

Biodiversity

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5.1 INTRODUCTION AND FRAMEWORK

This synthesis and assessment report builds on an extensive scientific literature and series of recent assessments of the historical and potential impacts of climate change and climate variability on managed and unmanaged ecosystems and their constituent biota and processes. It identifies changes in resource conditions that are now being observed, and examines whether these changes can be attributed in whole or part to climate change. It also highlights changes in resource conditions that recent scientific studies suggest are most likely to occur in response to climate change, and when and where to look for these changes. As outlined in the Climate Change Science Program (CCSP) Synthesis and Assessment Product 4.3 (SAP 4.3) prospectus, this chapter will specifically address climate-related issues in species diversity and rare ecosystems.

In this chapter the focus is on the near-term future. In some cases, key results are reported out to 100 years to provide a larger context but the emphasis is on next 25–50 years. This nearer-term focus is chosen for two reasons. First, for many natural resources, planning and management activities already address these time scales through development of long-lived infrastructure, forest rotations, and other significant investments. Second, climate projections are relatively certain over the next few decades. Emission scenarios for the next few decades

do not diverge from each other significantly because of the inertia of the energy system. Most projections of greenhouse gas emissions assume that it will take decades to make major changes in the energy infrastructure, and only begin to diverge rapidly after several decades have passed (30–50 years).

The potential impacts of climate change on biological diversity at all levels of biological and ecological organization have been of concern to the scientific community for some time (Peters and Lovejoy 1992; IPCC 1990; Lovejoy and Hannah 2005). In recent years, the scientific literature has focused on a variety of observed changes in biodiversity and has continued to explore the potential for change due to variation in the physical climate system (IPCC 2001; IPCC 2007; Millennium Ecosystem Assessment (MEA) 2005). The focus of the chapter is mainly, although not exclusively, on ecosystems within the United States; in some areas, little work has been done here but analogs exist in other regions. Because there have been several recent comprehensive reviews of the overall topic (Lovejoy and Hannah 2005; Parmesan 2007; IPCC 2007), we will not attempt another encyclopedic review in this chapter. Instead, the chapter will focus on the particular issues of particular concern to U.S. decision-makers, as outlined in the governing prospectus. The chapter also explores the implications of



changes in biological diversity for the provision of ecosystem services (MEA 2005) and, finally, the implications of these findings for observation and monitoring systems. In each of the following sections, we provide a summary of current examples in the literature of the topics identified. There are inevitably some topics that have not been explored, although a growing literature exists (e.g., Poff et al. 2002). This is purely a function of the governing prospectus for the assessment.

This chapter thus summarizes and evaluates the current knowledge, based on both observed and potential impacts with respect to the following topics:

- Changes in Distributions and Phenologies in Terrestrial Ecosystems
- Changes in Coastal and Near-Shore Ecosystems
- Changes in Pests and Pathogens
- Changes in Marine Fisheries and Ecosystems
- Changes in Particularly Sensitive Ecosystems
- Ecosystem Services and Expectations for Future Change
- Adequacy of Monitoring Systems

5.2 CHANGES IN DISTRIBUTION AND PHENOLOGIES IN TERRESTRIAL ECOSYSTEMS

As previous chapters have demonstrated, terrestrial ecosystems are already being demonstrably impacted by climate change. Changes in the geographic distribution of species and timing of specific biological processes such as pollination or migration have long been expected because, as is widely known, over the long-term these are often controlled by large-scale patterns in climate. In this section, we examine some of those specific changes as they have been analyzed in the recent literature.

5.2.1 Growing Season Length and Net Primary Production Shifts

There is evidence indicating a significant lengthening of the growing season and higher net primary productivity (NPP) in the higher latitudes of North America, where temperature increases are relatively high. Over the last 19 years, global satellite data indicate an earlier onset of spring across the temperate latitudes by 10–14 days (Zhou et al. 2001; Lucht 2002), an increase in summer photosynthetic activity (Zhou et al. 2001), and an increase in the amplitude of the annual CO₂ cycle (Keeling 1996); climatological and field observations support these findings (Figure 5.1).

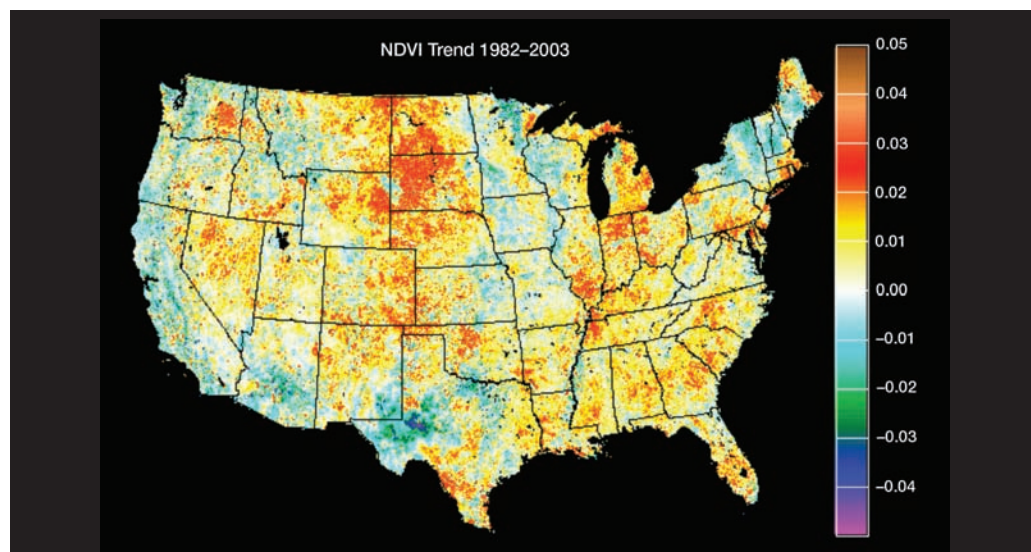


Figure 5.1 Changes in U.S. vegetation observed by satellite (NDVI, or Normalized Difference Vegetation Index) between 1982 and 2003 (NDVI units per year). The NDVI reflects changes in vegetation activity related to climate variability, land-use change, and other influences, and shows substantial trends in much of the conterminous United States. Figure provided by J. Hicke, University of Idaho, based on data from C. Tucker, NASA Goddard Space Flight Center.



In the higher latitudes in Europe, researchers detected a lengthening of the growing season of 1.1 to 4.9 days per decade since 1951, based on an analysis of climate variables (Menzel et al. 2003). Numerous field studies have documented consistent earlier leaf expansion (Wolfe et al. 2005; Beaubien and Freeland 2000) and earlier flowering (Schwartz and Reiter 2000; Cayan et al. 2001) across different species and ecosystem types. Accordingly, NPP in the continental United States increased nearly 10 percent between 1982–1998 (Boisvenue and Running 2006). The largest increases in productivity have been documented in croplands and grasslands of the central United States, as a consequence of favorable changes in water balance (Lobell et al. 2002; Nemani et al. 2002; Hicke and Lobell 2004).

Forest productivity, in contrast, generally limited by low temperature and short growing seasons in the higher latitudes and elevations, has been slowly increasing at less than 1 percent per decade (Boisvenue and Running 2006; Joos et al. 2002; McKenzie et al. 2001; Caspersen et al. 2000). The exception to this pattern is in forested regions that are subject to drought from climate warming, where growth rates have decreased since 1895 (McKenzie et al. 2001) and longer growing seasons have reduced productivity in forested subalpine regions (e.g., Monson et al. 2005; Sacks et al. 2007). Recently, widespread mortality over 12,000 km² of lower-elevation forest in the Southwest is consistent with the impacts of increased temperature and the associated multiyear drought (Breshears et al. 2005) even though previous studies had found productivity at treeline had increased (Swetnam and Betancourt 1998). Disturbances created from the interaction of drought, pests, diseases, and fire are projected to have increasing effects on forests and their future distributions (IPCC 2007). These changes in forests and other ecosystem types will cascade through the trophic structure with resulting impacts on other species.

5.2.2 Biogeographical and phenological shifts

Evidence from two meta-analyses (Root et al. 2003; Parmesan and Yohe 2003) and a synthesis (Parmesan 2006) on species from a broad array of taxa suggests that there is a significant impact from recent climatic warming in the form

of long-term, large-scale alteration of animal and plant populations including changes in distribution (Root and Schneider 2006; Root et al. 2003; Parmesan 2003). If clear climatic and ecological signals are detectable above the background of climatic and ecological noise from a 0.6°C increase in global mean temperature over roughly the last century, by 2050 the impacts on ecosystems are very likely to be much larger (Root and Schneider 2006).

Movement of species in regions of North America in response to climate warming is expected to result in shifts of species ranges poleward, and upward along elevational gradients (Parmesan 2006). Species differ greatly in their life-history strategies, physiological tolerances, and dispersal abilities, which underlie the high variability in species responses to climate change. Many animals have evolved powerful mechanisms to regulate their physiology, thereby avoiding some of the direct influences of climate change, and instead interact with climate change through indirect pathways involving their food source, habitat, and predators (Schneider and Root 1996). Consequently, most distributional studies, which incorporate integrated measures of direct and indirect influences to changes in the climate environment, tend to focus on animals while phenological studies, which incorporate measures of direct influences, focus on plants and insects. Although most studies tend to separate distributional and phenological effects of climate change, it is important to keep in mind that the two are not independent and interact with other changing variables to determine impacts to species (Parmesan 2006). In addition, most of the observed species responses have described changes in species phenologies (Parmesan 2006). This section will cover both by major taxa type.

Parmesan (2006) describes three types of studies documenting shifts in species ranges: (1) those that measure an entire species' range, (2) those that infer large-scale range shifts from observations across small sections of the species' range, and (3) those that infer large-scale range shifts from small-scale change in species abundances within a local community. Although very few studies have been conducted at a scale that encompasses an entire species' range (amphibians (Pounds et al. 1999; Pounds et al. 2006), pikas



(Beever et al. 2003), birds (Dunn and Winkler 1999), and butterflies (Parmesan 2006, 1996)), there is a growing body of evidence that has inferred large shifts in species range across a very broad array of taxa. In an analysis of 866 peer-reviewed papers exploring the ecological consequences of climate change, nearly 60 percent of the 1,598 species studied exhibited shifts in their distributions and/or phenologies over the 20- and 140-year timeframe (Parmesan and Yohe 2003). Field-based analyses of phenological responses of a wide variety of different species have reported shifts as great as 5.1 days per decade (Root et al. 2003) with an average of 2.3 days per decade across all species (Parmesan and Yohe 2003).

5.2.2.1 MIGRATORY BIRDS

For migratory birds, the timing of arrival to breeding territories and over-wintering grounds is an important determinant of reproductive success, survivorship, and fitness. Climate variability on interannual and longer time scales can alter phenology and range of migratory birds by influencing the time of arrival and/or the time of departure. The earlier onset of spring has consequences for the timing of migration and breeding in birds that evolved to match peak food availability (Visser et al. 2006). It should be expected that the timing of migration would track temporal shifts in food availability caused by changes in climate and the advancement of spring.

The phenology of migration to summer and wintering areas may be disrupted for long-distance, continental migrations as well regional local or elevational migrations. Since short-distance migrants respond to changes in meteorological cues whereas long-distance migrants often rely on photoperiod, it has been assumed that the climate signature on changes in phenological cycles would be stronger in short distance than in long-distance migrants (Lehikoinen et al. 2004). If true, this would lead to greater disruption in the timing of migration relative to food availability for long-distance, continental migrants relative to short-distance migrants. Recent studies of long-distance migration provide evidence to the contrary. In a continental-scale study of bird phenology that covered the entire United States and Canadian breeding range of a tree swallow (*Tachycineta bicolor*) from 1959 to 1991, Dunn and Winkler (1999) documented

a 9-day advancement of laying date which correlated with the changes in May temperatures (Winkler et al. 2002; Dunn and Winkler 1999). In a study of the first arrival dates of 103 migrant bird species (long-distant, and very long-distant migrants) in the Northeast during the period 1951–1993 compared to 1903–1950, all migrating species arrived significantly earlier, but the birds wintering in the southern United States arrived on average 13 days earlier while birds wintering in South America arrived four days earlier (Butler 2003). MacMynowski and Root (2007) have found, in a study of 127 species over 20 years of migratory birds that use the migratory flyway through the central United States, that short-range migrants typically respond to temperature alone, which seems to correlate with food supply, while long-range migrants respond more to variation in the overall climate system.

Conversely, in a reversal of arrival order for short- and long-distance passerines, Jonzen et al. (2006) showed that long-distance migrants have advanced their spring arrival into Scandinavia more than short-distance migrants, based on data from 1980 to 2004. Similarly, in a 42-year analysis of 65 species of migratory birds through Western Europe, researchers found autumn migration of birds wintering south of the Sahara had advanced, while migrants wintering north of the Sahara delayed autumn migration (Jenni and Kéry 2003). Finally, a study that combined analysis of spring arrival and departure dates of 20 trans-Saharan migratory bird species to the United Kingdom found an 8-day advance in the arrival and the departure time to the breeding grounds, but with no change in the residence time. The timing of arrival advanced in relation to increasing winter temperatures in sub-Saharan Africa, whereas the timing of departure advanced in response to elevated summer temperatures in their breeding ground (Cotton 2003). But, without an understanding of how this change correlates with phenology of the food resource, it is difficult to discern what the long-term consequences might be (Visser and Both 2005).

As these studies suggest, when spring migration phenology changes, migrants may be showing a direct response to trends in weather or climatic patterns on the wintering ground and/or along

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the migration route, or there may be indirect microevolutionary responses to the selection pressures for earlier breeding (Jonzen et al. 2006). A climate change signature is apparent in the advancement of spring migration phenology (Root et al. 2003), but the indirect effects may be more important than the direct effects of climate in determining the impact on species persistence and diversity. Indeed, there is no *a priori* reason to expect migrants and their respective food sources to shift their phenologies at the same rate. A differential shift will lead to mistimed reproduction in many species, including seasonally breeding birds. There may be significant consequences of such mistiming if bird populations are unable to adapt (Visser et al. 2004). Phenological shifts in migration timing in response to climate change may lead to the failure of migratory birds to breed at the time of abundant food supply (Visser et al. 2006; Visser and Both 2005; Stenseth and Mysterud 2002) and, therefore, may have implications for population success if the shift is not synchronous with food supply availability. Understanding where climate change-induced mistiming will occur and the underlying mechanisms will be critical in assessing the impact of global climate change on the success of migratory birds (Visser and Both 2005). The responses across species will not be uniform across their ranges, and are thus likely to be highly complex and dependent on species-specific traits, characteristics of local microhabitats, and aspects of local microclimates.

5.2.2.1.1 Mismatches and extinctions

Many migratory birds, especially short-range migrants, have adapted their timing of reproduction to the timing of the food resources. A careful examination of food resource availability relative to spring arrival and egg-laying dates will aid in the understanding of impacts of climate change. There is a suite of responses that facilitates an adaptive phenological shift; a shift in egg-laying date or a shift in the period between laying of the eggs and hatching of the chicks. In a long-term study of the migratory pied flycatcher (*Ficedula hypoleuca*), researchers found that the peak of abundance of their food resource (caterpillars) has advanced in the last two decades and, in response, the birds have advanced their laying date. In years with an early caterpillar peak, the hatching date was

advanced and clutch sizes were larger. Populations of the flycatcher have declined by about 90 percent over the past two decades in areas where food for provisioning nestlings peaks early in the season, but not in areas with a late food peak (Both 2006).

Climate change will lead to changing selection pressures on a wide complex of traits (Both and Visser 2005). It is the mistiming of the migration arrival, the provisioning of food resources and the lay dates that drive population declines. Predicting the long-term effects of ecological constraints and interpreting changes in life-history traits require a better understanding of both adaptive and demographic effects of climate change. Environmental stochasticity has the most immediate effect on the risk of population extinction because of its effects on parameters characterizing population dynamics, whereas the long-term persistence of populations is most strongly affected by the specific population growth rate (Saether et al. 2005). Research focused on both will aid in the understanding of the impacts of climate change.

5.2.2.2 BUTTERFLIES

Since temperature determines timing of migration and distribution, it is not surprising that many studies have documented changes in phenology of migration and significant shifts in latitudinal and elevational distribution of butterflies in response to current-day warming. The migration of butterflies in spring is highly correlated with spring temperatures and with early springs. Researchers have documented many instances of earlier arrivals (26 of 35 species in the United Kingdom (Roy and Sparks 2000); 17 of 17 species in Spain (Stefanescu et al. 2004); and 16 of 23 species in central California (Forister and Shapiro 2003)). An analysis of a 113-year record of nine migrating butterflies and 20 migrating moths found increasing numbers of migrants with increasing temperature along the migration route in response to fluctuation in the North Atlantic Oscillation (Sparks et al. 2005).

Butterflies are also exhibiting distributional and/or range shifts in response to warming. Across all studies included in her synthesis, Parmesan (2006) found 30–75 percent of species had expanded northward, less than 20 percent had contracted southward, and the remainder were

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stable (Parmesan 2006). In a sample of 35 non-migratory European butterflies, 63 percent have ranges that have shifted to the north by 35–240 km during this century, and 3 percent that have shifted to the south (Parmesan et al. 1999). In North America, butterflies are experiencing both distributional shifts northward, with a contraction at the southern end of their historical range, and to higher elevations, as climate changes.

In a 1993–1996 recensus of Edith’s checkerspot butterfly (*Euphydryas editha*) populations, Parmesan et al. (1996) found that 40 percent of the populations below 730 meters had become extinct despite the availability of suitable physical habitat and food supply, compared to only 15 percent extinct above the same elevation (Parmesan 1996). Wilson et al. (2007) documented uphill shifts of 293 meters in butterfly species richness and composition in central Spain between 1967–1973 and 2004–2005, consistent with an upward shift of mean annual isotherms, resulting in a net decline in species richness in approximately 90% of the study region (Wilson et al 2007). In Britain, Franco et al. (2006) documented climate change as a driver of local extinction of three species of butterflies and found range boundaries retracted 70–100 km northward for *Aricia artaxerxes*, *Erebia aethiops* and 130–150 meters uphill for *Erebia ephron*; these changes were consistent with estimated latitudinal and elevational temperature shifts of 88 km northward and 98 meters uphill over the 19-year study period.

An investigation of a skipper butterfly (*Atalopedes campestris*) found that a 2–4°C warming had forced a northward range expansion over the past 50 years, driven by increases in winter temperatures (Crozier 2003, 2004). A study investigating the altitudinal and latitudinal movements of 51 British butterfly species related to climate warming found that species with northern and/or montane distributions have disappeared from low elevation sites, and colonized sites at higher elevations consistent with a climate warming, but found no evidence for a systematic shift northward across all species (Hill et al. 2002). A subsequent modeling exercise to forecast potential future distributions for the period 2070–2099 projects 65 and 24 percent declines in range sizes for northern and southern species, respectively (Hill et al. 2002).

5.2.2.2.1 Mismatches and extinctions

As is the case for birds, changes in timing of migrations and distributions are likely to present resource mismatches that will influence population success and alter the probability of extinction. Predictions of climate-induced population extinctions are supported by geographic range shifts that correspond to climatic warming, and a few studies have linked population extinctions directly to climate change (McLaughlin et al. 2002; Franco et al. 2006). As populations of butterfly species become isolated by habitat loss, climate change is likely to cause local population extinctions.

Modeling of butterfly distribution in the future under climate change found that while the potential existed to shift ranges northward in response to warming, lack of habitat availability caused significant population declines (Hill et al. 2002). Similarly, phenological asynchrony in butterfly-host interactions in California led to population extinctions of the checkerspot butterfly during extreme drought and low snowpack years (Singer and Harter 1996; Thomas et al. 1996; Ehrlich et al. 1980; Singer and Ehrlich 1979). A modeling experiment of two populations of checkerspot butterfly suggested that decline of the butterfly was hastened by increasing variability in precipitation associated with climate change. The changes in precipitation amplified population fluctuations leading to extinction in a region that allowed no distributional shifts because of persistent habitat fragmentation (McLaughlin et al. 2002).

Whether there is evidence of actual evolutionary change in insects in response to climate change is presently unclear. A study of the speckled wood butterfly (*Pararge aegeria*) in England found that evolutionary changes in dispersal were associated with reduced investment in reproduction, which affects the pattern and rate of expansion at range boundaries (Hughes 2003). But this result is only suggestive of a potential interaction of the factors that control the pattern and rate of expansion at range boundaries and the response to a changing climate system.

5.2.2.3 MAMMALS

Mammals are likely to interact with climate through indirect pathways involving their food source, habitat, and predators, perhaps more

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strongly than through direct effects on body temperature (Schneider and Root 2002), although Humphries et al. (2004) also demonstrate that overall bioenergetic considerations are important, especially in northern species. Over periods of geological time, mammals' geographic distributions have been demonstrated to respond to long-term changes in climatic conditions. Guralnick (2007) has shown that for mammal species of long duration in North America (i.e., those that have had good distributional records in both the Late Pleistocene and modern times), flatland species had large northward changes in the southern edge of their distributions as a response to the warming of the interglacial period. Montane species showed more upward and northward shifts during this time period, with the consequence that their overall ranges appeared to expand rather than to simply to track to new climatic conditions. Guralnick's results are not specific to the problems posed by recent changes in the physical climate system, or to projected changes, because these are happening much faster than interglacial warming. However, they are indicative of the direction of change that even mammal species are expected to undergo as the physical climate system changes.

Guralnick (2007) was not able to specify mechanisms by which such range adjustments occurred in his statistical analysis of existing data. It is likely, however, that climate change will alter the distribution and abundance of northern mammals through a combination of direct, abiotic effects (e.g., changes in temperature and precipitation) and indirect, biotic effects (e.g., changes in the abundance of resources, competitors, and predators). The similar results of Martinez-Meyer et al. (2004) suggest that the methods of modeling climate change response in mammals' geographic ranges as a function of changes in climate should provide robust results, at least over time periods that are long enough to allow the individual species to respond. In the United States, the General Accounting Office (2007) has identified several examples of mammals in the system of U.S. public lands for which the consequences of climate change are expected to be noticeable – among these are grizzly bears, bighorn sheep, pikas, mountain goats, and wolverines. In each case, the responses to climate-driven changes do not appear to be

direct physiological responses to temperature and precipitation as much as they are responses to changes in the distribution of habitats, and in particular the compression and loss of habitats at higher elevations in mountainous areas.

The pika is a particularly interesting example, as several populations appeared to be extirpated in the United States when resampled during the 1990s (Beever et al. 2003). The pika lives in talus habitats at high elevations in mountainous areas and has a very short active season during the growing season, when it gathers grass for food for survival during the winter months. Seven out of 25 previously reported (early 20th century) populations appeared to have disappeared. Beever et al. (2003) concluded that local extirpations were best explained in a multifactorial way, and that changes in climatic factors that affected available habitat and food supply were one of the important factors. Similar phenomena have been reported for a different species of pika in Xinjiang Province in China (Li and Smith 2005). Climate effects are known to be important in both situations.

5.2.2.4 AMPHIBIANS

Many amphibian species are known to be undergoing rapid population declines, and there has been considerable discussion in the literature about the degree to which climate change might be involved (Stuart et al. 2004; Pounds, 2001; Carey et al. 2001). Carey et al. (2001) constructed a large database that included sites at which amphibian declines had been documented, and others at which they had not been. There were correlations of global environmental change in the climate system with evidence of decline, but their conclusion was that it was unlikely that the change in climate itself was the principal source of mortality in those populations. Rather, they hypothesized that changes in the global environment may have acted as an enabling factor, leading to other, more immediate causes of pathology and population declines.

There is some evidence that amphibian breeding is occurring earlier in some regions, and that global warming is likely the driving factor (Beebee 2002; Blaustein et al. 2001; Gibbs and Breisch 2001). Some temperate-zone frog and toad populations show a trend toward breeding earlier, whereas others do not (Blaustein et al. 2001). Statistical tests (Blaustein et al. 2002)



indicate that half of the 20 species examined by Beebee (1995), Reading (1998), Gibbs and Breisch (2001), and Blaustein et al. (2001) are breeding earlier. Of the half not exhibiting statistically significant earlier breeding, they are showing biologically important trends toward breeding earlier that, if continued, will likely become statistically significant (Blaustein et al. 2002). When taken together, these important data suggest that global warming is indeed affecting amphibian breeding patterns in many species. There is, however, marked unevenness of climate-change effects on amphibian breeding. For example, Fowler's Toad, *Bufo fowleri*, a late breeder, has bred progressively later in spring over the past 15 years on the north shore of Lake Erie (Blaustein 2001).

Kiesecker et al. (2001), in their study of amphibian populations in the U.S. Pacific Northwest, which are declining, point out that there are potential interactions among a number of environmental factors, including interannual climate variability, exposure to UV-B radiation causing egg and embryo mortality, and persistent climate change. It is very difficult to use field studies by themselves to sort out the relative contributions of each. However, two of the best-known examples of a climate-mediated rapid decline in amphibian populations are provided by the Golden Toad and Harlequin Frog, both of which are found in Costa Rica in the Monte Verde cloud forest. Pounds and Crump (1994) documented disappearances of the previously abundant populations of both animals as a consequence of climate-mediated stresses, in this case initially with the severe El Niño episode of 1987.

The discovery of a new disease caused by a previously unknown chytrid fungus has complicated the picture somewhat. But several studies, summarized recently by Wake (2007), conclude that even with the presence of the chytrid fungus, climate change has clearly had an impact in many of the well-documented amphibian declines and extinctions. Wake (2007) also points out that in at least one case, declines have also been found in nearby lizard species in the same habitats, although lizards are not known to be susceptible to the chytrid fungus.

5.2.3 Geographical and Distributional Responses of Plants

In this assessment, the chapters on forests, arid lands and agriculture largely consider changes in either individual plant or ecosystem processes – e.g., photosynthesis and transpiration, soil respiration, allocation of carbon to above- and below-ground components of ecosystems, and overall carbon capture and sequestration. Those chapters, as well as a subsequent section in this chapter, also consider disturbances of different types as they affect ecosystem composition and processes, including fire, pests, and invasive species.

But a fundamental tenet of ecology is that the geographical distribution of plant species is determined in large part by climatic conditions. It is therefore natural to ask whether there is evidence of changes in plant distributions as a result of climate variability and change as well as in plant/ecosystem functional performance. It is also important to understand the degree to which changes might be expected to occur in the future in plant distributions, both at the functional level and at the individual species level.

Iverson and Prasad (2001) provide a comprehensive review of methods to determine both empirical and modeling approaches to understanding how vegetation responds to changes in climate. They point out that paleoecological observations demonstrate not only that tree species did respond to long-term changes in climate, but that they did so individually, leading to new combinations of species than previously existed. Iverson and Prasad (2001) show the results of statistical modeling for the potential future distribution of tree species in the eastern United States, using several different model-derived climate scenarios. Out of a pool of 80 common tree species, they conclude that some forest types (e.g., oak-hickory) are likely to expand, while others (maple-beech-birch) will likely contract, and still others (spruce-fir) are likely to be extirpated within the United States. Their results appear to be robust to different climate scenarios, and are consistent with what we know about these species in the paleo record.



Dirnbock et al. (2003) document both the existing relationships between the distribution of 85 alpine plant species in Europe and climate and land-use variables. They then use simple projections of both land-use and climate variables to assess the likely responses of these plants to changes in climate over the next several decades, concluding that climate forcings and land-use changes will interact substantially to determine future distributions.

Burkett et al. (2005) and the Government Accounting Office (2007) provide a number of current examples of vegetation changes that are clearly the result of responses to variability in climatic forcings, and supply mechanisms for those changes. Examples include changes in wetland vegetation in Michigan that occur as a result of the interaction of water withdrawals and drought occurrence, and extension of tree line in U.S. sub-Arctic and Arctic regions – the latter clearly responding to the observed large regional warming of the past several decades.

A growing community of ecosystem modelers using Dynamic Global Vegetation Models has developed a capability to simulate the changes in potential natural vegetation as a function of changes in the physical climate system (Cramer et al. 2001). These simulations can be used to investigate the potential for future changes in the distribution of plant functional types, and serve as a guide for assessing risk. Scholze et al. (2006) provide one such example, concluding that for an analysis that considered 16 different climate/atmospheric composition scenarios, there was a large risk of considerable change in forested ecosystems and freshwater supply in many regions around the world, including the eastern United States. However, such analyses do not include land management or land-use processes, and thus establish the potential for change, rather than serving as quantitative predictions of change.

5.3 CHANGES IN COASTAL AND NEAR-SHORE ECOSYSTEMS

Coastal and marine ecosystems are tightly coupled to both the adjacent land and open ocean ecosystems and are thus affected by climate in multiple ways. In the tropics, coral bleaching and disease events have increased, and in the Atlantic, hurricane intensity and destructive potential has increased. In temperate regions, there are demonstrated range shifts and possible alterations of ocean currents and upwelling strength. In the Arctic, there have been dramatic reductions in sea ice extent and thickness, as well as related coastal erosion. Marine species were the first to be listed as threatened species due to physical stresses that are clearly related to variability and change in the climate system (Federal Register 2006). Coastal and near-shore ecosystems are vulnerable to a host of climate change-related effects, including increasing air and water temperatures, ocean acidification, altered terrestrial run-off patterns, altered currents, sea level rise, and altered human pressures due to these and other related changes (such as development, shipping, pollution, and anthropogenic adaptation strategy implementation). This section will discuss some of the most prominent effects of climate change observed to date in the coastal and near-shore regions of the United States, with some consideration given to applicable examples from other parts of the world.

5.3.1 Coral Reefs

Tropical and subtropical coral reefs around the world have been known for some time to be under a wide variety of stresses, some of them related to changes in the climate system, and some not (Bryant et al. 1998). The United States has extensive coral reef ecosystems in both the Caribbean Sea and the Pacific Ocean. Coral reefs are very diverse ecosystems, home to a complex of species that support both local and global biodiversity and human societies. It has been estimated that coral reefs provide \$30 billion in annual ecosystem service value (Cesar et al. 2003), including both direct market values of tourism, and estimates of the market value of other services, such as provision of habitat for fish breeding, and protection of coastline. A



Coastal and near-shore ecosystems are vulnerable to a host of climate change-related effects, including increasing air and water temperatures, ocean acidification, altered terrestrial run-off patterns, altered currents, sea level rise, and altered human pressures due to these and other related changes (such as development, shipping, pollution, and anthropogenic adaptation strategy implementation).



variety of regional estimates of economic value (Cesar 2000) have also been made that show substantial variation in their totals, depending in part on which services are taken into consideration. In some small developing countries, coral reefs may supply substantial fractions of total economic return through their contribution to tourism and as habitat for coastal fisheries; even in the United States and Australia where coral reefs provide small fractions of the total revenue, they generate many billions of dollars and can be very important in regional economies (Hoegh-Guldberg et al. 2007).

Corals and tropical regions where they live are experiencing increasing water temperatures, a reduction in surface water pH (Ravens et al. 2005), and there is evidence for increasing storm intensity (Emmanuel 2005), as well as a host of other ongoing challenges created as a result of development/tourism, fishing, and pollution. The effects of climate change in marine systems is highlighted by the 2006 proposed listing as Threatened under the Endangered Species Act of two species of corals in the Caribbean (Federal Register 2006). The major threats that motivated the proposed listings of Elkhorn (*Acropora palmata*) and Staghorn (*A. cervicornis*) corals were disease, elevated sea surface temperatures, and hurricanes – all of which relate to climate change and its effects (Muller et al. 2007; Mann and Emmanuel 2006).

5.3.1.1 INCREASING TEMPERATURE AND ACIDIFICATION OF OCEAN WATERS

The El Niño-Southern Oscillation (ENSO) event of 1982–83 marked the first contemporary broad-scale coral reef bleaching and mortality event (Glynn 1984). Since then, there have been subsequent bleaching events including the 1997–98 ENSO. The rate of occurrence (annually in some cases), and almost global scale since the early 1980s is in stark contrast to the trend of the first half of the century in which bleaching events were localized and linked to local events (D’Elia et al. 1991; Glynn 1993). From 1876–1979 only three bleaching events were recorded, whereas 60 are on record between 1980 and 1993 (Glynn 1993). Bleaching is considered to be a stress response caused primarily by increased water temperature (Glynn 1993) and synergistically enhanced by increased irradiance levels (Fitt and Warner 1995; Jokiel

and Coles 1990; Lesser et al. 1990). Corals become stressed if exposed to slight increases in water temperature – increases of only 1 to 2°C over the average annual thermal maxima for days to weeks can result in a bleaching event (Hoegh-Guldberg 1999). Field studies have correlated increased temperatures with mass bleaching events (Brown 1997; Hoegh-Guldberg et al. 1997; Glynn 1993). Additionally, the National Oceanic and Atmospheric Administration (NOAA) “Hotspot” program (Goreau and Hayes 1994) predicted bleaching for most geographic regions where bleaching occurred in 1998, adding further weight to the assessment that elevated temperature is the primary trigger for bleaching (Hoegh-Guldberg 1999). The final effect of the 1997–98 bleaching event has been assessed, with estimates indicating that 10–16 percent of world’s living coral reefs died during this event. In the western Indian Ocean, coral reefs lost up to 46 percent of living, reef-building corals (Hoegh-Guldberg 2005).

In 2005, the Caribbean basin saw unprecedented water temperatures and some dramatic bleaching, followed by coral disease and mortality. The most dramatic monitored bleaching took place in the U.S. Virgin Islands, where National Park monitoring showed that at some sites 90 percent of the coral bleached. Afterward there appeared to be a period of recovery as water temperatures decreased. Unfortunately, this was short-lived as disease appeared in November of the same year on many of the previously bleached corals. To date there is an estimated 50 percent combined mortality from bleaching and disease in the Virgin Island National Park surveys. As of yet, there are no reports of recovery as amounts of mortality continue to increase (Eakin et al., *in press, accepted*). In the Florida Keys, equally massive bleaching was anticipated when temperatures exceeded 9-degree heating weeks in late August 2005 (NOAA Coral Reef Watch), and in fact some bleaching was observed. But the arrival of hurricanes Katrina and Rita reduced water temperatures and appear to have provided some respite for corals in the Keys. However, the same pattern of disease was seen in the Keys in those corals that did bleach, with bleaching setting in around mid-August, followed by disease in early September (Brandt, *in press, accepted*).

Both intensities and frequencies of bleaching events clearly driven by warming in surface waters have increased substantially over the past 30 years (Hughes et al. 2003). At least 30 percent of reefs globally have been severely damaged, and relatively simple projections based on temperature changes alone suggest that within the next several decades, as many as 60 percent of the world's reefs could be damaged or destroyed (Hughes et al. 2003). While there is some evidence of short-term recovery, in many locations the frequency of bleaching events could become nearly annual within several decades under a variety of reasonable climate scenarios (Donner et al. 2005). Such changes would be significantly more rapid and pose significant problems for coral reef management on a global scale (Hughes et al. 2003; Pandolfi et al. 2003; Hoegh-Guldberg et al. 2007).

Additionally, as CO₂ concentrations increase in the atmosphere, more CO₂ is hydrolyzed in the surface waters of the world's oceans, leading to their acidification (Orr et al. 2005; Hoegh-Guldberg et al. 2007) (Figure 5.2). The chemical reactions governing the dissolution of CaCO₃ in surface waters, and therefore the availability of material for building corals' calcium carbonate

skeletons (as well as those of other calcifying organisms) are pH-dependent, and increases in acidity can lead to decreases in available CaCO₃ (Yates and Halley 2006). During the past 200 years, there has been a 30 percent increase in hydrogen-ion concentration in the oceans, and it is anticipated that this will increase by 300 percent by the end of this century (Ravens et al. 2005). There is evidence from site-specific studies (Pelejero et al. 2005) that in the Pacific Ocean there is natural decadal variability in the pH levels that individual reefs actually experience, and that the variability matches well with Interdecadal Pacific Oscillation variability.

However, even though some reef species may be more resistant to increases in acidity than others, the longer-term decreases in ocean pH due to increased atmospheric CO₂ concentrations may be occurring much more rapidly than in the recent history. And, when these long-term trends occur in phase with the IPO, even relatively resistant reefs would be exposed to extremely low pH levels that they have not experienced before. There are predictions that oceans could become too acidic over the long term for corals – as well as other species – to produce calcium carbonate skeletons (Caldeira and Wickett 2003;

Atmospheric CO₂ Effects on Coral Reefs

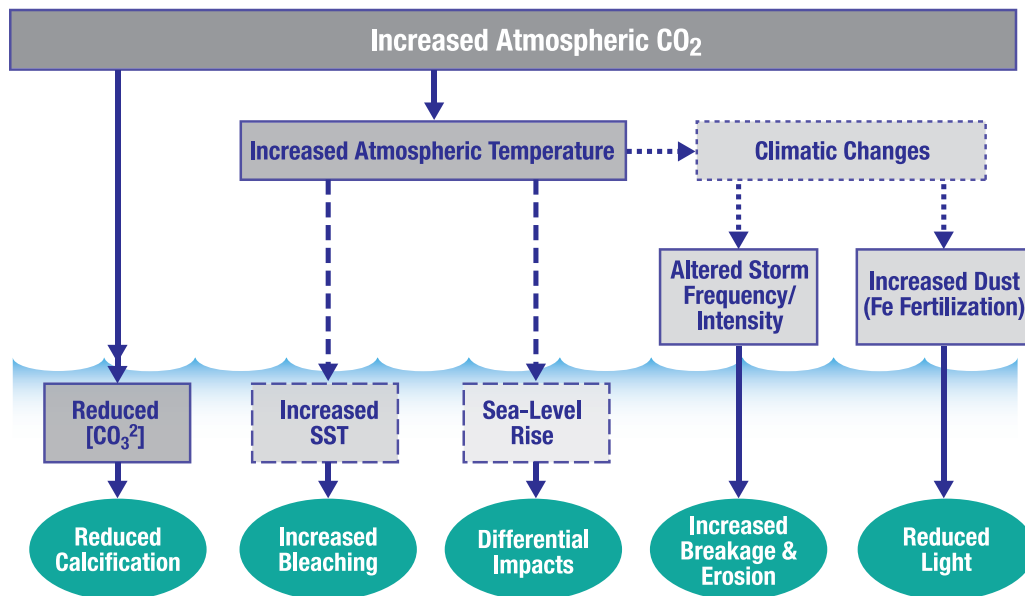


Figure 5.2 The figure above depicts various direct and indirect effects of changes in atmospheric CO₂ concentrations on coral reef ecosystems. Solid lines indicate direct effects, dashed lines indicate indirect effects, and dotted lines indicate possible effects. Fe = iron; SST = sea surface temperature; CO₃²⁻ = carbonate ion.



Hoegh-Guldberg 2005; Kleypas et al. 1999). More recent reviews of both experimental studies, modeling projections, and field observations suggest that the combination of changes in ocean surface temperatures, increasing ocean acidity, and a host of other stresses could bring coral reef ecosystems to critical ecological tipping points (Groffman et al. 2006) within decades rather than centuries, and that some regions of the ocean are already near that point from a biogeochemical perspective (Orr et al. 2005; Hoegh-Guldberg 2007).

Increasing sea surface temperatures are expected to continue as global temperatures rise. It is possible that these warmer waters are also increasing the intensity of the tropical storms in the region (Mann and Emmanuel 2006; Srivier and Huber 2006; Elsner 2006; Hoyos et al. 2006). As global temperatures rise, sea level will continue to rise providing additional challenges for corals. Increasing depths change light regimes, and inundated land will potentially liberate additional nutrients and contaminants from terrestrial sources, especially agricultural and municipal.

5.3.2 Coastal Communities and Ecosystems

5.3.2.1 WETLANDS AND BARRIER ISLANDS

The marine-terrestrial interface is vitally important for biodiversity as many species depend on it at some point in their life cycles, including many endangered species such as sea turtles and sea birds. In addition, coastal areas provide a wide variety of ecosystem services, including breeding habitat and buffering inland areas from the effects of wave action and storms (MEA 2005). There is a wide variety of different types of habitat in coastal margins, from coastal wetlands, to intertidal areas, to near-shore ecosystems, all of which are subject to a variety of environmental stresses from both the terrestrial, inland environments and from oceanic environments (Burkett et al. 2005). The additional proximity of large numbers of people makes coastal regions extremely important natural laboratories for global change.

Mangroves and sea grasses protect coastlines from erosion, while also protecting near-shore environments from terrestrial run-off. Sea level

rise, increased coastal storm-intensity and temperatures contribute to increased vulnerability of mangrove and sea grass communities (e.g., Alongi 2002). It has been suggested that the dominant sea grass species (*Zostera marina*) is approaching its thermal tolerance for survival in the Chesapeake Bay (Short and Neckles 1999). It has also been estimated that a 1-meter increase in sea level would lead to the potential inundation of 65 percent of the coastal marshlands and swamps in the contiguous United States (Park et al. 1989). In addition to overt loss of land, there will also be shifts in quality of habitat in these regions. Prior to being inundated, coastal watershed will become more saline due to saltwater intrusion into both surface and groundwater. Burkett et al. (2005) provide several excellent examples of documented and potential rapid, non-linear ecological responses in coastal wetlands to the combination of sea-level rise, local subsidence, salinity changes, drought, and sedimentation. Of particular concern in the United States are coastlines along the Gulf of Mexico and the Southeast Atlantic, where the combination of sea level rise and local subsidence has resulted in substantially higher relative, local rates of sea-level rise than farther north on the Atlantic Coast, or on the Pacific Coast (Burkett et al. 2005). In Louisiana alone, more than 1/3 of the deltaic plain that existed in the beginning of the 20th century has since been lost to this combination of factors. In the Gulf of Mexico and the South Atlantic, the ecological processes that lead to accretion of wetlands and continued productivity (Morris et al. 2002) have not been able to keep pace with the physical processes that lead to relative rising sea level (Burkett et al. 2005).

Barrier islands are particularly important in some regions where vulnerability to sea level rise is acute. In the northwest Hawaiian Islands, which were designated a National Monument in 2006, sea level rise is a threat to endangered beach nesting species and island endemics, including green sea turtles, Hawaiian monk seals, and the Laysan finch (Baker et al. 2006). Another example of an endangered island-locked species is the Key Deer, which is now limited to living on two islands in the Florida Keys. Their habitat is also at risk with most of the Keys at less than two meters above sea level. Median



sea level rise coupled with storm surges would inundate most of the available habitat either permanently or episodically, further threatening this endangered species.

5.3.2.2 ROCKY INTERTIDAL ZONES

Rocky intertidal habitats have been studied extensively with respect to their observed and potential responses to climate variability and change, both in Europe and in the United States (Helmuth et al. 2006; Mieszkowska et al. 2007; Mieszkowska et al. 2005; Bertness et al. 1999; Sagarin et al. 1999; Thompson et al. 2002; Mieszkowska et al. 2006; Barry et al. 1995). These systems react quite differently from wetlands because of the large differences in substrates. Nevertheless, the typical biota of gastropods, urchins, limpets, barnacles, mussels, etc., show reproductive, phenological, and distributional responses, similar in kind to responses of birds, butterflies, and mammals reported earlier in this chapter. However, Helmuth et al. (2006) point out that range shifts of up to 50 kilometers per decade have been recorded for intertidal organisms – far faster than documented for any terrestrial species to date.

Responses include reacting to changes in the thermal habitat, which results in heat stress, and subsequent low growth rates and early, stress-induced spawning of mussel species in New Zealand (Petes et al. 2007). Long time-series of observational data across several quite different taxonomic groups in the British Isles show consistent trends for species in response to strong regional warming trends observed since the 1980's, including: range extensions of northern species into previously colder waters; some range extension eastward of southern species into the English channel; a few species with southern range retractions; and several southern species showing earlier reproduction, greater survival rates, and faster growth rates than northern species (Mieszkowska et al. 2005). These responses are extremely similar to the biological responses shown by rocky intertidal species in the United States in several different locations (Bertness et al. 1999; Helmuth et al. 2006; Barry et al. 1995; Sagarin et al. 1999) on both the Pacific and Atlantic coasts. There is some suggestion in Europe that there could be

food-web level effects on the supply of food for shore birds, but interactions among shore bird predators, gastropods and other rocky intertidal organisms, and algal cover are complex and extremely difficult to predict (Kendall et al. 2004).

Thompson et al. (2002), Helmuth et al. (2005) and Helmuth et al. (2006) all point out that the observational base of responses of intertidal organisms to changes in climate is well enough understood that reasonable projections of future change can be made. However, knowledge of the particular physiological mechanisms for the individual species' responses is especially important (Helmuth et al. 2005) in order to distinguish the reasons for the variation in responses, and in order to understand how climate changes operate in these systems in the presence of other physical and biological stresses.

Because of its importance as a contributing stress to coastal and intertidal habitats, projections of mean sea-level rise have been important to understand. Projections for sea level rise by 2100 vary from 0.18 to 0.59 m ($\pm 0.1-0.2$) (IPCC 2007) to 0.5 to 1.4 m (Rahmstorf 2007). Some observational evidence suggests that recent IPCC estimates may be conservative and underestimate the rate of sea level rise (Meehl et al. 2007). The IPCC projection of 18–59 cm in this century assumes a negligible contribution to sea level rise by 2100 from loss of Greenland and Antarctic ice. Melting of the Greenland ice sheet has accelerated far beyond what scientists predicted even just a few years ago, with a more than doubling of the mass loss from Greenland due to melting observed in the past decade alone (Rignot and Kangaratnam 2006). The acceleration in the rate of melt is due in part to the creation of rivers of melt water, called *imoulines*, that flow down several miles to the base of the ice sheet, where they lubricate the area between the ice sheet and the rock, speeding the movement of the ice toward the ocean. Paleoclimatic data also provide strong evidence that the rate of future melting and related sea-level rise could be faster than previously widely believed (Overpeck et al. 2006).



5.4 CLIMATE CHANGE, MARINE FISHERIES AND MARINE ECOSYSTEM CHANGE

The distribution of fish and planktonic species are also predominately determined by climatic variables (Hays et al. 2005; Roessig et al. 2004) and there is recent evidence that marine species are moving poleward, and that timing of plankton blooms is shifting (Beaugrand et al. 2002; Hays et al. 2005; Richardson and Schoeman 2004). Similar patterns have been observed in marine invertebrates and plant communities

(Beaugrand et al. 2002; Sagarin et al. 1999), Southward et al. (1995) document extensive movement of ranges and distributions of both warm and cold-water species of fish and other marine life around the British Isles and northern Europe over the past several decades, with long-time series of data from fish landings. They point out that much of the original research on fisheries biology in these regions took place from the 1930s–1970s, a period of relative constancy in the marine climate system in these regions. Changes in distributions since then appear to be much more pronounced.

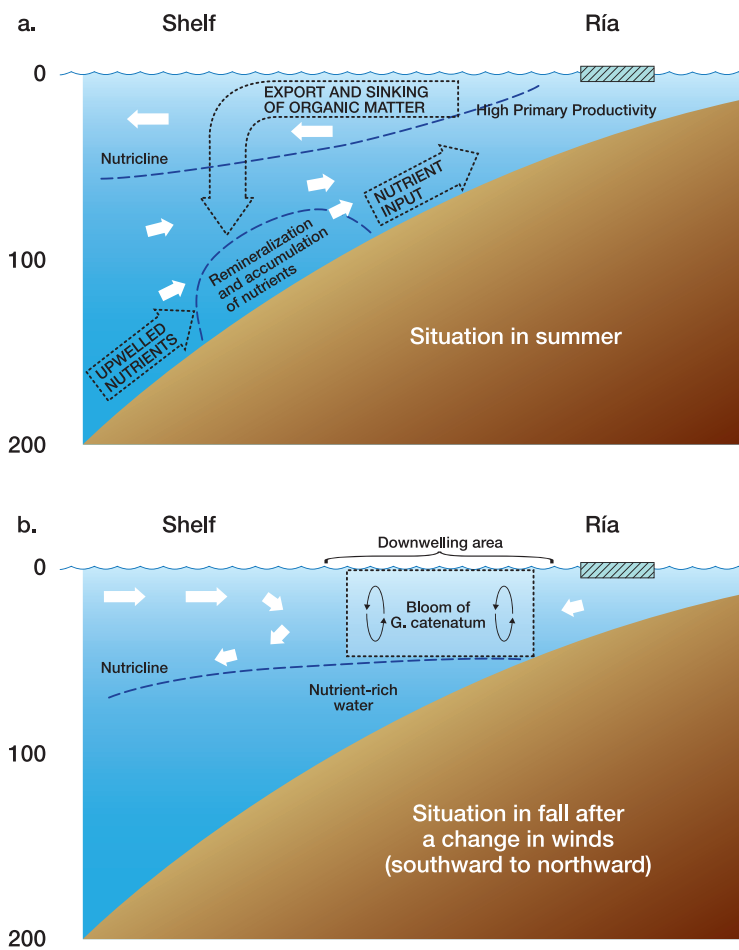


Figure 5.3 Diagram of nutrient dynamics. a) Summer: a profile view of a ria and the adjacent continental shelf, illustrating the “loop” consisting of upwelling-enriched primary production, which leads to export, sinking, and accumulation on the bottom of particulate organic matter. This organic matter decays and remineralizes, enriching the waters beneath the nutricline. b) Fall: After fall relaxation of upwelling, lighter oceanic surface water collapses toward the coast, producing a zone of downwelling in the ria. This depresses the nutricline and cuts off upwelling-produced enrichment of the photic zone. Vertically migrating dinoflagellates may access the nutrient pool beneath the nutricline and transport them upward to levels of higher illumination, where they can use them to support photosynthesis. From Bakun 1996.

Similar phenomena have been documented in Europe for Arctic and Norwegian cod in the Barents Sea (Dippner and Ottersen 2001), and Atlantic cod (Drinkwater 2005), where spawning, survival, and growth rates are affected in predictable ways by ocean temperature anomalies. In each case, the climate variability analyzed is tied to particular oscillations in the physical climate system (e.g., the North Atlantic Oscillation for cod), or to longer-term changes in climate. Fields et al. (1993) provide a general overview of the factors associated with the marine ecosystem responses to climate change. As in other systems examined in this report, the particular biological mechanisms of species responses are important in determining overall patterns. In addition, Hsieh et al. (2005) show that these large marine ecosystems are intrinsically non-linear, and thus subject to extremely rapid and large changes in response to small environmental forcings.

In coastal regions, decreased upwelling can decrease nutrient input to surface waters, reducing primary productivity (Soto 2002; Field et al. 2001). The food-web-level effects that such changes cause have been documented off the coast of Southern California after an abrupt, sustained increase in water temperature in the 1970s (Field et al. 1999). Conversely, climate change may alter wind patterns in ways that accelerate offshore winds and thus upwelling (Bakun 1990) (Figure 5.3).

Seven large marine ecosystems (LMEs) are recognized for U.S. waters: eastern Bering Sea, Gulf of Alaska, California Current, Gulf

of Mexico, southeast U.S. continental shelf, northeast U.S. continental shelf and the greater Hawaiian Islands. Each is being studied to varying degrees with regard to the impacts of climate variability and change on ecosystem structure, biodiversity and marine fisheries. Much of the research in these systems has been carried out by U.S. and Canadian scientists associated with the International Geosphere-Biosphere Programme GLOBAL Ocean ECOSystem Dynamics (IGBP-GLOBEC), or by scientists following GLOBEC standards. The GLOBEC model focuses on study of the coupling of physical forcing and biological response in fisheries-rich ecosystems, and is detailed at www.globec.org. This approach has been taken due to the tight coupling between physics and biology in the oceans as compared to terrestrial ecosystems (Henderson and Steele 2001).

It has been well established that the large basin-scale atmospheric pressure systems that drive basin scale winds can suddenly shift location and intensity at interannual-to-decadal time scales, with dramatic impacts on winds and ocean circulation patterns. These low frequency oscillations are known as the North Atlantic Oscillation (NAO), the Pacific Decadal Oscillation (PDO), and the El Niño-Southern Oscillation (ENSO). Perhaps the greatest discovery of the past 10 years is that these shifts have dramatic impacts on marine ecosystems.

The NAO has been strongly positive since the 1980s. Increases in the strength of the winds have resulted in dramatic impacts on Northeast Atlantic ecosystems. For instance, increased flow of oceanic water into the English Channel and North Sea has contributed to a northward shift in the distribution of zooplankton such that the zooplankton communities are dominated by warm water species (Beaugrand, 2004) with concomitant changes in dominance in fish communities from whiting (hake) to sprat (similar to a herring). Similar ecosystem shifts in the Baltic Sea have occurred where drastic changes in both zooplankton and fish communities have been observed (Kenny and Mollman 2006). Linkages between the NAO, zooplankton and fisheries have also been described for the Northwest Atlantic waters off eastern Canada and the United States. The recovery of the codfish populations, which collapsed in the early 1990s (presumably

as a result of overfishing), may be difficult due to changes in the structure of forage and food chains (Pershing and Green 2007).

In the North Pacific, the PDO refers to the east-west shifts in location and intensity of the Aleutian Low in winter (Mantua et al. 1997). Widespread ecological changes have been observed including increased productivity of the Gulf of Alaska when the PDO is in positive phase, resulting in dramatic increases in salmon production (Mantua et al. 1997), and a reversal of demersal fish community dominance from a community dominated by shrimps to one dominated by pollock (Anderson and Piatt 1991). Associated changes to the California Current ecosystem include dramatic decreases in zooplankton (McGowan et al. 1998) and salmon (Pearcy 1991) when the PDO changed to positive phase in 1977. There is also evidence that the large oscillations in sardine and anchovy populations are associated with PDO shifts, such that during positive (warm) phases, sardine stocks are favored but during negative (cool) phases, anchovy stocks dominate (e.g., Chavez et al. 2003).

ENSO is another major driver of climate variability. El Niño events negatively impact zooplankton and fish stocks resulting in a collapse of anchovy stocks in offshore ecosystems of Peru. Loss of anchovies, which are harvested for fish meal, affect global economies because fish meal is an important component of chicken feeds as well high-protein supplements in aquaculture feed. In waters off the west coast of the United States, plankton and fish stocks may collapse due to sudden warming (by 4–10°C) of the waters as well as through poleward advection of tropical species into temperate zones. Many of the countries most affected by ENSO events are developing countries in South America and Africa, with economies that are largely dependent upon agricultural and fishery sectors as a major source of food supply, employment, and foreign exchange.

5.4.1 Other climate-driven physical forces that affect marine ecosystems

The California Current (CC) example represents an excellent case study for one Large Marine Ecosystem. The CC flows in the North

In coastal regions, decreased upwelling can decrease nutrient input to surface waters, reducing primary productivity



Pacific Ocean from the northern tip of Vancouver Island (Canada), along the coasts of Washington, Oregon and California, midway along the Baja Peninsula (Mexico) before turning west. For planktonic organisms and some fish species, the northern end of the Current is dominated by sub-arctic boreal fauna whereas the southern end is dominated by tropical and sub-tropical species. Faunal boundaries, i.e., regions where rapid changes in species composition are observed, are known for the waters between Cape Blanco, Oregon/Cape Mendocino, California, and in the vicinity of Point Conception, California. Higher trophic level organisms often take advantage of the strong seasonal cycles of production in the north by migrating to northern waters during the summer to feed. Animals that exhibit this behavior include pelagic seabirds such as black-footed albatross and sooty shearwaters, fishes such as Pacific whiting and sardines, and gray and humpback whales.

5.4.2 Observed and Projected Impacts

Based on long-term observation records, global climate models, regional climate models, and first principles, there is a general consensus on impacts of climate change for the United States with regard to climate modes, biophysical processes, community and trophic dynamics and human ecosystems. The type, frequency and intensity of extreme events are expected to increase in the 21st century, however Meehl et al. (2007) suggest that there is no consistent indication of discernable changes in either the amplitude or frequency of ENSO events over the 21st century (Meehl et al. 2007). Climate models from the fourth IPCC assessment project roughly the same timing and frequency of decadal variability in the North Pacific under the impacts of global warming. By about 2030, it is expected that the *minima* in decadal regimes will be *above* the historical mean of the 20th century (i.e., the greenhouse gas warming trend will be as large as natural variability). Regional analyses suggest that for California, temperatures will increase over the 20th century with variable precipitation changes by region (Bell et al. 2004), which is consistent with global projections (Tebaldi et al. 2006).

Among other findings, IPCC assessment results for the United States suggest there will be a general decline in winter snowpack with earlier snowmelt triggered by regional warming (Hayhoe et al. 2004; Salathé 2005).

Additionally, warmer temperatures on land surfaces, contributing to low atmospheric pressure combined with ocean heating may contribute to stronger and altered seasonality of upwelling in western coastal regions (Bakun 1990; Snyder et al. 2003). Migration patterns of animals within the California Current (e.g., whiting, sardines, shearwaters, loggerhead turtles, Grey Whales) may be altered to take advantage of feeding opportunities. Recent disruptions of seasonal breeding patterns of a marine seabird (Cassin's Auklet) by delayed upwelling have been reported by Sydeman et al. (2006).

Warmer ocean temperatures will contribute to changes in upwelling dynamics and decreased primary production along the California Current. Global declines in NPP (as estimated from the SeaWiFS satellite sensor) between 1997 and 2005 were attributed to reduced nutrient enhancement due to ocean surface warming (Behrenfeld et al. 2006; Carr et al. 2006). A recent example during the summer of 2005 was characterized by a three-month delay to the start of the upwelling season resulting in a lack of significant plankton production until August (rather than the usual April–May time period). Fish, birds and mammals that relied upon plankton production occurring at the normal time experienced massive recruitment failure (Schwing et al. 2006; Mackas et al. 2006; Sydeman et al. 2006). In contrast, the summer of 2006 had some of the strongest upwelling winds on record yet many species again experienced recruitment failure, in part because there was a one-month period of no winds (mid-May to mid-June).

Snyder et al. (2003) suggest that wind-driven upwelling in the California current is likely to continue its long, 30-year increase in the future, as a function of changes in the physical climate. Such a change could lead to enhanced productivity in the coastal marine environment, and subsequent changes throughout the ecosystem.



5.5 CHANGES IN PESTS AND PATHOGENS

5.5.1 Interactions of Climate Change with Pests, Pathogens, and Invasive Species

Increasing temperatures and other alterations in weather patterns (e.g., drought, storm events) resulting from climate change are likely to have significant effects on outbreaks of pests and pathogens in natural and managed systems, and are also expected to facilitate the establishment and spread of invasive alien species. For the purposes of this chapter, pests and pathogens refers to undesirable outbreaks of either native or introduced insects or pathogens. Non-native species are those that are non-indigenous to a region, either historically or presently, while invasive species are those non-native species that harm the environment, the economy or human health. Initially, the most noticeable changes in plant and animal communities will most likely result from direct effects of climate change (for example, range expansions of pathogens, and invasive plants). The longer term consequences, however, may be the result of indirect effects such as disruptions of trophic relationships or a species decline due to the loss of a mutualistic relationship (Parmesan 2006).

Interactions between increasing global temperature and pests and pathogens are of particular concern because of the rapid and sweeping changes these taxa can render. While it is still difficult to predict specifically how climate change will interact with insect pests, or plant and animal diseases, some recent events have provided glimpses into the kinds of impacts that might unfold.

5.5.1.1 MOUNTAIN PINE BEETLE EXPLOSION

The mountain pine beetle (*Dendroctonus ponderosae*) is a native species that has co-existed with western conifers for thousands of years and plays an important role in the life cycle of North American western forests (Bentz et al. 2001; Powell and Logan 2001). However, the magnitude of recent outbreaks is above historical levels with historically unprecedented mortality (Logan et al. 2003). A recent outbreak in 2006 caused the death of nearly five million lodgepole pines (*Pinus contorta*) in Colorado, a four-fold increase from 2005. The infestation covers nearly half of all Colorado's forests. Such outbreaks are not confined to Colorado, but are also occurring in other parts of the United States and Canada, affecting tens of thousands of square miles of forest (Figure 5.4).



Figure 5.4 Aerial view of the U.S. Forest Service Rocky Mountain Research Station's Fraser Experimental Forest near Winter Park, Colorado, May 2007 and a mountain pine beetle (inset). The green strips are areas of forest that had been harvested decades earlier, and so have younger faster growing trees. The red and brown areas show dead and dying trees caused by bark beetle infestation. A more recent photo would show less contrast because, due to drought and beetle epidemic, mortality rates of young trees have also risen. Photo courtesy USFS, Rocky Mountain Research Station.

Multiple factors, including climate change, have been implicated in driving outbreaks in North America (e.g., Romme et al. 2006; Logan and Powell 2001; Logan et al. 2003). First, many North American conifer forests are primarily mature, even-aged stands due to widespread burning and heavy logging of the region during settlement 100 years ago. Mountain pine beetles prefer the mature trees resulting from these disturbances. Second, long-term drought stresses trees and makes them more vulnerable to the beetles because they cannot effectively defend themselves. Third, warmer summers also cause stress and increase growth rates of the insects, and, fourth, milder winters increase the chances of survival for the insect larvae (Romme et al. 2006; Powell and Logan 2005; Powell et al. 2000). While there is not yet definitive proof that climate change is behind the high levels of mountain pine beetle infestation, a recent study showed that over the last century Colorado's average temperatures have warmed (NRC 2007). It is therefore reasonable to expect that warmer temperatures in the future may lead to similar or more intensive events than those that are now occurring.

5.5.1.2 POLEWARD MIGRATION OF PLANT PESTS AND PATHOGENS

Latitudinal gradients in plant defenses and herbivory are widely known but the basis for these defenses (i.e., genetic versus environment) are not fully understood. A potential outcome under warming global temperatures is a relatively rapid poleward migration of pests and pathogens, and a relatively slower rate of adaptation (e.g., increased defense against herbivory) for plants. Biogeographic theory predicts increased insect herbivory (i.e., greater loss of leaf area to herbivores) in the lower latitudes relative to higher latitudes (MacArthur 1972; Vermeij 1978; Jablonski 1993). As with the mountain pine beetle described above, higher population densities of other herbivorous insects and therefore herbivory occur because dormant season death (i.e., winter dieback) of herbivores is absent, or greatly reduced at warmer temperatures, and/or plant productivity is generally greater than at higher latitudes (Coley and Aide 1991; Coley and Barone 1996). Because of this greater herbivory, plants are thought to be better defended or otherwise less palatable at low latitudes as a result of natural selection (e.g., MacArthur

1972; Hay and Fenical 1988; Coley and Aide 1991; Coley and Barone 1996). Alternatively, plants at low latitudes could be better defended because high latitude populations have had fewer generations since the last glaciation to evolve such defenses (Fischer 1960).

5.5.1.3 CLIMATE CHANGE AND PATHOGENS

Evidence is beginning to accumulate that links the spread of pathogens to a warming climate. For example, the chytrid fungus (*Batrachochytrium dendrobatidis*) is a pathogen that is rapidly spreading worldwide and decimating amphibian populations. A recent study by Pounds et al. (2006) showed that widespread amphibian extinction in the mountains of Costa Rica is positively linked to global climate change. To date, geographic range expansion of pathogens related to warming temperatures has been the most easily detected (Harvell et al. 2002), perhaps most readily for arthropod-borne infectious disease (Daszak et al. 2000). However, a recent literature review found additional evidence gathered through field and laboratory studies that supports hypotheses that latitudinal shifts of vectors and diseases are occurring under warming temperatures. Based on their review, Harvell et al. (2002) gathered evidence that:

- Arthropod vectors and parasites die or fail to develop below threshold temperatures.
- Rates of vector reproduction, population growth, and biting increase (up to a limit) with increasing temperature.
- Parasite development rates and period of infectivity increase with temperature.

Furthermore, Ward and Lafferty (2004) conducted an analysis that revealed that disease for some groups of marine species is increasing while others are not. Turtles, corals, mammals, urchins, and mollusks all showed increasing trends of disease, while none were detected for sea grasses, decapods, or sharks/rays. The effects of increasing temperature on disease are complex, and can increase or decrease disease depending on the pathogen (Ward and Lafferty 2004).

Expansion of an invader may not always be simply explained by warming temperatures. For example, Roman 2006 suggests that the north-



ern expansion of the invasive European green crab (*Carcinus maenas*) in North America was facilitated through the introduction of new lineages of *C. maenas* to Nova Scotia from the northern end of its native range in Europe. These northern populations may be better adapted to the colder temperatures found in northern Nova Scotia, relative to more southerly waters. Furthermore, the construction of a causeway and subsequent isuper portí in the Strait of Canso, Nova Scotia, appears to be at the epicenter of the high diversity of new *C. maenas* haplotypes (Roman 2006).

5.5.1.4 CLIMATE CHANGE AND INVASIVE PLANTS

Projected increases in CO₂ are expected to stimulate the growth of most plants species, and some invasive plants are expected to respond with greater growth rates than non-invasive plants (Dukes 2000; Ziska and George 2004; Moore 2004; Mooney et al. 2006). Some invasive plants may have higher growth rates and greater maximal photosynthetic rates relative to native plants under increased CO₂, but definitive evidence of a general benefit of CO₂ enrichment to invasive plants over natives has not emerged (Dukes and Mooney 1999). Nonetheless, invasive plants in general may better tolerate a wider range of environmental conditions and may be more successful in a warming world because they can migrate and establish in new sites more rapidly than native plants, and they are not usually limited by pollinators or seed dispersers (Vila et al., *in press, accepted*).

Finally, it is critical to recognize that other elements of climate change (e.g., nitrogen deposition, land conversion) will play significant roles in the success of invasive plants in the future, either alone or under elevated CO₂ (Vila et. al., *in press, accepted*). For example, several studies have brought to light the role of increasing nitrogen availability and the success of invasive grass species (e.g., Huenneke et al. 1990; Brooks 2003). Disturbance at both global and local scales has been shown to be an important factor in facilitating species invasions (e.g., Sher and Hyatt 1999; Mooney and Hobbs 2001; D'Antonio and Meyerson 2002), and land conversion that occurred more than 100

years ago may play a role in current invasions (Von Holle and Motzkin 2007). Recent work by Hierro et al. (2006), which compared the effects of disturbance on *Centaurea solstitialis* in its native and introduced ranges, suggests that disturbance alone does not fully explain invasion success. Instead, it appears that, for *C. solstitialis*, it is the combination of disturbance and escape from soil pathogens in the native range that has encouraged invasion.

5.6 PARTICULARLY SENSITIVE SYSTEMS

5.6.1 Impacts of Climate Change on Montane Ecosystems

Temperate montane ecosystems are characterized by cooler temperatures and often increased precipitation compared to surrounding lowlands. Consequently, much of that precipitation falls in the form of snow, which serves to insulate the ground from freezing air temperatures, stores water that will be released as the snow melts during the following growing season, and triggers vertical migration by animal species that cannot survive in deep snow. Changes in historical patterns of snowfall and snowpack are predicted as a consequence of global climate change, in part due to changes in spatial patterns of precipitation, and in part due to the warming that will result in more precipitation falling as rain rather than snow (Beniston and Fox 1996; MacCracken et al. 2001). Areas that historically have most of their annual precipitation as snow are now seeing more of it as rain; documentation of this trend comes from the Sierra Nevada Mountains, where Johnson found from analysis of a 28-year dataset (Johnson 1998) that below 2400 meters, less snow is accumulating and it is melting earlier. Diaz et al. (2003) (Figure 5.5) also reported that all the major continental mountain chains exhibit upward shifts in the height of the freezing level surface over the past three to five decades.

Increased variation in precipitation and temperatures is also predicted by climate change models, and Johnson (1998) also found that “Higher elevations exhibit greater variability, with most stations accumulating more snow and melting earlier. This could be the result of warmer air masses having higher moisture contents.”



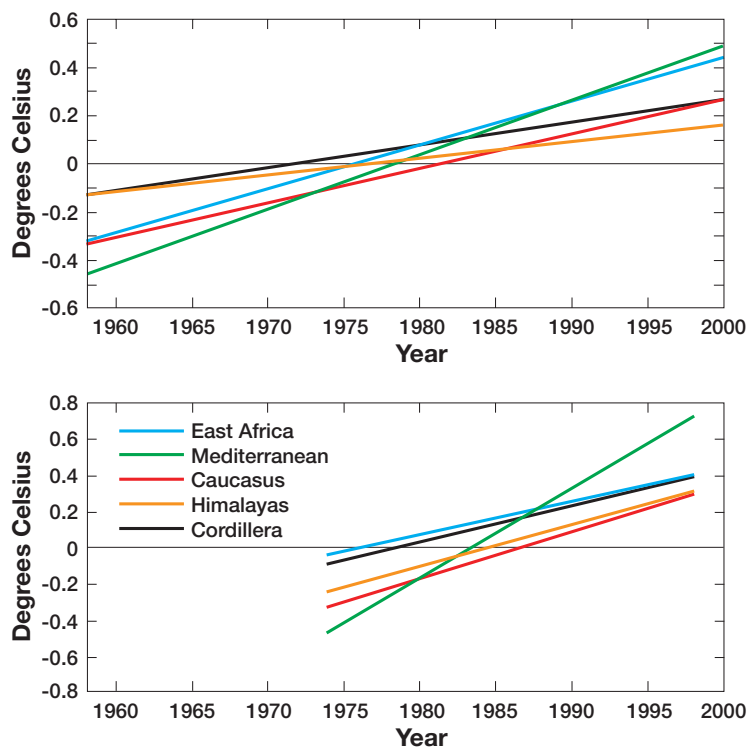


Figure 5.5 Linear trends in near surface air temperature for 5 different mountainous regions, based on the NCEP/NCAR Reanalysis data set. Top panel is for 1958–2000, lower panel for the period 1974–1998. From Diaz et al. 2003.

In addition to the influences of global climate change, which could affect both precipitation and temperature, regional effects can be important. For example, in the Colorado Rocky Mountains there are significant ENSO and PDO effects on winter precipitation. ENSO has also been shown to effect changes in freezing level in the American Cordillera (Diaz et al. 2003). Of course, all downstream water flows with headwaters in mountain areas are also affected by the variation in both timing and quantity of snowmelt (e.g., Karamouz and Zahraie 2004). These environmental changes are resulting in the disappearance of glaciers in most montane areas around the world. The changes in patterns and abundance of melt water from these glaciers have significant implications for the sixth of the world's population that is dependent upon glaciers and melting snowpack for water supplies (Barnett et al. 2005). Plant and animal communities are also affected as glaciers recede, exposing new terrain for colonization in an ongoing process of succession (e.g., for spider communities, see Gobbi et al. 2006).

One group of organisms whose reproductive phenology is closely tied to snowmelt is amphibians, for which this environmental cue is apparently more important than temperature (Corn 2003). Hibernating and migratory species that reproduce at high elevations during the summer are also being affected by the ongoing environmental changes. For example, marmots are emerging a few weeks earlier than they used to in the Colorado Rocky Mountains, and robins are arriving from wintering grounds weeks earlier in the same habitats (Inouye et al. 2000). Species such as deer, bighorn sheep, and elk, which move to lower elevations for the winter, may also be affected by changing temporal patterns of snowpack formation and disappearance.

The annual disappearance of snowpack is the environmental cue that marks the beginning of the growing season in most montane environments. Flowering phenology has been advancing in these habitats (Inouye and Wielgolaski 2003) as well as in others at lower altitudes, mirroring what is going on at higher latitudes (Wielgolaski and Inouye 2003). There is a very strong correlation between the timing of snowmelt, which integrates snowpack depth and spring air temperatures, and the beginning of flowering by wildflowers in the Colorado Rocky Mountains (e.g., Inouye et al. 2002; Inouye et al. 2003). For some wildflowers there is also a strong correlation between the depth of snowpack during the previous winter and the abundance of flowers produced (Inouye et al. 2002; Saavedra et al. 2003). The abundance of flowers can have effects on a variety of consumers, including pollinators (Inouye et al. 1991), herbivores, seed predators, and parasitoids, all of which are dependent on flowers, fruits, or seeds.

An unexpected consequence of earlier snowmelt in the Rocky Mountains has been the increased frequency of frost damage to montane plants, including the loss of new growth on conifer trees, of fruits on some plants such as *Erythronium grandiflorum* (glacier lilies), and of flower buds of other wildflowers (e.g., *Delphinium* spp., *Helianthella quinqueremis*, etc.) (Inouye 2008). Although most of these species are long-lived perennials, as the number of years in which



frost damage has negative consequences on recruitment increases, significant demographic consequences may result. These and other responses to the changing montane environment are predicted to result in loss of some species at lower elevations, and migration of others to higher elevations. Evidence that this is already happening comes from studies in both North America (at least on a latitudinal scale, Lesica and McCune 2004) and Europe (Grabherr et al. 1994). It is predicted that some animal species may also respond by moving up in elevation, and preliminary evidence suggests that some bumble bee (*Bombus*) species in Colorado have moved as much as a couple of thousand feet over the past 30 years (J. Thomson, personal communication).

5.6.2 Arctic Sea-Ice Ecosystems

Sea ice seasonally covered as much as 16,000,000 km² of the Arctic Ocean before it began declining in the 1970s (Johannessen et al. 1999; Serreze et al. 2007). For millennia, that ice has been integral to an ecosystem that provisions polar bears and the indigenous people. The ice also strongly influences the climate, oceanography, and biology of the Arctic Ocean and surrounding lands. Further, sea ice influences global climate in several ways, including via its high albedo and its role in atmospheric and oceanic circulation. In the past 10 years, the rate of decline in the areal extent of summer sea ice in the Arctic Ocean has accelerated, and evidence that the Arctic Ocean will be ice-free by 2050 is increasing (Stroeve et al. 2005, 2007; Overland and Wang 2007; Serreze et al. 2007; Comiso et al. 2008). Many organisms that depend on sea ice – ranging from ice algae to seals and polar bears – will diminish in number and may become extinct. Ecosystem changes already have been observed and are predicted to accelerate along with the rates of climate change. Many of the changes will not be readily obvious and may even counterintuitive. Here, we summarize expected changes and provide a few expected responses involving upper trophic levels that are thought to be illustrative.

At the base of the sea ice ecosystem are epontic algae adapted to very low light levels (Kühl et al. 2001; Thomas and Dieckmann 2002). Blooms of the those algae on the undersurface

of the ice are the basis of a food web leading through zooplankton and fish to seals, whales, polar bears, and people. Sea ice also strongly influences winds and water temperature, both of which influence upwelling and other oceanographic phenomena whereby nutrient rich water is brought up to depths at which there is sufficient sunlight for phytoplankton to make use of those nutrients (Buckley et al. 1979; Alexander and Niebauer 1981; Legendre et al. 1992).

Among the more southerly and seasonally ice-covered seas, the Bering Sea produces our nation's largest commercial fish harvests as well as supports subsistence economies of Alaskan Natives. Ultimately, the fish populations depend on plankton blooms regulated by the extent and location of the ice edge in spring. Naturally, many other organisms, such as seabirds, seals, walruses, and whales, depend on primary production, mainly in the form of those plankton blooms. As Arctic sea ice continues to diminish, the location, timing, and species make-up of the blooms are changing in ways that appear to favor marked changes in community composition (Hunt et al. 2002; Grebmeier et al. 2006). The spring melt of sea ice in the Bering Sea has long favored the delivery of organic material to a benthic community of bivalve mollusks, crustaceans, and other organisms. Those benthic organisms, in turn, are important food for walruses, gray whales, bearded seals, eider ducks, and many fish species. The earlier ice melts resulting from a warming climate, however, lead to later phytoplankton blooms that are largely consumed by zooplankton near the sea surface, vastly decreasing the amount of organic material reaching the benthos. The likely result will be a radically altered community favoring a different suite of upper level consumers. The subsistence and commercial harvests of fish and other marine organisms would also be altered.

Walruses (*Odobenus rosmarus*) feed on clams and other bottom-dwelling organisms (Fay 1982). Over a nursing period of two or more years, the females alternate their time between attending a calf on the ice and diving to the bottom to feed themselves. The record ice retreats observed in recent summers extend northward of the continental shelf such that the ice is over

An unexpected consequence of earlier snowmelt in the Rocky Mountains has been the increased frequency of frost damage to montane plants, including the loss of new growth on conifer trees, of fruits on some plants such as *Erythronium grandiflorum* (glacier lilies), and of flower buds of other wildflowers (e.g., *Delphinium* spp., *Helianthella quinquenervis*, etc.).



Whether the changes underway today will be survived by walrus, seals, polar bears, and other elements of the ecosystem will depend critically on the pace of change.

water too deep for the female walrus to feed (Kelly 2001). The increased distance between habitat suitable for adult feeding and that suitable for nursing young is likely to reduce population productivity (Kelly 2001; Grebmeier et al. 2006).

The major prey of polar bears and an important resource to Arctic Natives, ringed seals (*Pusa hispida*) are vulnerable to decreases in the snow and ice cover on the Arctic Ocean (Stirling and Derocher 1993; Tynan and DeMaster 1997; Kelly 2001). Ringed seals give birth in snow caves excavated above breathing holes they maintain in the sea ice. The snow caves protect the pups from extreme cold (Taugbøl 1984) and, to a large extent, from predators (Lydersen and Smith 1989). As the climate warms, however, snow melt comes increasingly early in the Arctic (Stone et al. 2002; Belchansky et al. 2004), and the seals' snow caves collapse before the pups are weaned (Kelly 2001; Kelly et al. 2006). The small pups are exposed without the snow cover and die of hypothermia in subsequent cold periods (Stirling and Smith 2004). The prematurely exposed pups also are more vulnerable to predation by Arctic foxes, polar bears, gulls, and ravens (Lydersen and Smith 1989). Gulls and ravens are arriving increasingly early in the Arctic as springs become warmer, further increasing their potential to prey on the seal pups.

Polar bears (Figure 5.6) are apex predators of the sea ice ecosystem, and their dependence on ice-associated seals makes them vulnerable to reductions in sea ice. While polar bears began diverging from brown bears (*Ursus arctos*) 150,000 to 250,000 years ago (Cronin et al. 1991; Talbot and Shields 1996; Waits et al. 1998), their specialization as seal predators in the sea ice ecosystem apparently is more recent, dating to 20,000 to 40,000 years ago (Stanley 1979; Talbot and Shields 1996). The bears' invasion of this novel environment was stimulated by an abundance of seals, which had colonized the region earlier in the Pleistocene (Deméré et al. 2003; Lister 2004). Adapting to the sea ice environment and a dependence on seals – especially ringed seals – exerted strong selection on the morphology, physiology, and behavior of polar bears.

The polar bear's morphological adaptations to the sea ice environment include dense, white fur over most of the body (including between foot pads), with hollow guard hairs; short, highly curved claws; and dentition specialized for carnivory. Physiologically, polar bears are extremely well adapted to feed on a diet high in fat; store fat for later future energy needs; and enter and sustain periods of reduced metabolism whenever food is in short supply (Derocher et al. 1990; Atkinson and Ramsay 1995). Feeding success is strongly related to ice conditions; when stable ice is over productive shelf waters, polar bears can feed throughout the year on their primary prey, ringed seals (Stirling and McEwan 1975; Stirling and Smith 1975; Stirling and Archibald 1977; Amstrup and DeMaster 1988; Amstrup et al. 2000). Less frequently, they feed on other marine mammals (Smith 1980, 1985; Calvert and Stirling 1990) and even more rarely on terrestrial foods (Lunn and Stirling 1985; Derocher et al. 1993). Polar bears exhibit the behavioral plasticity typical of top-level predators, and they are adept at capturing seals from the ice (Stirling 1974; Stirling and Derocher 1993).

Today, an estimated 20,000 to 25,000 polar bears live in 19 apparently discrete populations distributed around the circumpolar Arctic (Polar Bear Specialists Group 2006). Their overall distribution largely matches that of ringed seals, which inhabit all seasonally ice-covered seas in the Northern Hemisphere (Scheffer 1958; King 1983). Polar bears are not regularly found, however, in some of the marginal seas (e.g., the Okhotsk Sea) inhabited by ringed seals. The broad distribution of their seal prey is reflected in the home ranges of polar bears that, averaging over 125,000 km², are more than 200 times larger than the averages for terrestrial carnivores of similar size (Durner and Amstrup 1995; Ferguson et al. 1999). Most polar bear populations expand and contract their range seasonally with the distribution of sea ice, and they spend most of year on the ice (Stirling and Smith 1975; Garner et al. 1994). Most populations, however, retain their ancestral tie to the terrestrial environment for denning, although denning on the sea ice is common among the bears of the Beaufort and Chukchi seas (Harrington 1968; Stirling and Andriashek 1992; Amstrup and



Gardner 1994; Messeir et al. 1994; Durner et al. 2003). Dens on land and on ice are excavated in snow drifts, the stability and predictability of which are essential to cub survival (Blix and Lentfer 1979; Ramsay and Stirling 1988, 1990; Clarkson and Irish 1991).

The rapid rates of warming in the Arctic observed in recent decades and projected for at least the next century are dramatically reducing the snow and ice covers that provide denning and foraging habitat for polar bears (Overpeck et al. 1997; Serreze et al. 2000; Holland et al. 2006; Stroeve et al. 2007). These changes to their environment will exert new, strong selection pressures on polar bears. Adaptive traits reflect selection by past environments, and the time needed to adapt to new environments depends on genetic diversity in populations, the intensity of selection, and the pace of change. Genetic diversity among polar bears is evident in the 19 putative populations, suggesting some scope for adaptation within the species as a whole even if some populations will be at greater risk than others. On the other hand, the nature of the environmental change affecting critical features of polar bears' breeding and foraging habitats, and the rapid pace of change relative to the bears' long generation time (circa 15 years) do not favor successful adaptation. The most obvious change to breeding habitats is the reduction in the snow cover on which successful denning depends (Blix and Lenter 1979; Amstrup and Gardner 1994; Messier et al. 1994; Durner et al. 2003). Female polar bears hibernate for four to five months per year in snow dens in which they give birth to cubs, typically twins, each weighing just over 0.5 kg (Blix and Lentfer 1979). The small cubs depend on snow cover to maintain thermal neutrality. Whether it remains within the genetic scope of polar bears to revert to the ancestral habit of rearing in earthen dens is unknown.

Changes in the foraging habitat that will entail new selection pressures include seasonal mismatches between the energetic demands of reproduction and prey availability; changes in prey abundance; changes in access to prey; and changes in community structure. Emergence of female and young polar bears from dens in the spring coincides with the ringed seal's



Figure 5.6 Polar bear lounges near the Beaufort Sea, along Alaska's coastline. Image by Susanne Miller, from the U.S. Fish & Wildlife Service's digital library collection.

birthing season, and the newly emerged bears depend on catching and consuming young seals to recover from months of fasting (Stirling and Øritsland 1995). That coincidence may be disrupted by changes in timing and duration of snow and ice cover. Such mismatches between reproductive cycles and food availability are increasingly recognized as a means by which animal populations are impacted by climate change (Stenseth and Mysterud 2002; Stenseth et al. 2002; Walther et al. 2002).

The polar bear's ability to capture seals depends on the presence of ice (Stirling et al. 1999; Derocher et al. 2004). In that habitat, bears take advantage of the fact that seals must surface to breathe in limited openings in the ice cover. In the open ocean, however, bears lack a hunting platform, seals are not restricted in where they can surface, and successful predation is exceedingly rare (Furnell and Oolooyuk 1980). Only in ice-covered waters are bears regularly successful at hunting seals. When restricted to shorelines, bears feed little if at all, and terrestrial foods are thought to be of little significance to polar bears (Lunn and Stirling 1985; Ramsay and Hobson 1991; Stirling et al. 1999). Predation on reindeer observed in Svalbard, however, indicates that polar bears have some capacity to switch to alternate prey (Derocher et al. 2000).



Seal and other prey populations also will be impacted by fundamental changes in the fate of primary production. For example, in the Bering and Chukchi seas, the reduction in sea ice cover alters the physical oceanography in ways that diminish carbon flow to the benthos and increase carbon recycling in pelagic communities (Grebmeier et al. 2006). The resultant shift in community structure will include higher trophic levels. The exact composition of future communities is not known, nor is it known how effectively polar bears might exploit those communities.

Recent modeling of reductions in sea ice cover and polar bear population dynamics yielded predictions of declines within the coming century that varied by population but overall totaled 66% of all polar bears (Amstrup et al. 2007). Some populations were predicted to be extinct by the middle of the current century. While population reductions seem inevitable given the polar bear's adaptations to the sea ice environment (Derocher et al. 2004), quantitative predictions of declines are less certain as they necessarily depend on interpretations of data and professional judgments.

During previous climate warmings, polar bears apparently survived in unknown refuges that likely included some sea ice cover and access to seals. Within the coming century, however, the Arctic Ocean may be ice-free during summer (Overpeck et al. 2005), and the polar bear's access to seals will be diminished (Stirling and Derocher 1993; Lunn and Stirling 2001; Derocher et al. 2004). As snow and ice covers decline, polar bears may respond adaptively to the new selection pressures or they may become extinct. Extinction could result from mortality outpacing production, competition in terrestrial habitats with brown bears, and/or from re-absorption into the brown bear genome. Crosses between polar bears and brown bears produce fertile offspring (Kowalska 1965), and a hybrid was recently documented in the wild. Extinction through hybridization has been documented in other mammals (Rhymer and Simberloff 1996).

Predicted further warming inevitably will entail major changes to the sea ice ecosystem. Some ice-adapted species will become extinct; others

will adapt to new habitats. Whether the changes underway today will be survived by walruses, seals, polar bears, and other elements of the ecosystem will depend critically on the pace of change. Ecosystems have changed before; species have become extinct before. Critically important in our changing climate is the rapid rate of change. Biological adaptation occurs over multiple generations varying from minutes to many years depending on the species. The current rates of change in the sea ice ecosystem, however, are very steep relative to the long generation times of long-lived organisms such as seals, walruses, and polar bears.

5.7 ECOSYSTEM SERVICES AND EXPECTATIONS FOR FUTURE CHANGE

The Millennium Ecosystem Assessment (2005) is the most comprehensive scientific review of the status, trends, conditions, and potential futures for ecosystem services. It is international in coverage, although individual sections focus on regions, ecosystem types, and particular ecosystem services. The MEA categorized services as supporting, provisioning, regulating, and cultural (Figure 5.7). Some of these services are already traded in markets, e.g. the provision of food, wood, and fiber from both managed and unmanaged ecosystems, or the cultural services of providing recreational activities, which generate substantial revenue both within the United States and globally. The United States, for example, has a \$112 billion international tourism market and domestic outdoor recreation market (World Trade Organization 2002; Southwick Associates 2006).

Other services, in particular many cultural services, regulating services, and supporting services are not priced, and therefore not traded in markets. A few, like provision of fresh water or carbon sequestration potential, are mostly not traded in markets, but could be, and especially for carbon, there are many developing markets. In all cases, the recognition of a service provided by ecosystems is the recognition that they are producing or providing something of value to humans, and thus its value is shaped by the social dimensions and values of our societies as well as by physical and ecological factors (MEA 2005).



An example of an ecosystem service that has an increasingly recognized value is that provided by pollinators. Part of this increased recognition is a consequence of the recent declines in abundance that have been observed for some pollinators, particularly the introduced honey bee (*Apis mellifera*) (National Research Council 2006). The economic significance of pollination is underscored by the fact that about three-quarters of the world's flowering plants depend on pollinators, and that almost a third of the food that we consume results from their activity. The majority of pollinators are insects, whose distributions, phenology, and resources are all being affected by climate change (Inouye 2007). For example, an ongoing study at the Rocky Mountain Biological Laboratory (Pyke, Thomson, Inouye, unpublished) has found evidence that some bumble bee species have moved up as much as a few thousand feet in elevation over the past 30 years. Unfortunately, with the exception of honey bees and butterflies, there are very few data available on the abundance and distribution of pollinators, so it has been difficult to assess their status and the changes that they may be undergoing (National Research Council 2006).

Biological diversity is recognized as providing an underpinning for all these services in a fundamental way. A major finding of the MEA from a global perspective was that 16 out of 24 different ecosystem services that were analyzed were being used in ultimately unsustainable ways. While this finding was not specific to U.S. ecosystems, it does set a context for considering the consequences of documented ecosystem changes for services.

A subsequent question is whether any such changes in services can be reasonably attributed to climate change. The MEA evaluated the relative magnitudes and importance of a number of different direct drivers (Nelson et al. 2006) for changes in ecosystems, and whether the importance of those drivers was likely to increase, decrease, or stay about the same over the next several decades. The conclusion was that although climate change was not currently the most important driver of change in many ecosystems, it was one of the only drivers whose

importance was likely to continue to increase in all ecosystems over the next several decades (Figure 5.8).

5.8 ADEQUACY OF OBSERVING SYSTEMS

One of the challenges of understanding changes in biological diversity related to variability and change in the physical climate system is the adequacy of the variety of monitoring programs that exist for documenting those changes.

It is useful to think about such programs as falling into three general categories. The first is the collection of operational monitoring systems that are sponsored by federal agencies, conservation groups, state agencies, or groups of private citizens that are focused on particular taxa (e.g., the Breeding Bird Survey) or particular ecosystems (e.g., Coral Reef Watch). These tend to have been established for very particular purposes, such as for tracking the abundance of migratory songbirds, or the status and abundance of game populations within individual states, or the status and abundance of threatened and endangered species.

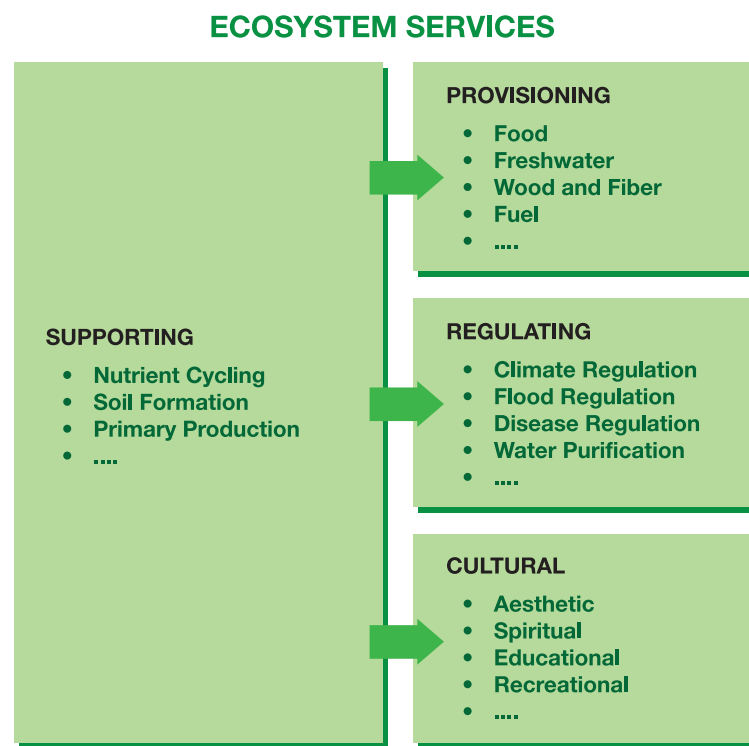


Figure 5.7 Categorization of ecosystem services, from MEA 2005.



The second category of monitoring programs is those whose initial justification has been to investigate particular research problems, whether or not those are primarily oriented around biodiversity. For example, the existing Long-Term Ecological Research Sites (LTERs) are important for monitoring and understanding trends in biodiversity in representative U.S. biomes, although their original justification was oriented around understanding ecosystem functioning. The yet-to-be established National Ecological Observatory Network (NEON) would also fall into this category. NEON's design for site locations samples both climate variability and ecological variability within the United States in a much more systematic way than ever before

done for a long-term research network, so there are likely to be very powerful results that can come from network-wide analyses.

The third category of monitoring systems is those that offer the extensive spatial and variable temporal resolution of remotely sensed information from Earth-orbiting satellites. These are not always thought of as being part of the nation's system for monitoring biological diversity, but in fact, they are an essential component of it. Remotely sensed data are the primary source of information on a national scale for documenting land-cover and land-cover change across the United States, for example, and thus they are essential for tracking

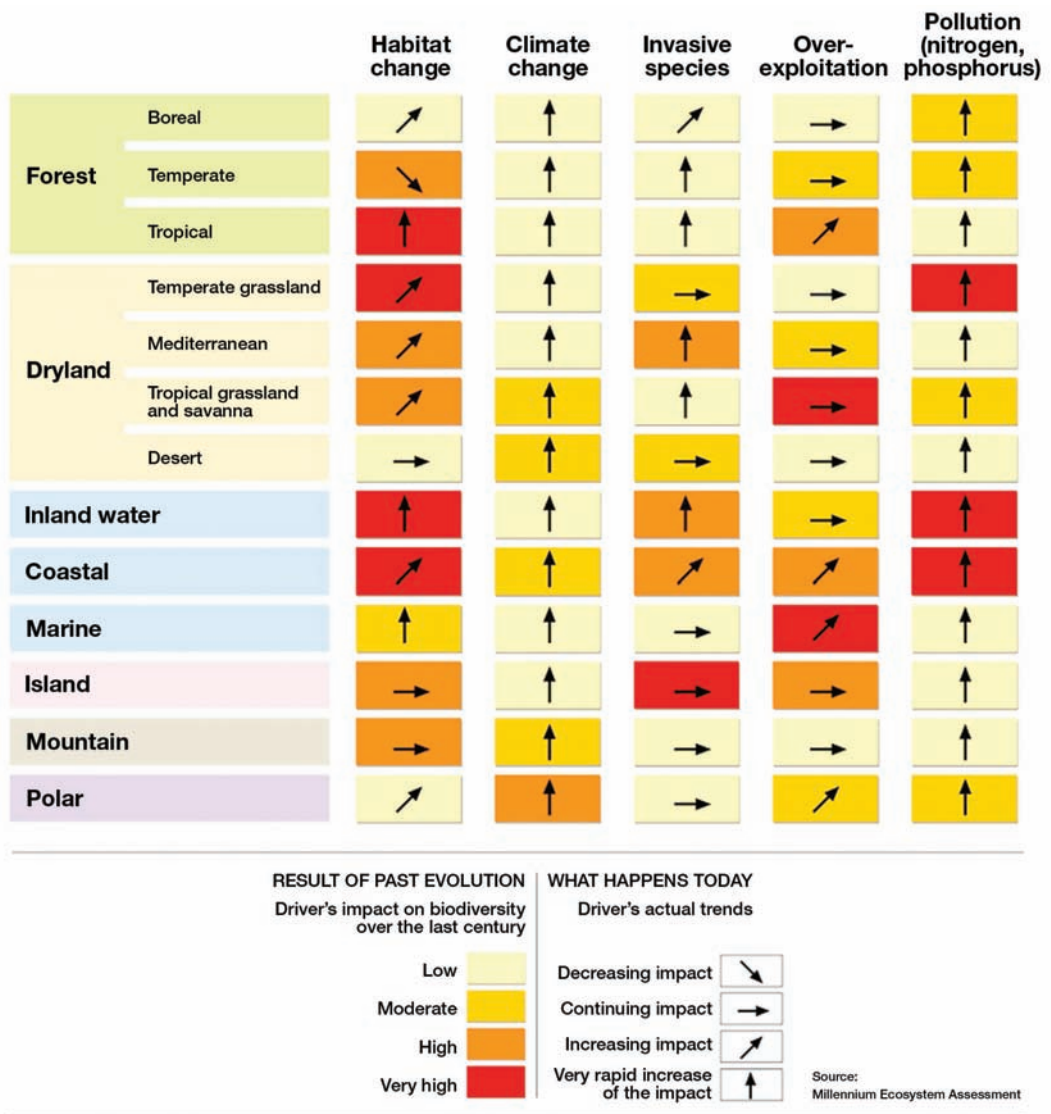


Figure 5.8 Relative changes in magnitude to ecosystem services caused by changes in habitat, climate, species invasion, over-exploitation, and pollution.

changes in perhaps the biggest single driver of changes in biodiversity, i.e. changes in habitat. Over the decades of the 1990s and 2000s, the remarkable profusion of Earth observation satellites has provided global coverage of many critical environmental parameters, from variability and trends in the length of growing season, to net primary productivity monitoring, to the occurrence of fires, to the collection of global imagery on 30-meter spatial resolution for more than a decade. Observational needs for biodiversity monitoring and research were recently reviewed by the International Global Observations of Land Panel, in a special report from a conference (IGOL 2006).

The National Research Council has recently released the first-ever Decadal Survey for Earth Science and Observations (NRC, 2007), which makes a comprehensive set of recommendations for future measurements and missions that would simultaneously enhance scientific progress, preserve essential data sets, and benefit a wide variety of potential applications. The report found that “the extraordinary U.S. foundation of global observations is at great risk. Between 2006 and the end of the decade, the number of operating missions will decrease dramatically, and the number of operating sensors and instruments on NASA spacecraft, most of which are well past their nominal lifetimes, will decrease by some 40 percent.”

Although there are lists of specifications for monitoring systems that would be relevant and important for recording changes in biodiversity associated with climate variability and change (e.g., IGOL 2007), at present there is no analysis in the literature that directly addresses the question of whether existing monitoring systems are adequate. For the moment, there is no viable option but to use existing systems for recording biodiversity changes, even if it means that the scientific community is attempting to use systems originally designed for alternate purposes.

The Government Accountability Office (2007) has documented extensively that one of the greatest perceived needs of federal land management agencies is for targeted monitoring systems that can aid them in responding to climate change. These agencies (e.g., U.S. Forest Service, National Park Service, Bureau of Land Management, NOAA, the U.S. Geological Survey) each face situations in which they recognize that they are already beginning to see the biological and ecological impacts of climate change on resources that they manage. GAO identified the improvement of monitoring capabilities to formulate effective adaptation and management responses as a priority. Remedying this situation was identified as a critical priority.

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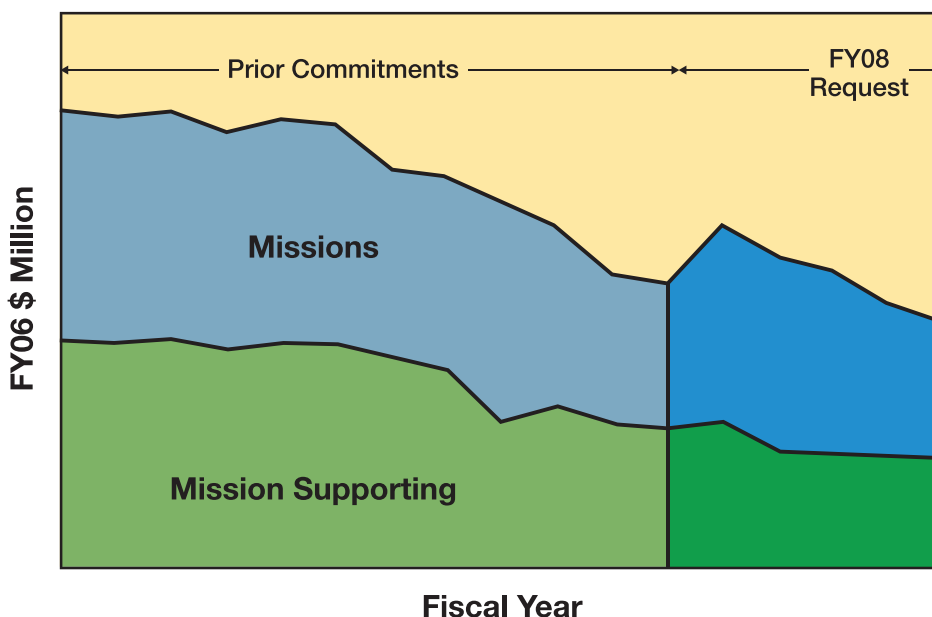


Figure 5.9 NASA budget for earth science research and applications demonstrations (1996–2012, fixed 2006 dollars).

5.9 MAJOR FINDINGS

In this section, we list the major findings from each section of the chapter, by topic heading. We then draw some general conclusions about the observed and potential impacts of climate change on biological diversity, the relationships to ecosystem services, and the adequacy of existing monitoring systems to document continuing change.

5.9.1 Growing Season and Phenology

- There is evidence indicating a significant lengthening of the growing season and higher NPP in the higher latitudes of North America where temperature increases are relatively high.
- Over the last 19 years, global satellite data indicate earlier onset of spring across the temperate latitudes by 10–14 days (Myneni, 2001; Lucht, 2002), an increase in summer photosynthetic activity (normalized difference vegetation index satellite estimates) (Myneni 2001) and an increase in the amplitude of annual CO₂ cycle (Keeling 1996), all of which are supported by climatological and field observations.
- Forest productivity, in contrast, which is generally limited by low temperature and short growing seasons in the higher latitudes and elevations, has been slowly increasing at less than 1 percent per decade (Boisvenue and Running 2006; Joos et al. 2002; McKenzie 2001; Caspersen et al. 2000).
- The exception to this pattern is in forested regions that are subject to drought from climate warming, where growth rates have decreased since 1895 (McKenzie 2001). Recently, widespread mortality over 12,000 km² of lower elevational forest in the southwest United States demonstrates the impacts of increased temperature and the associated multiyear drought (Breshears et al. 2005) even as productivity at tree line had increased previously (Swetnam and Betancourt 1998).
- Disturbances created from the interaction of drought, pests, diseases, and fire are projected to have increasing impacts on forests and their future distributions (IPPC 2007).

5.9.2 Biogeographical and Phenological Shifts

- Evidence from two meta-analyses (Root et al. 2003; Parmesan 2003) and a synthesis (Parmesan 2006) on species from a broad array of taxa suggests that there is a significant impact of recent climatic warming in the form of long-term, large-scale alteration of animal and plant populations.
- Movement of species in regions of North America in response to climate warming is expected to result in shifts of species ranges poleward and upward along elevational gradients (Parmesan, 2006).
- In an analysis of 866 peer-reviewed papers exploring the ecological consequences of climate change, nearly 60 percent of the 1598 species studied exhibited shifts in their distributions and/or phenologies over a 20- and 140-year time frame {Parmesan 2003}.
- Analyses of field-based phenological responses have reported shifts as great as 5.1 days per decade (Root et al. 2003), with an average of 2.3 days per decade across all species (Parmesan and Yohe 2003).

5.9.3 Migratory Birds

- A climate change signature is apparent in the advancement of spring migration phenology (Root et al. 2003) but the indirect effects may be more important than the direct effects of climate in determining the impact on species persistence and diversity.

5.9.4 Butterflies

- The migration of butterflies in the spring is highly correlated with spring temperatures and with early springs. Researchers have documented many instances of earlier arrivals (26 of 35 species in the United Kingdom (Roy and Sparks 2000); 17 of 17 species in Spain, (Stefanescu 2004); and 16 of 23 species in central California (Forister and Shapiro 2003).
- Butterflies are also exhibiting distributional and/or range shifts in response to warming. Across all studies included in her synthesis, Parmesan (2006) found 30–75 percent of



species had expanded northward, less than 20 percent had contracted southward, and the remainder were stable (Parmesan 2006).

5.9.5 Coastal and Near Shore Systems

- In the tropics there have been increasing coral bleaching and disease events and increasing storm intensity.
- In temperate regions there are demonstrated range shifts in rocky intertidal organisms, coastal fisheries, and in marine fisheries as well, and possible alterations of ocean currents and upwelling sites.

5.9.6 Corals

- Corals and tropical regions where they live are experiencing increasing water temperatures, increasing storm intensity (Emmanuel 2005), and a reduction in pH (Ravens et al. 2005), all while experiencing a host of other on-going challenges from development/tourism, unsustainable fishing and pollution. Acidification presents a persistent threat that is increasing in magnitude for shallow water corals and free-swimming calcifying organisms.
- Corals in many tropical regions are experiencing substantial mortality from increasing water temperatures and intense storms, both of which could be exacerbated by a reduction in pH. Increases in ocean acidity are a direct consequence of increases in atmospheric carbon dioxide.

5.9.7 Marine Fisheries

- Large, basin-scale atmospheric pressure systems that drive basin-scale winds can suddenly shift their location and intensity at decadal time scales, with dramatic impacts on winds and ocean circulation patterns. Perhaps the greatest discovery of the past 10 years is that these shifts also have powerful impacts on marine ecosystems.
- Examples of ecosystems impacts include increased flow of oceanic water into the English Channel and North Sea resulting in a northward shift in the distribution of zooplankton. As a result, the zooplankton community became dominated by warm

water species (Beaugrand, 2004) with concomitant changes in fish communities from one dominated by whiting (hake) to one dominated by sprat (similar to a herring).

- Similar (and drastic) ecosystem changes are known for the Baltic Sea (Kenny and Mollman 2006), where drastic changes in both zooplankton and fish communities were observed. Cod were replaced by sprat and dominance in zooplankton switched from lipid-rich (and high bioenergetic content) species to lipid-poor species.
- Linkages between the NAO, zooplankton and fisheries have also been described for the Northwest Atlantic, including waters off eastern Canada and the United States; Pershing and Green (2007) report a decrease in salinity, and an increase in biomass of small copepods (zooplankton).

5.9.8 Pests and Pathogens

- Evidence is beginning to accumulate that links the spread of pathogens to a warming climate. For example, the chytrid fungus (*Batrachochytrium dendrobatidis*) is a pathogen that is rapidly spreading worldwide, and decimating amphibian populations. A recent study by Pounds and colleagues (2006) showed that widespread amphibian extinction in the mountains of Costa Rica is positively linked to global climate change.
- To date, geographic range expansion of pathogens related to warming temperatures have been the most easily detected (Harvell et al. 2002), perhaps most readily for arthropod-borne infectious disease (Daszak et al. 2000). However, a recent literature review found additional evidence gathered through field and laboratory studies that support hypotheses that latitudinal shifts of vectors and diseases are occurring under warming temperatures.

5.9.9 Invasive Plants

- Projected increases in CO₂ are expected to stimulate the growth of most plants species, and some invasive plants are expected to respond with greater growth rates than non-invasive plants. Some invasive plants may have higher growth rates and greater maxi-



mal photosynthetic rates relative to native plants under increased CO₂, but definitive evidence of a general benefit of CO₂ enrichment to invasive plants over natives has not emerged (Dukes and Mooney 1999).

- Nonetheless, invasive plants in general may better tolerate a wider range of environmental conditions and may be more successful in a warming world because they can migrate and establish in new sites more rapidly than native plants and they are not usually limited by pollinators or seed dispersers (Vila et al., in press, accepted).
- Finally, it is critical to recognize that other elements of climate change (e.g., nitrogen deposition, land conversion) will play a significant role in the success of invasive plants in the future, either alone or under elevated CO₂ (Vila et al., in press, accepted).

5.9.10 Particularly Sensitive Systems

- One group of organisms whose reproductive phenology is closely tied to snowmelt is amphibians, for which this environmental cue is apparently more important than temperature.
- Hibernating and migratory species that reproduce at high elevations during the summer are also being affected by the ongoing environmental changes. For example, marmots are emerging a few weeks earlier than they used to in the Colorado Rocky Mountains, and robins are arriving from wintering grounds weeks earlier in the same habitats. Species such as deer, bighorn sheep, and elk, which move to lower elevations for the winter, may also be affected by changing temporal patterns of snowpack formation and disappearance.
- There is a very strong correlation between the timing of snowmelt, which integrates snowpack depth and spring air temperatures, and the beginning of flowering by wildflowers in the Colorado Rocky Mountains.

- An unexpected consequence of earlier snowmelt in the Rocky Mountains has been increased frequency of frost damage to montane plants, including the loss of new growth on conifer trees, of fruits on some plants such as *Erythronium grandiflorum* (glacier lilies), and of flower buds of other wildflowers (e.g., *Delphinium* spp., *Helianthella quinquenervis*, etc.). Although most of these species are long-lived perennials, as the number of years in which frost damage has negative consequences on recruitment increases, significant demographic consequences may result.

5.9.11 Arctic Sea-Ice Ecosystems

- Today, an estimated 20,000–25,000 polar bears live in 19 apparently discrete populations distributed around the circumpolar Arctic (IUCN Polar Bear Specialists Group 2005). Their overall distribution largely matches that of ringed seals, which inhabit all seasonally ice-covered seas in the northern hemisphere (Scheffer 1958; King 1983), an area extending to approximately 15,000,000 km².
- Most polar bear populations expand and contract their range seasonally with the distribution of sea ice, and they spend most of year on the ice (Stirling and Smith 1975; Garner et al. 1994).
- The rapid rates of warming in the Arctic observed in recent decades and projected for at least the next century are dramatically reducing the snow and ice covers that provide denning and foraging habitat for polar bears (Roots 1989; Overpeck et al. 1997; Serreze et al. 2000; Stroeve et al. 2007).
- During previous climate warmings, polar bears apparently survived in some unknown refuges. Whether they can withstand the more extreme warming ahead is doubtful (Stirling and Derocher 1993; Lunn and Stirling 2001).



5.10 CONCLUSIONS

Terrestrial and marine systems are already being demonstrably affected by climate change. This conclusion can be made with very high confidence. There are observable impacts of climate change on terrestrial ecosystems in North America including changes in the timing of growing season length, phenology, primary production, and species distributions and diversity. Some important effects on components of biological diversity have already been observed and have been increasingly well documented over the past several decades. This statement is true both for U.S. ecosystems, and ecosystems and biological resources around the world (IPCC 2007).

There is a family of other impacts and changes in biodiversity that are theoretically possible, and even probable (e.g., mismatches in phenologies between pollinators and flowering plants), but for which we do not yet have a substantial observational database. However, we cannot conclude that the lack of a complete observational database in these cases is evidence that they are not occurring – it is just as likely that it is simply a matter of insufficient numbers or lengths of observations.

It is difficult to pinpoint changes in ecosystem services that are specifically related to changes in biological diversity in the United States. The Millennium Ecosystem Assessment (2005) is the most recent, and most comprehensive, scientific assessment of the state of ecosystem services around the world, the drivers of changes in both ecosystems and services, the inherent tradeoffs among different types of ecosystem services, and what the prospects are for sustainable use of ecological resources. The MEA concludes that climate change is very likely to increase in importance as a driver for changes in biodiversity over the next several decades, although for most ecosystems it is not currently the largest driver of change. But a specific assessment of changes in ecosystem services for the United States as a consequence of changes in climate or other drivers of change has not been done.

We can think of the monitoring systems that have been used to evaluate the relationship between changes in the physical climate system and biological diversity as having three components.

- There is a plethora of species-specific or ecosystem-specific monitoring systems, variously sponsored by the U.S. federal agencies, state agencies, conservation organizations, and other private organizations. However, in very few cases were these monitoring systems established with climate variability and climate change in mind.
- Augmenting the monitoring systems that make routine measurements is a set of more specific research activities that have been designed to create time-series of population data and associated climatic and environmental data.
- The third component is spatially extensive observations derived from remotely sensed data. Some of these satellite data are primarily focused on land-cover, and thus are a good indicator of the major single driver of changes in biodiversity patterns. Others produce estimates of NPP and changes in the growing season, and thus reflect functional changes in ecosystems. However, similarly to the in situ monitoring networks, the future of space-based observations is not assured. The NRC (2007) has recently released a major survey of data and mission needs for the Earth sciences to address this issue, so we will not pursue it further here.

