grains (-27 to -19%), soybean (-8 to +12%), cash crops (-20 to -5%)] | *Africa*<br>[total agricultural production (-13 to -9%), GDP per capita (-10 to -7%), agricultural prices (-9 to +56%)];<br>*Asia*<br>[total agricultural production (-6 to 0%), GDP per capita (-3 to 0%), agricultural prices (-17 to +48%)];<br>*Latin America*<br>[total agricultural production (-15 to -6%), GDP per capita (-6 to -2%), agricultural prices (-8 to +46%)] | Yield impacts based on Rosenzweig and Parry (1994) values for "level 1" (farm-level) adaptations and $CO_2$ direct effects; yield impacts are weighted (by production) average of country-level yield changes; values for total agricultural production and per capita GDP include both yield and price impacts; range for agricultural prices is across food and cash crops, and GCMs |

in application of transient climate change scenarios to impact modeling between studies results in competing explanations about differences in impact estimates between the two types of climate change scenarios.

### 5.3.4.2. *Historical Analogs of Adaptation*

The agriculture sector historically has shown enormous capacity to adjust to social and environmental stimuli that are analogous to climate stimuli. Historical analogs of the adaptability of agriculture to climate change include experience with historical climate fluctuations, deliberate translocation of crops across different agroclimatic zones, rapid substitution of new crops for old ones, and resource substitutions induced by scarcity (Easterling, 1996). In the Argentine Pampas, the proportion of land allocated to crops has increased markedly at the expense of grazing land during historic humid periods, and vice versa during dry periods (Viglizzo *et al*., 1997). Historical expansion of hard red winter wheat across thermal and moisture gradients of the U.S. Great Plains provides an example of crop translocation (Rosenberg, 1982). At present, the northern boundary of winter wheat in China is just south of the Great Wall and the north edge of China, where large temperature increases are expected under climate change (Lin, 1997). Winter wheat planting has shifted from Dalian (38º54'N) to Shenyang (41º46'N) in Liaoning province. The shift was aided by introduction of freeze-resistant winter wheat varieties from high-latitude countries such as Russia, the United States, and Canada into Liaoning province (Hou, 1994, 1995; Chen and Libai, 1997). Rapid introduction of canola in Canadian agriculture in the 1950s and 1960s shows how rapidly farmers can modify their production systems to accommodate a new crop (National Research Council, 1991). Adaptation to declining groundwater tables by substituting dryland for irrigated crops in regions of the U.S. Great Plains is an example of substitutions to deal with water

***Table 5-4 (continued)***

| Study | Scope | Crops | Climate Scenario[a] | Yield Impact w/o Adaptation[b] | Yield Impact w/ Adaptation[b] | Socioeconomic Impact | Comments |
|---|---|---|---|---|---|---|---|
| *b) Studies of Yield and Production in Developed Regions, Nations, and Subnational Regions* | | | | | | | |
| Hulme *et al*. (1999) | Europe | Wheat | HadCM2—moderate (1% $yr^{-1}$) and low forcing (0.5% $yr^{-1}$) simulations for 2050 | +9 to +39%[g] (note that climate change impacts are indistinguishable from climate variability for 4 of 10 countries) | | | Includes direct effects of $CO_2$ |
| Antle *et al.* (1999b), Paustian *et al*. (1999) | Montana, USA | Winter wheat, spring wheat, barley | CCC | Climate change only (-50 to -70 %);<br>$CO_2$ fertilization only (+17 to +55%);<br>climate change + $CO_2$ (-30 to +30%) | | *With adaptation* [mean returns (-11 to +6%), variability of returns (+7 to +25%)]; *without adaptation* [mean returns (-8 to -31%), variability of returns (+25 to +83%)] | Scenarios include climate change plus $CO_2$ fertilization; yield impacts from Century model; adaptation modeled as change in crop rotation and management |
| Barrow and Semenov (1995) | 1 site in UK; 1 site in Spain | Wheat | Sensitivity analysis (+2,+4°C); downscaled UKMO high-resolution transient run (UKTR) | UK site only [+2°C (-7%), +4°C (-10%)]; both sites [+3°C (-14 to -5%), UKTR (-5 to +1%)] | | | Direct effects of $CO_2$ not considered |
| Dhakhwa *et al*. (1997) | North Carolina, USA (1 site) | Maize | GFDL, UKMO with equal and unequal day/night warming | -28 to -2% | | | Includes direct effects of $CO_2$ |
| Tung and Haith (1998) | New York, Indiana, and Oklahoma (1 site each) | Corn | GFDL | -24 to -15%[h] | -19 to -9%[h] | | Direct effects of $CO_2$ not considered; water supply also modeled; adaptations (change in variety, plant date, irrigation amount); assumes management practices currently optimal |
| Howden *et al*. (1999a) | Australia | Wheat | CSIRO 1996 | 9 to 37% | 13 to 46% | Gross margins (28 to 95%) | Assumes prices unchanged |

***Table 5-4 (continued)***

| Study | Scope | Crops | Climate Scenario[a] | Yield Impact w/o Adaptation[b] | Yield Impact w/ Adaptation[b] | Socioeconomic Impact | Comments |
|---|---|---|---|---|---|---|---|
| *b) Studies of Yield and Production in Developed Regions, Nations, and Subnational Regions (continued)* | | | | | | | |
| Brown and Rosenberg (1999) | USA corn and wheat regions | Corn, wheat | Three GCM-based 2x$CO_2$ scenarios distributed over time[i] (GISS, UKTR, BMRC) | Corn[j] [+1°C (-6 to +7%), +3°C (-17 to +4%), +5°C (-34 to -3%)]; wheat[j] [+1°C (-8 to +47%), +3°C (-20 to +37%), +5°C (-70 to -11%)] | | Change in production: Corn[j] [+1°C (-10 to +10%), +3°C (-20 to +5%), +5°C (-35 to -5%)]; wheat[j] [+1°C (-10 to +55%), +3°C (-25 to +45%), +5°C (-75 to -8%)] | $CO_2$ level corresponds to temperature change (365–750 ppm); dryland cropping only; planting date and growing season length allowed to vary in response to climate |
| *c) Studies of Yield and Production in Economies-in-Transition and Developing Regions, Nations, and Sub-National Regions* | | | | | | | |
| Alexandrov (1999) | Bulgaria (2 sites) | Winter wheat, maize | GISS, GFDL R-30, CCC, OSU, UK89, HCGG, and HCGS equilibrium scenarios; | Maize (-35 to -1%), wheat (+8 to +20%) | Maize (-24 to -10%)[k] | Net return with adaptation: maize (-29 to -12%)[k] | Includes $CO_2$ direct effects; adaptation (change in planting date) |
| | | | GFDL-T transient scenario at 2060s | Maize (-22%), wheat (+14%) | Maize (-21%)[l] | Maize (-26%)[l] | |
| Cuculeanu *et al.* (1999) | Romania (5 sites) | Winter wheat, maize | CCC, GISS | Wheat (+15 to +21%), dry maize (+43 to +84%), irrigated maize (-12 to +4%) | Irrigated maize (-18 to +8%)[m] | | Includes $CO_2$ direct effects; adaptation (new cultivars and changes in plant date, crop density, fertilizer amount) |
| Matthews *et al*. (1997) | Asia | Rice | Sensitivity analysis (+1, +2, +4°C); | +1°C (-7 to +26%)[n], +4°C (-31 to -7%)[n] | | | Includes $CO_2$ direct effects; adaptation (single to double cropping system, planting date shift, change in variety) |
| | | | GFDL, GISS, UKMO | -8 to +5%[o] | +14 to +27%[o] (with change in variety)[o] | Change in production: China with change in cropping system (+37 to +44%), region with change in variety (+13 to +25%)[o] | |

***Table 5-4 (continued)***

| Study | Scope | Crops | Climate Scenario[a] | Yield Impact w/o Adaptation[b] | Yield Impact w/ Adaptation[b] | Socioeconomic Impact | Comments |
|---|---|---|---|---|---|---|---|
| *c) Studies of Yield and Production in Economies-in-Transition and Developing Regions, Nations, and Subnational Regions (cont.)* | | | | | | | |
| Smith *et al*. (1996a) | The Gambia | Maize,<br>millet-early,<br>millet-late,<br>groundnuts | CCC, GFDL, GISS | -26 to -15%<br>-44 to -29%<br>-21 to -14%<br>+40 to +52% | | | $CO_2$ direct effects considered in all cases but Mongolia; adaptation in Mongolia consists of earlier seeding |
| | Zimbabwe | Maize | CCC, GFDL | -14 to -12% | | | |
| | Kazakhstan | Spring wheat,<br>winter wheat | CCC, GFDL[p],<br>incremental scenarios | -70 to -25%<br>-35 to +17% | | | |
| | Mongolia | Spring wheat | GFDL, GISS | -74 to +32% | -67 to -5%[q] | | |
| | Czech Republic | Winter wheat | Incremental scenarios | -3 to +16% | | | |
| Singh and Mayaar (1998) | Trinidad | Sugar cane | 4 synthetic scenarios, 1 GCM-based (CCC equilibrium) | -42 to -18% | | | Direct effects of $CO_2$ not considered |
| Magrin *et al*. (1997) | Argentina (pampas region) | Wheat, maize, soybean | GISS scenario for 2050 | Wheat (-15 to +15%), maize (-30 to -5%), soybean (+10 to +70%) | | Change in production: wheat (+4%), maize (-16%), soybean (+21%) | Includes direct effects of $CO_2$ (550 ppm) |
| Amien *et al*. (1996) | Indonesia | Rice | GISS transient | 2050<br>(-14 to -9%)[r] | | | Includes direct effects of $CO_2$ |
| Saseendran *et al*. (2000) | India (5 sites) | Rice | Synthetic (+1.5°C, +2 mm $day^{-1}$ precipitation) | -15 to -3%[s] | | | Includes direct effects of $CO_2$ (460 ppm) |
| Lal *et al*. (1999) | India | Soybean | Sensitivity analysis (+2,+4°C; ±20, ±40% precipitation) | -22 to +18% | | | Includes direct effects of $CO_2$ |
| Buan *et al*. (1996) | Philippines (6 sites) | Rice, corn | CCC, GFDL, GISS, UKMO | Rice (-13 to +9%), corn (-14 to -8%) | | | Includes direct effects of $CO_2$ |

**Table 5-4 (continued)**

| Study | Scope | Crops | Climate Scenario[a] | Yield Impact w/o Adaptation[b] | Yield Impact w/ Adaptation[b] | Socioeconomic Impact | Comments |
|---|---|---|---|---|---|---|---|
| *c) Studies of Yield and Production in Economies-in-Transition and Developing Regions, Nations, and Subnational Regions (cont.)* | | | | | | | |
| Karim *et al*. (1996) | Bangladesh | Rice, wheat | CCC, GFDL | Rice (-17 to -10%), wheat (-61 to -20%) | | | $CO_2$ direct effects not considered |
| Jinghua and Erda (1996) | China | Maize | GFDL, UKMO, MPI | -19 to +5%[t] | | Change in production: -6 to -3% | $CO_2$ direct effects not considered |

[a] GCMs are $2xCO_2$, unless otherwise noted.
[b] Unless otherwise noted, range is across GCM scenarios and site values are averaged; for sensitivity analyses, yield range is across sites.
[c] Range across countries and GCM scenarios; NA= North America, LA= Latin America, WE = Western Europe, EE = Eastern Europe, AS = Asia, AF = Africa.
[d] Range across three GCM scenarios, two levels of adaptation, and two baseline scenarios for 2060 (optimistic and pessimistic).
[e] World yields (weighted by production); range across crops.
[f] Range across countries in study.
[g] Range across countries and GCM scenarios.
[h] Range across sites.
[i] Temporal distribution accomplished using SCENGEN scenario generator.
[j] Change in global mean temperature (used as surrogate for time); regional temperature change may be higher or lower.
[k] One site only; range across three $2xCO_2$ equilibrium GCM scenarios (two sites and seven GCMs considered in *without adaptation* values).
[l] One site (two sites considered in *without adaptation* values).
[m] Range across two climate scenarios and degree of adjustment (e.g., different planting date options).
[n] Range across two crop models and $CO_2$ levels (1, 1.5, and $2xCO_2$).
[o] Range across crop model and GCM scenarios.
[p] Transient scenario.
[q] Adaptation not applied when climate change increases yields in the no-adaptation case.
[r] Average of “normal” and El Niño years; range across two sites.
[s] Range across sites.
[t] Range across sites and GCM scenarios.

becoming a scarce production resource (Glantz and Ausubel, 1988). None of these examples, however, deals specifically with an evolving climate change and all are historic—which limits confidence in extending their conclusions to future climate change.

### 5.3.4.3. Agronomic Adaptation of Yields

Increasing numbers of studies have investigated the effectiveness of agronomic adaptation strategies (e.g., adjustments in planting dates, fertilization rates, irrigation applications, cultivar traits) in coping with climate-induced yield losses and gains since the SAR (see Table 5-4). Considerable costs could be involved in this process, however—for example, in learning about and gaining experience with different crops or if irrigation becomes necessary. In some cases, a lack of water resulting from climate change might mean that increased irrigation demands cannot be met (see Section 4.7.2).

Methodologically, there has been little progress since the SAR in modeling agronomic adaptations. On one hand, the adaptation strategies being modeled are limited to a small subset of a much larger universe of possibilities, which may underestimate adaptive capacity. On the other hand, the adaptations tend to be implemented unrealistically, as though farmers are perfectly clairvoyant about evolving climate changes, which may inflate their effectiveness (Schneider *et al.*, 2000). Some studies find agronomic adaptation to be most effective in mid-latitude developed regions and least effective in low-latitude developing regions (Rosenzweig and Iglesias, 1998; Parry *et al.*, 1999). This finding clearly is supported across the studies summarized in Table 5-4, although the number of studies that include adaptation is not large. A small number of studies in Table 5-4 compare yield changes with and without agronomic adaptation. Percentage changes in yields across a range of climate change scenarios for those studies are shown in Figure 5-2. Each pair of vertical bars represents the range of percentage changes by crop, with and without adaptation, for each study. Clearly, adaptation ameliorates yield loss (and enhances yield gains) in most instances. The median adapted yields (mid-point of the vertical bars) shift upward relative to the median unadapted yields in six of the eight studies. Two studies do not show such an upward shift (Mongolia, Romania) because of peculiarities in the modeling (see figure caption). Adaptation ameliorates the worst yield losses in seven of the eight studies.

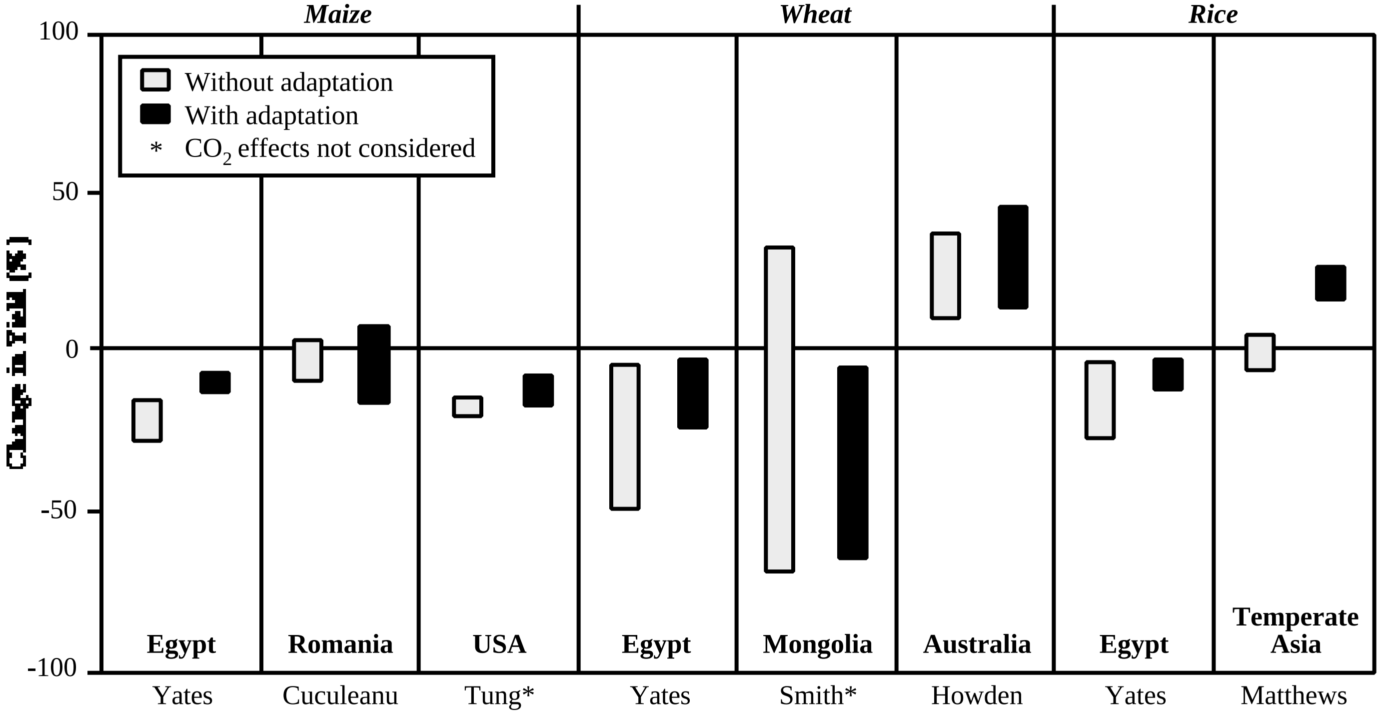

**Figure 5-2**: Ranges of percentage changes in crop yields (expressed in vertical extent of the vertical bars only) spanning selected climate change scenarios—with and without agronomic adaptation—from paired studies in Table 5-4. Each pair of ranges is differentiated by geographic location and crop. Pairs of vertical bars represent the range of percentage changes with and without adaptation. Endpoints of each range represent collective high and low percentage change values derived from all climate scenarios used in the study. Horizontal extent of the bars is not meaningful. Note that on the x-axis the last name of the lead author is listed as it appears on Table 5-4. See Table 5-4 for details on climate scenarios used and types of adaptation strategies modeled in each study. In the case of Mongolia, adaptation was not modeled when climate change resulted in positive yield change. In Romania, earlier planting of irrigated maize results in slightly lower yields than with current planting dates.

It is important to note, however, that differences in modeling methodology and aggregation of results often lead to conflicting conclusions in specific regions. For example, in two studies that used the same GCM scenarios, Matthews *et al.* (1997) simulate large increases, whereas Winters *et al.* (1999) simulate large decreases in rice yield with adaptation across several countries in Asia (see Table 5-4c). Hence, confidence in these simulations is low.

Important work has investigated the geographic distribution of crop potential under climate change. Carter and Saarikko (1996) used crop models to demonstrate a poleward shift in Finnish potential cereal cultivation by 100–150 km for each 1°C increase in mean annual temperature. Reyenga *et al.* (1999) indicate that climate change accompanied by a doubling of $CO_2$ is likely to enable expansion of existing Finnish wheat-growing areas into dry margins, considerably extending potential cropping areas. This expansion is moderated by dry conditions but not effectively enhanced by wetter conditions.

### *5.3.4.4. Adaptation of Livestock*

Significant costs can be incurred to counter the effects of climate change on animal production; moreover, the impact of a warmer climate in terms of costs is not linear: Larger changes in climate can increase costs exponentially (Hahn and Morgan, 1999). Possible benefits of climate change during cooler seasons are not well documented, but the benefits are likely to be less than the consequential negative hot weather impacts (Hahn *et al.*, 1992). The ability of livestock managers to cope with climate is demonstrated daily in their coping with normally varying conditions. A variety of management adaptations are available for livestock production systems. For example, Hahn and Mader (1997) outline a series of proactive management countermeasures that can be taken during heat waves (e.g., shades and/or sprinklers) to reduce excessive heat loads. Historical success in coping with climate variability suggests that livestock producers are likely to adjust to climate change successfully. Johnson (1965) provides examples from advances in genetics and breeding as related to the environment. These capabilities should allow adaptation to changing, less favorable circumstances associated with projected rates of climate change. However, coping can entail significant dislocation costs for certain producers. For individual producers, uncertainties associated with potential climate change imply additional risks related to how and when to adapt current production practices (Lewandrowski and Schimmelpfennig, 1999). Confidence in the foregoing projections of the ability of livestock producers to adapt their herds to the physiological stresses of climate change is difficult to judge.

The general lack of simulations of livestock adaptation to climate change is problematic. This absence of a well-developed livestock counterpart to crop modeling of adaptation assessments suggests a major methodological weakness. Hence, we give only low to moderate confidence in projections of successful livestock adaptability.

#### *5.3.4.5. Adaptation Effects on Farm and Regional Land Use and Income*

Reilly *et al*. (1996) review several studies that analyzed the effects of climate scenarios at the farm scale. One important implication of these studies is that changes in production activities and management adaptations could play an important role in mitigating the impacts of climate change.

Survey-based studies on farm-level decisionmaking in response to climate change have propagated since the SAR. Case studies in Canada suggest several farm-level adaptive strategies to which farmers may resort (Brklacich *et al.*, 1997; Smithers and Smit, 1997; Chiotti, 1998). In southern Ontario, Smit *et al.* (1997) found that farmers' choice of cultivar to plant in the current year is highly conditioned by the climate experienced the year before. There have been few quantitative, farm-level economic studies of response and adaptation to climate change since the SAR. In the U.S. midwest, Doering *et al.* (1997) used a crop-livestock linear programming model linked to the Century biogeochemistry model to investigate the impacts of climate change on 11 representative farms in the region under a doubled-$CO_2$ scenario. The analysis considered these farms under static technology and adapted technology; thus, the analysis could be regarded as representing farm-level economic responses to climate change as well as farm-level combined technological and economic adaptation. It showed that climate change may cause substantial shifts in the mix of crops grown in the upper midwest, with much less land planted to a corn-soybean rotation and more land devoted to wheat than now. Earlier planting of corn increased returns; hence, a more frost-resistant corn variety was found to be important to farm-level adaptation. Using a similar methodology, Parry *et al.* (1996) predict optimal agricultural land use in response to climate change in England and Wales.

Antle *et al.* (1999b) used an econometric-process simulation model of the dryland grain production system in Montana linked to the Century model (as reported in Paustian *et al.*, 1999) to assess the economic impacts of climate change in that region. Unlike other farm-level studies, this analysis is based on a statistically representative sample of commercial grain farms in the region, not on a small number of representative farms, so the results can be interpreted as representing the entire population of farms in the study area. Moreover, instead of using linear programming models, this analysis combines site-specific data and econometric production models in a stochastic simulation model, allowing representation of physical and economic heterogeneity in the region. Simulations were conducted for baseline and doubled $CO_2$ (Canadian Climate model) with observed production technology, with and without land-use adaptation and with and without $CO_2$ fertilization.

**Box 5-5. Extending Uncertainty in Crop Models to Uncertainty in Economic Analysis**

Several factors contribute uncertainty to modeling of impacts of climate change on agricultural systems (Parry *et al*., 1999). Crop modeling studies invariably highlight the need to develop confidence that the outputs are not "model-dependent." There is uncertainty because of the fact that yield estimates obtained in climate impact assessments vary from one impact model to another. In some cases, simulations across models may exhibit good agreement. Comparison of rice models showed that their predictions for potential production were quite close to observed values (Peng *et al.*, 1995). The Erosion Productivity Impact Calculator (EPIC), a generalized crop model, predicts observed yields most closely during years with extreme warmth, lending confidence to its ability to predict yields under climate change conditions (Easterling *et al.*, 1996). In other situations, agreement is not as good. In comparisons between wheat simulation models (Goudriaan *et al.*, 1994; Wolf *et al.*, 1996), grain yield predictions were markedly different between models. Such findings have stimulated work to compare the performance of different models and to analyze the underlying reasons for differences (Gregory *et al.*, 1999; Mearns *et al.*, 1999).

Spatial resolution of crop models is another important source of uncertainty in crop models. Crop models simulate processes that regulate growth and development at fine scales (a few kilometers), whereas climate change scenarios that drive crop models typically are produced by climate models operating at coarse scales (1,000 km or more) (Barrow and Semenov, 1995; Easterling *et al.*, 1998). Studies that use statistical downscaling techniques and nested limited area numerical models to increase the resolution of GCM scenarios of climate change have shown large simulated yield discrepancies between coarse-resolution (GCM) and fine-resolution (downscaled) climate change scenarios. Mearns *et al.* (1999) demonstrate for a site in Iowa that yields can change algebraic sign depending on the resolution of the climate change scenario: Maize yield decreases (-11%) with low-resolution climate change (CSIRO model) and increases (2%) with high-resolution climate change (RegCM limited area model nested within CSIRO model). Causes of variance in model output must be identified.

## Box 5-5. Extending Uncertainty in Crop Models to Uncertainty in Economic Analysis (continued)

Some analysts use statistical approaches to relate historical climate to yields as a basis for projecting yield response to future climate change. Spatial analog models use reduced-form econometric models that are based on historical data and estimate the relationship between economic variables such as asset values or value of production and climate variables. These models then can be used to simulate the effects of climate change on economic outcomes (Mendelsohn *et al.,* 1994). A strength of the spatial analog approach is that it embeds the complex responses of agricultural decisionmakers to spatial variations in climate. A disadvantage of some spatial analog models is that they do not represent market equilibrium. They can be used, however, to simulate price scenarios or be linked to an equilibrium model (e.g., see Adams *et al.*, 1999). Another key limitation is that these models do not represent productivity explicitly, so they cannot be linked to process models to incorporate the important effects of $CO_2$ fertilization on productivity and economic behavior. Kaufmann and Snell (1997) relate underlying determinants of yields to historical climate, to exploit the advantages of statistical analysis in capturing adaptation while utilizing the robustness of process-based crop modeling.

Integrated assessments of agricultural impacts of climate change (Table 5-4) often link outputs from climate models with crop process models and economic models that together predict changes in land use, crop choice, production, prices, and impacts on economic welfare of producers and consumers. The studies listed in Table 5-4 provide some indication of sensitivity to alternative climate models by comparing results from several different models or model variants. These comparisons show that model results are highly sensitive to alternative climate model inputs, reflecting the wide range of yield impacts that have been estimated with crop models. For example, Adams *et al.* (1999) report that net U.S. welfare changes from climate change range from –$16 billion to +$117 billion, depending on the climate scenario used. This surely illustrates Schneider and Moss’(Chapter 2) cascading uncertainty. By changing the assumptions about demand for U.S. exports, Adams *et al.* (1999) also find that the impacts of climate change on U.S. agriculture are highly sensitive to assumptions about the demand for U.S. exports, with impacts ranging from –$1.3 billion to +$123 billion. Many other assumptions are described in these studies (and, given the large size and complexity of these models, many other assumptions are not discussed explicitly) that are embedded in these models and that have not been subjected to sensitivity analysis. The limited sensitivity analyses that have been performed suggest that uncertainties in economic models alone are large and further imply that the economic impacts of climate change on agriculture, such as those presented in the SAR (Reilly *et al.*, 1996) and in Table 5-4, are given low confidence.

In view of the significant potential impacts of $CO_2$ on crop yields and the many interactions with climate and soil factors (Section 5.3.3), another significant question is how this effect is incorporated into integrated assessments of agricultural impacts. Some of the studies cited in Table 5-4 do not incorporate $CO_2$ fertilization effects; others use estimates of yield changes that are based on experiments or models that do incorporate these effects. Another question is whether experimental or simulated estimates of the $CO_2$ fertilization effect on yields is a realistic estimate of on-farm yield changes. Darwin and Kennedy (2000) found that estimates of the economic benefits of $CO_2$ fertilization are sensitive to the magnitude of yield change.

Taken together, it is clear that the results of existing agricultural impact studies must be assigned a low degree of confidence. How uncertain these results are has not been quantified. The implication is that econometric and integrated assessment modelers need to undertake sensitivity analysis to determine key assumptions and parameters and focus quantitative uncertainty analysis on those dimensions of the models.

With climate change, $CO_2$ fertilization, and adaptation, mean returns change by –11 to + 6% relative to the base climate and variability in returns increases by +7 to +25%, whereas without adaptation mean returns change by –8 to –31% and variability increases by +25 to 83%. These findings provide support for the hypothesis that ability to adapt plays a critical role not only in mean impacts but also in the spatial variability of impacts. They provide empirical support to the hypothesis advanced in the SAR that climate change is likely to have its greatest adverse impacts on areas where resource endowments are poorest and the ability of farmers to respond and adapt is most limited.

#### *5.3.4.6. Environmental and Natural Resource Consequences of Responses and Adaptation to Climate Change*

None of the economic studies in Reilly *et al*. (1996) analyze the environmental consequences of adaptation to climate change, such as increased demands on land and water resources. Adams *et al*. (1998) observe that this shortcoming remains true of most recent studies. However, Darwin *et al*. (1995) do include an analysis of impacts of climate change on land and water resources in a global model with the world subdivided into eight regions. They argue that competition from crop production could aggravate direct climate-induced losses of forests in moist tropical regions. IMAGE 2.0 simulations with future scenarios of limited $CO_2$ emissions show that increased deforestation increases agricultural capacity because of a smaller $CO_2$ fertilizaton effect on crops than if emissions continue on the current trajectory (Leemans, 1997).

Lewandrowski and Schimmelpfinnig (1999) draw implications for impacts on land and water resources, wild species, and natural ecosystems from the literature. They suggest that increased demand for irrigation predicted by these studies is likely to increase the opportunity cost of water and possibly reduce water availability for wildlife and natural ecosystems. However, it is difficult to go beyond such generalities with these aggregate models because most environmental impacts of agriculture are site-specific. Strzepek *et al.* (1999) show that some scenarios of climate change may reduce irrigation system reliability in the lower Missouri River in the U.S. corn belt, which may induce instream environmental stress. In many developing countries, current irrigation efficiencies are very low by developed-country standards. Irrigation effeciency in the Philippines in 1990 was 18%, compared to the global average of 43% (Asian Development Bank, 1998). Currently, 3,480–5,000 liters of water are used to produce 1.0 kg rough rice (equivalent to 640 g milled rice) in the Philippines (Baradas, 1999) and some neighboring countries. At those irrigation efficiencies, increased irrigation demand caused by climate change would strain irrigation supplies. Hence, one adaptation strategy is to increase irrigation efficiency.

Agriculture is a source and a sink of GHGs; hence, climate-induced agricultural land-use change is likely to impact soil carbon stocks in agricultural soils (Paustian *et al.*, 1996; Lal *et al.*, 1998; IPCC, 2000). Antle *et al.* (1999b) and Paustian *et al.* (1999) link a field-scale econometric-process simulation model to the Century ecosystem model to assess the impacts of climate change (from the Canadian Climate Centre model) on soil carbon in the dryland grain production system of the U.S. northern Great Plains. In a related set of studies, Antle *et al.* (2000) and Paustian *et al.* (2000) link a regional economic agricultural land-use model to the Century ecosystem model to assess the impacts of climate change on soil carbon in central U.S. cropland. These studies demonstrate that adaptive changes in land use and management are likely to have greater impacts on soil carbon than the direct effects of climate. Thus, adverse effects of climate change on soil carbon tend to be offset by the adaptive changes in land use that would be made by farmers in response to climate change. The degree of this offset depends on the magnitude of $CO_2$ fertilization effects on crop yields.

#### 5.3.4.7. Note on Costs of Adaptation

Adaptation is unlikely to come without cost. In a literature survey, Tol *et al.* (1998) conclude that adaptation costs (as opposed to net costs of damages) are not reported in most impact studies, especially in agriculture. Yet transition costs (e.g., to retrain farmers in new practices) and equilibrium costs (e.g., to develop additional irrigation or apply more fertilizer) may be considerable. The absence of a benefit-cost calculus for agricultural adaptation is a key deficiency. Existing studies also fail to account for the process of long-term, endogenous adaptation of technology in ways that are consistent with the extensive economic literature on that subject (Antle, 1996b). This process also will involve significant costs. An extensive body of economic research has studied the benefits and costs of agricultural research and has shown that institutions that are responsible for agricultural research adapt agricultural technology across space and time in response to relative resource scarcity (Hayami and Ruttan, 1985). Quiggin and Horowitz (1999) argue that changes in fixed capital for on-farm and off-farm infrastructure may be the most significant cost associated with adaptation to climate change.

### 5.3.5. *Modeling Impacts and Adaptation in a Global Economy*

Relatively few studies cited in the SAR linked estimates of yield responses to climate change with regional or global economic models to estimate production and welfare impacts. New studies that make such links provide important information on climate change-induced impacts on agriculture and on global and regional well-being (summaries are in Darwin *et al.*, 1995; Adams *et al.*, 1998; Lewandrowski and Schimmelpfennig, 1999). Recent contributions incorporate more crops and livestock, utilize Geographic Information System (GIS)-based land-use data, and link structural and spatial analogs (Table 5-4a). As noted in Section 5.3.2, the price of agricultural commodities is a useful statistic to summarize the net impacts of climate change on the regional or global supply/demand balance and on food security. Table 5-4a shows that the global model used by Darwin *et al.* (1995) and the U.S. model developed by Adams *et al.* (1998) predict that, with the rate of average warming expected by IPCC scenarios over the next century, agricultural production and prices are likely to continue to follow the downward path observed in the 20th century (see Section 5.3.2). As a result, impacts on aggregate welfare are a small percentage of GDPand tend to be positive, especially when the effects of $CO_2$ fertilization are incorporated. The only study that predicts real price increases with only modest amounts of climate change is Parry *et al.* (1999).

An important limitation of studies summarized in Reilly *et al.* (1996) is their focus on high-income regions of the world. Antle (1996b), Reilly *et al.* (1996), and Smith *et al.* (1996a) have suggested that impacts of climate change may be larger and more adverse in poorer parts of the world, where farmers and consumers are less able to adapt. The first study to address this question quantitatively in an aggregate regional analysis is Winters *et al.* (1999; they studied the impacts of climate change on Africa, Asia, and Latin America by using a computable general equilibrium model. Their analysis used larger price increases than those predicted by more recent studies in Table 5-4a, so this analysis should show more adverse impacts than an analysis based on prices from the more recent global studies.

The results summarized in Table 5-4a focus on the most vulnerable groups in poor countries—poor farmers and urban poor consumers. The results show that impacts on the incomes of these vulnerable groups would tend to be negative and in the range of 0 to –10%; in contrast, the impacts on consumer and producer groups predicted for the United States by Adams *et al.* (1998) ranged from –0.1 to +1%. Darwin (1999) reports results disaggregated by region and concludes that developing regions are likely to have welfare effects that are less positive or more negative than more-developed regions.

### *5.3.6. Vulnerability of the Agricultural Sector*

A population, region, or sector is vulnerable to climate change when serious deficits or unused opportunities remain after taking account of adaptation. Assessment of agricultural vulnerability to climate change calls attention to populations, regions, and sectors that may lose the means to satisfy basic needs (food security, progress toward development, a healthy environment) or fail to seize opportunities to improve social welfare. Particularly when such cases result in part from market failure, issues of consequential equity arise that signal policy interventions. Although no single measure of agricultural vulnerability exists, several indices together provide a sketch of vulnerability, including crop yields, crop prices, production, income, number of people at risk of hunger, rates of erosion, and irrigation demand.

#### *5.3.6.1. How Much Warming can Global Agriculture Absorb Before Prices Rise?*

Is there an amount of climate change to which the global food production system can adapt with little harm but beyond which it is likely to impose serious hardship? An answer can be sketched only with very low confidence at this time because of the combination of uncertainties noted above. As noted in Section 5.3.2, prices are the best indicator of the balance between global food supply and demand. They determine the access of a majority of the world’s population to an adequate diet. Two of three global studies reviewed here project that real agricultural output prices will decline with a mean global temperature increase of as much as 2.5ºC, especially if accompanied by modest increase in precipitation (Darwin *et al.,* 1995; Adams *et al.,* 1998). Another study (Parry *et al.,* 1999) projects that output prices will rise with or without climate change, and even a global mean temperature increase of ~1ºC (projected by 2020) causes prices to rise relative to the case with no climate change. When studies from the SAR are included with these more recent ones, there is general agreement that a mean global temperature rise of more than 2.5ºC could increase prices (Reilly *et al.,* 1996; Adams *et al.,* 1998; Parry *et al.,* 1999), with one exception (Darwin *et al.,* 1995). Thus, with very low confidence, it is concluded from these studies that a global temperature rise of greater than 2.5º C is likely to exceed the capacity of the global food production system to adapt without price increases. However, results are too mixed to support a defensible conclusion regarding the vulnerability of the global balance of agricultural supply and demand to smaller amounts of warming than 2.5ºC.

#### *5.3.6.2. Vulnerable Regions and Populations*

Although one may be reasonably optimistic about the prospects of adapting the global agricultural production system to the early stages of warming, the distribution of vulnerability among regions and people is likely to be uneven. As pointed out in Section 5.3.3, in the tropics—where some crops are near their maximum temperature tolerance and where dryland, nonirrigated agriculture predominates—yields are likely to decrease with even small amounts of climate change. The livelihoods of subsistence farmers and pastoral people—who make up a large proportion of rural populations in some regions, particularly in the tropics, and who are weakly coupled to markets—also could be negatively affected. In regions where there is a likelihood of decreased rainfall, agriculture could be substantially affected regardless of latitude. However, regional economic analysis (see Table 5-4) indicates that aggregate impacts on incomes even in the most vulnerable populations may not be large.

Clearly, in addition to the foregoing generality on productivity, other features of agricultural vulnerability are likely to vary widely among people, regions, nations, and continents (see Chapters 10–17). As noted in several places in this section and elsewhere (Downing *et al*., 1996a), the poor—especially those living in marginal environments—will be most vulnerable to climate-induced food insecurity. Parry *et al.* (1999) assessed the consequences of climate change for the number of people at risk of hunger as defined by the Food and Agriculture Organization (FAO, 1988) (see Table 5-4 for details of the study). By the 2080s, the additional number of people at risk of hunger as a result of climate change is estimated to be about 80 million. However, some regions (particularly in the arid and subhumid tropics) may be affected more. Africa is projected to experience marked reductions in yield, decreases in production, and increases in the risk of hunger as a result of climate change. The continent can expect to have 55–65 million extra people at risk of hunger by the 2080s under the HadCM2 climate scenarios. Under the HadCM3 climate scenario, the effect is even more severe, producing an estimated additional 70+ million people at risk of hunger in Africa. It should be noted, however, that these hunger estimates are based on the assumption that food prices will rise with climate change, which (as noted above) is highly uncertain as far as 80 years into the future.

Who are these extra people at risk of hunger likely to be? Downing *et al.* (1996b) suggest the following classes: rural smallholder producers, pastoralists, rural wage laborers, urban poor, and refugees and displaced people. In addition, they point to particular kinds of individuals: rural women, malnourished children, handicapped and infirm people, and the elderly.

## 5.4. Wildlife in Ecosystems

The overall mobility of wildlife and the fact that they are physiologically constrained by temperature and moisture make them effective indicators of climatic changes (Root, 1988a; Parmesan *et al.,* 2000). Evidence is presented below that shows that many different taxa from around the world already are exhibiting recognizable changes, such as poleward and elevational range shifts and changes in the timing of events such as breeding (see Table 5-3). Many factors (e.g., habitat conversion, pollution) pressure animals (see Figure 5-1). The information that follows indicates that the changes from such pressures could result in patterns that differ from those created by rapid climate change, which are created by the physiological constraints of organisms in response to climatic variables. For example, the pattern many species are showing of general poleward movement is not likely to be created by habitat conversion because such conversion generally does not occur less frequently along the poleward sides of many species ranges (e.g., along the northern boundaries in the northern hemisphere) than along those nearer the equator. Consequently, the balance of evidence suggests that, for animals that are exhibiting significant large-scale patterns of changes, the most consistent explanation is recent climatic change. Thus, like the proverbial "canaries in the coal mine," wildlife seem to be providing an important early indicator of how ecosystems might respond to the discernible human impact on climate that is contributing to its change (Santer *et al.,* 1996).

Much of the early work on the effects of climate change on ecosystems focused on vegetation. Animals (nondomestic animals or wildlife—these terms are used interchangeably) are important members of ecosystems and are affected by weather and climate (Andrewartha and Birch, 1954). Consequently, concerns about the impacts that rapid climatic change may have on wildlife and the risks these changes may impose on ecosystem services are assessed and summarized in this section.

### *5.4.1. State of Wildlife*

#### *5.4.1.1. Current Status of Endangered/Extinct Animals*

Recent estimates indicate that 25% (~1,125 species) of the world's mammals and 12% (~1,150 species) of birds are at a significant risk of global extinction (Stattersfield *et al.,* 1998; UNEP, 2000). One indicator of the magnitude of this problem is the speed at which species at risk are being identified. For example, the number of birds considered at risk has increased by almost 400 since 1994, and current population sizes and trends suggest an additional 600–900 soon could be added to these lists (IUCN, 1994; UNEP, 2000). The number of animals threatened with extinction varies by region (see Table 5-5). Global patterns of total diversity are reflected in the number of species at risk in each region, in that areas with more total species are likely to have more at risk. The number of threatened invertebrates in Table 5-5 is unrealistically low. The extinction rate of invertebrates in tropical forests alone has been estimated at 27,000 yr$^{-1}$, largely because of habitat conversion (Wilson, 1992).

#### *5.4.1.2. Species Status from Secure to Extinction: Ranking Risks*

Extinction often is caused by a combination of pressures acting over time (Wilson, 1992). Three traits of species populations that contribute to endangerment status are range size, distribution of suitable habitat within the range, and population size. Species that are most at risk often have small ranges, inhabit a unique type of habitat or one found in isolated areas (patchy in distributions), and/or typically occur at low population densities (Rabinowitz, 1981; Rabinowitz *et al.*, 1986). Using these criteria, signs that a species may be at risk include shrinking range, decreased availability of habitat within the range, and local or widespread population declines.

Species with restricted habitat requirements typically are most vulnerable to extinction (Pimm *et al*., 1995), including many

***Table 5-5****: State of some of the world's vertebrate wildlife. For each region, the table lists the number of critically endangered, endangered, and vulnerable species, separated by slashes (UNEP, 2000).*

| **Region** | **Totals** | **Amphibians** | **Reptiles** | **Birds** | **Mammals** |
|---|---|---|---|---|---|
| Africa | 102 / 109 / 350 | 0 / 4 / 13 | 2 / 12 / 34 | 37 / 30 / 140 | 63 / 63 / 163 |
| Asia and the Pacific | 148 / 300 / 739 | 6 / 18 / 23 | 13 / 24 / 67 | 60 / 95 / 366 | 69 / 163 / 283 |
| Europe and Central Asia | 23 / 43 / 117 | 2 / 2 / 8 | 8 / 11 / 10 | 6 / 7 / 40 | 7 / 23 / 59 |
| Western Asia | 7 / 11 / 35 | 0 / 0 / 0 | 2 / 4 / 2 | 2 / 0 / 20 | 3 / 7 / 13 |
| Latin America[a] and the Caribbean | 120 / 205 / 394 | 7 / 3/ 17 | 21 / 20/ 35 | 59 / 102 / 192 | 33 / 80 / 150 |
| North America[a] | 38 / 85 / 117 | 2 / 8 /17 | 3 / 12 / 20 | 19 / 26 / 39 | 14 / 39 / 41 |

[a] UNEPdata place Mexico in North America because of similarity of biomes.

endemic species that could be lost with loss of their habitat (Wyman, 1991; Bibby *et al.*, 1992; Stattersfield *et al.*, 1998). For example, the Sundarban, the only remaining habitat of Bengal tigers (*Panthera tigris tigris*) in Bangladesh, is projected to decrease considerably in size as a result of rising sea levels; Milliman *et al.* (1989) estimate a loss of 18% of the land by 2050 and as much as 34% by 2100. For tigers and the many other species that inhabit these forested wetland habitats, migration to higher ground probably would be blocked by human habitation of adjacent lands (Seidensticker, 1987; ADB, 1994). Many mountainous areas also have endemic species with narrow habitat requirements (Dexter *et al.*, 1995; Stattersfield *et al.*, 1998). With warming, habitats may be able to move up in elevation if the mountain is high enough. If not, the habitat could be lost (Still *et al.*, 1999). Some montane species that are susceptible to this change include forest birds in Tanzania (Seddon *et al.*, 1999), Resplendent Quetzal (*Pharomachrus mocinno*) in Central America (Hamilton, 1995), mountain gorilla (*Gorilla gorilla beringei*) in Africa, and spectacled bear (*Tremarctos ornatus*) in the Andes (Hamilton, 1995, and references therein). Protecting species that currently are vulnerable, endangered, or critically endangered (see Table 5-5) requires measures that, in general, reverse the trend toward rarity. Without management, there is high confidence that rapid climate change, in conjunction with other pressures, probably will cause many species that currently are classified as critically endangered to become extinct and several of those that are labeled endangered or vulnerable to become much rarer, and thereby closer to extinction, in the 21st century (Rabinowitz, 1981).

#### 5.4.1.3. *Wildlife Ties to Goods and Services*

Concern that species will become rare or extinct is warranted because of the goods and services provided by intact ecosystems and the species themselves. Most of the goods and services provided by wildlife (e.g., pollination, natural pest control; see Table 5-1) derive from their roles within systems. Other valuable services are provided by species that contribute to ecosystem stability or to ecosystem health and productivity. The recreational value (e.g., sport hunting, wildlife viewing) of species is large in market and nonmarket terms. Losses of species can lead to changes in the structure and function of the affected ecosystems, as well as loss of revenue and aesthetics (National Research Council, 1999).

### 5.4.2. ***Pressures on Wildlife***

Rapid climate change is only one of a long list of pressures on wildlife. Alone or in combination, these pressures may greatly increase species' vulnerabilities to rarity and extinction. Pressures such as land use and land-use change, introduction of exotic species, pollution/poisoning, and extreme climatic events and recent rapid climate change are of major concern and are discussed below. Other pressures include wildlife diseases, human persecution (e.g., overharvest, harassment), collisions with towers and other structures, collisions with cars and other forms of transportation, electrocutions, anthropogenic barriers to dispersal, war and other civil conflict, and wildlife trade (see Price *et al.*, 2000, for more information and references).

*Conversion of natural and semi-natural habitats* currently is quite extensive. Although climate change is starting to have observable effects on wildlife and the predicted future impact of climate change is expected to be large, the immediate principal current threat to the world's wildlife is habitat conversion (Vitousek *et al.*, 1997a). Roughly 80% of forests that originally covered the Earth have been cleared or degraded, and logging, mining, or other large-scale developments threaten 39% of what remains (UNEP, 2000). Roughly 65% of Asia's wildlife habitat has been converted to other uses, as well as nearly 75% of Australian rainforests (UNEP, 2000). Habitat conversion and degradation affect nearly 89% of all threatened birds and 83% of all threatened mammals (IUCN, 2000). Nearly 75% of all threatened bird species are found in forests (Stattersfield *et al.*, 1998), and tropical forests are the most species-rich terrestrial habitats; an estimated 90% of the world's species occur in moist tropical forests, which cover only 8% of the land area (UNEP, 2000).

*Introduction of exotic species*, intentionally and accidentally, has had deleterious effects on wildlife populations. Overall, 18% of threatened mammals in Australia and the Americas and 20% of the world's birds are challenged by introduced species (UNEP, 2000). Most extinctions on islands can be tied to introduction of species that prey on native species or destroy critical habitats (Stattersfield, *et al.*, 1998).

*Pollution and other biochemical poisonings* have direct and indirect effects on wildlife. Chemical contaminants have been detected in the tissues of species from around the globe, in regions such as Brazilian Pantanal wetlands (Alho Cleber and Vieira Luiz, 1997) and remote arctic habitats (UNEP, 2000). The most obvious effects of chemicals are direct poisoning events. These frequently are side effects of pesticide applications and can lead to losses of thousands of individuals (Biber and Salathe, 1991). In addition to affecting wildlife directly, contaminants can indirectly affect them by modifying their habitats.

*Stochastic and extreme climatic events* can cause deaths of large numbers of individuals and contribute significantly to determining species composition in ecosystems (Parmesan *et al.*, 2000). Hurricanes can lead to direct mortality, and their aftermath may cause declines because of loss of resources required for foraging and breeding (Wiley and Wunderle, 1994). Many of these extreme climatic events are cyclical in nature, such as sea surface temperature (SST) changes that are associated with the El Niño-Southern Oscillation (ENSO). Sea temperature increases associated with ENSO events have been implicated in reproductive failure in seabirds (Wingfield *et al.*, 1999), reduced survival and reduced size in iguanas (Wikelski and Thom, 2000), and major shifts in island food webs (Stapp *et al.*, 1999). Extreme drought in Africa was thought to have contributed to declines in the populations of many Palearctic migratory birds that wintered in the savanna and steppe zones of the Sahel (Biber and Salathe, 1991).

*Climate and climate change* are strong drivers of biotic systems. The distribution and survival of most species are moderated by climate (Root, 1988a,b,c; Martin, 1998; Duellman,1999). Although species have responded to climatic changes throughout evolutionary time (Harris, 1993), the primary concern today is the projected rapid rate of change. High species richness appears to be related to stable conditions; abrupt impoverishment of species has occurred during times of rapid change (Tambussi *et al.*, 1993).

*Synergistic effects* are likely to be quite damaging to animals. As habitat becomes more fragmented, barriers to dispersal or expansions of species' ranges could occur. This could force individuals to remain in inhospitable areas, decreasing the range and population size of species and ultimately leading to extinction (Rabinowitz, 1981). Fragmentation also may facilitate movement of invasive species into an area, leading to potential population declines through predation, competition, or transmission of disease (e.g., May and Norton, 1996). Increasing urbanization also could lead to increasing exposure to contaminants, which may make species less fit to survive changes in environmental conditions or weaken their immune systems (Pounds and Crump, 1987; Berger *et al.,* 1998). Human responses to climate change also may contribute to synergistic effects; for example, if new pest outbreaks are countered with increased pesticide use, nontarget species might have to endure climate- and contaminant-linked stressors.

## *5.4.3. Responses of Wildlife and Impacts on Goods and Services*

Findings indicate that many animals already may be responding to local climatic changes. Types of changes already observed include poleward and elevational movement of ranges, changes in animal abundance, changes in body size, and shifts in the timing of events such as breeding to earlier in the spring. These responses have been identified by a group of studies from around the world in a variety of different species (see Table 5-3). Far more information is available than can be summarized here. More detail on these changes is available in Hughes (2000) and Price *et al.* (2000).

### *5.4.3.1. Changes Exhibited by Animals*

Results from most studies that use large-scale data sets provide circumstantial (e.g., correlational) evidence about the association between changes in climate-related environmental factors and animal numbers or activities. Circumstantial evidence, though insufficient by itself, is highly suggestive when multiple studies examining a myriad of different species on all continents find similar results. Combined with smaller scale studies, experimental studies, and modeling studies that examine mechanistic connections between animals and climate change, the weight of evidence becomes even stronger. Such is the case for wildlife already exhibiting change related to climate forcings (see Table 5-3). The information given in the following subsections is a sampling of the types of studies that have examined the potential impacts of climate change on animals. The studies were selected for taxonomic and geographic inclusiveness and are not inclusive of the breadth of range of published studies. Information on more studies can be found in Table 5-3 and in Price *et al.* (2000).

#### *5.4.3.1.1. Shifts in animal ranges and abundances*

Ranges and abundances of prehistoric animals are known to have changed significantly over time (Goodfriend and Mitterer, 1988; Baroni and Orombelli, 1994; Coope, 1995). Currently, many species are undergoing range changes because of habitat conversion, land degradation (e.g., grazing, changes in fire regime), climate change, or a combination of factors. Possible climatically associated shifts in animal ranges and densities have been noted on three continents (Antarctica, Europe, and North America) and within each major taxonomic group of animals (see Table 5-3).

*Invertebrates:* Insect dispersal to favorable areas to make effective use of microclimatic differences is a common response to changing climate (e.g., Fielding *et al.*, 1999). The ranges of butterflies in Europe and North America have been found to shift poleward and upward in elevation as temperatures have increased (Pollard, 1979; Parmesan, 1996; Ellis *et al.*, 1997; Parmesan *et al.*, 1999). Warming and changed rainfall patterns also may alter host plant-insect relations, through community or physiological responses (e.g., host plant food quality) (Masters *et al.*, 1998).

*Amphibians and Reptiles:* Amphibians may be especially susceptible to climatic change because they have moist, permeable skin and eggs and often use more than one habitat type and food type in their lifetimes (Lips, 1998). Many amphibious species appear to be declining, although the exact causes (e.g., climate change, fungus, UV radiation, or other stresses) are difficult to determine (Laurance, 1996; Berger *et al.*, 1998; Houlahan *et al.*, 2000). Disappearance of the golden toad (*Bufo periglenes*) and the harlequin frog (*Atelopus varius*) from Costa Rica's Monteverde Cloud Forest Reserve appear to be linked to extremely dry weather associated with the 1986–1987 ENSO event (Pounds and Crump, 1994). Correlation between warming, reduced frequency of dry-season mist, and the timing of population crashes of four other frog species and two lizard species from the same cloud forest also has been found (Pounds *et al.*, 1999).

*Birds:* Bird ranges reportedly have moved poleward in Antarctica (Emslie *et al.,* 1998), North America (Price, 2000), Europe (Prop *et al.*, 1998), and Australia (Severnty, 1977). For example, the spring range of Barnacle Geese (*Branta leucopsis*) has moved north along the Norwegian coast, correlated with a significant increase in the number of April and May days with temperatures above 6°C (Prop *et al.,* 1998). The elevational range of some Costa Rican tropical cloud forest birds also apparently are shifting (Pounds *et al.,* 1999).

*Mammals:* Changes in mammal abundance can occur through changes in food resources caused by climate-linked changes or changes in exposure to disease vectors. For example, the Australian quokka (*Setonix brachyurus)* differs in susceptibility to Salmonella infections depending on climatic environmental conditions (Hart *et al.,* 1985).

### 5.4.3.1.2. Changes in timing (phenology)

*Invertebrates:* Warmer conditions during autumn and spring adversely affect the phenology of some cold-hardy species. Experimental work on spittlebugs (*Philaenus spumarius*) found that they hatched earlier in winter-warmed (3°C above ambient) grassland plots (Masters *et al.,* 1998).

*Amphibians:* Two frog species, at their northern range limit in the UK, spawned 2–3 weeks earlier in 1994 than in 1978 (Beebee, 1995). These changes were correlated with temperature, which also showed increasing trends over the same period.

*Birds:* Changes in phenology, or links between phenology and climate, have been noted for earlier breeding of some birds in Europe, North America, and Latin America (see Table 5-3). Changes in migration also have been noted, with earlier arrival dates of spring migrants in the United States (Ball, 1983; Bradley *et al.*, 1999), later autumn departure dates (Bezzel and Jetz, 1995), and changes in migratory patterns in Europe (Gatter, 1992).

### 5.4.3.1.3. Changes in morphology, physiology, and behavior

*Amphibians and Reptiles:* Correlations between temperature and calling rates have been found in Egyptian frogs (Akef Mamdouh and Schneider, 1995). Indian tree frogs show differences in behaviors that depend on their level of hydration (Lillywhite *et al.,* 1998). Painted turtles grew larger in warmer years, and during warm sets of years turtles reached sexual maturity faster (Frazer *et al.,* 1993). Physiological effects of temperature, primarily sex determination, also can occur while reptiles are still within their eggs (Gutzke and Crews, 1988).

*Birds:* Spring and summer temperatures have been linked to variations in the size of eggs of the Pied Flycatcher (*Ficedula hypoleuca*). Early summer mean temperatures explain ~34% of the annual variation in egg size between the years 1975 and 1994 (see Figures 5-3 to 5-5; Jarvinen, 1996).

*Mammals:* Body size is correlated with many life-history traits, including reproduction, diet, and size of home ranges. North American wood rat (*Neotoma spp.*) body weight has shown a significant decline that is inversely correlated with a significant increase in temperature over the past 8 years (Smith *et al.,* 1998). Juvenile red deer (*Cervus elaphus*) in Scotland grew faster in warm springs, leading to increases in adult body size (Albon and Clutton-Brock, 1988).

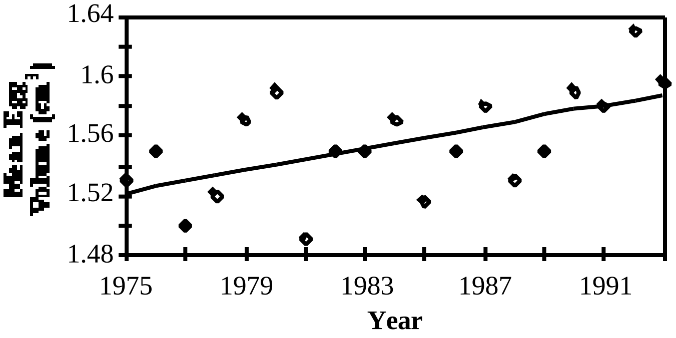

**Figure 5-3**: Change in egg volume of Pied Flycatcher birds over time. Regression line is y = 1.24 + 0.004 x (P<0.01).

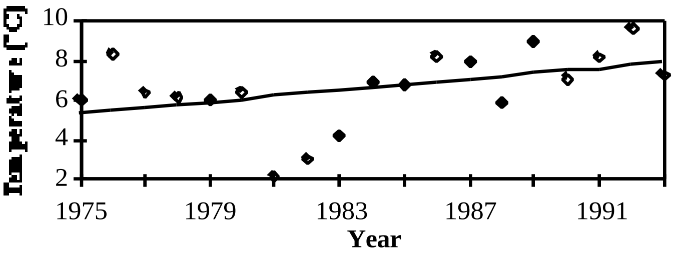

**Figure 5-4**: Change in the mean air temperature during the egg laying time period of the Pied Flycatcher. Regression line is y = -5.5 + 0.14 x (P<0.7).

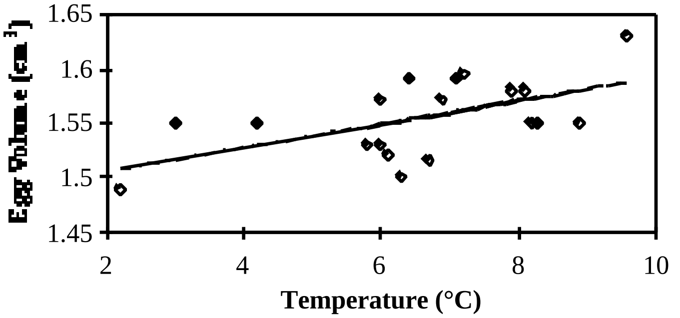

**Figure 5-5**: Correlation between egg volume of the Pied Flycatcher over the years and mean air temperature (regression is significant at P<0.01).

### 5.4.3.1.4. Changes in community composition, biotic interactions, and behavior

Differential responses by species could cause existing animal communities to undergo restructuring. This occurred in prehistoric plant communities because no analogous communities exist today (Davis, 1990). Similarly, Graham and Grimm (1990) found no present-day mammal communities that are analogous to some prehistoric animal communities.

*Invertebrates:* Potato aphids grown on plants kept in elevated $CO_2$ (700 ppm) showed a reduced response to alarm pheromones in comparison to those grown on plants in ambient $CO_2$ (350 ppm). Aphids were more likely to remain on leaves, possibly making them more susceptible to predators (Awmack *et al.*, 1997b).

*Amphibians:* Temperature and dissolved oxygen concentrations can alter the behavior of amphibian larvae, and changes in thermal environments can alter the outcome of predator-prey interactions (Manjarrez, 1996; Moore and Townsend, 1998).

*Birds:* In the UK, climate change may be causing a mismatch in the timing of breeding of Great Tits (*Parus major*) with other species in their communities (Visser *et al.*, 1998). The phenology of plants and some animals in the study area has advanced over the past 23 years, whereas breeding timing of Great Tits has not changed. This decoupling could lead to birds hatching when food supplies may be in low abundance (Visser *et al.*, 1998).

*Mammals:* Post *et al.* (1999) document a positive correlation between gray wolf (*Canis lupus*) winter pack size and snow depth on Isle Royale (USA). Greater pack size leads to three times more moose kills than in years with less snow. Fewer moose mean less browsing and thus better growth of understory balsam fir (*Abies balsamea*).

### *5.4.3.2. Model Projections of Wildlife Responses*

*Invertebrates:* Because changes in the distribution of disease vectors, such as mosquitoes, and crop and forest pests can have major human health and economic impacts, most modeling of changes in insect range and abundance have focused on pest species (Sutherst *et al.*, 1995). In southern Africa, models project changes in the ranges of tsetse flies, ticks, and mosquitoes (Rogers, 1996). Potential range changes of other disease vectors are discussed in Chapter 9.

High proportions of boreal forest insect pests overwinter as eggs. The strong link between patterns of minimum temperature and the location of outbreaks are best explained by the fact that eggs are killed when temperatures dip below a species' tolerance threshold (Sullivan, 1965; Austara, 1971; Virtanen *et al*., 1996). Modeling work that is based on these observations and projected climate data suggests that increasing nighttime winter temperatures may increase the frequency of these pest species in northern areas (Tenow, 1996; Virtanen *et al*., 1996, 1998), although warmer summer temperatures may reduce the intensity of outbreaks (Niemelä, 1980; Neuvonen *et al*., 1999; Virtanen and Neuvonen, 1999; but see Ayres, 1993).

*Amphibians and Reptiles:* In studies on altitudinal (Pettus and Angleton, 1967; Licht, 1975; Bizer, 1978; Berven, 1982a,b) and latitudinal (Collins, 1979) gradients, a general pattern of faster metamorphosis at smaller sizes occurs at high elevations and northern latitudes. Changes in these life history attributes may affect species' abundances as a result of susceptibility to predators or environmental extremes or changes in reproductive output (Calef, 1973; Travis, 1981). Species that inhabit high-altitude areas may be at particular risk from climate change because as temperatures increase, their habitats may disappear (Hamilton, 1995; Pounds *et al.*, 1999). In Australia, frog distributions are strongly correlated with patterns of annual rainfall, implying that frogs in these areas may be able to expand their ranges if precipitation increases (Tyler, 1994). Reptile ranges often correlate with temperature (Nix, 1986; Owen and Dixon, 1989; Yom-Tov and Werner, 1996), suggesting that ranges may shift with temperature change. Desert tortoises (*Testudo graeca graeca*) in southern Morocco already have shifted their ranges in response to drier conditions possibly resulting from land-use changes (Bayley and Highfield, 1996).

### Box 5-6. Penguins as Indicators of Climate Warming in Western Antarctic

Midwinter surface air temperatures in the Western Antarctic Peninsula (WAP) region have increased by 4–5°C over the past 50 years (Smith *et al.*, 1996b). Studies confirm that the spatial and temporal patterns of winter sea-ice development in the WAPhave changed during this time in response to rapid warming (Fraser *et al.*, 1992; de la Mare, 1997; Jacobs and Comiso, 1997; Loeb *et al.*, 1997). Chinstrap (*Pygoscelis antarctica*) and Adélie (*P. adeliae*) penguin populations also have changed during the past 25 years.

Although these two species are ecologically very similar, with diets and breeding ranges that overlap in the WAP (Volkman *et al.*, 1980), their winter habitat preferences are radically different. Adélies are obligate inhabitants of the pack ice, whereas Chinstraps are ice-intolerant, preferring to remain in close association with open water (Fraser *et al.*, 1992; Ainley *et al.*, 1994). The quality and availability of winter habitat is an essential determinant of survival and therefore a key factor regulating seabird populations (Birkhead and Furness, 1984). Adélie penguins have decreased by 22% whereas Chinstrap penguins have increased by more than 400% over the past 25 years (Fraser and Patterson, 1997; Smith *et al.*, 1999). This pattern supports the hypothesis that the increasing availability of open water as a result of warmer winters is favoring the survival of Chinstraps over the ice-dependent Adélies (see Fraser *et al.*, 1992).

*Birds:* In the prairie pothole region of the United States and Canada, a significant correlation between wetlands, duck numbers, and the Palmer Drought Severity Index has been found (Sorenson *et al.*, 1998). Projections of warming and drying for this region suggest that the number of wetlands and, correspondingly, the number of breeding ducks could be reduced.

Similar losses of wetlands have been projected for Africa (Magadza, 1996) and Australia (Hassal and Associates, 1998).

*Mammals:* Population reductions in mammals in African arid lands are possible if the incidence of drought increases (IPCC, 1998). In Australia, declines in several mammal species may occur if droughts increase in frequency or intensity (Caughley *et al.*, 1985; Roberston, 1986; Gordon *et al.*, 1988). Mountains (see Section 5.4.3.1), patchy habitats, and oceans can be barriers to range shifts. The Arctic Ocean is an obstacle to 25 species of Canadian mammals, with the collared lemming (*Dicrostonyx groenlandicus*) possibly losing at least 60% of its available habitat to climate change (Kerr and Packer, 1998).

The nutritional quality of some plant species has been found to decrease with increased $CO_2$ availability (Bazzaz, 1996). This could mean that herbivores might have to eat more. Most studies have dealt with insect or domestic mammals, but similar results are likely to hold for wild herbivorous mammals (Baker *et al.*, 1993; Bolortsetseg and Tuvaansuren, 1996). Relative changes in major plant lifeforms also can affect species distributions and populations densities; for example, in high-latitude rangelands, shrub abundance may increase and forb abundance may decrease (Chapin *et al.*, 1995), possibly leading to limitations on food supplies available to migrating caribou (White and Trudell, 1980).

### 5.4.3.3. *Impacts on Goods and Services with Market Values*

#### 5.4.3.3.1. *Control of pest species and disease vectors*

Wildlife plays a role in natural and agricultural ecosystems through reduction of injurious insect pests and disease vectors. In some forested ecosystems, birds, together with parasitoids and other insect predators, are effective predators of insect pests (Crawford and Jennings, 1989; see Price *et al.*, 2000). Insects, ants, and spiders also play key roles in reducing populations of pest insects. For example, when ants were removed from maize plots in Nicaragua, two pest species increased in abundance, and damage to the maize increased (Perfecto, 1991). The annual economic value of wildlife control of insect pests has been estimated to be hundreds of billions of U.S. dollars (de Groot, 1992; Pimentel *et al.*, 1992, 1993; Pimentel, 1997). This includes destruction of pests that are injurious to crops (estimated at US$90 billion year in the United States alone), as well as those affecting forests.

Climate change could impact many of these systems by decoupling predators from their prey (Root and Schneider, 1995). Studies in North America project reductions in the extent of distribution size of some of the species that feed on pests in forest, grassland, and agricultural ecosystems (see Price, 1995; Price *et al.*, 2000). This could lead to an increasing need to use pesticides, with accompanying health risks (human and wildlife) and economic costs (Kirk *et al.*, 1996; Colburn *et al.*, 1997; Herremans, 1998).

#### 5.4.3.3.2. *Pollinators*

Pollination is crucial to the reproduction of many plants, thus to maintenance of functioning ecosystems and biodiversity. Many factors influence the occurrence and density of insects (e.g., habitat conversion, excessive pesticide use). The principal mode of pollination of many plant species is by insects. Worldwide, an estimated 400 crop species are pollinated by bees and more than 30 other animal genera; possible crop loss in some species would be more than 90% in the absence of bees (Southwick, 1992; Buchmann and Nabhan, 1996). More than 100,000 different animals around the world pollinate 250,000 types of plants (Buchmann and Nabhan, 1996). The estimated annual value of wildlife pollination to commercial crops and pasture grasses is tens of billions of U.S. dollars (de Groot, 1992; Pimentel *et al.*, 1992, 1995). If pollination of noncommercial plants were added, the figure would be significantly higher.

#### 5.4.3.3.3. *Seed dispersal*

Seeds of some plants are dispersed by one or a few animal species. In Costa Rican tropical deciduous forests, as many as 60% of plants have their seeds dispersed by birds; the numbers are 75% in subtropical evergreen forests and 80% in montane evergreen forests (Bawa, 1995). On Samoa, the vast majority of seed dispersal in the dry season is mediated by flying foxes (Cox *et al.*, 1991). Plants commonly disperse via seeds passing through the digestive tracts of animals or with animals that cache seeds (Robinson and Handel, 1993; Lanner, 1996). Consequently, if the ranges of the appropriate animals become disjunct from even part of the ranges of specific plants, dispersal of the plants may suffer (see Price *et al.*, 2000, for other examples).

#### 5.4.3.3.4. *Decomposers and soil maintainers*

Invertebrates and microorganisms living on or below the soil surface provide needed goods and services to human societies (e.g., mixing and aeration of soils, decomposition of materials and human waste) (Daily *et al.*, 1997). These processes contribute to creation of fertile topsoil from organic matter and mineral components of the soil. Some of these organisms (e.g., ants) are susceptible to climatic changes, especially droughts (Folgarait, 1998). On a global scale, the estimated cost of replacing waste elimination services performed by these organisms is hundreds of billions of U.S. dollars annually; the estimated cost of replacing topsoil production services is tens of billions of U.S. dollars annually (de Groot, 1992; Crosson *et al.*, 1995; Pimentel *et al.*, 1995; Daily *et al.*, 1997).

#### 5.4.3.3.5. *Protection of endangered species*

Another way of assessing the value of wildlife is by examining how much is spent on its conservation. In fiscal year 1995, approximately US$330 million was spent on the conservation of threatened and endangered species in the United States and its territories (U.S. Fish and Wildlife Service, 1995). As pressures facing wildlife increase, the number of species that require conservation attention also will most likely increase.

#### *5.4.3.3.6. Subsistence hunting*

People in many parts of the world depend on wildlife for their daily nutritional needs. This is most pronounced in less-developed areas. For example, the Cree along James Bay in Canada harvest approximately 800,000 kg of animal food annually. The per capita replacement value of this harvest was estimated to be CDN$6,000 in 1986 dollars (Scott, 1987). The abundance of caribou (*Rangifer tarandus groenlandicus*) available for harvest by indigenous peoples could decrease as a result of increased temperatures, snowfall, and potential shifts in the timing of precipitation (Brotton and Wall, 1997; Ferguson, 1999). Adverse impacts also have been projected for other subsistence species, including marine birds, seals, polar bears (Stirling, 1997), tundra birds (Jefferies *et al.*, 1992), and other tundra-grazing ungulates (Jeffries, *et al.*, 1992; Gunn, 1995).

Wildlife species also are a significant source of food and medicine for people in many temperate and tropical countries, such as Botswana and Nigeria (McNeely *et al.*, 1990) and Australia (Bomford and Caughley, 1996). Among the Boran (Kenya), birds are used for medicines (ostrich oils) and their feathers for cleaning wounds (Isack, 1987). In many countries, climate change impacts such as reductions in wildlife populations may have the greatest impact on the lowest-income groups—those with the least ability to adapt if hunting opportunities decline (Arntzen and Ringrose, 1996).

#### *5.4.3.3.7. Recreational use of animals and ecotourism*

In many African countries, ecotourism to view wildlife is a major contributor to gross national product (GNP). Worldwide, ecotourism is estimated to provide US$500 billion to 1 trillion annually to the global economy (Munasinghe and McNeely, 1994). Changes in climate could reduce the populations of some of the species people are willing to pay to see (Mills *et al.*, 1995; Allen-Diaz, 1996).

### *5.4.3.4. Impacts on Goods and Services with Nonmarket Valuations*

Though difficult to measure, nonmarket values must be taken into account in discussing the state of wildlife. These values can be described in terms of the cultural, religious, scientific, and aesthetic importance of wildlife species (Bawa and Gadgil, 1997; Goulder and Kennedy, 1997; National Research Council, 1999). At times, monetary value can be assigned to nonmarket aspects, such as a tourist's willingness to pay to see wildlife in natural habitats (e.g., Edwards, 1991). Although this is difficult, monetary values might be able to be assigned to some of the wildlife services described in this section. The fact that a monetary value is difficult to assign should not diminish the importance of the service; for many, no substitute exists for the services provided by wildlife.

Tools used to assess nonmarket monetary values vary, but a few approaches have become common. One attempts to determine "existence value": people's willingness to pay to know that an animal simply continues to exist, even if they never see it (National Research Council, 1999). Similarly, an "option value" is based on a person's desire to be able to potentially interact with the animal some time in the future, and a "bequest value" is being assured that future generations will be able to use or view an animal (National Research Council, 1999). Listed below are some areas where it is particularly difficult to estimate a monetary value for animal-related services.

#### *5.4.3.4.1. Cultural and religious importance of animals*

Many indigenous peoples use wildlife as integral parts of their cultural and religious ceremonies. For example, birds are strongly integrated into Pueblo Indian (United States) communities. Birds are regarded as messengers to the gods and a connection to the spirit realm. Among Zuni Indians (United States), prayer sticks with feathers from 72 different species of birds are used as offerings to the spirit realm (Tyler, 1991). In Boran (Kenya) ceremonies, selection of tribal leaders involves rituals that require Ostrich (*Struthio camelus*) feathers. Birds also are used for tribal cosmology, meteorology, religion, and cultural ceremonies (Isack, 1987). Wildlife plays similar roles in cultures elsewhere in the world. Thus, shifts in the timing or the ranges of wildlife species could impact the cultural and religious lives of some indigenous peoples.

#### *5.4.3.4.2. Wildlife and scientific information*

Researchers have been studying spider silk to learn how this strong and elastic material could be manufactured (Xu and Lewis, 1990). Similarly, scientists have examined the structure of the small scales on butterfly wings to understand how they reflect light and dissipate heat. These studies may help engineers design better processes for manufacturing and designing computer chips (Miaoulis and Heilman, 1998). Researchers also have studied the regenerative properties of cells within rhinoceros horns and have identified potent antimicrobial chemicals from frogs and toads (Cruciani *et al.,* 1992; Moore *et al.*, 1992; Boskey, 1998).

#### *5.4.3.4.3. Ecosystem function and biodiversity maintenance*

Other valuable services are provided by species that contribute to ecosystem health and productivity. Reductions in or losses of species can lead to reduced local biodiversity and changes in the structure and function of affected ecosystems (National Research Council, 1999). The most well-known example of this kind of effect comes from marine systems, where the presence or absence of a starfish species has been found to greatly influence the species composition of intertidal habitats (Paine, 1974). Species in terrestrial systems also can have a strong influence on the biodiversity of their ecosystems; in many cases these effects are related to their functions as pollinators or seed dispersers (see Price *et al.*, 2000).

### *5.4.4. Adaptation by Humans for Changes in Goods and Services*

The ability of wildlife to adapt naturally to climate change is largely a function of available genetic diversity and the rate of change. This section deals not with natural adaptation but with human adaptation to changes in wildlife populations, in terms of conserving wildlife and replacing some of the goods and services that wildlife provides.

#### *5.4.4.1. Parks and Reserves*

One typical method to adapt to declines in wildlife populations has been establishment of refuges, parks, and reserves. Placement of reserves, however, rarely has taken into account potential rapid climate change, even though the problems of such change and reserve placement were discussed in the mid-1980s (Peters and Darling, 1985). Managers of current reserves and parks need to be encouraged to consider rapid climate change in developing future management plans (Solomon, 1994; Halpin, 1997). Developing a series of bioindicators to monitor the potential impacts of rapid climate change on parks and reserves may be possible (de Groot and Ketner, 1994).

In the United Kingdom, the Institute of Terrestrial Ecology has estimated that 10% of all designated areas (i.e., nature reserves) could be lost (e.g., to habitat degradation) within 30–40 years and that species distribution in 50% of designated areas could change significantly over the same period (UK DETR, 1999). In light of these changes, there is a need for a robust nature conservation system that can accommodate climate change.

In part, the disparity between siting reserves where wildlife species currently are versus where they may be in the future may stem from uncertainties in the rate and amount of projected climate change. If a species'range shifts out of a reserve created for its survival, the current reserve placement could even be considered maladaptive. However, if reserves are not created and species are lost to other pressures, the potential effects of climate change on species distributions are moot (see Box 5-7).

#### *5.4.4.2. Captive Breeding and Translocations*

Another way in which humans have dealt with endangered wildlife populations has been through captive breeding and translocations. These techniques have been put forward in the past as methods to deal with future population pressures caused by climate change (Peters, 1992). However, although captive breeding and translocation are effective tools for conservation of some species, they may be appropriate for only a handful of species (see Box 5-8).

**Box 5-7. Biodiversity Hotspots and Climate Change**

Biodiversity hotspots are areas with exceptional concentrations of endemic species facing extraordinary threats of habitat destruction. Twenty-five hotspots contain the sole remaining habitats for 133,149 (44%) vascular land plants and 9,732 (36%) terrestrial vertebrates (Myers *et al.*, 2000). The number of invertebrates is not known, but in light of the many interactions (e.g., pollination) between plants and invertebrates, especially insects, the concentrations of these animals probably parallel those of plants. Nine of these hot spots occur on islands, making them particularly vulnerable to sea-level rise and limiting or preventing the opportunity for many terrestrial animals to modify their range (Myers *et al.*, 2000). Because these 25 hotspots constitute only 1.4% of the Earth's land surface (Myers *et al.*, 2000), they provide an opportunity for planners to respond to the biodiversity crisis by giving priority status to conserving them.

Unfortunately, synergistic effects of climate change and requirements to conserve areas for species survival may complicate the situation. With warming temperatures, many species are expected to move poleward (Parmesan *et al.*, 1999) or upward in altitude (Pounds *et al.*, 1999). This implies that the locations of hotspot reserves may need to allow for such movement, which may require a somewhat larger region to be conserved. Even with these efforts, some species may not be conserved because they presently are as far poleward (e.g., fynbos region at the southern tip of South Africa) or as high in altitude (e.g., cloud forests in Costa Rica) as they can be.

#### *5.4.4.3. Replacing Lost Ecosystem Services*

Humans may need to adapt not only in terms of wildlife conservation but also to replace lost ecological services normally provided by wildlife. It may be necessary to develop adaptations to losses of natural pest control, pollination, and seed dispersal. Although replacing providers of these three services sometimes may be possible, the alternatives may be costly (Buchmann and Nabhan, 1996). Finding a replacement for other services, such as contributions to nutrient cycling and ecosystem stability/biodiversity, are much harder to imagine. In many cases, such as the values of wildlife associated with subsistence hunting and cultural and religious ceremonies, any attempt at replacement may represent a net loss.

In many agricultural/silvicultural systems, pesticides are used to prevent losses to pests (insects, pathogens, some vertebrates). In the past 50 years, pesticide use worldwide has increased more than 25-fold (Worldwatch Institute, 1999). The estimated cost of pesticide use in the United States in the mid-1990s was US$11.9 billion (equivalent to 4.5% of total U.S. farm production expenditures), and worldwide use was US$30.6 billion (Aspelin and Grube, 1999). Given that these values are for systems that still had some natural pest control, changes in wildlife distributions might necessitate changes in economic expenditures for pesticides. However, pesticides often kill more than the target species, possibly eliminating natural predators that keep pest populations low. For example, increased pesticide use in Indonesia between 1980 and 1985 led to the destruction of the natural enemies of the brown planthopper. Subsequent increases in planthopper numbers caused reductions in rice yields estimated to cost US$1.5 billion (FAO figures cited in Pimentel *et al.*, 1992).

Adaptation to loss of natural pollinators may be possible in some cases. Farmers sometimes lease bee colonies to pollinate their crops. Although this may be an option for the ~15% of

### Box 5-8. Limitations of Captive Breeding and Translocation to Conserve Biological Diversity Threatened by Climate Change

In some cases, threatened populations of sensitive species could be augmented or reestablished through captive breeding for reintroduction, especially if the degree of climate change proves to be small or moderate. In combination with habitat restoration, such efforts may be successful in preventing the extinction of small numbers of key selected taxa. Similarly, translocation of wildlife between areas within their natural range also might mitigate the effects of small to moderate climate change. This strategy has been applied successfully to augment or restore depleted populations of various species (Boyer and Brown, 1988).

Captive breeding for reintroduction and translocation are likely to be less successful if climate change is more dramatic. Such change could result in large-scale modifications of environmental conditions, including loss or significant alteration of existing habitat over some or all of a species'range (Myers *et al.*, 2000). Captive breeding and translocation therefore should not be perceived as panaceas for the loss of biological diversity that might accompany dramatic climate change, especially given the current state of the environment. Populations of many species already are perilously small; further loss of habitat and stress associated with severe climate change may push many taxa to extinction.

One limitation to captive breeding is the lack of space available to hold wildlife for breeding purposes. Zoos and offsite breeding facilities can be expected to accommodate no more than a small fraction of the number of species that might be threatened. Recent studies have indicated that no more than 16 snake species and 141 bird species could be accommodated and sustained in accredited North American zoos and aquariums in long-term management programs (Quinn and Quinn, 1993; Sheppard, 1995).

Captive breeding programs are expensive, and locating funding to support large numbers of programs could be difficult (Hutchins *et al.*, 1996). For example, it costs US$22,000 to raise a single golden lion tamarin in the United States and reintroduce it to its native Brazil (Kleiman *et al.*, 1991). Part of the cost associated with such programs includes the extensive scientific studies that must be conducted for the program to be successful. Reintroduction is technologically difficult and unlikely to be successful in the absence of knowledge about the species'basic biology and behavior (Hutchins *et al.*, 1996). Rearing and release strategies must be tested experimentally, and released animals must be monitored to assess the efficacy of various methods (Beck *et al.*, 1994). In the case of black-footed ferrets (*Mustela nigripes*) and golden lion tamarins, it took more than a decade to develop the knowledge base required for success.

If wildlife translocation involves moving species outside their natural ranges, other problems may ensue. Exotic species can have devastating effects on host ecosystems, including extinction of native fauna (McKnight, 1993). The unpredictable consequences of species introductions means that translocation is severely limited in its ability to conserve species that are threatened by climate change.

Finally, reintroduction and translocation programs cannot be successful if there is no appropriate habitat left for captive-bred or translocated animals to be released into (Hutchins *et al.*, 1996). Not all of the habitat components that are necessary for a species to survive can be translocated. Entire suites of plant and invertebrate species may be critical elements for a species to succeed in a new environment, but no techniques exist for translocating intact biological communities. Although captive breeding and translocation have potential value for well-studied animals, these strategies appear to be impractical for the vast number of species threatened by rapid climate change.

crops fertilized by domestic honeybees, it may not be an option for crops typically fertilized by wild pollinators or for the 250,000 types of wild plants that are pollinated by 100,000 different invertebrate species (Buchmann and Nabhan, 1996).

### 5.4.5. *Equity Issues*

People in many parts of the world are dependent on wildlife for all or part of their daily nutritional needs. A typical adaptive response to this situation would be to replace all or part of this food with store-bought products. This might be feasible in areas near developed societies but could become increasingly difficult in more remote communities. However, there is more to subsistence hunting than the capture of food. Subsistence hunting plays a major role in the culture of Cree communities in northern Canada. "The killing, preparation, sharing, and consumption of game is central to the seasonal renewal of social relations in Cree villages, and of a relationship to the land which is both secular and sacred in importance" (Scott, 1987). Even if compensations or substitutions for subsistence uses could be made, there still would be equity issues stemming from the loss of culture associated with this way of life.

Many of the aforementioned potential adaptations are more applicable to developed countries than in developing countries. For example, the use of leased honeybees is not applicable to crops fertilized by flying foxes or other wild animals. The same can be said of many forms of seed dispersal. Increased use of pesticides may require more capital than is available to small farmers in some developing countries (Pimentel *et al.*, 1992). Adaptations that may be practical for developed countries simply may not be equitable for developing countries.

### 5.4.6. *Vulnerabilities, Sensitivities, Uncertainties*

In trying to understand and predict potential impacts of climate change on wildlife species, some species and geographic areas are found to be at greater risk than others. Species with small populations, restricted ranges, and specific habitat requirements often are most vulnerable (see Section 5.4.1.2).

Migratory species may be especially vulnerable because they require separate breeding, wintering, and migration habitats. In many cases, one or more of these habitats could be at risk because of climate change and other habitat loss. For example, a large portion of the eastern population of the monarch butterfly (*Danaus plexippus*) winters in a small region of warm-temperate dry forest in Mexico. With climate change, this area is projected to contain trees that are more typical of a subtropical dry forest—probably unsuitable for wintering monarchs (Villers-Ruíz and Trejo-Vázquez, 1998). The relative vulnerability of shorebird migration sites in the United States varies, depending on local geomorphologic and anthropogenic factors, and these factors could exacerbate the effects of sea-level rise. For example, southern San Francisco Bay could lose most of its intertidal feeding habitat with a 2°C average temperature rise (medium confidence) (Galbraith *et al.*, 2001).

One key region of concern is the Arctic and Antarctic, where the temperature increase is projected to be large and changes to habitat availability and accessibility (e.g., freezing and thawing of sea ice and tundra) are expected. Such changes may hamper migration, reproduction, and survival of many species, including birds, polar bears (*Ursus maritimus*), caribou, and musk-oxen (Jefferies *et al.*, 1992; Stirling and Derocher, 1993; Gunn and Skogland, 1997; Stirling, 1997).

Many biological uncertainties exist in the understanding of ecosystem processes. Nevertheless, the balance of evidence suggests that projecting impacts of climatic change on a variety of wildlife species is possible (medium confidence). Laboratory and field studies have demonstrated that climate plays a strong role in limiting species'ranges (high confidence). Only a small fraction of all species have been monitored long enough to detect significant trends. Most monitored species that show significant trends have exhibited changes over the past few decades that are consistent with local warming and expected physiological responses (high confidence). However, potential *specific* changes in wildlife resulting from climate change can be projected only with low confidence for most species because of many possible contributing factors, such as habitat destruction and exotic invasive species. Some species clearly are responding to global change (see Section 5.4.3), and many more changes probably have gone undetected. Researchers are in the process of coupling these discernible changes with various biological theories regarding climate and species spatial and temporal patterns; through this process, we expect that reliable *general* projections can be and in fact are being made (high confidence).

Scientists also need to develop a better understanding of how all of the components of ecosystems work together. The role each species plays in ecosystem services, in wild and managed systems, is necessary to understand risks and possible surprises associated with species loss. Without this information, the probability of surprises associated with species loss is high (medium confidence).

## 5.5. Rangelands (Grasslands, Savannas, and Deserts)

Rangelands here are taken to include deserts (cold, hot, and tundra), grasslands (unimproved), scrub, chaparral, and savannas (after, e.g., Allen-Diaz, 1996). This section does not consider improved grasslands or croplands in detail because they are covered in Section 5.3. It does partly cover tundra because it is an important grazing system; that ecosystem is discussed in more detail in Section 5.9.

Ecosystems within rangelands are characterized by low-stature vegetation because of temperature and moisture restrictions; they are found on every continent (Allen-Diaz, 1996). They are adapted to great variations in temperature and rainfall on an annual and interannual basis, but they generally are confined to areas that have about one unit of precipitation to every 16 units of evapotranspiration (the ratio that is characteristic of drylands; Noble and Gitay, 1996). They often are referred to as "pulse systems" (Noble and Gitay, 1996).

In many countries, human activities in rangelands have evolved in response to variable and often unpredictable climate. Human practices include pastoralism; subsistence farming (Allen-Diaz, 1996); and, more recently, commercial ranching (Canziani and Diaz, 1998). Rangelands also are important for many national economies in terms of foreign cash (e.g., through tourism). They are important stores of biodiversity, including ancestors of many of the cereals (World Bank, 1995), and have high levels of endemism (Barnard *et al.*, 1998). Rangelands are used primarily for grazing and hence livestock production (Squires and Sidahmed, 1997). Thus, they are important for food (mostly livestock, but also wild fruits), fuelwood, wood poles for construction, and feed (Campbell *et al.*, 1997). Other key services are biodiversity, water cycle, and carbon stores. Some of these products and services can be given economic valuation; however, only a small component of the total economic value is represented by products that can be given a market value (Campbell *et al.*, 1997), which suggests that nonmarket values are quite important for some rangelands.

Rangelands are adapted to grazing and other disturbances, such as fire, flood, and insect herbivory (Allen-Diaz, 1996). Vegetation tends to be sparse and thus is not considered worth mechanical harvesting; however, the sparse grass/herbaceous cover is efficiently harvested by grazers (Williams, 1986). In many cases, episodic fires (Bock *et al.*, 1995) are important for providing new and lush growth for grazers, and fire sometimes is used to manage grass-woody shrub balance (Noble *et al.*, 1996), which is important for livestock and meat and wool production (see, e.g., Chapter 12). Human management can be critical to the status of these systems with or without climate change. Previous IPCC reports concluded that fluctuating rainfall and temperatures along with increased human activity (especially in more tropical systems) has led to land degradation and eventually desertification in many areas (Bullock and Le Houérou, 1996; Gitay and Noble, 1996; Canziani and Diaz, 1998).

In this section, food/fiber, biodiversity, and carbon stores are examined in detail (see Table 5-1). Water as a resource is considered elsewhere in the report, at the global level (Chapter 4) and in many of the regional chapters (Chapters 10-12).

### *5.5.1. Current Status of Key Goods and Services*

Estimates for rangeland cover vary between 31 and 51% of the land surface of the Earth (Allen-Diaz, 1996; WRI, 2000); the upper estimate includes tundra grasslands. Rangelands support human populations at low densities (Batchelor *et al.*, 1994) on almost every populated continent. The latest figures suggest a population of at least 938 million people—or about 17% of the world's population (WRI, 2000).

The World Resources Institute report (WRI, 2000) assesses the food/fiber production and biodiversity of rangelands as "fair" (on a scale of excellent, good, fair, poor, and bad) over the past 20–30 years, but the underlying biological ability of the rangelands to continue to support that productivity and biodiversity is declining, suggesting that productivity and the biodiversity assessment may not hold in the future. Livestock production in rangelands is estimated to be 65 Mt in 1998 (WRI, 2000), with global meat production estimated at 225 Mt. Nineteen percent of the world's centers for plant diversity are found in rangelands. For carbon stores, the WRI assessment of rangelands is "good"; again, however, the underlying ability is judged to be declining (WRI, 2000). IPCC (2000) estimates carbon stores in rangelands as 84 Gt C in vegetation (of a global total of 466 Gt), 750 Gt C in soils (of a global total of 2,011 Gt). WRI (2000) gives a range of 405–806 Gt for total carbon stores. Both estimates suggest that rangelands are important carbon stores.

Many of the people in rangelands rely on fuelwood for their daily cooking and heating needs. Estimated fuelwood use as a percentage of total energy use in 1993 for countries dominated by rangelands is about 60–90% (e.g., in Afghanistan, Mozambique, Swaziland); the world average is 6%, and Africa's average is 35% (WRI, 1998).

### *5.5.2. Major Pressures on Key Goods and Services*

Climate change is a pressure on key goods and services in a system that is responsive to climatic fluctuations. It is possible that climate change would lead to increased frequency of extremes of climatic events (such as drought and floods) driven through change in the frequency of ENSO events. Apart from climate change, other direct and indirect pressures from human activities can be important in the delivery of services from rangelands. These pressures include land-use change, which often leads to fragmentation (Allen-Diaz, 1996; Gitay and Noble, 1998; WRI, 2000); changes in the densities of livestock (WRI, 2000); competition for land and water; and altered fire regimes (Russell-Smith *et al.*, 1997). These pressures would have subsequent impacts that cannot be disaggregated from impacts of climate change.

### *5.5.3. Responses of Rangelands and Impacts on Goods and Services*

It has been suggested that for rangelands the possible effect of climate change may be trivial compared with the past and present impacts of human activities, including livestock grazing (Le Houérou, 1996). This may not be entirely true for all rangelands, but it does suggest that it would be difficult to separate the impacts of climate change from the impacts of many of the other pressures that are acting and will continue to act on the system. Subsequent sections assess the direct impacts of climate change; impacts from other aspects of global change, especially the pressures listed above, also are assessed (see

Box 5-9). Major impacts of climate change on key goods and services are mediated through changes in NPP; changes in plant community composition, structure, and forage quality (e.g., through changes in $C_3$ and $C_4$ plants); and changes in plant herbivory and phenology.

### 5.5.3.1. *Impacts of Changes in ENSO and Related Events*

Many of the world's rangelands are affected by ENSO events. There have been several ENSO events in the 1980s and 1990s that followed each other closely (Polis *et al.*, 1997). Thus, rangelands have been subjected to prolonged drought conditions with little recovery time. Simulation results have shown that ENSO events are likely to intensify (Chapter 2) under a doubled $CO_2$ scenario, with the result that dry areas within rangelands are likely to become drier and mesic areas will become wetter during ENSO events (Noble and Gitay, 1996). Thus, some rangelands are likely to experience more extremes of events, with subsequent changes in vegetation and water availability. These changes often are tracked by insect herbivory (Polis *et al.*, 1997), leading to additional impacts.

### 5.5.3.2. *Impacts of Increases in $CO_2$ and Climate Change on Plant Productivity, Species Composition, Decomposition, and Carbon Stores*

The effect of elevated $CO_2$ concentrations on decomposition, plant productivity, and carbon storage could be as large as the

**Box 5-9. Impacts of Some Pressures on Rangelands**

*Land-Use Change*

Major factors in land-use change are conversion of rangelands to croplands (Allen-Diaz, 1996; WRI, 2000) and increased human settlements, especially urbanization (Gitay and Noble, 1998), which lead to fragmentation (WRI, 2000). There have been large-scale changes in land use: For example, in the South Platte Basin in the United States, 40% of the land cover has been converted from rangelands to croplands. This can alter carbon stores, sometimes leading to soil carbon loss of as much as 50% (Allen-Diaz, 1996), but it also can lead to increased plant productivity through irrigated grain production (Baron *et al.*, 1998). Baron *et al.* (1998) conclude that subsequent impacts on the biogeochemical cycles of the basin and on land-atmosphere interactions can affect many rangelands. In many rangelands, native species that occur at low density are used for fuelwood. In some cases, fuelwood collection can lead to decreases in woody vegetation cover and possible land degradation.

*Livestock Production*

Some rangelands have high densities (>100 $km^{-2}$) of livestock, with livestock being moved to take advantage of the periodic growth, especially after rain and/or fires (WRI, 2000). Modeling studies show that increased grazing pressure (i.e., overgrazing) would cause grass and herbaceous productivity to fall below a certain threshold, resulting in increased and rapid rates of land degradation especially under drier and/or hotter climate conditions (Abel, 1997).

*Competition for Land and Water*

There is increased demand for water for direct human consumption and for irrigation (WRI, 2000). Food production obviously is positively affected by increased water use (see Section 5.3 and Chapter 4), but this water use is an added pressure on many rangeland ecosystems. Nomadic pastoralism, which was common in many rangelands until recently, allowed pastoralists to cope with the variable climate of the rangelands they inhabited. Land-use changes (to permanent agriculture, urban areas, conservation, and game reserves that have included loss of sources of permanent water) have led to overall loss of land available. Together with increased human population, this has led to competition for land and changes from pastoral communities to more market-orientated and cash-based economies (Allen-Diaz, 1996).

*Altered Fire Regimes*

The SAR did an extensive review of fires and rangelands. The projected increase in variability in climate led to the conclusion that the frequency and severity of fires will increase in rangelands (Allen-Diaz, 1996), provided that drought and grazing do not lead to a reduction in vegetation biomass. There is historical evidence that fire frequency has changed (increased and decreased, depending on vegetation biomass) in recent decades (see, e.g., Russell-Smith *et al.*, 1997), leading to changes in vegetation composition. Fires also are started by humans; for example, in Africa 25–80% of rangelands are burned every year, often to induce new plant growth (WRI, 2000). This has implications for short-term productivity but possibly long-term land degradation.

impact of climate change alone (Ojima *et al.*, 1993; Parton *et al.*, 1994; Hall *et al.*, 1995).

#### 5.5.3.2.1. Plant productivity

At the global level, plant production is projected to increase in grasslands when climate change and elevated $CO_2$ are combined (Parton *et al.*, 1994) but could be affected by potential changes in disturbance regimes and by land-use practices in some rangelands (Parton *et al.*, 1994, 1995). As $CO_2$ concentrations increase, transpiration per unit leaf area is expected to decrease and WUE will increase (Morgan *et al.*, 1994a; Read *et al.*, 1997; Wand *et al.*, 1999). Thus, increased $CO_2$ could lead to enhanced productivity (e.g., Hunt *et al.*, 1996; Owensby *et al.*, 1996), especially under low soil moisture conditions that are characteristic of rangelands (Wand *et al.*, 1999). However, Bolortsetseg and Tuvaansuren (1996)—using 30-year climatic data as a baseline with projected climate change scenarios that incorporate elevated $CO_2$—report a decrease in productivity in deserts but an increase in colder areas; Gao and Yu (1998)—using a regional model with elevated $CO_2$, a 20% increase in precipitation, and a 4°C increase in temperature—project that NPP of most steppes in China will decrease by 15–20% and NPP of woodland and shrublands and desert grasslands will increase by 20–115%. Productivity also is likely to increase because elevated $CO_2$ could result in enhanced nitrogen uptake (Jones and Jongen, 1996; Coughenour and Chen, 1997). However, the response varies between species: Dominant species show a less enhanced response than rarer species, suggesting a possible change in the composition and structure of the vegetation in rangelands (Jones and Jongen 1996; Berntson *et al.*, 1998). Slower depletion of soil water under increased $CO_2$ concentrations should favor plants that otherwise might do poorly under water stress (Polley *et al.*, 1997), thus altering the species mix.

#### 5.5.3.2.2. Carbon stores

Many rangelands tend to have large belowground stores (Tate and Ross, 1997). Studies suggest a strong interaction between $CO_2$ and temperature on soil carbon fluxes, possibly mediated through NPP. Soil carbon losses of about 2 Gt over 50 years under combined climate change and elevated $CO_2$ have been projected. These losses compare with carbon losses of about 4 Gt from climate change alone (Hall *et al.*, 1995; Parton *et al.*, 1995). Hall *et al.* (1995) project that tropical savannas will be soil carbon sinks under elevated $CO_2$ and climate change. For temperate grasslands, Thornley and Cannell (1997) suggest an annual carbon sink of 0.5–1.5 t $ha^{-1}$ $yr^{-1}$ under a scenario of elevated $CO_2$ and a temperature increase of 5°C; however, grasslands are likely to be a net source under that temperature increase alone. Based on some experimental studies and supported by observations around $CO_2$-venting spring areas in temperate grasslands of New Zealand, future carbon storage could be favored in soils of moderate nutrient status, moderate to high clay content, and low to moderately high soil moisture status (Tate and Ross, 1997). In contrast, Cook *et al.* (1999) show that long-term $CO_2$ enrichment by a natural $CO_2$-venting spring in a subarctic grassland is likely to result in slower soil carbon accumulation compared to the ambient $CO_2$ atmosphere. Soil carbon is greatly affected by management practices, and appropriate management in many rangeland systems could result in carbon sequestration (see below).

Productivity of rangelands also can be affected by changes (which could be a result of climate change) in soil fauna and flora, as well as microbial activity (Yeates *et al.*, 1997; Zaller and Arnone, 1997; Kandeler *et al.*, 1998). These studies, which have not looked at arid systems within rangelands, suggest that elevated $CO_2$ is likely to cause an increase in soil microbial activity and increased soil $CO_2$ production. NPP has been reported to be enhanced by elevated $CO_2$ in nodulated and mycorhizae-infected species (Allen-Diaz, 1996), although specific infection rates often are unchanged.

#### 5.5.3.2.3. Changes in species composition, $C_3$ and $C_4$ plants

$C_3$ and $C_4$ plants react differently to elevated $CO_2$ and climatic factors. In paleostudies in the northern Chihuahan Desert, Buck and Monger (1999) observe a major shift from $C_4$ grasses to $C_3$ desert shrub-dominated vegetation about 7,000–9,000 years BP, associated with a possible increase in aridity and a subsequent increase in $C_4$ grasses, with an increase in moisture about 4,000 years BP. Photosynthesis in $C_3$ plants is expected to respond more strongly to $CO_2$ enrichment than in $C_4$ plants (Mayeux *et al.*, 1991). If this is the case, it is likely to lead to an increase in the geographic distribution of $C_3$ plants (many of which are woody plants) at the expense of $C_4$ grasses (Noble and Gitay, 1996; Ehleringer *et al.*, 1997; Polley *et al.*, 1997). However, the impacts are not that simple. In pot experiments, elevated $CO_2$ is reported to improve water relations and enhance productivity in the $C_4$ shortgrass steppe grass *Bouteloua gracilis* (Morgan *et al.*, 1994b). In modeling and experimental studies, NPP of $C_3$ and $C_4$ grasses increase under elevated $CO_2$ for a range of temperatures and precipitation (Chen *et al.*, 1996; Hunt *et al.*, 1996; Owensby *et al.*, 1999) but could result in relatively small changes in their geographical distributions (Howden *et al.*, 1999b). There are additional interactions with soil characteristics and climatic factors. Epstein *et al.* (1998)—using existing distribution of grasslands in the U.S. Great Plains combined with modeling studies but not including the effect of elevated $CO_2$—found that $C_3$ plants are more productive in cooler, drier conditions and do particularly well in soils with high clay. There was a differentiation of the response of $C_4$ plants into two height classes: shortgrasses and tallgrasses. Productivity of $C_4$ shortgrasses increased with increasing mean annual temperature but decreased with mean annual precipitation and sand content, whereas $C_4$ tallgrass productivity tended to increase with mean annual precipitation and sand content and was highest at intermediate values of mean annual precipitation. Coffin and Lauenroth (1996) used gap models linked to a soil water model and found that $C_4$ grasses increase in dominance as a result of increases in temperature in all months. The coolest sites that currently are dominated by $C_3$ grasses are predicted to shift to dominance by $C_4$ grasses, whereas sites that currently are dominated by $C_4$ grasses have an increase in importance of this group. When the number of frost days is decreased, subtropical $C_4$ grasses may invade the more palatable $C_3$ grasses, leading to a decrease in forage quality. The rate and duration of this change is likely to be affected by human activity; high grazing pressure may mean more establishment sites for $C_4$ grasses (Panario and Bidegain, 1997).

#### 5.5.3.2.4. *Woody species, grass, and weeds mix*

There have been some suggestions that woody weed encroachment into herbaceous parts of rangeland (up to three-fold in some cases) may be a result of changes in regional climate (Brown *et al.*, 1997b) and may be facilitated by the increase in atmospheric $CO_2$ concentration since industrialization (Polley *et al.*,1996). However, the more plausible explanation for the present observed increase in woody weeds in many rangelands is that it is a result of land-use change (especially increased grazing and changes in fire regime) as well as land degradation (Bond *et al.*, 1994; Archer *et al.*, 1995; Brown *et al.*, 1997b; Gill and Burke 1999). A modeling study of semi-arid woodlands in Australia under elevated $CO_2$ and climate change suggests that future potential burning opportunities may increase as a result of the $CO_2$ response of the grass layer, which allows adequate fuel to build up more frequently, provided stocking rates are not increased commensurately (Howden *et al.*, 1999c). Thus, regional climate change and elevated $CO_2$, as concluded in the SAR, may change the balance from more herbaceous species (grasses and herbs) to more woody species (mainly shrubs), subsequently affecting productivity, decomposition, and fire frequency of the system, as well as forage quality (Allen-Diaz, 1996; Noble and Gitay, 1996) and regional carbon cycling (Gill and Burke, 1999).

#### 5.3.3.2.5. *Changes in phenology*

Increased variability of rainfall and temperature is likely to affect the phenology of plant and animal species (Gitay and Noble, 1998). It also would affect animal abundance and feeding behavior (see Section 5.4) and would be critical for some pests and diseases and availability of forage for livestock and other mammals (Watt *et al.*, 1995). Species composition changes also could occur. Brown and Carter (1998) manipulated the climate of small areas of temperate grasslands and found that weeds become established in gaps created through increased occurrence of spring or summer drought. Warmer winters induce a late spring drought—which, when combined with an imposed summer drought—leads to significant reduction in cover of nonweeds especially and an increase in weedy species. However, if summer moisture is increased, grasses become dominant. Thus, if climate change results in changes in summer and winter soil moisture and land use creates gaps, this is likely to affect species composition in some grasslands. Fuller and Prince (1996)—using satellite data and NDVI to detect vegetation change—found that there is an early greening of the Miombo woodlands. These woodlands are sensitive to the arrival of spring rains, thus might undergo changes if there is a shift in rainfall patterns.

### 5.5.3.3. *Biogeographical Shifts and Land Degradation*

Section 5.2 summarizes the main outputs of recent model outputs. Previous IPCC reports have documented boundary shifts in rangeland vegetation with adjoining vegetation as a response to past climate changes (see, e.g., Gitay and Noble, 1998). The SAR concluded that model outputs suggest significant changes in the boundaries of grasslands and deserts, with expansion of warm grass/shrub types and a decrease in tundra systems (Allen-Diaz, 1996). More recent studies by Allen and Breshears (1998) do not support this rapid change, except perhaps in cases of high drought-induced mortality. Human activities are likely to affect the final changes, especially in tropical and subtropical areas (Allen-Diaz, 1996; Villers-Rúiz and Trejo-Vázquez, 1998), and will negate any gains resulting from amelioration of climate (Gitay and Noble, 1998).

Palaeoecological evidence shows shifts in rangeland vegetation—for example, between the last glacial maximum (LGM) and the Holocene in the Sahara-Gobi desert belt (Lioubimtseva *et al.*, 1998). These shifts have resulted in changes in the carbon stores. Dry and cool conditions during the LGM (about 20,000 years BP) resulted in the spread of arid and semi-arid ecosystems at northern and southern margins of the desert belt. The southern limit of the Sahara migrated southward at least 400 km relative to its present position and almost 1,000 km southward compared to the mid-Holocene (about 9,000 years BP). The northern margin of the temperate deserts and dry steppes of central Asia shifted northward by about 300 km over Kazakhstan, southern Siberia, and Mongolia. During the last world deglaciation, the Sahara-Gobi desert belt was a sink for approximately 200 Gt of atmospheric carbon, but since the mid-Holocene it has been a source.

Modeling studies by Ni (2000), using the BIOME3 model (see Section 5.2) with present climate for the Tibetan Plateau and elevated $CO_2$ (550 ppm), suggest that there is likely to be a large reduction in temperate deserts, alpine steppe, desert, and ice/polar desert; a large increase in temperate shrubland/meadow and temperate steppe; and a general poleward shift of vegetation zones. Villers-Rúiz and Trejo-Vázquez (1998) found that projected impacts of climate change (using two climate change scenarios generated by the Canadian Climate Center) would result in expansion of some rangeland systems into moist forest areas in Mexico. However, under a GFDL scenario there would be an increase in the distribution of tropical humid and wet forests into rangelands. Zimov *et al.* (1995) suggest that shifts in vegetation boundaries in the rangelands could be influenced by mammalian grazing (in this case, the study area did not have livestock); thus, model projections should include the impact of mammal populations for realistic projections of future vegetation changes (see also Section 5.4).

Desertification tends to be associated with land degradation in rangelands (Pickup, 1998). However, desertification aggregates many land degradation processes and can be exacerbated by climate change. The main processes involved in land degradation include physical, chemical, and biological degradation (Lal *et al.*, 1989, 1999; see also Section 5.3.3) and result in soil compaction, destruction of soil surfaces (which often are kept intact by the presence cyanobacteria, lichens, and mosses), and salinization (Dregne, 1995; Pickup, 1998). These processes are sensitive to livestock trampling. The end result can be decreased water penetration, increased surface runoff, and exposure of soils to wind and water erosion, thereby changing the water and nutrient cycles (Belnap, 1995). In some cold deserts, changes to the soil surface and thus water penetration are quite critical because low vegetation cover, low surface rooting plants, few nitrogen fixers, and low soil temperatures result in slow recovery in nutrient cycles (Belnap, 1995). Under many circumstances, recovery of the soil surface in many arid and semi-arid parts of the world is estimated to take several hundred years, with nitrogen-fixing capability (mostly from the soil crust) taking at least 50 years—making them generally vulnerable to further desertification and soil degradation.

Land degradation is a nonlinear process with thresholds that can be triggered by climatic factors and human activities (Puigdefabregas, 1998). This has implications for the sensitivities and vulnerability of these systems. Villers-Rúiz and Trejo-Vázquez (1998) suggest that the impacts of climate change are likely to be minor compared to land degradation in parts of rangelands in Mexico where conversion of forests into grasslands is mainly for cattle ranching. Lavee *et al.* (1998) suggest on the basis of field and experimental work that if climate becomes more arid in Mediterranean to arid areas of Israel (through decreases in annual precipitation and the frequency and intensity of precipitation, together with increasing temperature), productivity would decrease—leading to decreased organic matter, soil permeability,and rates of infiltration and thus land degradation. The rate of change of these variables along the climatic transect is nonlinear. A step-like threshold exists at the semi-arid area, which sharply separates the Mediterranean climate and arid systems. This means that only a relatively small climatic change would be needed to shift the borders between these two systems. Because many regions of Mediterranean climate lie adjacent to semi-arid areas, the former are threatened by desertification in the event of climate change.

Irrigation in semi-arid areas is a major cause of secondary (or dryland) salinity. Elevated $CO_2$ increases WUE, so it may reduce irrigation needs, whereas warming may enhance water demand and exacerbate problems of secondary salinity (Yeo, 1999). Increased subsoil drainage under elevated $CO_2$ may increase the risk of secondary salinization and areas potentially affected (Howden *et al.*, 1999d).

#### *5.5.3.4. Changes in Biodiversity*

This subject is reviewed extensively in the SAR and the *Special Report on Regional Impacts of Climate Change*. More recent studies mostly suggest that recently observed changes in rangelands, especially animals (see Section 5.4), are occurring as a result of land degradation. For example, in Australia, land degradation from pastoral activity as well as the impacts of introduced feral mammals (goats, foxes, rabbits) has resulted in extinctions of small- and medium-sized mammals (Pickup, 1998). There have been past changes in plant biodiversity as a result of changes in rangeland vegetation. For example, Prieto (2000), using paleorecords from eastern plains of Argentina, found that there was a replacement of dry steppe by humid grasslands during the late glacial-Holocene transition. The plant composition indicates a frequently disturbed habitat with a lot of weedy species that did not exist in the records in previous ages.

### *5.5.4. Adaptation Options*

Human societies in rangelands would have to adapt to changes in climate, especially temperature and water availability. The SAR concludes that the lack of infrastructure and investment in resource management in many countries dominated by rangelands makes some adaptation options problematic (Allen-Diaz, 1996) but also makes these areas more sensitive to impacts of climate change (Gitay and Noble, 1998). Nevertheless, some adaptation options are available for many of the rangelands.

Specific examples of the interaction between climate change and management decisions may be highlighted better at the regional level. For example, in Australia, Pickup (1998) found that substantial shifts in rainfall have occurred over the past 100 years. If climate change results in further shifts in rainfall patterns, the major impacts are likely to be related to increased climate variability. Pastoral management decisions in these rangelands tend to be taken over the short term; wetter periods generate unrealistic expectations about land use and high stocking rates, which drier periods are unable to support. This has and would lead to land degradation.

#### *5.5.4.1. Landscape Management*

Rangelands consist of a mosaic of various ecosystem types (WRI, 2000) with soil and water processes as well as associated nutrient cycles that operate at the landscape or regional scale (Coughenour and Ellis, 1993). Human use of rangelands often affects landscape processes (e.g., water flow, soil erosion) and changes in processes such as productivity, decomposition, and fire. Thus, possible future adaptation options might have to be sought at the landscape level (Aronson *et al.*, 1998) and over long time frames (Allen-Diaz, 1996). Because many rangelands are in semi-arid and arid parts of the world, actions to reduce destruction of the soil crust (which are important for soil stabilization and nitrogen fixation) and thus land degradation are extremely important. These actions could include adjustment in the time and intensity of grazing (Belnap and Gillette, 1998). Restoration of degraded soils has vast potential to sequester carbon in soil and aboveground biomass (Lal *et al.*, 1999), although restoration could be costly (Puigdefabregas, 1998).

#### 5.5.4.2. Selection of Plants and Livestock

Selection of plants (e.g., legume-based systems) and animal species and better stock management are likely to be the most positive management options (Chapman *et al.*, 1996; Gitay and Noble, 1998). Rotational cropping and decreased use of marginal lands might be necessary in rangeland management. This might mean more intensive land management in some areas, leading to more reliable food supplies and perhaps reduction in methane production from livestock (decrease in methane production would be caused by improved forage quality; Allen-Diaz, 1996). Potential stocking rates could be determined via satellite imagery (Oesterheld *et al.*, 1998). However, the decision on stocking rates might still be made on a social basis, especially given the values associated with livestock in many pastoral rangeland communities (Turner, 1993).

#### 5.5.4.3. Multiple Cropping System and Agroforestry

As human population in rangelands increases and land use changes, some traditional practices are becoming less appropriate. Some of the options for sustainable agriculture could include efficient small-scale or garden irrigation, more effective rain-fed farming, changing cropping patterns, intercropping, or using crops with lower water demand (Lal, 1989; Batchelor *et al.*, 1994; Dixon *et al.*, 1994a; Dabbagh and Abdelrahman, 1998). Conservation-effective tillage is an option that could help to achieve improved productivity (Benites and Ofori, 1993). Agroforesry, using potential fuelwood species, also is an obvious option to alleviate land degradation as well as to meet some social needs. Management (e.g., appropriate coppicing in Uganda) or other practices (e.g., collecting only dead or fallen wood; Benjaminsen, 1993) are considered essential for maintenance of fuelwood species. Where woody weeds are increasing, they could be used for domestic fuel supply; otherwise, management options (e.g., use of fire along with regulation of grazing—Archer, 1995; Brown and Archer, 1999) might have to be implemented to control woody weeds.

#### 5.5.4.4. Role of Community Participation and Public Policy

Community participation in decisionmaking and management, along with public policy, can be a favorable and critical issue in implementing some adaptation options. This could result in better management of rangelands (Pringle, 1995; Allen-Diaz, 1996; Thwaites *et al.*, 1998), thereby probably meeting conservation objectives (Pringle, 1995). Decisions to be made might include:

- Determinations about appropriate stocking rates, which might require discussion and negotiations among stakeholders, especially because stocking rates might be more social than technically oriented (Abel, 1997)
- Choosing some agroforestry practices that fulfil local needs, especially because many communities rely on fuelwood (Benjaminsen 1993)
- Diversification, since some communities could get or have already gotten involved in tourism as a way of highlighting some of the unique flora, fauna, and landscape features, thereby conserving the systems, reducing some of the impacts, and obtaining cash (Hofstede, 1995).

In dealing with options for reducing the consequences of land degradation in the future, public policy may have a crucial role (Hess and Holechek, 1995), especially because decisions at the landscape level (which are likely to include many different land tenures) are going to be increasingly important. Policies could be developed to address multiple pressures and, over the long term, to encompass sustainable land management and could include investments by governments to improve rangeland status (Morton *et al.*, 1995; Pickup, 1998).

### 5.5.5. Vulnerabilities and Sensitivity to Climate Change

For the future of rangelands, it is important to reduce the vulnerability of these systems to climate change. This is likely to be achieved by considering social and economic factors that determine land use by human populations (Allen-Diaz, 1996). Soil stability and thus maintenance of water and nutrient cycles are essential in reducing the risk of desertification. Any changes in these processes could make rangelands particularly vulnerable to climate change. Land degradation is a nonlinear process with thresholds that make these systems sensitive and vulnerable (Puigdefabregas, 1998). Prevention of land degradation might be a cheaper option than restoration, which can be costly (Puigdefabregas, 1998). Some studies suggest that changes in rainfall pattern may make some vegetation types within rangelands more vulnerable (e.g., Miombo woodlands—Fuller and Prince, 1996) if growing periods could not shift or if these growing periods coincide with insect outbreak.

## 5.6. Forests and Woodlands

Forests and woodlands[2] provide many goods and services that society values, including food, marketable timber and non-wood products (fuel, fiber, construction material), medicines, biodiversity, regulation of biogeochemical cycles, soil and water conservation (e.g., erosion prevention), carbon reservoirs, recreation, research opportunities, and spiritual and cultural values. Forests play a key role in the functioning of the biosphere—for example, through carbon and water cycles (the latter is discussed in Chapter 4)—and hence indirectly affect the provision of many other goods and services (Woodwell and MacKenzie, 1995). Changes in global climate and atmospheric composition are likely to have an impact on most of these goods and services, with significant impacts on socioeconomic systems (Winnett, 1998).

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[2] In this section, “forest” includes both forest land and woodlands, unless otherwise specified.

Since the SAR, many studies have dealt with changes in the structure, composition, and spatial patterns of forests (e.g., VEMAPMembers, 1995; Smith *et al*., 1996a; Bugmann, 1997; Shriner *et al*., 1998). The biogeochemical literature has focused on the carbon cycle (e.g., Apps and Price, 1996; Fan *et al.*, 1998; Steffen *et al*., 1998; Tian *et al*., 1998; IPCC, 2000; Schimel *et al*., 2000). There is an expectation that directed land-management practices can either increase or retain carbon stocks in forests, thereby helping to mitigate increases in atmospheric $CO_2$ levels; this is discussed elsewhere (IPCC, 2000; see also TAR WGIII Chapter 4 and TAR WGI Chapter 3).

The influence of climate change on forests and associated goods and services is difficult to separate from the influence of other global change pressures such as atmospheric changes, land use, and land-use change resulting from human activities. The State-Pressure-Impacts approach outlined in Section 5.1 is used here as a framework to examine interrelated responses to global change and expected changes in supply of services from forests and woodlands (see Table 5-1). In this section, the focus is on some of the important pressures, impacts, and responses for three goods and services provided by forests and woodland ecosystems: carbon, timber, and non-wood goods and services. The impact on biodiversity in forests is covered in Sections 5.2 and 5.4. The state of the sector and the pressures acting on it, as well as possible responses, impacts, and adaptation opportunities, will differ among the regions of the world; here the focus is on global commonality.

### 5.6.1. Current Status and Trends

This section presents an overview of the current status and trends for forests in general; specific regional trends are presented in some of the regional chapters (Chapters 10–17).

The world's forests cover approximately 3,500 Mha (FAO, 1997a), or about 30% of the total land area (excluding Greenland and Antarctica). About 57% of the world's forests, mostly tropical, are located in developing countries. About 60% are located in seven countries (in order): the Russian Federation, Brazil, Canada, the United States, China, Indonesia, and the Democratic Republic of Congo.

In 1995, plantation forests were estimated to cover 81 Mha (2.3% of total estimated cover) in developing countries and about 80–100 Mha in developed countries (FAO, 1997a). They play an important role, particularly in the production of industrial roundwood and fuelwood, restoration of degraded lands, and provision of non-wood products (FAO, 1997a). Some countries obtain 50–95% of their industrial roundwood production from plantations that cover 1–17% of their total forest area (Sedjo, 1999). In many developing countries, plantations often occur as community woodlots, farm forests, and agroforestry operations.

Between 1980 and 1995, the area of the world's forests decreased by about 180 Mha (5% loss of total forest area in 15 years) as a result of human activities (FAO, 1997a). About 200 Mha were converted to agriculture (subsistence agriculture, cash crops, and ranching), but this loss was partially offset by about 40 Mha increase in plantations. In developed countries, forests increased over the same period by about 20 Mha through afforestation and natural regeneration on land no longer in use by agriculture, despite losses of forests to urbanization and infrastructure development. Loss of native forest in developing countries (tropical and nontropical) appears to have slowed during 1990–1995, with an overall loss of about 65 Mha (FAO, 1997a). However, other changes such as fragmentation, nonsustainable logging of mature forests, degradation, and development of infrastructure—all leading to losses of biomass—have occurred over large areas. Of about 92 Mha of tropical closed forest that underwent a change in cover class during 1980–1990, 10% became fragmented forest and 20% was converted to open forest or extended forest fallow (FAO, 1996). There are no global estimates of forest degradation, but data from specific areas give an indication of the extent of degradation. Logging practices damage and degrade more than 1 Mha $yr^{-1}$ of forest in the Brazilian Amazon; surface fires (e.g., those in 1998) may burn large areas of standing forest in these regions (Cochrane *et al.*, 1999; Nepstad *et al.*, 1999; see also Chapter 14). These authors conclude that present estimates of annual deforestation for Brazilian Amazonia capture less than half of the forest area that is impoverished each year—and even less during years of severe drought. In the boreal zone, there has been continuing encroachment by agriculture and development of infrastructure (roads, survey lines, wellheads, etc.) that open access to primary forests. Preliminary estimates of these effects for Canada, for example, indicate a net loss of 54,000–81,000 ha $yr^{-1}$ of forest over the period 1990–1998 as a result of various activities (Robinson *et al.*, 1999).

#### 5.6.1.1. Carbon Pools and Flux

Carbon pools in the world's forests are estimated to be 348 Gt C in vegetation and 478 Gt C in soil (to 1 m) (updated since Dixon *et al.*, 1994b, by Brown, 1998). Bolin and Sukumar (2000) based their numbers on Dixon *et al.* (1994b). The largest vegetation and soil carbon pools are in tropical forests (60 and 45% of the total, respectively) because of their large extent and relatively high carbon densities. Carbon stocks in forests vary, depending on the type of forest in relation to climate, soil, management, frequency of disturbances, and level of human-caused degradation.

Based on traditional carbon inventories, terrestrial ecosystems were shown to be carbon sources during the 1980s in the SAR (Brown *et al.*, 1996); a high- and mid-latitude forest sink was exceeded by the source from low-latitude forests. However, recent work—using atmospheric measurements and modeling—suggests that terrestrial ecosystems appear to be net sinks for atmospheric carbon, even when losses from land-use change are taken into account (Bolin and Sukumar, 2000). For the 1980s, a net storage increase of 0.2 ± 1.0 Gt C $yr^{-1}$ by terrestrial ecosystems (largely in forests) was estimated as the difference between a net emission of 1.7 ± 0.8 Gt C $yr^{-1}$ from land-use changes (primarily in the tropics) and global terrestrial uptake of 1.9 ± 1.3 Gt C $yr^{1}$ (Bolin and Sukumar, 2000). In the 1990s, estimated net emissions from land-use change decreased slightly to 1.6 ± 0.8 Gt C $yr^{-1}$, and global terrestrial uptake increased to 2.3 ± 1.3 Gt C $yr^{-1}$, resulting in a net terrestrial uptake of 0.7 $\underline{+}$1.0 Gt C $yr^{-1}$ (Bolin and Sukumar, 2000). This terrestrial net sink of carbon arises as the net effect of land-use practices (agricultural abandonment and regrowth, deforestation, and degradation); the indirect effects of human activities (e.g., atmospheric $CO_2$ fertilization and nutrient deposition); and the effects of changing climate, climatic variation, and disturbances. The relative importance of these different processes is known to vary strongly from region to region.

Regional source and sink relationships also have been inferred by techniques of inverse modeling of observed atmospheric $CO_2$ gradients and circulation patterns (Ciais *et al.*, 1995; Fan *et al.*, 1998). These estimates are relatively imprecise and are difficult to relate to those that are based on forest inventory data. For example, the study by Fan *et al.* (1998) suggests that 1.4 Gt C $yr^{1}$ was taken up by terrestrial biota in North America in the 1980s. However, mechanistic models and measurements based on forest inventories do not agree with the magnitude or spatial distribution of this carbon sink (Holland and Brown, 1999; Potter and Klooster, 1999).

Temperate forests are considered to be net carbon sinks at present, with estimates of 1.4–2.0 t C $ha^{-1}$ $yr^{1}$ (Nabuurs *et al.*, 1997; Brown and Schroeder, 1999; Schulze *et al.*, 1999). These findings are consistent with recent estimates of carbon in woody biomass, based on statistics for 55 temperate and boreal countries; these statistics indicate a general increase in forest biomass from the 1980s to the 1990s (UN-ECE/FAO, 2000c). Changes in forest management (reduction of harvest levels, increased regeneration effort, and administrative set-asides), as well as changes in the environment (N and $CO_2$ fertililization), appear to have contributed to this trend, but the relative contribution of different factors varies among forest regions and countries (Kauppi *et al*., 1992; Houghton *et al.*, 1998, 1999; Brown and Schroeder, 1999; Liski *et al.*, 1999; Nadelhoffer *et al.*, 1999).

In boreal forests, carbon budgets vary strongly among different forest types (Apps *et al.*, 1993; Bonan, 1993, Shvidenko and Nilsson, 1994). Although some boreal forest regions currently appear to be net carbon sources (Shepashenko *et al.*, 1998; Kurz and Apps, 1999), others appear to be net sinks, varying between 0.5 and 2.5 t C $ha^{-1}$ $yr^{-1}$ (Shvidenko and Nilsson, 1994; Jarvis *et al.*, 1997). The difference between annual increment and net fellings reported to the FAO (reported in UN-ECE/FAO, 2000c) does not account for changes in the frequency and severity of disturbances that have a large influence on source and sink relationships in boreal forests (Kasischke *et al.*, 1995; Kurz and Apps, 1999; Bhatti, 2001). For example, detailed analyses of forest inventory data together with observed changes in disturbance over time indicate that Canadian forest ecosystems changed from a modest sink (0.075 Gt C $yr^{-1}$) from 1920–1970 to a small net source of 0.050 Gt C $yr^{-1}$ in 1994 (Kurz and Apps, 1999). Similarly, in Russia between 1983 and 1992, managed forest ecosystems in the European portion, where disturbances were relatively controlled, were a sink of 0.051 Gt C $yr^{-1}$ but a net source of 0.081–0.123 Gt C $yr^{-1}$ in the less intensively managed Siberian forests of the east (Shepashenko *et al.*, 1998). Factors that were not included in these analyses that may offset losses of biotic carbon from disturbed forests include nitrogen deposition and $CO_2$ fertilization (Chen *et al.*, 2000; Schimel *et al.*, 2000), but experimental verification of these influences is not yet possible from inventory data.

In the tropics, forests are still reported to be a net carbon source as a result of land-use change. Although some studies suggest net carbon uptake in some tropical forests (Grace *et al.*, 1995; Phillips *et al.*, 1998), losses associated with high rates of forest clearing and degradation exceed such gains. The magnitude of the net carbon source from the tropics has been reported to be about 0.1 Gt C $yr^{-1}$ lower in the first half of the 1990s than in the 1980s (Houghton and Hackler, 1999; Houghton *et al.*, 2000; Houghton, 2001), mostly because of reduced rates of deforestation in the 1990s (FAO, 1996, 1997a).

In summary, carbon stored in forest ecosystems appears to be increasing, and at an increasing rate with about 0.2 Gt C $yr^{-1}$ being stored in the 1980s and 0.7 Gt C $yr^{-1}$ in the 1990s. Most storage occurs in temperate forests, with a small net source from tropical forests and boreal forests varying depending on the disturbance regime they experience.

### *5.6.1.2. Timber and Non-Wood Products*

Forests contribute to GDPin three main ways: industrial wood products, fuelwood, and the economic impacts of recreation and non-wood products (e.g., mushrooms). Regional estimates generally are available only for the first (industrial) component, which captures the direct value of harvesting timber and the value added by manufacturing. At a global level, forestry contributes approximately 2% to GDP(FAO, 1997a)—6% in Africa, 3% in South America, 2% in North and Central America, and 1% in Europe. In developing countries, forestry contributes 4% to GDP; in developed countries the contribution is 1%. Total industrial timber production in 1997 was 1.5 billion $m^3$, with more than 60% coming from developed countries (FAO, 1997a).

Although income and population growth influence demand for industrial timber, recycling and technological change (e.g., use of wood chips for manufactured products) can affect the quantity harvested from forests. Total industrial wood harvests have remained relatively constant over the past 20 years, even as global population and incomes have increased (FAO, 1998). Global per capita consumption of wood (including fuelwood and roundwood) is about 0.6 $m^3$ $yr^{-1}$; this level of consumption has been relatively stable over the past 40 years (Solberg *et al.*, 1996). Global fuelwood production in 1996 is estimated to be 1.9 billion $m^3$, with 90% of this production occurring in developing countries (FAO, 1997a). In 1994, annual per capita fuelwood consumption in developing countries was 0.39 $m^3$, versus 0.16 $m^3$ in developed countries (FAO, 1997a). It is estimated that 2 billion people rely on wood and charcoal for fuel (mostly derived from forests), and ensuring an adequate and sustainable supply will continue to be an important pressure on forests.

Non-wood forest products (NWFP)—such as edible mushrooms, nuts, fruits, palm hearts, herbs, spices, gums, aromatic plants, game, fodder, rattan, medicinal and cosmetic products, resins, and the like—make important contributions to household income, food security, national economies, and environmental objectives of conservation of biodiversity (FAO, 1997a). It is estimated that about 80% of the population of the developing world depends on NWFP to meet some of their health and nutritional needs. Several million households worldwide depend heavily on these products for subsistence consumption and income.

### *5.6.2. Pressures on Forests and Woodlands*

Forests have many pressures acting on them that result in changes to their structure and composition, as well as their function (see Figure 5-1). These structural changes, in turn, alter the function of forests in the physical climate system (Sellers *et al.*, 1990; Apps, 1993).

#### *5.6.2.1. Climate Variability and Climate Change*

Changes in climatic conditions affect all productivity indicators of forests (NPP, NEP, and NBP; see Box 5-1) and their ability to supply goods and services to human economies. The effects on forested area and forest productivity, however, vary from location to location, with gains in some regions and losses in others. Furthermore, the impacts vary among different measures of ecosystem productivity. For example, in boreal and alpine forests—where short growing seasons and heat sums are limiting factors to growth—NPP of many forest stands may increase with increasing temperature (Bugmann, 1997; but see Barber *et al.*, 2000), whereas NEP decreases as a result of increased decomposition (Schimel *et al.*, 1994; Valentini *et al.*, 2000; but see Giardina and Ryan, 2000). If higher temperatures lead to summer drought, even NPPmay decrease as a result of lowered photosynthetic rates associated with reduced stomatal conductance (Sellers *et al.*, 1997), exacerbating the decrease in NEP from decomposition. If drier conditions also result in increased fires, biomass and soil carbon losses may result in negative NBP (Wirth *et al.*, 1999; Apps *et al.*, 2000).

Projected changes in forest area, structure, NPP, and NEP as a result of climate change vary by forest type and biome (Neilson *et al.*, 1998). Climate change also is likely to include changes in seasonality (Myneni *et al.*, 1997), timing of freeze-thaw patterns (Goulden *et al.*, 1998), the length of the growing season, nutrient feedbacks (Tian *et al.*, 1998), disturbance regimes (Kurz *et al.*, 1995), and diurnal temperature patterns (Clark and Clark, 1999). Changes in intra-annual variation that fall outside the historical norm for a particular region also may have catastrophic effects—for example, through local climatic extremes or through late and early frost (Repo *et al.*, 1996; Ogren *et al.*, 1997; Colombo, 1998). These factors are likely to influence the distributional range of some tree species (Macdonald *et al*., 1998; Rehfeldt *et al.*, 1999a). Changes in precipitation may not have immediate effects on mature and old-growth forests, which have well-established root systems, but are likely to have pronounced effects on regeneration success for some species following disturbance, such as harvest or fire (Hogg and Schwarz, 1997; Price *et al.*, 1999a,b).

#### *5.6.2.2 Changes in Disturbance Regimes*

At the landscape scale, changes in the disturbance regime introduce instabilities in forest age-class distributions (Bhatti *et al.*, 2000) and eventually in the distribution of plant species. If changes in disturbances are caused or accompanied by changes in environmental conditions, the responses of the forest ecosystem can be very complex. Changes in disturbance regimes, spatially and over time, can be exacerbated or mitigated by human activities (e.g., by fire ignitions and fire suppression).

##### *5.6.2.2.1. Pressures from fires*

Large areas of mixed savanna-woodlands in dry tropical zones of Africa, South America, Australia, and large areas of tropical humid forests burn every year (WRI, 2000). These fires are part of the natural seasonal cycle of growth, decay, and combustion and are ignited by lightning strikes. However, humans have long played a significant role in modifying fire regimes by changing the season and frequency of burning and consequently vegetation composition and structure (Goldammer and Price, 1998). During the 1990s in tropical humid areas, major fires have occurred in the Brazilian Amazon, Mexico, and Indonesia (Kalimantan and Sumatra) and were particularly severe in 1997–1998 during an El Niño episode (FAO, 1997a; Nepstad *et al.*, 1999). Fire also is a serious threat to native forests in many parts of tropical and nontropical developing countries. China, for example, suffered large losses in a single fire event in 1987, with 1 Mha (Anon., 1987) to 1.3 Mha (Cahoon *et al.*, 1994) burned. Fire prevention and suppression capability depends on available infrastructure such as imagery, roads, machinery, and human capital. In general, developed countries are better able to manage fire in regions with roads; developing countries may lack one or all of the factors.

The Indonesian fires of 1997–1998 were associated with a significant, but not unique, drought over much of the region. Estimates of the area burned vary from 96,000 ha to more than 8 Mha (Harwell, 2000). Most of this area was not forest but scrub, grassland, and agricultural lands. Almost no fires occurred deep within undisturbed primary forest; most were associated with land-clearing for new settlements or plantations or with logging operations. The most persistent fires were seven clusters of fires along the edges of degraded peat-swamp forests in southern Sumatera and Kalimantan (Legg and Laumonier, 1999). The extent and persistence of these fires, and similar fires in Brazil, show the importance of interaction between climate and human actions in determining the structure and composition of tropical forests, land-use patterns, and carbon emissions.

In the boreal zone of Canada, there has been a marked increase in fire since about 1970, after a 5-decade decrease (Kurz *et al.*, 1995). The area of boreal forest burned annually in western North America has doubled in the past 20 years (0.28% in the 1970s to 0.57% in the 1990s), in parallel with the observed warming trend in the region (Kasischke *et al.*, 1999), despite much improved detection and suppression technology. Similar trends have been noted for Eurasian forests (Shvidenko and Nilsson, 1994, 1997; Kasischke *et al.*, 1999). Whether these changes are the result of human-induced climate change or are a result of natural climatic variability is not certain (Clark *et al.*, 1996; Flannigan *et al.*, 1998). Changes in the disturbance regime over periods of decades result in changes in forest age-class distribution to younger versus old forests (Kurz and Apps, 1999). These changes will reduce the landscape-averaged biomass stock and dead organic matter pools, including soils (Bhatti *et al.*, 2000). Hence, changes in NBPoccur on scales of years to decades.

Fire frequency is expected to increase with human-induced climate change, especially where precipitation remains the same or is reduced (Stocks *et al.*, 1998). A general but moderate increase in precipitation, together with increased productivity, also could favor generation of more flammable fine fuels. Miranda (1994) suggests an increase in risk, severity, and frequency of forest fires in Europe. Stocks *et al.* (1998) used four GCMs and found similar predictions of an earlier start to the fire season and significant increases in the area experiencing high to extreme fire danger in Canada and Russia. Some regions may experience little change or even decreases in fire frequency, where precipitation increases or temperature does not rise (as in eastern Canada, where regional cooling has led to decreased fire frequency—Flannigan *et al.*, 1998). In most regions, there is likely to be an increased risk of forest fires, resulting in a change in vegetation structure that in turn exacerbates this risk (Cramer and Steffen, 1997).

During the past decade, forest fires in developed countries generally have become smaller, with the exception of the former Soviet Union (FAO, 1997a), Canada (Kurz *et al.*, 1995; Kurz and Apps, 1999), and the United States (Sampson and DeCoster, 1998). Where observed, the slight decline in forest areas burned per year is believed to be in part a result of improved prevention, detection, and control of fires. However, many such protected forests have increasing fuel loads and an abundance of dead and dying trees that eventually will make them more susceptible to catastrophic fires (e.g., Sampson and DeCoster, 1998). Several authors suggest that climate change is likely to increase the number of days with severe burning conditions, prolong the fire season, and increase lightning activity, all of which lead to probable increases in fire frequency and areas burned (Price and Rind, 1994; Goldammer and Price, 1998; Stocks *et al.*, 1998).

#### *5.6.2.2.2. Pressures from diseases and insect herbivory*

Insect outbreaks can be extremely important disturbance factors (Hall and Moody, 1994); during outbreaks, trees often are killed over vast areas (Hardy *et al.*, 1986; Candau *et al.*, 1998). Under climate change, damage patterns caused by insects may change considerably, particularly those of insects whose temporal and spatial distributions strongly depend on climatic factors. The ecological interactions are complex, however (see Box 5-10).

In temperate and tropical regions, insect and disease outbreaks are reported mostly for plantation forests; relatively less is known about native forests (FAO, 1997a). In the boreal zone, insect-induced mortality was a significant part of the changing disturbance regime for Canada in the period 1920–1995 (Kurz *et al.*, 1995). Insect mortality accounted for the loss of approximately 76 Mha in that period, with a near tripling of the average annual rate after 1970 (Kurz and Apps, 1999). Similar trends have been observed for Russian forests, where recent annual insect damage and disease mortality affecting as much as 4 Mha was reported by Shvidenko *et al.* (1995). In Siberian and Canadian forests, insect damage is estimated to be of the same magnitude as fire loss (Krankina *et al.*, 1994; Fleming and Volney, 1995; Kurz *et al.*, 1995; Shvidenko *et al.*, 1995; Shvidenko, 2000). Changes in drought conditions appear to play an important role in insect outbreaks (Volney, 1988; Sheingauz, 1989; Isaev, 1997).

There is likely to be an increase in declines and dieback syndromes (Manion, 1991) caused by changes in disease patterns, involving a variety of diseases. For example, in temperate and boreal regions, there may be increased incidence of canker diseases in poplars and other tree species. Some canker diseases increase in severity with decreased bark moisture content brought on by drought (Bloomberg, 1962). As another example, *Armillaria* root disease is found throughout the world and causes significant damage on all forested continents (Kile *et al.*, 1991), through mortality and growth loss. This disease—one of the largest threats to regeneration in the productive forest of the Pacific Northwest of North America—has surfaced as a result of past management practices (Filip, 1977) but may be exacerbated by changing climate. Under present conditions, *Armillaria* root disease causes losses of 2–3 million $m^3$ $yr^{-1}$ in the forests of Canada's Pacific Northwest (Morrison and Mallett, 1996). More recently, Mallet and Volney (1999) report a 43% decrease in annual volume increment and a 23% loss in annual height increment in lodgepole pine caused by *Armillaria* root disease. The incidence of *Armillaria* root disease can be expected to change under warmer or drier conditions. Significant damage has been observed in forests that have undergone drought stress (Wargo and Harrington, 1991). Morevoer, in regions such as the Pacific Northwest where the mean annual temperature presently is below the optimum

**Box 5-10. Complex Interactions: North America's Southern Boreal Forests, Pests, Birds, and Climate Change**

The eastern spruce budworm (*Choristoneura fumiferana*) is estimated to defoliate approximately 2.3 Mha in the United States (Haack and Byler, 1993) and affects 51 million $m^3$ of timber in Canada annually (Fleming and Volney, 1995). Although the budworm usually is present at low densities, budworm densities can reach 22 million larvae $ha^{-1}$ during periodic outbreaks (Crawford and Jennings, 1989). Outbreaks can extend over 72 Mha and last for 5–15 years, killing most trees in mature stands of balsam fir (Crawford and Jennings, 1989; Fleming and Volney, 1995).

Weather is thought to play a role in determining the budworm's range. Outbreaks frequently follow droughts (Mattson and Haack, 1987) or start after hot, dry summers (Fleming and Volney, 1995). Drought stresses host trees and changes plant microhabitats (Mattson and Haack, 1987). Moreover, the number of spruce budworm eggs laid at 25°C is 50% greater than the number laid at 15°C (Jardine, 1994). In some areas, drought and higher temperatures also shift the timing of reproduction in budworms so that they may no longer be affected by some of their natural parasitoid predators (Mattson and Haack, 1987). Weather, at least in central Canada, also may play a role in stopping some outbreaks if late spring frosts kill the tree's new growth on which the larvae feed.

Control of some populations of eastern spruce budworm may be strongly aided by bird predators, especially some of the wood warblers (Crawford and Jennings, 1989; but see Royama, 1984). Birds can consume as much as 84% of budworm larvae and pupae when budworm populations are low (approximately 100,000 $ha^{-1}$), but once larvae populations exceed 1,000,000 $ha^{-1}$, bird predation is unable to substantially effect budworm populations. This predatory action of birds works in concert with those of other predators, mostly insects.

The spruce budworm's northern range may shift northward with increasing temperatures—which, if accompanied by increased drought frequency, could lead to outbreaks of increasing frequency and severity that lead to dramatic ecological changes (Fleming and Volney, 1995). Increasing temperatures also might reduce the frequency of late spring frosts in southern boreal forests, perhaps increasing outbreak duration in some of those areas.

A changing climate also might decouple some budworm populations from those of their parasitoid and avian predators (Mattson and Haack, 1987; Price, 2000). Distributions of many of the warblers that feed on spruce budworms could shift poleward, perhaps becoming extirpated from latitudes below 50°N (Price, 2000). Replacing biological control mechanisms with chemical control mechanisms (e.g., pesticides) ultimately may yield a different set of problems; there are economic and social issues relating to large-scale pesticide application (see Section 5.4.4.3).

(25°C) for *Armillaria* growth, a warmer climate is likely to result in increased root disease and rate of spread.

##### *5.6.2.2.3. Pressures from other disturbances*

Additional disturbances are associated with extreme weather events such as hurricanes, tornadoes, unexpected drought or heavy rainfall, flooding, and icestorms that lead to extensive mortality and ecosystem change (e.g., Lugo *et al.*, 1990; Walker and Waide, 1991). Such events generally are highly localized and take place in a relatively short period of time but have long-term economic impacts (Haight *et al.*, 1995) and effects on ecosystems (Pontailler *et al.*, 1997). There is some evidence of recent increases in damage from such extreme events (Berz, 1999; see also Chapters 8 and 9).

#### *5.6.2.3. Changing Demand for Forest Goods and Services*

Future demand for industrial wood products depends on income growth, population growth, technological change, growth in human capital, changes in tastes and preferences, and institutional and political change (Solberg *et al.*, 1996). Changes in other markets also can influence demand for wood products. For example, increases in the price of substitutes, such as steel and concrete building materials, would increase the demand for industrial timber. In light of these driving factors, recent timber market assessments have predicted that industrial harvests will increase by 1–2% $yr^{-1}$ (Solberg *et al.*, 1996; Brooks, 1997; FAO, 1997b; Sohngen *et al.*, 1999). These results contrast with those in the SAR (Solomon *et al.*, 1996), which concluded that global demand for industrial fiber would exceed global supply in the next century.

There is some debate about which forests are likely to be harvested in the future. Some authors contend that most supply will come from new industrial plantations, secondary growth forests, and enhanced management, rather than from native forests (FAO, 1997b; Sohngen *et al.*, 1999). The proportion of global timber from subtropical plantations (presently 10%) may increase to 40% by 2050 (Sohngen *et al.*, 1999). Non-native species, such as eucalypts and pines, are favored in these regions because the costs of management and harvesting are low compared to those in temperate and boreal forests (Sedjo and Lyon, 1990; Sedjo, 1999). Recent estimates use global timber market models that incorporate management responses to prices across a wide range of forests. Under most price scenarios, subtropical plantations with 5- to 20-year rotations are a financially attractive alternative (Sedjo, 1999). However, higher demand still may put pressures on native forests even if plantation establishment and forest management responds to price increases (Solberg *et al.*, 1996; Brooks, 1997).

Policies that raise prices for substitute products, such as non-wood building materials made from steel or plastics, may increase timber demand, increase non-native plantation establishment, and cause additional pressures on native forests. Substituting non-wood products for wood products could increase carbon emissions as well (Marland and Schlamadinger, 1995; Schlamadinger *et al.*, 1997).

Increased reliance on plantations may have positive and negative ancillary consequences. For example, most of the plantations established for industrial purposes involve nonindigenous species, and the environmental effects of these plantations are not fully evident. However, most plantations have been established on former agricultural lands, which begins the process of restoring forests (Lugo *et al.*, 1993). Furthermore, plantations may reduce harvest pressures on natural forests. Despite increased reliance on plantations, however, industrial harvests in native forests along accessible roadways are likely to continue (Johns *et al.*, 1996).

The relationship between income and fuelwood demand is nonlinear. As incomes rise and infrastructure grows, households substitute alternative fuel sources (e.g., natural gas, fuel oil). Brooks *et al.* (1996), for example, suggest that fuelwood harvests will increase by 17% by 2050 under a low-GDP-growth scenario, but only by 4% under a high-growth scenario. Fuelwood harvest depends on the extent of substitution by alternative methods of heating and cooking. Currently, use of wood for energy on a large scale does not appear to be cost-effective relative to other energy sources, but if future energy prices rise, the demand for wood as a source of energy could rise.

#### *5.6.2.4. Land-Use Change*

Land-use change, including deforestation, is still considered to be a large pressure on global forests (Alcamo *et al.*, 1998). Several factors contribute to deforestation in tropical regions, including income and population growth, road-building policies, and other government policies (Southgate, 1998). Conversion to agriculture is impractical in boreal systems because of low productivity and high access costs, but some boreal forests continue to be converted to second-growth, managed forests. In temperate regions, conversion of agricultural lands back to forest has increased with agricultural productivity and falling prices (Kuusela, 1992; UN-ECE/FAO, 2000a). There is some evidence that these trends will continue, although at a lower rate (UN-ECE/FAO, 2000a). Factors affecting these trends include urbanization, agricultural yields and prices, timber prices, access and conversion costs, and subsidy programs (such as those that promote afforestation for environmental reasons, including mitigation of climate change impacts).

#### *5.6.2.5. Other Pressures*

*Fragmentation* of forest landscapes as a result of climate change, land-use practices, and disturbance is expected to take place in advance of larger scale biome shifts (Fahrig and Merriam, 1994; Shriner *et al.*, 1998). Fragmentation can change biodiversity and resiliency (Sala *et al.*, 2000). Fragmentation of the landscape can occur as a result of disturbance (natural or anthropogenic) or more gradually from successional responses to environmental changes (NBIOME SSC, 1992).

*Pressures from air pollution and air quality:* There is evidence of decline in forest condition as a result of air pollution, especially in areas adjacent to industrial areas and large cities (e.g., deposition of heavy metals, sulfur, nitrogen, and ozone) (Nilsson *et al.*, 1998). Of major concern is increased nitrogen deposition caused by industrial processes and agriculture (Vitousek *et al.*, 1997b). Nitrogen deposition is higher in northern Europe than elsewhere (Vitousek *et al.*, 1997b). Low-level increases in nitrogen deposition associated with air pollution have been implicated in productivity increases over large regions (Schindler and Bayley, 1993; Vitousek *et al.*, 1997b). Temperate and boreal forests, which historically have been nitrogen-limited, appear to be most affected (Townsend and Rastetter, 1996; Vitousek *et al.*, 1997b). In other areas that become nitrogen-saturated, other nutrients are leached from the soil, resulting in forest dieback (Vitousek *et al.*, 1997b)—counteracting, or even overwhelming, any growth-enhancing effects of $CO_2$ enrichment.

Tropospheric ozone has been shown to impact the structure and productivity of forest ecosystems throughout industrialized countries (Chameides *et al.*, 1994; Weber *et al.*, 1994; Grulke *et al.*, 1998) and is likely to increase in extent with further industrial development and agriculture management (Chameides *et al.*, 1994). It has been suggested that the impact of ozone damage is reduced, but not eliminated, by increasing $CO_2$ (Tingey *et al.*, 2001).

In developed countries, the major impacts of air pollution on forest services are likely to be on recreation and non-wood products. Air pollution has been shown not to have significant impacts on industrial timber markets in the United States (Haynes and Kaiser, 1990), although European timber market studies suggest potentially larger local effects (Nilsson *et al.*, 1992). Although increasing industrialization in developing countries that have less restrictive air pollution requirements could have effects on local industrial or fuelwood markets, these increases are not expected to have major effects on global timber supply.

### 5.6.3. Responses by Forests and Woodlands and Impacts on their Goods and Services

Assessment of responses and impacts must distinguish between transient and equilibrium situations. Because forests are composed of long-lived organisms, responses to climate change and resulting impacts may take a long time to propagate through the system. To the extent that global change (climate and land-use change) proceeds faster than the life cycle of many late-succession trees, transient responses will predominate. Forest structure today is the result of activities and events that occurred many years (>100) ago; hence, responses and parameters measured today are not in equilibrium with present conditions. Many state variables, such as carbon pools, are expected to change with a time delay. Furthermore, forest responses to climate change and resulting impacts may extend longer than the change in climate.

#### 5.6.3.1. Responses and Impacts: Carbon Storage

There are numerous ways in which forests respond to climate change and other pressures. These responses affect their ability to store carbon. Responses include changes in species distribution, NPP, NEP, and NBP; pests and disease outbreaks; and elevated $CO_2$, as well as changes in climate variability and weather extremes.

##### 5.6.3.1.1. Forest and species distribution

Models that predict changes in species distribution suggest reduced forest carbon storage as climate changes (King and Neilson, 1992; Smith and Shugart, 1993; Kirilenko and Solomon, 1998; Woodward *et al.*, 1998), although the change in forest carbon stocks depends on species migration rates (Solomon and Kirilenko, 1997; see also Section 5.2). Where seed availability and dispersal are impeded (e.g., by fragmented landscapes), achieved/realized productivity may remain below the potential for some time (resulting in carbon losses) unless aided by human intervention (Iverson and Prasad, 1998; Sohngen *et al.*, 1998; Iverson *et al.*, 1999). Pitelka and Plant Migration Workshop Group (1997) point out, however, that increases in weed species that take advantage of human mobility may be an adverse effect.

Changes in forest distribution as a result of climate change are likey to occur subtly and nonlinearly (Davis and Botkin, 1985; Prentice *et al.*, 1993; Neilson and Marks, 1994; Tchbekova *et al.*, 1994; Bugmann *et al.*, 1996; Neilson and Drapek, 1998). Prediction of changes in species distribution is complicated by the current lack of precise predictions of environmental changes themselves (especially precipitation) and responses of species to these changes. Price *et al.* (1999b) show that responses to precipitation are greatly dependent on assumptions made about species parameters and the temporal pattern of rainfall. The most rapid changes are expected where they are accelerated by changes in natural and anthropogenic disturbance patterns (Overpeck *et al.*, 1991; Kurz *et al.*, 1995; Flannigan and Bergeron, 1998).

At the stand level, climate-induced changes in competitive relationships are likely to lead to dieback and replacement of maladapted species, causing changes in stand population and productivity (e.g., Cumming and Burton, 1996; Rehfeldt *et al.*, 1999b). In addition, increases in locally extreme events and disturbances (fires, insects, diseases and other pathogens) developing over different time scales (Wein *et al.*, 1989; Campbell and McAndrews, 1993; Campbell and Flannigan, 2000) also lead to regionally specific increases in mortality and dieback. Resulting changes in age-class distributions and productivity are likely to have short- and long-term impacts on carbon stocks (Kurz *et al.*, 1995; Fleming, 1996; Hogg, 1997, 1999; Fleming and Candau, 1998).

##### 5.6.3.1.2. NPP and NEP

Some modeled responses of forest ecosystems to climate change suggest that forests could increase carbon storage during climate change (Xiao *et al.*, 1997; Prinn *et al.*, 1999). However, recent work has provided new experimental evidence on the response of vegetation uptake (NPP) and ecosystem losses (affecting both NEP and NBP) to observed changes in climate over the past century. The net balance between NPP(usually assumed to increase with warming, though this assumption is challenged by data offered by Barber *et al.,* 2000), heterotrophic respiration (often assumed to increase with warming, though this is challenged, for example, by Giardina and Ryan, 2000, and Liski *et al.*, 1999—at least for mineral soil components) and disturbance releases (often ignored but shown—for example, by Kurz and Apps, 1999—to be important in boreal estimates of NBP) is no longer as clear as the SAR asserts. Present research continues to improve scaling of localized responses (at the stand level, where increases and decreases in NPP and NEP are observed) to the global scale.

Research reported since the SAR confirms the view that the largest and earliest impacts induced by climate change are likely to occur in boreal forests, where changes in weather-related disturbance regimes and nutrient cycling are primary controls on productivity (Kasischke *et al.*, 1995; Kurz *et al.*, 1995; Yarie, 1999). The impacts are exacerbated by characteristic ecosystem time constants (rotation length, mean residence time of SOM, etc.) that are long compared to other forest ecosystems. In boreal forests, recent warming has been shown to change seasonal thaw patterns (Goulden *et al.*, 1998; Osterkamp and Romanovsky, 1999); increase growing season length (Keeling *et al.*, 1996; Myneni *et al.*, 1997); and, if accompanied by summer drought, reduce NPP (Sellers *et al.*, 1997; Barber *et al.*, 2000). These trends are expected to continue on average (see Table 3-10), although short-term modulations will occur (e.g., in association with ENSO events) (Black *et al.*, 2000).

The SAR (Kirschbaum *et al.*, 1996) concludes that in lowland humid tropics, temperatures already are close to optimum ranges for year-round growth. Hence, an increase in temperature as a result of climate change is likely to have a marginal effect on forest processes. However, research in a lowland tropical forest in Costa Rica has shown that the annual growth in six major species of this forest (with markedly different growth rates and life histories), over a period of 13 years, was highly negatively correlated with annual mean minimum (nighttime) temperatures (Clark and Clark, 1999). Although annual tree growth varied among the six species, there was a highly significant interannual coherence of growth among species.

#### 5.6.3.1.3. *Insect herbivory, pests, and diseases*

Some evidence suggests that insect populations already are responding to climate change (Fleming and Tatchell, 1995). In general, current forecasts of the response of forest insects and other pathogens to climate change are based on historical relationships between outbreak patterns and climate. These forecasts suggest more frequent or longer outbreaks (Thomson and Shrimpton, 1984; Thomson *et al.*, 1984; Mattson and Haack, 1987; Volney and McCullough, 1994; Carroll *et al.*, 1995; Cerezke and Volney, 1995; Brasier, 1996; Roland *et al.*, 1998). Outbreaks also may involve range shifts northward, poleward, or to higher elevations (Williams and Liebhold, 1997). All of these responses will tend to reduce forest productivity and carbon stocks, although the quantitative extent of these changes is hard to predict (see Box 5-10).

#### 5.6.3.1.4. *Elevated $CO_2$*

At the time of the SAR, no experiments on intact forest ecosystems exposed to elevated $CO_2$ had been performed. Since then, several FACE experiments have been implemented and are beginning to show interesting results. In a 13-year-old loblolly pine plantation (North Carolina), $CO_2$ levels have been maintained at 200 ppm above ambient. After 2 years, the growth rate of the dominant trees increased by about 26% relative to trees under ambient conditions (DeLucia *et al.*, 1999). Litterfall and fine root increment also increased under the $CO_2$-enriched conditions. Total NPP increased by 25%. The study concludes, however, that stimulation is expected to saturate not only because each forest stand tends toward its maximum carrying capacity (limited by nutrient capital) but also because plants may become acclimated to increased $CO_2$ levels.

Research on $CO_2$ fertilization, however, has taken place only over a short fraction of the forest ecosystem's life cycle. Questions of saturation of response can be addressed through longer term experiments on tree species grown under elevated $CO_2$ in open-top chambers under field conditions over several growing seasons (Norby *et al.*, 1999). A review of such experiments by Norby *et al.* (1999) found that the evidence shows continued and consistent stimulation of photosynthesis, with little evidence of long-term loss of sensitivity to $CO_2$; the relative effect on aboveground dry mass was highly variable but greater than indicated by seedling studies, and the annual increase in wood mass per unit of leaf area increased. Norby *et al.* (1999) also found that leaf nitrogen concentrations were lower in $CO_2$-enriched trees, but not as low as seedling studies indicated, and the leaf litter C/N ratio did not increase. In the majority of $CO_2$ chamber experiments, the decrease in the percentage of nitrogen in litter was matched by an increase in the percentage of lignin. Moore *et al.* (1999) have suggested, however, that lower litter quality caused by $CO_2$ fertilization may have offset the expected temperature-induced increase in decomposition. A longer term perspective still is needed because long-term trends cannot be extrapolated directly from relatively short-term experiments on individual trees (Idso, 1999). Field experiments on elevated $CO_2$ provide inconclusive evidence at this time to predict overall changes in carbon storage in forests.

#### 5.6.3.1.5. *Climate-induced changes in variability and weather extremes*

The net rate of carbon storage in forests varies as a result of interannual variability in rainfall, temperature, and disturbance regimes. For example, transient simulations with the Terrestrial Ecosystem Model (TEM) suggest that forests of the Amazon basin during hot, dry El Niño years were a net source of carbon (of as much as 0.2 Gt C $yr^{-1}$), whereas in other years they were a net sink (as much as 0.7 Gt C $yr^{-1}$) (Tian *et al.*, 1998). Notably, source and sink strength varied across the basin, indicating regional variation. Similarly for the United States, the net rate of carbon storage varied from a net source of about 0.1 Gt C $yr^{-1}$ to a net sink of 0.2 Gt C $yr^{-1}$ (Schimel *et al.*, 2000). The rate of carbon accumulation in undisturbed forests of the Amazon basin reported by Tian *et al.* (1998) was approximately equal to the annual source from deforestation in the same area. For the United States, Houghton *et al.* (1999)—using inventory data—estimate a sink of 0.35 Gt $yr^{-1}$ during the 1980s as a result of forest management and regrowth on abandoned agricultural lands, whereas Schimel *et al.* (2000) indicate that 0.08 Gt $yr^{-1}$ stored from 1980 to 1993 could have been a result of carbon fertilization and climate effects. Thus, for the United States, the rate of carbon accumulation in forest regrowth on abandoned agricultural and harvested managed forest lands appears to be as large as or larger than the direct effects of $CO_2$ and climate (Schimel *et al.*, 2000). The SAR (Kirschbaum *et al.*, 1996) suggests that modeled trends described above are unlikely to continue under projected climate change and high elevated $CO_2$ concentrations. However, climatic variability (including ENSO events) is likely to increase under a changed climate (see Table 3-10), which may increase interannual variation in regional carbon uptake—as demonstrated by Tian *et al.* (1998) and Schimel *et al.* (2000).

### 5.6.3.2. *Responses and Impacts: Timber and Non-Wood Goods and Services*

#### 5.6.3.2.1. *Response to locally extreme events*

In addition to fire and insect predation, other episodic losses may become increasingly important in response to locally extreme weather events. For example, in Europe, wind-throw damage has appeared to increase steadily from negligible values prior to about 1950; wind-throws exceeding 20 million $m^3$ have occurred 10 times since then (UN-ECE/FAO, 2000b). Losses in 1990 and 1999 were estimated at 120 million and 193 million $m^3$, respectively—the latter equivalent to 2 years of harvest and the result of just three storms over a period of 3 days (UN-ECE/FAO, 2000b). The unusual 1998 icestorms in eastern North America caused heavy damage to infrastructure as well as large areas of forest, the extent of which is still being assessed (Irland, 1998).

Financial returns to forest landowners decline if these episodic events (including fire and insect predation) increase (Haight *et al.*, 1995), although the impact on global timber supply of such episodic losses is unlikely to be significant. Local effects on timber and non-wood goods and services may be significant, although timber loss may be ameliorated through salvage logging. Moreover, the location of the loss is particularly important for non-wood products and services. For example, forest fires in recreational areas are known to have an impact (Englin *et al.*, 1996).

#### *5.6.3.2.2. Industrial timber*

Recent industrial timber studies link equilibrium ecological models to economic models to measure market impacts to potential climate change and include adaptation (Joyce *et al.*, 1995; Perez-Garcia *et al.*, 1997; Sohngen and Mendelsohn, 1998; McCarl *et al.*, 2000; Sohngen *et al.*, 2000). Conclusions from regional studies are similar to those in the SAR (Solomon *et al.*, 1996) for temperate regions, suggesting that wood supply in these regions will not be reduced by climate change. Studies in the United States that consider only changes in forest growth find small negative or positive impacts on timber supply (Joyce *et al.*, 1995; McCarl *et al.*, 2000). Studies that consider growth effects and species redistribution effects (dieback of existing forests, followed by redistribution), as well as alternative economic scenarios, find that welfare economic impacts in U.S. timber markets could change by –1 to +11% (Sohngen and Mendelsohn, 1998). Generally, consumers are predicted to benefit from increased supply and lower prices. Producers in some regions of the United States may lose because of lower prices and dieback, although productivity gains could offset lower prices (Sohngen, 2001).

In contrast to the SAR (Solomon *et al.*, 1996), more recent global market studies suggest that climate change is likely to increase global timber supply and enhance existing market trends toward rising market share in developing countries (Perez-Garcia *et al.*, 1997; Sohngen *et al.*, 2000). One study that uses TEM (Melillo *et al.*, 1993) to predict growth changes finds that global timber growth rises, global timber supply increases, prices fall, and consumers and mill owners benefit (Perez-Garcia *et al.*, 1997). Landowners in regions where increased timber growth does not offset lower prices perceive losses. A study using the BIOME3 model (Haxeltine and Prentice, 1996) suggests that producers in temperate and boreal forests could be susceptible to economic losses from short-term dieback effects and lower prices, although long-term (>50 years) supply from these regions is predicted to increase (Sohngen *et al.*, 2000). Alternatively, studies that do not consider global market forces, timber prices, or adaptation predict that supply in boreal regions is likely to decline (Solomon and Leemans, 1997).

#### *5.6.3.2.3. Recreation and non-wood forest products*

There is considerably less published literature available to assess the effects of climate change on nonmarket services from forests (Wall, 1998), including recreation and non-wood forest products. Climate change is likely to have direct effects on forest-based recreation. For example, lengthening of the summer season may increase forest recreation (Wall, 1998). Changes in the mean and variance in daily temperature and precipitation during peak seasons will affect specific activities differently, however. In the United States, some studies suggest that higher temperatures may negatively affect camping, hiking, and skiing but positively affect fishing (Loomis and Crespi, 1999; Mendelsohn and Markowski, 1999; Joyce *et al.*, 2000).

Climate change also will have indirect effects on forest recreation. For example, changes in the structure and function of natural forests that are used for recreation could alter visitation patterns by causing users to substitute alternative sites. Alternatively, some recreational industries may have large adaptation costs—for example, snow-making costs in skiing areas may increase (Irland *et al*., 2001). In addition, institutional factors are likely to play a strong role in mediating the response of recreation because much of it occurs on public, natural forests where adaptation may be less economically feasible. Many forest-based activities, such as hunting, rely on management decisions by agencies that will have to adapt to climate change as well (Brotton and Wall, 1997).

## ***5.6.4. Adaptation Options and Vulnerability of Goods and Services***

### *5.6.4.1. Adaptation in Timber and Non-Wood Goods and Services*

In markets, prices mediate adaptation through land and product management. At the regional and global scales, the extent and nature of adaptation will depend primarily on prices, the relative value of substitutes, the cost of management, and technology (Joyce *et al*., 1995; Binkley *et al.*, 1997; Perez-Garcia *et al*., 1997; Skog and Nicholson, 1998; Sohngen and Mendelsohn, 1998). On specific sites, changes in forest growth and productivity will constrain, and could limit, choices of adaptation strategies (Lindner, 1999, 2000).

Forest management has a history of long-term decisions under uncertain future market and biological conditions (e.g., prices, pest infestations, or forest fires). Most adaptation in land management will occur in managed forests; it will include salvaging dead and dying timber, replanting new species that are better suited to the new climate, planting genetically modified species, intensifying or decreasing management, and other responses (e.g., Binkley, 1988; Joyce *et al.*, 1995; Perez-Garcia *et al.*, 1997; Sohngen and Mendelsohn, 1998; Lindner, 1999). Climate change is not likely to cause humans to convert highly productive agricultural land to forests (McCarl *et al.*, 2000), even with afforestation incentives (e.g., the Kyoto Protocol) (Alig *et al*., 1997). Adaptation estimates are sensitive to assumed rates of salvage and choices made in regenerating species during climate change (Sohngen and Mendelsohn, 1998; Lindner, 1999). In product management, adaptation includes substituting species in the production process for solid wood and pulpwood products, shifting harvests from one region to another, and developing new technologies and products, such as wood products manufactured with adhesives (McCarl *et al.*, 2000; Irland *et al.*, 2001). In some cases, producers may need to adapt to changes in wood quality (Gindl *et al.*, 2000).

In addition to adaptation through traditional management, agroforestry, small woodlot management, and windrows (or shelterbelts) could provide numerous adaptation options for maintaining tree cover and fuelwood supplies in developing countries. Afforestation in agroforestry projects designed to mitigate climate change may provide important initial steps towards adaptation (Sampson *et al*., 2000). Although agroforestry is not expected to play a large role in global industrial timber supplies, it may have important regional implications.

Recreation users can adapt by substituting recreational sites as forests respond to climate change, but this will have impacts on recreational industries. Substitution will be easier in regions where transportation networks are well established. Adaptation will be more difficult in regions where recreation depends on particular forest structures (e.g., old-growth forests) that are negatively affected by climate change and for which there are no close substitutes (Wall, 1998).

#### 5.6.4.2. *Vulnerability Associated with NPP, NEP, and NBP*

Given their inertia, forested systems may exhibit low vulnerability and low climate sensitivity, unless drought and disturbance are driving factors (Peterken and Mountford, 1996). Extant forests may persist and appear to exhibit low vulnerability and low climate sensitivity (see Section 5.2), but they may be climate sensitive in ways that are not immediately apparent. Thus, their vulnerability may occur as a reduction in quality (degradation) even where the forest persists as an entity. Increases in disturbances, however, may lead to rapid structural changes of forests, with replacement by weedy species (Overpeck *et al*., 1990). For example, an increase in area burned by fire or destroyed by invading insects and disease could rapidly undergo changes in species composition, successional dynamics, rates of nutrient cycling, and many other aspects of forest ecosystems—with impacts on goods and services provided by these forests. From this perspective, in the context of all of the direct and indirect impacts of climatic change and their interactions, the potential vulnerability of forests is high.

The main processes that determine the carbon balance of forests—photosynthesis, plant and heterotrophic respiration, and disturbance releases—are regulated by different environmental factors. Carbon assimilation is a function mainly of available light, temperature, nutrients, and $CO_2$, and this function shows a saturation characteristic. Respiration is a function of temperature; it increases exponentially with temperature. Thus, close correlation between GPP and NPP does not exist (allocation plays an important role). Similarly, there is no strict correlation between NPP and NEP (high NEP is possible at high and low rates of assimilation and respiration). Disturbances such as harvest or fire export carbon from forests, bypassing respiration; thus, NBP is not expected to correlate with NEPand NPP. Photosynthesis, plant and heterotrophic respiration, and disturbance are vulnerable in different ways and are quite sensitive to climate change and other global change forces.

#### 5.6.4.3. *Vulnerability of Unmanaged Systems*

Increases in disturbances such as insect infestations and fires can lead to rapid structural and functional changes in forests (species composition, successional dynamics, rates of nutrient cycles, etc.), with replacement by weedy species. The effects of these vulnerabilities in unmanaged systems on goods and services vary. They are not likely to have large effects on market products given that unmanaged forests constitute an increasingly small portion of timber harvests. They could have large impacts on local provision of timber products, NWFP, and fuelwood. Local services from forests could be highly vulnerable, particularly if the services are tied to specific forest functions that change (e.g., hiking in old-growth forests).

#### 5.6.4.4. *Vulnerability in Managed Systems*

Managed systems are vulnerable to direct impacts on NPP (related to volume production) and species composition (related to timber quality), as well as market forces. The global supply of market products exhibits low vulnerability because timber markets have high capacity to adapt to change. Temperate and boreal producers are vulnerable to dieback effects and lower prices caused by potential global increases in timber growth. Managed forests in subtropical regions have low vulnerability given their high growth rates and short rotation periods, which provide multiple opportunities to adapt to changes.

## 5.7. Lakes and Streams

Lake and stream ecosystems include many familiar places: large and small lakes, permanent and temporary ponds, and streams—from tiny, often temporary rivulets at headwaters to powerful floodplain rivers discharging from our continents.

Freshwaters (lakes and rivers), which are so valuable for the sustainability of life as we know it, constitute only 0.0091% of the Earth's surface waters by volume. This is a great deal less than 0.5% for groundwater and 97.3% for oceans (Cole, 1994). Lakes and rivers are used intensively for recreation and are aesthetically valued. These values include fishing, hunting, swimming, boating, skating, and simply enjoying the view.

This section focusses on physical and biological processes and how they affect goods and services from lakes and streams that are regulated by these processes. The emphasis is on food, carbon, and biodiversity. Hydrology and water supply are covered in Chapter 4; oceans and coastal systems are covered in Chapter 6. Hydrological goods and services are covered largely in Chapter 4, but are considered here as an interactive influence of climate change on inland waters. Wetlands are covered in Section 5.8.

### *5.7.1. Status of Goods and Services*

Products from lakes and streams include fisheries (fish, amphibians, crustaceans, and mollusks) and aquaculture; services include biodiversity, recreation, aesthetics, and biogeochemical cycling. A detailed list of services from freshwater ecosystems includes many items that are undervalued in economic terms and often are unrecognized and unappreciated—for example, their role in the carbon cycle.

Reported catches in freshwater and inland fisheries were 7.7 Mt of biomass in 1996 (FAO, 1999b) and 15.1 Mt from aquaculture. China accounts for 80% of aquaculture production. Actual landings in capture fisheries are believed to be two to three times larger owing to nonreporting (FAO, 1999c). About 100 fish species are reported in world catches—primarily cyprinids, cichlids, snakeheads, catfish, and barbs. Asia and China report the largest catches; Africa is second; North America ranks relatively low. River and large reservoir fisheries are important.

Recreational fish catches are included in the foregoing world catches and were reported to be 0.48 Mt in 1990 (FAO, 1992). This is likely to be an underestimate because only 30 (of 200) countries reported recreational catches. Recreational fishing is increasing in developed and developing countries. The number of anglers is estimated at 21.3 million in 22 European countries, 29.7 million in the United States, and 4.2 million in Canada (U.S. Fish and Wildlife Service, 1996; Department of Fisheries and Oceans, Canada, 1998). Total expenditures for recreational angling are in the billions of dollars—for example, $38 billion in the United States in 1996 (U.S. Fish and Wildlife Service, 1996) and $4.9 billion in Canada in 1995.

Freshwaters are known for high biodiversity and endemism owing to their island-like nature, which leads to speciation and reduces invasions of competitors and predators. Of about 28,000 fish species known on Earth, 41% are freshwater species and 58% are marine, of which 1% spend part of their lives in freshwater (Moyle and Cech, 1996). Individual east African Rift Valley lakes contain species flocks of almost 250 cichlid species. In Lake Baikal in Russia, 35% of the plants and 65% of the animals are endemic (Burgis and Morris, 1987). At the global level, biodiversity in many lakes has been decreasing in recent decades, with many species becoming extinct (Naiman *et al*., 1995b). The trend is likely to continue, with many species that now are listed as endangered or threatened becoming extinct (IUCN, 1996). The causes for these extinctions are likely to be related to the many pressures listed below.

Inland waters play a major role in biogeochemical cycling of elements and compounds such as carbon, sulfur, nitrogen, phosphorous, silica, calcium, and toxic substances. The general roles are storage, transformation, and transport (Stumm and Morgan, 1996). Storage is important because sediments and associated minerals accumulate in the bottom sediments of lakes, reservoirs, and floodplains. Transformation includes organic waste purification and detoxification of various human created compounds such as insecticides. Water movement redistributes these spatially.

Of special interest here is the role of freshwaters in carbon storage and $CO_2$ and methane ($CH_4$) release. Organic carbon from primary production in lakes and adjacent riparian lands accumulates in sediments; estimates are that 319 Mt $yr^{-1}$ are buried in the 3.03 million $km^2$ of small and large lakes, reservoirs, and inland seas worldwide (USGS, 1999; see also Stallard, 1998). This estimate excludes amounts for peatlands (96 Mt $yr^{-1}$). It is three times greater than estimates for the ocean in absolute terms (97 Mt $yr^{-1}$) even given the relatively small area of inland waters. Lakes also are a source of GHGs. Lakes become supersaturated with dissolved $CO_2$, and net gas exchange is from the lake to the atmosphere (Cole *et al*., 1994). For example, during summer in Lake Pääjärv, Finland, the amount of carbon from respiration in the water column was greater than that produced by phytoplankton and sedimentation combined (Kankaala *et al.*, 1996). Pulses of $CO_2$ from the water column and $CH_4$ from sediments are released to the atmosphere during spring and fall mixing of the water column of dimictic lakes (Kratz *et al.*, 1987; Riera *et al.*, 1999; Kortelainen *et al.*, 2001). Methane releases, especially from the littoral zone, can be significant in lakes and reservoirs (Fearnside, 1995, 1997; Alm *et al.*, 1997a; Hyvönen *et al.*, 1998).

Hydroelectric power plants generally are assumed to emit less $CO_2$ than fossil fuel plants. However, a hydroelectric reservoir may contribute more to GHGs over 100 years of operation than a fossil fuel plant that produces an equivalent amount of electricity (Fearnside, 1997). Emissions are likely to be high in the first few decades and then decrease. This is exemplified by a Brazilian hydroelectric reservoir that is simulated to release a large quantity of $CO_2$ during the first 10 years after filling (*ca.* 5–27 Mt $CO_2$ gas $yr^{-1}$) but relatively low amounts from years 30 to 100. Releases of $CH_4$ were simulated to be high for at least 100 years (about 0.05–0.1 Mt $CH_4$ $yr^{-1}$, with actual estimates for 1990 of 0.09 Mt) (Fearnside, 1995, 1997).

### *5.7.2. Pressures on Goods and Services*

People expect drinkable, swimmable, boatable, and fishable freshwaters. However, rivers and lakes have many pressures. These pressures include land and water use for urbanization, agriculture, and aquaculture; hydrologic engineering structures such as dams, dykes, channelization, and construction of drainage and connecting canals; water extractions for industry, drinking water, irrigation, and power production; water pollution with toxics, excess nutrients, and suspended sediments; capture fisheries; and invasion of exotics. UV-B is an interactive pressure with climate change owing to the change in water clarity influenced by drought. Under these conditions, UV-B penetrates farther and causes more damage in clear waters than in murky waters.

Human demand for water in many areas will increase more rapidly with climate change and population increases and leave fewer waters unmodified by water projects (see Chapter 4). Water projects interact with many aspects of climate change as related to natural resource and environmental management. Additional dams will increase the difficulty of managing migratory fish populations in streams. Sedimentation that occurs above dams will reduce downstream transport of sand, sediments, and toxic substances. Lakes and reservoirs with increased water withdrawals will reduce the suitability of the littoral zone for fish spawning and nursery areas. Diversions of water by canals, ships, or pipes will transport exotics into new watersheds and confound biodiversity and exotics issues.

### *5.7.3 Responses of Lakes and Streams and Impacts on their Goods and Services*

Many responses of lakes and streams to climate change were documented in the SAR by Arnell *et al.* (1996), and more have been added by Cushing (1997) and Domoto *et al.* (2000). Responses include warming of waters; reductions in ice cover; reduction in dissolved oxygen in deep waters; changes in the interaction between waters and their watersheds; changes in biogeochemical cycling; greater frequencies of extreme events, including flood and drought; changes in growth, reproduction, and distribution of organisms; and poleward movement of climate zones for organisms. Only a few are mentioned in the following subsections.

#### *5.7.3.1. Physical Conditions*

#### *5.7.3.1.1. Ice cover*

To date, the only limnological properties measured or simulated at a global scale are lake and river ice phenologies. Ice-cover durations for inland waters have decreased over the entire northern hemisphere (Magnuson *et al.*, 2000) (see Figure 5-6). Change from 1846 to 1995 averaged 8.7 days later freeze and 9.8 days earlier break-up; these changes correspond to a 1.8°C increase in air temperature. Lake Suwa in Japan, Tornionoki River in Finland, and Angara River in Siberia have longer records; ice phenologies have been changing in the direction of warming since about the early 1700s, but at slower rates than during 1846–1995. Interannual variability in freeze dates, thaw dates, and ice duration are increasing (Kratz *et al.*, 2001). Interannual variability was greater from 1971 to 1990 than from 1951 to 1970 in 184 lakes in the northern hemisphere.

For the Baltic Sea, which contains freshwater communities near shore and to the north, simulation until 2050 reduced the extent of ice cover from 38% of the area to 10%, and simulation until 2100 reduced ice cover to zero (Haapala and Leppäranta, 1997). For U.S. lakes, ice-on date, ice-off date, ice duration, ice thickness, and the continuity of ice cover all responded to a $2xCO_2$ change (Fang and Stefan, 1998). Mean duration at 209 locations distributed evenly across the country declined by 45 days (60%), and ice thickness by 21 cm (62%). Simulated

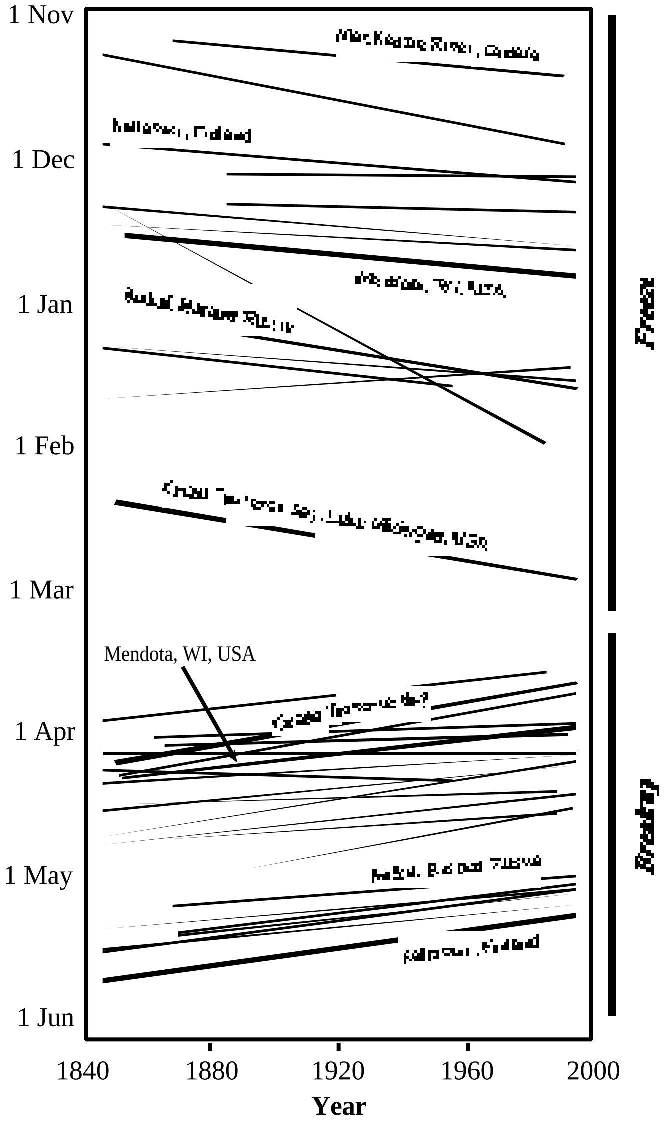

**Figure 5-6**: Historical trends in freeze and breakup dates of lakes and rivers in the Northern Hemisphere; 37 of the 39 trend slopes are in the direction of warming (modified from Magnuson *et al*., 2000).

profiles of deepwater oxygen concentrations and water temperatures also changed dramatically in lakes across the United States (Fang and Stefan, 1997).

Lake ice influences biogeochemical cycling, including gas exchange with the atmosphere, fish habitat availability (through changes in pH and dissolved oxygen), biodiversity, and seasonal succession (Arnell *et al.*, 1996; Cushing, 1997). Decreased ice cover should reduce winterkill of fish.

Ice break-up date influences the productivity of diatoms beneath the ice of Lake Baikal (Granin *et al.,* 2001). Seasonal succession of phytoplankton in small lakes is altered partly because of variation in the incoculum of algae beneath the ice in different years (Adrian and Hintze, 2001). The mechanism for enhanced diatom productivity in Lake Baikal is a combination of nutrient enrichment immediately beneath the ice owing to the extrusion of salts as the water freezes and vertical mixing that results from the greater density of the water with extruded salts immediately beneath the ice. Reductions in ice thickness and increases in snow would be expected to reduce diatom production.

River ice plays an important biological role by regulating flow aeration and oxygen concentrations under the ice (Prowse, 1994; Chambers *et al.*, 1997); dissolved oxygen can approach critical levels for river biota (e.g., Power *et al*., 1993). Decreases in the duration of the river-ice season or increases in the size and frequency of open-water sections where re-aeration can occur will reduce anoxia. Warmer winters favor the formation of mid-winter break-ups produced by rapid snowmelt runoff, particularly those initiated by rain-on-snow events. Such events significantly impact benthic invertebrate and fish populations where late-season break-ups are the norm (Cunjak *et al*., 1998).

Ecological impacts will be influenced most by changes in break-up timing and intensity. Physical disturbances from break-up scouring and flooding influence nutrient and organic matter dynamics, spring water chemistry, and the abundance and diversity of river biota (Scrimgeour *et al.*, 1994; Cunjak *et al.*, 1998). Ice-induced flooding supplies the flux of sediment, nutrients, and water that are essential to the health of freshwater delta ecosystems (e.g., Lesack *et al.*, 1991; Prowse and Conly, 1998). Even the mesoscale climate of delta ecosystems depends on the timing and severity of break-up flooding (Prowse and Gridley, 1993).

Climate-induced change in ice cover or the timing and severity of its break-up will affect the movement and deposition of sediment and associated contaminants. Stable cover leads to deposition of sediment particles that would remain in suspension without ice cover. Environmental contaminants with an affinity for adsorption to fine-grained sediments are more likely to be deposited during ice-covered, low-flow season than during open-water season. The break-up plume contains the winter-long accumulation of contaminated sediment (Milburn and Prowse, 1996). Sediment deposition zones often are where biological productivity is greatest, such as on river deltas (Milburn and Prowse, 1998). Because growth rates of aquatic organisms in cold regions are low and because higher trophic level animals are longer lived, potential exists for greater life-long accumulation of contaminants in these aquatic ecosystems than in more temperate regions.

River ice is a key agent of geomorphologic change; it is responsible for creating numerous erosional and depositional features within river channels and on channel floodplains (e.g., Prowse and Gridley, 1993; Prowse, 1994). During break-up, these processes cause channel enlargement and removal and succession of riparian vegetation. Climatic conditions that alter the severity of such events will greatly influence river morphology and riparian vegetation.

#### *5.7.3.1.2. Direct use of water*

Intensification in direct uses of water will negatively influence "natural" waters and could seriously reduce services provided by these ecosystems. Some values and services of lakes and rivers would be degraded by human responses to their greater demands for water for direct human use through dam construction, dyke and levee construction, water diversions, and wetland drainage (Postel and Carpenter, 1997). Such activities can alter water temperatures, recreation, pollution dilution, hydropower, transportation, and the timing and quantity of river flows; lower water levels; destroy the hydrologic connection between the river and the river floodplain; reduce natural flood control, nutrient and sediment transport, and delta replenishment; eliminate key components of aquatic environments; and block fish migration. At risk are aquatic habitat, biodiversity habitat, sport and commercial fisheries, waterfowl, natural water filtration, natural floodplain fertility, natural flood control, and maintenance of deltas and their economies (Ewel, 1997).

Water-level increases as well as decreases can have negative influences on lakes and streams. A recent example is from Lake Baikal, where the combined influences of a dam on the outlet and increased precipitation has increased water level by 1.5 m and decreased biodiversity and fishery production (Izrael *et al.*, 1997; ICRF, 1998).

### *5.7.3.2. Fisheries and Biodiversity*

Potential effects of climate change on fisheries are documented in the SAR by Everett *et al.* (1996); freshwater fisheries in small rivers and lakes are considered to be more sensitive to changes in temperature and precipitation than those in large lakes and rivers. Changes in temperature and precipitation resulting from climate change are likely to have direct impacts on freshwater fisheries through changes in abundance, distribution, and species composition. Because current exploitation rates tend to be high or excessive, any impact that concentrates fish will increase their catchability and further stress the population.

Assessing the potential impacts of climate change on aquaculture is uncertain, in part because the aquaculture industry is mobile and in a period of rapid expansion. There is no question that large abundances relative to traditional wild harvests can be produced in small areas and that in many cases these sites can be moved to more favorable locations. Changes in groundwater may be especially significant for aquaculture. In tropical areas, crustaceans are cultured in ponds (Thia-Eng and Paw, 1989); fish frequently are cultured in cages. Commonly cultured species such as carp and tilapia may grow faster at elevated temperatures, but more food is required and there is an increased risk of disease. Temperature is a key factor affecting growth, but other factors relating to water quality and food availability can be important. Modeling studies suggest that for every 1°C average increase, the rate of growth of channel catfish would increase by about 7%, and the most favorable areas for culture would move 240 km northward in North America (McCauley and Beitinger, 1992; see Figure 5-7, right panel). The southern boundary also would move northward as surface water temperatures increase, perhaps exceeding lethal ranges on occasion. Growth would increase from about 13 to 30°C and then fall off rapidly as the upper lethal limit of about 35°C is reached (McCauley and Beitinger, 1992). Thus, aquaculture for traditional species at a specific location may have to switch to warmer water species.

Warmer conditions are more suitable for warmer loving flora and fauna and less suitable for cold-loving flora and fauna. Warmer temperatures, however, lead to higher metabolic rates, and if productivity of prey species does not increase, reductions in growth would occur at warmer temperatures (Arnell *et al.*, 1996; Magnuson *et al.*, 1997; Rouse *et al.*, 1997). Rates of natural dispersal across land barriers of less mobile species poleward or to higher altitudes are not likely to keep up with rates of change in freshwater habitats. Species most affected would include fish and mollusks; in contrast, almost all aquatic insects have an aerial life history stage, thus are less likely to be restricted. Some streams have a limited extent and would facilitate only limited poleward or altitudinal dispersal. This could be especially problematic as impoundments restricting movements of organisms increase in number. Coldwater species and many coolwater species would be expected to be extirpated or go extinct in reaches where temperatures are at the warmer limits of a species range. Many lakes do not have surface water connections to adjacent waters, especially in headwater regions where interlake movement would be limited without human transport of organisms across watershed boundaries.

Exotics will become a more serious problem for lake and stream ecosystems with warming. In the northern hemisphere, for example, range extensions occur along the northern boundaries of species ranges and extinctions occur along the southern boundaries, in natural waters and in aquaculture operations (Arnell *et al.*, 1996; Magnuson *et al.*, 1997). In addition, loss of habitat for biodiversity will result from warmer, drier conditions interacting with increases in impoundment construction.

Distributions of fish are simulated to move poleward across North America and northern Europe. In northern Europe, Lehtonen (1996) forecasts a shrinking range for 11 coldwater species and an expanding range for 16 cool- and warmwater

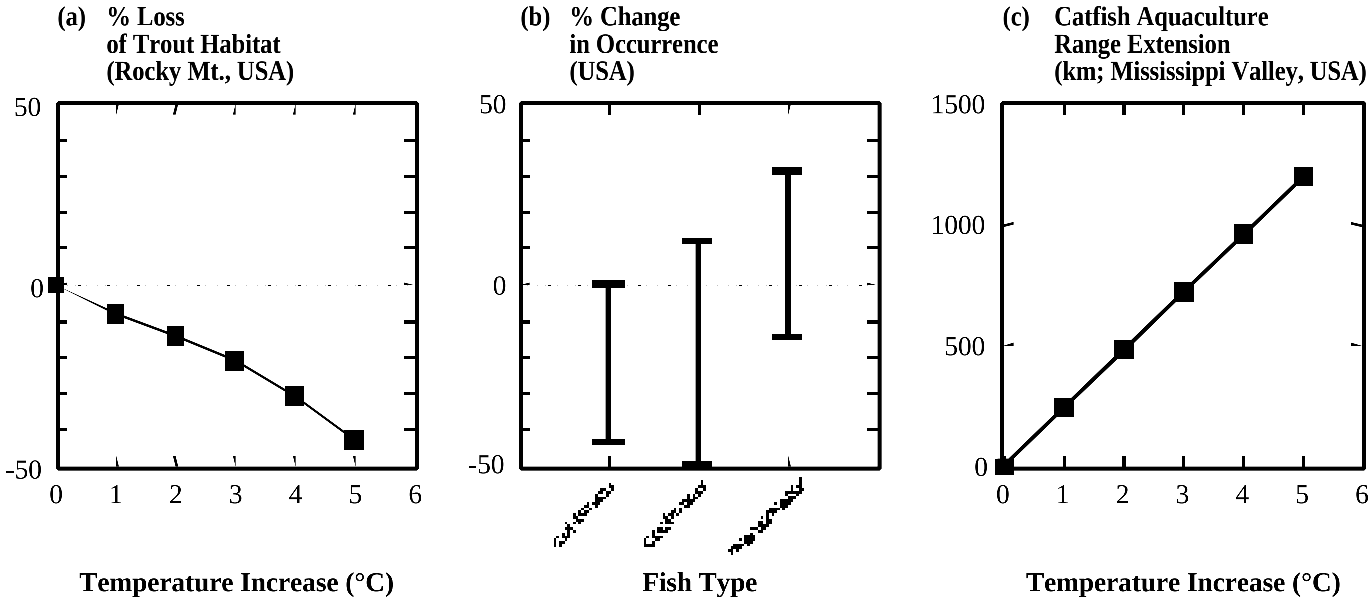

**Figure 5-7**: Simulated changes in thermal habitat for fish in the continental United States. Left panel is modified from Keleher and Rahel (1996). Center panel is modified from Mohseni and Stefan (2000) and Fang *et al.*(1998). Right panel is modified from McCauley and Beitinger (1992). Simulation is for a 2x$CO_2$ climate from the Canadian Climate Model and represents air temperature increases of 3–6.5ºC in different parts of the United States. Coldwater fish include trout and salmon; coolwater fish include yellow perch, walleye, northern pike, and white sucker; and warmwater fish include sunfish (black basses, bluegill, pumpkinseed) and common carp.

fish. In simulations for Finland by Lappalainen and Lehtonen (1995), lake whitefish lost habitats progressively toward the north, whereas brown trout did better, at least in the north. Coolwater fish were forecast to spread northward through the country. In simulation studies, boundaries of individual warmwater species ranges in were projected to move northward by 400–500 km in Ontario, Canada (Minns and Moore, 1995), and southern boundaries of coldwater fishes were projected to move 500–600 km northward in the southeastern United States (estimated from Fang *et al.,* 1998). In simulations based on an elevated $CO_2$ scenario using the Canadian Climate Centre model, warmwater fish were projected to benefit in shallow eutrophic and mesotrophic lakes around the United States, owing to reduction in winterkill, but habitable lakes and streams for coolwater fish and especially coldwater fish were projected to decline, owing to summer kill (Fang *et al.*, 1998; Mohseni and Stefan, 2000). Habitat changes from various studies over large regions (see Figure 5-7, center panel) projected 0–43% reductions for coldwater species, 50% reductions to 12% increases for coolwater species, and 14% reductions to 31% increases for warmwater fish. Changes differ among ecosystem types and areas, depending on latitude and altitude.

A dramatic picture of the regional decline in trout habitat in the Rocky Mountain region of the western United States is provided by Keleher and Rahel (1996) (see Figure 5-7, left panel). Even a 1°C increase in mean July air temperatures is simulated to decrease the length of streams inhabitable by salmonid fish by 8%; a 2°C increase causes a reduction of 14%, a 3°C increase causes a 21% decline, a 4°C increase causes a 31% reduction, and a 5°C increase causes a 43% reduction. There also is likely to be a increased fragmentation of inhabitable areas for the North Platte River Drainage in Wyoming (Rahel *et al.*, 1996).

### *5.7.3.3. Biogeochemical Cycling and Pollution*

Understanding of interactions of climate change with other human-caused pressures on lakes and streams is still in its infancy. No simulation models have been developed to assess the combined effects of these pressures on these systems.

#### *5.7.3.3.1. Eutrophication*

Nutrient cycling would be altered by climate change in ways that could exacerbate existing water quality problems such as eutrophication (see Figure 5-8). Eutrophication of lakes results when nutrient inputs from catchments and recycling from bottom sediments are large. The result is excessive production of algae; blue-green algae reduce water quality for recreation and drinking. Deep coldwater habitats become anoxic, owing to greater rates of decomposition of sinking organic mater (Horne and Goldman, 1994).

Interaction between climate change and eutrophication is complex, and projections are somewhat contradictory (see Figure 5-8) because climate-influenced processes have interacting and often opposing effects (Magnuson *et al.*, 1997; Schindler, 1997). Consider, for example, phosphate release from anoxic sediments. In a warmer climate, the longer period of summer stratification would increase the likelihood that anoxia develops below the thermocline (Stefan and Fang, 1993); this would increase the solubility of phosphates in sediment and increase nutrient recycling. At the same time, warmer climates would reduce the duration of ice cover in lakes, which would reduce winter anoxia and decrease sediment phosphate release in winter. This is further complicated by water column stability.

In Lake Mendota, Wisconsin (Lathrop *et al.*, 1999), it is not surprising that one-third of observed year-to-year variation in summer water clarity is associated with variability in runoff (see Figure 5-8); more nutrient input leads to higher populations of phytoplankton, which reduce water clarity. Runoff would be influenced by differences in precipitation and the frequency of extreme rainfall events during autumn, winter, and spring. Precipitation trends differ around the world, and there is evidence for increased frequency of extreme rainfall events that may occur in different seasons of the year (see Chapter 4). Wetter climates or climates with more extreme rainfall events would increase export of nutrients and sediment to lakes and streams; dryer climates or those with more even rainfall would reduce export to lakes and streams. Extreme rainfall events would export more if they occurred at seasons when the earth was bare in agricultural watersheds.

In Lake Mendota, the other two-thirds of the variation in water clarity also is related to climate (see Figure 5-8). One-third is through climatic influences on vertical mixing, where warmer summers lead to greater water column stability, less recycling from deep water, and greater water clarity. The remaining one-third is related to the abundance of herbivorous zooplankton that eat phytoplankton. A warming climate would allow invasion of new species of fish that forage on the herbivorous zooplankton; in Lake Mendota, it probably would be the gizzard shad that lives in reservoirs south of Wisconsin (see Dettmers and Stein, 1996).

Several other complexities would lead to different results, depending on the change in precipitation patterns. For example, in a dryer climate, increases in water residence time (Schindler *et al.*, 1996a) would increase the importance of nutrient recycling within lakes, and storage of nutrients in the sediments of lakes would be reduced (Hauer *et al.*, 1997). This is complicated by changes in light penetration that occur if dissolved organic carbon (DOC) inputs are reduced (Magnuson *et al.*, 1997; Schindler, 1997). DOC can reduce light penetration, so a reduction in DOC input would increase light availability and could lead to increases in primary production in deeper water. This influence would depend on whether climate becomes dryer or wetter.

Several empirical and simulation studies support the idea of increased eutrophication with climate change. Results of these studies contradict the expectations of reductions in nutrient loading from catchments in drier climates and greater stability of the water column in warmer climates.

Empirical relations (Regier *et al.*, 1990; Lin and Regier, 1995) suggest that annual primary production by phytoplankton, zooplankton biomass, and sustained yields of fisheries all increase with temperature. Simulations for Lake Erie (Blumberg and Di Toro, 1990) and smaller lakes (Stefan and Fang, 1993) indicate that climate change leads to more eutrophic conditions with respect to loss of oxygen beneath the thermocline in summer. Ogutu-Ohwayo *et al.* (1997) suggest that recent changes in the regional climate of Lake Victoria in Africa may have reduced physical mixing and contributed to increases in deepwater anoxia and thus nutrient recycling contributing to eutrophication.

### *5.7.3.3.2. Acidification*

Acidification of streams and recovery of acidified lakes would be altered by climate changes (Yan *et al.*, 1996; Magnuson *et al*., 1997; Dillon *et al.*, 2001). In addition to direct atmospheric deposition of acids, sulfates deposited in the catchment are transported to streams during storm events as pulses of acidity. Lakes would receive less buffering materials in dryer climates and more in wetter climates. Lakes high in the landscape that

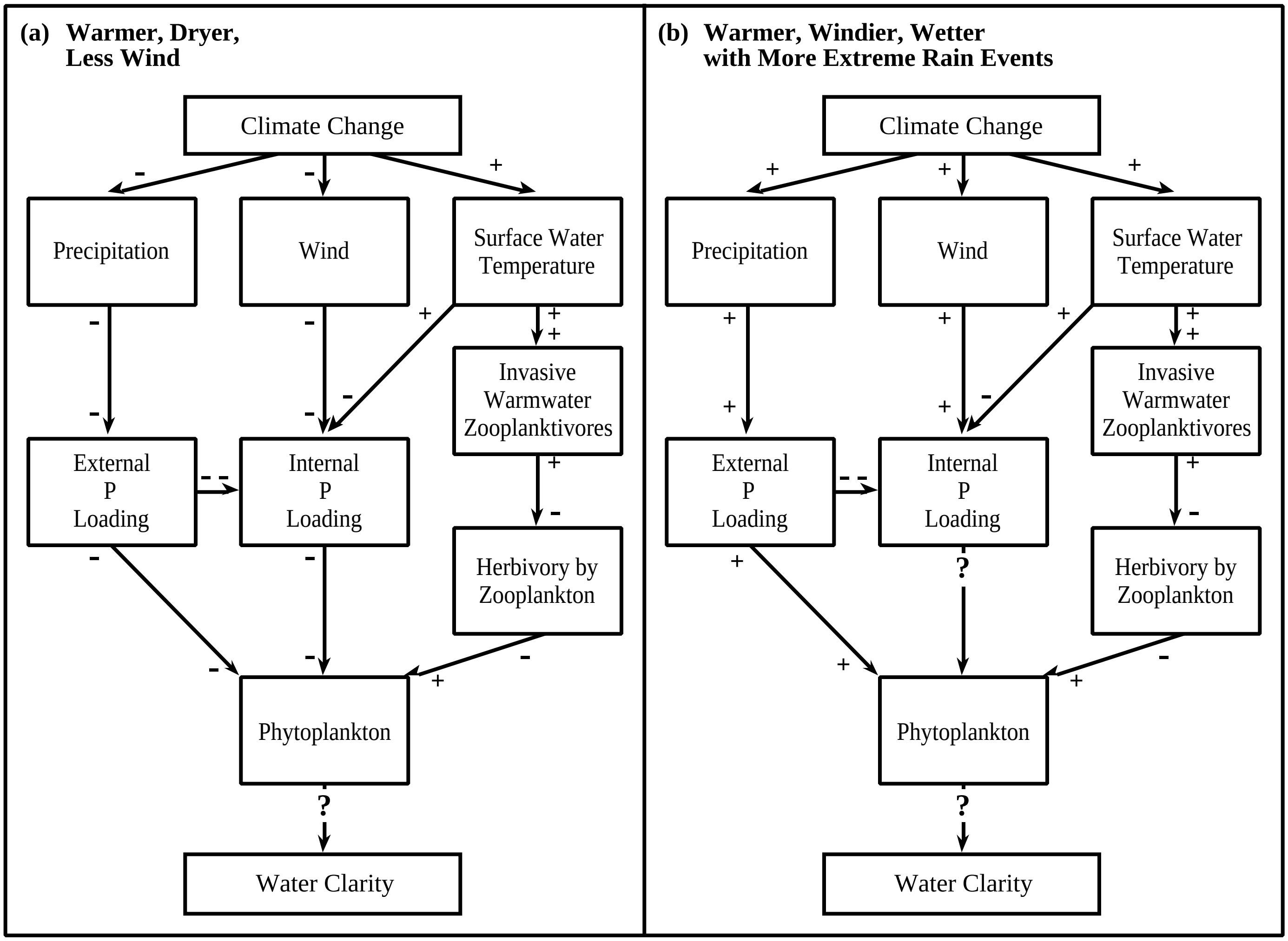

**Figure 5-8**: Diagram of complex interactions between climate change, watershed and lake processes, and water clarity of a eutrophic lake [modified from Lathrop (1998) and information in Lathrop *et al*. (1999)]. The left panel diagrams a warmer, dryer climate with less wind; the right panel diagrams a warmer, windier, and wetter climate with more extreme rain events. In both cases, the altered climate would be expected to change the water quality of the lake, but the complexity of relations leads to uncertain future water clarities. A "+" means an increase and a "**-**" means a decrease in the condition or process; a "**?**" means conflicting expectations. Greater blooms of phytoplankton lead to lower water clarity, and reduced blooms lead to greater clarity. Blooms depend on external and internal loading of phosphorus (P). Dryer climates lead to less external loading, whereas wetter climates or more episodic rains lead to more external loading. Warmer climates lead to warmer surface waters and increased vertical stability in the water column, thus less mixing and internal loading. Warmer waters also allow potential invasion by warmer loving, zooplaktivorous fish that can reduce zooplankton species that in turn reduce algal populations. Windier climates result in increased vertical mixing, thus greater internal loading.

receive less groundwater in dry years would be more vulnerable to acidification (Webster *et al.*, 1996).

#### *5.7.3.3.3. Toxics*

Climate change would interact with biogeochemical transport and transformation of toxics such as mercury, zinc, and pesticides (see brief reviews in Magnuson *et al.*, 1997; Schindler, 1997; and related physiological understanding in Wood and McDonald, 1997). The expected influences are poorly known as well as complex and variable, depending on the toxin, the organism, and the climate scenario. Enough is known to say that toxic stresses are not independent of climate change.

### *5.7.3.4. Other*

#### *5.7.3.4.1. Recreation*

Winter recreational opportunities would decline with warmer climates. Declines in safe ice conditions would reduce all ice-related activities such as ice fishing, ice skating, ice boating, and snowmobiling on lakes and rivers. Recreational uses of lakes that become more eutrophic are likely to be degraded by lower water clarity and increased blooms of noxious blue-green algae. Nonmarket values for water-based recreation are in direct conflict with greater direct uses of that water and a warmer and drier climate. These values ranged from $3–65 per thousand $m^3$ of water for fishing, rafting, and river recreation in general in Colorado (Postel and Carpenter, 1997).

UV-B radiation can be harmful to freshwater organisms (Bothwell *et al.*, 1994; Williamson and Zagarese, 1994; Williamson *et al.*, 1996). Absorption of UV-B is lower in clearwater lakes. Reduction of colored DOC entering lakes during drier conditions results in greater transmission and thus greater harm to organisms (Schindler *et al.*, 1996a; Yan *et al.*, 1996; Schindler, 1997; Williamson *et al.*, 1996).

#### *5.7.3.4.2. Heterogeneity in response*

Responses of lakes and streams to climate change are spatially heterogeneous. Much of the local heterogeneity depends on the type of water body being considered: lake or stream, large or small, shallow or deep, eutrophic or oligotrophic, and so forth. Spatial heterogeneity also results from spatial differences in climate drivers themselves with latitude, altitude, and distance from the coast or a large lake. Inappropriate responses with large expenditures of local resources will result unless the spatial heterogeneity of local responses is understood and can be predicted.

Two ideas that help explain local heterogeneity in responses of lakes and streams to climate change from a landscape perspective are the stream continuum (Vannote *et al.*, 1980; Minshall *et al.*, 1985) and the position of a lake in the landscape (Kratz *et al.*, 1991; Kratz *et al.*, 1997; Magnuson and Kratz, 2000; Riera *et al.*, 2000). Both concepts are geomorphic legacies resulting from the location of the water body in the hydrologic flow field. Headwater streams will be more shielded from warming relative to lowland streams because cool groundwater sources are more important; in forest catchments, they are more shaded from radiation, and in mountain catchments they are at higher, cooler altitudes (Hauer *et al.*, 1997). For lakes, changes in chemical inputs with changes in rainfall will be greater in upland lakes than in lowland lakes. Upland lakes are supplied more by dilute precipitation than by solute-rich groundwater or overland flow. The chemistry of headwater lakes is extremely responsive to climate-driven changes in groundwater inputs because they tend to have smaller volumes and shorter residence times (Krabbenhoft and Webster, 1995).

Differences in the extent of connected wetlands or other sources of DOC contribute to local heterogeneity in responses. Export of DOC to lakes and streams increases in wetter times and decreases during drought. This, in turn, changes light penetration and vertical distribution of solar heating and, in lakes, the depth of thermocline and the relative magnitude of cold and warm thermal habitat for fish (Schindler *et al.*, 1996b). Penetration of UV-B, thus the damaging effects of that radiation (Schindler, 1997), also will differ among lakes. Lakes without large sources of DOC that leach in during wetter periods will respond less to climate changes in these respects.

Shorter term, more stochastic patterns in catchments alter the behavior of lake ecosystems. Consider a drought that increases the likelihood of forest fires in the watersheds of boreal lakes in Ontario (Schindler, 1997). Burned areas typically are patchy and may or may not include the catchment of a given lake. If they do include the lake's watershed, there is an initial increase in the input of solutes to streams and the lake. The lake also is under greater influence of wind mixing with trees gone, which would deepen the thermocline and again alter the relative magnitude of cold and warm thermal habitat for fish.

Predicted responses of threatened anadromous Pacific salmonid stocks in the Columbia basin of the northwestern United States are varied. Climate-related factors that are important to successful reproduction include temperature, the river hydrograph (peak and annual flow), and sedimentation (Neitzel *et al.*, 1991). Based on expected changes in these streams and 60 stocks across the basin, impacts on 23% were judged to be negative, 37% positive, and 40% neutral.

## ***5.7.4. Adaptation Options***

Human responses to climate change could further exacerbate the negative impact on aquatic ecological systems. For example, human responses to a warmer climate and the variety of precipitation scenarios, as well as human population increases, are likely to place greater demands on freshwaters to meet water needs for drinking, industry, and irrigation, along with an increase in water management projects (see Chapter 4).

Potential results would include fewer free-flowing streams and greater fluctuations in water level. These changes would cause a loss of ecosystem services and products from "natural/ unmodified" lakes and streams. Conflicts between developers and those wishing to reduce development pressure on lakes and streams probably would intensify as freshwater becomes either more scarce or more abundant.

One adaptation to climate change would entail poleward transportation of fish, mollusks, and other less vagil organisms across watershed boundaries to cooler waters (Magnuson *et al.*, 1997). Historical introductions and range expansions of species have resulted in extinction or extirpation of preexisting fauna owing to ecological interactions, especially predation by the invader. Well-known examples include the sea lamprey, alewife, smelt, and zebra mussel to the Laurentian Great Lakes in North America (Mills *et al.*, 1994); the peacock bass in Panama (Zaret and Paine, 1973); and the Nile perch into Lake Victoria (Kaufman, 1992). Other examples being documented include the decline or extirpation of cisco and yellow perch in small Wisconsin lakes in the United States with the arrival of rainbow smelt (Hrabik *et al.,* 1998) and the loss of cyprinids (minnows) and the decline of lake trout with the introduction of black bass and rockbass into Ontario lakes in Canada (Casselman, 2000; Jackson, 2000; Vander Zanden, 2000). The history of invasions exemplifies the homogenization of continental faunas (Rahel, 2000), losses in native organisms, and large changes in ecosystem processes and structure—all with economic consequences.

Fisheries (capture, aquaculture, and recreational) and associated introductions would exacerbate biodiversity and exotic problems in a warming climate. A substantial number of inland fisheries are associated with introduced species and stock enhancement. Tilapia, carp, grass carp, and rainbow trout are commonly introduced or stocked. An estimated 38% of the recreational catch is of non-native fishes. The practice of introducing species apparently has resulted in 1,354 introductions of 237 species into 140 countries from pre-1900 to the mid-1980s (FAO, 1988). Despite recognition of the importance of wild native species and increasing appreciation of the importance of biodiversity, impacts of climate change are likely to be managed by introducing better adapted species, if past management practices prevail. In the United States, an average of 38% of the recreational fishery is for non-native species, 75% of which now reproduce naturally (Horak, 1995). As climate change impacts affect the abundance of native (and non-native) species, it is probable that the past precedent of introducing non-native species would be applied to maintaining catches and recreational fishing stability.

Moving species poleward to adapt to changing climate zones for lakes and streams is fraught with scientific uncertainties and human conflict. The exotic can become a superabundant pest species, with negative effects on native organisms—including extirpation and extinction. The invasion ecology of organisms is not a predictive science; many surprises would be expected. The case has been made that managing with exotics increases the instability of the fish community and fish management problems and includes many unexpected consequences (Magnuson, 1976). Introduction of warmer water fauna on top of regional fauna that are having increasing problems from warming climates is likely to be a controversial adaptation.

Climate changes may provide development opportunities for aquaculture if traditional wild fisheries are less stable and markets favor the stability of the aquaculture product. Aquaculture represents a mobile technology that potentially can move with changing climate to the best conditions for culture (see Figure 5-7).

### *5.7.5. Vulnerability*

Specific vulnerable elements include reduction and loss of lake and river ice, loss of habitat for coldwater fish, increases in extinctions and invasions of exotics, and potential exacerbation of existing pollution problems such as eutrophication.

Adaptation to climate change may induce other negative effects related to secondary pressures from new hydrologic engineering structures, poleward transport by humans of fauna and flora adapted to warmer lakes and streams, and interactions resulting from increased stocking and relocation of recreational and aquacultural endeavors.

Inland waters are affected hydrologically, physically, chemically, and biologically by climate change (Arnell *et al.*, 1996, Cushing, 1997). One reason for their vulnerability is that lakes, rivers, and wetlands integrate and reflect human and natural events in their watersheds and airsheds (Naiman *et al.*, 1995b). Interactions with changes to their watersheds, riparian shorelines, and human use of water combine to make lake and stream ecosystems vulnerable. Potential changes in quantity and quality of water reduce the ability of these waters to provide goods and services.

## 5.8. Inland Wetlands

Wetlands are defined here as any area of land where the water table is at or near the surface for some defined period of time, leading to unique physiochemical and biological processes and conditions that are characteristic of shallow flooded systems (Mitch and Gosselink, 1993; Oquist *et al.*, 1996). However, it should be noted that many other definitions also are in use, and care is required in comparing wetland areas in different regions of the world (Lugo *et al*., 1990; Finlayson and van der Valk, 1995; Finlayson and Davidson, 1999).

Wetlands described in this section include those dominated by forested and nonforested vegetation; those on highly organic soils such as peatlands (partially decomposed plant material); and those on mineral soils, as is often typical of riverine systems. Peatlands are peat-accumulating wetlands and usually are divided into bogs and fens, both of which may be forested.

Peatlands are found at all latitudes, from the Arctic to the tropics. The growing vegetation layer of bogs is totally dependent on atmospheric inputs for their water and solute supply; consequently, surface peat layers are acidic and poor in nutrients. Fens receive additional water from overland flow and/or groundwater and typically have near-neutral pH and higher nutrient levels.

Wetlands provide many services and goods, many of which are indirect values (e.g., recreation, education, biodiversity, wild food) and are difficult to quantify. Key ecosystem services and goods considered here include biodiversity, carbon sinks, food, and fiber (peat and wood).

## *5.8.1. State of Wetland Services*

### *5.8.1.1. Habitat and Biodiversity*

Global estimates for the area of wetlands vary according to the definition of wetland used. Spiers (1999) reports global estimates for natural freshwater wetlands of 5.7 million km$^2$. The estimate by Matthews and Fung (1987) is 5.3 million km$^2$, which represents approximately 4% of the Earth's land surface, although a more recent estimate by Lappalainen (1996) is somewhat larger, at 6.4 million km$^2$. This compares well with Finlayson and Davidson's (1999) estimate of about 7 million km$^2$, including 1.3 million km$^2$ of rice paddy.

Maltby and Proctor (1996) estimate that peatlands cover about 4 million km$^2$ (±4%), constituting about 75% of wetlands. More than 90% of peatlands are in temperate, boreal, and subarctic regions. The total area of tropical peatlands is estimated to be 0.37–0.46 million km$^2$ (i.e., approximately 10 % of the global resource), but the full extent is uncertain (Immirzi *et al.*, 1992).

Many wetlands have irregular wetting and drying cycles, driven by climate. To date, little attention has been given to the impact of climate change on these less regular cycles of wetlands in semi-arid and arid regions (Sahagian and Melack, 1998). Changes in the area of these wetlands can be immense but could be monitored by using area-based parameters—for example, functional parameters and wetland extent expressed in terms of ha-days (Sahagian and Melack, 1998).

Species that form wetland plant communities are adapted to varying degrees to life in a flooded environment. These phenomena show large spatial variability, and different species show varying degrees of susceptibility to them, so it is not surprising that wetland vegetation exhibits such a high degree of variation in species composition (Crawford, 1983). Peatland plant communities have been observed to change over long periods of time, reflecting the peat accumulation process and leading to gradually drier conditions. This inherent changeability of wetland communities results largely from their occurrence in environments where a single extremely variable habitat factor—water supply—is predominant (Tallis, 1983). Consequently, land use and climate change impacts on these ecosystems can be expected to be mediated through changes in the hydrological regime.

Primary production in wetland communities is highly variable (Bradbury and Grace, 1983; Lugo *et al.*, 1988). Generally, wetland communities—which are dominated by trees, sedges, and grasses—have higher production rates than those characterized by shrubs and mosses. Organic matter produced in many wetlands is accumulated partially (2–16%—Päivänen and Vasander, 1994) as peat. A necessary antecedent condition for peat formation and accumulation is an excess of water stored on the mineral soil or sediment surface. This arises in humid climatic regions where precipitation exceeds evaporation (Ivanova, 1981; Clymo, 1984) or in more arid climatic regions where lateral inputs of water via surface runoff and/or groundwater seepage are sufficient to exceed evaporative demand (Glaser *et al.*, 1996).

Tropical peatlands play an important role in maintenance of biological diversity by providing a habitat for many tree, mammal, bird, fish, and reptile species (Prentice and Parish, 1992; Rieley and Ahmad-Shah, 1996; Page *et al.*, 1997); some of these species may be endemic or endangered. In common with other peatlands, tropical systems have a significant scientific value that goes beyond their plant and animal communities.

Within the peat also lies a repository of paleoenvironmental and paleogeochemical information that is extremely important in understanding past climatic conditions. These paleorecords are used to estimate rates of peat formation or degradation, former vegetation, climatic conditions, and depositional environments (Morley, 1981; Cecil *et al.*, 1993; Moore and Shearer, 1997).

### *5.8.1.2. Carbon Sink*

The importance of peat-accumulating wetlands to global change is via the large carbon store accumulated over millennia—and the risk that this store would be released to the atmosphere in conditions modified by global change (e.g., fires). The carbon store in boreal and subarctic peatlands alone has been estimated at 455 Gt with an annual sink of slightly less than 0.1 Gt (Gorham, 1991). Tropical peatlands also are a considerable store (total of 70 Gt), containing as much as 5,000 t C ha$^{-1}$, compared with an average of 1,200 t C ha$^{-1}$ for peatlands globally (Immirzi *et al.*, 1992; Diemont *et al.*, 1997). Estimates of the annual sink in tropical peatlands vary from 0.01 Gt (Sorensen, 1993) and 0.06 Gt (Franzen, 1994) to 0.09 Gt (Immirzi *et al.*, 1992)—emphasizing the lack of reliable data.

Optimal conditions for carbon sequestration appear to be in areas with mean annual temperatures between 4 and 10°C (Clymo *et al.*, 1998), which prevail in much of the southern boreal and temperate zones. The present carbon accumulation rate for boreal and subarctic bogs and fens is estimated as 0.21 t C ha$^{-1}$ yr$^{-1}$ (Clymo *et al.*, 1998). Rapid carbon accumulation rates also have been estimated for some tropical peatlands (Neuzil, 1997); retrospective values of Indonesian peatlands range from 0.61 to 1.45 t C ha$^{-1}$ yr$^{-1}$ (Neuzil, 1997).

Another important long-term (>100 years) sink for carbon in forested wetlands is wood biomass. Based on growth data (Shepard *et al.*, 1998) and conversion factors (Turner *et al.*, 1995) for bottomland hardwood forests, southern U.S. swamps, for example, sequester 0.011 Gt C $yr^{-1}$.

A small but significant proportion of organic matter in wetland soils is transformed into methane in the metabolism of methanogenic bacteria. Methane production is a characteristic feature in all wetland soils; the rate is governed largely by substrate availability and temperature (Shannon and White, 1994; Mikkelä *et al.*, 1995; Schimel, 1995; Bergman *et al.*, 1998; Komulainen *et al.*, 1998). Part of the methane produced in anoxic soil is oxidized by methanotrophic bacteria in aerobic surface layers.

The ratio of methane production to consumption determines the magnitude of the flux from the soil to the atmosphere; this rate is governed largely by the depth of the aerobic layer (Roulet *et al.*, 1993; Shannon and White, 1994). The role of vascular plants in providing continuous substrate flux for methanogenesis and as a transport pathway to the atmosphere has been stressed (Whiting and Chanton, 1993; Schimel, 1995; Frenzel and Rudolph, 1998). Slow fermentation of organic matter in growing peat layers of bogs has been cited as the factor that theoretically limits the final volume a bog may reach during its development (Clymo, 1984).

#### 5.8.1.3. Food and Fiber Production

Wild berries growing on peatlands are an important natural resource in many regions of the boreal zone (Reier, 1982; Yudina *et al*., 1986; Salo, 1996). In Finnish peatlands alone, the annual biological yield of wild berries may exceed 150 million kg, of which approximately 10% is picked, with a value of US$13.5 million (Salo, 1996). In North America, cranberry (*Vaccinium macrocarpon*) is commercially cultivated in peatlands (Johnson, 1985). The use of peatlands for agriculture has a long history; presently, 10,000 $km^2$ are under this land use (Immirzi *et al.*, 1992).

The importance of wetlands in North America as waterfowl habitat has long been recognized (Mitch and Gosselink, 1993). The commercial value of these wetlands is not direct; it comes through the added value of the activities of hunters in local economies. Although there are few estimates of this service, much of the wetland conservation effort in North America has focused on conservation, enhancement, and creation of habitat.

Products from tropical forested wetlands include rattans, resins, latex, fungi, fruit, honey, and medicinal plants, sale of which provides revenue for local communities. Exploitation of fish from swamp forests and associated waterways also can supply a modest income and is an important source of protein for local human populations (Immirzi *et al.*, 1996; Lee and Chai, 1996).

Forested wetlands are valuable for wood production, mostly as a result of wetland modification. For example, in Finland, these wetlands produce about 18 million $m^3$ of timber annually—nearly 25% of the total annual increment (Tomppo, 1998). Forested wetlands in the southern United States produce 39 million $m^3$ of timber, of which 33 million $m^3$ are removed annually (Shepard *et al.*, 1998).

Direct harvest of forest resources from tropical peatlands yields several important products, ranging from timber and bark to non-timber products (Immirzi *et al.*, 1996; Lee and Chai, 1996). Southeast Asia's peat swamps yield some of the most valuable tropical timbers—in particular, ramin (*Gonystylus bancanus*) (Ibrahim, 1996).

Peat has been used as a domestic energy source in northwestern and central Europe for centuries (Feehan and O'Donovan, 1996). The present volume of industrial peat harvesting is estimated at 71 million $m^3$, most of it in Finland and Ireland (Asplund, 1996). The peatland area occupied by peat harvesting is rather small in comparison to the area of land uses such as agriculture and forestry; for example, in Finland 24 million $m^3$ of energy peat are harvested from an area of 530 $km^2$. Employment aspects may be important in peat harvesting because most peat sites are located in remote areas where few industrial jobs are available. Peat harvesting may offer an estimated 550 permanent/seasonal jobs for each 1 million $m^3$ of produced peat (Nyrönen, 1996). Canada has a small but prosperous peat harvesting industry for horticulture and medical uses (Rubec *et al.*, 1988; Keys, 1992). Between 1986 and 1990, Canadian shipments were 662,000–812,000 t $yr^{-1}$ (Keys, 1992).

### 5.8.2. Pressures on Wetland Services

Pressures on wetlands are likely to be mediated through changes in hydrology, direct and indirect effects of changes in temperature, and land-use change. There would be interactions among these pressures and subsequent impacts on services and good from these ecosystems.

#### 5.8.2.1. Changes in Hydrology

Climate change will affect the hydrology of individual wetland ecosystems mostly through changes in precipitation and temperature regimes. Because the hydrology of the surface layer of bogs is dependent on atmospheric inputs (Ingram, 1983), changes in the ratio of precipitation to evapotranspiration may be expected to be the main factor in ecosystem change. However, work on the large peatland complexes of the former Glacial Lake Aggazzi region indicate that the hydrology of bogs cannot be considered in isolation or independent of local and regional groundwater flow systems (Siegel and Glaser, 1987; Branfireun and Roulet, 1998) and that groundwater flow reversals, even in ombrotrophic peatlands, can have an impact on their water storage and biogeochemistry (Siegel *et al.*, 1995;

Devito *et al.*, 1996). From the perspective of assessment of climate variability and change of peatlands, these systems need to be viewed in the broader context of their hydrogeological setting.

Fen, marsh, and floodplain wetlands receive additional water influx from the surrounding basin, including underground sources—sometimes from a considerable distance. This means that climate change impacts are partially mediated through changes in the whole basin area or even further afield where groundwater reserves do not correspond with surface basins. These changes also may affect the geochemistry of wetlands. Recharge of local and regional groundwater systems, the position of the wetland relative to the local topography, and the gradient of larger regional groundwater systems are critical factors in determining the variability and stability of moisture storage in wetlands in climatic zones where precipitation does not greatly exceed evaporation (Winter and Woo, 1990). Changes in recharge external to the wetland may be as important to the fate of the wetland under changing climatic conditions as the change in direct precipitation or evaporation on the wetland itself (Woo *et al.*, 1993).

### *5.8.2.2. Changes in Temperature*

Temperature is an important factor controlling many ecological and physical functions of wetlands. Primary productivity and microbial activity are both controlled to a certain extent by temperature conditions. Temperature also affects evapotranspiration rates and has an impact on the water regime. Because higher temperatures and drying of the surface soil usually occur together and interactively affect the ecosystem processes, it is not always possible to separate their impacts.

Although Gorham (1991) suggests that the effects of temperature will be overshadowed by the impacts of water-level drawdown on northern peatlands, the impacts of temperature increases on wetlands on permafrost may be drastic (Gorham, 1994a). A fairly small increase in temperature might initiate large-scale melting of permafrost, with thermokarst erosion and changed hydrological regimes as a consequence (Billings, 1987). Work by Vitt *et al.* (1994) and Halsey *et al.* (1997) has demonstrated clearly the dynamic association between the distribution of peatlands, peatland types, and the presence or absence of permafrost in North America. This association is strong enough that it has been used as a proxy method for inferring climatic variability during the Holocene (Halsey *et al.*, 1995). This might imply shifts from black spruce/*Sphagnum*/lichen communities on permafrost to wetter fen communities, with subsequent changes in carbon cycling.

### *5.8.2.3. Land-Use Change*

Land-use change may create multiple pressures on wetland habitats. Area estimates of the scale of direct development of tropical peatlands vary and provide only an imprecise picture of the current situation (Immirzi *et al.*, 1992; Maltby and Immirzi, 1996). In southeast Asia, agriculture and forestry are the major peatland land uses. Toward the end of the 1980s, it was estimated that in Indonesia alone 3.7 Mha (18% of the total peat swamp forest) had undergone some form of development (Silvius *et al.*, 1987).

Cultivation of tropical peatlands involves measures that radically change the hydrological regime and consequently influence vegetation and soil processes. Forests are cleared and effective drainage installed. In many places in southeast Asia, cultivation of horticultural and estate crops has met with mixed success, and some previously converted peatlands have been abandoned, although peat-forming vegetation has failed to reestablish (Immirzi *et al.*, 1992). Reasons for failure include poor water management and persistent infertility of the soil (Rijksen *et al.*, 1997).

The total area of tropical peatland drained or otherwise altered during forestry management is not known. Silvius *et al.* (1987) suggest that as much as 0.11 million km$^2$ of peatlands in Indonesia (i.e., as much as 50% of the total resource) are possibly being exploited for forestry purposes. In Malaysia, most of the remaining peat swamp forest outside limited conservation areas has been logged (Immirzi *et al.*, 1992). Sustainable-yield forestry is likely to be the most appropriate form of land use for peat swamp forest, but such methods applicable to peat swamps have yet to be developed, let alone implemented (Immirzi *et al.*, 1992).

Use of peatlands for forestry usually brings about smaller changes in the ecosystem. In floodplain swamps and peatlands of the more continental areas of North America, often only tree stands are managed (Dahl and Zoltai, 1997), but in northwestern Europe and the southeastern United States, forestry use includes artificial drainage (Richardson and McCarthy, 1994; Päivänen, 1997). Some 0.15 million km$^2$ have been drained for forestry, mostly in Scandinavia and Russia (Päivänen, 1997). In these cases, much of the original vegetation (Laine *et al.*, 1995) and functions (Aust and Lea, 1991; Minkkinen *et al.*, 1999) are preserved during forestry management. About 70% of the expansive peatlands in North Carolina (6,000 km$^2$) have been entirely or partially degraded through draining, ditching, or clearing (Richardson and Gibbons, 1993).

Peat harvesting for energy or horticultural use has the most drastic impact on the ecosystem; vegetation is removed with the topsoil prior to harvesting, and most of the accumulated peat gradually is extracted (Nyrönen, 1996).

## ***5.8.3. Impacts on Wetland Services***

Generally, climatic warming is expected to start a drying trend in wetland ecosystems. According to Gorham (1991), this largely indirect influence of climate change, leading to alteration in the water level, would be the main agent in ecosystem change and would overshadow the impacts of rising temperature and longer growing seasons in boreal and subarctic peatlands. Monsoonal areas are more likely to be affected by more intense rain events over shorter rainy seasons, exacerbating flooding and erosion in catchments and the wetlands themselves. Similarly, longer dry seasons could alter fire regimes and loss of organic matter to the atmosphere (Hogenbirk and Wein, 1991).

### *5.8.3.1. Habitat and Biodiversity*

Climate change may be expected to have clear impacts on wetland ecosystems, but there are only a few studies available for assessing this. There are some laboratory studies concerned with the responses of individual plant species or groups of species (Jauhiainen *et al.*, 1997, 1998a,b; van der Heijden *et al.*, 1998). Based on these studies, however, it is difficult to predict the responses of plant communities formed by species with somewhat varying environmental requirements.

The response of wetland plant communities to drought has received some attention in temperate freshwater wetlands (Greening and Gerritsen, 1987; Streng *et al.*, 1989). Stratigraphical studies in peatlands have shown hydroseral succession whereby swamp and fen communities gradually develop into bog communities (Tallis, 1983). These changes are largely autogenic, connected to growth of wetland communities and caused by past climatic variability or artificial drainage. An alternative approach to observe the vegetation-environmental change succession has been to use space as a time substitute by mapping different plant communities onto climatic and hydrological surfaces (Gignac *et al.*, 1991). This approach has shown tight coupling between various peatland plants, climate, hydrology, and resultant chemistry—and even for trace gas exchange (Bubier, 1995)—and has been used to infer certain aspects of peatland development through macrofossil analysis (Gorham and Janssens, 1992; Kuhry *et al.*, 1993).

Much is known about how vegetation changes as a result of water-level drawdown following drainage for forestry in northwestern Europe. Drying of surface soil initiates a secondary succession whereby original wetland species gradually are replaced by species that are typical of forests and heathlands (Laine and Vanha-Majamaa, 1992; Vasander *et al.*, 1993, 1997; Laine *et al.*, 1995). Plants living on wet surfaces are the first to disappear, whereas hummock-dwelling species may benefit from drying of surface soil. In nutrient-poor peatlands, bog dwarf-shrubs dominate after water-level drawdown; at more nutrient-rich sites, species composition develops toward upland forest vegetation (Laine and Vanha-Majamaa, 1992; Minkkinen *et al.*, 1999).

The effect of sea-level rise on wetlands has been addressed in several assessments. In northern Australia, extensive seasonally inundated freshwater swamps and floodplains are major biodiversity foci (Finlayson *et al.*, 1988). They extend for approximately 100 km or more along many rivers but could be all but displaced if predicted sea-level rises of 10–30 cm by 2030 occur and are associated with changes in rainfall in the catchment and tidal/storm surges (Bayliss *et al.*, 1997; Eliot *et al.*, 1999). Expected changes have been demonstrated by using information collected from the World Heritage-listed Kakadu National Park, but the scenario of massive displacement of these freshwater wetlands can be extended further afield given similarities in low relief, monsoonal rainfall, and geomorphic processes (Finlayson and Woodroffe, 1996; Eliot *et al.*, 1999). In fact, the potential outcome of such change can be seen in the nearby Mary River system, where saline intrusion, presumably caused by other anthropogenic events, already has destroyed 17,000 ha of freshwater woodland and sedge/grassland (Woodroffe and Mulrennan, 1993; Jonauskas, 1996).

Mechanisms by which environmental factors and biotic interactions control wetland biodiversity are not well understood (Gorham, 1994b). The effects of water-level drawdown after drainage for forestry indicate that the shift in species composition from bog and fen species to forest species only slightly affects plant species richness of individual sites (Laine *et al.*, 1995). In regions dominated by forests, there would be clear reduction in regional diversity as landscapes become homogenized after drainage (Vasander *et al.*, 1997).

The inherent changeability of wetland communities, resulting from spatial and temporal variability in water supply (Tallis, 1983), may be the key factor in the response of wetland communities to climate change. Because there may be differences between species in adaptation potential, community structures would change, and there would be profound effects on the nature of the affected wetlands, as discussed by Gorham (1994a). The response of wetland plant communities to changing environment may have fundamental effects on the species diversity of these ecosystems.

Because of spatial and temporal variability in ecosystem processes, development of systems models for wetlands is becoming an important assessment tool. A fully coupled peatland-climate model has not yet been developed, but there have been some significant advances in modeling various components of the peatland and/or wetland biogeochemical system (Harris and Frolking, 1992; Roulet *et al.*, 1992; Christensen and Cox, 1995; Christensen *et al.*, 1996; Walter *et al.*, 1996; Granberg, 1998), and several process-level models are now used at the global scale (Cao *et al.*, 1996; Potter *et al.*, 1996; Potter, 1997).

Elevated $CO_2$ levels will increase photosynthetic rates in some types of vegetation (e.g., $C_3$ trees and emergent macrophytes) (Bazzaz *et al*., 1990; Idso and Kimball, 1993; Drake *et al*., 1996; Megonigal and Schlesinger, 1997; see also Section 5.6.3.1). Responses of nonvascular vegetation, such as sphagna, have been less clear (Jauhiainen *et al.*, 1994, 1997, 1998a; Jauhiainen and Silvola, 1999). A study in Alaskan tussock tundra found that photosynthetic rates in *Eriophorum vaginatum* quickly adjusted downward such that rates with elevated and ambient $CO_2$ were similar after 1 year (Tissue and Oechel, 1987). However, a sustained increase in net ecosystem carbon sequestration was observed when elevated $CO_2$ treatments were combined with a 4°C increase in temperature (Oechel *et al.*, 1994).

Many $C_3$ plants respond to elevated $CO_2$ with a decrease in stomatal conductance (Curtis, 1996), which could reduce transpiration rates. Because transpiration is an important pathway for water loss from many ecosystems (Schlesinger, 1997), including wetlands (Richardson and McCarthy, 1994), reductions in transpiration rate could affect the position of the aerobic-anaerobic interface in wetland soils (Megonigal and Schlesinger, 1997).

### *5.8.3.2. Carbon Sink*

It has been suggested that water-level drawdown and increased temperature will decrease carbon sequestration in subarctic and boreal peatlands, especially in more southern latitudes (Gorham, 1991). However, conclusions are hampered by the diversity of possible responses.

Impacts of climate change on wetland carbon sink can be measured directly by using the eddy covariance method or chamber techniques (Crill *et al.*, 1988; Fowler *et al.*, 1995; Alm *et al.*, 1997b; Aurela *et al.*, 1998) and modeling the balances (Frolking *et al.*, 1998). In recent years, several high quality, continuous, snow-free season time series of net ecosystem exchange have been obtained for wetlands and peatlands in the subarctic zone (Burton *et al.*, 1996; Friborg *et al.*, 1997; Schreader *et al.*, 1998), the boreal zone (Shurpali *et al.*, 1995; Jarvis *et al.*, 1997; Kelly *et al.*, 1997; Lafleur *et al.*, 1997; Roulet *et al.*, 1997; Suyker *et al.*, 1997; Goulden *et al.*, 1998), and the temperate zone (Happell and Chanton, 1993; Pulliam, 1993). Extrapolations of temporally detailed enclosure measurements along with estimates of the export of DOC have been undertaken to derive peatland carbon balances (Waddington and Roulet, 1996; Carroll and Crill, 1997). These studies point out the difficulty of assessing the sink/source strength of a peatland within a reasonable level of certainty because of errors introduced in the scaling process (Waddington and Roulet, 2000).

For northern peatlands, much is known about the effects of water-level drawdown on the carbon balance, based on studies carried out in peatlands drained for forestry (Glenn *et al.*, 1993; Roulet *et al.*, 1993; Laine *et al.*, 1995; Martikainen *et al.*, 1995; Minkkinen and Laine, 1998). These studies may be used cautiously to represent the climate change impact because the effect of drainage on ecosystem structure and functioning is similar to that predicted after drying caused by climate change in northern latitudes (Laine *et al.*, 1996). The change observed in vegetation structure after water-level drawdown directs biomass production to the shrub and tree layer; in most cases, primary production and biomass increased (Laiho and Finér, 1996; Laiho and Laine, 1997; Sharitz and Gresham, 1998). Simultaneously, litter production increased (Laiho and Finér, 1996; Laiho and Laine, 1997; Finér and Laine, 1998), and the litter was more resistant to decomposition (Meentemeyer, 1984; Berg *et al.*, 1993; Couteaux *et al.*, 1998). There is some evidence that part of the carbon from decomposing litter is stored in the peat profile down to 0.5-m depth (Domisch *et al.*, 1998). These alterations to vegetation production and litter flow into soil have been observed to keep the net carbon accumulation rate into the soil of boreal bogs in most cases at the level prior to water-level drawdown, sometimes exceeding this level (Minkkinen and Laine, 1998). However, increased duration and shortened return periods of extreme droughts may have detrimental effects on the peat carbon balance, as indicated by the results of Alm *et al.* (1997b).

Water-level drawdown will cause a decrease in $CH_4$ emissions as substrate flux to anoxic layers is decreased and consumption of $CH_4$ in the thicker aerobic layer is enhanced (Glenn *et al.*, 1993; Roulet *et al.*, 1993; Martikainen *et al.*, 1995; Roulet and Moore, 1995). It has been suggested that reduced $CH_4$ emissions after water-level drawdown, together with an increase in tree biomass and a fairly small change in carbon sequestration into peat, may even decrease the greenhouse effect of these ecosystems (Laine *et al.*, 1996).

Therefore, it is not immediately clear that a warmer, drier climate necessarily will lead to a large loss of stored peat for all peatland types. The feedback between climate and peatland hydrology and the autogenic nature of peatland development is poorly understood. Clymo (1984) first developed this idea, and it has recently been used to show how surface topography on peatlands is preserved (Belyea and Clymo, 1999). Hilbert *et al.* (1998) have expanded on the work of Clymo (1984) and developed a model of peatland growth that explicitly incorporates hydrology and feedbacks between moisture storage and peatland production and decomposition. Their studies suggest that some peatland types (e.g., most bogs) will adjust relatively quickly to perturbations in moisture storage.

Future rates of carbon sequestration in swamps on mineral soils will depend largely on the response of trees to changes in hydrology, temperature, and elevated $CO_2$ concentrations. The aboveground productivity of temperate zone swamp forests is strongly regulated by the extent of soil saturation and flooding, and the long-term effect of changes in hydrology on growth will depend on the position of forests along the current hydrological gradient (Megonigal *et al.*, 1997). Drier conditions may increase NPP on extensively flooded sites and decrease it on dry and intermediate sites.

The combination of high soil carbon density and rapid warming in peatlands underlain by permafrost have raised concerns that northern peatlands may become net carbon sources rather than sinks (Lal *et al.*, 2000). Indeed, working in tussock and wet tundra, Oechel *et al.* (1993) estimate that these systems are now net sources of 0.19 Gt C, caused mainly by melting of permafrost and lowering of the water table. Botch *et al.* (1995) report that peatlands of the former Soviet Union are net sources of 0.07 Gt C $yr^{-1}$. Other parts of the boreal zone may have become enhanced sinks as a result of recent warming (Myneni *et al.*, 1997), and continued warming could change the equation to favor net carbon storage—as suggested by warming experiments in arctic ecosystems (Hobbie, 1996).

Higher temperatures may affect carbon cycling of other wetlands as well. Increased photosynthetic activity of deep-rooted wetland plants, such as sedges, may enhance substrate availability for methanogenesis—which, together with higher temperatures, might lead to higher $CH_4$ emissions (Valentine *et al.*, 1994; Bergman *et al.*, 1998; Segers, 1998) where water level would remain near the soil surface.

Land uses such as agriculture and forestryalways change carbon fluxes in ecosystems. High carbon losses have been reported for agricultural crop production on drained wetland in Europe and North America, as much as 10–20 t C $ha^{-1}$ $yr^{-1}$ (Armentano and Menges, 1986). Studies have shown that agriculture on peat soils may contribute significantly to nitrous oxide ($N_2O$) emission (Nykänen *et al.*, 1995a). Kasimir-Klemedtsson *et al.*, (1997) conclude that agricultural practices on organic soils lead to a net increase in emissions of GHGs because of large fluxes of $CO_2$ and $N_2O$, over decreases in emissions of $CH_4$.

As agricultural management fundamentally alters the processes of wetlands and gradually leads to decreases in wetland area (Okruszko, 1996). Large areas of wetlands have been lost in Russia, Europe, and North America by complete drainage and conversion to other land uses. It has been estimated that 53% of the original 89 Mha of wetlands in the coterminous United States were lost by the 1980s (Shepard *et al.*, 1998), much of it to agricultural conversion. Development for agriculture also can have offsite effects—for example, reduced water quality that impacts fisheries (Notohadiprawiro, 1998). Arable agriculture always transforms wetlands into sources of GHGs to the atmosphere (Armentano and Menges, 1986; Okruszko, 1996), with the exception of $CH_4$.

Consequences of the development of tropical peatlands include lowering of the water table, which promotes peat oxidation and decomposition. Peat loss and subsidence can occur at very fast rates—as much as 0.9 cm per month (Dradjad *et al.*, 1986). Eventually, shrinkage and oxidation may lead to loss of the entire peat profile and exposure of underlying nutrient-poor substrates or potential acid sulphate soils (Maltby *et al.*, 1996; Rieley *et al.*, 1996). In subcoastal situations, this may be followed by marine inundation.

When wetland use for forestry involves only management of existing tree stands, the impacts on functions and processes may be small, and the ecosystem may remain within the wetland concept (Aust and Lea, 1991; Minkkinen *et al.*, 1999). However, if artificial drainage is included, decay of organic matter is enhanced, with consequent increases in $CO_2$ emissions from peat (Glenn *et al.*, 1993; Silvola *et al.*, 1996). The results of the carbon balance change reported are highly variable, depending on methods used and climatological differences. Losses of peat carbon have been reported by Braekke and Finer (1991) and Sakovets and Germanova (1992), whereas increased post-drainage carbon stores have been reported by Anderson *et al.* (1992) and Vompersky *et al.* (1992). Based on a large cross-sectional data set from forest drainage areas in Finland, it was shown that carbon accumulation in peat soil increased in southern parts of the country (annual mean temperature 3–4.5°C) but decreased in northern Finland (with mean temperature of 0–1°C). The carbon accumulation increase was clearest for nutrient-poor bog sites (Minkkinen and Laine, 1998). Cannell and Dewar (1995) have concluded that drainage and planting of conifers on organic soils produces little long-term change in soil carbon stores because enhanced organic matter oxidation is compensated by increased litter production of the tree stand.

Water-level drawdown after drainage decreases $CH_4$ emissions from peatland (Glenn *et al.*, 1993; Roulet *et al.*, 1993; Martikainen *et al.*, 1995; Roulet and Moore, 1995). Increased consumption in the surface soil may even form a small $CH_4$ sink in some cases (Glenn *et al.*, 1993; Roulet *et al.*, 1993; Fowler *et al.*, 1995; Martikainen *et al.*, 1995; Roulet and Moore, 1995; Komulainen *et al.*, 1998). The effect of drainage on $N_2O$ emissions has been reported to be fairly small and restricted to fen sites (Martikainen *et al.*, 1993).

Forest harvesting in tropical swamp forests can result in changes to the quality and quantity of organic matter inputs from vegetation, and—as the work of Brady (1997) has shown—if tree root mats decline, net accumulation of peat also may decline. Where selective logging is combined with artificial drainage, decomposition and subsidence of peat may proceed at rates of 3.5—6.0 cm $yr^{-1}$ (Brady, 1997). In contrast with peatlands of the temperate and boreal zone, there has been poor success with establishing forestry plantations on tropical peatlands.

Peat harvesting totally changes the structure and functioning of the original ecosystem by removing the vegetation and finally most of the accumulated peat deposit. This has a fundamental impact on the GHG balances of harvesting sites: $CH_4$ emissions almost stop (Nykänen *et al.*, 1995b), but the whole accumulated carbon store forms a $CO_2$ source to the atmosphere during harvesting and combustion (Rodhe and Svensson, 1995), even if peat combustion may replace imported energy in countries with no other major domestic energy sources.

Recent findings have shown that restoration of cut-away peatlands after harvesting soon initiates colonization of peatland plants (Smart *et al.*, 1989; Tuittila and Komulainen, 1995; Campeau and Rochefort, 1996; Wheeler, 1996; Wind-Mulder *et al.*, 1996; Ferland and Rochefort, 1997; LaRose *et al.*, 1997; Price *et al.*, 1998) and may restart carbon accumulation (Tuittila *et al.*, 1999).

Elevated $CO_2$ is likely to stimulate $CH_4$ emissions in a wide variety of wetland ecosystems, including freshwater marshes (Megonigal and Schlesinger, 1997) and rice paddies (Ziska *et al.*, 1998). Studies in northern peatlands have been equivocal: One study reports a maximum increase of 250% (Hutchin *et al.*, 1995) and another no increase (Saarnio *et al.*, 1998). Because $CH_4$ is a more powerful GHG than $CO_2$, wetlands amplify the greenhouse effect of elevated $CO_2$ by converting a portion of this gas to $CH_4$.

Increased nitrogen deposition may alter the species composition of wetland communities (Aerts *et al*., 1992) and their production, leading to higher production and net accumulation rates in peatlands where production is nitrogen limited (Aerts *et al.*, 1995). There is some indication that nitrogen inputs may affect trace gas emissions from peat soils by increasing emissions of $N_2O$ and sometimes decreasing those of $CO_2$ and $CH_4$ (Aerts, 1997; Aerts and Ludwig, 1997; Aerts and Toet, 1997; Regina *et al.*, 1998).

#### *5.8.3.3. Food and Fiber*

Services that involve artificial drainage might even benefit from climatic warming and additionally lowered water levels. Moya *et al.* (1998) report that rice biomass and seed yield is increased by $CO_2$ concentrations of 200 and 300 ppm above ambient, but these increases are diminished or reversed when air temperature is elevated by 4°C. In the northern latitudes of Scandinavia, forest production in drained peatlands clearly is favored by higher temperatures (Keltikangas *et al.*, 1986) and lower water-table levels.

### *5.8.4. Adaptation Options for Wetlands*

Most wetland processes are dependent on catchment-level hydrology, which is being changed by land-use changes at fairly large scales. Thus, it may be very difficult if not impossible to adapt to the consequences of projeted climate change. For key habitats, small-scale restoration may be possible if sufficient water is available. In cases where wetlands are used for arable agriculture, the impact on the carbon balance could be controlled by the choice of cropping method, including alternative crops and depth of drainage.

### *5.8.5. Vulnerability of Functions and Key Services of Wetlands*

The types of inland wetlands that are most vulnerable to global change (i.e., experience the largest changes) are difficult to ascertain. As concluded in the SAR, arctic and subarctic ombrotrophic bog communities on permafrost would change drastically after thawing of the frost layer and might be considered prime candidates in the vulnerability assessment, together with more southern depressional wetlands with small catchment areas. The increasing speed of peatland conversion and drainage in southeast Asia will place these areas at a greatly increased risk of fire. This will be one of the principal factors in determining the viability of tropical systems.

Global change impacts on wetlands would cause changes in many of the ecosystem services of wetlands. Especially vulnerable are functions that depend on a high degree of water availability. Services that involve artificial drainage might even benefit from climatic warming and additionally lowered water levels. For instance, in northern latitudes of Scandinavia, forest production in drained peatlands clearly is favored by higher temperatures (Keltikangas *et al.*, 1986).

## 5.9. Arctic and Alpine Ecosystems

Arctic and alpine ecosystems are characterized by low human population densities. However, they provide important goods and services locally and globally. At the local scale, they are the resource base of many indigenous cultures and provide recreation, food, and fiber to people in adjacent regions. At the global level, arctic and boreal regions play an important role in the world's climate system. They contain 40% of the world's reactive soil carbon (McGuire *et al.*, 1995); influence global heat transport through their impact on regional water and energy exchange with the atmosphere; and determine freshwater input to the Arctic Ocean, which influences bottom-water formation and thermohaline circulation of the oceans. Alpine regions are important sources of freshwater and hydropower for surrounding lowlands. Changes in these goods and services would have socioeconomic impacts throughout the world.

Chapter 16 presents information on the effects of climate on the physical environment of polar regions. The impacts of climate change on high-mountain systems and arctic tundra are extensively reviewed in the SAR by Beniston and Fox (1996) and Allen-Diaz (1996), respectively. This section emphasizes the effects of climatic change particularly on ecosystem production, carbon stores, biodiversity, and water flow—which, in turn, can affect the productivity especially of alpine ecosystems.

### *5.9.1. State and Trends of Goods and Services*

#### *5.9.1.1. Arctic Ecosystems*

Arctic and alpine tundra each occupy 4 million $km^2$ (Körner, 1999). Approximately 25% of the tundra is ice-covered, 25% shrub-dominated, and the rest dominated by herbaceous plants. Soil carbon stocks in boreal and tundra peatlands are large (see Sections 5.6 and 5.8).

High-latitude warming that has occurred in the Arctic region since the 1960s (Chapman and Walsh, 1993; Serreze *et al.*, 2000) is consistent with simulations of climate models that predict increased greenhouse forcing (Kattenberg *et al.*, 1996). Climatic change has been regionally variable, with cooling in northeastern North America and warming in northwestern North America and northern Siberia. The warming results from a change in the frequency of circulation modes rather than gradual warming (Palmer, 1999). Precipitation (P) and surface evaporation (E) have increased at high latitudes, with no significant temporal trend in the balance between the two (P-E) (Serreze *et al.*, 2000).

Permafrost underlies 20–25% of the northern hemisphere land area (Brown *et al.*, 1997a). Ice that forms during periods of cold climate frequently constitutes a high proportion (20–30%)

of the volume of these frozen soils (Brown *et al.*, 1997a). Consequently, melting of permafrost can lead to surface collapse of soils forming thermokarst, an irregular topography of mounds, pits, troughs, and depressions that may or may not be filled with water, depending on topography. Permafrost temperatures have warmed in western North America by 2–4°C from 1940 to 1980 (Lachenbruch and Marshall, 1986) and in Siberia by 0.6–0.7°C from 1970 to 1990 (Pavlov, 1994), whereas permafrost cooled in northeastern Canada (Wang and Allard, 1995)—patterns that roughly parallel recent trends in air temperature. However, the magnitude of warming and patterns of interannual variation (roughly 10-year oscillations) in permafrost temperatures are not readily explained as a simple response to regional warming (Osterkamp *et al.*, 1994). Changes in permafrost regime probably reflect undocumented changes in the thickness or thermal conductance of snow or vegetation, in addition to changes in air temperature (Osterkamp and Romanovsky, 1999). Thermokarst features are developing actively in the zone of discontinuous permafrost (Osterkamp and Romanovsky, 1999), particularly in association with fire and human disturbance, but there are no long-term records from which to detect trends in the regional frequency of thermokarst.

In contrast to the long-term trend in tundra carbon accumulation during the Holocene, flux measurements in Alaska suggest that recent warming may have converted tundra from a net carbon sink to a source of as much as 0.7 Gt C yr$^{-1}$ (Oechel *et al.*, 1993; Oechel and Vourlitis, 1994). The direction and magnitude of the response of carbon exchange to warming may be regionally variable, depending on climate, topography, and disturbance regime (Zimov *et al.*, 1999, McGuire *et al.*, 2000). During recent decades, peak-to-trough amplitude in the seasonal cycle of atmospheric $CO_2$ concentrations has increased, and the phase has advanced at arctic and subarctic $CO_2$ observation stations north of 55°N (Keeling *et al.*, 1996). This change in carbon dynamics in the atmosphere probably reflects some combination of increased uptake during the first half of the growing season (Randerson *et al.*, 1999), increased winter efflux (Chapin *et al.*, 1996), and increased seasonality of carbon exchange associated with disturbance (Zimov *et al.*, 1999). This "inverse" approach generally has concluded that mid-northern latitudes were a net carbon sink during the 1980s and early 1990s (Tans *et al.*, 1990; Ciais *et al.*, 1995; Fan *et al.*, 1998; Bousquet *et al.*, 1999; Rayner *et al.*, 1999). At high northern latitudes, these models give a wider range of estimates, with some analyses pointing to a net source (Ciais *et al.*, 1995; Fan *et al.*, 1998) and others to a sink (Bousquet *et al.*, 1999; Rayner *et al.*, 1999).

High-latitude wetlands and lakes account for 5–10% of global $CH_4$ fluxes to the atmosphere (Reeburgh and Whalen, 1992). These fluxes increase dramatically with thermokarst (Zimov *et al.*, 1997), acting as a potentially important positive feedback to global warming.

Satellite imagery suggests an increase in NDVI (a measure of "greenness") from 1981 to 1991 (Myneni *et al.*, 1997), although interpretation is complicated by changes in sensor calibration (Fung, 1997). If these changes in satellite imagery are an accurate reflection of vegetation activity, changes in NDVI could help explain the increase in seasonal amplitude in atmospheric $CO_2$ observed at high northern latitudes (Keeling *et al.*, 1996; Randerson *et al.*, 1999). This also is consistent with the increased biomass of shrubs in the arctic tundra (Chapin *et al.*, 1995).

#### 5.9.1.2. Alpine Ecosystems

Climatic changes observed in alpine areas generally have paralleled climatic patterns in surrounding regions, with the most pronounced warming at high latitudes, in the Alps, and in Asia and the least pronounced changes in tropical alpine regions (Diaz and Bradley, 1997). Precipitation generally has increased, with the most pronounced changes in winter, leading to increased snow depth (Beniston, 1997).

Regional trends in climate have led to shrinkage of alpine and subpolar glaciers, equivalent to a 0.25 ± 0.1 mm yr$^{-1}$ of sea-level change or 16% of the sea-level rise in the past 100 years (Dyurgerov and Meier, 1997). This trend has been regionally variable, with Asia contributing 45% of this sea-level rise and arctic islands an additional 18%. The net mass reduction of the alpine glaciers has been most pronounced since 1980, when regional warming was greatest.

Climatic warming observed in the Alps has been associated with upward movement of some plant taxa of 1–4 m per decade on mountaintops and loss of some taxa that formerly were restricted to high elevations (Braun-Blaunquet, 1956; Grabherr *et al.*, 1994). In general, direct human impacts on alpine vegetation from grazing, tourism, and nitrogen deposition are so strong that climatic effects on goods and services provided by alpine ecosystems are difficult to detect (Körner, 1999). Soil carbon stocks per unit area in alpine ecosystems are only one-third as great as those in the arctic because greater topographic relief promotes greater drainage and decomposition than in the arctic (Körner, 1995b).

### 5.9.2. Responses of Arctic and Alpine Ecosystems and Impacts on their Goods and Services

#### 5.9.2.1. Impacts Resulting from Changes in Climate on Arctic Ecosystems

Changes in climate are likely to be the greatest cause of changes in goods and services in the arctic (Walker *et al.*, 2001). Projected climatic warming of 4–10°C by the end of the century probably would cause substantial increases in decomposition, nutrient release, and primary production. The net effect on the carbon balance will depend primarily on soil moisture (McKane *et al.*, 1997; McGuire *et al.*, 2000), which cannot be projected with confidence. In general, surface soils on slopes are expected to become drier as thaw depth increases. Lowlands may experience substantial thermokarst, impoundment of water, and reduced aeration.

Plant production frequently is limited in the arctic by excessive moisture and slow turnover of nutrients in soils, and warming and drying of soils is likely to enhance decomposition, nutrient mineralization, and productivity. Many of these changes in productivity may be mediated by changes in species composition and therefore are likely to lag changes in climate by years to decades (Chapin *et al.,* 1995; Shaver *et al.,* 2000). Threshold changes in productivity associated with poleward movement of the treeline is likely to experience time lags of decades to centuries because of limitations in dispersal and establishment of trees (Starfield and Chapin, 1996; Chapin and Starfield, 1997).

The net effect of warming on carbon stores in high-latitude ecosystems depends on changes in the balance between production and decomposition. Decomposition initially may respond more rapidly than production, causing trends toward net carbon efflux (Shaver *et al.,* 1992; Smith and Shugart 1993).

Warming-induced thermokarst is likely to increase $CH_4$ flux to the atmosphere in lowlands, particularly peatlands of northern Canada and western Siberia (Gorham, 1991; Roulet and Ash, 1992; see also Section 5.8) and the loess-dominated “yedoma” sediments of central and eastern Siberia (Zimov *et al.,* 1997). Fires and other disturbances are likely to affect the thermokarsts; however, the role of these disturbances—which can be mediated with changes in regional climate in inducing thermokarst—are poorly understood.

Changes in community composition associated with warming are likely to alter feedbacks to climate. Tundra has a three- to six-fold higher winter albedo than boreal forest, but summer albedo and energy partitioning differ more strongly among ecosystems within tundra or boreal forest than between these two biomes (Betts and Ball, 1997; Eugster *et al.,* 2000). If regional surface warming continues, changes in albedo and energy absorption during winter are likely to act as positive feedbacks to regional warming as a result of earlier melting of snow and, over the long term, poleward movement of the treeline. Surface drying and a change in dominance from mosses to vascular plants also would enhance sensible heat flux and regional warming in tundra (Lynch *et al.,* 1999; Chapin *et al.,* 2000).

Poleward migration of taxa from boreal forest to the Arctic tundra will depend not only on warming climate but also on dispersal rates, colonization rates, and species interactions and therefore may exhibit substantial time lags. The arctic historically has experienced fewer invasions of weeds and other exotic taxa than other regions (Billings, 1973). Some of the most important changes in diversity in the arctic may be changes in the abundance of caribou, waterfowl, and other subsistence resources (see also Section 5.4). Changes in community composition and productivity may be particularly pronounced in the high arctic, where much of the surface currently is unvegetated and is prone to establishment and expansion of additional vegetation (Wookey *et al.,* 1993; Callaghan and Jonasson, 1995).

#### *5.9.2.2. Impacts of Land-Use Change*

Extraction of oil and mineral resources is likely to be the greatest direct human disturbance in the arctic. Although the spatial extent of these disturbances is small, their impacts can be far-reaching because of road and pipeline systems associated with them (Walker *et al.,* 1987). Roads in particular open previously inaccessible areas to new development, either directly related to tourism and hunting or to support facilities for resource extraction.

Changes in goods and services in alpine ecosystems are likely to be dominated by changes in land use associated with grazing, recreation, and other direct impacts, as a result of their proximity to population centers (Körner, 1999; Walker *et al.,* 2001). Many of the alpine zones with greatest biodiversity, such as the Caucasus and Himalayas, are areas where human population pressures may lead to most pronounced land-use change in alpine zones (Akakhanjanz and Breckle, 1995). Direct impacts from human activities are likely to be most pronounced in lower elevational zones that are most accessible to people (Körner, 1999). Improved road access to alpine areas often increases human use for recreation, mining, and grazing and increased forestry pressure at lower elevations (Miller *et al.,* 1996). Overgrazing and trampling by people and animals may tend to destabilize vegetation, leading to erosion and loss of soils that are the long-term basis of the productive capacity of alpine ecosystems.

Alpine areas that are downwind of human population or industrial centers experience substantial rates of nitrogen deposition and acid rain (Körner, 1999). Continued nitrogen deposition at high altitudes, along with changes in land use that lead to soil erosion, can threaten provision of clean water to surrounding regions. Nitrogen deposition occurs primarily during the winter and is transmitted directly to streams during snowmelt, so it readily enters water supplies (Körner, 1999). Many places that use water from alpine areas depend on slow release of the water by melting of snowfields in spring and summer. Warming is likely to create a shortened snowmelt season, with rapid water release creating floods and, later, growing-season droughts. These changes in seasonality, combined with increased harvest in montane forests, would amplify floods.

### ***5.9.3. Adaptation Options***

Opportunities for adapting to expected changes in high-latitude ecosystems are limited because these systems will respond most strongly to globally induced changes in climate rather than to regionally controlled and regulated factors. The effects of climate and disturbance regime on regional productivity and carbon balance may be difficult to alter at the local or regional scale. The most important opportunities for mitigation will be protection of peatlands, yedoma sediments, and other carbon-rich areas from large-scale hydrological change. It is unlikely that expected changes in fire or thermokarst regime can be altered except in areas of high population densities. Adaptation options for forests that occur in these areas are presented in Section 5.6.

The most important opportunities for adaptation in arctic ecosystems may exist for culturally important resources such as reindeer, caribou, waterfowl, and specific plant taxa (see Section 5.4). Careful management of these resources could minimize climatic impacts on indigenous peoples. Another adaptation option is diversification. Many high-latitude regions depend strongly on one or a few resources—such as timber, oil, reindeer, or wages from fighting fires. Economic diversification would reduce the impacts of large changes in the availability or economic value of particular goods and services.

In alpine regions, major opportunities for adaptation relate to protection from large-scale changes in land use or pollutant levels. Because these causes of change are local to regional, wise management could minimize changes.

### *5.9.4. Vulnerability*

Goods and services provided by many arctic regions depend on the physical integrity of permafrost and therefore are vulnerable to warming-induced thermokarst. The large carbon stocks in these regions are vulnerable to loss to the atmosphere as $CO_2$ or $CH_4$, which could act as positive feedbacks to global warming. A few key taxa of animals—such as reindeer, caribou, and waterfowl—are critical cultural resources. Often these taxa are concentrated in particular habitats, such as riparian areas or high-Arctic oases. Ecological changes that modify the population dynamics of these species or modify the impact on their habitats could have important social consequences.

High levels of endemism in many alpine flora and their inability to migrate upward mean that these species are most vulnerable. In addition, alpine soils are vulnerable to losses from erosion, which would radically reduce the goods and services that these regions could provide.

## 5.10. Research Needs across Ecosystems

Most of the foregoing sections mention the need for further research in many areas. Results are needed for future assessments of the impacts of climate change on goods and services provided by ecosystems, adaptation options, or identification of vulnerable systems or regions. It also is clear that most of the research to date has focused on impacts and some on adaptation, with little effort on the vulnerability of systems or regions. Key areas where research is needed to improve future assessments and decrease uncertainties in existing knowledge are as follows (order does not imply priority):

- Long-term monitoring of agricultural processes for detection of impacts of climate change and climate variability and development of possible adaptations to these impacts
- Coupling of pest and crop/animal process modeling to estimate climate-induced pest impacts on crops and livestock
- Pest and disease interaction with climate change and the impact of these interactions on many ecosystems
- Development of coupled biophysical-economic modeling of farm-level decisionmaking under climate change
- Improved understanding of agricultural vulnerability and adaptive capacity
- Improved understanding of interactive effects of climate change and management on carbon source-sink relations in many ecosystems
- Determination of mechanisms that create the often complex changes in wildlife populations, through a systems-type approach to problemsolving such as strategic cyclical scaling (SCS)[3]
- Development of better understanding of the link between biodiversity and ecosystems functions (role each species plays in ecosystem goods and services is necessary to understand the risks and possible surprises associated with species loss)
- The impact of land-use change on biodiversity and the consequence of these impacts on good and services from ecosystems
- The impact of the added stress of climate change on many ecosystems that are under pressure from human activities
- Exploration of management options (including individual species range) that could be used to adapt to the impacts of global change (allowing ecosystems to continue to provide essential goods and services in a changed and rapidly changing world)
- Exploration of the role of integrated management as a means for providing better management options for many ecosystems
- Long-term experiments on intact natural ecosystems across their native range to study the interactive effect of climate change, elevated $CO_2$, nutrient/pollution deposition, soil moisture, and WUE of plants
- The interaction of elevated $CO_2$, increased temperature, and changes in soil moisture with nitrogen deposition and land-use change in influencing the goods and services provided by ecosystems
- Valuation of nonmarket goods and services, such as recreation and NWFP
- Management of exotics and achieving desired community structures (research and alternative models for such activities needed in terrestrial and aquatic systems)

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[3] SCS has three basic steps: detection, attribution, and validation (Root and Schneider, 1995). Many types of ecological data would improve the ability to conduct studies based on the SCS approach. First, additional monitoring systems are needed that would allow long-term data to be collected over a broad scale. Such data would provide the means to detect patterns of change more efficiently. Second, specific small-scale studies examining possible cause of a shifting pattern are needed. After several such studies are completed, these data could be used to validate the cause or mechanism.

- Development of a predictive approach for identifying and dealing with spatial heterogeneity, since responses to climate change differ greatly in adjacent areas (increasing the risk of inappropriate actions with large expenditures of local resources)
- Development of systems models for peatlands, including analysis and modeling of the dynamic interactions between climate change impacts, vegetation development, and carbon exchange between wetland ecosystems and the atmosphere at several spatial scales (from stands to regions to the globe)
- The role of fire and other disturbances on many ecosystems and their role in trace-gas budgets
- Time lags by which productivity, decomposition, and disturbance in ecosystems respond to climatic change
- Assessment of the vulnerability of unique biological resources that are culturally important to many indigenous peoples.

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