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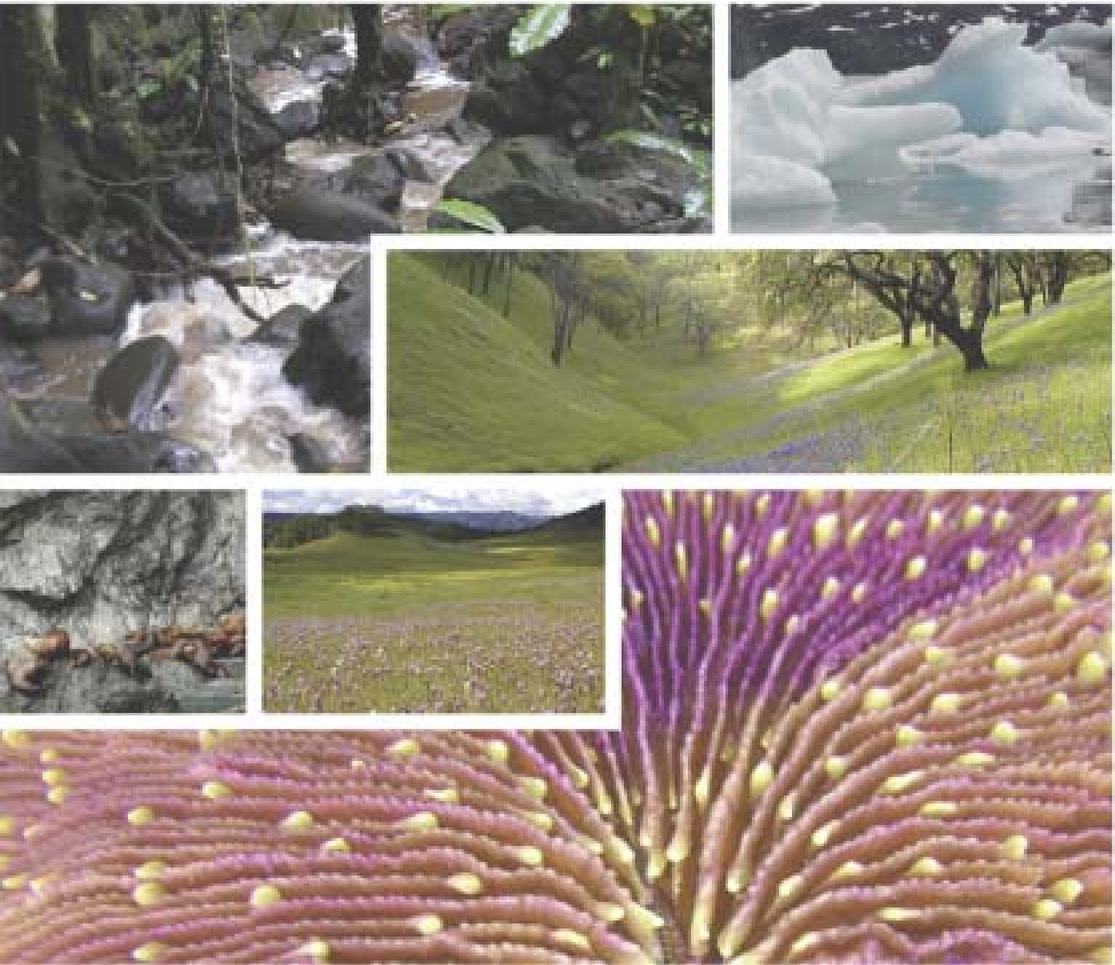
BUYING TIME:

A User's Manual for Building Resistance and Resilience to Climate Change in Natural Systems



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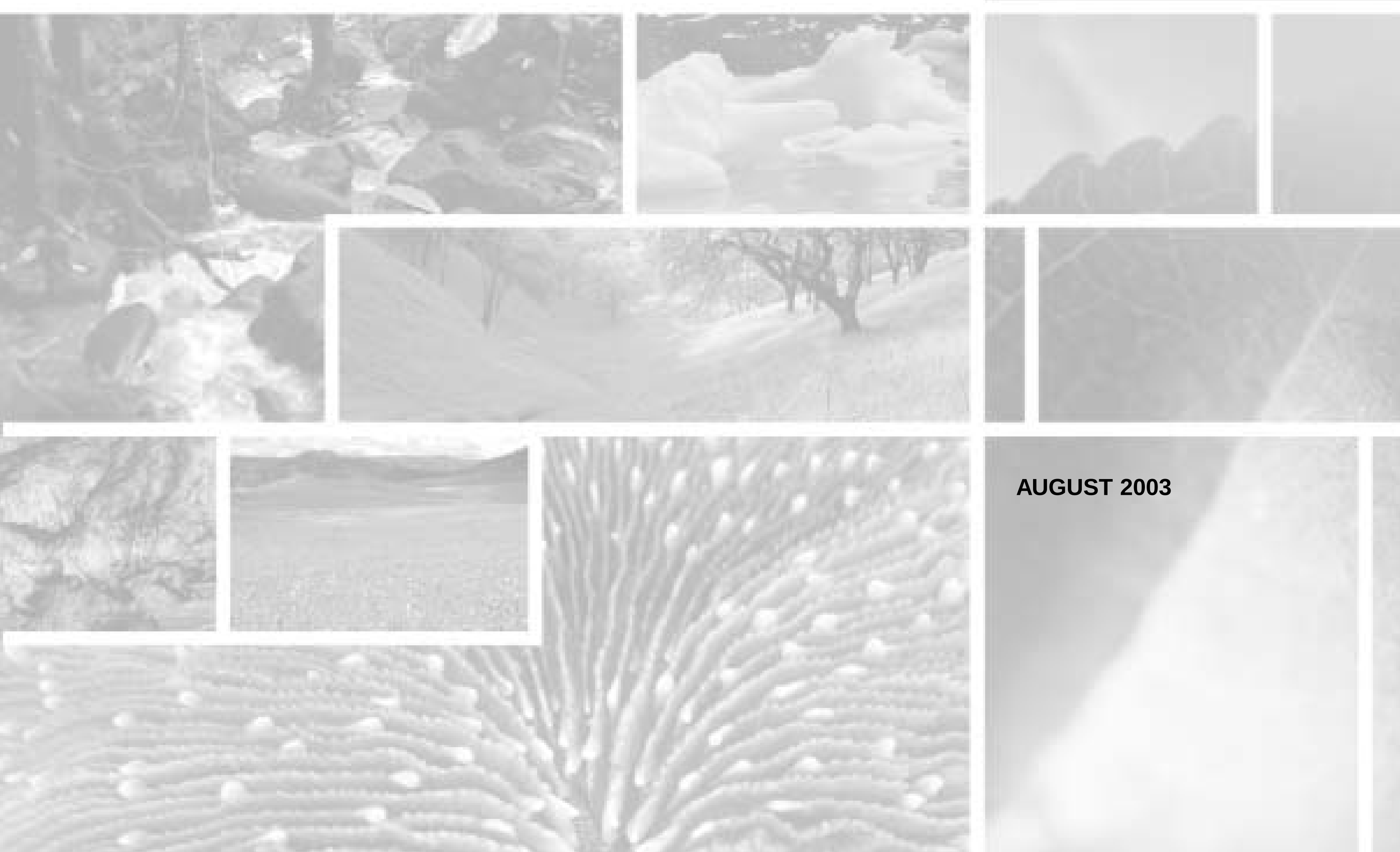

AUGUST 2003

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AS WE STAND AT THE beginning of the new millennium, the threats to nature and protected areas are unprecedented. While some progress has been made and strategies such as protected areas have been successful in preserving biodiversity in some places, new threats are arising. None of these threats is as great as global climate change and none will have such large implications for the way natural resource managers plan and implement conservation strategies.

While global climate change is seemingly difficult to understand and plan for, planning is essential, as the conservation approaches of the past may not work in an ever-changing warmer world. New strategies, led by deep cuts in the heat-trapping gases that causes climate change, predominately carbon dioxide from the burning of fossil fuels, may at least buy some time for ecosystems to adapt in the years and decades ahead. However, if CO_2 emissions are not reduced quickly and deeply, some of those treasured ecosystems will not survive.

Climate change is happening now and nature is experiencing its impacts first. Whether one looks at coral reefs, mangroves, arctic areas or montane regions, climate change poses a complex and bewildering array of problems for ecosystems. The key question is, what can be done—in addition to the rapid reduction of CO_2 emissions now—to increase the resiliency of these ecosystems to climate change?

WWF compiled “Buying Time: A User’s Manual for Building Resistance and Resilience to Climate Change in Natural Systems” for natural resource managers who are ready to confront the impacts of climate change. While far from comprehensive, this manual brings together assessments and potential initial adaptation strategies for various biomes.

Written by experts, the manual addresses all of the major biomes with practical ideas of how to begin increasing the resiliency of ecosystems and plan our protected areas in response to the threat of climate change. Some of these strategies are in line with the conservation strategies we have been working on for years—reducing fragmentation, building corridors, reducing threats, and increasing resiliency in general. Natural resource managers in the field must begin planning to buy time for these biomes while the world works to switch from coal to clean power, a key strategy to reduce the causes of climate change.

Looking at the models and gathering empirical evidence, WWF is recommending that natural resource managers start now to build climate change adaptation strategies into their preservation philosophies and plans. This manual is a first step to assist managers in doing so.

While testing and implementing these new conservation strategies, managers should also communicate the threat that climate change poses to their biome to local, regional and national authorities. Resource managers have an important role to play in the climate change debate by using examples of changes seen in their regions as indicators of the need for rapid and deep cuts in CO_2 emissions.

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Building Resistance and Resilience to Climate Change

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A VAST ARRAY OF SCIENTIFIC literature now makes it abundantly clear that the climate is changing and ecosystems are being affected by these changes. Much as awareness has been raised about invasive species, environmental contaminants, altered hydrology, and habitat fragmentation, conservation practitioners must now address climate change. This manual aims to assist natural resource and protected area managers as they begin to consider how to respond to this growing threat.

The need to respond couldn't be clearer; effects of climate change are now visible around the world. This year two large-scale studies demonstrated a global ecological fingerprint of climate change. To resolve differences between economists and biologists over the strength of climate change effects in natural systems, Parmesan and Yohe (2003) analyzed data on more than 1700 species to show with a "very high confidence" level (Intergovernmental Panel on Climate Change definition) that climate change has already altered range boundaries and phenology. A separate study using data from 143 studies found that 80% of species studied showed trait changes consistent with climate change-driven predictions (Root et al., 2003).

In each of the biomes discussed in this manual there are examples of specific climate-related changes. The thickness and extent of arctic sea ice has reduced dramatically; in 2002 ice coverage was the smallest it has been since satellite records began (Serreze et al., 2003). Coral bleaching has drastically changed some reef communities and while bleaching was once a rare and localized occurrence it is now a global phenomenon (Glynn, 1993; Hoegh-Guldberg, 1999). Montane glaciers are shrinking around the world, and rates of retreat are generally accelerating (Haeberli et al., 1999). Increases in surface water temperatures in Lake Tanganyika, a deep tropical lake in East Africa, have reduced annual mixing, causing nutrient depletion in the upper layers of the lake and have reduced primary productivity by 70% since 1975 (Verburg et al., 2003). In the highland rainforests of Monteverde, Costa Rica, the lifting of the cloud base associated with increased ocean temperatures has been linked to the disappearance of 20 species of frogs (Pounds et al., 1999). On a rocky shoreline in central California, where annual mean ocean temperature has increased 1ºC over the past 60 years, researchers have documented an increase in southern animal species and a decrease in northern animal species (Sagarin et al., 1999).

Current Trajectory & The Limits of Adaptation

Over the past century the average global temperature has risen 0.7 ºC as atmospheric CO_2 concentrations have risen from ~280 ppm to 370 ppm. Emissions scenarios from the IPCC (2001) suggest that if humans do not act to reduce emissions, we will see CO_2 levels of 550 ppm within the next 40 to 100 years, roughly a doubling of pre-industrial concentrations. This range of scenarios implies an additional increase in temperature of 1 to 5.8 ºC. Recent papers suggest that the upper end of this range is more likely and that even higher temperatures are possible, especially if climate sensitivity has been underestimated (Caldeira et al., 2003). This higher range of temperatures will also mean greater sea level rise and greater potential changes in precipitation and oceanic currents.

Clearly most systems will be dramatically challenged and subsequently altered by changes of this magnitude. It is unlikely that any local strategies could provide adequate protection for biodiversity under these conditions (Figure 1). Conserving biodiversity will therefore require a two pronged approach. First, greenhouse gas emissions must be dramatically reduced in order to slow the rate and extent of global climate change. Under current emissions scenarios, all outcomes result in dramatic changes beyond the reach of adaptive measures. WWF proposes that we reduce greenhouse gas emissions to limit change to less than a 2 ºC average temperature increase above pre-industrial levels. Second, assuming that we can limit the rate and extent of change, we will still need to respond to the change that is already inherent in the system and buy some time for ecosystems as emissions are reduced. Because the effects of greenhouse gases in the atmosphere have a substantial lag time we are locked into additional change from the concentrations of greenhouse gases already in the atmosphere today. This does not mean that reducing emissions is futile; rather, it means that we must reduce emissions quickly and deeply and take local action to protect biodiversity by increasing the resistance and resilience of natural systems so they can better survive the changes to come.

Figure 1. The effects of climate change on biodiversity for three thermal thresholds. (Adapted from C. Parmesan, pers. comm., based on a preliminary survey of existing literature)

TEMPERATURE CHANGE (°C)	EFFECT ON BIODIVERSITY
2	Some species lost Possible management options exist
4	Many species lost Few management options (those that exist will be extremely expensive)
6	Dire

What is Resilience Building in Response to Climate Change?

We think of this manual as a small first step in developing successful adaptation strategies for natural systems conservation. The chapters outline the significant features of each biome, what threats are already affecting these biomes, likely affects of climate change on each biome, possible strategies for increasing resilience within the particular biome and finally offer suggestions and examples of how these strategies might be implemented.

Increasing the resilience of a natural system is a standard goal of conservation; intact ecosystems have more resources for withstanding stresses. Natural systems are already affected by an array of stresses, from fragmentation to pollution to invasive species. Climate change will add another layer of stress to this complex matrix of interactions. Increasing ecosystem resilience to climate change will require even greater vigilance and longer term planning. Suggestions in these chapters fall into three broad categories.

1. PROTECT ADEQUATE AND APPROPRIATE SPACE

   Ecosystems with high biodiversity and those that maintain crucial structural components are thought to recover more easily from climatic disturbances. Traditional conservation methods such as creating protected areas, whether in terrestrial or marine areas, will thus have another justification in the next several decades. It will become increasingly important, however, to take into account projected impacts of climate change when designing new protected area systems, and to expand spatial scales through buffer zones and corridors to aid species migration. In particular, planners should look for climate refugia-areas that experience less change than others. Planning reserves will now require an eye for potentially dramatic future changes in protected areas; thinking about not only current but future configurations of habitats, communities, and ecosystems. Managers will need to be even more strategic, creative and flexible in designing protection strategies to address traditional land uses, existing threats, and also climate change stresses. Protecting not just space but functional groups, keystone species, climatic refugia, and multiple microhabitats within a biome to provide adequate representation is essential.

2. LIMIT ALL NON-CLIMATE STRESSES

   Climate change is not occurring in a vacuum. There are myriad stresses affecting natural systems, including habitat fragmentation, overharvest, invasive species, and pollution. A limited body of research on interactions between climate and non-climate stresses suggests synergistic responses (McLusky et al., 1986). For example, when rainbow trout *(Salmo gairdneri)* are exposed to the pesticide permethrin over a range of temperatures, the toxicity increases as temperature increases (Kumaraguru and Beamish, 1981). To support ecosystem resilience you must reduce the number of simultaneous insults faced by that ecosystem. Fortunately many stressors are more locally controllable than climate change. In a marine system this may mean establishing “no-take zones” to reduce fishing pressure and associated habitat destruction. In a freshwater system this may require limiting the concentration of toxic substances in effluent from an upstream industry. It may mean protecting alpine watersheds by limiting extraction of water by downslope agriculture and cities, or limiting harmful grazing practices in grasslands. Forests could require limiting fragmentation from road construction and logging. None of these tasks are easy, but they are approachable on a local level and they can increase the overall resilience of the system.

3. USE ACTIVE ADAPTIVE MANAGEMENT AND STRATEGY TESTING

Given uncertainty about the exact nature of ecosystem impacts of and responses to climate change, effective management will require a responsive and flexible approach. The success of various conservation approaches should be continually reassessed, and approaches adjusted as new information becomes available. In instances where impacts are relatively clear, active intervention to increase adaptive capacity coupled with monitoring is necessary. Such intervention may include assisted migration or reintroduction of species, non-chemical control of pest or disease outbreaks, prescribed burning or other fire management strategies to lessen the impact of increasingly severe and frequent wildfires, controlling invasive species and decreasing nutrient-enhanced run-off into marine and freshwater ecosystems. Where extinction in the wild is inevitable, *ex situ* conservation of species via the collection of germplasm, seedbanks, gardens, aquaria or zoos can be used as an option of last resort.

Regardless of the management strategy, on-going monitoring is essential to assure that actions are truly of the “do no harm” variety. Monitoring with adaptive management sets up an *in situ* experiment, providing data for modification of management strategies and allowing for exchange of results between protected areas for better strategy development world-wide.

Methodologies for Evaluating Climate Change Impacts

Chapters in this report evaluate existing and predicted effects of climate change on different biomes. Impacts that have already been seen are reported from case studies in the scientific literature. Predicted impacts are approached in a number of ways. At a large scale, it is possible to predict major shifts in biome types by combining biogeography models such as the Holdridge Life Zone Classification Model with general circulation models (GCMs) that project changes under a doubled CO_2 scenario. Biogeochemistry models simulate the cycling of nutrients between both biotic and abiotic components of the ecosystem for assessing the impacts of changes in temperature, precipitation, soil moisture, and other climatic factors that give clues to ecosystem productivity. Dynamic global vegetation models integrate biogeochemical processes with dynamic changes in vegetation composition and distribution. Comparing present trends in species and communities with paleological data also provides indications of how they will weather future climate change (Hansen et al., 2001).

The Role of the U.N. in Developing National Adaptation Plans

Under the United Nations Framework Convention on Climate Change and the Kyoto Protocol, countries are required to develop National Adaptation Plans. These plans must include strategies for natural systems, but very little work has been done thus far to ensure that these crucial natural resources are included in national adaptation strategies. It is WWF's hope that this manual will provide a tool to assist in completion of the biodiversity portion of National Adaptation Plans so that countries have strategies in place to buy time for biodiversity and natural resources. Numerous conversations are currently underway around the world noting the need for further support, both financial and intellectual, for adaptation strategies. While adaptation plans cannot in any way replace mitigation, they are vitally important to ensure that all countries receive the support necessary to protect their natural resources, especially those countries where such support is currently unavailable. Thus far support has been inadequate, especially for developing countries.

The Importance of Integrating Climate Change Threats Into Conservation Plans

Conservation planning is the key to protection of biodiversity and ecosystem function. The majority of planning to date has focused on issues relating to space; designing reserves to protect moderately "pristine" tracts of land or water. While we have protected only a fraction of the area needed to meet recommended spatial goals, we must also start addressing threats that originate outside reserves and protected areas. Environmental threats like climate change require that we extend conservation planning beyond the boundaries of protected areas, and into a future in which ecosystems and biomes may be quite different than they are today.

We must also realize that while it is incumbent on us to take action now to design, test and adopt conservation strategies that respond to climate change, these efforts are not the long-term solution. Even the best-designed approaches to increasing resistance and resilience to climate change will work only for changes of a few degrees at most. In essence, we are only buying ecosystems time, but time they desperately need while efforts are made to stabilize atmospheric concentrations of greenhouse gases and limit the rate and extent of climate change.

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Grasslands at a Crossroads: Protecting and Enhancing Resilience to Climate Change

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THE STRUCTURE AND FUNCTION OF the world's grasslands makes them one of the most vulnerable to global climate change of any terrestrial ecosystem (Sala et al., 2000; IPCC, 2001a). The low-stature of vegetation confers high light availability, leaving many grasslands naturally vulnerable to invasive species (Wagner, 1989), especially following disturbance (Hobbs and Huenneke, 1992). A critical role for fire in maintaining plant community structure in many grassland types leaves them vulnerable to vegetation change should changes to temperatures and precipitation occur that are sufficient to alter biomass and fire frequency. Grasslands are also vulnerable due to human environmental impacts, including conversion to agriculture (both cropland and grazing land) (Dale et al., 2000; Ricketts and Dinerstein, 2001), the introductions and spread of invasive species (Mack et al., 2000; Mooney and Hobbs, 2000), the proliferation of roads (Forman and Alexander, 1998; Forman, 2000), alterations to fire regimes (D'Antonio, 2000; Dale et al. 2000), and pollution that alters soil fertility and rates of plant growth (Schlesinger, 1997; Lejeune and Seastedt, 2001). Combine the above natural and anthropogenic vulnerabilities, and it is apparent that when faced with human-induced global climate change (Schlesinger, 1997; IPCC, 2001b), what remains of the world's grassland ecosystems as we know them are in trouble (Forseth, 1997).

Fortunately, however, the fate of grassland ecosystems faced with climate change, which has the potential to favor different groups of species and alter ecosystem processes, is not yet sealed. Although in some regions, habitats are far more degraded and in need of restoration and recovery than others (see Ricketts and Dinerstein, 2001 and the World Wildlife Fund's Ecoregions website, *http://www.worldwildlife.org/ecoregions/index.htm*, for an overview of the world's grassland types and assessment of their conservation status), scientists, land managers, and policymakers, with the help of private landowners, still have time to devise and implement adaptations that will be needed to protect and conserve grasslands from threats posed by global climate change. This chapter summarizes current scientific knowledge concerning potential steps that will be required to achieve this goal.

It first outlines the components of grassland ecosystems that will prove crucial to their resilience to climate change and describes human environmental impacts to them. Resilience is defined as the ability to withstand not only possible episodic climate changes, but also possible long-term directional changes (Malcolm and Markham, 1996); it may vary depending on the ecosystem component in question (Lavorel, 1999). The chapter then outlines potential impacts of predicted climate changes on grasslands given their current human-altered environmental condition. Finally, it proposes adaptations that may prove useful for preventing or minimizing these effects, and discusses complex challenges that could arise during translation of proposed strategies into the management plans and policies that will be required to maintain and restore grassland resilience.

Crucial Components of Grassland Ecosystems

The condition of vegetation and soils will prove critical to grassland resilience to climate change. Healthy, vigorous stands of native vegetation are likely to be more resilient to warming temperatures and increasing frequency and duration of droughts than degraded grasslands because their roots have access to deeper soil moisture and they are better able to compete with invasive species (Goodwin et al., 1999; Enloe et al., 2000; Stohlgren et al., 2001; Gelbard, 2003). Healthy plant cover is critical for intercepting rainfall, maximizing infiltration and soil water supply, reducing overland flow, and preventing nutrient losses due to erosion (Noss and Cooperrider, 1994). A healthy level of soil organic matter is important for soil aggregate formation, fertility, stability, water movement and holding capacity, and aeration, and therefore influences plant growth. Plant growth and vigor is largely determined by physical, biotic, and disturbance factors that limit primary productivity, including water availability (Sala et al., 1988), nutrient cycling (West, 1983; Schlesinger, 1997; Evans and Belnap, 1999), the suite of competing species (Grace and Tilman, 1990), healthy soil biota (Allen et al., 1992; Belnap and Lange, 2001), and the native disturbance regime (e.g., herbivory, fire; Mack and Thompsen, 1982; Collins, 1992; D’Antonio, 2000). Soil water-holding capacity and precipitation patterns are the major determinants of water availability for plants (Jenny, 1980), which in turn is the major determinant of plant growth (Schlesinger, 1997).

Abundance and relative composition of plant species are increasingly recognized to depend on various ecological processes such as herbivory and fire (Soule and Terbourgh, 1999; Ricketts and Dinerstein, 2001). Changes to the above ecosystem components due to climate change, human environmental impacts, or synergisms therein have the potential to influence grassland composition and/or function, and therefore resilience (Mack, 1989; Forseth, 1997; Mooney and Hobbs, 2000). The status of these ecosystem components differs among grasslands (Carpenter, 2001; Dellafiore, 2001; Ricketts and Dinerstein, 2001; Seymour and Rowan, 2001), depending on such factors as the degree to which their native species and soil biota are adapted to the current disturbance regime (Mack, 1989; Milchunas and Lauenroth, 1993).

Stresses and Vulnerabilities Due to Factors *Other* Than Climate Change

Numerous human disturbances and stresses are causing substantial degradation to the crucial components of grassland ecosystems outlined above (Walker, 1995). These include habitat fragmentation and loss, the spread of invasive species, alteration of fire regimes, and pollution; and all may act alone or synnergistically, leaving grasslands particularly vulnerable to climate change (Walker and Steffen, 1997; Dale et al., 2000).

HABITAT FRAGMENTATION AND LOSS

Throughout the world, grasslands have been degraded by livestock grazing, converted to agriculture and urban centers, and fragmented by roads (Dale et al., 2000).

LIVESTOCK GRAZING: Livestock grazing is undoubtedly the major land use of the world's remaining grasslands (IPCC, 2001a), and it is listed as a threat to these systems on all continents except for Antarctica (Richardson et al., 2000; Carpenter, 2001; Dellafiore, 2001; Dinerstein and Louks, 2001; Hobbs, 2001; Ricketts and Dinerstein, 2001). Its impacts are variable depending not only on the intensity, timing, duration, and type of grazing animal, but on environmental conditions and the ecosystem and plant life form in question (Mack, 1989; Noy-Meir et al., 1989; Milchunas and Lauenroth, 1993; Harrison et al., 2003). Livestock grazing diminishes grassland resilience by compacting soils, thus reducing infiltration and soil water capacity and drying out surface soils, which may increase vulnerability to drought and accelerate desertification (Schlesinger et al., 1990; Robertson, 1996, Fig. 1). It causes substantial degradation to riparian habitats (Belsky et al., 1999), which are hot spots of native biodiversity within grasslands, and provide connectivity between grassland patches (Noss and Cooperrider, 1994; Stohlgren et al., 1998). Livestock cause erosion and topsoil loss and destroy biological soil crusts (living crusts composed of lichens, mosses, algae, and cyanobacteria that cover the soil surface between individual grasses, shrubs, and trees in many semi-arid grasslands, and that are critical for maintaining soil fertility and plant vigor; Belnap and Lange, 2001) that may result in considerable nutrient losses (Archer and Smeins, 1991; Evans and Belnap, 1999; Belnap and Lange, 2001). They also cause localized increases in soil fertility where they concentrate, such as near water sources and under trees (Dalhgren et al., 1997; Belsky and Gelbard, 2000). Vegetation within such nutrient hot spots tends to be dominated by invasive species, and these sites may act as foci for further invasion (Belsky and Gelbard, 2000). The result of changes to soil fertility and structure caused by livestock, combined with seed introductions of invasive species and selective grazing of palatable species over weedy ones frequently results in reductions in both the vigor and reproduction of native species and an acceleration of the spread of invasives (Buffington and Herbel, 1964; Schlesinger et al., 1990; Archer et al., 1995; Belsky and Gelbard, 2000; Hobbs, 2001; IPCC, 2001a). These impacts leave grasslands highly vulnerable to climate changes that may favor invasive species (Baskin, 1998).

Fig. 1. Intensive sheep grazing favors exotic annuals such as *Erodium cicutarium* in Carrizo Plain National Monument (Fig. 1a; photo by J. Gelbard) and by causing terracing that compacts soils, reducing infiltration and increasing overland flow and topsoil loss (Fig. 1b; photo by J. Gelbard).

Figure 1a.

Figure 1b.

CONVERSION TO AGRICULTURE: Vast expanses of grassland habitat have been converted to agriculture, whose spread has often left remaining patches surrounded within a sea of crop fields, irrigated pastures, vineyards, and orchards (Noss and Cooperrider, 1994, Dale et al., 2000, Ricketts and Dinerstein, 2001). Grasslands from the Argentine Pampas (Dellafiore, 2001), to the Kazakh steppe (Ponomarenko, 2001), to the American tallgrass and Palouse prairies (Ricketts and Dinerstein, 2001), have been almost entirely converted to agricultural production, leaving few intact habitats remaining. Such isolation leaves grassland species little room to migrate in response to climatic changes. Agricultural impacts to grasslands stretch beyond habitat fragmentation and loss. For example, due to changes in the level of surface water evaporation, irrigated agriculture may alter regional climate in such a way as to influence plant community composition (Baron et al., 1998).

An impact to grasslands that often accompanies agricultural conversion is pesticide use. The application of toxic pesticides may poison soils, alter below-ground communities, weaken native plants, and lead to the loss of native species or invasions of different types of exotics (Frenkel, 1970; Myers and Kent, 1996; Tyser et al., 1998; Dale et al., 2000). Even supposedly weed-specific herbicides such as Clopyrid (Transline), which is used to control yellow starthistle *(Centaurea solstitialis)*, may lead to the loss of functional groups such as asters and legumes (DiTomaso et al., 2000), potentially reducing nitrogen availability and destabilizing plant community composition.

PROLIFERATION OF ROADS: Roads are the entry points for virtually all human environmental impacts to grasslands, including conversion to agriculture and subdivisions, livestock grazing, off-road vehicle (ORV) use, arson fires, and the spread of invasive species (Forman and Alexander, 1998). Of particular concern for grassland resilience, roads are the pathways by which invasive plants are spread away from ports and other original points of introduction and throughout a geographic region (Frenkel, 1970; Belsky and Gelbard, 2000; Gelbard and Belnap, 2003). For example, seeds of 124 plant species, a high proportion of which were exotic, were collected from an automobile as it was driven more than 15,000 km throughout Central Europe (Schmidt, 1989). In Australia, researchers collecting seeds from vehicles in Kakadu National Park found 14 exotic species not known to be present in the area (Lonsdale and Lane, 1994). Whether in Africa (Milton and Dean, 1998), the Middle East (Holzapfel and Schmidt, 1990), Europe (Schmidt, 1989), the United States (Gelbard and Belnap, 2003), or Australia (Lonsdale and Lane, 1994), this typically results in roadsides being highly invaded habitats. In Serengeti National Park, for example, road shoulders contained ruderal species that did not occur in naturally disturbed or climax communities (Belsky, 1987). In the western United States, the noxious weed, yellow starthistle occurred in 73% of quadrats 10 meters from roads, but only 21% of quadrats greater than 1000 m from roads (Gelbard and Harrison, 2003). Roadsides in the Middle East were dominated by Mediterranean species, while adjacent habitats were dominated by Saraho-Arabian species (Holzapfel and Schmidt, 1990).

Figure 2. Off-road vehicle (ORV) damage to plants and soils in a rare serpentine grassland.

Photo by J. Gelbard.

The construction and improvement of roads can be considered an important agent of land cover change. Gelbard and Belnap (2003) estimated that construction of 10 kilometers of paved roads converts as much as 12 hectares of natural habitat to highly invaded roadside habitats that can act as conduits for the spread of invasive species. Conversely, roadless habitats are often refuges for native species (Soule and Terbourgh, 1999), but such habitats are typically rare at the landscape scale. In an area of northern California, for example, grasslands greater than 1000 meters from roads composed only 1.5% of the landscape (Gelbard and Harrison, 2003). This study provided a striking illustration of how the expansion of a road network fragments and degrades grasslands, leaving them highly vulnerable to additional vegetation changes induced by climatic shifts.

An impact to grasslands that is directly related to roads is off-road vehicle (ORV) use, which damages vegetation and soils and spreads invasive plant seeds (Fig. 2; Webb and Wilshire, 1983; Noss and Cooperrider, 1994). Inadequate attention in management plans and insufficient enforcement of existing regulations have resulted in hundreds of thousands of kilometers of unauthorized ORV routes across grassland landscapes.

URBANIZATION: Throughout many areas of the world, grasslands occupy valley bottoms and other habitats of low topographic position, which are the most accessible for development (West, 1983). As a result, they have been especially vulnerable to habitat loss due to urbanization (Gerlach et al., 1998; Fig. 3).

Figure 3. Untold extents of grassland habitat have been lost to urbanization.

Photo by J. Gelbard.

SPREAD OF INVASIVE SPECIES

The introduction and spread of exotic and otherwise invasive plant species *(exotic* species are not native to an ecosystem, while *invasive* species are those that spread rapidly from original points of introduction) poses a considerable threat to many of the world's remaining grasslands. Invasive species possess traits that make them highly adaptive to climate change, including tolerance of a wide range of environmental conditions, resistance to human disturbances, and rapid rates of growth, seed production, and dispersal (Baskin, 1998; Dukes and Mooney, 1999; IPCC, 2001a). They include not only plants, but also pathogens such as barley yellow dwarf virus, which afflicts some native bunchgrass species and may act in conjunction with moisture stress caused by competition with exotic annuals to contribute to high bunchgrass seedling mortality (Malmstrom, 1998). Impacts of invasive species include species endangerment (Wilcove et al., 1998), reductions in biodiversity (Rosentreter, 1994) and wildlife habitat (Bedunah, 1992), alterations to ecosystem processes such as fire frequency (D'Antonio and Vitousek, 1992), and nutrient cycling and hydrology (Vitousek et al., 1997), increases in topsoil loss (Lacey et al., 1989), alterations to soil microclimate (Evans and Young, 1984), and reductions in land value and livestock forage capacity (Sheley and Petroff, 1999; Naylor, 2000). The most severe impacts of invasive species often occur where they alter the disturbance regime (D'Antonio, 2000; Levine et al., 2003). The invasive species problem can often be considered a subset of many of the human impacts described in this section (Archer et al., 1995; Knops et al., 1995; Belsky and Gelbard, 2000).

ALTERED FIRE REGIMES

Humans have caused dramatic alterations to grassland fire regimes (D'Antonio, 2000), which are often important determinants of plant community composition (Collins, 1992; Ricketts and Dinerstein, 2001; Keeley, 2001). In some grassland types, such as the Intermountain West, USA, shortening of the fire return interval due to invasions has favored fire-tolerant plant species, resulting in the loss of fire-intolerant bunchgrasses, shrubs, and associated wildlife (D'Antonio and Vitousek, 1992; Knick and Rotenberry, 1997; D'Antonio, 2000). Other impacts of increased fire frequency include accelerated nutrient losses through volatilization and mortality of biological crust organisms (Schlesinger, 1997; Belnap and Lange, 2001), and losses of rare species (Rosentreter, 1994). Fuel breaks created to suppress fire may further alter grasslands by acting as conduits for the spread of invasive species into uninfested habitats (Keeley, 2001). The area impacted by increased fire frequency may expand with climate change, and thus this threat can also be considered a potential climate change impact (Current and Future Stress, pg.39).

In other grasslands, human-caused decreases in fire frequency have negatively influenced ecosystem health, especially where dominant native species are more fire tolerant than invaders (e.g., Collins, 1992; Heady, 1995; Dale et al., 2000). In California grasslands, for example, low intensity fire can favor native bunchgrasses such as *Nassella pulchra* (Heady, 1995; DiTomaso et al., 1999; Keeley, 2001), while fire suppression results in the build-up of a thick layer of exotic annual grass thatch that hinders the establishment and reproduction of many native species (Menke, 1989; Heady, 1995).

POLLUTION

INCREASED ATMOSPHERIC CO$_2$: The rise in atmospheric CO$_2$ concentrations caused by human activities (Schlesinger, 1997) may have variable effects on grasslands, depending on environmental conditions such as soil fertility, climate, and the species mix

(Forseth, 1997; Dukes and Mooney, 1999; IPCC, 2001a). One effect of increased CO_2 concentrations is increased water use efficiency of plants, because the amount of time that they need to leave stomata open for CO_2 uptake is reduced (Forseth, 1997; Schlesinger, 1997). Experiments have shown that as a result, heightened CO_2 levels increase soil moisture, which may negatively impact grasslands where such conditions favor invasive species (Hobbs and Mooney, 1989; Dukes and Mooney, 1999; Dukes, 2000).

The effect of increased CO_2 on grasslands may also depend on the mix of C3 vs. C4 species in a community. Plants with the C3 photosynthetic pathway such as many bunchgrasses (e.g., *Stipa* spp.) and mesquite *(Prosopis* spp.) may increase photosynthetic rates up to concentrations of 1000 parts per million CO_2 and beyond (Pearcy and Ehleringer, 1983), and therefore may show enhanced growth rates as CO_2 concentrations increase. In contrast, plants with the C4 photosynthetic pathway, such as many *Bouteloua* species of the Great Plains, USA, are already saturated by CO_2 at current atmospheric levels and are expected to show little if any increase in growth with rising CO_2 levels (Woodward et al., 1991; Forseth, 1997). The result may be to favor C3 over C4 species, though the likelihood that this will occur remains uncertain (Walker and Steffen, 1997), and may depend on environmental conditions such as moisture availability (Forseth, 1997) and the current disturbance regime (Mack and Thompsen, 1982; Milchunas and Lauenroth, 1993). Since rising temperatures may increase terrestrial CO_2 outputs into the atmosphere by increasing decomposition rates, especially in northern latitudes where soils contain substantial carbon stores (Schlesinger, 1997), effects of rising CO_2 levels on grasslands may be augmented by climate change (Current and Future Stress, below).

NITROGEN DEPOSITION: Doubling of nitrogen (N) inputs into the terrestrial N cycle as a result of human activities is leading to accelerated losses of biological diversity among plants adapted to efficient use of N and animals and microorganisms that depend on them (Vitousek et al., 1997). Increased N availability degrades grasslands by favoring weedy species and reducing native species richness (Huenneke et al., 1990; Wedin and Tilman, 1996; Schlesinger, 1997, Weiss, 1999; Scherer-Lorenzen et al., 2000; Lejeune and Seastedt, 2001), especially in habitats characterized by higher temperatures and precipitation (Walker and Steffen, 1997). For example, Lejeune and Seastedt (2001) concluded that N deposition has been an important contributor to the spread of knapweeds *(Centaurea* species). Experimental studies have shown N addition to result in losses of species diversity in grasslands (Huenneke et al., 1990; Wedin and Tilman, 1996), especially when combined with disturbance (Hobbs, 1989), including by resulting in losses to mycorrhizal species that are important to native persistence (Egerton-Warburton and Allen, 2000). Effects of climate change on grassland community composition may be especially pronounced in grasslands impacted by N deposition (Zavaleeta et al., 2003).

CFCS AND STRATOSPHERIC OZONE DEPLETION: The loss of stratospheric ozone (O_3) due to human activities has resulted in increased UV-B radiation at the Earth's surface (Schlesinger, 1997). Plant response to UV-B radiation spans from little effect to large reductions in photosynthesis and growth, depending on environmental conditions (Forseth, 1997), but may result in changes in competitive balance in terrestrial plant communities (Barnes et al., 1988).

OTHER POLLUTANTS: Pollutants such as tropospheric ozone (O_3), which is phytotoxic, and sulfur and nitrogen oxides (SOx, NOx), which combine with rainwater to form acid rain, may weaken native species, reducing their ability to withstand some of the above stresses (Schlesinger 1997, Walker and Steffen, 1997). In more mesic grasslands where acid deposition results in losses of basic cations via leaching (Schlesinger, 1997), it may result in nutrient limitations that weaken native plants, increasing their vulnerability to stress induced by drought, disease, and competition with invasive species, and therefore to climate change.

Current and Future Stress and Vulnerability Due to Climate Change

PROJECTED CLIMATE CHANGES TO GRASSLAND ECOSYSTEMS

General circulation models (GCMs) predict grassland ecosystems to experience climatic changes including higher maximum (daytime) and minimum (nighttime) temperatures, and more intense precipitation events (IPCC, 2001b). They predict precipitation to increase over northern mid-high latitude grasslands with larger year-to-year variation, including increased frequency of droughts and floods associated with El Niño events, and project both regional increases and decreases for low latitudes (IPCC, 2001b). Within each grassland type, the nature and degree of impacts caused by climate changes is likely to depend on both environmental conditions and individual species or plant life form, due to differences in competitive abilities, migration rates, and responses to disturbance (Forseth, 1997; Dukes and Mooney, 1999; Malcolm and Pitelka, 2000; Buckland et al., 2001; Zavaleta et al., 2003). Location of sites within a grassland region may also be important, as ecotonal habitats are most likely to experience vegetation changes (Neilson, 1993). The degree to which human-caused disturbances and stresses (Stresses and vulnerabilities, pg.33) have already degraded resilience is also likely to prove critical to the future response (Forseth, 1997; Malcolm and Markham, 1996; Schlesinger, 1997). Unlike past climatic changes, the current suite of species in most grasslands often includes species intentionally and accidentally introduced from all over the world, leaving considerably greater potential for climatic shifts to cause or accelerate large-scale vegetation change (Mack, 1989; Mooney and Hobbs, 2000).

POTENTIAL IMPACTS OF INCREASED PRECIPITATION

In grasslands predicted to experience an increase in precipitation (IPPC, 2001b), such as those of the western United States, this may alleviate moisture stress resulting from rising temperatures. However, it may also accelerate nutrient cycling (e.g., N mineralization), increasing both productivity (Breymeyer et al., 1996) and the germination, seedling survival, and spread of many invasive species (Hobbs and Mooney, 1989; Baskin, 1998; Dukes and Mooney, 1999; Sutherst, 2000). In hot desert grasslands (e.g., within Australia, the Sonoran and Chihuahuan deserts of Mexico and the USA), GCMs predict an increase in the frequency of intense precipitation events and flash floods. This climatic prediction, combined with the likelihood that rising temperatures will increase oxidation of soil organic matter important to soil stability (Walker and Steffen, 1997), is likely to increase erosion and nutrient loss, especially in already-disturbed habitats. If such climatic changes are accompanied by a decrease in less-intense precipitation events, grasslands may experience a net loss in water availability, since in more arid systems, much of the water that falls in the form of intense events often runs through the system and is not absorbed.

POTENTIAL IMPACTS OF DECREASED PRECIPITATION

Other grasslands, including continental interiors such as the Canadian Prairie, Central Australia, the African Serengeti, and the Great Plains, USA, are expected to experience a *decrease* in precipitation (IPCC, 2001b). Possible consequences include decreased productivity and water availability and increased risk of wildfire (Baron et al., 1998; IPCC, 2001a), which may result in losses of rare species and soil biota, disruption of N cycling, and associated impacts to grassland flora and fauna (Knick and Rotenberry, 1997; D’Antonio, 2000; Belnap and Lange, 2001). Since biological soil crusts may act as a physical barrier to the establishment of some invasive species (Mack, 1989), large-scale reductions in crust cover due to increased frequency of drought and fire may accelerate the spread of exotic annuals such as cheatgrass *(Bromus tectorum)* (Stohlgren et al., 2001; Gelbard and Belnap, 2003). In other grassland habitats, however, including in northern Europe (Buckland, et al. 2001), South Africa (Richardson et al., 2000), and California (Hobbs and Mooney, 1989), an increase in the frequency and duration of drought may cause declines in the abundance of some exotic species, allowing for the recovery of some natives.

POSSIBLE SYNERGISMS AMONG CLIMATE CHANGES AND HUMAN ENVIRONMENTAL IMPACTS

Synergisms among climate change and other human impacts, such as among grazing, ORV use and increased frequency of drought, may also negatively impact grasslands. For example, effects of drought on vegetation are likely to be more severe where trampling disturbances have reduced infiltration and soil water capacity (Schlesinger et al., 1990; Baron et al., 1998). As noted above, warming temperatures and higher precipitation may increase biomass and therefore fire frequency and intensity, especially where grasslands are dominated by summer-dry exotic annuals (D’Antonio, 2000). One result may be a feedback in which increased fire frequency results in the displacement of native bunchgrasses and shrubs, opening still more habitat to invasion (D’Antonio and Vitousek, 1992). In other possible synergisms, the probability that increased precipitation and CO_2 concentrations (which are likely to further increase with climatic warming due to oxidation of soil carbon stores, especially in northern latitudes; Schlesinger, 1997) will alter plant community composition is likely to be highest in areas subjected to high nitrogen deposition (Dukes and Mooney, 1999; Sala et al., 2000, Zavaleta et al., 2003), while an increase in the frequency of drought may have particularly severe consequences for rare species (Tilman and El-Haddi, 1992), especially in the presence of human impacts to vegetation and soils described above. Of course, one never knows what surprises Mother Nature may spring upon grasslands in the future (Malcolm and Markham, 1996; Schneider and Root, 1996; Broeker, 1997), as potential climate changes and their consequences remain difficult, if not impossible, to predict with a high degree of certainty (Walker and Steffen, 1997).

Adaptation Options

Many possible strategies (adaptations) may prove useful for maintaining and restoring the resilience of grassland ecosystems to climate change.

Represent grassland types across environmental gradients in protected areas: Because we do not know precisely which grassland types will be most sensitive to climate change, maintaining all types in replicated protected areas across environmental gradients (to protect against the loss of individual reserves to catastrophic events), with management appropriate to the native disturbance regime, will help to ensure that resilient types persist (Noss and Cooperrider, 1994; Soule and Terbourgh, 1999; Dale et al., 2000). Reserve systems should include heterogeneous topographic, soil, and management conditions to maintain full suites of native species (Noss and Cooperrider, 1994; Markham and Malcolm, 1996; Halpin, 1997; Soule and Terbourgh, 1999, Fuhlendorf and Engle, 2001). They should also be designed to protect rare species and recognize where isolation may hinder their ability to migrate as the climate changes (Markham and Malcom, 1996)

Protect relict and native-dominated communities as appropriate per system: Maintenance of relict grasslands (Driscol, 1964; Mason et al., 1967; Jeffries and Klopatek, 1987; Ambos et al., 2000) is crucial because these communities both serve as models for habitat restoration and help us to understand how grasslands altered versus unaltered by human activity are affected by climate change. Since many relict communities are isolated in remote sites surrounded by altered habitats, they could prove vulnerable to climatic shifts (Halpin, 1997). Isolation may also have beneficial effects, however, such as by protecting relict communities from the introduction of exotic plant seeds and diseases that afflict native species (Gelbard, 2003).

Minimize fragmentation by land use changes and roads: The negative effects of ecosystem fragmentation are abundantly documented worldwide (Noss and Cooperrider, 1994; Soule and Terbourgh, 1999; Dale et al., 2000). Core grassland habitats distant from roads and human disturbances are often refuges for native species, but roads are so widespread that such habitats are typically rare at the landscape scale (Gelbard and Harrison, 2003). Conversely, fragmentation of grassland ecosystems by roads and land use changes contributes to reductions in native biodiversity, both by eliminating grassland habitats entirely and by facilitating the spread of invasive species (Forman and Alexander, 1998; Gerlach et al., 1998; Gelbard and Belnap, 2003). Activities that fragment grasslands should be minimized, while roadless and otherwise relatively intact core habitats should receive some form of protection (Noss and Cooperrider, 1994; Dale et al., 2000).

Connectivity is the antithesis of fragmentation. Corridors or stepping-stones of suitable habitat may facilitate the migration of species in response to climate change, and are important for maintaining species migrations and gene flow (Noss and Cooperrider, 1994; Dale et al., 2000). However, connectivity can also be detrimental where it provides a conduit for the spread of invasive species and diseases that afflict native species (Markham and Malcolm, 1996). In grassland ecosystems, connectivity should be considered where it is critical for maintaining gene flow among populations of rare species and in avoiding fragmentation (Noss and Cooperrider, 1994). Maintenance of intact riparian habitats—often hotspots of biological diversity in grasslands—is one approach for retaining connectivity among grassland patches (Noss and Cooperrider, 1994).

Practice low-intensity, sustainable grazing practices: Proper livestock management is critical for maintaining grassland components crucial to resilience. Grazing should be maintained in grasslands where native species are adapted to it (Baker, 1978; Mack, 1989), while land managers should consider reducing or removing grazing from marginal lands (e.g., semi-desert and desert grasslands) and systems where the predominant native species lack a long evolutionary history of grazing by large hooved herbivores (Mack, 1989; Milchunas and Lauenroth, 1993). Invaded or otherwise degraded grasslands are likely to pose more complex management challenges (Billings, 1990; Archer and Smeins, 1991; Brenton and Klinger, 1994), and may require active restoration. Within grazed grasslands, it is important to maintain heterogeneity of management at the landscape-scale and mimic grazing patterns of native herbivores to maximize native biodiversity (Noss and Cooperrider, 1994; Fuhlendorf and Engel, 2001). Such measures are likely to help promote the resilience of grasslands to climate change by helping to protect native species from invasions and by maintaining favorable soil nutrient and moisture conditions (Gelbard, 2003).

Prevent and control the spread of invasive species, including pathogens: Since invasive species may be favored over natives under many climate change scenarios, and cause substantial degradation to grasslands (Stresses and vulnerabilities, pg.33), preventing and controlling their spread is critical for maintaining grassland resilience (Baskin, 1998; Mack et al., 2000; Mooney and Hobbs, 2000).

Grassland restoration: Where possible, land managers should conduct grassland restoration, including reintroductions of native species, eradication or control of invasive species, inoculations with soil biota important to native plant vigor, nutrient cycling, and decomposition (e.g., mycorrhizae, biological soil crust organisms), and restoration of native disturbance regimes (Soule and Terbourgh, 1999).

Maintenance of natural fire regimes: Fire regimes exert a profound influence on the health and heterogeneity of grassland vegetation (Collins, 1992; Hartnett et al., 1996; D'Antonio, 2000; Ricketts and Dinerstein, 2001). Maintaining or restoring native fire regimes will be an important component of efforts to maintain and restore grassland resilience (Collins, 1992; D'Antonio et al., 2001; Dale et al., 2000).

Provide buffer zones: The fixed boundaries of protected areas are not well-suited to a dynamic environment unless individual areas are extremely large (Noss and Cooperrider, 1994; Halpin, 1997; Soule and Terbourgh, 1999). With changing climate, buffer zones might provide suitable conditions for shifting of populations to lands bordering reserves as conditions inside reserves become unsuitable. Another function for buffer zones may be to act as barriers to the spread of new invaders away from roads. For example, establishing strips of dense, ungrazed perennial bunchgrasses, or rocky, infertile substrates may help prevent roadside verges from acting as conduits for the spread of a recently introduced exotic species into the adjacent plant community (Roche et al., 1994; Enloe et al., 2000; Gelbard and Belnap, 2003).

Identify and protect functional groups and keystone species: Maintaining the natural diversity of species and functional groups, and dominance (e.g., cover) of native species, is a sound overall strategy for enhancing resilience to climate change (Malcolm and Markham, 1996; Dukes, 2002). Several recent studies have demonstrated increased tolerance to environmental extremes and recovery potential as native species richness or cover increases. For example, Dukes (2002a) found that yellow starthistle *(Centaurea solstitialis)* grown in monoculture responded strongly to CO_2 enrichment, increasing aboveground production by 70%, but when grown in competition with common grassland species, showed parallel, but non-significant increases. In some grasslands, native abundance may also increase resistance to invasion, especially at the neighborhood scale (Levine, 2000; Gelbard, 2003). For example, plots dominated by established monocultures of the native perennial grass, *Nassella pulchra*, along with the late season annual forbs, *Hemizonia congesta* and *Lessingia hololeuca* (Dukes 2002b) resisted *Centaurea* invasion. These species, like starthistle, complete their life-cycles late in the growing season and utilize deep soil moisture, suggesting that plant communities are most resistant to invasion where they contain a high abundance of native species with similar life-history characteristics to introduced exotics (Roche et al., 1994; Dukes, 2001, 2002b). It follows that maintaining not only native richness or cover, but also functional group richness, may increase the stability of grassland ecosystems, by increasing the likelihood of protecting native species and life forms with similar life history characteristics to introduced invaders.

Protect climatic refugia at multiple scales: It makes sense to identify past climatic refugia wherever possible and focus conservation efforts on these areas so they can again function as refugia during present and future periods of climate change (Noss, 2000).

Strategies for Protecting and Managing Grasslands Faced with Climate Change

The appropriateness and feasibility of implementing the above adaptations will vary considerably among grassland types, depending on the evolutionary history of their predominant species and the current vulnerabilities of the system. Below, guidelines are provided to help direct decisions concerning how to maximize grassland resilience to climate change.

LANDSCAPE LEVEL LAND-USE PLANNING

Implementation of many of the proposed adaptations for maintaining and restoring resilience will require conservation-oriented planning that allows examination of local decisions in a regional context (Soule and Terbourgh, 1999; Dale et al., 2000). Optimally, counties, states, conservation organizations, and land trusts should work cooperatively to develop large-scale land use plans that aim to achieve the complementary goals of determining which habitats should receive some form of protection, and which lands are less biologically important and thus suitable for development (Noss and Cooperrider, 1994; Soule and Terbourgh, 1999; Dale et al., 2000).

One objective of landscape planning should be to represent each grassland type, especially its relict communities and sites that contain rare species, in replicated protected areas across environmental gradients, with management appropriate to the native disturbance regime (Current and Future Stress, pg.39). The degree to which this option is feasible will depend on the proportion of habitat that remains intact. Where few if any large patches of native habitat remain, this option will not be feasible. However, where either intact habitats, or degraded, but undeveloped sites suitable for restoration are available, they can be prioritized to receive some form of protection (Noss and Cooperrider, 1994). Additional adaptations that can be considered during the landscape planning process are described below.

PREVENT AND SLOW THE PROLIFERATION OF INVASIVE SPECIES

When it comes to buffering grasslands against global climate change, a significant challenge will be to protect native communities from the threat of invasive species (Stresses and vulnerabilities, pg.33; Current and Future Stress, pg.39). By virtual consensus, invasion ecologists agree that strategies for achieving this goal should focus on the causes of invasions (such as seed sources and disturbances that increase vulnerability to invasion), not just the invasions themselves (Hobbs and Humphries, 1995; Sheley and Petroff, 1999; Mack et al., 2000). Just as individuals who pay attention to threats posed by an illness after they have become sick are more likely to suffer reduced personal health, resulting in the need for potentially expensive medical treatments, grassland managers who pay attention to threats posed by an invasive species only after their lands have become invaded are more likely to suffer reduced ecosystem health, resulting in the need for expensive control treatments. It is increasingly apparent, moreover, that when it comes to confronting the invasive species problem, an aggressive, prevention-oriented, and adaptive approach is favorable not only ecologically, but also economically (Hobbs and Humphries, 1995; Randall, 1996; Sheley and Petroff, 1999; Mack et al., 2000; Naylor, 2000).

A prevention-oriented approach requires stemming the influx of propagules via the numerous vectors that managers can control, such as vehicles, livestock, outdoor recreationalists, and those related to international trade and horticulture (Mack and Lonsdale, 2001) (it would be virtually impossible to prevent seeds from being introduced via natural vectors such as streams, wind, and native wildlife). This will require persistent monitoring of grasslands adjacent to roads and trails, and in pastures and other hot-spots of seed introduction, as well as rapid eradication of incipient infestations. In addition, it will require coordination with neighboring landowners to prevent adjacent lands from providing seed sources for recolonization.

Preventative management also requires minimizing disturbances that increase grassland vulnerability to invasion, especially those caused by livestock, ORVs, road maintenance operations, and outdoor recreationalists (Sheley and Petroff, 1999). Although the influence of any one of these factors may differ depending on such factors as soil fertility and moisture (Gelbard and Harrison, 2003), and the ecosystem or plant community type (Stohlgren et al., 2001; Harrison et al., 2003) in question, all that are affecting the vulnerability of a site to invasion should be addressed. Examples of possible measures are noted in the paragraphs below.

Attention to detail, as well as political willpower, will make or break invasive species management efforts. For example, both opportunities and restrictions posed by natural climatic variation need to be considered. “Free” years of control provided by climatic extremes such as drought (for some exotic species; e.g., yellow starthistle) and exceptionally wet years (for others; e.g., barb goatgrass; *Aegilops triuncialis)* should be aggressively followed up with at least a second year of eradication treatments, while reseeding of competitive native species may only succeed during exceptionally wet years. The responsiveness of adaptive management, especially the efficiency with which monitoring results are translated into on-the-ground changes, will prove critical to success as managers seek to prepare for the possibility that climate change will accelerate the spread of invasive species.

PREVENT AND MINIMIZE ROAD IMPACTS RELATING TO INVASIVE SPECIES

Roads are a major key to maintaining grassland resilience to climate change. One of the main reasons for their importance is that climate change is predicted to accelerate the spread of invasive species, and roads provide a major conduit for invasion (Stresses and vulnerabilities, pg.33; Current and Future Stress, pg.39;Strategies for protecting and managing grasslands, pg. 44). Measures that may prove useful for minimizing road impacts include:

Avoiding road construction in roadless habitats and on vulnerable (i.e., more fertile) soil types (Gelbard and Belnap, 2003; Gelbard and Harrison, 2003) to minimize the introductions and spread of invasive species and avoid creating corridors for ORV use and (intentional or accidental) setting of fire;

Carefully timing road maintenance to avoid favoring invasive species and spreading their seeds (Benefield et al., 1999);

Using native species in soil stabilization and revegetation operations to avoid the likelihood of introducing species that could invade neighboring grasslands under changing climatic conditions (Bugg et al., 1997);

Ensuring that roadfill used in road maintenance operations is not contaminated with weed seeds;

Conducting aggressive monitoring of roadside vegetation. This would allow for adaptive management in which roadside maintenance activities can be experimentally manipulated to work toward decreasing both the susceptibility of roadside habitats to invasion and the likelihood that roadside invaders will spread into adjacent natural ecosystems (Gelbard and Belnap, 2003).

IMPLEMENT SCIENCE-BASED AND ADAPTIVE LIVESTOCK MANAGEMENT STRATEGIES

Since livestock grazing is the predominant use of the world's remaining grasslands and can cause profound ecological impacts (Stresses and vulnerabilities, pg.33; Current and Future Stress, pg.39), this measure will prove critical for maintaining resilience. Too often, grazing practices fail to consider the best available scientific advice concerning how to graze livestock in such a manner as to minimize negative ecological consequences (Archer and Smeins, 1991; Noss and Cooperrider, 1994; Belsky and Gelbard, 2000; Dale et al., 2000; Hobbs, 2001). For example, numerous studies have demonstrated the important role of timing when using grazing as a weed control tool (summarized by Tu et al., 2001), but grazing on commercial ranches is typically continuous (Wagner, 1989), with livestock rarely if ever removed or rotated during times known to favor invasive species. In many cases, this is likely due to a lack of flexibility on commercial ranches (Tu et al., 2001), but in others it may simply be due to lack of awareness of timing effects. Appropriate grazing strategies will differ by ecosystem, and local environmental conditions (Harrison et al., 2003), but may include:

Limit grazing to habitats where native species and soil biota are adapted to it (Baker, 1978; Mack, 1989), and to degraded habitats where invasions are likely to worsen following livestock removal (e.g., Brenton and Klinger, 1994). Nurturing the recovery of soils and native vegetation following livestock removal requires a long-term view, especially in more arid systems (e.g., Brejda, 1997; Anderson and Inouye, 2001; Valone et al., 2002). Native recolonization may require not only a native seed source, but also climatic conditions that occur only once every decade or longer. Recovery of many native species may also be more pronounced on sites characterized by favorable moisture conditions (Billings, 1990; Belsky and Gelbard, 2000). For example, after 58 years of protection from grazing in the western United States, native plant density and cover increased on northern exposures, although not on southern and western exposures (Monsen, 1994). The same holds true for declines in exotic species following livestock removal, as demonstrated in British Columbia, where cheatgrass began to decline only after 30 years of exclusion (McLean and Tisdale, 1972).

Design management on a site-specific basis depending on soil and moisture conditions, the predominant native species, life forms, and exotics that pose a threat (Noss and Cooperrider, 1994; Randall, 1996; Holecheck et al., 1998). Altering the timing, intensity, and duration of grazing as well as the type of grazing animal may all help to minimize negative impacts to native species and soils (Archer and Smeins, 1991; Holecheck et al., 1998; DiTomaso, 2000; Belnap and Lange, 2001; Masters and Sheley, 2001);

Conduct intensive monitoring as part of adaptive management to minimize excessive disturbances to vegetation and soils (Noss and Cooperrider, 1994).

By preventing and minimizing selective grazing of palatable native species, limiting the extent of trampling disturbances that reduce soil water capacity and injure native plants, maintaining nutrient cycles and biological soil crusts, minimizing the spread of invasive species, and maintaining native disturbances regimes, the above livestock management measures will help to protect and enhance grassland resilience, including to climate change.

GRASSLAND RESTORATION

The vast area of grassland habitats that are currently degraded undoubtedly makes the task of restoration appear daunting. In grasslands where exotic annuals have overrun large expanses of habitat, such as those in Australia (Hobbs, 2001), Chile (Arroyo et al., 2000), California (Heady, 1995), and the Intermountain West, USA (Mack, 1989), a long-term, multi-pronged strategy involving restoration of the native disturbance regime (including letting natural fires burn, conducting controlled burns, and, where necessary, removing livestock), re-introductions of native species and soil biota, and eradication or control of invasive species will likely be needed (Soule and Terbourgh, 1999; U.S. Bureau of Land Management [BLM], 1999; Belnap and Lange, 2001).

Of course, the appropriate tools may or may not be available, depending on the location of a site (Soule and Terbourgh, 1999). For example, controlled burning is not feasible and may be illegal in more developed or air pollution-sensitive areas. It may also require some type of incentive if it is to be conducted on many private lands, where ranchers lose that forage for a season. In addition, since natural fires sometimes hinder restoration and recovery (D’Antonio, 2000; Harrison et al., 2003), some degree of fire suppression may still be required (Soule and Terbourgh, 1999). In grasslands that remain relatively intact, such as those of the Great Plains, USA (Ricketts and Dinerstein, 2001), management improvements combined with native reseeding and weed control are more likely, in the short-term, to restore grassland components important to resilience. Since the science of grassland restoration remains in its early stages (Young, 1999), an adaptive approach will almost certainly be needed to restore and maintain the composition and structure of grasslands faced with climate change.

Recommendations for Action and Policy

Policy and management actions required to implement the above strategies include measures that can be implemented at international, federal, state, county, community, and individual levels. At the international level, for example, considerable changes to trade policies will be needed to help stem the introductions and spread of invasive species, and such policies will need to be vigorously defended in both biological and economic terms in World Trade Organization dispute panels (McNeely, 2000; Van Driesche and Van Driesche, 2001). At national, state, and local levels, appropriate measures will often differ between public and private lands, and perhaps among cultures (Dale et al., 2000).

Private landowners will need financial impetus and know-how to implement the suggested adaptations. It will be important for governmental agencies and non-governmental organizations (NGOs) to use incentives to convey a sense that changing management will provide an opportunity to improve economic well-being and quality-of-life. Conversely, incentives that accelerate habitat loss by encouraging suburbanization, such as tax deductions for home mortgage interest payments, and duties that discourage conservation, such as expensive estate taxes that landowners can sometimes only pay by subdividing grasslands, should be replaced with incentives for conservation (Myers and Kent, 1998; Dale et al., 2000). The same holds true for subsidies for agriculture, water use, and roadbuilding throughout the world, without which many grassland habitats could not affordably be converted to or maintained as agriculture, overgrazed, or roaded (Myers and Kent, 1998).

It will also be important to educate the public about the availability of incentive programs. Both NGOs and governmental agencies will prove key as grassland owners seek assurances that taking steps to increase resilience will not threaten their livelihoods. Financial and quality-of-life concerns must be addressed by providing both direct (e.g., conservation easements, conservation rewards, safe harbor agreements, habitat conservation plans, grants, cost-share programs) and indirect (e.g., education, technical support) benefits to make grassland conservation and restoration economically and socially desirable, and thus more politically feasible (McNeely, 2000; Dasgupta et al., 2000; Dale et al., 2000; Naylor, 2000).

On public lands, implementing the recommended strategies for maintaining resilience will require changes—some major and some minor—to existing land use policies and management plans (Dale et al., 2000). For example, subsidies, without which livestock grazing on marginal habitats would not be economically feasible for many ranchers in the western United States, should be eliminated to allow market forces to influence the decision of whether grazing can continue (Myers and Kent, 1998). Similarly, cuts to government spending on the maintenance of existing, but non-essential roads could provide an incentive to close seldom-used roads that may still be acting as corridors for the spread of invasive species. In addition, an incentive to change road maintenance schedules to avoid exacerbating invasions could be created by making eligibility for international or federal transportation grants contingent on implementing the proposed adaptations. This could help, for example, to stem the problem of roadside mowing to prevent fire often resulting in the dominance of roadside grasslands by noxious weeds (Benefield et al., 1999).

Implementing the proposed adaptations will undoubtedly pose considerable challenges. The most effective options for maximizing resilience may not always appear socioeconomically feasible, may not be legal, and may meet with considerable political resistance (Lee, 1993; Walker and Steffen, 1997; Soule and Terbourgh, 1999). Take for example the issue of restoring native fire regimes. From a scientific standpoint, a mixed strategy in which managers let natural fires burn and conduct controlled burns in systems where frequent fire is important for maintaining native vegetation, but suppress fire in grasslands where it is naturally infrequent (e.g., Sagebrush/bunchgrass steppe; West, 1983), might be appropriate. In the short-term, however, translating this scientific information into appropriate fire policy and management decisions is sure to prove contentious. While economic benefits of such an approach (e.g., reduced cover of invasive species, increased forage quality) could increase public support for this type of fire policy in the long-term, such adaptations are unlikely to succeed without broad public support from the beginning (Soule and Terbourgh, 1999). Thus, in addition to devising management strategies to maintain and restore resilience, ecologists and land managers will need to work together with economists, sociologists, communication specialists, policymakers, and the public to devise economic and public relations strategies aimed at gaining the support of skeptical policymakers and landowners (Lee, 1993; McNeely, 2000; Robertson and Hull, 2001; Dale et al., 2000).

Communication and education strategies may include illustrations of how delaying management improvements might prove more costly than taking immediate action (e.g., Walker and Steffen, 1997; Naylor, 2000; Ayres, 2001). Management guidance and education materials can be made available via direct mailings, over the internet via public outreach, at local farmers' markets and community centers, and at meetings of involved stakeholder organizations such as commerce, farming, ranching, and outdoor recreation associations.

Funding for devising and implementing adaptations, typically a critical factor limiting the effectiveness of such functions as adaptive management, may come from international organizations such as the United Nations (e.g., UNEP), International Union for the Conservation of Nature, the World Commission on Protected Areas, and the Global Environment Facility. National organizations such as the United States Invasive Species Council, state-level agencies such as agriculture and transportation departments, and private foundations may also provide financial support. Technical expertise for implementing adaptations could be provided by groups such as NGOs, and university experts and graduate students.

Examples of Existing Adaptations

Changes in grassland management that promote resilience are already being implemented on both public and private lands. In the southwestern United States, the Malpais Borderlands Group—a coalition of ranchers, state and federal agencies, The Nature Conservancy, and private foundations—have banded together to devise a grass-banking approach to conservation and restoration. Ranchers whose lands have suffered sufficient declines in productivity to limit their ability to graze livestock are now threatened with the prospect of subdividing their lands to provide needed income. However, by joining the group, they can gain access to a grass bank on which to graze livestock—the 110,000 hectare Gray Ranch—and receive restoration and recovery treatments such as native reseeding and rest from grazing, as well as technical assistance and monitoring from grassland management experts, in exchange for taking steps such as (1) donating a conservation easement on their land to the group, which conserves grassland habitat by maintaining their ranch as open space, while providing economic benefits such as reduced estate taxes; and (2) allowing natural fires to burn, which helps beat back invasive woody plants and stimulates the recovery of native grasses and wildflowers (Page, 1997). Of course, risks such as degradation of the grassbank, itself, will need to be avoided, and this approach to grassland conservation and restoration remains an experiment in progress. However, the Malpais Group has clearly devised an innovative approach for landscape-scale conservation of privately owned grasslands, one that can certainly be modified to serve as the basis for other large-scale private lands conservation efforts.

An emerging approach with intriguing prospects for improving the health of publicly-owned grasslands is voluntary retirement of public lands grazing permits in the western United States (Salvo and Kerr, 2001). To encourage livestock removal from public lands where soils and native species are vulnerable to or have been degraded by livestock impacts, NGOs are seeking to facilitate the purchase of grazing permits for above-market value. This program would provide an opportunity for private land owners to cash in on the value of their public lands grazing permits, without the loss of their public grazing leases increasing pressure on them to subdivide and develop their privately-owned parcels. Removing livestock from these public lands, in turn, would reduce the considerable deficits that the federal government incurs through its public lands grazing program (Myers and Kent, 1998), including by (1) maintaining livestock infrastructure on marginal lands, (2) repairing livestock-caused degradation, and (3) controlling predators. It would increase resilience to climate change by allowing for restoration of native disturbance regimes, plant communities, and soils on grasslands where livestock are an exotic ecological force.

Finally, the practice of grassland restoration is accelerating (Harnett et al., 1996; DiTomaso et al., 1999, 2001; US BLM, 1999). Due to the increasing realization that establishment of native species will require restoration of native disturbance regimes, land managers are conducting controlled burns and reintroducing native herbivores (Menke, 1992; Harnett et al., 1996; DiTomaso et al., 1999). Although many of these practices require continued research, preliminary results are encouraging (e.g., D’Antonio et al., 2001). For example, three years of controlled burns almost entirely eliminated the cover and seed production of yellow starthistle and increased native bunchgrass cover by more than five-fold (DiTomaso et al., 1999). In a tallgrass prairie, bison grazing and prescribed fire resulted in significant increases to plant species diversity and spatial heterogeneity (Hartnett et al., 1996).

By recognizing that land management should differ among grasslands depending on their natural disturbance regime, environmental conditions, and conservation status, by reigning in destructive land use practices, and, where possible, by restoring habitats, including through the use of incentive-based conservation strategies, scientists, land managers, policymakers, and the public can work together to maintain and restore the resilience of grassland ecosystems to climate change. Of course, this chapter has provided only a general overview of potential adaptations, and the ability of the proposed strategies to confer resilience is not limitless in relation to climate change. For example, while these measures are most likely to be effective, in the short-term, in less degraded grasslands, they may take decades to restore resilience in thoroughly degraded grasslands. Even then, if governments and corporations continue to resist the need for decisive measures to address the ultimate causes of climate change, such as the growing human population's greenhouse gas emissions and conversion of natural habitats to human uses (Schlesinger, 1997), there may be little that land managers can do to prevent continued degradation, fragmentation, and loss of grasslands. At a time when humanity has reached a crossroads, in one direction faced with a growing extinction crisis that is certainly affecting grasslands (Sala et al., 2000), while in the other faced with the possibility that environmental solutions abound with opportunities to improve economy and quality-of-life (Myers, 1996; Daily et al., 1997; Myers and Kent, 1998; Dasgupta et al., 2000), we owe it to ourselves and to future generations to protect and maintain the world's grasslands from the potential consequences of global climate change.

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Forest Ecosystems Threatened by Climate Change: Promoting Long-term Forest Resilience

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CURRENT RATES OF CLIMATIC WARMING are the highest they have been in the last 10,000 years (IPCC, 1996a). Against this backdrop are forests and forest ecosystems, which have persisted for hundreds of millions of years. During this time major fluctuations in the climate have caused vegetation to modify their composition, structure and function, or risk extinction. Forests as a biome have tolerated such climatic changes through their ability to migrate, relatively unencumbered, to suitable new habitat. Past changes have also occurred at a much slower pace than those seen today, allowing forests time to adapt. Many of today's forests, however, have undergone serious fragmentation and degradation from roads, agriculture and development, and are thus impeded in their ability to migrate as their local climate changes (Noss, 2000). It is in combination with these threats that the impacts of unprecedented rates of climate change can compromise forest resilience, and distribution (IPCC, 2001).

The following sections lay out the chief stresses to each of the different forest types from tropical regions to boreal forests, as well as the crucial components of each system that must be maintained for healthy ecosystem functioning. Then evidence of current impacts on each major forest type is reviewed, together with projections of likely future impacts determined from General Circulation Models (GCMs) and other scientific research. Analysis of these existing and future non-climate threats applied together with an overlay of climate change impacts for a particular forest type or ecoregion will help conservationists understand what protection and other management options are available to enhance forest resilience in the face of climate change.

Stresses and Vulnerabilities Due to Factors Other Than Climate Change

Forests cover almost a third of the Earth's landmass, representing the most substantial reservoir of terrestrial biological diversity. Scientists estimate that forests contain as much as two-thirds of all known terrestrial species (FAO, 2000). However, many forest-dwelling large mammals, half the large primates, and nearly 9% of all known tree species are at some risk of extinction (WRI, 2000).

About half of the world's original and intact "frontier" forest has been lost since the dawn of agriculture some 8,000 years ago (FAO, 2000). Just one fifth of original forests exist in large and relatively natural ecosystems (Dirk et al., 1997). Most forest loss has occurred in the last three decades, largely due to human impacts. Deforestation has been most absolute in the Temperate Zone, where only a fraction of the original intact forest still remains in scattered fragments (FAO, 2000). Close to 15 million hectares of largely tropical natural forests are lost each year, equal to an area the size of Nepal (FAO, 2000). In addition, 1.5 million hectares are converted to forest plantations every year. The major cause of forest loss is conversion to other land uses as well as fragmentation from logging, agriculture, and settlement. Scientists assert that this has already resulted in the beginning of the 6th great extinction, and the first to be perpetuated by humans (Chapin et al., 2000).

Although forest cover is stable in the boreal region, and slightly increasing in the temperate zone after prior deforestation, these trends hide the fact that the *quality* of these forests has deteriorated drastically, impoverishing wildlife habitat for species dependent upon intact forest and unraveling ecosystem functions (FAO, 2000).

An overview of existing and future non-climate threats to each of the major forest types is below, followed by a summary of possible or likely impacts from climate change itself. A broad-brush analysis of these threats in combination with a discussion of the crucial components for each forest type provides a baseline for evaluating how resilient each type will be in the future once an overlay of probable climate change impacts are applied. Examples of threats to particular forest ecoregions are given in some instances to illustrate a more detailed level of analysis of vulnerability to climate change.

TROPICAL AND SUBTROPICAL MOIST BROADLEAF FORESTS

Tropical and Subtropical Moist Forests are extremely sensitive to disturbances such as plowing, overgrazing, and excessive burning due to limiting climatic and soil conditions. The dominant current threats to this forest type aside from climate change include conversion to agricultural land as well as large-scale commercial logging. An area the size of Ireland is lost due to conversion every few years within the tropical and subtropical region (FAO, 2000). This forest loss, combined with the introduction of exotics on islands, is the primary driver behind a high rate of species extinction (WWF, 2003a).

Looking at the immediate pressures to the health and functioning of a specific ecoregion is necessary before applying an overlay of climate change scenarios in order to help identify management options. The Eastern Arc Montane Forests of Kenya and Tanzania are among the oldest mountain ranges in Africa, and provide an illustrative example. As is true with most mountainous regions in Africa, this forest system is isolated from other similar areas by great expanses of lowland habitats. Isolation has produced a high level of endemism with many local species of plants and animals restricted to single mountain ranges. These microhabitats have experienced relatively moist conditions for long periods of time, even during drying trends throughout much of the rest of Africa. Very little of the Eastern Arc Montane Forests is under protected status, though all of the mountains are under pressure from agricultural expansion on lower slopes, firewood collection, and grazing (WWF, 2003b). Careful analysis of these existing and future non-climate threats applied together with an analysis of climate change impacts will help conservationists understand what protection and other management options are available to augment migration possibilities for endemic species that may be challenged with excessive changes in temperature and precipitation regimes within this ecoregion.

TROPICAL AND SUBTROPICAL DRY BROADLEAF FORESTS

Tropical and subtropical dry forests require large natural areas in order to maintain large predators and other vertebrates, to buffer sensitive species from hunting pressure, and to absorb occasional large fire events. Many dry forest species are reliant upon water sources and the persistence of riparian forests. In general, dry forests are extremely sensitive to excessive burning and deforestation, overgrazing and the introduction of exotic species. Restoration is possible though can be particularly challenging in intensely degraded areas. (WWF, 2003c)

In the Mexican Dry Forest, a growing urban population, increasing tourism and exploitation of wildlife are extremely serious threats to the ecoregion. Changing land uses for road construction, perennial plantations, and ranch farms are also a source of concern. (WWF, 2003d) A detailed layout of these threats, together with information on the distribution of sensitive species would serve as a preliminary step to analyzing this ecoregion’s vulnerability to climate change.

TROPICAL AND SUBTROPICAL CONIFEROUS FORESTS

Healthy tropical and subtropical coniferous forests are of sufficient size to withstand disturbances such as fire, windthrow, and outbreaks of disease. Several species, as well as successional processes are dependent on fire, however. Late-successional species, which are highly sensitive to logging and fragmentation due to slow regeneration rates, require special attention in management plans. Exotics also can have extensive impacts on these forests.

The Mesoamerican Pine-Oak Forests of El Salvador, Guatemala, Honduras, Mexico, and Nicaragua contain among the world’s most extensive subtropical coniferous forests, with many plant and animal species limited to particular locales. This ecoregion is also under pressure from fragmentation due to commerical logging, overgrazing by livestock, and conversion to agriculture.

TEMPERATE BROADLEAF AND MIXED FORESTS

Temperate broadleaf and mixed forests have been degraded and deforested to a great extent through habitat conversion, resource extraction, and through the introduction of exotic species (Wilcove et al., 1998). Carnivores inhabiting these forests have large home ranges, and forests must be of adequate size to maintain resilience to large-scale disturbance events such as fire. Late-successional forests are vital for the survival of many plants, lichen, fungi and invertebrates in this ecosystem. (WWF, 2003e) Some species inhabiting these forests show great sensitivity to fragmentation. This list includes breeding songbirds exposed to parasitism or elevated nest predation, as well as many forest understory species whose migration to other suitable forest is hindered by deforested areas. The loss of large native predators has substantial cascading effects on forest structure and ecology; and the introduction of exotic species can have large impacts on native communities as well. However, restoration potential for this forest type is high.

The Southwest China Temperate Forests, home to the giant panda *(Ailuropoda melanoleuca)* and many other rare species, contain some of the richest assemblages of temperate forest trees in the world and exhibit high endemism and unusal biogeographic patterns. Conversion to agriculture and fragmentation and degradation for timber are the principal threats to this ecoregion. (WWF, 2003f) A map of the distribution of where fragmentation and degradation is occuring, together with an analysis of crucial habitat will provide the baseline necessary to analyze climate change threats so that conservation plans can be adjusted accordingly.

TEMPERATE CONIFEROUS FORESTS

Temperate coniferous forests likewise require sufficient patch sizes in order to maintain larger carnivores which are extremely wide-ranging, as well as species that track resources that may vary widely in time and space with disease outbreaks, fires, and cone production. Large carnivores are highly sensitive to encroachment upon their home ranges, and many species are sensitive to habitat fragmentation, particularly of late-successional species which may regenerate slowly. Fires are necessary in many temperate forests in order to maintain successional processes. (WWF, 2003g)

The Altai-Sayan Montane Forests are experiencing a number of threats, including clearance of the forest, over-collection of plants and hunting along the banks of larger rivers and in heavily populated areas such as the Kusnetsk Basin, Salair, Alatau Kuznetsk and southwestern Altai. Overgrazing and associated erosion is a problem in some alpine and sub-alpine areas, as is mining and the threat of extensive wildfires that have engulfed large areas within the ecoregion. (WWF, 2003h)

BOREAL FORESTS

Boreal forests suffer from a variety of atmospheric threats unrelated to climate change, such as acid rain, UV-B radiation (from depletion of atmospheric ozone), and smog. There are also direct threats, including degradation from logging, mining and oil extraction, pipelines, and roads. Extreme weather events, such as ice storms, are highly damaging to trees (and could become increasingly prevalent as climate patterns shift).

At the same time, it is important to take into account those aspects of boreal forests that are crucial to its long-term viability. Natural disturbances such as fires, which cycle nutrients, and pests and diseases, which allow the introduction of early successional species by killing off adult trees, are often carefully balanced. (Sekula, 2000). In the Canadian Boreal Forests, man-made activities threaten the extent of the forest. Mining activities in the north, and logging in the southwest are constraints to healthy ecosystem function.

MEDITERRANEAN FORESTS

Mediterranean forests have largely been degraded through conversion to agriculture, pasture and to urban development. Fire, logging, the introduction of exotics, and intensive grazing are all present threats as well. Native mediterranean forests are very sensitive to habitat fragmentation, grazing, and alteration of fire regimes (through overburning or fire suppression). The loss of natural groundwater in many Mediterranean regions also has large-scale impacts on biodiversity through the alteration of riverine and floodplain systems. Protection of riparian areas, and blocks of native habitat large enough to sustain regular fire events by leaving sufficient unburned areas to retain genetic diversity are crucial (WWF, 2003i).

MANGROVES

Mangroves play an integral role in the coastal ecosystem, and are of invaluable local and global ecological, economic and social importance. Mangroves live in estuarine settings, acting as a buffer between marine and freshwater systems. In this capacity they are known to act as a filter of local water and can protect shorelines from eroding forces. Mangrove forests also protect seagrass beds and coral reefs from deposition of suspended matter that is transported seaward by rivers. This forest type inhabits waterlogged, salty soils along coasts in the tropics and subtropics where they experience tidal flow. Mangroves are dependent upon a relatively stable hydrographic and salinity regime, and they are susceptible to pollution and the alteration of salinity levels.

Mangroves currently suffer from large-scale conversion and degradation (Ellison and Farnsworth, 1997). In some countries such as India, the Philippines and Vietnam, over 50% of mangrove ecosystems have been lost in the last hundred years. Mangrove trees are harvested for timber, fuel, or pulp, and are also cleared for aquaculture or other development. Deterioration of a mangal occurs when the mangrove is clearcut, when the water flow pattern is disrupted, or when the water level becomes too high so that the aerial roots are unable to obtain oxygen. Pollution, particularly oil, can also interfere with the exchange of gas from mangal roots.

The Gulf of Guinea contains Africa’s most extensive mangroves, which help to stabilize a large part of the West African shoreline. Forming a dense barrier between sea and land, the mangrove is a crucial food reservoir for coastal people who rely on its supply of shrimp and crabs, as well as its wood for fuel. It is also a vital host to a number of endangered species, including the African manatee, pygmy hippopotamus, and clawless otter. The area is currently under high stress from urbanization, industrialization, and agriculture, and is experiencing impacts from timber and petroleum exploitation around the Gulf coast.

Assess Present and Future Stress and Vulnerability Due to Climate Change

A warmer climate and changes in precipitation patterns will cause disparate effects on forest ecosystems, making some contract while others will expand. Increases in CO_2 will compound this effect in some systems while dampening the impacts in other systems. Together this indicates that many areas, especially those habitats along environmental gradients, will be subjected to change, and if the population cannot adapt or move with changes in climate, they will face extinction.

Globally, it has been estimated that at least one-third of the world's remaining forests may be adversely impacted by climate change over the next century (IPCC, 1995). Climate change may force species to migrate or shift their ranges far faster than they are able to, thereby disrupting existing ecosystems (Kirilenko et al., 2000; Stewart et al., 1997). Forests may experience changes in fire intensity and frequency, increased susceptibility to insect damage or diseases, and extreme weather events which they may not be adapted to survive (IPCC, 2001).

Predictions for the impacts climate change will continue to have on forests are gathered in a number of ways. At a large scale, it is possible to predict major shifts in biome types by combining biogeography models such as the Holdridge Life Zone Classification Model with general circulation models (GCMs) that project changes under a doubled CO_2 scenario. Biogeochemistry models simulate the gain, loss and internal cycling of carbon, nutrients, and water-impact of changes in temperature, precipitation, soil moisture, and other climatic factors that give clues to ecosystem productivity. Dynamic global vegetation models integrate biogeochemical processes with dynamic changes in vegetation composition and distribution. Studies on particular species comparing present trends with paleological data also provide indications for how species will weather or adapt to future climate change. (Hansen et al., 2001)

Forests are both directly and indirectly impacted by climate change. The direct impacts of warming temperatures and changes in precipitation patterns or extreme weather events on forests are already evident in certain tree and animal species (IPCC, 2001). Even small changes in temperature and precipitation can have significant affects on forest growth and survival (e.g., for certain species of pine; Rehfeldt, 1999; and in tropical montane cloud forests; Hilbert et al., 2001; Pounds et al., 1999), particularly those in threshold areas or at the margins of an ecosystem. Higher temperatures increase water loss through evapotranspiration, which result in drier conditions, as well as decreasing a plant's efficiency of water use (NRC, 2002). Increases in temperature can also have dramatic implications for the timing of flowering and fruiting for plants (Beaubien and Freeland, 2000; Bazazz, 1998), and can also directly affect growth rates and other physiological factors that will cause species to migrate or become extinct. Forests will also be directly threatened by changes in the seasonality of precipitation and increases in extreme weather events such as hurricanes, flooding, lightning, and wind storms (Dale et al., 2001; Hansen et al., 2001; IPCC, 2001; NAST, 2001; Peterson, 2000). Forest characteristics and age-class structure play a large role in determining how a forest will respond to changes in moisture conditions (NRC, 2002). Mature forests have well-established root systems, and are therefore better able to withstand drought-like conditions, whereas younger forests and post-disturbance stands are more vulnerable to decreases in moisture (Gitay et al., 2001). Some species are also more tolerant to variable moisture conditions than others.

Disturbances are a natural part of the functioning of forest ecosystems, and are integral in bringing about succession. Most forests are in some state of reestablishment after disturbances, which themselves result in a change in ecosystem function as species composition and the structure of the forest changes. However, an extreme change in forest structure and function can take place when disturbances exceed their natural range of variation (Dale et al., 2000). Climate change affects forests both directly and indirectly through disturbances such as fire (Flannigan et al., 2000), drought (Howden et al., 2003), introduced species (Simberloff, 2000), insect and pathogen outbreaks (Ayres and Lombardero, 2000), hurricanes (Loope and Giambelluca, 1998), wind storms (Peterson, 2000) and ice storms (Irland, 2000) (Dale et al., 2000). Impacts can be seen across an array of spatial scales, from the leaf to the forest landscape, and can include a reduction in leaf function, deformed tree structure, tree death, altered regeneration patterns through the destruction of seed banks, a disruption in the physical environment from soil erosion and nutrient loss, and increased patchiness of forest communities (Dale et al., 2000). Because trees survive for long periods of time and take many years to become established, many climate change impacts on forests will be expressed through alterations in disturbance regimes (Franklin et al., 1991; Dale et al., 2000; Dale et al., 2001).

Other indirect effects of climate change on forests are often difficult to detect due to the complex and interdependent nature of ecosystem components. Yet, many indirect effects are just as serious if not more so than some direct effects, due to the cascading nature of the relationships (Hansen et al., 2001). For example, an alteration of the timing of flowering and fruiting caused by temperature changes can have a relatively minor impact upon the plant species, but the cascading effects on animal species dependent upon the fruit could be substantial (Bazazz, 1998). Likewise, changes in precipitation patterns caused by climate change will likely cause greater conversion of intact forest areas as some agricultural lands experience drought and erosion.

The impacts on forests from elevated levels of atmospheric CO_2 have been studied, though the results are “neither clear nor conclusive” (NRC, 2002; Gitay et al., 2001). Higher concentrations of CO_2 generally improve efficiency of water use as plants open their stomata less and thereby reduce water loss through transpiration, though disparate results for overall plant growth have been shown, depending on the species, individual tree age, and length of study period (NRC, 2002). Moreover, plants have been shown to adjust to higher CO_2 levels such that the higher absorption rates can decrease over time (Gitay et al., 2001). Difficulties of modeling the effects of elevated CO_2 concentrations are compounded when other anthropogenic emissions are considered. For example, ozone (O_3) offsets potential benefits of CO_2 on plant productivity (Karnosky et al., 1999; Isebrands et al., 2001); while nitrogen oxides may enhance forest growth in nitrogen-limited systems (Robinson et al., 2002).

Within the next 50-100 years, changes to ecosystem functions and plant demographic processes will be the imminent threats, though in the long term, large shifts in forest types are likely to occur (Hilbert et al., 2001). Looking broadly at forest types, boreal forests are expected to be impacted severely through a reduction in extent since warming will be greatest at the poles (IPCC, 2001). In the tropics, the impacts of sea-level rise are predicted to be significant for mangroves as they are inundated in many areas (IPCC, 2001). In tropical forests more generally, the effects of drought and changes in seasonality will compound existing threats of fragmentation and degradation. Across all forest types, some of the most vulnerable will be island or relict forest communities, including highly fragmented forests surrounded by agricultural or urban development and forest systems on remote islands whose migration opportunities are hindered either latitudinally or altitudinally, as in the case with tropical montane cloud forests (Dudley, 1998). Individual species especially vulnerable are those with limited geographical ranges with low rates of germination or survival of seedlings and those with limited seed dispersal or other migration capabilities (IPCC, 2001). Those species inhabiting the boundaries of heat or drought tolerant limits may be especially at risk. More detailed information on the extent of impacts for each major forest type is outlined below.

Dudley (1998) provides an overview of major categories of changes expected in forest ecosystems:

DISTURBANCE: climate change will increase the degree of disturbance, through extreme weather events such as storms and as a result of smaller but ultimately more pervasive changes to seasonality, rainfall and temperature. Climate change will thus add to those other forms of human disturbance, which are currently fragmenting and altering forest ecosystems.

SIMPLIFICATION: the net effects of problems with tree reproduction and species' migration rates in areas experiencing severe climate change will tend to cause problems for slower growing species and instead favour fast growing, short-lived weed and invasive species. The result will be an acceleration of a trend that is already occurring as a result of other forms of human interference, namely, the replacement of species-rich forests by species-poor forests.

MOVEMENT: is likely both geographically and altitudinally, as growing conditions alter. The ability of trees to migrate fast enough to keep pace with climate change is still largely unknown and will depend upon many other factors. The extent to which ecological conditions change will depend on a complex mixture of factors: for example, warmer conditions could encourage trees to move up-slope while accompanying droughts might have the reverse effect.

AGE REDUCTION: disturbance, increased forest fires, changes in pest patterns and the transition of whole communities will encourage an existing trend towards the replacement of old-growth forests with younger stands. This has particularly important implications for biodiversity, as many of today's threatened species are those confined to older habitats.

EXTINCTION OR EXTIRPATION: some of the most vulnerable forest habitats, including relict species at the edge of their ecological niche and some particularly threatened systems, such as mangroves on low-lying islands, could disappear altogether. Species could also disappear from some forests that appear to be surviving the changes relatively well.

CLIMATE CHANGE IMPACTS ON TROPICAL MOIST FORESTS

Although warming will be greatest at the poles, the impacts of climate change on tropical forests can still be substantial due to their relative sensitivity to climatic variables. Phenological processes are highly correlated with climatic signals. Any changes in climate can have significant impacts on a forest ecosystem. Changes in plant phenology are said to be the most immediate indicators of climate change (Corlett and LaFrankie, 1998). Related to this are the coevolutionary relationships that are highly specialized and could become disrupted if a species is affected by a changed phenological event or physiologically (Bazzaz, 1998).

Perhaps most importantly, high species diversity per unit area in tropical moist forests creates narrow niches that may be severely impacted by a reduction in biodiversity through elimination of species when migration to other suitable habitats is not an option (Bazzaz, 1998).

The sensitivity of hydrological regimes in tropical forests due to existing levels of deforestation could be exacerbated through a change in structure and function, particularly in low-lying areas where floods may increase in frequency and intensity (Bonell, 1998). The predicted decline in the rainfall in the Amazon Basin and the intensification of the Indian monsoon will have large-scale effects on availability of water for tropical forests (Bazzaz, 1998).

The frequency and intensity of fire are likely to increase due to climate change in tropical moist forests, where fire is naturally rare or nonexistent. The interrelationships of fire, climate and forest ecology are complex, and will depend upon a particular forest type. However, in some places, an increase in drought could cause desiccation of forest that could spur more fires. Alternatively, increased precipitation results in more biomass which provides a larger fuel load that in turn makes forests that are already vulnerable—due to framentation and other human-induced threats—more susceptible to large-scale fire events. Most tropical moist forests are not resistant to fires, which could cause a large-scale loss of biodiversity. These forests will experience a change in species composition increasingly favoring xeromorphic, pyrophytic and generally species-poor plant communities that create a positive feedback as the new ecosystem will be increasingly liable to burn. Extreme cases would result in desertification. (Dudley, 1998; Goldammer and Price, 1998)

At a course level, distributional shifts in forest types have been predicted for a number of tropical forest regions using Global Circulation Models that focus on temperature and precipitation changes. A 1 °C rise in temperature would increase the productivity of rainforests as a whole as long as all other factors are held constant. However, changes in rainfall patterns combined with warming can produce sizeable shifts in the distribution of forest types. Increased rainfall enhances growth of tropical moist forests (holding fire, pests, and the effects of other factors constant), while decreased precipitation could shift existing tropical moist forests to favor woodlands and tropical dry forests. (Hilbert, in press). Several of the member countries to the United Nations Framework Convention on Climate Change within the tropics have conducted an analysis of expected distributional changes in forest types. For example, the composition of forest types in Thailand is estimated to change dramatically, with subtropical forests declining and tropical forests in the southern region of the country increasing due to increases in precipitation combined with an increase in temperature (OEPP, 2000). In general, many shifts in distribution are expected to take place at the edges of forest types and in ecotonal areas between rainforest and more open forest areas (Hilbert, 2001).

TROPICAL MONTANE CLOUD FORESTS

Tropical Montane Cloud Forests (TCMF) are an important subset of moist tropical forests from a climate change perspective. Even small-scale shifts in temperature and precipitation are expected to have serious consequences for tropical forests in the high mountains; indeed, many of the impacts have already caused species extinctions. TCMFs are especially vulnerable because they reside in areas with steep gradients associated with their ecosystem boundaries where the climatic conditions are highly specific and therefore catered to endemic biota. Atmospheric warming is raising the altitude of cloud cover that provides TCMF species with moisture via predictable and prolonged immersion in clouds (Pounds et al., 1999). The habitat for TCMF species will shift up the slopes of mountains as they follow the retreating cloud base, forcing populations of reptiles and other species into an increasingly smaller area.

The extreme sensitivity of the microclimates of TCMFs to climate change make a good case for using these systems as a ‘listening post’ for detecting climate change (Loope et al., 1998). Several examples exist around the globe. In the highland rainforests of Monteverde, Costa Rica, the lifting of the cloud base associated with increased ocean temperatures has been linked to the disappearance of 20 species of frogs. (Pounds et al., 1999). The Wet Tropics World Heritage Area of northeastern Queensland in Australia provides another example of highland tropical habitat whose complex topography has resulted in highly specialized ‘cool islands’ housing extremely rare fauna that will be threatened by a warmer, drier local climate in the next few decades (Hilbert, 2001). A doubled CO_2 scenario will shift relative humidity up-slope by hundreds of meters during the winter dry season when forests are most reliant on moisture from clouds. With only one degree of warming, the highland rainforest habitat in this area could decrease the amount of habitat by as much as half, causing one-third of endemics to decrease. It is entirely probably that 30-50 species and most highland faunal species will disappear with average temperature increases of 1-5 °C (Hilbert et al., 2001). Yet another example exists in East Maui, Hawaii, where the steep microclimatic gradients in montane tropical forests, combined with increases in interannual variability in precipitation and hurricanes are expected to produce a situation where endemic biota will likely be displaced by non-native plants and animals (Loope et al., 1998).

CLIMATE CHANGE IMPACTS ON TROPICAL DRY FORESTS

The bulk of reported and predicted impacts of climate change on tropical dry forests concern the desiccation and related fire risk of this forest type. A simulation using the

Holdridge Classification Model integrated with GCMs under a doubling of CO_2 in the next century for Tanzania shows that a decline in precipitation and increase in ambient temperature will produce a shift from subtropical dry forest and subtropical moist forest to tropical very dry forest, tropical dry forest, and small areas of tropical moist forests. Predictions of increased disease and predation of seeds and seedlings are expected to cause a change in species composition and structure in the Kitulandgalo Forest Reserve as some species will fail to establish themselves. (Mwakifwamba and Mwakasonda, 2001) In Thailand, subtropical dry forest that currently covers little over one percent of the country's total forest area may disappear entirely as it is replaced by tropical dry forests and tropical very dry forest emerging in the north and northeast of the country (OEPP, 2000).

A change in the precipitation, either through an annual increase or decrease is expected to make tropical dry forests subject to greater risk from forest fires in the immediate term. For areas where precipitation is expected to decrease, thus prolonging the dry seasons, increased desiccation makes the forest system more likely to ignite. Reduction of Net Primary Productivity (NPP) combined with the growing impacts of grazing and farming, however, will lead to a landscape which is more sparse and unable to support the spread of fires. In areas that are predicted to experience enhanced precipitation, an increase in plant biomass will lead to a more continuous source of fuel to support more frequent and intense fires. However, a trend of increased fire occurrence tends to lead eventually to a general decrease of fires due to the reduction of fuelbeds over time. (Goldammer and Price, 1998)

CLIMATE CHANGE IMPACTS ON TEMPERATE FORESTS

The effects of climate change are already apparent in temperate forests such as in the United States and Canada, where a 1-2 °C increase in ambient air temperature and changes in precipitation have been documented for the past century (Watson et al., 1998). Forest decline and dieback along the Atlantic and Pacific coasts are thought to be related to increased levels of CO_2 (Mueller-Dombois, 1992). A further increase in ambient air temperature is expected to shift the range of suitable habitat for temperate forests northward – in the case of the United States, between 100 to 530 km during the next century (Iverson et al., 1999; Iverson and Prasad, 2001). Assuming a 2 °C warming over the same time period, tree species will be forced to migrate at a rate of approximately 1-3 miles per year, which is much too rapid for the bulk of temperate species, except for those whose migration is aided by birds carrying their seeds. In general, a shift in species composition to more heat-tolerant 'fast adapters', even grasslands, is likely (NAST, 2001). Indeed, species associated with human-dominated landscapes as well as exotics which are more tolerant to climatic extremes have already greatly expanded their ranges in the United States (Drake et al., 1989; Flather et al., 1999).

Flora and fauna inhabiting temperate forests will be affected physiologically, as well as through a loss of habitat and an increase in extreme weather events such as droughts, floods, windstorms, and wildfires that could increase tree mortality. As forests shift, they will become increasingly vulnerable to additional disturbances, such as diseases and other pests, as well as to fires as natural species regimes are altered (NAST, 2001). Warming can be particularly detrimental to many tree species when increased temperature speeds up the development of insects and pathogens, and thereby increases rates of infestations (Hansen et al., 2001). Increased desiccation in fragmented temperate forests, such as those in the Bassit upland and the Jebel Saheiliyeh in Syria may be seriously threatened through an increase in wildfires (Dudley, 1998).

Some models predict an overall increase in forest productivity with increased temperature, though other climatic-induced disturbances may counteract this. In addition, increases in productivity of one system may have negative effects in linked systems. For example, increases in forest productivity in the Great Plains could decrease the availability of water to aquatic ecosystems such as the Mississippi River intercoastal waterways. (NAST, 2001)

The National Assessment Synthesis Team (NAST) of the U.S. Global Change Research Program finds that a vast degree of biodiversity change is expected in the United States (NAST, 2001). The maple/beech/birch forests currently present in the Appalachian Range from New England and as far south as West Virginia and western Pennsylvania could disappear. The aspen and birch forests in the Upper Great Lakes region could be lost, while patches of red spruce forest communities may become extinct in its present range. Assuming a gradual warming, these forests will likely be replaced by oak/pine and oak/hickory forests, though a significant total loss of forest cover may occur if warming is more extreme and coupled with increased droughts, wildfires and/or insect infestations. An increase in the total area of forests burned could be on the order of 25-50% as increases in productivity create larger fuel loads that spark fires when coupled with more intense periodic droughts. (NAST, 2001; Howden et al., 2003). Some predict that warming in cooler areas such as in the northern United States and western mountains near the Canadian border will increase tree richness (Currie, 2001) and benefit reptiles and amphibians; though a quarter overall decrease in bird and mammal richness is also predicted, especially in the eastern U.S. (NAST, 2001; Hansen et al., 2001).

Similar trends are predicted for temperate forests in Scandinavia, where Norway spruce and beech are expected to move north and possibly replace other forest communities such as those with dwarf birch. Likewise, in China, cold temperate coniferous forests and mixed temperate coniferous and boreal forests will likely be reduced (Dudley, 1998). A comparative analysis of vegetation distribution in the Montseny mountains of Catalonia in northeastern Spain over the last half century shows a replacement of cold-temperate forests with Mediterranean forests that is correlated with warming combined with the cessation of traditional land management. Beech *(Fagus sylvatica)* and heather *(Calluna vulgaris)* are being replaced by holm oak *(Quercus ilex)* forest at mid-altitudes (800-1400 m), which has resulted in isolation and degradation of beech communities through its shift upwards in altitude. Within the isolated beech stands, the trees are more defoliated (by 30%), recruitment is 41% lower, and holm oak recruitment is three times higher than in more stable and continuous beech stands. (Peñuelas and Boada, 2003)

CLIMATE CHANGE IMPACTS ON BOREAL FORESTS

Warming is being felt more at the poles, with an expected rise of 4-5 °C during winter that may go as high as 10 °C over the next century according to some models (Sekula, 2000). The overall effect of this warming will affect the species composition and other ecological services of two-thirds of boreal forests (Kirschbaum and Fischlin, 1996), and cause a dramatic loss of between 25% and 40% of boreal forest area, as gains in the north are unable to keep up with replacement by temperate species in the south (Stone, 1996). Boreal forests will be replaced at lower latitudes by temperate forest species, grasslands, and in Russia, forest-steppe. Boreal forest habitat is predicted to migrate poleward by 300-500 km in the next century (IPCC, 1996b). In Russia, a reduction in boreal forests on the order of 19% and loss of productivity is predicted through General Circulation Models (Krankina et al. 1997). Plant hardiness zones have already shifted in accordance with warming over the last century in Canada, most notably in the western part of the country (McKenney et al., 2001).

Migration will be inhibited to a great extent, however, by inhospitable tundra soils and lack of biota that is necessary for colonization. The rate of colonization by tree species differs greatly, depending on the seed dispersal rate and range of tolerance, for example with white spruce colonizing at 100-200 km/100 years, and 4-8 km/century for Scots pine (IPCC, 1996b). An average rate of 25 km/century through seed dispersal and propagule establishment (Solomon, 1992) will mean that the rate of warming is likely to be approximately 10 times faster than that needed for successful species migration (Jardine, 1994). Moreover, tree species will be impeded from successful migration due to barriers such as habitat fragmentation and competition from more hardy exotic species (Iverson and Prasad, 2001; James, 2001; Collingham and Huntley, 2000).

Many tree species will also suffer physiological problems or changes in the timing and rate of seed production (Stewart et al., 1997) which will inhibit growth and reestablishment rates and in the long term, successful migration. Plants with narrow temperature tolerances, slow growth characteristics (Kirschbaum and Fischlin, 1997) and limited dispersal mechanisms (e.g. heavy seeds) will be the most vulnerable (Thompson et al., 1998). A new composition of species favoring early successional trees and shrubs over slow-growing woody species will be the net result (which will also entail a total loss of stored carbon). Further, a drastic change in species composition and loss of overall habitat with even 2 °C warming near the poles will cause a loss of ecosystem functionality as species richness is diminished. A decrease in habitat through this scenario would result in the loss of animals inhabiting the boreal Great Basin mountain ranges on the order of 10-50% (IPCC, 1996b).

An average rise in temperature of 1 °C over Canada in the last century has had an impact on plant physiology and phenology (NRC, 2002). At mid to high latitudes (45°N and 70°N), warming has corresponded with increased plant growth and the length of the growing season (Myneni et al., 1997). Warming that was accompanied by a decrease in precipitation has had a detrimental impact on plant growth in some tree species in western Canada such as aspen poplar, where reduced tree ring growth has been associated with drought events (Hogg et al., 2001). Reproductive timing in tree species has also been recorded, for example the trembling aspen in Alberta has begun blooming 26 days earlier over the last century (Beaubien and Freeland, 2000), and the bud break of white spruce in Ontario is also taking place earlier (Colombo, 1998).

Boreal forests are expected to become victims of increased insect infestations with the onset of warmer conditions. Many temperate pests such as the mountain pine beetle, normally limited by cold in boreal ranges, will expand their range, and many boreal tree species will be ill-equipped to deal with their infiltration, especially if ecosystem health is already compromised. At the same time, local boreal pests will be given longer periods to flourish under warmer winters. (Stewart et al., 1997; Sekula, 2000) Fleming et al. (2001) document historical trends that show that spruce budworm outbreaks increase the frequency of wildfires by increasing the amount of dead plant matter that serves as fuel for fires.

Warmer and drier conditions will also lead to changes in hydrological regimes, and have large-scale impacts through more frequent, severe, and widespread forest fires. Climate will be the underlying cause of ecosystem change via fire, which is the major disturbance regime in the boreal ecosystem. Studies have shown that fire frequency as well as the total area burned has increased in the last 20 to 40 years in accordance with warmer temperatures (Schindler, 1998; Kasischke et al., 1999; Stocks et al., 2000). A longer fire season, drier conditions and an increase in lightning storms are projected to increase the fire season severity in accordance with climate change (Stocks et al., 1998; Goldammer and Price, 1998). An initial increase in fires will aid the migration of fire-adapted species to germinate, as well as providing nutrients to the soil. However, over time, increased fragmentation due to fire will inhibit chances for migration. The result will be a shift of age class distributions toward younger forests (and a decrease in stored carbon).

CLIMATE CHANGE IMPACTS ON MEDITERRANEAN FORESTS

Hulme and Sheard (1999) provide an overview of the most notable effects of a warming and drying trend in the Mediterranean region, including a decrease in water availability and a corresponding increase in fires, both of which will have significant impacts upon regional biodiversity. Increased dessication brought on by climate change will further compound a trend towards reductions in groundwater supplies due to intensification of farming and urbanization. In the Iberian Peninsula (Spain and Portugal), the annual mean temperature has risen by nearly 1.6 °C over the last century, where the warmest years occurred in 1989, 1995 and 1997. Precipitation is expected to decrease by an average of 5-15% annually. Winters are expected to get wetter while the other seasons are expected to become drier, especially summer. Fires, while a natural component to the ecosystem, are becoming increasingly threatening, and an increase in occurrence and area burned is strongly correlated with increasing daily maximum temperature and decreasing humidity. In 1994, a fire burned almost half a million hectares of Spanish forest and shrubland and killed 31 people. A loss of Mediterranean woodland habitat will have a noted impact upon the endangered Iberian lynx due to a reduction in two of its vital food sources—rabbit and duck.

CLIMATE CHANGE IMPACTS ON MANGROVES

The principal climate change-induced threat to mangroves comes from sea-level rise. Floods in the East African coastal region point to high vulnerability in this region (IPCC, 2001). In Bangladesh and India, the biodiversity of the Sunderbans is at great risk due to rising seas; a one meter rise in sea level will likely cause the Sunderbans and the tiger with it to disappear (IPCC, 2001).

Rising sea-levels bring about changes in sediment dynamics, erosion and changes in salinity, all of which could compromise the ability of a mangrove system to survive. Many communities will be inundated while others will suffer from changes to hydrological and salinity regimes. Inundation has been shown to bring about a decrease in photosynthesis and increase salinity and salt stress during floods in the Everglades National Park in Florida (Dudley, 1998). Moreover, sea level rise is expected to take place at about twice the rate at which sediment build-up—necessary for the mangrove's survival—will occur and cause the sinking of many deltas. Lastly, "erosion will reduce the size of mangroves, by cliff erosion on the seaward edge that undercuts mangrove roots, through sheet erosion across the swamp surface and by cutting away of tidal creek banks" (Dudley, 1998).

Canvass and Assess Adaptation Options

After completing a vulnerability analysis to determine how a forest system may be impacted by changing climatic conditions, the next step is to look at the range of adaptation options available in order to promote resiliency. An effective vulnerability analysis will determine which components of the system—species or functions, for example—will be most vulnerable to change, together with consideration of which parts of the system are crucial for ecosystem health. An array of options pertinent to adapting forests to climate change are available, both to apply to forest communities at high risk from climate change impacts as well as for those whose protection should be prioritized given existing resiliency. Long-term adaptation of species will be enabled where natural adaptation processes such as migration, selection, and change in structure are allowed to take place due to sufficient connectivity and habitat size within the landscape.

Forest adaptation options are not dissimilar from traditional forest conservation methods; however great emphasis is placed upon increasing spatial and temporal scales, protection of key forest communities, managing specifically for increased disturbances, and flexibility given uncertainties and surprises surrounding what climate change will bring. The menu of options below all seek to maintain the health of forest biodiversity as the overarching vehicle for successful adaptation. Criteria for sustainable forest management outlined in the Montreal Process of the United Nations Conference on Environment and Development include maintenance of forest ecosystem health, conservation of biodiversity, maintenance of forest productivity, and conservation of soil and water resources (MPWG, 1998). Sustainable forest management criteria provide a framework into which adaptation strategies can be incorporated (NRC, 2002).

1. REDUCE PRESENT THREATS

The most apparent strategy for improving forest resilience to climate change is to promote overall ecosystem health. As discussed earlier, a variety of present non-climate threats to forest systems exist, namely conversion, fragmentation, and degradation. Identifying and targeting for action the reduction of priority threats in a particular forest system will go far in ensuring that forest structure, composition, and function that increase a forest's resilience are maintained.

2. AVOID FRAGMENTATION AND PROVIDE CONNECTIVITY

Noss (2000) provides an overview of the negative effects of ecosystem fragmentation which are abundantly documented worldwide. "Edge effects" threaten the microclimate and stability of a forest as the ratio of edge to interior habitat increases. Eventually, the ability of a forest to withstand debilitating impacts is broken. Fragmentation of forest ecosystems also contributes to a loss of biodiversity as exotic, weedy species with high dispersal capacities are favored and many native species are inhibited by isolation.

The importance of minimizing road networks deserves special consideration. Roads exacerbate the effect of a warmer climate to increase the incidence and rate of invasions of forests by pest and disease by encouraging the dispersal of invasive exotic species. They also restrict the dispersal of less mobile native species.

3. MAXIMIZE SIZE OF MANAGEMENT UNITS—DECISION-MAKING ON A LARGE, BIOGEOGRAPHIC SCALE

The fossil record provides evidence that species have adapted to changing climates by shifting their ranges. Protected areas established to conserve a particular species may not contain appropriate habitat in the next few decades. Overlaying a climate change scenario upon existing protected areas and other management units will give managers an indication of where a given habitat will occur to enable revision of management boundaries. However, given the rather crude estimation of climate change impacts within particular locales as well as the largely unknown change in ecosystem dynamics that will occur, it is prudent to give forest systems the maximum allowance of habitat in which to migrate (Noss, 2000). Attention should be given in the design of protected area networks to the need for north-south as well as altitudinal migration opportunities. The solution in this case does not rely solely on a reorientation of protected area boundaries, but also to a paradigm shift where decision-making regarding land uses takes place on a large, biogeographic scale to include potential habitat outside reserves (Hilbert, In Press).

4. PROVIDE BUFFER ZONES AND FLEXIBILITY OF LAND USES

The fixed boundaries of protected areas are not well suited to a dynamic environment unless individual areas are extremely large. With changing climate, buffer zones might provide suitable conditions for shifting of populations to lands bordering reserves as conditions inside reserves become unsuitable (Noss, 2000). Buffer zones increase the patch size of the interior of the protected area and overlapping buffers provide migratory possibilities for some species (Sekula, 2000). Buffer zones must be large, and managers of protected areas and surrounding lands must demonstrate considerable flexibility by adjusting land management activities across the landscape in response to changing habitat suitability. A specific case for a buffer zone surrounding tropical montane cloud forests can be made based on research that shows the upwind effects to deforestation of lowland forests causes a raising of the cloud base (Lawton et al., 2001).

5. REPRESENT FOREST TYPES ACROSS ENVIRONMENTAL GRADIENTS

Representing the full range of habitat types is a traditional conservation method, to set aside areas for scientific study, as a node of comparison against disturbed areas, and as a means of conserving species that may be too difficult to manage separately. The uncertainty about the precise type and distribution of impacts necessitates maintaining a full spectrum of forest types within protected areas to enable some resistant and resilient types to persist. (Noss 2000)

6. PROTECT MATURE FOREST STANDS

Primary forests have been shown to be particularly resilient to climate change (Franklin et al., 1991). Mature trees are better able to weather large-scale disturbances than recently established forests (Brubaker, 1986), thereby providing a refuge for species reproduction once favorable climatic conditions return (Noss, 2000). While shifts in composition along environmental gradients are still expected in established forests (Franklin et al., 1991), the effects are expected to be much slower, thereby giving species more time to adapt.

7. PROTECT FUNCTIONAL GROUPS AND KEYSTONE SPECIES

Maintaining the natural diversity of species and functional groups in forests is a sound overall strategy for enhancing both resistance and resilience to climate change. Several recent studies have demonstrated increased tolerance to environmental extremes and recovery potential as species richness increases. Species diversity in turn promotes the “redundancy” or number of species present in critical functional groups. Functional groups include various kinds of producers, pollinators, seed dispersers, predators, parasites, decomposers, and so on. Thus, it is not just species diversity that matters, but also species composition. Both may enhance the stability of a forest ecosystem. Efforts to identify keystone species and functional groups will help forest managers maintain natural patterns of abundance and distribution. (Noss, 2000)

8. PROTECT CLIMATIC REFUGIA

“Across continents, at both temperate and tropical latitudes, topographically diverse areas allowed habitats and lineages to persist through altitudinal shifts and, in many cases, to diverge during periods of climate change" (Noss, 2000). Climatic refugia are important for maintaining assemblages of species typical of past climates. Identifying and protecting areas will decrease the non-climate stresses species in these places may be under to enhance their chances for survival so that they may function as refugia during future climate change. In North America, such areas include the southern Appalachians, the valleys of major rivers in the southeastern coastal plain of North America, and the Klamath-Siskiyou region of northwestern California and southwestern Oregon. Major refugia in Europe include Iberia, Italy, the Balkans, and the Caucasus. In Central America, riparian habitats have been important refugia, especially along the Caribbean coast. (Noss, 2000)

9. MAINTAIN NATURAL FIRE REGIMES

The frequency and intensity of fires are known to correlate with changing climatic conditions. However, the relationships of fire to a particular forest ecosystem, and the different management decisions that are required vary greatly for different systems. Fire suppression has actually been shown to bring about a decrease in biodiversity in some areas where fire is a natural component of the ecosystem (Noss et al., 1995; Noss and Peters, 1995), while in other areas, particularly in the tropics, human-set fires have had catastrophic consequences for biodiversity (Trapnell, 1959; Van Schaik and Kramer, 1997; Dudley, 1998). Noss (2000) provides an overview of the complexity of fire management for different forest ecosystems, and provides some general guidelines. "Regional differences in fire ecology imply that fire policies established in response to concerns about climate change should not be uniform; rather, they should be established based on what is known of the fire ecology of each region and forest type…A mixed strategy in which managers let many natural fires burn, protect old growth from stand-replacing fires, and manage other stands through prescribed burning and understory thinning, is probably the optimal approach" (Noss, 2000). Some high value smaller areas may require protection from fire (Stocks et al., 1998). For example, Hirsch et al. (2001) promote the integration of 'fire-smart landscapes' that reduce the intensity and spread of wildfire and its impacts through harvesting, regeneration and stand tending.

10. ACTIVELY MANAGE PESTS

Climate change has been associated with increased infestations of insects, disease, and exotic species (Williams and Liebhold, 1995). This has been abundantly documented in cooler climates, where increases in temperature and decreases in precipitation have led to increases in attacks by spruce budworm in boreal forests, for example (Fleming et al., 2001). In ecosystems where pests are predicted to have a significant impact on the system, an active management program to reduce the negative effects of the pest could be devised (Howden et al., 2003). Prescribed burning is an option for reducing vulnerability to pest outbreaks in some temperate and boreal regions (Wheaton, 2001). Nonchemical pesticides have been proposed as means to reduce leaf mortality from insects (Johnson, 2001). Other nonchemical options such as baculoviruses are being investigated for their potential use in attacking pest species such as spruce budworm while leaving other species and the environment relatively unharmed (NRC, 2001).

11. SILVICULTURAL TECHNIQUES TO PROMOTE FOREST PRODUCTIVITY

Because climate change will likely have differential impacts upon different species and age classes of trees, a straightforward, ‘no regrets’ strategy is to apply silvicultural techniques that maintain a diversity of age stands and mix of species (Krankina et al., 1997). These measures will contribute to maintaining the productivity of the forest system as climate changes. Other silvicultural activities, such as collecting salvage cuttings from ice storms, may reduce the amount of long-term damage arising from future storms (Irland, 2000).

12. PREVENT CONVERSION TO PLANTATIONS AND PRACTICE LOW-INTENSITY FORESTRY

Forestry operations that minimize soil disturbance and utilize less clearcutting and chemical pollutants help reduce the invasion of exotic species, loss of carbon from soil, and the potential loss of mycorrhizae. The size of canopy openings and removal of biomass from sites should both be reduced. These methods are more likely to promote the resistance and resilience of forests to climate change than intensive forestry operations. (Noss, 2000)

13. MAINTAIN GENETIC DIVERSITY AND PROMOTE ECOSYSTEM HEALTH VIA RESTORATION

Adaptation to climate change via selection of resilient species depends upon genetic variation. Efforts to maintain genetic diversity should be applied, particularly in degraded landscapes or within populations of commercially important trees (where genetic diversity is often low due to selective harvesting). In such places where genetic diversity has been reduced, restoration, especially using seed sources from lower elevations or latitudes, can play a vital role in maintaining ecosystem resilience (Noss, 2000). Hogg and Schwarz (1997) suggest that assisted regeneration could be used in southern boreal forests in Canada where drier conditions may decrease natural regeneration of conifer species. Similarly, genotypes of beach pine forests in British Columbia may need assistance in redistributing across the landscape in order to maintain long-term productivity (Rehfeldt et al., 1999). In addition, species can be specifically selected for replanting that are known to be more resilient to impacts in a given landscape. For example, trees with thick bark can be planted in areas prone to fire to increase tree survival during increased frequency and severity of fires (Dale et al., 2000).

14. ASSIST MIGRATION WITH SPECIES INTRODUCTIONS TO NEW AREAS

Management programs specially designed for tree species that will be especially impacted by climate change may be necessary in some areas. The primary method of ensuring the species’ survival will be to consider introducing the species that are ‘climatologically trapped’ to more suitable or safe habitats (Sekula, 2000). This may take placed outside the species’ present range, though new areas may become more appropriate for the species under new climatic conditions. Dore et al. (2000) suggest that careful introduction of selected tree species in the Boreal Transition Ecozone may prove more ecologically and economically viable than current forms of agriculture. In all cases, special attention should be given to the species' history and potential effects of its introduction (PSRS, 2003), and the reintroduction should be well planned and coordinated with other management programs (Sekula, 2000).

15. PROTECT MOST HIGHLY THREATENED SPECIES *EX SITU*
    For some forest ecosystems, such as the cloud forests of tropical mountains, climate change is already (or will soon become) the dominant threat to an extent that mitigation efforts will not prevent some loss of species. In these situations, *ex situ* preservation of species in zoos and botanical gardens may be the only way to avoid extinction. Collections should include sufficient genetic diversity to allow adaptation to uncertain conditions in reintroduction sites. (Noss, 2000)

For the Great Plains island forests in North America, Henderson et al. (in press) recommend a "discovery, provenance and breeding program, encompassing both extant tree species within island forest and possible new species introductions, with the objective of establishing which varieties and species are best adapted to the range of probable future climates in island forests" (Henderson et al., in press). Specific measures could include: "collection of seed from dry microsites within and outside island forests, determination of related tree species to those now extant which might add resiliency, use of plantation trial sites within or adjacent to island forests or outside where such sites might serve as analogues for future moisture conditions at island forest sites" (Henderson et al., in press).

Identify and Select Adaptation Strategies

The selection of adaptation strategies will depend largely upon the existing and eminent non-climate stresses to the forest system, that is, the baseline upon which climate change will exert additional pressure. The overlay of climate-induced threats pointing to particular species or system vulnerabilities will aid in the choice of strategies. For example, in areas where forests are degraded due to species extraction, such as within many mangroves, reforestation of mangal species in accordance with knowledge of future changes to the hydrological system are likely to be the most obvious preventative measure to increase the chances for survival during the onset of sea-level rise.

Once managers are armed with knowledge of non-climate and climate threats to a system, it will be important to develop a vision for what the management plan is trying to achieve. As Millar notes (PSRS, 2003), the goal should not be to stop change or preserve a species, population or landscape in its current or former condition. Especially for highly vulnerable systems, "change may be inevitable, and resisting it could lead to abrupt and undesired consequences in the future" (PSRS, 2003). Forest composition and distribution that occurred before large-scale settlement (the 1800s) developed in response to the harshest period of the Little Ice Age which ended in the late 19th century, and are probably not good models for climate change in the present and future (PSRS, 2003).

As noted above, making forests more resilient to climate change necessitates a multipronged approach targeting both forests at high risk as well as prioritizing protection for those that contribute significant genetic potential through their existing resiliency. Determining which category a particular forest community fits within will help managers choose from the suite of adaptation options.

The first basic strategy for all forests should be preventative measures, such as protection through the creation of national parks, reserves, or buffer zones. There are 44,000 protected areas worldwide, covering an area equivalent to China and India combined. However, this constitutes only about 12% of the world's forests having legal protection of one sort or another since the dawn of the modern conservation movement in the late 1800s. WWF's Forests for Life campaign estimated that in 1996 only about 6% of the world's forests were formally set aside for strict protection purposes. Moreover, many of the existing protected areas are protected in name only, and are seriously degraded or under pressure from illegal logging, poaching, mining and other threats.

Where some category of protection—strict or otherwise—does not appear likely, avoiding fragmentation through the minimization of road networks, and practicing low-intensity forestry are sound options. Preventing damage that will compromise a system's resilience to climate change is usually much less costly than actions to restore it.

For forests at high risk to climate change, a range of strategies to deal with known disturbances are possible. As a prevention measure, one approach is to manage for the disturbance, for example by decreasing the density of tree planting to reduce susceptibility to drought, or removing trees vulnerable to ice or wind storms. Other disturbances, including fires, and pests can also be managed through preventive measures and thereby manipulate the intensity or frequency of the disturbance. Another approach is to mitigate the forest disturbance itself. This can be done, for example, by limiting the introduction of non-native species or by using prescribed burns. Forests can also be manipulated with the aim of reducing vulnerability after the disturbance has occurred. Recovery efforts can be employed after the disturbance or managed in an ongoing process, for example for adding structural elements that create shade for the reestablishment of vegetation, or through the planting of late successional species to speed up succession. Short-term mitigation efforts will be necessary in some instances in order to support certain gene pools until a stable habitat is identified. (NAST, 2001)

When choosing among adaptation strategies, managers will be increasingly challenged to view the role that climate change plays in perpetuating human-related changing conditions within forests. This realization will necessitate a fresh examination of threats such as overgrazing, invasive species, or fires in light of the role that climate change plays. (PSRS, 2003).

Lastly, the importance of decision-making for ecological change on a large biogeographical scale that incorporates socio-economic and development priorities can not be stressed enough. Adaptation strategies that may be deemed necessary within a small forest community could change once placed upon a larger ecological landscape. Consideration of competing priorities—both among conservationists and through the larger human and development landscape—will create a more realistic management plan with better chances for success.

Implement or Recommend Actions and/or Policies

Once a vulnerability analysis has been conducted, and adaptation strategies chosen, it is necessary to decide how the actions will be implemented. Forest managers for a given system will likely be aware of existing management plans, and it is important to take stock of how the adaptation strategy would fit with these plans. If no clear management plans exist, it is conceivable to produce one focusing specifically on increasing resilience to climate change. Ideally, adaptation management strategies in response to climate change will serve as another layer in a comprehensive forest management plan that has as its objective the overall health of the forest ecosystem. For example, many WWF ecoregional visions are adding vulnerability to climate change as another component that will drive conservation decisions. Such anticipatory adaptation plans take climate change into account during the planning process (NRC, 2002), and will better ensure synergies with other management priorities. It is also usually much less costly to adjust management practices in advance of future changes rather than to wait for evidence of large-scale damage.

A number of scientific, governmental and NGO institutions are acquiring expertise in the area of climate change impacts and adaptation. It will be fruitful to seek partnerships with these institutions at the beginning of any project to analyze climate impacts and options to increase resilience. The United Nations Environment Program is implementing a new program which is being jointly executed by the System for Analysis Research and Training (START) and the Third World Academy of Sciences (TWAS), with funding from the U.N. Global Environment Facility called “Assessments of Impacts and Adaptations to Climate Change (AIACC)”. The aim of this project is to “enhance the scientific capacity of developing countries to assess climate change vulnerabilities and adaptation, and generate and communicate information useful for adaptation planning and action” (AIACC website, 2003). For boreal systems, Environment Canada, the federal environmental agency, founded in 1999 the Science, Impacts and Adaptation Project with $15 million in funding to better understand risks from climate change and formulate responses (Environment Canada, 2003). The Potsdam Institute for Climate Impact Research in Germany includes in its mission to contain human-induced climate change to a tolerable level, and provide suitable measures to adapt to the unavoidable warming of the planet, particularly in the poorest developing countries (Potsdam Institute website, 2003). Likewise, a number of research institutes within universities have sprung up in the last several years to study forest vulnerability to climate change and adaptation.

Partner institutions and other stakeholders—especially those that will be impacted by any management decisions—can assist in the implementation of adaptation plans. A critical component of any adaptation project will be to devise a rigorous and goal-oriented monitoring system for the given management area and surrounding landscape to the extent possible in order to detect whether management techniques are having the desired effect. Monitoring is also important given the relative uncertainty with which climate change will change forest systems, especially via the complex relationships between species structure, composition, and functioning. Many monitoring programs exist for fire, insects or pathogens, but few exist to monitor reserve areas or for other disturbance events such as landslides or ice storms (NAST, 2001). The results of monitoring will also enable lessons to be drawn from adaptation management efforts, and to compare these with similar ‘control’ landscapes or other adaptation projects in different regions with similar habitat type. These lessons could be disseminated through reports, websites, and other educational materials.

Examples of Existing Adaptation Strategies

The United Nations Environment Programme’s World Conservation Monitoring Centre (WCMC) has begun a project called “Forests in Flux” with the goal to define priority areas for networks of forest protected areas across the globe. A crucial component of the project is “review[ing] and assess[ing] predicted responses of forest ecosystems to climate change, focusing specifically on key issues for conservation, particularly ecosystem tolerance and loss of biodiversity” (WCMC, 2003). An ‘ecosystem response (forests and climate change) database” has been established through the project in order to assess current knowledge of predicted responses of forest ecosystems through summaries of model specifications, parameters, nature of predictions, references and a listing of collaborators. Ecosystems identified as highly vulnerable have been flagged for more detailed analysis.

The AIACC initiative mentioned above includes a number of on-going adaptation projects relevant to the forest sector in developing countries. For example:

A project in Southern Africa is “developing more realistic ways of predicting the response of plant, animal, bird and reptile species to a changing climate in the presence of a changing and fragmented landscape. The project will analyze a range of adaptation options to determine which are the most effective, cost-efficient, and robust” (AIACC South Africa project website, 2003).

In Malawi, Zambia, Zimbabwe, and Mozambique, a project is being carried out to “assess vulnerability and explore adaptation options to climate variability and extreme events in the Miombo region…that lie within the drainage basin of the Zambezi River” (AIACC Miombo project website, 2003). The project is developing datasets and scenarios of land use change, a regional integrated model, and tools for analyzing regional impacts and adaptation, including a case study documenting climate—land use—people—ecosystem linkages in order to guide adaptation plans.

A project in Southeast Asia is analyzing “the impacts of climate change and associated land use and cover change on water resources, forest ecosystems, and social systems of watersheds…” (AIACC Southeast Asia project website, 2003).

In China, climate change impacts and possible adaptation strategies for key sectors are being determined through an integrated approach, including workshops, surveys, multi-stakeholder consultations, ecological modeling, GIS and remote sensing, and multi-criteria decision-making. (AIACC China project website, 2003).

The adaptation strategies suggested above will be key measures in the short and medium term to maximizing the resilience of forest systems impacted by climate change. Even if emissions were drastically reduced today, human-induced changes in temperature and precipiation will cause wide-ranging disturbances upon ecosystems within the next century, due to the long lifespan of emissions in the atmosphere. Therefore, the only long-term option to ensuring healthy forest ecosystems is to reduce emissions of greenhouse gases now so that we will not leave a legacy of forests fraught with problems far worse than those reviewed here.

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CHAPTER 3
Montane

Designing Strategies to Increase the Resilience of Alpine/Montane Systems to Climate Change

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MOUNTAIN REGIONS HAVE HIGH levels of biodiversity, due particularly to the compression of life zones into a small horizontal distance (Jeník, 1997; Körner, 2002). Many factors combine to create these high levels of biodiversity at the regional scale. One is the combination of steep altitudinal gradient, topographic variation and range of aspects, providing a rich variety of habitats at all scales. A second set of factors extends over geological time. As mountain ranges have developed, species have been able to migrate along new pathways, exploiting ecological niches as they have opened up. Yet interruptions in mountain-building phases, subsequent erosion, and changes in climate—especially ice ages—have also isolated species, so that they have evolved in different ways, a major reason for high levels of endemism in mountain areas. A third set of factors relates to human activities. Some mountain ecosystems have remained relatively unmodified by humans, while others have been heavily cultivated or otherwise altered for centuries or millennia. Regardless of degree of modification, montane regions often have higher levels of biodiversity than adjacent lowlands.

Although alpine terrain makes up only 3% of Earth’s continents, it houses 10,000 species of higher plants, or 4% of the global total (Gough et al., 1994). Alpine plants are small in size, allowing a large number of individuals to live in the available microhabitats, leading to high levels of alpha diversity, which may be further increased by the isolation of individuals at the patch scale and frequent disturbance. Levels of genetic diversity are also high, due to polyploidy and a high degree of self-incompatibility (Körner, 2002). Despite this wealth of biodiversity, mountain ecosystems often have low resilience to anthropogenic impacts, taking longer to recover than many lowland ecosystems.

Mountains also play key roles in global climate processes, especially through interactions with convection and air mass systems, causing precipitation through the orographic effect, and by condensing and storing freshwater in the form of snow and ice (Bandyopadhyay et al., 1997). This function is important to billions of people who rely on mountains as ‘water towers’ for drinking and other domestic use, fisheries and agricultural irrigation, hydroelectric power, industrial use and transport or recreation (Mountain Agenda, 1998).

In addition to the variety of ecosystem services listed above, montane and alpine ecosystems also have a high intrinsic value in many of the world's mountain protected areas (Notaro and Signorello, 2001) and provide the 'capital' of tourism (Krippendorf, 1984). Consequently, changes in landscapes can have major effects on tourism markets, not only causing severe economic and social impacts for mountain people, but also affecting the budgets of mountain protected areas.

Finally, the fact that many alpine and montane ecosystems are situated on the boundaries between countries—and also the administrative areas of individual states—can lead to particular challenges in defining management strategies at the level of the ecosystem or mountain range. Thus, the alpine/montane biome is poorly placed to respond to conservation efforts in general, and climate change in particular.

Stresses and Vulnerabilities Due to Factors Other Than Climate Change

POLLUTION

Pollution from airborne contaminants affects all mountain areas, particularly those downwind of urban/industrial areas. Aerosols, which enter the snowpack and hydrological system via precipitation often far from their source, can have a detrimental impact on the freshwater environment, reducing the quality of water in alpine lakes and therefore having negative impacts on biodiversity (Koinig et al., 1998). Over the past century, there has been a global increase in nitrogen deposition (Vitousek et al., 1997), as well as increases in the deposition of nitrates and sulfates at high-altitude sites (Döscher et al., 1995). Acidification of rainfall can kill or seriously damage trees and smaller flora and reduce feeding quality for wildlife (Hamilton et al., 1997). High levels of tropospheric ozone can reduce the growth of plants and mosses and may affect the relative competitiveness of species (NEGTAP, 2001). Conversely, stratospheric ozone depletion due to anthropogenic emissions allows higher and more biologically damaging levels of ultraviolet radiation to reach Earth's surface, with negative effects on both flora and fauna (Bjorn et al., 1998; Blaustein et al., 1995; Kiffney et al., 1997).

AGRICULTURE

Agricultural activities often extend up mountainsides and have important effects on mountain environments, including poisoning of groundwater by agricultural chemicals or slurry and numerous physical impacts such as the construction of vehicular tracks or the installation of more agri-industrial forms of processing. High-altitude soils are thin and are therefore much more sensitive to disturbance, and take much longer to recover. Grazing and burning in many of the world's upland areas have reduced the biodiversity of mountain pastures (Körner, 2002), and in these sensitive areas much of the natural land cover has been significantly altered. The changes associated with agricultural abandonment often have a role in increasing the vulnerability of alpine/montane areas, particularly where the balance of the water supply is altered, or the stability of steep or terraced terrain is not maintained (MacDonald et al., 2000). Conversely, some alpine plants have extended locally into grasslands opened up by burning and grazing by introduced herbivores in New Zealand (Halloy and Mark, 2003) and other parts of the world. Thus, through a series of complex processes, agriculture has both reduced the extent of natural alpine and montane communities and allowed their extension into previously unavailable terrain.

FORESTRY

Forest plantations, often composed of alien species, are common in alpine and montane regions. Many aspects of forestry alter runoff and drainage patterns; forest road construction, planting, thinning, and harvesting can lead to the contamination of watercourses with sediments, machine oils, and other contaminants. (Hamilton and Bruijnzeel, 1997). Forestry also affects natural successional patterns, decreasing habitat area for species that depend on more established, old-growth forests. As with agriculture, forestry activities have also increased the extent of many alpine and montane habitats, allowing a downslope expansion into areas from which forests have been removed by cutting, burning, and/or grazing (Wielgolaski, 1997).

FRESHWATER ABSTRACTION

People whose supply of freshwater depends on mountain areas are often far removed from those areas, and thus relatively unaware of negative impacts of their water use on mountain ecosystems. Vulnerabilities associated with freshwater abstraction from mountain systems relate to the alteration of river channels; the construction of dams; and over-abstraction for water-thirsty cities and agriculture, often far downstream (Bandyopadhyay et al., 1997). All of these factors can influence aquatic and riparian habitat and the species dependent on them; as well as groundwater resources, sometimes over large areas.

TOURISM AND RECREATION

Tourism-related development affects diverse mountain environments, although it must be recognised that this phenomenon is distributed very unevenly at vary spatial scales across the world’s mountain ranges (Price et al., 1997). Erosion, fires, trampling or other loss of flora by people, horses, and off-road vehicles, hunting and fishing, disturbance of fauna (the presence of people in and flying above habitats), and pollution (litter and human waste) are all serious problems related to human recreational use of mountain ecosystems (Buckley, 2000). Soil and biological systems in alpine/montane environments take far longer to recover from such impacts than in the lowlands (Gordon et al., 1998), and are therefore more vulnerable to such disturbances. Some high-impact mountain tourist industries, such as skiing and snowboarding resorts, may experience strong declines in days of operation due to climate change.

INFRASTRUCTURE DEVELOPMENT

Communications facilities (towers and masts for radio, TV, mobile phones, military communications, etc.) have been installed on many mountains. In most cases, this involves the construction of a road, which removes and damages habitat and may lead to long-term soil erosion. The road is then often used for access not only for the maintenance of the communications facilities, but also for hunting and other recreational activities. All of these activities may endanger populations of animals and plants, especially endemic species restricted to summits.

ALIEN SPECIES

Particularly because of their isolation, mountain species are often not able to compete with alien species, especially when their introduction is linked to habitat destruction. This is a problem in many mountain areas, but particularly those on isolated islands such as those of Hawaii (Loope and Giambelluca, 1998) and New Zealand (Halloy et al., 2001).

Present and Future Stress and Vulnerability Due to Climate Change

Increased concentrations of atmospheric CO_2, the primary cause of climate change, are likely to affect the physiology of alpine and montane plants. The exact outcomes are unclear, and will vary from species to species and be affected by factors such as inter-specific competition, grazing, physical site conditions, and levels of nitrogen deposition (Körner, 1999). As mountain climates are generally characterised by great variability and cycles at both diurnal and seasonal scales, some mountain biota are already stressed, which will influence their response not just to CO_2, but to all aspects of climate change (Price and Haslett, 1995).

ECOLOGY

A likely impact of climate change on mountain environments is upslope altitudinal migration of vegetation climatic belts, generally—but not always—leading to a decrease in their area, and the loss of the coldest climatic zones at the summits (Halpin, 1994). Thus all affected zones will experience increased vulnerability (Parry, 2000). Migration of vegetation zones around mountains to a different aspect is also possible, with implications for certain floral assemblages such as lichens, which depend greatly on living on a certain aspect for maximum insolation. Migration is typically severely restricted as a spatial response in mountain areas, however, because of their topography (Huntley and Baxter, 2002) and, often, the availability of suitable soils (Theurillat et al., 1998). Thus upslope migration will probably result in the contraction and fragmentation of populations of plants and associated fauna in the present montane, alpine, and nival belts. A key factor in the survival of plant populations will therefore be their dispersal capabilities, which tend to be low in alpine species; for instance, only 35% of alpine specialist plants in New Zealand were capable of long distance dispersal, as compared to 65% of non-specialists (McGlone et al., 2001). This is one of a range of key characteristics that make many alpine and montane species particularly sensitive to changes in climate—other characteristics are that they are: at the edge of their range; geographically localised; genetically impoverished; slow reproducers; or highly specialised (McNeely, 1990).

Changes in season length, and particularly earlier snowmelt dates and higher soil temperatures, could affect flowering and other aspects of plant phenology (Dunne et al., 2003; Thórhallsdóttir, 1998), as well as the annual cycles of pollinators, disease-causing insects, and other organisms. Equally, alien and invasive plant species may benefit more from climate change than native species, to the detriment of the latter (Dukes and Mooney, 1999). All of these trends will cause changes in inter-specific competition: some species will do better, but others will die. Thus, plant community composition will change, and some species that are now relatively uncommon, existing in isolated patches, will increase in abundance. Thus, while some species will benefit from climate change, others—particularly rare and endemic species—are likely to gradually become extinct, beginning with those found only at the highest altitude (Halloy and Mark, 2003). The impact on montane and alpine floral diversity could therefore be significant, especially in isolated mountain ranges (Körner and Spehn, 2002).

As current habitats change and are lost, fauna will also have to adapt or be forced to migrate upwards and polewards, following the altitudinal and latitudinal shifts of climate and vegetation belts, in order to survive. For insectivorous and carnivorous animals, the combination of climate and land-use change may lead to the loss of key elements of their diet, for both year-round residents and migrants that arrive earlier because of climate change, as shown for robins migrating to the Rocky Mountains of Colorado (Inouye et al., 2000). Also, as mentioned earlier, migration is a restricted strategy for isolated mountain areas or those at the continental margins (e.g., the Scandinavian mountains, the New Zealand mountain ranges, or the northern Rocky Mountains in Alaska). Thus, predicting rates of change in populations of both flora and fauna is highly complex, depending not only on changes in climate but on changes in biotic interactions as has been shown, for instance, for amphibians at Monteverde, Costa Rica (Pounds et al., 1999).

CRYOSPHERE

The shrinkage of glaciers is a global phenomenon, and rates of retreat are generally accelerating (Haeberli et al., 1999). Tropical glaciers are particularly sensitive because of the lack of seasonality and the fact that ablation is year-round (Arnell and Liu, 2001). In a number of mountain areas, the rapid melting of glaciers is leading to an increased risk of glacier lake outburst floods (GLOFs), which cause total destruction of habitats along affected watercourses, as well as loss of life and severe damage to infrastructure (Richardson and Reynolds, 2000). The melting of permafrost will lead to slope instability, loss of foundation stability for structures, and increased damage from freeze-thaw cycles (Haeberli, 1995).

For every °C increase in temperature, the snowline rises in altitude about 150 m (Parry, 2000). As a result, less snow will accumulate at lower elevations than at present. However, there could be more snow accumulation above the freezing level as both precipitation and temperatures increase. This may cause more frequent avalanches (Extreme events, pg. 94) and hinder the movement of large ungulates and skiers, but at the same time provide added protection to small plants and mammals (Scott and Suffling, 2000). In general, the trend will be for the snowpack to become unstable, snowpack duration to be reduced, and the profile of permanent snowpatches to change. These phenomena already appear to be resulting in changes in the emergence dates of marmots in the Rocky Mountains of Colorado (Inouye et al., 2000) and decreases in the area and occurrence of plant communities and endemic mammals in the Australian Alps (Green and Pickering, 2002).

Secondary effects of cryospheric dynamics are mainly with regard to hydrology. In alpine lakes in Norway, increased winter snowfall negatively correlates with the growth of brown trout *(Salmo trutta)* (Borgstrom, 2001). More generally, increased melt from receding glaciers and icefields will also increase water flows initially, although this effect will subside as the glaciers disappear. This will occur relatively quickly in the temperate mountain regions, and even faster in the tropics. Changing storage and release rates for precipitation will alter the timing of peak flows downstream; populations of aquatic species relying on late summer and early autumn melt may go extinct (Murray, pers. comm.). At the same time, a longer snow-free season, leading to a longer dry period, may lead to increased fire frequency.

HYDROLOGY

Hydrological stresses derive from the dependence of terrestrial, riparian, and aquatic ecosystems on freshwater from mountain sources. Changing precipitation patterns have been recorded world-wide (Folland and Karl, 2001) and inevitably affect runoff patterns, as will the changes in cryospheric dynamics noted above. As a greater proportion of total precipitation falls as rain rather than snow, the lag time due to snow precipitation and storage until meltwater enters the river system is reduced; in addition, melting glaciers will add a further volume of water into the systems until they too are gone. Then overall flow will decrease dramatically and become more variable (Arnell and Liu, 2001).

Increases in flows tend to increase sediment transport, turbidity, and bank erosion. Conversely, decreases in flows can increase pollution loads. Droughts, or decreased and earlier runoff from snow and ice melt may have important impacts on fish populations, especially those that rely on adequate late summer and early autumn flows for spawning (Levy, 1994). Changes in the seasonal distribution of precipitation may have other effects; for instance, an increase in summer precipitation could affect seed formation in species which have evolved to set seed during a dry summer period (Murray, pers. comm.). Given the importance of mountain water to downstream populations and economies, decreased and less reliable runoff may lead to increased demands for water storage in alpine and montane areas.

EXTREME EVENTS

In addition to more variable precipitation, it is predicted that the incidence of severe storms, floods, droughts and—in tropical and subtropical areas—tropical storms and hurricanes will increase (Arnell and Liu, 2001). Increases in flooding frequency are perhaps among the most feared physical impacts of climate change (Arnell, 1996), with critical impacts for ecosystems, human populations, and infrastructure in mountain areas and downstream. However, there have been few studies of this problem, mainly due to the difficulties involved with identifying credible scenarios for changes in flood-producing climatic events. Furthermore, many floods have multiple causes; they may be purely a product of rainfall or snowmelt or a combination of the two (Arnell and Liu, 2001). Impacts of floods can include greatly increased sediment loads and bedload transport and river channel scour, with losses in riparian ecosystems and negative impacts on aquatic flora and fauna (Wohl, 2000). Conversely, low summer flows and droughts also have negative impacts on riparian and aquatic ecosystems, and cause reductions in water storage, which could lead to demands for new storage infrastructure.

Rockfalls, landslides, and debris flows are likely to increase as a result of increases in precipitation or porewater, or by increased mechanical weathering, with release zones at higher altitudes (Rebetez et al., 1997). As mentioned earlier, more frequent and/or larger avalanches will result from changes in weather patterns, including more frequent high-intensity winter storms, and other changes in the cryosphere or permafrost layers (Evans and Clague, 1997). For both landslides and avalanches, there would be changes in runout zones, which would be likely to increase in area; the vegetation would be kept at earlier successional stages. This could be beneficial for species adapted to these habitats (Suffling, 1993). Increased avalanche management could scare wildlife and affect patterns of visitor use, as they might not wish to visit areas where active avalanche control is being practiced (Murray, pers. comm.).

OTHER CURRENT OR FUTURE STRESSES

There are a host of additional stresses that may affect montane systems as climate change progresses. These include:

- soil changes, with influences on the growth of plants (Kundzewicz and Parry, 2001; Theurillat et al., 1998);

- changes in fire frequencies, with effects on heathlands, grasslands and forest ecotones (Gitay et al., 2001);

- changes in cloudiness, humidity, and precipitation in areas covered by tropical montane cloud forests, and resultant changes in these ecosystems and associated hydrological regimes (Foster, 2001);

- increasing wind velocities, especially associated with intensified tropical cyclones (Giorgi and Hewitson, 2001) affecting evapotranspiration rates, fire probabilities, wind erosion, etc.;

- changes in populations of insects and diseases and their impacts on host plant or wildlife populations (Scurrah, pers. comm.), and the effect of the spread of malaria, leishmaniasis, dengue fever and other diseases on tourism and travel markets in mountain areas currently free of these diseases (McMichael and Githeko, 2001).

Potential Adaptation Options

Adaptation is a potentially useful tool to alleviate some of the negative impacts of climate change, or to take advantage of the positive effects. Adaptation will not halt climate change; rather, it is a strategic tool to manage vulnerability and alleviate impacts to ecosystems. Adaptations can be either in response to observed climate changes or anticipatory. Anticipatory adaptation is more difficult as it relies on sound scientific predictions of the likely impacts of climate change in specific areas, but there is often great uncertainty about these effects, particularly in mountain areas (Smith, 1997). However it is very unlikely that adaptation after the fact could prove successful.

Mountain areas are characterized by a great diversity of situations: ecological, economic, cultural, administrative, political, etc. There is also great diversity in the number and extent of protected areas in particular mountain ranges. Many mountain ranges have no protected areas (e.g., the Atlas, Papua New Guinea, mountains of Myanmar), and existing reserves often protect only the summits with no provisions for connectivity among reserves or the maintenance of entire ecosystems (Hamilton, 2002). Climate change provides an added impetus to the need to address the conservation of alpine and montane biodiversity at regional and global scales, recognizing that, as many mountains are the frontiers between protected areas, administrative areas or states, transboundary cooperation is often necessary.

IDENTIFYING FUTURE CHANGE

General Circulation Models (GCMs) have been widely used to try to predict climate change. When coupled with atmosphere-ocean circulation theories (Atmosphere-Ocean General Circulation Models or AOGCMs), these are the most powerful tools currently available for climate change prediction (Giorgi and Francisco, 2001) (although see Hannah 2003, this volume regarding Regional Circulation Models). They are most useful at the regional scale: e.g., sizeable portions of continents, such as northern Europe, or western North America. However, as mountain terrain and microclimates are complex systems with many interlinkages and autovariations, prediction models may become insufficient, a problem exacerbated by the fact that few mountain ranges have adequate data for validation (Beniston et al., 1997; see discussion by Hannah, this volume, on strengths and weaknesses of models). While different climate models are generally in agreement on certain issues, such as an intensified summer monsoon (Cusbasch and Meehl, 2001), they also predict quite different futures for the same region not only in terms of temperature and precipitation (Hay et al., 2000; Whetton et al., 2001), but also the frequency of El Niño events (Cusbasch and Meehl, 2001), which are key drivers for the climates of sub-tropical and tropical mountain regions. Nevertheless, such models have been used to predict environmental changes and species distribution under future climates in mountain areas, e.g., in Canada (Scott and Suffling, 2000), Mexico (Townsend Peterson et al., 2002) and Tibet (Ni, 2000).

Other approaches for defining possible future climates include scenario techniques (e.g., Parry et al., 1988), which may be more appropriate in mountain regions as climatic data are generally temporally and spatially limited (especially at higher altitudes), yet much local expert knowledge is often available. Scenario modeling approaches have been used to evaluate the potential distribution of mammals on isolated mountains in the southwest USA (McDonald and Brown, 1992); alpine plants in New Zealand (Halloy and Mark, 2003), South Africa (McDonald et al., 2002), Norway (Sætersdal and Birks, 1997), and the Alps (Gottfried et al., 1999; Lischke et al., 1998); and plant species and vegetation types in western North America (Fagre and Peterson, 2002).

STRATEGY DESIGN

It is vital to identify what is to be protected: which species, species assemblage(s), habitat(s), or landscape(s) should take priority? In alpine and montane zones, there is likely to be competition between several ecosystem types which could become established under altered climate regimes, and these often require very different management strategies (Price and Haslett, 1995). For those concerned with conservation, it is also important to recognize the need for the continued availability of key resources, including personnel and infrastructure, and the need to operate within a regional cultural and economic context. According to the Intergovernmental Panel on Climate Change (IPCC) (Gitay et al., 2001), adaptation options are limited in mountain areas because these biomes are the most vulnerable and will respond most strongly to changes in climate.

To understand likely vulnerabilities of natural and cultural resources to climate change, it may be appropriate to conduct an integrated assessment (e.g., Yin and Cohen, 1994) or climate sensitivity analysis (Peine and Berish, 1999), including the following stages:

- identify ecosystems, ecotones, species, and ecosystem processes that are particularly sensitive to climate change;
- evaluate protected area and forest resource monitoring programs as to their capability to detect ecosystem dynamics associated with climate change;
- utilise results of the analysis for education materials and services;
- devise a regional research and monitoring agenda;
- identify all significant non-climate related threats and plans to minimize them.

While quantitative approaches will be essential at all of these stages, they should be complemented with approaches drawing on traditional ecological knowledge (Thomson et al., 2000). Knowledge derived from long-term experience is important for addressing an issue as long-term and complex as climate change.

Possible adaptation options are grouped into eight sets below: protected areas, conservation networks, bioregional approaches, participation and active management, monitoring, infrastructure, supporting policies and minimizing non-climate-related stresses. In all cases, the key principle is to maintain the maximum variety of possible options, recognizing that they will not necessarily help species and ecosystems at the highest elevations. In addition, means to minimize non-climate stresses (Stresses and vulnerabilities, pg. 90)—for instance, minimizing pollution and pressures from land-use practices and change, appropriate and controlled tourism and infrastructure development, limiting water use, control of alien species—should always be implemented as integral components of adaptation strategies.

PROTECTING ADEQUATE AND APPROPRIATE SPACE

Given the prospect of upslope shift of habitat space, one key element of adaptation is to ensure that each mountain protected area—and especially its core zone(s) if zoning is practised—has as great a range of elevations, slope aspects, and habitat mosaics as possible, and is as large as possible. The principle of maximizing diversity also applies to protected area systems, which should include several replicates of different ecosystem types, to permit adjustment to rapid climate shifts and survive major events (e.g., hurricanes and tropical storms) (Barborak, pers. comm.). Consideration may be given to designing mountain protected areas to be climatic refugia, in particular where this has occurred in the past, so that these areas can act as potential habitat for climatic migrant species. This requires, however, adequate connectivity within the landscape (Markham and Malcolm, 1996), as discussed below.

CONSERVATION NETWORKS

A conservation network is a further strategy towards the protection of landscapes, habitats, or species threatened by climate change (Bennett, 1999; Bridgewater, 1996; Hannah, 2001). Adaptation is allowed in such a network through species migration via buffer zones, protected corridors (especially along watercourses), matrices or landscape connections, ‘stepping stones’ through anthropogenically altered terrain (e.g., agricultural land, planted forests, urban areas). Such networks need to be designed carefully—in mountain areas this is particularly important where spatial variability and migration routes for range-shifts are limited. Protected area systems should be designed to maximise connections, corridors or landscape units. These must be able to cross political boundaries, especially in mountains, which often form such boundaries. Such a system must be both dynamic and large-scale (regional to global), requiring regional co-ordination, a focus on biodiversity hotspots, and a proactive adaptation strategy (Hannah, 2001). Such networks are under development in a number of mountain regions, including the Albertine Rift, the Andes, the Apennines, the Australian Alps, the Rocky Mountains, and the Western Ghats (Hamilton, 2002).

At the global scale, one key initiative is UNESCO’s World Network of Biosphere Reserves (WNBR). There are currently 408 sites in 94 countries; nearly half of these are in mountain areas (Price, 2001). They differ from national protected areas in that they are designated internationally, forming a global network; have transitional zones (‘zones of cooperation’) with flexible boundaries; and can straddle international boundaries. Integrated monitoring of biosphere reserves is being undertaken at the global scale through the Biosphere Reserve Integrated Monitoring Program (BRIM), which includes floral, faunal, and socio-economic data. While not all biosphere reserves are currently managed according to the principles laid down in the Statutory Framework of the WNBR

(Price, 2002), they do represent a unique structure for addressing many of the adaptation options listed previously, and for global exchange of experience and best practice. A new program involving mountain biosphere reserves within the Mountain Research Initiative is currently under development (www.mri.unibe.ch).

BIOREGIONAL APPROACHES

From the Convention on Biological Diversity to the EU Natura 2000 and other regional networks to smaller-scale action plans, biodiversity strategies now acknowledge that bioregional approaches are necessary (Johnson et al., 2001). They can be used as a tool for adapting to global climate change in the following ways:

- large regions accommodate full ecosystem functions and habitats, fostering ecological viability over the long term;
- different zones can be used to experiment on and study the impacts of climate change;
- monitoring can happen in a more controlled manner in protected core areas;
- adaptive management responses can be tried in both buffer zones and more widely within the bioregion, especially in areas adjacent to protected areas;
- effective partnerships involving many stakeholders take account of the multiple needs of the strategic approach, including economic sustainability and social acceptance;
- strategic models can be developed to work within limits imposed by increasing fragmentation.

Bioregional strategies allow managers to establish and maintain protected area boundaries which are flexibly designed to adjust to changing climatic regimes, and if necessary, to move upslope with protected habitat(s), providing buffer zones and refugia. Such approaches also take into consideration resource uses at the regional scale, recognising that climate changes will result in changed pressures from local people as conditions for cultivation and settlement change (Price and Barry, 1997).;

Particularly because alpine and montane ecosystems do not usually cover large contiguous areas, their conservation has to be considered within the context of the management of surrounding forest ecosystems (whether or not within protected areas) and other regional land uses, using conservation networks and bioregional approaches. Again, the great variety of land ownership and use patterns in mountain regions around the world makes it difficult to be prescriptive. Nevertheless, one relatively common feature of mountain regions is communal ownership and/or management, particularly at higher altitudes, especially for grazing. Even where these institutions have lapsed or been removed by government action, the cultural roots often remain and can be built on. This is a critical link to modern approaches to conservation, which recognise the need to involve local people in the management of projected areas (e.g., Stevens, 1997; Stolton and Dudley, 1999), whether as park staff or involved in economic activities based on either the utilisation of resources within protected areas (e.g., grazing, collecting plants and mushrooms, hunting) or providing services to tourists. In Hawaii, the value of the traditional ahupua`a system, involving experts and diverse stakeholders, has been specifically recognized as appropriate for adapting to climate change (Shea et al., 2001).

PARTICIPATION AND ACTIVE MANAGEMENT

All of the spatial approaches mentioned above require the explicit consideration of land uses within and adjacent to mountain protected areas, and therefore the active involvement of local people as partners in the conservation of biodiversity. The involvement of local people in conservation is particularly important in mountain areas, where spatial networks are essential to minimise the risk of loss of small isolated populations and to allow more mobile species to move along altitudinal and ecological gradients. Equally, people whose families have been established in a mountain area for many generations will have extensive traditional ecological knowledge with regard to issues such as responses to past periods of environmental change (Glantz, 1988) and the existence of key resources, such as the last streams to dry up in periods of drought. Especially, but not only, in developing countries, achieving conservation goals in mountain areas—whose people are often among the poorest at national scales (Huddleston et al., 2003)—may require the coordination of a number of approaches aimed at improving local people’s livelihoods in order to decrease their dependence on natural resources. This can include supporting local people in the management and conservation of natural resources, providing academic and skill enhancement training or financial support, or investing in small businesses or infrastructure, such as micro-hydro schemes (Sherpa, pers. comm.). NGOs often have key roles to play in such initiatives. They, the protected area, and other conservation-oriented personnel may be involved in specific management strategies such as:

- prioritizing actions to protect key threatened species, particularly ‘keystone species’, such as important pollinators or seeders. This is of particular value in mountain areas where altitudinal range may be utilised to maintain high levels of biodiversity in the face of greater biological redundancy;

- protecting valuable species in situ and avoiding or reducing additional stress on their habitats, for instance through the control of grazing or recreational pressures;

- constructing snow barriers to ‘catch’ snow potentially increasing the likelihood that snow is available for plants and animals that depend on it during the winter months (Good, pers. comm.);

- modifying site conditions to ensure that decreasing plant communities are replaced by other naturally occurring communities of lesser extent, e.g., snowpatch communities by others that occur naturally in areas with little or no snow—an approach comparable to that implemented successfully in highly eroded areas following many years of domestic stock grazing in the Australian Alps (Good, pers. comm.);

- propagating endangered species *ex situ* and, possibly, transplanting them to sites where they have a reasonable chance of survival (Good, pers. comm.; Halloy and Mark, 2003);

- management of invasive/alien species according to the strategy: 1) prevent any new introductions of invasives; 2) ensure early detection and eradication of new infestations; 3) contain any established populations that are too large to eradicate; 4) prevent the largest infestations from spreading (Wittenberg and Cock, 2001);

- releases of water from dams to allow the survival of riparian and aquatic populations and the continuity of key annual activities (e.g., fish spawning) (Parkstrom, pers. comm.).

MONITORING

A key element of every adaptation strategy must be to monitor both its implementation and the changes in the physical and biological environment (e.g., with long-term meteorological stations and regular biological surveys). Monitoring should be interdisciplinary and integrated, and is also necessary to assess model-based predictions (Fagre and Peterson, 2002). While specific monitoring approaches must be developed and implemented for individual protected areas and/or mountain ranges, it is also worth considering involvement in one or more of the relevant international programs that have been, or are being, developed for mountain areas.

The Global Observation Research Initiative in Alpine Environments (GLORIA) aims to establish an effective long-term observation network for detecting the effects of climate change on mountain biota on a global scale. The GLORIA Multi-Summit approach (Pauli et al., 2001) provides such a method, and is designed to compare biodiversity patterns along fundamental climatic gradients, vertically as well as horizontally. GLORIA is a contribution to Activity 1 of the global Mountain Research Initiative (Becker and Bugmann, 2001), developed within the context of the International Geosphere-Biosphere Programme (IGBP), International Human Dimensions Programme on Global Environmental Change (IHDP) and the Global Terrestrial Observing System (GTOS). It includes four activities:

1) Long-term monitoring and analysis of indicators of environmental change in mountain regions;

3) Integrated model-based studies of environmental changes in different mountain regions;

4) Process studies along altitudinal gradients and in associated headwater basins;

5) Sustainable land use and natural resource management

All of these activities, as well as those considered in the previous paragraphs, require suitably-trained on-the-ground, research, and management personnel. The knowledge of local people (‘traditional ecological knowledge’) may also be valuable both for monitoring and for developing testable hypotheses for research (Danby et al., 2003).

INFRASTRUCTURE

To facilitate adaptation measures, physical infrastructure is essential, particularly in the many mountain areas where accessibility is a challenge. Management for the survival of specific species and habitats—as well as travel, both seasonally or year-round—may require the stabilisation of slopes, especially after damage resulting from extreme events. Given the limited availability of resources, mapping of risks may be necessary to assist in prioritising actions (Rossi, pers. comm.).

POLICIES

The essential underpinning to all adaptation options comprises appropriate plans and policies, both for specific areas (e.g., protected areas and administrative regions) and for sectors and agencies (Kumaran, pers. comm.). These need to identify problems and priorities, and include appropriate legal provisions and economic instruments to ensure their application. Again, given the fact that many alpine and montane ecosystems straddle state or administrative frontiers, transboundary instruments (Sandwith et al., 2001; Villeneuve et al., 2002) may often be necessary.

MINIMIZING NON-CLIMATE-RELATED STRESSORS

Interactive effects between climate-related and non-climate-related stressors are common, and organisms and communities that are already stressed may be less resistant and resilient to the challenges posed by a changing climate. Thus any attempts to mitigate effects of climate change must consider ways to reduce the influence of other anthropogenic stressors. Siting reserves in remote locations relatively unaffected by human activities is one way to do this; when creating reserves in less pristine environments, every effort should be made to control human impacts. Even human activities that have been sustainable in the past may cease to be so as climate change alters mountain ecosystems which, as discussed earlier, tend to be less resilient than other terrestrial ecosystems even under the best of conditions. It may be necessary to revisit extractive and development uses of montane regions, and increase our efforts to further limit them. Issues such as water abstraction may require particular attention as human uses compete with biodiversity needs as water becomes scarcer in mountain regions.

Implementation of actions and/or policies

As noted in Policies section, above, adequate information and appropriate plans and policies, including economic instruments, are essential for adaptation. Individual protected areas require effective management plans which are implemented, but it is critical that these are embedded in, and supported by, government policies that recognise that the conservation of biodiversity requires appropriate information and has to be undertaken at the landscape scale, within the context of regional economies—and that many environmental services and other benefits derive from the wise management of mountain resources, particularly those of protected areas (Phillips, 1998). This means that regional and national planning and land-use plans and policies should incorporate guidelines and indicators that ensure that biodiversity conservation targets are met—as required under the Convention on Biological Diversity to which most governments are signatories.

ECONOMIC AND LEGAL INCENTIVES

To assure the implementation of such plans and policies, new economic approaches are being developed. Examples include schemes in Colombia, Costa Rica, and Ecuador under which downstream irrigators and hydro-electricity companies pay upstream landowners to ensure reliable flows (Koch-Weser and Kahlenborn, 2004); and other schemes in Latin America through which landowners are paid to maintain biodiversity through national funds deriving from energy taxes, debt swaps, and carbon offsets (Barborak, pers. comm.). While the funds deriving from these schemes have mainly been provided to people owning forests (Pagiola et al., 2002), the same people often own or manage land higher in watersheds; and grazing regimes often have important consequences on runoff patterns, as well as biodiversity. Key points are that:

- economic incentives are generally needed to ensure the stewardship of private land and water resources within wider conservation networks, and must be supported by appropriate legal mechanisms;
- compensation payments must be based on recognition by the ‘downstream’ beneficiaries of the (increasing) scarcity of resources for which they are willing to pay compensation;
- these payments should be high enough to serve as an incentive to upstream land owners/managers to change their practices; often payments should be linked to the provision of other services and facilities, e.g., for education, health, and access;
- the long-term success of conservation strategies, especially outside protected areas, requires understanding and acceptance—thus existing laws, customs, and institutions must be taken into account, and targeted education is often essential;
- some key areas of land need to be managed for specific conservation goals; this may require purchase, which implies the availability of appropriate funds (Barborak, pers. comm.);
- funding has to be consistent; there have been too many ‘boom-and-bust’ large projects (Kumaran, pers. comm.).

FINANCIAL RESOURCES

In many mountain areas around the world, tourism is a major element of regional economies (Godde et al., 2000). Revenues from tourism are both of importance for development, and also current or potential sources of income for biodiversity conservation, whether in protected areas or in wider networks. The need to link biodiversity conservation and development is evident in the many projects of the Global Environmental Facility (GEF) in mountain ecosystems, which are the topic of one of the 10 operational programs with regard to biological diversity within the GEF Operational Strategy. In 2002 this program was funding 107 mountain projects—including initiatives in Central and South America and the Caribbean, East and Southern Africa, and Central Asia (Global Environment Facility, 2002). Also of note, given the likelihood of the expansion of invasive species into montane and alpine zones, is the Secretariat of the Global Invasive Species Program, supported by the World Bank, with projects in areas such as the mountain fynbos of South Africa (World Bank, 2002).

As well as multilateral funding, bilateral and NGO funding can be directed towards activities that contribute to adaptation in relation to climate change, again stressing the links between development (including tourism) and biodiversity conservation. Such activities may be within one country or at regional scales, though transboundary projects can run into funding difficulties where mountain ranges fall within more than one of the administratively-defined regions of a funding agency, e.g., the Albertine Rift which straddles both the East and Central African regions (Plumptre, pers. comm.). Funding, for instance through trust funds, may also be essential for the purchase of key parcels of land, e.g., adjacent to the core areas of existing protected areas, or to create corridors or buffer zones (Barborak, pers. comm.). National-level funding, from government agencies and/or NGOs, is essential for all aspects of adaptation, from research and monitoring through planning and land acquisition to active management activities.

Conclusion

Even if the resources can be found to implement many of the actions discussed above, with support from local people, governments, and the private sector, there are likely to be significant losses of many alpine and montane species in coming decades—though it should also be recognised that certain species will benefit from climate change. Climate change is a new factor of change affecting montane and alpine ecosystems and species, adding to the existing stresses described in Stresses and vulnerabilities section, pg. 90, deriving from local pollution, agriculture, forestry, freshwater abstraction, tourism and recreation, infrastructure development, and the introduction of alien species. Reductions in these direct stresses are an essential prerequisite to all adaptation strategies; to be ensured through actions at all levels from individual behavior to the effective implementation of national and international legal instruments.

Yet the key focus for adaptation is not in the mountains, but among the people, governments, and businesses of an increasingly urbanized planet whose people are ever more dependent both on the fossil fuels whose combustion causes climate change and on water deriving from mountain areas. To make these links, those responsible for mountain protected areas should take the lead in promoting the development of, and utilising, state-of-the-art technologies in alternative energy sources and energy conservation. These are often particularly appropriate in mountain areas because of their high solar radiation, steep watercourses, and windiness, and because costs of connection to regional or national grids and of transmission are high (Schweizer and Preiser, 1997). Similarly, state-of-the art methods of water use and management should be implemented in mountain protected areas. The implementation of such technologies locally will have minimal direct effect on alpine and montane ecosystems. However, well-illustrated, clear interpretative materials and programs addressed to the millions of people who visit mountain protected areas could have an important impact in raising awareness of the vital heritage they protect—and the ways in which it is endangered by climate change.

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Building Resilience in Arctic Ecosystems

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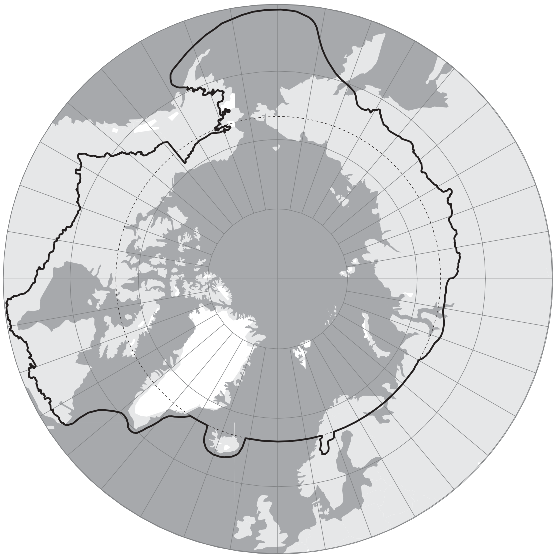
IN THE ARCTIC, CLIMATE CHANGE is expected to be among the most rapid and extensive of any region on Earth. Warmer temperatures and melting ice dramatically altering the structure and function of arctic ecosystems over the next century. However, the impacts of climate change cannot be considered in isolation from other forces of change affecting the region. For example, while arctic habitats are relatively unmodified by human activities, economic development in the region depends on the exploitation of both living and non-living resources. As a consequence, there is added potential for impacts on biodiversity.

The challenge this chapter faces is to provide suggestions for action that can be taken in response to cummulative impacts, the details of which are poorly understood, while the effects are likely to be widespread and substantial.

Crucial Aspects of Arctic Ecosystems

While various definitions exist to describe the geographical extent of the Arctic, we use the boundary defined by the Conservation of Arctic Flora and Fauna (CAFF) working group in this discussion (Figure 1). It encompasses the ecosystems found from the forest-tundra transition zone northward on land, and in adjacent marine areas north to the arctic basin.

Figure 1: The CAFF boundary is used in this discussion to delimit the Arctic. Source: CAFF.



Since the Arctic is a vast geographical area distinguished by a variety of landforms and complex interactions between land, water and the atmosphere, the characteristics of biodiversity in arctic ecosystems are unique (Hansell et al., 1998). Arctic ecosystems contribute significantly to the diversity of life on Earth since they contain many species, habitats, ecological processes and biological adaptations not found elsewhere on the planet.

Relative to other ecosystems, the Arctic is species-poor. On an evolutionary time scale, the terrestrial ecosystems are very young and rapidly evolving (Jonasson et al., 2000). Many species in the Arctic live within marginal habitats at the edge of their ranges. A species at the edge of its range tends to be highly sensitive to environmental conditions, so the abundance and distribution of its population can undergo wide swings as conditions change. While few endemic genera and species are found in the Arctic, there are relatively more endemic subspecies, and the genetic diversity within a species can be high (Jonasson et al., 2000). The few endemic species that are found in the Arctic are adapted to a range of habitats and food resources that otherwise would be occupied by competing species if they were present.

The dominant feature of the marine environment is sea ice. Pack ice is present at the North Pole year round permitting limited light penetration into the water column and resulting in low primary production. Around the periphery of the Arctic Ocean and in the marginal shelf seas, ice forms and melts annually. Seasonal ice cover varies from year to year. The annual release of freshwater during the spring melt creates a highly productive environment. Large numbers of organisms from all trophic levels can be found along ice edges, leads and polynyas where the interaction of ice, sunlight and water currents is greatest (Stirling, 1997).

These highly productive waters sustain important fishery resources. Fisheries in northern waters account for about 47% of total global production (FAO, 2002). The western Bering Sea has the largest fish biomass for Pacific cod and cod-like fishes, the world's largest pollock fishery is in the eastern Bering Sea, and the Barents' capelin stock is potentially the largest in the world (Duda and Sherman, 2002; NOAA et al., 2003). Other commercially valuable species found in northern waters include herring, halibut, salmon, and shrimp.

The Arctic has been home to indigenous peoples for millennia. Their historical patterns of resource use reflected the environmental conditions of the day. These conditions often varied from season to season and year to year, and settlement patterns correspondingly adapted to resource availability. Today, the traditional use of living resources for hunting, trapping, fishing and reindeer husbandry remains an important part of the culture of many arctic indigenous peoples. The importance of this special relationship to land and resources in the Arctic is reflected in the growing number of political and institutional arrangements at the local, national and international levels. Home rule in Greenland, land claims in Alaska and parts of northern Canada, and the continuing discussions of such matters in Scandinavia and Russia, has led to varying degrees of self-government, land ownership, and influence over ecosystem and resource management decisions by indigenous peoples in the Arctic.

As part of their natural function, arctic ecosystems provide goods and services that are crucial both locally and globally. To maintain the flow of ecosystem goods and services it is critical to maintain ecosystem function including the cycling of water, the cycling of nutrients, the flow of energy, and biodiversity (Pimm, 1986). In the Arctic, local goods and services include the resource base that provides food, supports cultural traditions, opportunities for recreation, research and education, economic activity and energy resources. The Arctic is also intimately linked by air masses, ocean currents, river systems and migratory species to other ecosystems around the globe, thereby providing global goods and services. Some examples of these include: the storage of 40% of the world's reactive soil carbon, global heat transport through ocean currents and energy exchanges with the atmosphere, freshwater input into the Arctic Ocean which may influence the thermohaline circulation of the global oceans, and fisheries contributions to the global food supply (IPCC, 2001a). The Arctic is a source of potentially large feedbacks to the global climate system. The net effect or time scale of these feedbacks is poorly understood; nevertheless, the impacts of ecosystem changes on the ability of arctic ecosystems to provide these various goods and services will be felt across the world.

Stresses Due to Factors Other Than Climate Change

The influence of human activity can now be detected in even the remotest regions of the Arctic. Climate change is only one of the many factors affecting the future of arctic ecosystems. Industry, infrastructure development, and large-scale resource exploitation are advancing in arctic ecosystems. Major economic and community development benefits are expected in connection with these activities. At the same time serious threats to the cultural, spiritual and environmental heritage of the Arctic are also likely.

Forces of change in arctic ecosystems include overfishing, shipping, the exploitation of natural resources, pollution and the long-range transport of contaminants, and increased ultraviolet (UV) radiation due to the depletion of stratospheric ozone (AMAP, 2002; CAFF, 2001; IPCC, 2001b). These non-climate stresses may impede the ability of arctic ecosystems to respond to climate change.

OVERFISHING

Today, global production from capture fisheries and aquaculture is currently the highest on record and is very important for global food security. While recent increases in capture fisheries production have come mainly from the southeast Pacific, fleets based in the eight arctic countries are among the top producers and account for more than 14% of global fish landings (FAO, 2002).

Unfortunately, the number of populations that are overfished, as well as the indirect effects of fisheries on marine ecosystems through bycatch and habitat destruction, indicate that management has failed to achieve its principal goal of sustainability (Botsford et al., 1997). Since 1975, several stocks in the Barents Sea have suffered from overfishing leading to substantially reduced yields in the 1970s and 1980s. While strong fisheries management measures have since been put into place, actual catches of fish have frequently exceeded the advised limits (Nakken, 1998).

In addition, certain harvesting methods pose a serious threat to non-target species. Extensive bottom trawling for cod and prawns, as well as sea-floor dredging for scallops, takes place throughout the Arctic. Such fishing techniques can alter the structure and diversity of complex benthic communities by leveling the seafloor, removing slow-growing sedentary organisms, and increasing suspended sediment loads.

SHIPPING

Shipping has a significant impact on the marine environment through atmospheric emissions, noise, leakage of antifouling agents, and operational discharges of sewage, garbage, sludge, bilge, solid oil and waste oil (Table 1). Discharges resulting from accidental events such as collisions, groundings, and fires have major environmental impacts. Among wildlife, the biophysical conditions of the Arctic have resulted in specialists that are long-lived, slow to mature, and have conservative reproductive strategies. The loss of mature adults has a disproportionate impact on the ability of the population to renew its numbers (Musick, 1999). Large oil spills and similar accidents can be devastating because mature adults are the most impacted, leaving behind a younger population that is slow or unable to recover.

Table 1: Maritime operations and their impact on the marine environment.
Source: after PAME (2000).

Activity/operation	Issue of concern
Onboard production of oily wastes, sewage and garbage	Illegal discharges to sea
Discharge of ballast water of foreign origin	Biodiversity impacts due to competition or predation from introduced species
Loading and unloading activities	Increased risk of discharges of oil and bilge water
Tanker traffic	High accident potential
Heavy bunker oil as cargo and fuel	Substantial negative impacts when discharged
Operation in areas of high ice concentration	Increased accident risk and greater risk of marine pollution
Tugging / Towing	Increased accident risk
Cruise / Passenger vessels	Large-scale operational discharges plus increased accident risk near ice

Climate change will favor increased shipping along high-latitude routes (IPCC, 2001b). Moreover, exotic organisms and pathogens carried in ships' ballast water have devastated native flora and fauna all over the globe, and in the Arctic where most organisms are likely to be at the margins of their thermal tolerances, warming may give exotic species increased opportunities to survive, spread, and form problem populations.

RESOURCE EXPLOITATION

If not properly managed, commercial exploitation of the Arctic's natural resources—including fisheries, wildlife, oil and gas, and minerals—can seriously threaten biodiversity, ecosystem function, and traditional cultural practices. Cumulative environmental effects assessment recognizes that the effects on the environment from individual human activities can combine and interact with each other to cause effects that may be different in nature or extent from the effects of the individual activities themselves (Hegmann et al., 1999).

The National Research Council in the United States (2003) undertook a study of the cumulative environmental effects of oil and gas activities on Alaska's North Slope since 1965. The NRC's findings indicate that Alaska's environment and culture have been significantly affected by oil infrastructure and activities. Effects have accumulated from road building, damage from off-road travel, interference with subsistence activities, and social changes in local communities—although these are noted to be both positive and negative.

Animal populations have been affected in a number of ways. Bowhead whale migrations seem to have been displaced by the noise of seismic exploration. In addition, garbage and food produced in oil field operations have resulted in higher than normal densities of predators (e.g. polar bears, arctic foxes, and glaucous gulls) which prey on the eggs, nestlings and fledglings of birds. As a result, the reproduction rates of some bird species in industrial areas, including various geese, eiders and shorebirds, are in some years insufficient to balance death rates (NRC, 2003).

In the Arctic there is a significant degree of overlap between biologically important areas and the continental shelf where considerable reserves of hydrocarbons exist (Stirling, 1997). Threats associated with the operation of offshore facilities include the risk of blowout, discharges of produced water, chemicals and drilling muds, leaks from pipelines, sea floor disruption, and increased shipping to and from installations and processing facilities. A large oil spill in marine waters would likely have substantial effects on animal populations because current cleanup methods can remove only a small fraction of spilled oil under conditions of broken ice. At present there is just one truly offshore facility operating in the Arctic, the Northstar facility in Alaska, however planning for a number of new facilities is currently underway.

CONTAMINATION AND POLLUTION

Environmental contaminants occur at relatively high levels in some arctic species and, in many cases, the long-term biological and ecological effects are not yet known. Local pollution from industrial activities, such as mining and oil and gas development, can negatively affect the environments surrounding these developments. However, in the mid-1980s, persistent organic pollutants (POPs) were discovered to be accumulating in arctic food webs (Downie and Fenge, 2003). These include PCBs, pesticides like DDT, dioxins, furans, and brominated flame retardants which are transported to the Arctic through air, river and ocean currents from points of origin further south. POPs accumulate in the body fat of animals, so marine mammals and scavengers at higher trophic levels are most susceptible. Mammals such as the polar bear, arctic fox, killer whale, northern fur seal, and birds such as the glaucous gull, all tend to carry high contaminant loads (CAFF, 2001).

Mercury, lead and cadmium are all present in the Arctic. According to the most recent assessment of arctic pollution, there is a trend of increasing mercury levels in marine birds and mammals in the Canadian Arctic and possibly also western Greenland; cadmium levels in some seabirds are high enough to cause kidney damage; and while levels of anthropogenic radionuclides in the Arctic are declining, releases from reprocessing plants have resulted in increases in levels of technetium-99 and iodine-129 in the European Arctic (AMAP, 2002).

OZONE DEPLETION

The release of the synthetic chemicals called chlorofluorocarbons (CFCs) into the atmosphere, through various industrial processes and other human activities, causes ozone depletion in the stratosphere. The problem is most pronounced over polar regions during spring because of cold temperatures and lack of sunlight. For each 1% decrease in stratospheric ozone, the amount of UV-B radiation reaching the Earth's surface increases by 1.5% to 2%. Enhanced UV-B radiation can have subtle but long term impacts on ecosystem processes that reduce nutrient cycling and indirectly decrease productivity (Fergusson and Warlde, 1998).

Since the signing of the Montreal Protocol in 1987, which banned the use of CFCs, the rate of ozone destruction has slowed to around 3-5% per decade. There is preliminary evidence of a reversal in the Earth's ozone decline (Newchurch et al., in press), however, global warming may interfere with ozone recovery by altering the way in which air mixes in the lower stratosphere. Model simulations suggest that warming over Northern Hemisphere continents could delay arctic ozone recovery (Shindell, 2003), thus posing a longer term threat of ecosystem damage due to increased UV radiation.

Present and Future Stress Due to Climate Change

The Arctic is already demonstrating clear evidence of change consistent with what is expected to result from warming temperatures, and matches trends that have been projected by general circulation models. According to the most recent assessment carried out by the Intergovernmental Panel on Climate Change (IPCC), extensive land areas in the Arctic show a 20th century warming trend in air temperature by as much as 5°C accompanied by an increase in precipitation (IPCC, 2001b). As a result, regional studies have been undertaken to assess the impacts of climate change in the Mackenzie Valley (Cohen, 1997), Bering Sea (Weller and Anderson, 1999), and Barents Sea (Lange, 2002). Recognizing the general vulnerability of arctic ecosystems, the Arctic Council[3] has commissioned the Arctic Climate Impact Assessment (ACIA). The goal of ACIA is to gather knowledge on climate change and ultraviolet radiation in order to provide reliable information to governments and people; environmental, human health, social, and economic impacts will be included in the assessment (ACIA, 2000). A peer-reviewed volume synthesizing the current science of arctic climate change will be published late in 2004.

Early results from ACIA show that while the greatest warming is expected over the Arctic Ocean, by 2070 temperatures will on average increase by 3.4°C for the area north of 60° latitude; this is double the projected global average. Changes of this magnitude will irrevocably alter arctic ecosystems, leaving behind a natural world that will be nearly unrecognizable to today’s arctic residents.

Patterns of climate change within the Arctic are complicated, as they vary in rate and magnitude by region and by season. Some anticipated impacts of climate change in terrestrial and marine ecosystems are summarized below, as are vulnerabilities.

TERRESTRIAL ECOSYSTEMS

Arctic landscapes have already begun to reveal the impacts of climate change. Regions underlain by permafrost have decreased in extent. A general warming of ground temperatures and the thickening of the active soil layer has been observed in many areas, particularly at the southern margins of the Arctic close to and south of the treeline (IPCC, 2001b).

Important ecological changes that appear to have been triggered by warming are also being documented in terrestrial ecosystems across the circumpolar north. Reduced nutritional value of caribou and moose browse, decreased water availability, and increased forest fire tendencies have been detected (Weller and Lange, 1999). The Arctic Transitions in the Land-Atmosphere System (ATLAS) program has recorded an advance and infilling of trees at the treeline, as well as an expansion of shrubs in the northern portion of the Arctic (Chapin, 2002). These vegetation changes may also explain some of the summer warming that has recently been recorded in northern Alaska and the other regions of the Arctic.

The effects of climatic warming on local environmental conditions have not gone unnoticed by the indigenous communities living in the Arctic. Krupnik and Jolly (2002) documented observations of change from the perspective of twenty-three indigenous communities. Elders report that they are seeing species of birds and animals farther north than previously recalled, and new species are appearing that have never been seen before. Elders also note that the weather is more unpredictable than it used to be (Jolly et al., 2002).

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[3] The Arctic Council (www.arctic-council.org) is a high-level intergovernmental forum established by the countries whose borders cross the Arctic Circle: Canada, Denmark, Finland, Iceland, Norway, the Russian Federation, Sweden, and the United States. Decisions and priorities are set with active participation and full consultation of arctic indigenous representatives.

Changes in arctic climate are expected to continue throughout the 21st century and persist for many centuries to come, bringing with them major physical, ecological, sociological and economic transformations (IPCC, 2001b). The greatest changes in temperature are projected to take place during the winter months; extreme cold temperatures are expected to be less severe and occur less often. Precipitation in terrestrial ecosystems will increase by 10-20% in summer months, and by 5-80% in winter months (IPCC, 2001b). In almost all regions, these projections are well beyond the range of variability of the current climate patterns.

As warming occurs, there will be changes in species compositions with a tendency for poleward and elevational shifts in species assemblages, the establishment of new assemblages of species, and the loss of some polar species (IPCC, 2001b). Vast areas of the Arctic may develop entirely different ecosystems from those that exist currently (Everett and Fitzharris, 1998). Ecosystem models project that the area of tundra will decrease by two-thirds from its present size, due to an expansion of the boreal forest (Everett and Fitzharris, 1998). Vegetation changes that result from warming are expected to further enhance regional temperature increases in the Arctic. Chapin (2002) has found that altering the vegetation of the tundra from its current state of no shrubs to one of being shrub-dominated would increase mean July temperatures by 1.5 to 3.5 °C.

Modifications to wildlife populations are also projected to result from warmer temperatures in the Arctic, including changes in population size, structure, and migration routes. A consensus has not been reached on the probable impacts of climate change on ungulates. On the one hand, changes in the timing and location of food sources, an increase in parasites and insect-borne disease, and more insect harassment may lead to declines in animal populations such as caribou and muskox (Gunn, 1995). On the other hand, caribou appear to be highly resilient to changing environmental conditions because they are generalist feeders (Callaghan et al., 1998). Insects will benefit from a warmer Arctic, since many insects are constrained from expanding north due to cold winter temperatures (Parmesan, 1998). Permafrost melting may substantially alter ecosystems in situations where the melted permafrost results in landslides, leading to poor water quality that is detrimental to fish and other wildlife.

Small increases in temperature will amplify the melting of snow and ice, the hydrology of the North is particularly susceptible to warming since snow and ice drive virtually all of the major hydrological processes and related aquatic ecosystems in the Arctic (IPCC, 2001b). The runoff regime is expected to be driven increasingly by rainfall, with less seasonal variation in runoff. Because the ice cover on lakes and rivers will be thinner, freeze up later, and breakup earlier, the rates of primary productivity in aquatic ecosystems will be affected (Rouse et al., 1997). Also related to hydrology is the ability of arctic wetlands to act as a source or sink for carbon dioxide (CO_2) and methane. Peatlands may dry out because of increased evaporation and plant transpiration, thus becoming a source of greenhouse gases. With a doubling of atmospheric CO_2 concentrations, the southern boundary of peatlands in Canada is projected to move northward 200-300 km

(Gignac and Vitt, 1994). It is still uncertain as to whether the current status of northern peatlands as a global sink for CO_2 will change to a source.

MARINE ECOSYSTEMS

Arctic sea ice is both an indicator of climate change (through changes in extent and thickness) as well as a factor capable of influencing global climate through ice-albedo feedbacks and thermohaline circulation.

Sea ice is sensitive to temperature changes in the air above as well as in the ocean below. Winter maximum ice extent decreased by approximately 3% per decade through the 1980s and 1990s (Parkinson et al., 1999) while summer minimum extent has shrunk by 9% per decade over the same period (Comiso, 2002). New period-of-record minima have been reached several times over the past ten years (Serreze et al., 2003). Ice thickness, as observed from submarine transects, is estimated to have decreased by 40% between 1958 and 1997 (Rothrock et al., 1999), however, models have shown that these observations may not necessarily be true for the whole of the Arctic Ocean (Holloway & Sou, 2001).

Global warming, as forced by observed greenhouse gases and tropospheric sulfate aerosols, has been shown to cause reductions in the area covered by sea ice (Vinnikov et al., 1999). Probable changes in sea ice over the next century will alter the albedo of the surface, creating a feedback to the global climate system. Sea ice is important because it reflects more incoming solar radiation than the sea surface (i.e., it has a higher albedo). Therefore, a reduction in sea ice gives a positive feedback on climate warming. Significant changes in albedo over large areas also have the potential to produce a nonlinear, accelerated change (IPCC, 2001b).

The Arctic Ocean plays a significant role in the thermohaline circulation of the world's oceans. Most coupled ocean-atmospheric models show a weakening of thermohaline circulation from the North Atlantic by 2100 due to increased freshwater input from large arctic rivers and melting of the Greenland ice sheet. Although there remains considerable uncertainty as to the likelihood of an irreversible collapse of global thermohaline circulation, perturbations caused by increased freshwater inputs resulting in a reorganization of global ocean circulation can lead to abrupt climate change (IPCC, 2001b; Manabe and Stouffer, 1994).

The most recent assessment of circumpolar climate change indicates that primary production in marine environments will likely increase, species assemblages will shift northward, and ice-associated species will decline (IPCC, 2001b). It is important to recognize, however, that changes in water temperature, ocean currents and sea ice regimes will be non-uniform across arctic marine ecosystems and our understanding of marine biodiversity is insufficient to draw reliable conclusions about the probable effects. While impacts on individual species and species groups can be generalized by changes in physical habitats, food availability, and predator-prey relationships we should expect surprises along the way! Arctic marine food webs can be very complex but with only a few key species connecting the different levels, so changes in one trophic level can easily propagate to others.

CHANGES TO PHYSICAL HABITATS

The changes in physical habitats likely to result from different water temperatures and sea ice regimes will produce both winners and losers. Warmer temperatures will favor many fish species while the reduction of sea ice will have substantial impacts on marine mammals (IPCC, 2001b). Temperature appears to be a major determinant in several aspects of fish ecology and recruitment seems to be significantly better in warm years than in cold years; the same is true for growth (Loeng, 1989). Distributions of fish stocks that are determined by water temperature are likely to retract north while new species introductions will arrive from southern waters.

There will be negative consequences for marine mammals and seabirds dependent on ice for breeding and foraging (Tynan and DeMaster, 1997). Changes in the extent and type of ice cover will reduce the ability of polar bears to access prey, forcing them to move north or to stay on land for longer periods, thus increasing nutritional stress and lowering reproductive success (Stirling and Derocher, 1993). Polar bears in Wapusk National Park in western Hudson Bay are expected to be extirpated from the park (Scott and Suffling, 2000), although population numbers there are currently stable (Lunn et al., 2002). Ice-associated seals, especially ringed seals, are particularly vulnerable to changes in the extent and character of sea ice because they depend on ice for so many aspects of their life cycle (Tynan and DeMaster, 1997). True arctic cetaceans—bowhead, beluga and narwhal—spend much of their time in areas with significant amount of ice cover, having specialized in foraging for ice-associated species, and will likely see increased competition from migratory whale species (Kovacs, 2003).

CHANGES TO FOOD AVAILABILITY

Thinner ice cover will increase the solar radiation penetrating to the underlying water thereby increasing photosynthetic production. Earlier melting of the ice in spring will also extend the growing season, although this is dependent on local changes in upwelling, vertical mixing, and freshwater inputs (IPCC, 2001b). Of increasing interest are the effects that the Arctic Oscillation and North Atlantic Oscillation (AO/NAO) have on ecological processes (Stenseth et al., 2002). This atmospheric pattern accounts for major variations in weather and climate around the world and may affect the relative timing of food requirement and food availability known as the “match-mismatch hypothesis” (Cushing, 1990). The concept of match and mismatch is essential in food-web energy transfers: a match implies that the predators are located in the same space and at the same time as their prey, while mismatch implies that they are not.

Match and mismatch are important for primary production in terms of grazing (which drives food webs) and sedimentation (which controls nutrients) (Sakshaug, 2003; Sakshaug et al., 1992), however it can also be applied to the foraging success of many seabirds

(Kovacs, 2003). Seabirds are vulnerable to changes in prey availability particularly during the breeding season where a major constraint on breeding distribution is the distance between suitable nesting sites on shore and feeding zones at sea. If the temporal distribution of macrozooplankton shifts outside of the reproductive season, or their spatial distribution moves away from nesting proximity, then a mismatch occurs which can result in reproductive failure. Matches and mismatches are both likely to occur and suggests that the impacts of climate change on sea birds will vary geographically (Kovacs, 2003).

CHANGES IN PREDATOR-PREY RELATIONSHIPS

As individual species adjust to climate change, interactions among species will also change, especially in predator-prey relationships. If ice-inhabiting seals or walrus, for example, are forced to haul out on land to complete their molt or maintain proximity to food sources, they could be more exposed to new predators such as grizzly bears and wolves (Lowry, 2000). At the same time, studies of fish species in the Barents Sea show that changes in sea temperature may increase metabolic rates of cod and result in an increase in consumption of capelin by 100,000 tons per degree Celsius (Bogstad and Gjøsæter, 1994).

Some of the changes in predator-prey relationships may actually represent the natural adaptive capacity of the species themselves. In western Hudson Bay, for example, where direct impacts to polar bears have been observed due to a reduced feeding period in the late spring and early summer when seals—especially young ringed seals—are most available (Stirling et al., 1999), studies of alternative prey species are underway. These include assessments of species which may change in their relative and absolute abundance in relation to changes in sea ice and winter availability of open water. It is possible that populations of bearded and harbour seals may increase in western Hudson Bay as the amount of open water during winter increases. If so, these alternate prey species will become more important in the diet of polar bears there, potentially prolonging the survival of the western Hudson Bay population, at least in the short term while there is still enough annual ice present in the bay (Stirling pers. comm.).

VULNERABILITIES

Arctic ecosystems are often considered to be fragile. However, some arctic scientists are skeptical of this assessment since many arctic species are generalists that can survive over a wide range of conditions, making them highly disturbance adapted. The high genetic diversity and the vast numbers of sub-species within arctic populations are a reflection of how arctic species have adapted to local conditions over time. This natural adaptive capacity is an important characteristic that can help protect arctic species from the impacts of environmental changes. Endemic species, however, are vulnerable to both competition by species invading from the south and habitat loss; this is especially true for ice-associated species such as the walrus, ringed seal and polar bear. In addition, the Arctic's low-diversity, species-poor ecosystems have limited functional redundancy (CAFF, 2001). In such ecosystems, certain ecological roles may depend on relatively few species. As a consequence, while an individual species may be resilient to environmental alteration, the system as a whole may be vulnerable to the inability of a single species to adapt, especially if that species performs a key ecosystem function.

Climate change has the potential to have significant effects on commercial and industrial activity in the Arctic, resulting in both positive and negative economic impacts (IPCC, 2001b; Maxwell, 1997; Weller and Lange, 1999). Longer, warmer summers could increase tourism and the number of visitors to the region. Increases in precipitation could require costly upgrades and redesign of tailing dams and water diversion structures in the mining industry. In addition, an increase in the length of the annual frost-free period could affect access to many oil and gas exploration sites, currently reached via winter roads built on frozen ground. In areas where permafrost is susceptible to climate change, melting may cause the foundations of buildings to shift unevenly, the rupture and buckling of pipelines and storage tanks, and the structural integrity of older buildings, water supplies and waste disposal infrastructure to be threatened. Failure to retrofit old structures as conditions change will pose a risk of serious polluting events.

Due to the potential implications of global warming for traditional ways of life, resource development and conservation, the projected impacts of climate change are of considerable concern to residents of the North. Northern indigenous peoples have demonstrated their resilience to change over time; however, the cumulative effects of climate change and human development may result in unexpected challenges to cultural sustainability. Shifts in the habitat and diversity of food species due to climate change could impact the cultural and religious lives of some indigenous peoples (Gitay et al., 2002). Changes to traditional diets have already resulted in an increased incidence of diabetes, heart disease and obesity in some indigenous populations, and any additional dietary changes may be detrimental to the health of northern residents. Langdon (1995) concluded that the combination of alternative cultural lifestyles and altered subsistence opportunities resulting from global warming may pose the greatest threat of all to the continuity of indigenous cultures.

Assessment of Adaptation Options

If ecosystems are to adapt successfully to climate change, effective management must support and build on species’ and ecosystems’ natural resilience while reducing vulnerabilities.

In an environmental context, resilience is the ability of ecosystems, habitat types and species to maintain a relatively constant state in the face of disturbance and stress, and to recover quickly after a temporary disturbance (Noss, 2001). Some arctic species are more resilient to climate change than others, for example by being better able to migrate to more favorable habitat; thus, in the marine environment pelagic communities are generally more resilient than benthic communities.

Vulnerability is the susceptibility of ecosystems, habitat types and species to the adverse impacts of change. As defined by the IPCC, vulnerability is a function of the character, magnitude, and rate of change of climatic variation to which a system is exposed, as well as its sensitivity and adaptive capacity. Some arctic species—for example, those that depend on sea ice—are more vulnerable to the effects of climate change than others.

Because global warming is expected to change many of the defining characteristics of the Arctic, there are few options for adapting to climate change. In fact, the best chance arctic ecosystems have for long-term biodiversity conservation is to slow, and eventually stop, anthropogenic climate change. However, two traditional conservation approaches can be applied to support the resilience of species and ecosystems: habitat protection and the reduction of non-climate stresses.

HABITAT PROTECTION

The establishment of protected areas is the conventional way to protect landscapes, ecosystems, and habitats. Identifying and protecting key areas prior to development is a luxury in many areas of the Arctic that is no longer possible in some of the more developed parts of the world. However, industrial development and commercial activity in the Arctic are increasing pace, and experience from other regions shows that once major development begins, options for large-scale habitat protection are foreclosed. While the establishment of protected areas is important for conservation, it is highly unlikely that the more traditional approach to designating protected areas in the Arctic will be adequate in the long-term to protect biodiversity, given the magnitude of ecosystem impacts that are projected to arise from climate change. New and creative approaches to protecting habitats will be required.

The vast majority of arctic ecosystems remain unprotected, especially in the marine environment. Yet arctic governments have committed to protecting a circumpolar network of protected areas under CPAN[4]. Establishing a representative network linking terrestrial, coastal and marine ecosystems must be a top priority. Special emphasis should be on representing habitat types across environmental gradients following the expected path of climate change and shifting habitats, and protecting heterogeneous ecosystems and habitats with high species diversity. Protected area design should be guided by the following.

PROTECT RESILIENT AND VULNERABLE ENVIRONMENTS

An urgent priority for scientists and managers is to identify and protect particularly resilient and vulnerable habitat types, species, and populations/stocks. This analysis should include resilience and vulnerability to the effects of climate change, as well as the cumulative effects of non-climate stresses. Many sources of knowledge can contribute to such an analysis, including traditional knowledge, field studies and modeling.
Protect keystone species

Certain species are so essential to ecosystem function that their disappearance, or a sharp reduction in their numbers, can result in the disappearance or reduction of other species. These so-called keystone species thus play a key role in the maintenance of healthy ecosystems. Resource managers should identify keystone species for their ecosystems and implement conservation strategies for them, such as precautionary harvesting targets.

PROTECT ALONG CLIMATIC GRADIENTS

While protected areas can be established to either shelter keystone species or to conserve representative ecosystem types, perhaps the most significant influence on the ability of ecosystems in the Arctic to adapt to climate change is the ability of arctic flora and fauna to move northwards with rising temperatures. Protected area strategies will need to ensure the continuity of habitat areas along environmental gradients.

PROMOTE CONNECTIVITY

The degree of connectivity between protected areas, along with the uses adjacent to them, influence the resilience of flora and fauna to climate change (Feenstra et al., 1998). Removing impediments to migration and preventing the creation of new impediments will be critical to facilitate the northward movement of species with climate change. This can be achieved by linking protected areas through the establishment of migration corridors.

PROTECT CLIMATE REFUGIA

Due to localized climatic conditions, some habitats are more resistant to the effects of climate change than others. These areas are known as climate refugia. A priority for scientists and managers should be to identify potential climate refugia for arctic species; an example might be the high Canadian archipelago where polar bears have easy access to both feeding areas on the ice and denning areas on land. A conservation priority should be to protect climate refugia from non-climate stresses and include them in reserve systems that link climate-vulnerable habitat to refugia.

AVOID FRAGMENTATION

Many arctic species require non-fragmented habitats to maintain healthy populations. Similarly, indigenous peoples require large stretches of undisturbed landscapes in order to maintain their traditional practices (CAFF, 2001). Although the Arctic still boasts of some of the largest stretches of undisturbed habitats on the planet, shifting ecosystems caused by climate change, and habitat disturbance and fragmentation caused by development activities, are increasing threats and may become barriers to the northward migration of species. One of the best methods available to avoid habitat fragmentation is for land-use planning processes to incorporate this issue at the pre-development stage.

PROTECT AT MULTIPLE SCALES

Biodiversity conservation strategies need to be implemented at the local, regional and circumpolar scales. Such strategies are essential for maintaining healthy ecosystems and conserving arctic biodiversity. It is important to network protected areas on a circumpolar basis because no single country can ensure habitat protection for migratory species, and because critical areas that must be protected to maintain the biodiversity and productivity of the entire arctic ecosystem will often fall under multiple jurisdictions.

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[4] The Circumpolar Protected Areas Network (CPAN) was established in 1998 and is a subgroup of CAFF, the Arctic Council's working group for the Conservation of arctic Flora and Fauna.

A strategic plan to establish a circumpolar network of protected areas has been completed by CAFF and endorsed by the eight arctic countries (CAFF, 1997). The plan specifies national and circumpolar actions to fill gaps in habitat protection. While CPAN was not established exclusively to combat climate change, this network will help to maintain ecosystem function and the flow of goods and services from and within arctic ecosystems, goals that are consistent with facilitating the natural response of ecosystems to climate change.

The initial burst of creating new protected areas, particularly in the Russian Arctic, has since come to a standstill. While protected areas currently cover approximately 15% of the terrestrial Arctic area, they are unevenly distributed across ecosystems and habitats; over 35% of arctic glaciers are protected, but less than 5% of the forest tundra (CAFF, 2001). In the coming years, CPAN will need to re-establish the momentum for creating protected areas and increase both protection of marine areas as well as productive terrestrial habitats.

REDUCTION OF NON-CLIMATE STRESSES

There is growing evidence that healthy species and ecosystems are more resilient to environmental change, including climate change (Burton, 2001). In general, the stress caused by climate change will lessen resilience to non-climate stresses such as pollution and vice versa, meaning that currently acceptable contaminant thresholds may eventually be too high. Therefore, another key adaptation strategy for arctic ecosystems is the dramatic reduction of non-climate stresses.

HARVESTING REGIMES

Harvesting wild species is perhaps the most common form of natural resource use across all regions and peoples of the Arctic (Freese, 2000). Since the distribution of flora and fauna in the Arctic is likely to change with the changing climate, changing the location or intensity of harvesting activities may become necessary. In those situations where populations of harvested species are showing serious declines, substituting store bought foods for traditional foods may need to be considered. However, this option would likely prove to be problematic for many indigenous communities, particularly in remote areas where food costs are high and where traditional foods are an important way of life.

Since hunting and fishing are of such importance to the health and well-being of arctic peoples, a better option is to manage habitats and harvesting activities to ensure an ecologically sustainable supply of wild species into the future. Freese (2000) conducted a review of what must be done to ensure that the use of wild species in the Arctic does not compromise the biodiversity and ecological integrity of the region over the long term. This review provides fifteen guidelines[5] that concern the consumptive use of wild species. If implemented, the cumulative impacts of climate change and harvesting activities on populations of wild species could be avoided.

In addition, stricter fishing quotas and the establishment of no-take zones are necessary to address the problem of overfishing. Nakken (1998) advises that management authorities take a more precautionary approach when setting total allowable catch limits due to the discrepancies that exist between recommended limits and actual catches. His research indicates that setting limits at or below estimates for total allowable catch could lower the rate of exploitation and stabilize or even increase the yields of most commercial fish stocks.

SHIPPING STANDARDS

Climate change will likely result in an expansion of shipping due to year-round access along the whole of the Northern Sea Route and Northwest Passage, as well as new oil and gas developments throughout the Arctic. This implies a significantly increased risk of accidents. While shipping will always entail some level of environmental risk, much can be done to reduce this risk, inter alia:

- The quality standards of ships operating in arctic waters must be high. Compliance with quality standards must be ensured by strict port controls of the vessels.

- Discharge regulations in the Arctic must be very strict, for example by granting high-latitude waters special area status under various annexes of the International Convention for the Prevention of Pollution from Ships (MARPOL). Compliance with the regulations must be ensured through surveillance and monitoring, coupled with effective sanctions toward violations.

- Ship traffic must be steered away from the most sensitive locations, for example by the use of mandatory shipping routes and “no-go” areas.

- Close surveillance and monitoring of shipping activity is needed to prevent collisions and for early detection of ships in distress.

- The whole, or parts, of the Barents Sea should be granted status as a Particularly Sensitive Sea Area (PSSA) by the International Maritime Organization. Within a PSSA, a wide range of protective measures can be applied to reduce the risks involved in shipping.

ARCTIC POLLUTION

Local sources of pollution are regulated by legislation within each arctic nation. However, because many contaminants originate from outside the Arctic, the mitigation of arctic pollution requires international action. Indigenous peoples were instrumental in drawing international attention to the issue of long-range transport of POPs (Downie and Fenge, 2003). Subsequently, international negotiations led to the signing of the Stockholm Convention, a global treaty signed by 151 countries that regulates the disposal, use and release of some of the worst offenders in this category of pollutants; however, about 20 nations still need to ratify the convention before it enters into force.

Unfortunately, ongoing monitoring of pollutants in the Arctic shows that other chemicals are increasing in volume, especially brominated flame retardants and organic mercury which both cause effects similar to those chemicals already banned by the Stock-

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[5] Available at http://www.wwf-canada.org/NewsAndFacts/Supplemental/ConsumptiveUseOfWildSpecies.pdf

holm Convention (AMAP, 2002). Regional and global agreements on these and other dangerous contaminants are necessary, as is sufficient testing and regulation of both commonly used and new chemicals. Support must also be given to the European Union's proposed regulatory system called REACH—Registration, Evaluation and Authorization of CHemicals—which will bring much needed scrutiny to the health and environmental hazards of chemicals in current use.

INVASIVE SPECIES

Invasive species are any species—including its seeds, eggs, spores, or other biological material capable of propagating that species—that are not native to an ecosystem and whose introduction does or is likely to cause economic or environmental harm. While not all invasive species cause ecological stress, some can displace or eliminate native species, disrupt interactions between native species, reduce biological and genetic diversity by hybridizing with native species, introduce parasites or diseases, and cause socio-economic and recreational impacts (CAFF, 2001). Some studies have shown that ecosystems with high biodiversity have higher resistance to exotic invasions. Therefore the Arctic, with its low biodiversity, might be more vulnerable to exotic invasions than other regions in the world. Presently, the magnitude of the threat of invasive species on arctic environments is unclear; however, the potential impacts of this threat warrant further investigation and precautionary action on species introductions, especially since climate change is expected to result in the migration of new species into the region.

TOURISM IMPACTS

The Arctic has seen considerable growth in tourism over the last decade. CAFF estimates that more than 1.5 million tourists visited the region in the year 2000, but when the entire state of Alaska is included in this estimate, the count approximately doubles. Nature and culture are the primary attractions in the Arctic. There is evidence that tourism can benefit conservation efforts by helping to provide an economic value for nature and wildlife, and creating support for protected areas (UNEP, 1995). While many visitors to the region come to see pristine wilderness, they are also more likely to be aware of environmental issues, and through their experience can become advocates for Arctic conservation in their home countries.

However, tourism is also a potential threat to the environment (CBD, 2002). Common concerns are increased fragmentation through permanent infrastructure, damage to soils and vegetation, wildlife disturbance, and increased waste and pollution. Ship-based tourism accounts for a large portion of the total tourism numbers in the Arctic, as it provides tourists with comfortable access to the remotest areas; in some arctic destinations, cruise tourism is the primary means of travel to and within the area. In this context, questions of waste water management, ballast water and exotic species introductions, and preparedness in case of accidents must be raised.

In cooperation with the tourism industry and other stakeholders, WWF has developed guidelines[6] for reducing impacts and increasing benefits of tourism for nature and local people in the Arctic. Following these guidelines will help to reduce the potential stresses that tourism can have on Arctic ecosystems.

Guidelines for Selecting Strategies

The pace at which change is occurring requires that comprehensive adaptation strategies be developed by arctic nations in the near-term. Under the conditions of a changing climate, the challenge is to adjust human activities in such a way as to protect ecosystem function, maintain the flow of goods and services, and enhance the resilience of arctic ecosystems. The best defenses against biodiversity impacts in the face of climate change will require comprehensive strategies that involve local residents and ecosystem-based management.

INDIGENOUS PARTICIPATION

Human habitats in the region are diverse with settlements ranging from small, remote communities to modern, industrial cities. Arctic communities have both formal (e.g. based on resource extraction) and informal (e.g. based on hunting and herding) economies that are dependent on the living and non-living resources throughout the region.

Out of a total population of around 2 million, there are about 500,000 indigenous people living in the Arctic. Their historical presence in the Arctic demonstrates the incredible resilience of indigenous cultures through time, nonetheless, indigenous peoples are more sensitive to climate change than non-indigenous groups because climate change will directly affect their traditional hunting habitats and subsistence species. Certain forms of hunting are delayed or abandoned under poor ice conditions and traditionally important species such as salmon, herring, walrus, seals, whales, and various birds are likely to undergo shifts in range and abundance (IPCC, 2001b).

The importance of involving indigenous peoples in selecting adaptation options and developing comprehensive management strategies cannot be over-emphasized. One of the clear challenges to understanding past and current climate change in the Arctic is that our scientific understanding is based on records that are often short-term, fragmentary, or both (Huntington, 2002). For generations, arctic residents have made first-hand observations of climate and environmental variability. This intimate knowledge of the land provides specific and detailed insights into changing local and regional conditions. Traditional ecological knowledge adds an important dimension to the understanding of climate change impacts that cannot be achieved through models or other scientific methods. Because the combined perspectives of science and traditional knowledge generate a broader understanding of environmental change than either knowledge system can accomplish on its own (Kofinas, 2002), understanding and addressing climate change in the Arctic cannot be done adequately without incorporating the views of indigenous peoples (Huntington, 2002).

Residents of the Arctic are best positioned to identify local vulnerabilities and develop locally-appropriate adaptation responses, especially since vulnerabilities to climate change vary across the region. In addition, involving indigenous peoples in adaptation decision-

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[6] Available at http://www.ngo.grida.no/wwfap/core/about/guidelines.html

making respects the magnitude of the impacts of climate change on their traditional ways of life, and the patterns of existing and emerging self-governance and land ownership.

ECOSYSTEM-BASED MANAGEMENT

Resource managers must take into account long-term and cumulative impacts of human activities and environmental change. Through a broad and transparent process, managers should set 30-year regional or ecosystem-wide management goals that focus on conserving ecosystem structure and function. Ecosystem-based management frameworks offer an inclusive process to assess ecosystem health and the shared goals among stakeholders, and will typically yield large-scale spatial management plans regulating various types of use (e.g. through zoning, access, protection, quotas, etc.).

Ecosystem-based management (EBM) is based on the precautionary principle, which seeks to minimize the risk of damage, in particular when knowledge is scarce or non-existent. The following overview is derived from *Policy Proposals and Operational Guidance for Ecosystem-Based Management of Marine Capture Fisheries* (Ward et al., 2002) and is offered here as a means by which to approach managing the multiple stresses and competing opportunities within arctic ecosystems.

The concept of EBM has evolved over the past few decades in response to two characteristics of managed natural systems:

1. That exploited natural resources are highly connected to their surrounding ecosystems and this connectivity can have major effects on their productivity; and

2. The exploitation of natural resources can have effects on other resources and on other (non-utilized) species as well as aspects of the ecosystems where the resources occur, and these direct and indirect effects can have very major consequences for related or dependent species.

These two properties can be summarized as (1) the effect of the environment on the resource being exploited, and (2) the effect of resource exploitation on the environment. EBM attempt to address both of these environmental and ecosystem interactions.

Some approaches to EBM advocate a strictly ecological focus to maintain the capacity of an ecosystem to deliver desired goods and services. Other approaches extend the EBM concept to include human goals and aspirations for ecosystems. These latter approaches recognize the highly managed nature of terrestrial systems in particular, and that the notion of *sustainability* is driven by the socio-economic and cultural context within which resource management must reside (Pirot et al., 2000). It is this approach that needs to be applied to our use of resources and activities in all arctic environments.

Despite the diversity of views and experience with EBM in various jurisdictions, reasonable consensus is emerging across a broad range of different resource sectors (forestry, civil society, marine) about basic principles of EBM (Harwell et al., 1996; NOAA, 1999; Pirot et al., 2000; Ward et al., 1997). These principles can be summarized as:

1. Maintaining the natural structure and function of ecosystems, including the biodiversity and productivity of natural systems and identified important species, is the focus of management.

2. Human use and values of ecosystems are central to establishing objectives for use and management of natural resources.

3. Ecosystems are dynamic; their attributes and boundaries are constantly changing and consequently interactions with human uses also are dynamic.

4. Natural resources are best managed within a management system based on a shared vision and set of objectives developed among stakeholders.

5. Successful management is adaptive and based on scientific knowledge, continual learning and embedded monitoring processes.

Integrated approaches propose managing ecosystems on a regional basis and considering all uses in the context of their impacts on biodiversity. These approaches to resource use and biodiversity conservation entail agreements from all users to reduce activities that may degrade specific areas or values of conservation importance, but permit activities to occur in areas where they do not threaten regional biodiversity objectives. The regional management approach identifies specific uses that are acceptable, and identifies complementary protected areas to ensure that biodiversity is maintained. Within this context, ecosystem-based management is consistent with building ecosystem resilience to the impacts of climate change.

Recommendations

Biologically speaking, the term adaptation is used to describe the evolution of organisms or species through time; in this chapter, adaptation is defined as human-driven initiatives to support the natural resilience of arctic ecosystems and reduce their vulnerability to the adverse consequences of climate change. In this sense, adaptations are conscious, planned decisions by human beings that may result in an autonomous response by ecosystems.

The threats to arctic ecosystems are not limited to small areas or single species. In the long term, the resilience of arctic flora, fauna and peoples depends heavily on both global and local actions to preserve the integrity of arctic ecosystems. The only response to the magnitude and diversity of pressures facing the Arctic is to manage the human activities that adversely impact biodiversity.

STABILIZE LEVELS OF GREENHOUSE GASES IN THE ATMOSPHERE

The best chance arctic ecosystems have for long-term conservation is to slow, and eventually stop, anthropogenic climate change. This requires broad scale global action on reducing greenhouse gas emissions. Resource managers have an important role to play here by engaging on climate policy and using examples of change seen in their systems as indicators of the need for action.

The goal of the United Nations Framework Convention on Climate Change (UNFCCC) is to stabilize levels of greenhouse gases in the atmosphere to levels that prevent dangerous anthropogenic interference with the climate system. While mitigation is generally not considered an adaptation option *per se,* in the long term, this strategy will have the greatest benefit in terms of reducing the vulnerability of arctic ecosystems to climate change, and supporting the natural adaptive capacity of arctic ecosystems to adjust to a new climate. Since the dominant response of arctic species to climate change is believed to be relocation rather than evolutionary adaptation, immediate action to reduce atmospheric CO_2 concentrations will give arctic ecosystems more time to adapt to a changing climate.

SUSTAINABLE DEVELOPMENT

Overall, there will be increased human activity in the Arctic as a result of climate change, especially as improved sea access will enhance opportunities for expanded fisheries, new transportation routes, and development of the vast offshore oil and gas reserves. If managed wisely, the resources can bring long-term wealth and development to the region. This, however, demands that renewable resources, which can continue to support local economies and cultures long after non-renewable resources have been depleted, are not sacrificed for short-term economic gains.

WWF promotes the Conservation First principle, which was designed to balance nature conservation and industrial development. Conservation First means there should be no new or expanded large-scale industrial development in the Arctic until areas of high conservation value are identified and protected. This will safeguard important cultural and wildlife areas from industrial development for the long term. It also provides planning certainty and predictability for communities, investors, developers, government, and other stakeholders.

Conservation First is an important part of a broader ecosystem approach to the management of arctic regions. What is new about Conservation First is its focus on timing: that area protection must take place before industrial development begins and forecloses options for proactive conservation. Successful programs will:

1. Operate within a supportive policy framework.

2. Recognize economic, social and cultural interests as factors that may affect resource management.

3. Recognize ecological values and incorporate them into management.

4. Provide adequate information on exploited species and habitats to ensure that development is low risk.

5. Ensure that the resource management system is comprehensive and inclusive, based on reliable data and knowledge, and that it uses an adaptive approach.

6. Consider environmental externalities within the resource management system (Ward et al., 2002).

The Convention on Biological Diversity acknowledges that substantial investments are required to conserve biological diversity. It also points out that, in return, conservation will bring significant environmental, economic and social benefits. Implementing the Conservation First principle in the Arctic has three major advantages:

FOR COMMUNITIES: It conserves renewable natural resources and ecosystems that have been the basis for human survival in the Arctic for thousands of years and will be the basis for long-term, sustainable development in the future.

FOR CONSERVATION: It secures the survival of key species, ecosystem components, and processes that are important to and representative of the region. Some areas also have ecosystem functions far beyond the region itself, for example as havens for migratory species, ground-water preservation, or moderators for larger-scale climate processes.

FOR BUSINESS: The process allows conflicts to be identified and resolved before major investments are made, providing certainty and predictability for investors, developers, governments, conservationists, and other stakeholders.

Large-scale exploitation of non-renewable resources, with the accompanying growth of infrastructure and industry, provides local economic and community benefits, but industrial development also poses serious threats to the cultural, spiritual, and environmental heritage of the Arctic. Conservation First ensures that these threats are minimized. It provides a way to maintain the integrity of arctic ecosystems as an integral part of planning for development.

Conclusion

The uncertainties concerning climate change projections are large. They stem from the gaps in knowledge of ocean and atmospheric processes, from our assumptions about the future, from the models themselves, and from inherent limitations in our ability to project the climate. Uncertainties pose a significant challenge to any effort to develop robust management strategies but the lack of complete understanding does not preclude effective ecosystem management. While we know relatively little about the nature of changes to biodiversity, enough is known to justify immediate conservation action (Costanza et al., 1998; Ludwig et al., 1993).

Projected climate change will have sweeping impacts on arctic ecosystems and careful management of natural resources will be required to secure healthy ecosystems for the future. Management efforts must be based on what is known about ecosystem components and their interrelationships, as well as the likely effects of and on human activities. As our understanding of the changes to biodiversity improves, management responses must also adapt. In many cases, adapting to climate change will not involve adopting entirely new courses of action; rather, the strengthening and the expansion of existing conservation practices may be adequate (Burton, 2001).

Discussions and development of adaptation responses in the Arctic are currently only in their very early stages. Because of the vulnerability of the region, the expected rate and magnitude of change, and some of the uncertainties related to what is known about ecosystems in general and how they will be impacted, there are few concrete management prescriptions that can be recommended and none that are being tested or implemented at the present time. Nevertheless, in order to lay the groundwork for future responses to climate change it is extremely important for resource managers to begin including climate considerations in their management plans.

Although some important groundwork has been laid, much more work remains in order to ensure the resilience of arctic communities, ecosystems and the traditions of indigenous peoples. Some guiding principles are as follows:

- Resource managers should inventory the ecosystems in their regions and the interrelationships among them, and then assess the probable impacts that climate change will have on them, along with specific adaptation options to ameliorate those impacts.

- Resource managers should only implement adaptation options that are suitable to the local circumstances in their regions, and then monitor their effectiveness in building resilience.

- Resource managers need to work closely with arctic residents (particularly indigenous peoples) in adaptation research, planning, decision-making, implementation, and monitoring of arctic habitats.

- Resource managers need to encourage mechanisms to enhance information flow among researchers and policy-makers on climate change policy and adaptations in practice.

- Resource managers need to document the changes that are taking place in the Arctic and communicate these changes broadly as arguments for international action on emissions reductions.

- To support the work of resource managers, arctic nations need to develop cooperatively a vision that guides the allocation of efforts and resources towards adaptation activities.

- To provide resource managers with more time to develop strategies to facilitate natural systems' adaptation, arctic nations need to be vigilant to ensure that interna-

> tional emission reduction targets beyond the Kyoto Protocol are negotiated in order to achieve the stabilization of greenhouse gases in the atmosphere at levels that prevent dangerous anthropogenic interference with the climate system.

It is important to recognize that adaptation will not be enough to completely protect arctic ecosystems from the forces of climate change. If there is to be any chance of conserving arctic ecosystems in the long term, international efforts toward mitigating the causes of climate change must be put into place. Effective management of natural resources will only buy ecosystems additional time to adjust to a changing climate until broad global action on reducing greenhouse gas emissions takes effect.

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Designing Reserves to Sustain Temperate Marine Ecosystems in the Face of Global Climate Change

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TEMPERATE MARINE ECOSYSTEMS include a wide array of habitats: estuaries, marshes, seagrass and kelp beds, rocky coastlines, sandy, muddy, and cobble shores, the deep sea and the open ocean. To cover the details of conservation and climate change in each habitat type would turn this chapter into a lengthy book. Instead, I will highlight those aspects of the marine realm in general that set it apart from terrestrial ecosystems and discuss their implications for both reserve design and responses to climate change. Because of their relative accessibility, coastal habitats have received the most attention from marine conservation biologists and the public alike, and information presented in this chapter reflects this bias. Still, we should remember that we are only just beginning to understand the rich diversity of life in the deep and open seas, and the ways in which apparently distinct marine ecosystems interact with each other. While designing conservation strategies with only partial information about the ecosystems in questions is difficult, we ignore unfamiliar habitats in our conservation planning at our peril.

Temperate seas are rich in life, and have played a central role in human history. Both coastal and open seas are critical sources of food for humans, and traditionally supplied other resources as well. People turn to oceans and coasts for recreation, employment, and spiritual fulfillment. Population, development, and economic activity are often concentrated in coastal areas, both providing incentive to keep coasts attractive and healthy, and threatening their health and beauty. Despite this, marine ecosystems and organisms are often poorly understood. The first part of this chapter, then, addresses key differences between marine and terrestrial ecosystems.

Although the science of marine reserves is relatively young, reserves have demonstrated success as a conservation tool. The establishment of a reserve will, on average, double the density, triple the biomass, and increase the size and diversity of organisms present by 20-30% (Halpern, 2003). To ensure the long-term success of reserves, we need address their strengths and weaknesses in responding to climate change, which is now an inevitability. West and Salm (2003) have suggested designing reserves to enhance the ability of communities within reserves to be “resistant and resilient” in the face of climate change. What aspects of marine ecosystems and reserves will best serve this purpose? What aspects are most vulnerable to climate change? Parts two and three of this chapter examine current and predicted threats to and changes in marine ecosystems, both as a result of climate change and other factors.

There are already several excellent guides on marine reserve design (e.g. Kelleher 1999; Salm et al 2000). Parts four and five of this chapter provide a brief review of what's known about reserve design and success, discuss it in the context of global climate change, and suggest approaches to reserve design that might be particularly relevant in the face of global climate change. The Strategies for implementation section, pg. 163, offers suggestions for reserve planners wishing to incorporate these suggestions into their own planning. Finally, the Existing adaption programs section, pg. 164, reviews existing approaches to adaptation in temperate marine reserve planning and suggests a few areas where more information would be particularly helpful.

Unique Aspects of the Marine Environment/marine Conservation

Like terrestrial ecosystems, marine ecosystems are diverse mosaics of interacting physical and biological elements. In many other regards, however, marine ecosystems are quite distinct, and any approach to marine conservation must be based on a solid understanding of the unique physical, biological, and sociopolitical aspects of the marine environment. Several key differences are reviewed below; further reading on differences between marine and terrestrial habitats and their implications can be found in e.g. Carr et al. (2003), Day and Roff (2000), or Strathmann (1990). The overriding point is to realize that ecological knowledge gained from terrestrial systems may not be directly applicable to marine systems.

PHYSICAL DIFFERENCES

THREE-DIMENSIONALITY

Compared to air, water is relatively dense, viscous, nutrient-rich, and wet. Thus buoyancy is relatively easy, desiccation isn't a problem, and there is less need for elaborate support structures. As a result, the marine world is strikingly three-dimensional. While terrestrial life exists by and large within 100 m of the Earth's surface, marine communities exist from the ocean's edges to its abyssal planes thousands of meters below the surface. Many organisms spend all or significant parts of their lives in the water column. Thus, in addition to covering about 70% of Earth's surface, the oceans make up about 99% of Earth's biosphere.

TEMPERATURE

The thermal regimes of intertidal and open ocean habitats are distinct both from each other and from those in most terrestrial habitats. Intertidal organisms experience extremes of temperature and desiccation, in some cases experiencing temperature swings of 20 °C between immersion and emersion. Differences in the timing of low tides (midday vs. early morning) can override latitude in determining degree of thermal stress (Helmuth et al., 2002); cyclic changes in the timing of low tides due to lunar oscillations can have significant effects on intertidal zonation (Denny and Paine, 1998). Some intertidal organisms are living close to their thermal tolerance limit (e.g. Stillman and Somero, 1996), in which case even small changes in temperature could lead to dramatic changes in community structure (Bertness et al., 1999). Conversely, because of water's high heat capacity, open ocean and subtidal organisms are generally spared any rapid temperature shifts akin to the day-night or seasonal shifts in terrestrial ecosystems (Harley et al. in press). The buffering capacity of oceans may cause ocean temperatures to rise more gradually than air temperatures as climate change progresses.

PHYSICAL CONNECTIVITY

Waves and currents cause nutrients, sediment, and pollution to mix and travel much more freely in the marine environment than in most terrestrial environments. Thus marine communities may depend on nutrients coming from remote sources, and likewise may be negatively affected by pollutants from distant locales. Nutrient pollution from the Mississippi River has rendered thousands of square kilometers of the Gulf of Mexico uninhabitable creating an area referred to as "the dead zone" (National Science and Technology Council, 2000). On a more positive note, connectivity means that marine populations are less sensitive to habitat fragmentation than terrestrial systems.

ECOLOGICAL DIFFERENCES

THE BENTHOS AND THE PLANKTON

On a very basic level, the marine realm can be divided into two categories: the benthos and the plankton. Benthic organisms are those that live in or on the sea floor. While the most diverse benthic communities are in the photic zone, there is also substantial diversity in the deep sea, beyond the reach of sunlight. Vibrant hydrothermal vent communities thrive in an ecosystem based on chemosynthesis rather than photosynthesis. The availability of water-borne food allows marine ecosystems to support an array of filter feeders, animals that take bacteria, algae, other animals, and debris from the water passing overhead (or overfoot, in the case of barnacles). There are no terrestrial equivalents to filter-feeders (unless one considers spiders).

The oceans also contain an ecosystem for which there is no terrestrial equivalent: the pelagic realm. The most obvious inhabitants of the pelagic world are nektonic organism, strong swimmers like whales and fish who can move independently of oceanographic currents. Less visible but equally important are planktonic organisms, those with weak or absent motility who exert relatively little control over their large-scale movement.

LIFE HISTORIES

Most marine animals have complex life histories wherein adult and larval stages are physically, ecologically, and physiologically distinct. Adults are often relatively sessile, while larvae may travel for tens or thousands of kilometers through the water column. Thus larval supply, dispersal, and recruitment are critical in determining the persistence of adult populations, or the ability of populations to recover from disturbances. Understanding these phenomena will also be critical in designing reserves that can succeed in the long term, particularly as climate change alters current patterns and other elements of marine ecosystems that determine where and when larvae recruit to adult populations. Despite their importance, our understanding of many aspects of larval transport and recruitment, particularly as they affect reserves, is still fairly rough. Many models treat larval dispersal as a spatially symmetrical process, ignoring the existence of unidirectional transport by currents, and model settlement as evenly distributed in space, despite the common existence of regions of high and low settlement (Gaines et al., 2003). The concept of source and sink populations suggests that certain populations may depend mostly or entirely on larvae supplied by distant populations to persist. Protection of adult populations and habitat may thus be insufficient for population persistence.

In addition to having planktonic larvae, many marine invertebrates release gametes directly into the water column, where sperm may become limiting. Since individuals of many sessile or sedentary species cannot or do not move long distances in search of mates, declining population density may lead to significant decreases in fertilization success. This phenomenon is suspected to play a role in the decline of at least one commercially important species, the white abalone (Hobday et al., 2000).

The fecundity of marine animals—producing thousands of eggs per season is quite common—led many to think that marine populations could not be overfished. However, mortality rates for larvae and juveniles are also extremely high, and current rates of population decline show that high fecundity is no panacea.

POPULATION CONNECTIVITY

Because of the abundance of life in the water column and the movement of water over vast distances by currents, marine populations are thought to be much more interconnected than terrestrial ones. Numerous studies of population genetics bear this out, at least for species with planktonic larvae (reviewed in Grosberg and Cunningham, 2001). Adult rafting may add to the connectivity of distant populations. As would be expected, marine species tend to have greater genetic polymorphism and less spatial or geographic structure than terrestrial ones (Carr et al., 2003). This may ignore the existence of cryptic species, however.

The depth of the ocean, the scale of movement of marine organisms, and, for planktonic organisms, their small size makes tracking movement difficult. Thus our knowledge of the extent of movement by marine organisms is fairly limited. One approach to determining movement of planktonic organisms is to assume that they track oceanographic current patterns, and recent models indicate that currents patterns rather than other environmental variables may be responsible for some marine biogeographic boundaries (e.g. Gaylord and Gaines, 2000). Nonetheless, it is clear that even weak swimmers are able to affect their horizontal distribution by altering their vertical position (reviewed in Forward and Tankersley, 2001; Hill, 1998). This has been particularly well demonstrated for larvae of estuarine crab species (e.g. Garrison, 1999), and recent evidence suggests that fish larvae also have mechanisms that allow them to recruit back to natal populations (Warner et al., 2000). Analysis of genetic structure can provide some clues as to the extent of genetic intermixing of populations across a wide geographic range, although it doesn't give information about who moves where. More recently, investigators have used chemical signals preserved in calcium carbonate structures (e.g. bones, statoliths) to infer geographic movement of fish and invertebrate larvae (Thorrold et al., 2002).

FOOD WEB STRUCTURE AND COMPLEXITY

In general, marine food webs are more complex than terrestrial food webs. While terrestrial ecosystems exhibit relatively little lateral transfer of energy or materials, the diverse array of planktonic organisms and planktivores in marine systems allows for significant lateral transfer. Unlike terrestrial carnivores, which actively seek out their prey, many planktivores generally depend on currents to bring their meals to them. In addition, the turnover rate of marine autotrophs is rapid relative to terrestrial ones; phytoplankton divide approximately daily, depending on location, and the majority of macroalgae are annual or short-lived.

NATURAL VARIABILITY AND RATE OF RESPONSE TO CHANGE

Marine ecosystems, by and large, exhibit a more rapid response to environmental changes in climate than do terrestrial systems, probably because of the relatively short lifespan of many key marine organisms (Carr et al., 2003). Such rapid ecosystem changes have been observed in response to cyclic environmental shifts such as the El Niño Southern Oscillation (e.g. Lubchenco et al., 1993) and the Pacific Decadal Oscillation (Hollowed et al., 2001). Many marine species, particularly those lacking planktonic dispersal, also exhibit large interannual variability in population size (Eckert 2003), making it more difficult to identify long-term trends in response to changes in climate.

The rate of response to change, in particular to temperature shifts, may also be related to the relatively large proportion of thermoconformers in marine as compared to terrestrial ecosystems. The vast majority of marine animals are cold-blooded; although many have some techniques for altering their body temperature, they track environmental temperatures more closely than warm-blooded animals.

SOCIOPOLITICAL DIFFERENCES

VISIBILITY AND FAMILIARITY

Aside from intertidal areas, the marine world is largely invisible to humans, except with respect to commercial fishery productivity. Also, the structure of marine communities is conceptually foreign to most people. Because there are no terrestrial equivalents to many marine life histories and community structures, it is difficult for people to appreciate what matters. Also, much of the devastation of marine life occurs beyond the reach of the human eye: destructive bottom trawling is invisible to most, but has a profound effect not just on the previous communities of those areas of the seabed, but on populations of commercially important species. It is this invisibility that allows people to see destructive programs like carbon sequestration in the deep sea as relatively benign. Because we have not explored the deep sea in any great detail, there is a false impression that it is a barren wasteland.

OWNERSHIP AND USE

Currently, less than 1% of oceans are included in reserves, compared to approximately 6% of the terrestrial world (Carr et al., 2003). Marine resources in most regions have been managed as common-pool resources w here open access is the rule. While some coastal cultures have ownership rights for marine resources, in the global market the oceans and their resources are by and large expected to be available to anyone. This has lead to a "tragedy of the commons," wherein there is more incentive for over-extraction than for protection. Unlike terrestrial food industries, in which both plants and animals are harvested extensively and are mostly domestically raised, marine harvest is primarily wild-caught animals. Given the environmental problems associated with marine aquaculture, however, whether or not switching to farmed marine fish would ease the pressure on wild populations is unclear. Perhaps related to this general lack of ownership, marine ecosystems are rarely the result of positive human intervention; in other words, there are few marine equivalents to setting fires to maintain open grasslands, irrigating land to create rice paddies, or the like. While the idea of active intervention to shape terrestrial ecosystems reflects attitudes common in human societies, our approach to marine ecosystems and perhaps conservation as well seems more guided by a desire for an absence of human activity.

Current Stresses on Temperate Marine Ecosystems, Other Than Climate Change

Marine ecosystems face myriad stresses, of which climate change is only one. To adequately address climate change, we must do so in the context of other known stressors. An adaptation strategy for climate change that increases the risk from other stressors would serve little use.

PROBLEMS ASSOCIATED WITH COMMERCIAL FISHERIES

OVERHARVEST AND TROPHIC CASCADES

It is estimated that 90% of the world's large fish, both open ocean and groundfish species, have been lost; once an industrial fishery opens, it takes only 10-15 years to achieve this level of reduction (Myers and Worm, 2003). Although overexploitation of marine stocks is not a new phenomenon, the advent of industrial fishing technology has allowed unprecedented rates and levels of depletion (reviewed in Pitcher, 2001).

It is not simply the loss of fish that matters; the loss of apex predators or keystone species can have profound and long-lasting ecological effects (reviewed in Coleman and Williams, 2002). A classic example is the overharvest of sea urchin predators (or potentially of the predators of urchins' predators) leading to increases in urchin populations and thus decreases in kelp abundance (Estes and Palmisano 1974). Loss of kelp forests can lead to declines in many fish species relying on kelp forests as nursery grounds, and to increases in other populations (Lubchenco et al., 1993). The loss of large pelagic fish may lead to increases in gelatinous zooplankton such as ctenophores and jellyfish (Mills, 1995).

FISHERIES-ASSOCIATED HABITAT DESTRUCTION

Many fisheries use mobile gear that drags along the sea floor, such as trawls and dredges. Such practices lead to long-term decreases in numbers of individuals, species diversity, and biomass, as well as changes in the physical and chemical nature of the habitat (Veale et al., 2000; Kaiser et al., 2000). Areas that have been subjected to long-standing dredging or trawling tend to be dominated by small-bodied organisms that are less likely to get captured by bottom trawls, although whether overall changes in community structure are due to selective species removal or decreased habitat complexity is unclear (National Research Council, 2002).

A different sort of habitat destruction occurs when driftnets are accidentally lost or purposefully discarded when they wear out or are damaged. They then drift through the ocean, entangling animals from fish to mammals.

DISEASES ASSOCIATED WITH AQUACULTURE FACILITIES

When animals are transported around the globe for aquaculture, unwanted organisms inevitably accompany them (Bower et al., 1994). In addition, aquaculture facilities are often overcrowded and stressful, providing a perfect incubator for diseases (e.g. Wagner et al., 2002; Avnimelech and Ritvo, 2003). Diseases are transmitted between farmed and wild animals through escapees and outfall.

NON-NATIVE SPECIES

Most invasions by non-native species occur accidentally as a result of shipping (Ruiz et al., 2000), although even estuaries with no international shipping may have numerous non-native species (Wasson et al., 2001). Invasions also occur due to purposeful introduction of non-natives for aquaculture, primarily fish, bivalves, and crustaceans; these purposeful introductions are often accompanied by the accidental introduction of predators, parasites, or other hangers-on accompanying the species of interest. A well-characterized example of this is the introduction of several predatory oyster drills along with Pacific oyster to the northwest coast of North America. Non-native species have also been introduced by scientific researchers: witness “Freeman’s blight,” a species of tunicate introduced to New England by a visiting researcher at the Marine Biological Laboratories, and now a major pest up and down the coast. Private and commercial aquaria are the likely sources of the “killer alga” *Caulerpa taxifolia* in both California and the Mediterranean. *C. taxifolia* is an aggressive competitor, which outcompetes native seagrasses and algae, thereby radically altering community structure. In most cases, some effort is made to contain the non-native species, but rarely are such efforts entirely successful. Indeed, attempts at biocontrol of both invasive and native pests are yet another source of exotic species (reviewed in Secord, 2003).

Non-native species can have numerous impacts on their new ecosystems, including altering physical structure, changing decomposition rates, out-competing natives for resources or shifting competitive hierarchies, and, of course, eating native species (reviewed in

Grosholz, 2002). Invasive snails have been implicated in the decline and local extinction of native mud snails in California as a result of more efficient feeding and production of offspring (Byers, 2000), and invasive mussels have changed an unstructured mud flat into a mussel reef (Crooks, 1998). The latter invasion completely transformed both the physical and biological structure, resulting in a community assemblage completely distinct from that in the surrounding unaltered mud flats. Invasive species play key roles in structuring rocky intertidal zonation as well (e.g. Leonard et al., 1999). In addition to species- and community-level effects, introduced species can alter trophic transfer within entire ecosystems. A non-native clam in San Francisco Bay (perhaps the most invaded estuary in the world) is such an efficient filter feeder that it has shut down the usual spring phytoplankton bloom, shifting primary production from planktonic to benthic ecosystems (Alpine and Cloern, 1992). For a review of both some effects of and problems associated with assessing the ecological impact of invaders, see Parker et al. (1999) and Grosholz (2002).

PHYSICAL DISTURBANCE

COASTAL DEVELOPMENT

Human populations have always been concentrated in coastal areas, and it is no different today. As the human population continues its exponential growth and lifestyles in many cases become increasingly resource-intensive, coastal marine communities around the world are showing signs of strain. Around 60% of the world’s population lives within 100 kilometers of the coast (Vitousek et al., 1997), although this area encompasses only 20% of Earth’s landmass. Thirty percent of all land within 100 kilometers of the coast has been at least moderately altered by human use (World Resources Institute, 2000). For many important types of coastal ecosystems, such as seagrass beds, there simply aren’t sufficient data to know the extent of loss, although in some highly developed areas loss approaches 90% (Burke et al., 2001). Destruction of wetlands reduces nursery grounds for commercially important species such as oysters and salmon, decreases filtration of input from terrestrial sources, and increases coastal erosion and flooding. Levees starve beaches and wetlands.

TOURISM

While people who spend time in particular habitats are more likely to be invested in preserving those habitats, even the most environmentally-conscious visitors are not without effect. For instance, a coralline alga-dominated intertidal community in New Zealand showed a 50% decline in meiofaunal biomass after only five days of trampling (Brown and Taylor, 1999). Negative affects of trampling have also been documented on rocky shores in Europe (e.g. Fletcher and Frid, 1996) and North America (e.g. Brosnan and Crumrine, 1994), as well as in salt marshes (Chandrasekara and Frid, 1996), although a study in South Africa found no long-term effects of trampling (Bally and Griffiths, 1989). Although snorkeling and scuba diving are not as common in temperate marine systems as they are in tropical systems, they do occur, and can cause noticeable effects.

Tourism has numerous effects on the physical environment as well. Increased use of a coastal area generally increases nutrient input, particularly when high-level sewage treatment is not available. In some cases, beaches are created artificially to encourage tourism, and beachfront areas are altered to become more “attractive” to tourists. Sunscreens, bug repellants, and other topically applied chemicals may wash off into the seawater, accumulating to noticeable levels in highly-used areas.

NOISE AND LIGHT POLLUTION

Although probably less important than many other anthropogenic influences, both increases in noise and light can have negative effects on many marine animals. The presence of strong light from shore has been implicated in the migration of sea turtle hatchlings away from rather than towards the ocean (Witherington and Martin, 1996), and light pollution may also cause disturbed vertical or horizontal migration by squid and zooplankters.

For whales that depend on sonar for migration and communication, noise pollution, particularly sonar, may cause a number of effects, including death, deafness, stranding, and altered behavior (National Research Council, 2003). Excess noise may also make it hard for juvenile fish to find their natal reef (Graydon, 2003). Although the use of sound as a location cue has not been documented for temperate marine organisms other than cetaceans, it may have an effect.

CHEMICAL POLLUTION

Ideally, marine reserves can be located so as to reduce the possibility of chemical pollution. If a pollutant is fairly non-dispersive and has a known source, it is possible to design reserves to avoid the source, or to contain the pollutant in case of an unanticipated spill. In contrast, highly dispersive pollutants are difficult to contain, and pollutants coming from multiple small or mobile sources are difficult to avoid.

OIL SPILLS

One of the best-studied oil spills was the 1989 Exxon Valdez Spill in Prince William Sound, Alaska, USA. Monitoring of long-term affects of this spill shows not only that spills have long-lasting effects, but that recovery rates vary tremendously among habitats. Although most kelp forest taxa recovered within two years, over 80% of subtidal eelgrass taxa had not recovered even six years after the spill, despite a relatively rapid recovery of the eelgrass itself (Dean and Jewitt, 2001). Similarly, Driskell et al. (2001) found that even seven years after the spill, rocky intertidal community structure had still not returned to normal. Two studies of recovery in soft-sediment communities found only incomplete recovery ten years after the spill.

Invertebrate taxa also vary strongly in their sensitivity to oil spills. A fine-sand amphipod community took over ten years to recover from the Amoco Cadiz oil spill off the coast of Brittany, France (Dauvin, 1998), while polychaetes in the same region seemed to be relatively unaffected (Gesteira and Dauvin, 2000). Such differential sensitivity would likely lead to shifts in population structure following oil spills, as was indeed seen in waters off Cape Cod, Massachusetts, USA following a 1969 oil spill (Sanders, 1977).

The decreased mixing of marine waters predicted to accompany global warming in many regions may alter the effects of oil spills, increasing residence time on the surface and slowing biological degradation. While higher temperatures, which could increase biodegradation rates, may counteract this effect, the overall effect of increased temperatures on ecological effects of oil spills is unclear. For enzymatic repair processes, increased temperature may speed recovery, up to a point (e.g. Hoffman et al., 2003); conversely, increased temperature may be a stressor, and combine synergistically with other stressors (e.g. coral bleaching, Fitt and Warner, 1995).

POINT SOURCE POLLUTION

Point source pollution is pollution coming from an identifiable source such as a factory outfall or a stream running through a heavily used agricultural area. While point sources often produce large quantities of pollutants, they are relatively easy to pinpoint and in many countries are regulated at regional or national levels. Traditional point sources have been industrial, dumping large quantities of heavy metals and polycyclic aromatic hydrocarbons (PAHs) into marine waters. More recently, coastal aquaculture facilities have become a more wide-spread problem, draining excess antibiotics and fertilizer into nearby waters. Another fairly recent type of point-source pollution is desalination plants. As fresh water becomes an increasingly valuable commodity, it is likely that more and more desalination plants will be built to serve the needs of coastal cities. The environmental effects of the hypersaline outflow from such facilities are as yet poorly understood.

NON-POINT SOURCE POLLUTION

In heavily populated areas, non-point sources can be as or more important than point sources in polluting marine ecosystems. Antifouling paint and oil and gas leaks from private boats can be problematic, as can the combined pesticides and fertilizers used by individuals on their lawns and gardens. Reducing non-point source pollution is difficult, and requires a concerted community education effort, as well as regulations governing the sale and use of toxic chemicals.

NUTRIENT POLLUTION

Excess nutrients entering the water due to fertilizer runoff, untreated human and animal waste, or sewage outfalls leads to eutrophication, oxygen depletion and the creation of “dead zones” as has been documented in a number of locations worldwide. The Gulf of Mexico is perhaps the most famous, with fertilizer from the Mississippi Basin leading to the annual formation of an anoxic area encompassing thousands of square kilometers (Malakoff, 1998). Nutrient over-enrichment can also cause harmful algal blooms, which injure or kill marine animals including fish, birds, and mammals (Conley et al., 2002; Anderson, 1997). Increased water temperatures and the resultant stratification that may accompany climate change make the occurrence of anoxic zones more likely, and may worsen existing zones. Ironically, it has also been suggested that increases in nutrients from agricultural runoff and sewage have allowed the recovery of fisheries that had been starved of nutrients. This has been studied off the coast of Egypt, where the Aswan Dam virtually stopped the input of terrestrial nutrients from the Nile (Nixon, 2003).

SEDIMENTATION

Both increases and decreases in terrestrial sediment input have been documented as problems in marine ecosystems. The construction of bulkheads and seawalls actually increases shoreline erosion and results in “starved” beaches, where a decline in the amount of incoming sediment causes an increase in substrate coarseness as well as increasing erosion rates. Damming major rivers also radically reduces sediment as well as water input into coastal areas. Conversely, poor land use practices and the clearing of soil-stabilizing vegetation in river catchments have lead to increases in sedimentation in other areas. Increased sedimentation can be particularly problematic for seagrass beds.

THERMAL POLLUTION

The use of seawater to cool power plants may cause significant alterations in local thermal regimes. In Florida (USA), some manatees now congregate around thermal effluent rather than natural heat sources such as thermal vents and artesian springs (Packard et al., 1989). Warm water outflow has also caused alteration in benthic communities in California (USA) near the Diablo Canyon nuclear power plant (Tissot et al., 1991). These later studies are particularly interesting in that community-level changes did not reflect predictions based on individual-level thermal sensitivities.

Observed and Predicted Effects of Climate Change on Temperate Marine Ecosystems

The all-pervasive nature of climate change—its effects are likely to be felt globally and throughout the depths of the ocean—demands that its effects be considered in reserve design and planning. Changes in climate are not a new phenomenon. Paleoecological records provide insight into effects of past cycles of global warming and cooling, and cyclical changes in climate regime in certain regions have allowed us to look at short-term (year-long to decades-long) changes in ecosystems. This information, combined with experimental work on individual organisms and communities and mathematical models, provides the basis for most predictions about future effects of global climate change. In making such extrapolations, however, there are several caveats. The causes of global climate change are not the same as the causes of regional regime shifts, and we should not assume that the effects will be the same either. Also, small-scale experiments or observations don’t capture the complex interactions characteristics of marine ecosystems. Trophic cascades such as those described by Estes and Palmisano (1974) are one example of unanticipated chains of events; there are likely many other such unrecognized connections that will determine the ultimate effects of climate change.

PHYSICAL CHANGES

Seascapes generally classified based on factors such as temperature regime, hours of sunlight, extent of ice cover, etc. While some of these factors (hours of sunlight) will not be significantly affected by climate change, others will, creating new combination of physical variables and thus new seascapes.

TEMPERATURE CHANGES

While the global mean temperature will rise as a result of climate change, it is possible that in certain areas, average sea temperature may drop as a result of changing current and upwelling regimes. Also, it is unclear for many organisms what aspects of temperature change matter most: maximum, minimum, or mean temperature, or rate of change and thermal history, measured daily, seasonally, or annually.

RISING SEA LEVEL

As global temperatures increase, sea level is expected to rise both as a result of thermal expansion of sea water, and as a result of melting glaciers and ice caps. In some areas, temperature-related sea level increase is exacerbated by decreased rates of soil-building in wetlands due to decreased sediment input either because of drought or shoreline modification. Some salt marshes won't be able to keep up with sea level rise (Moorhead & Brinson 1995). Even if they can, landward shoreline armoring may interrupt their natural advance upshore. Some coastal areas are sinking because of decreases in sediment input without any reduction in erosion. Areas such as the Rhône, Po, and Ebro deltas in the Mediterranean that are subsiding are further threatened by sediment starvation.

Changes in sea level may also affect the type of coastal system present in any given region. Recent research suggests that sea level rise over the 18,500 years has changed the Southern California Bight from a productive, diverse rocky coastline to a less productive, less diverse sandy coastline (Graham et al., 2003).

Changes in sea level have also been implicated in paleontological patterns of diversification and extinction in a number of marine taxa (e.g. House, 1989). In general, shallow seas are the most productive marine areas; increasing sea level increases the extent of shallow seas, and may increase overall marine productivity. Falling sea level, which reduces the area available for shallow-seas organisms, is associated with several mass extinction events.

INCREASING ULTRAVIOLET RADIATION

In marine organisms, UV causes death and deformity, alters behavior, sex ratio, and development, and can change community composition (reviewed in Hansen and Hoffman, 2001; Häder et al., 1998). Despite the common misconception that ultraviolet (UV) does not penetrate deeply enough into marine waters to play a significant role in marine ecosystems, both UVA and UVB cause measurable effects to down to several meters depth (e.g. Karentz and Lutze, 1990). In the clearest tropical waters, levels of UVA detectable by deep-sea shrimp can penetrate to 500-600 m (Frank and Widder, 1996).

UV levels in marine ecosystems are changing due to stratospheric ozone depletion, alterations in solar luminosity, and changes in water clarity due to a number of influences.

There is still only a rudimentary understanding of how elements of global change such as altered precipitation will affect UV levels in marine ecosystems, although the changes in sedimentation rates and primary productivity discussed above will certainly influence the depth to which UV penetrates. While progress is being made in reducing the production of ozone-depleting chemicals, the same cannot be said for other factors influencing UV levels in marine environments. Additionally, changes in vertical water circulation will affect the length of time planktonic organisms spend in the high-UV surface waters.

ALTERED NUMBERS/INTENSITY OF STORM EVENTS

Although increased severity and frequency of storm events has been predicted to result from global climate change (Houghton et al., 2001), data at this point are inconclusive. Wave heights have increased globally over the past 30 years , and higher sea levels will increase the height of storm waves and surges, increasing the frequency of extreme events. Storms may also have increased precipitation intensity; this would increase both erosion and salinity stress for coastal marine ecosystems.

Interannual climate fluctuations such as the El Niño/Southern Oscillation (ENSO) or the Interdecadal Pacific Oscillation, which strongly influence number and intensity of storms, may themselves be affected by global climate change (Timmermann et al., 1999). Although it is clear that such periodic climate oscillations have varied in frequency and intensity in the past, the causes of such variation is uncertain (Tudhope et al., 2001; Urban et al., 2000)

WATER CHEMISTRY

The decreased freshwater input and increased evaporation expected to accompany global warming trends may lead to increased salinity in small or shallow semi-enclosed bodies of water such as tide-pools and estuaries. The implications of such changes are unclear, as many tide-pool and estuarine organisms are adapted to alterations in salinity. This does not mean, however, that they will be able to compensate for any future changes.

Increased CO_2 concentrations may lead to a gradual decrease in the ocean's pH and calcium carbonate concentration. Studies have shown that calcification rates of many marine organisms, including corals, coralline algae, and coccolithophorids, are decreased by increases in CO_2 (reviewed in Gattuso and Buddemeier, 2000).

Additionally, there will likely be alterations in seasonal levels of freshwater input: more rain rather than snow in winter may lead to more runoff in winter and less in summer. For animals such as herring that respond to drops in salinity as a spawning cue, altered timing and amount of freshwater influx could have profound effects.

UPWELLING AND CURRENTS

Numerous changes in water movement are expected to occur as a result of increased temperature. These include increased stratification and stability of the water column due to surface water warming, and consequently reduced upwelling, both near-shore and open-ocean (summarized in Soto, 2002; Field et al., 2001). Decreased upwelling in turn decreases nutrient input to surface waters, leading to a decline in primary productivity. Effects of decreased phytoplankton reverberate through the food web, leading to reduction in density of zooplankton and likely secondary consumers as well. Such changes have been documented off the coast of Southern California following an abrupt and sustained increase in water temperature in the 1970s (reviewed in Field et al., 1999).

Conversely, climate change may alter wind patterns in ways that increase offshore winds and thus upwelling (Bakun, 1990). Accurately predicting local changes in upwelling regimes, then, requires understanding the relative strength of changes in offshore winds vs. surface warming.

There may also be altered current regimes, such as reduction in area and a coastward migration of shelf-tidal fronts, a reduction in strength of the Gulf Stream, and an increase in strength of the Labrador Current (Soto, 2002). Such large-scale changes in currents would result in changes in larval transport, affecting both how many larvae recruit to adult populations, and where the larvae end up.

ALTERED RATES OF SEDIMENT INPUT

Changes in amount and timing of rainfall will affect input of sediment as well as of fresh water. Winter rainstorms have the potential to wash greater amounts of sediment into rivers, since in many regions there is less vegetation to hold soil in place in winter than in summer. Increases in sedimentation will also occur in regions experiencing decreasing sea ice cover, as land is exposed to the erosional forces of waves for longer parts of the year.

On the other hand, sediment input may decrease for a number of reasons. Increasing coastal development is leading to increasing hardening of coastlines. Seawalls, bulkheads, and riprap all decrease the amount of sediment reaching adjacent coastal areas, resulting in sediment-starved beaches. Reduced sediment input resulting from the damming of rivers will be exacerbated by the drier conditions expected in much of the world. Changes in grain size and character of sediment-starved beaches can lead to dramatic changes in community composition and structure.

Climate change may increase the problem of human alteration of shorelines. Coastal homeowners wary of increased numbers or intensity of storms and rising sea level may build more and bigger bulkheads, further starving shorelines in developed areas. This negative feedback cycle can already be seen in highly erosional coastlines like the Wadden Sea.

BIOLOGICAL CHANGES

EFFECTS ON INDIVIDUALS

Temperature can affect individual organisms both behaviorally and physiologically, with effects ranging from mild to lethal. Not surprising, then, is the range of adaptations and sensitivities organisms display to both absolute temperature and to changes in temperature (Hochachka and Somero, 2002). Animal species inhabiting the high intertidal, for instance, can generally survive significantly higher temperatures than their low intertidal congeners, and susceptibility to thermal stress can occur on multiple levels (reviewed in Somero, 2002).

Temperature also affects growth rate and development, potentially affecting generation times or juvenile size. In over 90% of aquatic ectotherms studied, increased rearing temperature resulted in decreased organismal size (Atkinson, 1995). While average individual size does decrease with latitude in many invertebrate species, it is unclear whether changes in ocean temperature directly affect individual size.

POPULATION-LEVEL EFFECTS

For many species, temperature serves as a cue for reproduction (Olive, 1995). If species synchronize their reproduction with such environmental factors as food availability using cues sensitive to global climate change, such as temperature, climate change may disrupt this synchrony and thus alter developmental success (Olive, 1995). Bhaud et al. (1995) describe the disruptive effects of even short-term temperature fluctuations on the reproduction of some marine polychaetes.

Temperature can also affect reproduction by altering sex ratios. Temperature-dependent determination of sex occurs in all sea turtles (Davenport, 1997) as well as some copepods (Voordouw and Anholt, 2002) and fish (Pavlidis et al., 2000; Goto et al., 1999; Conover, 1984). Changing climate cycles will potentially skew sex ratios to the extent that population growth is threatened.

EFFECTS ON COMMUNITIES

Within any community, it is likely that different members of the community will respond differently to various elements of global climate change depending on their sensitivity to and tolerance for various aspects of environmental change. Changes in temperature, sedimentation rate, ultraviolet radiation, and disturbance regime can shift competitive interactions, favoring some species over others. Species with high genetic diversity and short generation times may be able to undergo evolutionary adaptation, while genetically depauperate species are less likely to do so. Mobile species may track preferred temperatures, while less mobile species must either adapt or disappear locally. For instance, northward range expansions following warming trends have been documented for sardine populations off the coast of California (Lluch-Belda et al., 1992). On a rocky shoreline in central California, where annual mean ocean temperature has increased 1 °C over the past 60 years, researchers have documented an increase in southern animal species and a decrease in northern animal species (Sagarin et al., 1999). Likewise, a study of kelp forest fish in Southern California found that since the 1970s there had been a 50% drop in cold-water, northern species, and a 150% increase in warm-water, southern species (Holbrook et al., 1997).

Predicting changes in communities in response to climate change may not be as simple as understanding species thermal biology, however. Fox and Morin (2001), for instance, found that protist populations with density-dependent regulation were relatively immune to effects of temperature change, while Davis et al. (1998) found that interspecific interactions made effects of temperature changes difficult to predict for three species of fruit fly. For intertidal barnacles, interspecific interactions shifted from competitive to facilitative as environmental conditions became harsher (Bertness et al., 1999).

Global warming could also alter the strength of interaction between keystone predators and their prey, potentially causing strong shifts in intertidal communities by affecting feeding by a keystone species (Sanford, 1999). On the coast of Oregon, decreasing temperature slowed the feeding of two key rocky intertidal predators by around 30% (Sanford, 2002). Thus the timing and number of upwelling events determined the reproductive output of some intertidal prey species.

BIOGEOGRAPHIC EFFECTS

The role of temperature and desiccation in determining zonation patterns in rocky intertidal habitats is well established. In general, biotic interactions such as competition and predation set a species' lower boundary, and physical stress sets the upper boundary. Species thus exist at tidal heights where predators are scarce or absent, but abiotic conditions fall within survivable ranges. A study in Washington State found that intertidal bands of mussels and barnacles were broader on exposed than sheltered coastlines; in less wave-exposed locations, temperature stress lowered the upper limit for these species, but predation by a sea star, which sets their lower limit, remained unchanged (C. Harley, pers. comm.). If climate change lowers the upper limit of barnacles and mussels in all locations, it will likely lead to an overall decrease in the populations of these animals, since expanding downward will not be an option where predatory sea stars are present. A similar interaction between thermal stress and predation was found to exclude one species of alga from south-facing slopes, where high temperatures during emersion pushed their upper limit down into the zone of high predation (Harley, 2003). The lesson for climate change is that this alga may go locally extinct because its upper limit will shift down while its lower limit remains stationary.

On a larger scale, species ranges may shift poleward in response to global warming. Such a shift may be dictated by a species' need to stay within a particular water temperature range, or, in the case of intertidal organisms, to stay within an acceptable range of air temperatures (Lubchenco et al., 1993). Such range shifts appear to be occurring in some marine environments (Sagarin et al., 1999).

NON-NATIVE SPECIES

Climate change is likely to affect the relative success of biological invaders. Stachowicz et al. (2002), for instance, found that warmer temperatures favored non-native over native ascidians. In some cases, marine animals have been imported to new environments with the understanding that colder temperatures in the new location would limit the exotic's ability to reproduce (e.g. Pacific oysters). As temperatures increase and limits to reproduction decline, such species will spread more readily throughout their new home. Alterations in currents may bring larvae from populations of introduced species to new locations, and disruption of native, natural ecosystems by climate change may render them more vulnerable to successful invasion by non-natives. If stocks of native marine organisms become too unstable or depleted, there may be an increase in aquaculture of non-natives species, and a consequent increased risk of invasion/dominance by these species.

DISEASES

Climate change has been linked with increases in marine disease outbreaks (Harvell et al., 1999; Hofmann et al., 1999) and range extensions in marine parasites (Ford, 1996). Warming trends associated with ENSO cycles correlate with shellfish diseases on the US Atlantic and Gulf coasts (McLean and Tsyban, 2001). Increased outbreaks may be due to increased pathogen transmission or decreased host resistance. Decreased resistance may result from behavioral or physiological changes in marine populations, and may also be affected by interactive effects with pollutants. Many new diseases are caused by host shifts rather than the appearance of previously unknown organisms (Harvell et al., 1999); as mentioned previously, increased in aquaculture may also increase the global transmission of diseases. Indeed, transfer of infected stocks may have been responsible for some of the most devastating disease outbreaks in farmed bivalves (discussed in Harvell et al., 1999)

Considerations and Suggestions for Marine Reserve Design With a Focus on Possible Adaptation Options

A number of criteria have been proposed for evaluating potential marine reserve sites, including ecological, social, and economic considerations (see Roberts et al., 2003 for ecological criteria, and refs therein for other approaches). Although it is clear that the biophysical characteristics of temperate marine ecosystems will change, uncertainty about the nature and rate of change is compounded by our lack of knowledge about how complex ecological interactions will respond to change. The UNFCCC suggests the possibility of natural ecosystem adaptation to climate change, if we limit the rate of change sufficiently. Also key to successful adaptation is looking at interactions between climate change and other anthropogenic stressors on marine systems: which factors will be exacerbated by climate change and which will be ameliorated?

SPATIAL CONSIDERATIONS

CREATE NETWORKS OF RESERVES

It has been suggested that fewer large reserve might be better for protecting biodiversity while numerous smaller reserves might be more effective for fisheries enhancement; given the range of dispersal distances of marine larvae, however, many biologists now believe that networks of marine reserves will be more effective than single large reserves (e.g. Hastings and Botsford, 2003). Networks allow a greater diversity of habitats to be covered over a greater geographical range for the same amount of total area protected. Increasing the geographical range included in a network will become increasingly important if the anticipated latitudinal species migrations occur in response to climate change. Although there are few empirical data on size and spacing of reserves within networks, these issues have been the focus of recent theoretical work. Based on a review of published dispersal distances, Shanks et al. (2003) suggested that reserves of 4-6 km2 would be sufficient to retain short-distance dispersers. Roberts et al. (2001) noted that even reserves as small as 5 km^2 have been shown to be effective in increasing densities of commercially valuable species. Spacing between reserves and location of reserves within the network area is less clear. Long-distance dispersers may travel between 20 and 4400 km, and remain in the plankton for almost 300 days (Shanks et al., 2003). Their exact route is unknown. As discussed previously, the common assumption has been that planktonic marine larvae by and large follow ocean currents, but at least some larvae have behavioral mechanisms to aid in local retention.

DESIGN RESERVES TO PROVIDE TEMPERATURE REFUGIA

One "natural adaptation" to climate change is for species ranges to shift to higher latitudes. Such latitudinal changes have been demonstrated for numerous terrestrial species (Parmesan and Yohe, 2003), and for marine species on the west coast of North America during a recent warming trend there (reviewed in Field et al., 1999). To the extent that this migration allows the preservation of species that would otherwise go extinct, marine reserves should attempt to provide robust corridors for such poleward movement. Designing latitudinally extensive networks serves this purpose. Many species may not be able to migrate fast enough to keep pace with climate change, however (Malcolm and Markham, 2000).

It may also be possible for some populations to shift their habitat into deeper, cooler water as global warming progresses. As mentioned previously, however, downward range expansions may be prohibited for some species by the presence of competitive dominants or predators, and photosynthetic organisms are limited by the depth of the photic zone. Also, in a study of rocky intertidal community response to warming, Sagarin et al. (1999) found no overall downward shift in tidal range, although there was evidence of a latitudinal shift.

While shifting habitats to greater depth and higher latitudes is one way of finding cooler temperatures, most regions have identifiable "hot spots" and "cold spots" (see, e.g., Helmuth et al. 2002) due to factors such as upwelling, shade, subhabitats, timing of tides, and freshwater input. Designing reserves to include "cold spots" may reduce thermal stress from climate change. Areas with a high degree of vertical mixing or rapid water movement, for instance, have lower average temperatures than stratified areas. Such areas may also be higher in nutrients and thus provide a more stable food supply for zooplankton and planktivores. Areas of shoreline with ample shading and cooler microhabitats (cracks, crevices, etc.) can also provide temperature refugia and increase survivorship (Moran and Emlet, 2001).

ESTABLISH RESERVES IN TRANSITIONAL ZONES BETWEEN BIOGEOGRAPHIC REGIONS AS WELL AS IN CORE AREAS

Species are often assumed to be more successful and stable in the core region of their geographic distribution, and conditions are assumed to become increasingly stressful towards the edges of the species’ range. Thus, in the absence of climate change, it has been assumed that to preserve a species, it may be preferable to locate reserves away from range edges. As climate changes, however, the “best” area for a species may shift away from what had been the core of its range. Thus creating reserves throughout a species’ range, or at least including locations near its poleward boundaries, is a safer approach.

Additionally, this “abundant center” concept of species distribution has recently been challenged. In nine of 12 temperate marine invertebrate species studied by Sagarin and Gaines (2002), areas of peak density occurred in sites near the edges of the species’ range as well as in the center. These findings underline the importance of designing spatially heterogeneous networks of reserves.

ADD AN “INSURANCE FACTOR” TO RESERVE SIZE CALCULATIONS

To buffer against periodic severe disturbances such as hurricanes or oil spills, researchers have suggested adding an “insurance factor” when calculating the necessary size of a reserve (Allison et al., 2003). The insurance factor is simply extra area added to the reserve, and should assure that a reserve’s function goals are met despite catastrophes. The amount of extra area needed as insurance can be calculated based on the anticipated frequency of and vulnerability to disturbances in the region covered by the reserve or reserve network. Such an approach could also be effective in buffering against possible effects of climate change, and would work well in conjunction with other bet-hedging approaches. This approach may be ineffective in protecting many spatially restricted habitats and ecosystems, however.

INCLUDE A DIVERSITY OF HABITATS AND BIOGEOGRAPHIC REGIONS

Traditional reserve planning has been focused on single-species conservation, and as such has focused on the selection of critical habitat areas for the species of interest. While such considerations should continue to be included in planning, particularly for reserves whose aim is increased yield of commercially valuable species, there is a growing recognition that to be successful in the long term, reserves must be designed more broadly. Rare or vulnerable habitats should be included in reserves even if they have no known connection to species of interest; such habitats may play ecological roles we are unaware of, and would be hard to restore/replace. Also, such habitats contribute to the overall habitat heterogeneity, another key element of reserve design. Habitat heterogeneity, or the presence of multiple habitat types, has at least two potential benefits. First, habitat diversity generally corresponds with overall biodiversity, since a heterogeneous habitat provides more microhabitats and opportunities for specialization. Second, preserving multiple habitat types within a reserve or a network of reserves may benefit species that live in different habitats during different stages of their life. From the climate change perspective, protecting multiple habitat types increases the odds that at least some of the protected habitats will be resistant or resilient to the effects of climate change.

Including a diversity of habitats in marine reserve networks is also important for species that use different habitats at different times of year or during different life stages.

On a somewhat larger scale, reserve design should also maximize biogeographic representation. Although there is as yet no universally accepted set of biogeographic classifications for marine ecosystems, several such classifications have been proposed. Roff et al. (2003) present a table of possible factors to consider in designating marine biogeographic regions, and discuss local variation in the relative importance of these factors.

DESIGN RESERVES BASED ON FEATURES OF THE ENVIRONMENT UNLIKELY TO CHANGE

While which organisms will inhabit any stretch of shoreline decades in the future may be unpredictable, there are some elements of existing ecosystems whose persistence is more likely. Day and Roff (2000) termed these "enduring and recurrent features." Physiographic features such as topography will persist regardless of climate change, and can influence the biological communities that develop in any given area. Areas of high relief, for instance, are usually associated with greater species diversity and richness than areas of low relief. This may be in part because high relief provides a greater diversity of habitat, and in part because high relief also creates upwelling, which increases the supply of nutrients and oxygen, and thus productivity. Currents, waves and mixing related to fetch and tidal forces are also unlikely to change; as mentioned above, areas with strong currents and stronger wave action may experience less warming than calmer waters.

Certain oceanographic features are also enduring or recurrent. Convergence and divergence zones occurring in the open ocean, for instance, have fairly predictable locations in temperate waters of the southern hemisphere, although their location is not predictable in northern temperate waters. Oceanographic features may be more affected by climate change than physiographic features, however. For instance, upwelling is a fairly regular occurrence on the Pacific coast of the Americas, but increased frequency or strength of ENSOs might decrease the amount of upwelling significantly. The potential decrease of climate-driven upwelling underscores the importance of including topography-driven upwelling zones in reserves. Local topography and current patterns may also determine areas of higher or lower larval settlement (Shanks and Wright, 1987).

DESIGN RESERVES AROUND AREAS THAT SHOW RESISTANCE AND RESILIENCE TO CLIMATE CHANGE

Researchers and planners concerned with coral reef conservation have begun to identify locations that are resistant to bleaching, and factors that allow for more rapid recovery from bleaching (resilience) (West and Salm, 2003). If temperate events equivalent to bleaching in terms of sensitivity to climate change can be identified, they may assist in developing a similar approach for temperate marine ecosystems. For instance, if particular species are found to be most sensitive to the effects of climate change, areas where those species continue to thrive may be areas with natural resistance to climate change, either because of cooler conditions or some other combination of factors. Similarly, habitats or locations that recover quickly from more short-term changes in climate such as the Pacific Decadal Oscillation may be best able to cope with longer-term climate change. By identifying local areas with increased resistance or resilience, reserve planners can enhance the capacity of particular communities to persist (West and Salm, 2003).

SOCIOPOLITICAL CONSIDERATIONS

USE ADAPTIVE MANAGEMENT STRATEGIES

Because we have relatively little hard data on how well marine reserves work, and because the factors influencing reserve success may change over time, many scientists and practitioners are recommending the use of adaptive management strategies (e.g. Salafsky et al., 2002; Botsford et al., 2003; Parma et al., 1998). This approach requires practitioners to monitor the success of reserves and the on-going effects of design and management choices, and to adjust management practices in response to this information. This will allow the evolution of the most appropriate management practices for each reserve or network of reserves, and begin to provide much-needed information to aide in the design of future reserves. Such an adaptive, iterative approach is particularly important in creating reserves that will be most robust or resilient relative to climate change: there is so much uncertainty in our knowledge of the physical and ecological effects of climate change that future flexibility is essential.

Adaptive management is also important, as it allows reserve managers to respond to changes not only in the ecology of reserves, but in the sociopolitical context of reserves. It is likely that climate change will affect the relationship of local communities to the marine environment; such changes could significantly affect the effectiveness of reserves. For instance, the size needed for reserve effectiveness depends on fishing pressure outside the reserve; if climate change alters fishing pressure, the effectiveness of the reserve would likewise change.

ESTABLISH A TEMPORALLY RESPONSIVE APPROACH THAT ALLOWS YOU TO TRACK COMMUNITIES/SPECIES OF INTEREST AS CLIMATE CHANGES

Setting up a successful reserve in the first place is challenging; the idea of adjusting the boundaries of and restrictions in reserves over time may seem like it simply prolongs the struggle. However, the potential usefulness of such temporally-responsive reserve management is such that the idea is worth at least considering. In conversations with stakeholders and other planners, bring up the idea that to fulfill the management goals identified by the group, reserve boundaries may have to change in the future, just as many ranchers rotate their animals among different fields. If members of the community understand the importance and potential benefits of reserves they may be willing to do what it takes to ensure reserve success in the long term. Also, the periodic reassessments of reserve locations and activities required by this approach provide an opportunity to reengage community members and strengthen their commitment to the reserve.

INVOLVE STAKEHOLDERS FROM THE BEGINNING

While most discussions of reserve design focus on where to site reserves and what activities to allow, there is increasing awareness that committed and invested community stakeholders can be as essential to the long-term success of a reserve as appropriate selection of location and ecological characteristics (Salm and Clark, 2000; Kelleher, 1999). Given the degree of uncertainty involved in planning reserves in the face of climate change, the commitment of local communities to the ongoing success and effective management of reserves gains even greater importance. If people living and working near reserves feel responsible for the reserve, they are less likely to engage in destructive activities, more likely to police themselves and other community members, and more likely to support changes in reserve location or restrictions as necessitated by climate change or other factors.

To maximize community investment in a reserve, community involvement and empowerment must be part of even early planning. If people feel that they have been excluded at any stage of planning, reserve managers may start to look like adversaries rather than colleagues. Also important is taking the time to understand how local communities relate to marine resources, which community members use or control various aspects of the marine environment. This requires sensitivity to both culture and gender issues.

IDENTIFY CLEAR GOALS AND CRITERIA FOR MEASURING SUCCESS

In the past, reserve and conservation efforts have often focused on "species of interest," either culturally or commercially important or endangered species. It has been suggested (e.g. Simberloff, 1997; Franklin, 1993) that a single-species focus leads to a myopic approach in which overall ecosystem stability and function may be sacrificed; additionally, cryptic or unknown species will remain unprotected. An ecosystem approach, in contrast, may be more likely to support the species of interest.

LIFE HISTORY CONSIDERATIONS

Because marine populations are much more open than terrestrial ones, and marine organisms often have complex life cycles with two or more ecologically distinct life history phases, the appropriate size and location of reserves is tricky to determine. The degree of larval and juvenile dispersal can radically affect both the ideal size of a reserve and its ability to fulfill different goals (i.e. preserving biodiversity vs. enhancing fisheries; Botsford et al., 2003), and adult, larval, and juvenile stages must be considered for reserves to succeed. For species with distinct feeding and spawning grounds, both types of habitat must be preserved. Unfortunately, the full life history is understood for relatively few species, and may be quite different even for closely related species (Grantham et al., 2003; Strathmann, 1987). While larvae of some species recruit locally, in other cases adult populations may depend largely on larvae arriving from upcurrent locations. A decline in a particular adult population may reflect local problems for those adults, problems in the larval habitat, or problems with an upcurrent adult population that is a significant source of recruits.

Because knowledge of larval dispersal is spotty but dispersal distance seems critical to reserve sustainability (Botsford et al., 2003), the best approach may be to design reserve networks using a "bet-hedging" strategy. As mentioned earlier, Shanks et al. (2003) suggest making reserves large enough to sustain locally-recruiting populations (4-6 km in diameter), and spacing reserves such that larvae with longer dispersal distances can recruit to adjacent reserves (10-20 km spacing). Reserve placement also needs to reflect both major ocean currents and smaller currents and eddies, since the extent to which larger- and smaller-scale currents matter for larval dispersal and recruitment is unclear. Different habitats seem to have different frequencies of dispersing vs. non-dispersing larvae; a recent study on the west coast of the United States found that over 60% of sandy intertidal species had non-planktonic larvae, as compared to less than 22% of species in rocky intertidal and sandy subtidal habitats (Grantham et al., 2003).

Despite the ability of planktonic larvae to travel hundreds of kilometers, it appears that larvae of at least some species have mechanisms to enhance larval retention. These may reflect a complex balance of larval behavior, currents, salinity, and even sound. By assessing the existence of such mechanisms for populations of interest, reserves can be designed to give the best chance of larval retention. For instance, some larval fish use sound to locate appropriate adult habitats (Leis and Carson-Ewart, 2003, and refs therein). In this case, restricting use of motorized craft or other sources of extraneous noise might help increase or maintain larval recruitment.

In designing reserves to benefit particular species, it is essential to first identify the most vulnerable life stages of that species. Often the stage that is most visible, well understood, or accessible receives the most attention, with no consideration of which life stage is most in need of protection, or whose protection would do the most to boost adult populations.

CONTROLLING NON-CLIMATOLOGICAL STRESSES

Coastal ecosystems are intricately linked with the land they adjoin. Reserves should be located away from major sources of terrestrial pollution such as agricultural runoff, sewage outfall, or industrial outfall. While point sources of pollution are relatively easy to identify, non-point sources may be harder to avoid. In general, locating reserves away from heavily populated areas or areas that receive a lot of human use will help to reduce the negative impacts of non-point pollution. Locating reserves away from human use areas also reduces the risk of coastline alterations leading to sediment starvation of beaches, or conversely excessive sediment input due to poor land use practices and erosion. Ideally, marine reserves can be linked with terrestrial reserves, so that the major sources of terrestrial input (sediment, nutrients, fresh water) are preserved along with the marine ecosystem they serve.

Limiting anthropogenic stresses in marine reserves becomes even more critical when considering mitigation of climate change effects. Anthropogenic stressors can act synergistically with increasing temperature and other elements of climate change; minimizing non-climate stressors thus mitigates the negative effects of climate change as well. For instance, the recent increase in number and extent of harmful algal blooms (HABs) is thought to result from both warming and increased nutrient pollution (Mudie et al., 2002). Limiting excess nutrient input in areas near reserves would decrease the likelihood of devastating HABs affecting the reserve.

Identifying and Selecting Strategies for Your Particular Location

Given the variability of biophysical and social systems in and affecting marine ecosystems, there can be no single “right” way to create a reserve. Each reserve must be designed to best meet the needs and attitudes of local ecosystems and inhabitants. WWF has identified this location-based approach as a key to reserve success, along with the existence of stakeholder involvement, community enforcement, a clear management plan, legal protection, monitoring, financial sustainability, and sufficient numbers of well-trained personnel.

SOCIOPOLITICAL ISSUES OF RESERVE DESIGN

It is essential to identify the stakeholders in your region, their background knowledge and economic options, and what restrictions on use they would be willing to accept. As mentioned above, stakeholder input and support is a significant predictor of reserve success. Locally based, sustainability-driven resource users may have a different degree of commitment than profit-driven users, and the percent of users who are local and permanent (ostensibly having a more long-term view) vs. transient (tourists, big business) will have a significant impact on what strategy works best. Ideally all users may be convinced that the long-term sustainability of their enterprise, be it commercial or sustenance-oriented, is intricately linked with the long-term health of the marine ecosystem. Achieving stakeholder buy-in allows for a greater reliance on voluntary compliance and self-monitoring rather enforcement, which is often both more expensive and less effective. Given the financial uncertainty inherent in conservation planning, the more self-sustaining reserves can be, the better.

BIOPHYSICAL ASPECTS OF RESERVE DESIGN

Given the limited time and money available to planners, along with our still uncertain understanding of ecosystem response to climate change, most reserve planning will occur without what would be considered the ideal amount of information. Nonetheless, taking the time to inventory certain key aspects of the marine ecosystems in your region is essential.

First, map habitat types in your region, including location and area covered. The map should cover both structural elements of habitat (sand, kelp forests, seagrass beds, boulders, etc.) and functional elements of habitat (e.g. spawning grounds for particular species, nursery grounds, etc.). Identify key biotic and abiotic variables controlling species distribution in your area, and how they’ll be affected by climate change. For instance, what types of catastrophes does your region experience (how extreme, how frequent, how prolonged, rate of onset), and how will climate change likely change those? Is your region likely to experience sea temperature increases or decreases? Remember, there’s a great degree of local variability in the effects of climate change: just because warming will be the norm, doesn’t mean your area will warm up. What major current regimes in your area, if any, will likely be affected by climate change (e.g. ENSOs, NAO, PDO, NO)? Consider the tidal regime in your region. Areas in which low tides occur in the middle of the day will have more temperature stress on intertidal organisms (Helmuth et al. 2002), and will thus need different planning strategies than less-stressed areas.

Strategies for Implementation

"Best practice" approaches for planning reserves that will provide the best chance of success in the face of global climate change are not wildly different from "best practice" approaches for reserve planning as they exist now. Certain elements of reserve planning may gain increased importance, however, and are emphasized below.

REGIONAL AND TRANSBOUNDARY COOPERATION

Transboundary agreements and treaties, already important because of the open nature of marine ecosystems, may be essential to meet new spatial refugia or larval transport needs, or to reduce stressors other than climate change to acceptable levels. Also, given uncertainty about how communities and ecosystems may rearrange themselves as climate change progresses, the possible inclusion of temporally-sensitive management options may become increasingly important.

MONITORING

The science and practice of marine reserves are fairly young. There are not yet any time-tested design tools or best management practices, although there is wide-spread recognition that they must rapidly be developed if marine reserves are to fulfill their promise as conservation tools. Systematic monitoring, evaluation, and information-sharing will enable us to understand the factors essential for reserve success, and the particular challenges faced in variety of situations. The adaptive management approach suggested previously requires on-going input of information so that effectiveness can be iteratively evaluated and reserve management appropriately adapted as the environment changes and as more information becomes available. This approach enhances the probability that a reserve will achieve its long-term goals.

Enlisting the help of volunteers can be a good way to encourage community involvement and investment in the reserve. It also familiarizes you with the nature of your reserve, its cycles and processes. Also, monitoring lets you know whether or not your reserve is working. If your reserve isn't working, you need to redesign the reserve, alter restrictions on use, or establish another reserve. Which option you choose will depend on your best assessment of why the reserve isn't working. Further, more data are needed to help us design effective reserves in the future and assess the extent to which reserves can solve our conservation problems.

What to monitor depends on reserve goals, size, and budget. Some basic variables of interest are number, size, density and biomass of species inside, adjacent to, and far from the reserve; ideally, data should be collected before as well as after establishing the reserve to provide a baseline. Replicate counts inside the reserve are essential to ensure adequate and accurate sampling. Monitoring often focuses on vertebrates (Halpern

2003), but should be focused on overall biodiversity. This would provide more information for understanding ecosystem function and stability.

Existing Adaptation Programs and Some Needed Information

EXISTING PROGRAMS

There are few if any marine reserves that have been designed and monitored specifically with climate change adaptation in mind. Nonetheless, the need to incorporate potential effects of climate change in reserve design is gaining attention (e.g. Soto, 2002; MPA News, 2001), and suggestions for doing so are beginning to appear. Parks Canada, for instance, surveyed all national parks in Canada to identify sites and species most at risk from climate change, and predicting probable effects of climate change on each park (David Suzuki Foundation, 2000).

Australia has recently created the world's largest marine reserve, the Heard Island and McDonald Islands Marine Reserve in the Southern Ocean (Australian Antarctic Division, 2003). In addition to maintaining these subantarctic islands in a relatively pristine state, this reserve will be specifically monitored for possible effects of climate change in a region that has seen little direct anthropogenic impact.

INDICATOR SPECIES, COMMUNITIES, AND PROCESSES

As we head into an uncertain climate future, it would be useful to know which species, communities, or processes might serves as harbingers of changes to come. Identifying species most sensitive to climate change ("indicator species") would give us early warning signs of ecosystems on the brink of collapse, or early indications of successful conservation approaches. Identifying the most sensitive elements of ecosystems may also help us anticipate unexpected or indirect effects of climate change on ecosystems.

Species that live in variable environments (e.g. seasonal, intertidal) tend to have broader latitudinal ranges and occupy more biogeographic provinces than species from more stable environments. Species with broad geographic ranges also tend to be more long-lived on a geological scale than species with restricted ranges. Thus it might be expected that species naturally occurring in stable habitats or with geographically restricted ranges would be more susceptible to some aspects of climate change, and might make a good first guess as indicator species (Harley et al., in press).

Another broad-brush approach to predicting potential sensitivity of species to climate change centers on the prediction that mobile organisms, which can escape stressful situations behaviorally, should have a narrower tolerance range for a variety of stressors than sessile organisms, which must deal with whatever stressors come their way (Huey et al., 2002). Thus mobile organisms may prove more sensitive indicators of environmental change that sessile ones.

Since species do not live in a vacuum, it may also be useful to know which biological communities are most sensitive to climate change. Roberts et al. (2003) suggest that communities that depend on biological rather than physical matrices, such as eelgrass beds or mangrove swamps, are often the least resistant or resilient. Such communities may require particular attention in reserve design and sustainability.

LONG-TERM, LARGE-SCALE DATA SETS

While numerous models exist to predict the future of climate change and its effects on ecosystems, there is a strong need for real-world data to complement these models. Studies documenting effects of short-term climate changes such as ENSOs are an excellent first step, as are studies of the effects of thermal effluents on marine communities; both approaches should be expanded to include more geographic regions. Because such phenomena are much more short-lived and smaller-scale than global climate change, however, we must also accumulate data on community structure over broad geographic and long temporal scales.

Conclusions

When it comes to marine reserves and climate change, our level of knowledge is inadequate, and will remain so for the foreseeable future (see the excellent discussion of uncertainty in climate change modeling in Hannah, this volume). We cannot wait for certainty before taking action, however; we must make the best decisions we can based on our current understanding of marine ecosystems and climate change, and incorporate bet-hedging and flexibility into our planning.

The suggestions in this chapter are not in any way meant to undercut the importance of global efforts to halt global climate change, or to imply that immediate global reduction of greenhouse gas emissions is not critical. Since global climate is not under local control, however, and since some level of global climate change is now inevitable, local and regional planners need to take what actions they can to mitigate the effects of climate change. Also, reserves are a critical element of conservation planning, and failure to consider climate change when designing reserves may decrease the effectiveness of reserves dramatically.

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Increasing the Resistance and Resilience of Tropical Marine Ecosystems to Climate Change

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CHAPTER 6
Tropical Marine

TROPICAL MARINE ECOSYSTEMS ARE rich in biodiversity and provide productivity to support almost one billion people every year in Asia alone (Kaufman and Dayton, 1997). They include coral reefs, mangrove forests, seagrass beds and vast pelagic systems, and support local subsistence fishing as well as international commercial fisheries. Not only do coral reefs support biodiversity and provide sustenance for communities, but they also support extensive tourism activities around the world. Coral reefs and mangroves offer protection from coastal erosion, while seagrasses and mangroves both trap sediment from terrestrial run-off. Mangroves act as a filtration system for estuarine and fresh water. These habitats also serve as nurseries for many invertebrates and fish. The complex ecology of these systems is well appreciated but not fully understood. Efforts toward greater understanding are being constantly challenged as theses systems are altered by local anthropogenic stresses (fishing, development, extraction) and global climate change.

The resistance (ability to withstand change) and resilience (ability to recover from change) of an ecosystem determine how well it can deal with this barrage of challenges (Noss, 2001). Conservation efforts can enhance resistance and resilience to climate change by alleviating the overall pressures on the system, giving it more flexibility to mobilize its natural defenses. For example, coral reefs that experienced greater disturbance prior to a bleaching event tend to have a poor ability to recover (Brown, 1997a). The following chapter offers an introduction to the effects of climate change on tropical marine systems, as well as an overview of the types of strategies that might be adopted to increase the resistance and resilience of tropical marine protected areas. This chapter focuses mostly on coastal systems, touching on pelagic where information exists.

What Stresses Threaten Tropical Marine Systems Other Than Climate Change?

DEVELOPMENT

Tropical marine ecosystems are currently threatened by an array of local stresses associated with human activities. Although coastal regions encompass only 20% of Earth's landmass, almost 60% of the world's population lives within 100 kilometers of the coast (Vitousek et al., 1997). Growing human populations and the desirability of coastal property increases coastal development, resulting in the loss or degradation of coral reefs, mangroves and sea grasses. Conversion of reef habitat to land for development has a “complete and irreversible” effect on reefs, effectively destroying them. Such development is extensive in Egypt, the Seychelles, the Maldives, Singapore and some south Pacific atolls (Spalding et al., 2001; Bryant et al., 1998). Historically, 75% of all tropical coasts where inhabited by mangroves, but this is no longer the case (Farnsworth and Ellison, 1997). Accurate estimates of global mangrove loss are not available, but a recent survey characterized 55% of sites as “threatened” due to clear-cutting and reclamation (Farnsworth and Ellison, 1997). Seagrass loss is also poorly quantified, but at least 90,000 hectares were lost in the decade prior to 1996, with 45,000 hectares of that in Australia and 25,220 ha in the United States (Short and Wyllie-Echeverria, 1996). Losses were primarily due to dredge and fill activities and related changes in water quality. These numbers almost certainly underestimate global seagrass losses, which some researchers put at nearly 90% (Burke et al., 2001).

POLLUTION

Pollution is ubiquitous in marine waters and the tropics are no exception. Run-off from industry, cities and agriculture contain pesticides, metals and nutrients; oil and chemical spills are other pollutant sources, along with deposition of atmospherically transported compounds such as persistent organic compounds and mercury. Even in remote pelagic systems, many high level predators such as tuna have high tissue concentrations of these compounds and there is evidence that levels have been increasing over the past 20 years (Nakagawa et al., 1997). All of these pollutants disturb the oligotrophic waters that characterize most tropical marine systems; since many tropical marine environments have limited populations of biodegrading microbial fauna, they may be more sensitive to pollution than other systems. For example, on Laysan Island in the northwest Hawaiian Islands, there was a carbamate pesticide spill, likely due to a container washed off a passing ship (David et al., 2001). While the container was not found, its existence was inferred from an area referred to as “The Dead Zone”, which was identified in 1988. Surveys of the beach have found dead insects, ghost shrimp, Laysan albatross *(Diomedea immutabilis)* and, most significantly, Laysan finches *(Telespiza cantans)*, an endemic and endangered species (Woodward et al., 1998). Under normal terrestrial soil conditions, such pesticides are readily biodegraded. However tropical and subtropical sand beaches seem to have slower degredation of xenobiotics (Siegrist et al., 1994, Campbell et al., In Prep). As a result, mitigation of this site in 2002 required the removal of all of the contaminated sand for treatment. This raises substantial concern regarding the fate and effects of all contaminant spills in oligotrophic regions.

Corals in particular are affected by local nutrient run-off from land. Run-off changes the chemistry of the near-shore waters to favor algae over corals, and is often associated with greater siltation, decreasing coral growth rates (Koop et al., 2001; Ferrier-Pages et al., 2000; Pittock, 1999; Shimoda et al., 1998; Carpenter et al., 1998). Mangrove systems have not escaped the impacts of pollution either. Following oil spills, mangroves suffer defoliation, loss of associated species and death, with recovery times on the order of 15 to 20 years (Burns et al., 1993; Ellison and Farnsworth, 1996). There are fewer studies on the effects of pollution on seagrasses. Some seagrass communities have been adversely affected by oil spills (Zieman et al., 1984) and they are known to accumulate heavy metals, although no detrimental responses have been noted (Nienhuis, 1986). Herbicides, not surprisingly, have been found to adversely affect seagrasses. Photosynthetic activity was depressed in three species of seagrass *(Cymodocea serrulata, Halophila ovalis and Zostera capricorni)* when exposed to concentrations of diuron that were found in marine sediments off the coast of Australia (Haynes et al., 2000).

TOURISM

Tourism is a powerful tool for increasing public awareness of biodiversity and support for conservation efforts and funding, but it can create additional stress as well. Over-visitation and poorly managed visits cause severe degradation of tropical ecosystems. Poorly trained divers and snorkelers, as well as ill-placed boat anchorages damage coral reefs and seagrass beds. Tourism increases coastal development, pollution and extraction due to the increased number of people at a given location each year (Spalding et al., 2001). In some cases, seagrasses have even been removed from hotel beaches to create a more “aesthetically pleasant swimming zone” (Daby, 2003). Mangroves are also removed for development, and are trampled by other human activities.

OVERHARVEST OF FISHERIES

Fisheries are being depleted globally (Myers and Worm, 2003) and there is exceptional pressure on the tropics (Jackson et al., 2001). This includes large commercial fishing operations, such as tuna and other pelagic species, as well as small-scale efforts such as dynamite and cyanide fishing that damage not only fish populations but the entire community and reef-structure as well. In Tanzania, for example, dynamite fishing has been practiced for at least 30 years, resulting in substantial degradation of coral reef communities with loss of fish, invertebrates and coral (Guard and Masaiganah, 1997). Cyanide fishing was developed for the live fish trade and is now common on coral reefs in much of Southeast Asia. Sodium cyanide is “squirted” onto target fish, stunning them and allowing for easy collection. Although application of the poison is somewhat local, it spreads to cause lethal or sub-lethal impact on other organisms in the surrounding environment. In the Philippines, it has been in practice since the early 1960s (Halim, 2002).

OTHER EXTRACTIVE USES

While some coral rock is collected for the aquarium trade, coral reefs are also “mined” for building materials, especially in regions where there is little terrestrial rock. In the Maldives, for example, over a 13-year period, 93,450 m^3 of coral was extracted from a single atoll (Brown and Dunne, 1988). Such a large coral removal has reduced local fish and invertebrate populations, and compromised the physical protection that the reef offered to the island. Coral remnants left after mining seem to have very limited and slow recovery (Brown and Dunne, 1988).

Mangroves are extensively harvested for wood and wood-products. In the past their bark was collected for tannin production, but the dominant use of mangroves now is fuel and building materials (Ellison and Farnsworth, 1996).

INVASIVE SPECIES

Tropical marine systems are also affected by the spread of non-native species. The majority of these species are introduced accidentally (Carlton and Geller, 1993), although there are some cases of intentional introduction, especially for aquaculture (Randall, 1987). In Hawaii, for example, a recent survey of the archipelago determined that of the 23,150 species identified, 5,047 were nonindigenous (Eldredge, 2000); of these 343 are marine or estuarine and most were introduced through hull fouling or ballast release (Eldredge and Carlton, 2002).

Mangrove ecosystems are relatively protected from invasive species because of the unique environment in which they live. There are very few plants that are halophytes (salt tolerant), making the pool of potential invaders quite limited (Lugo, 1997). However, there are examples following hurricanes where disturbance opens up an opportunity for an introduced species (Loope et al., 1994). Additionally, mangroves themselves can be invasive. Red mangroves were introduced to Hawaii in 1902 from Florida in an effort to support coastlines. There are now on-going efforts to remove this species from the islands. While seagrass communities are almost certainly affected by invasive species, there is little literature on the effects.

What is the Effect of Climate Change on Tropical Marine Systems, Now and In the Future?

While it is clear from the previous section that there are already myriad threats to tropical marine ecosystems, it should be noted that at least some of these threats are beginning to be addressed through conservation efforts, especially in the case of coral reefs. However, Wilkinson (2002) noted that, “These improvements could be largely negated if the predicted threat posed by Global Climate Change of increasing sea surface temperatures and concentrations of CO_2 in seawater cause catastrophic bleaching and result in major reductions in the capacity of corals to calcify and grow.” This sort of dire assessment indicates how crucial it is that the impacts of climate change be considered and planned for in any contemporary conservation efforts. A variety of emergent stressors associated with climate variability and change, reviewed below, are of increasing concern to scientists and mangers.

INCREASING SEA TEMPERATURES

While water has a high heat capacity, it is not immune to changes in atmospheric temperature; as air temperatures rise, so will water temperatures. Ocean temperature changes actually lag behind air temperature changes to such a degree that even if atmospheric CO_2 concentration were stabilized today, the oceans would continue to warm for another century (Albritton et al., 2001). This may be a particular challenge to marine species that have relied on the thermal buffering capacity of the world’s oceans to maintain a relatively consistent environment. The response of these species to rising sea temperatures will depend on their thermal tolerances and the thermal tolerances of their competitors. Most species have an optimal physiological temperature range for respiration and growth (and photosynthesis in the case of primary producers). Outside that range, individuals are energetically challenged. In the case of some seagrasses, such as the eelgrass *Zostera marina*, increasing temperature favors respiration over photosynthesis, decreasing the seasonal growth optimum (Marsh et al., 1986; Short and Neckles, 1999). Other seagrasses in other parts of the world, however, increase photosynthesis with increasing temperature (Perez and Romero, 1992). For regions where seagrasses are living near their thermal maximum, such as shallow lagoons and near warm effluent from power plants, a 2 ºC increase in sea temperature will be detrimental as these populations are already living closer to their thermal limit, and a 4-5 ºC increase would result in extensive mortality (Edwards, 1995). Increasing water temperatures can also result in seagrasses being outcompeted by algal species, including epiphytic algae (Neckles et al., 1993).

The most dramatic effect of increasing temperature in tropical marine ecosystems is coral bleaching. Coral bleaching is defined by the loss of symbiotic dinoflagellates (zooxanthellae) or their pigments by the host coral animal (Glynn, 1993; Brown, 1997a). Normally, the symbiotic dinoflagellates provide the coral host with additional energy through photosynthetic activity. Long-term loss of the dinoflagellates can result in death of the affected coral (Harriott, 1985). Bleaching is considered to be a stress response caused primarily by increased water temperature (Glynn, 1993) and synergistically enhanced by increased solar irradiance levels (Jokiel and Coles, 1990; Lesser et al., 1990; Fitt and Warner, 1995). The water temperature need only increase by 1 to 2 ºC over the average annual thermal maxima for days to weeks to result in a bleaching event (Hoegh-Guldberg, 1999). These conditions have led some to rank climate change as potentially “the single greatest threat to reefs worldwide” (West and Salm, 2003).

SEA LEVEL RISE

Sea level is predicted to rise between 20 and 80 cm over the next century (IPCC, 2001a). This is due to both the thermal expansion of water and the melting of terrestrial ice masses (glaciers and ice sheets). Sea level rise will affect intertidal and coastal ecosystems by inundating them with water and affecting the availability of light, as well as altering patterns of water movement both intertidally and subtidally. For seagrasses, distribution and abundance are determined by salinity, light, depth and currents (Short and Neckles, 1999); rising sea levels could therefore dramatically alter seagrass communities and their composition.

Stable mangrove forests require stable sea level; prior to sea-level stabilization 6,000 years before present, large mangrove communities did not exist (Ellison and Stoddart, 1991). Thus, rapid sea level rise will likely be the greatest climate change challenge to mangrove ecosystems (Field, 1995). During past changes in sea level (8-9 mm/year) mangroves have migrated landward or seaward as necessary (Parkinson et al., 1994); with coastal development the ability of mangroves to migrate may be severely limited

(Ellison and Farnsworth, 1996). Reduced rates of sediment input due to coastal armoring and the damming of rivers will further limit the ability of mangrove forests to keep up with sea level rise (Field, 1995). The Intergovernmental Panel of Climate Change (IPCC) has recognized the severity of the threat of sea level rise to mangroves, and past floods in east Africa's coastal regions demonstrate the high vulnerability of this region (IPCC, 2001b). In Bangladesh and India, the mangroves of the Sunderbans are at great risk due to rising seas; a one meter rise in sea level will likely cause the Sunderbans and the tigers living there to disappear (IPCC, 2001b).

Under ideal conditions coral reef growth would likely be able to keep up with sea level rise predicted over the next century, but coral condition is not ideal and will become less so as the climate continues to change (Buddemeier and Smith, 1988). Bioerosion can erode reefs at rates up to 6 mm/year (Eakins, 1992), and changes in ocean chemistry will slow coral growth and may decrease the strength of coral skeletons (see further discussion of this under section on effects of Increasing Atmospheric CO_2). If bioerosion rates exceed calcification rates, then corals will not be able to keep up with sea level rise and may "drown".

EXTREME WEATHER EVENTS (HURRICANES, CYCLONES, ENSO PATTERNS)

The role of climate change in increasing the frequency and intensity of extreme weather is not well established (IPCC, 2001a). However if such effects do occur, this increases the importance of mangroves and coral reefs, which buffer coastlines against storm surge and high winds (Edwards, 1995). Increased frequency and intensity of storms would also threaten these critical ecosystems. Storm damage to corals, especially from hurricanes, is well documented throughout the tropics (Woodley, 1992; Harmelin-Vivien and Laboute, 1986; Done, 1992). Between December 1982 and April 1983, six hurricanes hit French Polynesia, and reef surveys following these events showed 50 to 100% damage depending on depth (Harmelin-Vivien and Laboute, 1986). Done (1992) points out that in some high-energy regions, like the Great Barrier Reef, damage from cyclone-generated waves is already dramatic with reef commonly lost to such events. Hurricane Gilbert "disturbed" the Cayman Islands in 1988 to the extent that reef surveys eight years later required no quantitative transects for reefs of less than 8 m depth (Riegl, 2001). Additionally, historic *Acropora palmata* zones were no longer present.

Seagrasses also suffer significant damage from current levels of tropical storms (Short and Wyllie-Echeverria, 1996) and are often slow to recover (Williams, 1990). Non-climatic pressures further exacerbate this slow recovery. Should storm frequency increase, some seagrass beds may not be able to recover sufficiently between storms. In Hervey Bay, on the coast of Queensland, Australia, a 100,000-hectare community was lost after two flood events and a cyclone in 1992, coupled with pressures from terrestrial nutrient and sediment run-off and shrimp trawling (Preen et al., 1995).

The damage of hurricanes to mangroves, through high winds and flooding, is less well studied. Of the reports that do is exist is one from Belize where large hurricanes have a frequency of about one every 30 years (Murray et al., 2003). Hurricane Hattie in 1961 had such force that it altered the coastline, elevating former mangrove habitat above saline influence and allowing it to become a pine forest (Murray et al., 2003).

The predicted increase in major climatic events, such as El Niño/Southern Oscillations (ENSO) (Timmermann et al., 1999; IPCC, 2001b), may have drastic effects on fish stocks, especially when combined with stressors such as overfishing. Reduced survival and growth rate, and altered migratory routes can all be caused by ENSO events, exacerbating the effects of intensive harvesting (Miller and Fluharty, 1992). Both 1972-73 and 1997-98 ENSO events significantly reduced Peruvian anchovy populations (Caviedes and Fik, 1992; Pfaff et al., 1999). ENSO events cause temporary range shifts, as well as introducing changes in reproductive physiology, egg and larvae survival, recruit and adult biomass, and fish schooling behavior (Jordán, 1991).

The ENSO event of 1982-83 marked the first contemporary broad scale coral bleaching and mortality event (Glynn, 1984). Since then, there have been subsequent bleaching events including the 1997-98 ENSO event. The rate of occurrence (annually in some cases) and almost global scale since the early 1980's 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, 1991; Glynn, 1993). From 1876-1979, only three bleaching events were recorded, whereas 60 are on record from 1980-1993 (Glynn, 1993). The increase in bleaching suggests that anthropogenic alterations of the environment are responsible, such as increases in annual sea surface temperature and occurrence of ENSO events (Hoegh-Guldberg, 1999; Pittock, 1999). The future for corals in regard to bleaching is grim. It has been suggested that by 2020 bleaching events like that caused by the 1997-98 ENSO will become "commonplace" (Hoegh-Guldberg, 1999). There has also been some correlation between ENSO events and disease outbreaks in coral and oysters (Harvell et al., 2002). Opportunistic pathogens can exacerbate the impact of bleaching events. For example, the gorgonian coral *Briareum asbestinum* suffered extensive mortality when affected simultaneously by bleaching and a suspected cyanobacterial pathogen (Harvell et al., 2001).

Clearly hurricanes, tropical cyclones and ENSO events have substantial effects on tropical marine systems even now. For the most part species in this region have evolved with these pressures and have clear cycles of recovery. However, this recovery is markedly slow even at historic frequencies of disturbance. If climate change does indeed increase the frequency and severity of these weather types, it could alter the tropical marine seascape, with more rapidly growing and less disturbable species becoming dominant.

INCREASING ATMOSPHERIC CO_2

Besides acting as a greenhouse gas, increasing atmospheric CO_2 will result in increased dissolved CO_2 in the water column, which in turn will cause a reduction in the pH of the oceans. As a result, less carbon will be biologically available to calcium-carbonate-forming organisms. To date pH-related changes in aragonite saturation levels have resulted in a 6-11% decrease in biogenic aragonite and calcite precipitation rates; a doubling of CO_2 is expected to result in another 8 to 17% reduction (Kleypas et al., 1999).

This would weaken shells and other calcium carbonate structures, as well as slow growth rates of marine invertebrates with calcium carbonate skeletons; these effects would be particularly pronounced in coral communities (Kleypas et al., 1999). Hoegh-Guldberg (1999) estimates that by 2050 calcification rates could be reduced by 14-30%.

Changes in available carbon may also affect seagrasses, as there is variability among species in how well they compete for available carbon (Short and Neckles, 1999). Also, as epiphytic algal growth is enhanced due to increased temperature and CO_2, in addition to local eutrophication, seagrasses will have to grow faster to keep pace with their epiphytes (Short and Neckles, 1999). Mangroves have been shown over the course of a single year to increase growth and reach reproductive maturity faster under enhanced CO_2 conditions (Farnsworth et al., 1996), however ecological ramifications are not clear. Field (1995) suggests that the real effect of enhanced CO_2 cannot be ascertained without a "long-term" experiment in which CO_2, water stress and nutrient stress are all monitored. For example, studies on marshes under elevated CO_2 conditions have shown changes in response over time, such as increased release of methane and other greenhouse gases, following initial increases in primary productivity (Dacey et al., 1994).

ULTRAVIOLET RADIATION

Ultraviolet (UV) radiation increases are generally equated with stratospheric ozone depletion. While there is some ozone thinning in the tropics (Madronich et al., 1995) and ultraviolet radiation is already naturally highest in the tropics, climate change induced tropical warming may also result in increased UV-penetrance into the water column. Warming results in doldrum conditions, causing increased stratification of the water column and decreased dissolved organic matter [including chromophoric dissolved organic matter (CDOM)], leading to higher levels of UV deeper in the water column (Vodacek et al., 1997; Siegel and Michaels, 1996). This means that as thermal stress increases, so does the stress of UV radiation. The negative synergistic effects of temperature and UV radiation on coral are well established (Lesser et al., 1990; Dunne and Brown, 2001). UV is also expected to affect seagrass photosynthetic ability (Larkum and Wood, 1993), although they may have some ability to protect themselves from slow changes through increased production of UV-blocking pigments (Dawson and Dennison, 1996).

Mangroves and seagrasses are both sources of CDOM and other UV-blocking compounds that can be transferred to the marine water column when plants die or leaves are lost (Lovelock et al., 1992; Stabenau et al., In Prep). Therefore, as mangroves and seagrass systems suffer degradation and losses due to various stresses, a result may be continued exacerbation of UV stress to all systems.

EFFECTS ON SPECIES OF CONCERN

Seagrasses provide a primary food source for dugongs, manatees and green turtles (Lanyon et al., 1989), all species of conservation concern, and loss of seagrass beds or shifts in species composition could have significant impacts on these species. In addition to the potential loss of food source and habitat (seagrasses, coral reefs), turtles face an additional threat from climate change: skewed sex ratios. Sex determination in turtles is temperature dependent (Davenport, 1997). Higher temperatures lead to more female turtles with only 1.5 ºC separating the sexes (Morreale et al., 1982). A two-ºC increase in temperature is expected to dramatically skew the sex ratio and a four-ºC increase would virtually eliminate male offspring (Janzen, 1994).

What are Possible Options for Increasing the Resilience and Resistance of Tropical Marine Systems to Climate Change?

The basic premise of increasing resistance and resilience of ecosystems in responses to a changing climate is very similar to that used in designing conservation strategies to protect biodiversity from any threat. Given the scope and multi-faceted effects of climate change, the challenge is to broaden our thinking on both spatial and temporal scales, employ greater rigor in setting limits and enforcement, and make sure that a few new facets are considered. Here the strategies are divided up into three categories: 1) create sufficient space for change, 2) reduce all non-climate stresses and 3) identify resistant and resilient populations for special protection. While most of the effort to date on these approaches has focused on coral reefs, and the examples following reflect that, it should be noted that they are also applicable to other coastal, tropical marine systems.

CREATING SUFFICIENT AND APPROPRIATE SPACE

1. CREATE RESERVES THAT CONTAIN REPRESENTATIVE SYSTEM TYPES (CORAL REEF, MANGROVE, SEAGRASS) ACROSS ENVIRONMENTAL GRADIENTS

Human development is unlikely to cease anytime soon. It is thereby crucial that areas be protected from human encroachment as a first step in biodiversity protection. There are a few things to keep in mind when designing reserves in response to climate change. While many of these overlap with standard conservation requirements, climate change may require additional diligence in meeting these requirements.

a) It is not enough simply to have space: it must be ecologically significant space. Done (2001) describes “the perfect regional configuration” of a marine protected area (MPA) as one which contains a “full suite” of regional biodiversity. He also suggests the need for MPAs to encompass large areas, depth gradients and a high diversity of species. Maximazing the heterogeneity of the reserve is crucial (see #4 below)

b) Reserves also require effective management and enforcement. The absence of these activities leads to “paper parks” with no real functionality (Westmacott et al., 2000).

c) Although the effects of climate change will not stop at the borders of reserves, they can provide an area in which non-climate stresses can be more effectively limited. If this does in fact confer resilience, decrease adverse effects, or increase recovery potential, then reserves may act as the seed populations for numerous sites beyond reserves that were not experiencing the same benefits.

d) One reserve of each habitat type is also insufficient. “Replication” of habitats in multiple protected areas is necessary so that loss of a single reserve would not mean loss of the only remaining examples of habitats protected by the reserve.

Designing protected areas in response to climate change will require additional care for some tropical marine systems. For example, communities dependent on biological rather than physical matrices, such as seagrass beds or mangroves, are often the least resistant or resilient to environmental perturbations (Roberts et al., 2003).

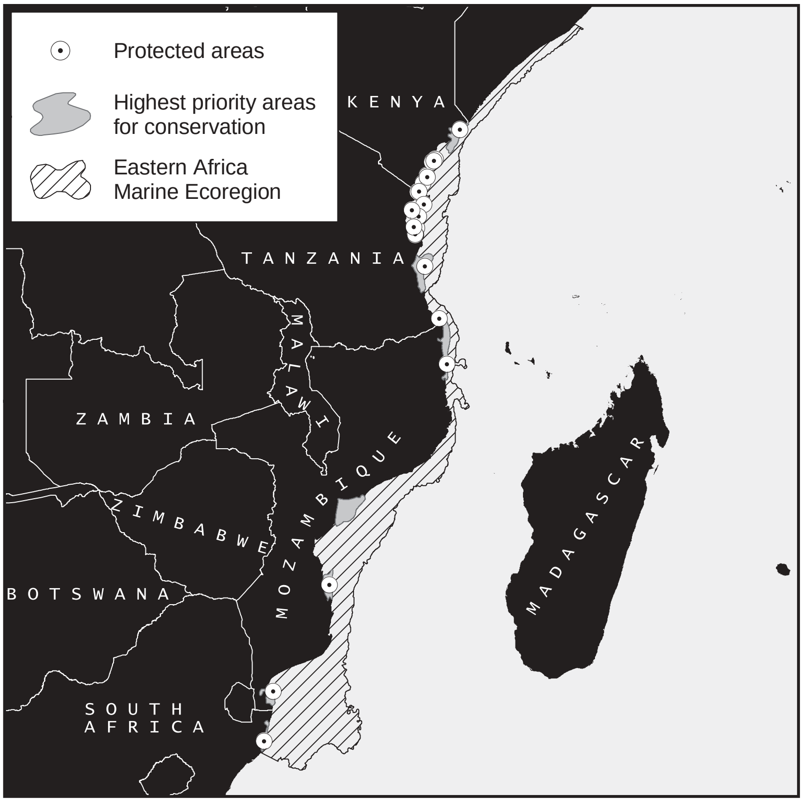
2. CREATE NETWORKS OF RESERVES

The “single large or several small” debate regarding reserve size has a long history in both terrestrial and marine conservation circles, but reserve networks are increasingly recognized as a powerful tool for the protection of marine ecosystems. Such networks confer a number of advantages. First, they allow coverage across a gradient of biogeographic and oceanographic conditions without the social, political and economic complexity of establishing a single large reserve. This is especially important where such gradients and the communities dependent on them cross political boundaries. The Eastern Africa Marine Ecoregion (Olson and Dinerstein, 1998), for example, is a network of marine protected areas that includes 4600 km of shoreline covering an extensive latitudinal and thermal gradient along the coasts of five countries (Somalia, Kenya, Tanzania, Mozambique and South Africa) (Figure 1). Networks are also more effective than single reserves at supporting organisms with a diversity of dispersal distances, which in marine flora and fauna may range from centimeters to thousands of kilometers (Shanks et al., 2003). By protecting large total areas while keeping individual protected areas somewhat smaller, networks are better able to provide source populations for recolonization of damage areas in other reserves in the network, as well as areas outside the network (Done, 2001; Westmacott, 2000). Finally, they have the added advantage of creating replicate reserves, spreading the risk while increasing the probability that representative biodiversity will avoid complete loss with a single damaging event.

3. PROTECT CLIMATIC REFUGIA

Identifying locations that are more stable during periods of global climate change can be very useful for conservation. These sites may have strong currents, upwelling or other oceanographic features that make them less prone to thermal fluxes. For example, at the south end of the island of Sulawesi, and along the Makassar Strait (between Borneo and Sulawesi), there is a region for which a 17-year analysis (1985-2001) showed no thermal anomalies greater than 1°C in many parts of the region. It is hypothesized that this is due to high current flow in this region, or other oceanographic features. (Kassem et al., 2002). Such areas may offer at least temporary refuge from warming waters resulting from climate change, and should be considered as sites or extensions of sites for conservation efforts (Done, 2001; West and Salm, 2003). Other local physical features that may create thermal refugia include proximity to deep water, shade, high wave energy and turbidity (West and Salm, 2003). Some of these have the added benefit of reducing exposure to light, including UV radiation.

Figure 1 Creating networks of reserves such as the String of MPAs along the East coast of Africa, is one approach to increasing resilience.



4. PROTECT PHYSICAL AND BIOLOGICAL HETEROGENEITY

It will not always be clear what aspects of a system confer resistance or resilience. This requires adoption of a bet-hedging strategy in which heterogeneous areas are selected for protection, providing as many options as possible. There has been limited research on this approach; however, a nice example exists involving two populations of checkerspot butterfly *(Euphydryas editha bayensis)*, with strong climatic influences on population extinction rates (McLaughlin, 2002). The population living in the more homogeneous habitat went extinct first, and prior to its extinction had more dramatic fluctuations in response to climatic changes than the population from the more heterogeneous site. This indicates that in addition to reserve size, it is important the protected area also encompass a variety of habitat types, including small-scale variability. Even for species for which we believe we know the ideal habitat type, it is possible that as the climate changes their habitat requirements will also change.

5. RESTORATION OF DEGRADED HABITATS

Some ecosystems have been dramatically degraded, but retain sufficient importance from an ecological or global biodiversity standpoint that it is worth trying to resurrect them. Mangrove forests, for example, have been substantially degraded globally yet serve an important ecosystem function when intact. In Vietnam (see details below under Case Studies) and east Africa, efforts are being made to restore mangrove forests to improve coastal protection and estuarine condition. They may also convey some advantage for nearshore coral reef systems, releasing protective CDOM (Lovelock et al., 1992; Stabenau et al., In Prep). Thus restoration of mangroves is advantageous not only for the mangrove trees themselves, but the adjacent biodiversity as well.

Restoration of seagrass beds is also recognized as environmentally and economically important, and a number of techniques for conservation and restoration have been developed (Fonseca et al., 1998). Some restoration techniques harness the natural behavior of wild animals: workers in the southern United States have installed bird roosts to provide natural fertilization for troubled seagrass beds (NOAA, 2003).

REDUCE OR ELIMINATE NON-CLIMATE STRESSES

6. ELIMINATE DESTRUCTIVE FISHING PRACTICES AND OVERFISHING

Reducing the damaging effects of destructive fishing will require creating regulations to limit these practices, as well as creation of no fishing zones to ease recovery and enforcement (Westmacott et al., 2000). To alleviate the root cause of the pressure it will be necessary to develop alternative livelihoods (Westmacott et al., 2000) or alternative techniques for fishing communities (Bryant et al., 1998). For example, in the Philippines where cyanide fishing has been a problem for decades, a government and NGO (International MarineLife Alliance) partnership has created the Cyanide Fishing Reform Program. This program trains fishermen in alternative fishing practices while the government is increasing enforcement of anti-cyanide fishing regulations. Efforts are being made to export the program to Indonesia where cyanide fishing is also a widespread concern.

7. REDUCE POLLUTION, INCLUDING TERRESTRIAL OF NUTRIENTS AND PESTICIDES

Limited research on interactions between climate and non-climate-related stresses, indicates synergistic responses (McLusky et al., 1986). For example, when rainbow trout (*Salmo gairdneri*) are exposed to the pesticide permethrin over a range of temperatures the toxicity increases as temperature increase (Kumaraguru and Beamish, 1981). Additionally, adverse effects of nutrient run-off on coastal ecosystems are well established (Koop et al., 2001) and less disturbed reefs generally have a greater likelihood of recovering from bleaching events (Brown, 1997a). Efforts to reduce pollution will generally focus on activities outside of reserves. This may require improving terrestrial land use practices to decrease nutrient and sediment run-off, eliminating local use of persistent pesticides or improving the quality of effluent from municipal and industrial sources. In the case of atmospheric deposition of contaminants the efforts will need to be similar to those needed to reduce greenhouse gas emissions. However, some atmospherically transported compounds, such as mercury, also have local sources. Mercury is used extensively in gold mining which in many cases occurs in watersheds adjacent to tropical marine systems.

8. REDUCE DAMAGING EXTRACTION

Tropical marine ecosystems have long been a source of materials to sustain life, be it food, clothing or even building materials. As human populations have grown, the extent of these extractive activities has become unsustainable. In addition to quantifying the problem of coral mining in the Maldives, Brown (1997b) has attempted to delineate alternatives for local construction. She estimates that, through increasing use of concrete blocks created from coral sand and dead coral material, local demand could be satisfied until 2050. Reducing extraction pressure on these slow recovery systems (corals, mangroves and seagrasses) will make them better able to respond to the stress of climate change. Additionally, decreasing removal of organisms and populations, increases the potential for greater genetic diversity in these systems, again increasing the likelihood that they will be better able to respond to the stress of climate change (e.g. Dodd and Rafii, 2001).

PROTECT RESISTANT AND RESILIENT POPULATIONS OR COMMUNITIES

9. IDENTIFY THOSE POPULATIONS OR COMMUNITIES THAT HAVE ENDOGENOUS FACTORS WHICH MAKE THEM LESS SUSCEPTIBLE TO THE EFFECTS OF CLIMATE CHANGE

Some populations or communities may possess endogenous factors that enhance their ability to deal with the added stress of climate change. For example, some coral bleaching events are caused by high temperatures exacerbated by UV (Lesser et al., 1990; Dunne and Brown, 2001). Zooxanthellae produce compounds called mycosporine-like amino acids (MAAs), which can effectively act as sunscreens. Concentrations of MAAs vary between species and populations (Gleason, 1993; Shick et al., 1996) and they can be induced to varying degrees by exposure to UV. It may be that some populations with naturally higher MAA concentrations, or an enhanced ability to induce these compounds during times of stress, are less prone to bleaching under combined UV/heat stress conditions. Such endogenous factors may be present in a population as a result of surviving a previous bleaching event, effectively a preadaptation (West and Salm, 2003; Done, 2001).

10. MAINTAIN DIVERSE GENE POOLS, AND NATURAL DIVERSITY OF ECOSYSTEMS

This is also a bet-hedging strategy. Since it is currently not possible to predict exactly how any location will change, it is best to retain a range of response options in ecosystems. Evolutionarily this protection from the unknown has come from genetic diversity. It is uncertain what gene or trait might confer a future advantage, but the more options you have the greater the likelihood that you have a combination that will survive (see review in Dodd and Rafii, 2001). It is similarly beneficial to maintain diversity at the species level; you never know which species will be the key to helping the system through a stressful period.

This list is by no means complete. It should be seen as a starting point for developing ideas and more importantly, for developing field testing strategies to begin assessing how these will work in your own system.

Adequate monitoring of individual sites is essential to best assess how each ecosystem is being affected and what strategies can be implemented. For example, you can only protect climate refugia if you know where they are. This monitoring must be coupled with field testing to see which strategies are most successful. In the case of climate change, it is necessary to start implementing some strategies—which can be seen as "do no harm"—before they are fully tested. Otherwise, as the effects of climate change become more apparent and pressing, it will be too late to implement many of the strategies and expect to have an enhancement in resilience or resistance.

What is Necessary to Implement These Strategies?

The first step is to gain the interest of informed resource managers and local communities. They will need to have sufficient levels of concern for their systems in relation to climate change. The next step requires involvement of stakeholders in the resilience-building process. Much of the stress of protected areas is caused by those who live outside the reserve through their competition for or damaging use of resources. More rigorous no-take and protective status enforcement, more and larger protected areas and limiting of development and extraction even outside of protected areas will not happen unless stakeholders can understand the benefit to them and are made part of the process in delineating this new level of protection. Finally, there must be vested long-term resources and vision for maintaining the protections that are created. Climate change is not a threat that can be solved in the short term. Systems will need to be resilient for decades to come and the stress will become more intense as atmospheric concentrations of CO_2 increase and temperatures rise.

Case Studies

It is crucial that we learn more about which strategies are effective and what additional factors may play a role in increasing the resistance and resilience of natural systems to climate change. Unfortunately, while there are many ideas about possible strategies, there is limited empirical evidence to aid those trying to make decisions. Waiting for full experimental conclusions before taking action is not an option. Climate change is already affecting ecosystems, and waiting could allow windows of conservation opportunity to close. There are a few studies that have begun to assess the success of various strategies and may serve as models for developing other studies.

MANGROVES

Vietnam has an extensive coastline (3000 km) with the majority of its populations living in the lowland alluvial plains. Since 1945, there has been a 45% loss in mangrove cover in Vietnam (Jameson et al., 1995), mostly due to conversion of mangrove forests to agriculture, including shrimp aquaculture (Tri et al., 1998). As previously mentioned, mangroves play an important role in coastal protection, and the plains previously protected by mangroves are threatened by rising sea level (potentially a third of the Red River's delta will be inundated) and a possible increase in the frequency of storms, such as tropical cyclones. As a result, Red Cross/Red Crescent societies are supporting efforts to restore mangroves to enhance protection for these regions. Since the project began in 1997, 18,000 ha of mangroves have been planted along 100 km of coast. The increased mangrove forest cover is improving conditions for associated species, including 109 species of bird. Harvestable marine resources also seem to be increasing in number:

areas with restored mangroves are no longer having to purchase "sea products" such as crabs from other provinces. (D. V. Tao, Pers. Comm.)

CORAL REEFS

Protecting coral reefs from the effects of bleaching might seem an impossible task. While nothing can be done to lower the temperature of the warming oceans, actions can be taken that limit other stresses that make reefs more susceptible to bleaching. In 2000, then-Governor Tauese Pita Fiti Sunia declared his intention to achieve a goal of protecting 20% of American Samoa's coral reefs in "no take" MPAs. To improve the likelihood that these MPAs would meet their long-term goals of protecting biodiversity, WWF developed a project to help identify not only the sites that are less likely to bleach and more likely to recover, but also to determine what factors account for these differences. To this end, the project is comparing seven sites with varying levels of formal protection, impact from high nutrient terrestrial run-off and concentrations of endogenous protective factors in their corals. Researchers survey the coral reefs quarterly for bleaching and recovery, monitor nutrient concentrations and water temperatures and assess the concentration of the sunscreening mycosporine-like amino acids in corals at each site. The project includes sites in protected areas such as the National Park of American Samoa and the Fagatele National Marine Sanctuary. Additionally, researchers are working with local stakeholders (managers, villages) to not only conduct the research but to begin discussions on possible response to study findings. In March of 2003, there was a bleaching event in the waters of American Samoa associated with elevated water temperatures. Bleaching ranged from 5 to 30% at the study sites. The recovery of the reefs from this bleaching event is now being tracked. WWF hopes to be able to use results over the next two years to assist in making management decisions for coral reefs in response to climate change. To this end a Coral Bleaching Monitoring Protocol is also being developed and efforts are underway to expand the approach of the American Samoa project to other coral reefs around the world.

Conclusion

Protection of natural resources from climate change will require a two-pronged approach. In addition to taking the types of measures outlined in this chapter, it will also be necessary to limit the rate and extent of anthropogenic climate change. This does not require that resource managers become advocates, but it may be useful to start documenting the impacts of climate change, the cost of preparing for the changes and the prognosis for success in responding at a local level. This information could be shared at management and scientific fora to indicate the need for broader-scale action.

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CHAPTER 7
Freshwater

Protecting Freshwater Ecosystems in the Face of Global Climate Change

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FRESHWATER ECOSYSTEMS HAVE BEEN critical to sustaining life and establishing civilizations throughout history. Humans rely on freshwater systems not only for drinking water, but also for agriculture, transportation, energy production, industrial processes, waste disposal, and the extraction of fish and other products. As a result of this dependence, human settlements worldwide are concentrated near freshwater ecosystems, with over half of the world's population living within 20 km of a permanent river (Small and Cohen, 1999).

In addition to humans, an enormous array of plants, animals, and microorganisms depend on freshwater ecosystems for their survival. Although freshwater ecosystems contain only 0.01% of the Earth's water and cover a small fraction of the planet's surface, rivers, lakes and wetlands harbor a disproportionately high fraction of the Earth's biodiversity. Freshwater fishes alone account for over one-fourth of all living vertebrate species. (McAllister et al., 1997).

Because freshwater ecosystems continuously channel precipitation from the surrounding landscape through the interconnected lakes, rivers, and wetlands that lie below, they can be surprisingly sensitive to distant activities. Increasing human water needs and extensive land alteration has contributed to the decline of countless freshwater species. Freshwater biodiversity is now more threatened than terrestrial biodiversity, and the projected mean future extinction rate of North American freshwater animals is about five times higher than for terrestrial animals, and comparable to predicted extinction rates for tropical rainforest communities (Ricciardi and Rasmussen, 1999).

The long-term protection of freshwater species is largely dependent upon identifying the underlying physical processes of freshwater systems that are most vulnerable to change, and determining how changes in these physical features might affect the resident flora and fauna. For this purpose, it is useful to divide freshwater ecosystems into rivers, lakes, and wetlands. In this chapter, rivers and streams are considered to be channelized bodies of water that generally display continuous flow, and lakes are relatively still bodies of water that can be either connected (through rivers, streams, etc.) or isolated from other bodies of water. Wetlands (also known as marshes, swamps, fens, bogs, floodplains, or depressions) are areas where the water table is at or near the surface, and vegetation is submerged for at least part of the year. Many, although not all, wetlands are connected to or interact strongly with lakes and rivers.

Because freshwater ecosystems are sensitive not only to water temperature, volume, and flow, but also to variability in these factors, rivers, lakes and wetlands are expected to display a wide variety of changes in response to global climate change. The purpose of this chapter is to provide information and strategies for designing and managing reserves that will allow freshwater ecosystems to withstand and/or adapt to climate change. The chapter begins with a review of current threats to freshwater ecosystems and some of the observed and predicted effects of global climate change on these systems. Next, general suggestions are presented for designing and managing freshwater reserves to increase resistance and resilience to climate change. The chapter concludes with a discussion of methods for selecting and implementing specific adaptation strategies to accommodate global climate change in freshwater ecosystems.

Current Threats to Freshwater Ecosystems

Because freshwater ecosystems depend strongly on physical features such as water quantity, quality and flow, many of the threats to these ecosystems involve activities that alter fundamental physical characteristics. Freshwater ecosystems throughout the world are threatened by human activities that directly alter system hydrology, such as construction of physical barriers to flow, water extraction, and filling or draining of shallow habitats. Pollution of waterways with toxic substances and excessive nutrients, as well as destructive land use practices in areas surrounding freshwater ecosystems, lead to reductions in water quality. While the above threats directly affect physical features of freshwater ecosystems, the introduction of exotic species primarily affects native biota. The invasion of freshwater ecosystems by non-native species is rapidly becoming one of the most serious threats to freshwater communities. Overexploitation of animals associated with freshwater ecosystems, particularly freshwater fishes, is also a continuing problem. Finally, penetration of harmful UV-B radiation into water bodies is increasing in many areas due to interactions between a number of anthropogenic factors, and a range of negative impacts on freshwater communities may result.

ALTERATION OF HYDROLOGY

PHYSICAL BARRIERS TO FLOW

Humans have constructed a variety of physical barriers, including dams, levees and dikes, to prevent flooding, generate power, supply water for irrigation or municipal water supplies, and provide recreational opportunities. Dams have been built on every continent except Antarctica; they are prevalent in developed countries and their rate of construction is increasing rapidly in developing nations. Dams have traditionally been viewed as an environmentally-friendly and sustainable means of ensuring water supply, controlling floods, and generating power without polluting the environment. However, retaining water and altering its natural flow can lead to large changes in aquatic and terrestrial habitats, both above and below dams.

Initial flooding above dams to create reservoirs can result in massive losses of terrestrial habitat; for example, India lost approximately 479,000 ha of forest land to various river valley projects from 1950-1975 (Goldsmith and Hildyard, 1984). Stagnation and low flow rates in reservoirs can lead to large changes in water temperature, including variations in seasonal peak temperatures and a reduction in natural temperature variation (Baron et al., 2003). Silt that is normally carried down rivers accumulates behind dams, and costly removal procedures are sometimes necessary to ensure that dams remain functional. In addition, the decomposition of flooded vegetation above dams may release significant quantities methane and other greenhouse gases, particularly in tropical areas where plant biomass is high (McCully, 1996); the magnitude of this greenhouse gas release compared to fossil fuel burning has been widely debated (Rudd et al., 1993; Gagnon and Chamberland, 1993; Fearnside, 1997).

Dams also cause dramatic changes in downstream flow regime, where seasonal, dynamic flows are replaced by steady water release for energy production, or intermittent large releases to lower reservoir levels. The temperature variability of water released from the bottom layer of reservoirs is low compared to that of natural stream flows, and oxygen content may be reduced. The loss of natural silt in released water can lead to a range of damaging effects downstream, including changes in chemical composition, river bank erosion, and massive habitat loss and erosion in coastal deltas and floodplains.

Finally, dams pose a significant barrier to diadromous fish and other migrating animals. In the Brazilian Amazon, where many fish undergo long-distance migrations during the rainy season, 79 dams are either planned or currently in existence (Pringle et al., 2000), and hydroelectric development is considered to be the greatest threat to Amazonian fisheries in the near future (Bayley and Petrere, 1989; Goulding et al., 1996). On the Columbia River in the northwestern United States, 19 major dams have already contributed to the extinction of at least 106 local stocks of pacific salmon (McGinnis, 1994).

Humans also build obstructions such as levees and dikes to prevent water from flowing laterally over river banks during high flows—in this way, land adjacent to rivers can be developed with less risk of seasonal flooding. However, these barriers disrupt connections between rivers and valuable floodplain habitats, which serve as refugia and spawning grounds for many animals, and are often sites of high biodiversity.

WATER DIVERSION/WITHDRAWAL

In order to meet the agricultural and municipal water needs of a growing population, large quantities of water are diverted or withdrawn directly from rivers, lakes, or the underlying water table. A large proportion of the world’s population is currently experiencing water stress, and human water needs are expected to increase dramatically in the coming decades due to projected population growth and increased development (Vörösmarty et al., 2000).

Withdrawal of water for human needs can reduce the total amount of water available to aquatic biota (resulting in low stream flows and declining lake levels), reduce seasonal variability in flows, and lead to large losses of habitat, particularly valuable edge habitats that are used for spawning or rearing (Tyedmers and Ward, 2001). Withdrawal of groundwater for agriculture in adjacent areas can completely dry up valuable wetlands, even if the habitat itself is protected (McDowall, 1984). Water that is withdrawn from freshwater systems is often returned in the form of household wastewater, agricultural runoff, or industrial cooling water, with reduced quality (increased pollutants, changes in nutrient load) and altered temperature.

FILLING OR DRAINING SHALLOW HABITATS

Wetlands throughout the world have been filled or drained for development or agriculture (Brinson and Malvarez, 2002; Junk, 2002). These habitats have traditionally been regarded as useless in their natural state, and many countries have provided subsidies to encourage the conversion of wetlands to agricultural land. Thankfully, the valuable ecosystem services that wetlands provide, including water purification, groundwater recharge, and flood control, are becoming more widely appreciated (National Research Council, 1995).

INVASIVE SPECIES

Apart from direct alterations of system hydrology, the most serious threat to freshwater ecosystems in many areas is the presence of invasive, non-native species. Invaders are often adaptable generalists who breed and disperse quickly, endangering native species through highly efficient competition, predation, or habitat alteration.

DELIBERATE INTRODUCTIONS

The deliberate introduction of non-native species for commercial or recreational fishing is widespread; for example, approximately 80% of the alpine lakes in the western United States have been stocked with non-native fish (Bahls, 1992). In many areas, lakes and streams are continuously re-stocked to maintain populations. Although these fisheries generate enormous income, the effects on freshwater communities can be devastating. Hundreds of native fish species in North America are threatened, and competition or predation by non-natives is the primary threat to many of these species (Allan and Flecker, 1993; Schindler, 2001). In Lake Victoria, increased predation and competition due to the introduction of the Nile perch and several non-native tilapiine species (in addition to ongoing environmental changes in the lake), led to the extinction of up to 200 endemic cichlid species within a few decades (Ogutu-Ohwayo, 1990; Witte et al., 1992).

ACCIDENTAL INTRODUCTIONS

Many species are introduced to and spread throughout freshwater ecosystems accidentally—for example, non-native species can be attached to boats or transported in ballast or bilge water. The zebra mussel, which is native to Europe, was accidentally introduced to the Great Lakes in 1986 through ballast water. It has altered water chemistry through highly efficient filter feeding, outcompeted populations of native mussels, and fouled boats, docks, and power plant intakes, costing millions of dollars in damage (Cooley, 1991; Effler et al., 1996; Nalepa et al., 1996). Several species of native mussels, clams, and commercially important fish are threatened directly or indirectly by its presence (Nalepa et al., 1996; Roberts, 1990).

Non-native plants can also cause damage to aquatic ecosystems. The water hyacinth, which is native to the Amazon River, has been introduced to freshwater ecosystems on several continents, including Africa and North America. It grows on the surface of water, blocking light, decreasing oxygen levels, and changing water chemistry (Gopal, 1987). Entire food webs have been altered as a result, and fish populations have been reduced or eliminated in some areas (Gowanloch, 1945; Timmer and Weldon, 1967). In the United States, water hyacinth has spread to rivers, lakes and lagoons throughout the country, forming dense mats that block canals and drainage pipes, prevent swimming and boating, and impair waterway navigation (Buker, 1982). In Lake Victoria, water hyacinth "hot spots" have persisted near areas of urban, industrial, and agricultural pollution, despite efforts to eradicate the floating vegetation (Lake Victoria Environmental Management Project, 2003)

POLLUTION

Freshwater ecosystems are polluted by a variety of human activities, from large-scale agriculture and industry to everyday behaviors, such as driving cars and fertilizing lawns. Large quantities of pollution often enter freshwater systems from point sources, such as industrial or municipal sewage outflows; for example, 23.4 billion tons of sewage and industrial waste was dumped into the Yangtze River in 2001, threatening human health and the survival of the endangered Yangtze River dolphin (Young, 2002). Thus, the focus on protecting water quality in many countries has been on preventing point source pollution (i.e. U.S. Clean Water Act of 1972). However, nonpoint source pollution is far more significant in many cases. Airborne pollutants can enter the atmosphere and travel long distances, entering lakes and waterways in otherwise pristine locations. Pollutants dissolved in runoff from the surrounding landscape may account for the greatest source of pollution in many freshwater ecosystems—for example, it has been estimated that 80% of the nutrients (nitrogen and phosphorus) that pollute U.S. waterways derive from nonpoint sources such as agricultural and urban runoff (Shaw and Raucher, 1993).

NUTRIENT POLLUTION

Runoff from fertilizers used in commercial agriculture or private yards adds large amounts of nitrogen and phosphorus to freshwater ecosystems. This can be especially problematic in lowland areas and in lakes or rivers with developed shores. The added nutrients lead to excess growth of algae (which is sometimes toxic), resulting in reduced water clarity and light penetration. Because of this increased primary productivity, the activity of decomposing, oxygen-consuming bacteria increases and oxygen levels decline. Shifts in the food web and alterations in bottom-water habitat can lead to changes in species composition and distribution. For example, the density, distribution and relative abundance of aquatic plants can change after eutrophication (Schmieder, 1997), and valuable fish species are often replaced by less desirable fauna that can tolerate low oxygen levels (Egerton, 1987). Natural eutrophication is a normal state in the succession of lakes as they age, but polluted runoff has led to early eutrophication and changes in the community structure of many naturally oligotrophic (nutrient-poor) lakes.

TOXIC POLLUTION

Toxic pollution in freshwater ecosystems can devastate local biota and endanger human food sources. Most toxic pollution derives from industry (i.e. dioxin, PCBs) or agriculture (pesticides such as DDT and toxaphene). Heavy metals, such as arsenic, zinc, selenium, and mercury are also released from mining and other industrial activities. Heavy metals and toxic compounds can become volatized in warmer parts of the globe, enter the atmosphere, and re-condense in cooler areas, often contaminating pristine sites and indigenous food supplies (Dewailly et al., 1989; Kidd et al., 1995; Wania and Mackay, 1993). These pollutants cause massive mortality events, such as fish kills, and are found in concentrations considered unsafe for human consumption in many aquatic animals, such as freshwater mussels (Lau et al., 1998). Because many pollutants accumulate in fatty tissues, they are magnified in the food chain and can reach concentrations in fish-eating birds and mammals that are up to 10 million times higher than those found in polluted water (Schindler et al., 1995).

Acid rain containing high levels of sulfuric and nitric acids is also a serious threat to many freshwater ecosystems, particularly lakes at high altitudes and latitudes. These pollutants enter the air mainly through the burning of fossil fuels, are transported atmospherically to distant parts of the globe, and are finally released into freshwater ecosystems through precipitation. In lakes susceptible to acidification (where acids are not readily neutralized by the soil or water), lake pH is lowered dramatically, and species composition and abundance can change as a result (Schindler et al., 1985; Wright and Schindler, 1995).

LAND USE

Destructive land use practices that result in vegetation loss anywhere within the drainage basin of a river can have negative impacts on freshwater ecosystems. The forests and native plant communities surrounding lakes, rivers, and wetlands help protect water quality and quantity by filtering and storing runoff. Changes in land use brought about by agriculture and urbanization (such as deforestation, chemical fertilization, and paving) lead to increased runoff with higher levels of nutrients and other pollutants. In addition, more sediments are washed into the system and water turbidity rises, with negative impacts on fish, filter feeders, aquatic plants, and bacteria (Baron et al., 2003; Megahan et al., 1992).

The vegetation immediately surrounding water bodies (i.e. riparian or floodplain vegetation) is particularly important to freshwater ecosystem health, as it shades water bodies (regulating water temperatures and providing thermal refugia) and supplies organic material such as falling leaves, insects, and woody debris to freshwater systems. The deforestation of Amazonian floodplains is considered to be one of the major forces behind the decline of Amazonian fisheries, as deforestation in these seasonally flooded areas has led to massive erosion, increased sediment load, and a decrease in large, woody debris in the river (Goulding et al., 1996). Although its function has not been studied extensively in tropical rivers, large, woody debris has been shown to play numerous important roles in temperate river systems, including altering flow to create habitat heterogeneity (i.e. stepped channels and deep pools) and helping to determine river channel form and stability. Woody debris also has a long residence time in most water bodies, and provides substrate, food, and shelter for a wide range of plants and animals (Bilby and Bisson, 1998; Petts, 2000).

OVEREXPLOITATION

Fish and shellfish harvests have declined sharply in the last few decades, and in many cases this decline is due to commercial or recreational overexploitation of these resources (Naiman et al., 1995). Overharvesting of aquatic species has occurred repeatedly in freshwater ecosystems throughout the world, in part because high levels of natural population variability can mask the effects of overexploitation until population declines are severe and irreversible (Ludwig et al., 1993). Attempts to bolster declining populations, such as with the release of hatchery fish, often exacerbate the problem (Hilborn, 1992; Meffe, 1992). Species inhabiting lakes can be particularly vulnerable to overexploitation in areas where there is not continual recruitment from populations outside the lake (Abell et al., 2002).

Hunting or extermination of mammals associated with freshwater ecosystems (i.e. beavers, muskrats) can affect not only the biological community, but also the physical structure of freshwater ecosystems, as the activities of some species lead to flooding and the creation of important wetland habitats. Hunting in tropical rivers and floodplains can add additional pressure to endangered species such as manatees, turtles, river otters and caiman (Junk, 2002).

EXPOSURE TO ULTRAVIOLET-B RADIATION

Exposure to high levels of Ultraviolet-B (UV-B) radiation can have a range of harmful effects on living organisms, and damage to the ozone layer has caused an increase in UV-B radiation of up to 50% in some alpine areas (Schindler et al., 1996). Although the ozone layer protects both terrestrial and aquatic ecosystems from much of the sun's UV-B radiation, some aquatic animals may be more vulnerable to the radiation that reaches the Earth's surface because they have historically been afforded a high level of protection by natural characteristics of the waters they inhabit (Williamson, 1995). UV-B radiation generally only penetrates the top layer of water bodies; the depth of UV-B penetration depends on water clarity and on the concentrations of dissolved organic carbon (DOC) and/or chromophoric dissolved organic matter (CDOM), which are derived from the breakdown of plant materials or dead organisms.

Animals living in clear or shallow bodies of water, those that are sessile or can't sense UV-B radiation, and those that are restricted to the upper layer of water during the daytime (i.e. small zooplankton taking refuge from downward migrations of predators) may be particularly vulnerable to increases in UV-B radiation (Williamson, 1995). Primary productivity has decreased as a result of increased UV-B radiation in some cases, but productivity could also increase, depending on whether the system is nutrient or grazer-limited (Williamson, 1995). Many vertebrates, including amphibians, lay transparent eggs with little UV-B protection in bodies of water (Hansen et al., 2002). UV-B radiation can reduce hatching success, retard larval growth rates, and cause morphological abnormalities in amphibians, and increased exposure to UV-B has been implicated as a possible factor in (although not the sole cause of) recent worldwide declines in amphibian populations (Blaustein et al., 1996; Blaustein and Wake, 1995; Gardner, 2001). However, the danger of UV-B radiation in freshwater ecosystems remains site-specific; for example, 85% of the potential amphibian breeding sites sampled in the U.S. Pacific Northwest are protected by naturally-occurring levels of DOC in the water (Palen et al., 2002).

The threat of UV-B exposure may be magnified by other human-induced stresses. For example, CDOM derived from healthy, forested drainage basins is far more effective at attenuating UV-B than colorless DOC from other sources (Schindler et al., 1996), and deforestation decreases the input of CDOM to freshwaters. In addition, acidification of lakes causes a decline in DOC concentrations, and thus UV-B radiation may actually be responsible for many of the negative effects attributed to lake acidification (Schindler et al, 1996).

Anticipated and Observed Effects of Climate Change on Freshwater Ecosystems

Human activities within the last century have led to a dramatic rise in atmospheric concentrations of carbon dioxide and other gases that contribute to the greenhouse effect. Within the next century, carbon dioxide concentrations are expected to at rise to levels at least twice as high as those present in pre-industrial times, and global climate is expected to change in a number of ways as a result. This climate change will primarily affect freshwater ecosystems through changes in water temperature, quantity, and quality (Shuter and Meisner, 1992), as well as through changes in the timing and duration of flows. Some of the expected physical and biological effects of climate change will affect all freshwater ecosystems, while others are specific to rivers, lake or wetlands.

PHYSICAL EFFECTS COMMON TO ALL FRESHWATER ECOSYSTEMS

TEMPERATURE CHANGES

There is widespread consensus that the greenhouse effect will lead to a global rise in air temperature, with mean surface temperatures increasing 1.5 to 5.8 °C by the year 2100 (Houghton et al., 2001). Temperatures are expected to increase more at higher latitudes, and in many of these regions the effects of global warming have already been documented; in Canada, mean air temperatures, water temperatures, and evaporation have all increased in the past 20-30 years (Schindler, 2001), and ice cover durations over lakes and rivers have decreased over the entire northern hemisphere by almost 20 days since the mid-1800’s (Magnuson et al., 2000).

However, in most cases, the effects of global warming on air and water temperatures are likely to be far more complicated that a gradual increase in average temperatures. In many regions, daily minimum air temperatures have increased more than daily maximum temperatures, leading to a reduction in the diurnal temperature range (Easterling et al., 1997). Both observational studies and models of future climate change suggest that there will be more hot summer days and fewer cold waves (Easterling et al., 2000). Regionally, temperatures are likely to become more variable, and this increased variability (i.e. a 1 °C increase in the standard deviation of temperature) will lead to a far greater frequency of extreme temperature events than a similar change in the mean temperature would (Meehl et al., 2000).

Rising temperatures are generally expected to lead to an increase in glacial melting, although increased winter precipitation could compensate for ice loss in some areas (Arnell et al., 2001). Many simulations suggest that glacier melting will depend strongly on the rate of temperature change; for example, Oerlemans et al. (1998) predicted that in the absence of increased precipitation, a rise of 0.4 °C per decade would eliminate nearly all of their study glaciers by 2100, while a rise of 0.1 °C per decade would only lead to a 10-20% loss of glacier volume. Tropical glaciers may be especially sensitive to global warming, as the equilibrium line between ice accumulation and melting is more sensitive to changes in air temperature (due to the lack of seasonality in tropical temperatures), and because glacial melting is significant year-round (Kaser et al., 1996).

PRECIPITATION CHANGES

Since 1900, surface precipitation has generally increased in mid- and high-latitude areas, and decreased in the tropics and subtropics (Easterling et al., 2000). Current models of global climate change suggest that annual precipitation is likely to increase further in high and mid-latitudes and most equatorial regions, but decrease in the subtropics (Carter et al., 2000). However, predicted regional changes in precipitation are far less certain, and in many areas, the size of predicted precipitation changes due to global warming are small compared to those due to natural multi-decadal variability (Arnell et al., 2001).

However, even slight changes in average precipitation could lead to substantial increases in the variability of precipitation events; because the size of precipitation events is not normally distributed about the mean, a change in average precipitation will also cause a change in variability (Meehl et al., 2000). Climate change models predict that global warming will generally lead to more extreme events, such as heavy 1-day and multi-day precipitation (Easterling et al., 2000), and an increase in the frequency of extreme rainfall has been observed in the United States and the UK (Karl and Knight, 1998; Osborn et al., 2000). In addition, most countries that have experienced a significant increase or decrease in precipitation also experienced a disproportionate change in the amount of precipitation falling during extreme precipitation events (Easterling et al., 2000). Tropical storm intensity has increased, and is expected to increase further in some areas, such as southwest Asia (Anderson et al., 2002), but in other areas tropical storm frequency and intensity is expected to remain the same or decline (Easterling et al., 2000).

WATER QUANTITY AND FLOW CHANGES

Although precipitation is one of the main factors determining water availability and flow, other factors such as evaporation, soil moisture, groundwater recharge, and glacial and snowmelt are also critical. Evaporation is generally expected to increase due to increasing mean temperatures. However, the exact amount of evaporation that will occur at a given site is determined by a host of other factors, including soil characteristics, the amount of water available, vegetation cover, and plant transpiration (which is affected by temperature and atmospheric carbon dioxide concentrations; Arnell et al., 2001). Soil moisture will depend on soil characteristics and the magnitude of local precipitation changes, and soil infiltration and water-holding capacity will in turn determine the volume of run-off. For example, drier soils often shows reduced water infiltration, and less extreme freezing events can reduce water infiltration in limestone soils (Boix-Fayos et al., 1998); reduced water infiltration could lead to greater run-off and an increase in flooding events. Groundwater recharge is affected by both the amount of precipitation and the duration of the recharge season, as well as by evaporation and soil moisture (Arnell et al., 2001). Although climate change is likely to lead to some changes in groundwater recharge, freshwater ecosystems that primarily receive input from groundwater are likely to experience smaller changes in water temperature and quantity than those dominated by precipitation.

Water flows depend primarily on precipitation in tropical and arid regions. In tropical river systems, seasonal heavy rainfall events already surpass the natural infiltration rates of soil, leading to high sediment input and dangerous levels of pesticide runoff from agricultural lands (Pringle, 2000); increased extreme rainfall events in these areas could lead to further water quality problems. In higher latitude regions, temperature changes will affect water flow through changes in snowmelt and the form of falling precipitation. In large parts of eastern Europe, European Russia, central Canada, and California, a major shift in streamflow from spring to winter has already been observed, because elevated temperatures cause precipitation to fall as rain rather than snow (Dettinger and Cayan, 1995; Westmacott and Burn, 1997). Similarly, glacier-fed rivers, lakes and wetlands (in temperate and tropical regions) may experience increased flows due to glacial melting, even in the absence of increased precipitation (Arnell et al., 2001).

Many models predict that extreme water flow events such as floods are likely to increase, due to heavier individual rainfall events (Reynard et al, 1998) or increased overall precipitation (Panagoulia and Dimou, 1997). Some models also predict seasonal shifts in peak flooding seasons (Saelthun et al., 1998). Finally, models of climate change suggest that hydrological droughts should increase in frequency, but this has only been observed in some areas, such as Hungary and China (Easterling et al., 2000). The effects of climate change on low flow conditions appear to be sensitive to the storage capacity of the system; basins with little groundwater storage capacity may experience more frequent droughts because they do not benefit as much from winter groundwater recharge (Arnell et al., 2001).

WATER QUALITY CHANGES

The input of chemicals, sediments, organic matter, nutrients and pollutants to freshwater ecosystems are all likely to be affected by climate change. Both simulations and direct observations indicate that increased precipitation can increase water alkalinity via enhanced weathering and input of base cations to streams or lakes (Avila et al., 1996; Sommaruga-Wögrath et al., 1997). Intense storm events following prolonged dry periods can lead to increased flushing of sediments or nitrates into water bodies (Arnell et al., 2001).

Rising temperatures and changes in precipitation are likely to cause changes in the biomass, production, and composition of terrestrial communities surrounding lakes, rivers and wetlands (Carpenter et al., 1992). These changes may affect the supply of organic matter to freshwater systems, shading and light (including UV-B) penetration, as well as the characteristics of runoff entering the system (i.e. DOC and nutrient concentrations, sediment load). Rising temperatures and evaporation may cause an increase in fires in some regions (this has been documented already in Canada; Schindler et al., 1990). Fires in regions surrounding water bodies could lead to increased nutrient input (from burnt vegetation) leading to eutrophication, increased sediment load, reduced input of organic matter (DOC, woody debris), and reduced protection from winds.

Water pollution could be affected in many ways by climate change. Increased volatilization of pesticides, PCBs, and heavy metals in warm and warming regions with condensation elsewhere will lead to increased pollutant loads in water bodies at higher latitudes and altitudes. Increased runoff could lead to increased pollution from agricultural and urban sources, while decreased water levels could lead to concentration of pollutants from point sources and the atmosphere (Schindler, 2001). Increased glacial melt has been shown to increase the concentrations of pollutants in glacial-fed streams and lakes by releasing organic pesticides and PCBs deposited in glacial ice during earlier decades when pollution levels were higher (Blais et al., 2001).

INCREASING HUMAN WATER NEEDS AND EXTRACTION

Rising air temperatures and evaporation are likely to contribute to increasing human water use and water shortages. By far, the largest proportion of current and predicted future water use is for agricultural irrigation; in 1995, 67% of all water withdrawals and 79% of all water consumed worldwide was used for agriculture, whereas municipal, or domestic, use represents only about 9% of withdrawals (Arnell et al., 2001).

Based on predicted population increases and development scenarios, water withdrawals are expected to increase 23-49% over 1995 levels by the year 2025 (Raskin et al., 1997). Municipal water withdrawals may be offset by declining per capita usage in some countries, and increases in water usage by the industrial sector in developing countries (particularly in Asia and Latin America) may be partially ameliorated by greater industrial efficiency (Arnell et al., 2001). Agricultural water usage, on the other hand, is expected to increase due to higher evaporation rates and because larger areas of land are likely to be under cultivation (to feed an growing human population). Areas where agriculture accounts for a large portion of the GNP and water is already scarce (i.e. many African countries) are likely to be hardest hit by climate-associated water shortages (Smith and Lenhart, 1996). Alterations in the way that water is priced (i.e. reducing agricultural water subsidies) could play an important role in promoting more efficient water use and thus decreasing human water needs and extraction.

BIOLOGICAL CHANGES COMMON TO ALL FRESHWATER ECOSYSTEMS

EFFECTS ON PHYSIOLOGY AND LIFE HISTORY

Temperature change alone is known to affect a range of physiological processes and life history traits. Higher ambient temperatures increase the metabolic demands of many animals; for example, even at sub-lethal temperatures, warming would lead to a several-fold increase in the energy requirements of lake trout *(Salvelinus nomaycush)* (McDonald et al., 1996). The effects of higher metabolism on growth may depend on food availability. In zooplankton with an adequate food supply, increased temperatures lead to a dramatic rise in feeding, assimilation, growth, and reproductive rates (Schindler, 1968), and local species richness can increase as a result (Stemberger et al., 1996). Increased temperatures can also lead to a rise in the frequency of toxic algal outbreaks, and in their toxicity to other animals (Hallengraeff, 1993).

Temperature affects body size in many aquatic animals; increased rearing temperature causes a reduction in body size at a given developmental stage in over 90% of cold-blooded, aquatic animals studied (Atkinson, 1995). Temperature also determines the sex of offspring in the American alligator *(Alligator mississippiensis)* and several groups of turtles (Conover, 1984). In one population of painted turtles *(Chrysemys picta)*, offspring sex was shown to be highly correlated with mean July air temperatures; statistical analyses indicate that a 2 °C rise in air temperatures would drastically skew sex ratios, and a 4 °C rise would virtually eliminate all males from the population (Janzen, 1994). The phenology, or timing of life history events, is also affected by ambient temperature; for example, breeding migrations and spawning dates have begun to occur earlier in several species of amphibians as a result of climate warming (Beebee, 1995).

Life history traits are intricately linked with water quantity and seasonal flow in numerous aquatic animals. In tropical rivers, many fish undergo feeding and spawning migrations of several thousand kilometers that are dependent on predictable, seasonal flooding events (Junk, 2002), and extreme flow levels may be necessary for maintaining populations of a number of other species (Poff and Ward, 1989; Poff et al., 1997). On the other hand, intense flooding can scour streambeds, displacing organic matter, bottom-feeding organisms, and small fish fry, and substantial increases in flood frequency could cause a shift in species composition, possibly eliminating many species (Poff et al., 1997).

The timing and duration of the breeding season in wading birds that frequent freshwater wetlands is strongly tied to water levels (Butler and Vennesland, 2000), and could be affected by either increases or decreases in precipitation and water table levels. Frogs are particularly sensitive to decreased precipitation; low precipitation in Puerto Rico has been correlated with drastic declines in frog populations (Stewart, 1995), and the extinctions of four frog species in Costa Rican cloud forests have been linked to a series of severe population declines following extreme El Niño-associated droughts (Pounds et al., 1999). Some inhabitants of seasonal wetlands, such as fairy shrimp, are entirely dependent on precipitation-filled, ephemeral vernal pools to complete short, highly-specialized life cycles (Eriksen and Belk, 1999).

EFFECTS ON COMMUNITY COMPOSITION AND DYNAMICS

Climate warming is likely to alter the composition of many communities, as different species will have different thermal tolerances and interactions between species may intensify as a result of reduced resources and habitat availability. In algae, thermal tolerance can affect the outcome of competition for nutrients and alter community composition (Rhee and Gotham, 1981). Where water levels or the size of suitable thermal habitats decrease, biotic interactions (including human overexploitation) may intensify as a result of increased densities of aquatic animals. For example, caddisfly competition in streams that is usually limited to the summer season persisted year-round in years when drought eliminated normal winter density reductions (Feminella and Resh, 1990). Climate change may affect motile vs. non-motile species differently, leading to differences in species distribution; in the marine intertidal zone, vertical distribution of marine invertebrates is tightly correlated with temperature, whereas the distribution of motile species is not (Huey et al., 2002).

In temperate or high-elevation tropical systems, where water temperatures are currently cool, climate warming is expected to facilitate the spread and establishment of non-natives, especially those from warmer climates (Stachowicz et al., 2002). Once thermal barriers to invasion are removed, native species may be displaced by invaders with a competitive advantage, such as warmwater, omnivorous fish with fast life cycles (Carpenter et al., 1992). Natural high flows help minimize the success of non-native fish by removing species that are poorly adapted to dynamic river environments; the restriction of flooding by reservoirs has already helped facilitate the proliferation of exotic fish in many river systems (Baron et al., 2003). Reduced flooding due to climate change could similarly allow non-native fish to become established in areas where the current flow regime would otherwise exclude them. Several species of exotic marine invertebrates may benefit from warming temperatures at higher latitudes, as earlier recruitment in warmer years helps them gain a competitive advantage over native species (Stachowicz et al., 2002); a similar breeding strategy may be present in invasive freshwater invertebrates.

Migration may be the only option for many animals that cannot adapt to increasing temperatures. During past eras of climate change, most plants and animals displayed range shifts, rather than morphological change, in response to changing environmental conditions (Noss, 2001). However, the rate of warming expected in the next 100 years is over ten times higher than warming after the last ice age (De Groot and Ketner, 1994), and it is unknown whether plants and animals will be able to migrate quickly enough to keep up with climate change (Malcolm and Markham, 2000). Meta-analyses indicate that hundreds of plant and animal populations have already shown highly significant, non-random changes in range boundaries, in the direction expected to result from climate change (Parmesan and Yohe, 2003), and temperature is known to the affect geographic distribution of fish, diatoms, and other aquatic animals (Meisner and Shuter, 1992; Reynolds, 1984).

Unfortunately, the ability of many freshwater species to migrate is restricted by bodies of water that preclude migration or that don’t allow movement in the correct direction. Migration is impossible from many isolated lakes and wetlands, and numerous major river systems run from east to west, precluding latitudinal migration. In the southwest and southern Great Plains (USA), nearly all major river systems run from east to west, and these systems contain some of the hottest free-flowing water on Earth. Many native species in these areas are already living near their thermal tolerance limits, and the combination of increased warming and the lack of northern migration routes could cause extinctions of up to 20 species of endemic fish (Matthews and Zimmerman, 1990).

Even where migration is possible, it is unlikely that entire communities and ecosystems will be transplanted intact. Species differ in their levels of tolerance of environmental change, and in their abilities to adapt or migrate in response. Differences in sensitivity to temperature change are likely to alter community dynamics. For example, communities composed of several species of *Drosophila* that were allowed to migrate between different thermal habitats did not simply shift to new zones when temperatures were increased to simulate global warming (Davis et al., 1998). Interactions between species and with parasites introduced to the system altered the relative abundance of species in different thermal zones in ways that were non-intuitive and difficult to predict. Thus, simple “climate mapping”, or assuming that species range boundaries will shift smoothly with changes in temperature, oversimplifies the effects that climate change may have on communities.

EFFECTS OF CLIMATE CHANGE ON LAKES

PHYSICAL EFFECTS ON LAKES

Increased mean surface temperatures are likely to lead to increased water temperatures and evaporation in many lakes, in both temperate and tropical areas (Schindler, 2001; Zinyowera et al., 1998). If precipitation does not increase enough to compensate, this could lead to reductions in outflow and/or lake volume. Important spawning and rearing habitat near the edges of lakes (Tyedmers and Ward, 2001) would be lost if lake levels declined, and lake characteristics based on water outflow could change dramatically. Lakes that currently supply outflow to downstream systems may become endorheic (with no outflow), and endorheic freshwater lakes may become saline (Schindler, 2001). The African Great Lakes are particularly sensitive to climatic effects on outflow, as current outflow is small (i.e. only 6% of water input to Lake Tanganyika leaves as riverine outflow), and even minor declines in precipitation (10-20%) are expected to completely close these basins (Bootsma and Hecky, 1993).

Changes in mean air temperatures have been shown to increase water temperatures in both temperate and tropical lakes, resulting in a range of physical and biological effects. Water temperatures in Lake Tanganyika, a deep tropical lake in East Africa, have risen by 0.2 °C at the lake bottom and 0.9 °C at 100 meters since 1913 (Verburg et al., 2003). Although the resulting change in the temperature gradient is relatively small, the vertical gradient in water density (which depends on temperature) has tripled. This sharpened density gradient has reduced annual mixing, which normally supplies nutrients from decomposition on the lake bottom to surface waters. The lack of nutrients in the upper layers of the lake has led to a 70% reduction in primary productivity since 1975 and an increase in water clarity and light penetration (Verburg et al., 2003).

Temperate lakes display sharper thermal gradients and larger seasonal changes in water temperature than tropical lakes. Lakes at higher latitudes and altitudes currently experience seasonal thermal stratification, in which they are covered by ice in the winter (with cool, relatively constant temperatures below), and develop a thermal gradient (thermocline) in the summer as surface waters warm up. Increased ambient temperatures have led to an earlier onset of thermal stratification, longer ice-free periods, and deeper thermoclines (thus smaller bottom layers) in many temperate lakes (Schindler et al., 1990). In addition, longer periods of thermal stratification with little vertical mixing results in reduced oxygen concentrations near the bottom of lakes.

Lake chemistry may be affected by climate change in a number of ways. Drought and decreased groundwater flow may make some lakes more susceptible to acidification, as groundwater often contains acid-neutralizing chemicals important to lake buffering (Schindler, 2001). However, the overall pH and chemical balance of lakes may be affected by temperature and precipitation changes in ways that are site-specific and difficult to predict; a number of European alpine lakes that experienced a 1 °C increase in temperature over 10 years actually displayed an increase in pH, as well as trends in sulfate and nitrogen concentrations that were opposite to trends in atmospheric deposition of these compounds (Sommaruga-Wögrath et al., 1997). These changes are most likely explained by increased biological activity and enhanced weathering of surrounding substrates due to high precipitation.

Concentrations of colored DOC, which derives from surrounding vegetation and provides the greatest protection from UV-B radiation, has been shown to decline in high latitude lakes as a result of climate warming, reduced streamflows, and lowered water tables (Schindler et al., 1996). Lake acidification (which can itself be caused by climate change) will exacerbate this effect (Schindler et al., 1996). Conversely, increased precipitation should increase DOC concentrations, and vegetation shifts due to climate warming (such as altitudinal shifts in the tree line) may protect higher lakes by increasing DOC concentrations (Palen et al., 2002).

Finally, the physical effects of climate change on temperate lakes can be synergistic and complex. For example, lakes at the Experimental Lakes Area in the boreal forests of northwestern Ontario experienced an average increase in air temperature of 2 °C over 20 years (Schindler et al., 1990). This resulted in an increase in mean and maximum water temperatures, an ice-free period that was 20 days longer, and an 30% increase in evaporation. The area experienced below-average precipitation and reduced runoff into lakes. More frequent fires caused a rise in nutrient input, which combined with higher temperatures to increase phytoplankton abundance and diversity. Reduced terrestrial input to the lakes (because of fires and clear-cutting) reduced DOC concentrations and led to clearer lakes with less protection against UV-B radiation. Increased solar penetration and higher winds (due to the loss of trees) led to deeper thermoclines and a reduction in bottom habitat (Schindler et al., 1990).

BIOLOGICAL EFFECTS ON LAKES

Although the effects of climate warming on the biotic communities of tropical lakes has received little attention, large decreases in primary productivity due to climate warming (Verburg et al., 2003) are likely to have a significant impact on the rest of the food chain. In temperate lakes, the radical physical changes in lake chemistry and thermal stratification resulting from climate change can have a range of effects on biological communities. As temperate freshwater fisheries play an important economic role in many countries, a great deal of research into the effects of climate change on freshwater ecosystems has focused on temperate freshwater fish.

Temperate fish can be divided into three major guilds according to their “fundamental thermal niche”, or the temperature at which the choose to spend most of their time and at which they experience optimal growth, activity levels and swimming performance (Shuter and Meisner, 1992): coldwater fish such as salmon and trout (with an optimal temperature around 15 °C), coolwater fish such as perch (with an optimal temperature around 24 °C), and warmwater fish such as carp and catfish (with an optimal temperature around 28 °C).

Seasonal thermal stratification in lakes allows all three guilds to co-exist because of variations in life history strategies (Shuter and Meisner, 1992). In the winter, cool- and warmwater fish are inactive, and cold water fish are active but experience little growth. Temperatures are optimal and growth rates highest for coldwater fish in the spring and fall, while the summer is optimal for cool- and warmwater fish. During the summer, the upper layers of the lake become too hot for coldwater fish and they are restricted to the cooler bottom layer, where oxygen levels are low and competition for food is fierce.

Because climate warming leads to longer periods of thermal stratification, coldwater fish will be restricted to these bottom layers for longer periods of time, and deeper thermoclines brought about by climate change may reduce the area of bottom layers and further intensify competition for food (Shuter and Meisner, 1992). Climate change may also lead to shorter spring and fall seasons, when temperatures are optimal for coldwater fish. An overall rise in water temperatures will lead to increased metabolic demands, but coldwater fish will generally have reduced access to prey. Warmer temperatures may make winter slightly more favorable, but not enough to compensate for losses during other seasons (Shuter and Meisner, 1992). Overall, coldwater fish are likely to experience decreased growth rates and increased heat mortality (Tyedmers and Ward, 2001).

Some populations may be able to adapt to thermal changes; a shift in preferred temperature from 15 °C to 20-21 °C has been observed in one population of lake trout (Salvelinus namaycush) in Canada (Sellers et al., 1998), and other populations of coldwater fish display adaptive behavior, in which they spend most of their time in coldwater refugia but make occasional feeding forays into warmer water (Snucins and Gunn, 1995). However, the majority of coldwater fish populations are likely to experience range shifts, with contractions near the low-latitude and low-altitude limits of their current range, and expansions to higher latitudes if migration is possible. Suitable habitat for coldwater fish in the continental United States may decline by as much as 50% (Eaton and Scheller, 1996), and range contractions of coldwater species could eliminate some of the world's most valuable fisheries (Schindler et al., 1990).

Conversely, however, climate warming and changes in thermal stratification may have positive effects on cool- and warmwater species, reducing winter kills, lengthening the growth season, and increasing available habitat, both locally and regionally, if poleward migration is possible (Shuter and Meisner, 1992). If lakes are not nutrient-limited, productivity is likely to increase (due to increased primary productivity and growth rates); overall fish catch may increase, but there are likely to be changes in the relative abundance of fish species (Tyedmers and Ward, 2001).

Finally, although most studies examine the effects of climate change on only a few species of fish, negative effects on one species can have an impact on the entire community. For example, a summer kill of planktivorous herring in a Wisconsin (USA) lake reduced predation on zooplankton by 50%, which led to an increase in large zooplankton and intensified zooplankton grazing, causing a substantial reduction in phytoplankton abundance (Kitchell, 1992). In addition, while many studies concerning lake water level focus on the effects of declining levels, rising water levels due to regional or seasonal increases in precipitation could also have negative impacts. In Lake Baikal, for example, the construction of a dam along with increased precipitation led to a 1.5 m rise in the level of the lake, and a subsequent decline in fish biodiversity and production (Izrael et al., 1992).

EFFECTS OF CLIMATE CHANGE ON RIVERS

The effects of climate change on rivers are likely to vary widely depending on latitude. Temperate rivers, like temperate lakes, will be affected primarily by temperature changes, while changes in precipitation timing and quantity could have dramatic effects on tropical rivers.

PHYSICAL EFFECTS ON RIVERS

Increases in air temperature will strongly influence water temperature in many rivers (particularly smaller rivers and streams) because the surface to volume ratio of rivers is high (Tyedmers and Ward, 2001). Increasing air temperatures will result in warmer water throughout rivers, from the headwaters to the mouth, as well as reduced oxygen levels. Rivers that are fed primarily by groundwater will be buffered against increasing seasonal variability in temperature (as groundwater temperature generally equals average annual air temperature), and may serve as thermal refugia in some areas, supplying relatively cooler water during hot seasons and warmer water during cold seasons.

Temperate rivers experience seasonal thermal cycles similar to temperate lakes, with uniform cold temperatures in winter (sometimes accompanied by ice cover) and longitudinally stratified temperatures in summer, with lower temperatures at groundwater-fed headwaters and higher temperatures downstream (Shuter and Meisner, 1992). High latitude rivers are already experiencing shorter periods of ice cover and earlier ice break-up (Magnuson et al., 2000), and many of the beneficial functions of ice jams (river scouring, changes in river channel morphology, flooding of riverine wetlands) may be compromised.

Flow regime is a critical component of river ecosystems. Mean flow may increase or decrease depending on changes in average precipitation, evaporation, soil moisture, and groundwater recharge, but seasonal shifts in flow may be more significant to freshwater ecosystems (Carpenter et al., 1992). Many rivers will experience altered timing or duration of high and low flows due to changes in seasonal variability of precipitation, frequency of extreme precipitation events, and timing of snowmelt. Spring snowmelts are likely to occur earlier due to warming, and winter flows are likely to increase in areas where winter precipitation falls as rain instead of snow. A shift in peak flows from spring/summer to winter will reduce the cooling effect of snowmelt on summer river temperatures (Tyedmers and Ward, 2001).

Where precipitation increases, stream flows may increase in volume and floods may become more frequent. Extreme flooding events and landslides could remove important woody debris from rivers and destabilize river channels (Carpenter et al., 1992). Where precipitation decreases, stream flow volume may also decrease, and reductions in runoff will lower the concentrations of DOC and organic matter in rivers. Increased evaporation could also lead to reduced streamflow, even in the absence of precipitation changes. Summer and ephemeral streams in arid regions (which provide critical habitat for many animals) are more vulnerable to drying up. A reduction in natural flooding events could eliminate many of the beneficial physical effects of seasonal flooding, such as creating floodplain habitat, displacing exotic plants, and determining river channel form.

BIOLOGICAL EFFECTS ON RIVERS

In tropical rivers, variations in air and water temperature are generally small, and water temperature is mainly regulated by shade and rainfall, rather than by cool groundwater refugia. The rainy and dry seasons of the tropics lead to large, predictable seasonal variations in precipitation and annual flooding of adjacent grasslands and forests, which provide abundant food and breeding grounds for fish. Thus, the life histories of tropical river fish are more strongly affected by changes in water level than by changes in temperature (Meisner and Shuter, 1992). As the rainy season draws to a close and floodplains dry up, members of the “whitefish” guild, who are sensitive to reduced oxygen levels, retreat to the main river channel with the receding floodwaters. “Blackfishes”, who are more tolerant of or adapted to low oxygen levels, remain in marginal floodplain habitats that become disconnected from the river and may even dry up completely (Welcomme, 1979). Some of these species, such as the lungfishes, are able to aestivate (cocoon in the mud) and breathe air when their water supply evaporates.

Climate change may affect the both the timing and extent of flooding in tropical rivers, although these effects are currently difficult to predict and will vary regionally. According to some estimates, the tropics will experience the smallest change in temperature, but the largest changes in precipitation, with rainfall becoming more variable, both within and between years (Houghton et al., 2001). Changes in floodplain dynamics will directly affect fish populations and fisheries yield, as growth rates and overall fish catch is correlated with the area of flooded land (Meisner and Shuter, 1992; Welcomme and Hagborg, 1977).

Fish communities in temperate rivers will experience effects similar to those in temperate lakes. Coldwater fish that are restricted to cool refugia at headwaters during the summer will experience increased competition, reduced growth, and possible range shifts (Shuter and Meisner, 1992). Warmer water and decreased oxygen content at headwaters may have negative impacts on eggs and larvae often placed there (Carpenter et al., 1992). Diadromous stocks that migrate long distances during the peak of summer may experience higher rates of pre-spawning mortality because of increased metabolic needs and disease outbreaks (Tyedmers and Ward, 2001). Even in stocks that do not perform summer migrations (such as the Adams River sockeye salmon) climate change is likely to result in a net population decline due to reduced juvenile emergence, growth and survival (Henderson et al., 1992). Some invertebrates in northern rivers require a prolonged period of exposure to nearly 0ºC water, followed by spring warming, in order for eggs to hatch. The release of warmer water in the winter from dams has resulted in massive local extinctions of invertebrates for tens of kilometers downstream (Lehmkuhl, 1974), and overall river warming would be expected to have a similar effect.

EFFECTS OF CLIMATE CHANGE ON WETLANDS

PHYSICAL EFFECTS ON WETLANDS

Increased air temperatures are likely to have a drying effect on many wetlands, unless increased precipitation compensates for evaporation. Shallow and ephemeral habitats, such as depressional wetlands (with no channelized flow in or out) or wetlands in arid areas could be lost entirely, especially if precipitation declines and groundwater is extracted for human needs (Gitay et al., 2001). Overall, a drier climate is likely to lead to contractions and loss of wetland habitat, as well as increased habitat fragmentation. Conversely, increased precipitation could lead to flooding, expansion and deepening of wetland habitat, and increased connectivity. However, increased precipitation or extreme flooding may also lead to an increased input of sediment and pollutants, and could destroy some wetlands if vegetation or other important habitat features are completely submerged.

Arctic and subarctic bogs located over permafrost could suffer dramatic changes in hydrological regime if rising temperatures lead to permafrost melting and wetland drainage. Increased decomposition in thawed northern peat bogs, as well as the increased risk of catastrophic fires due to drier peat, could release large amounts of carbon dioxide into the atmosphere, contributing to further global warming (Gorham, 1991)

Coastal freshwater wetlands are particularly sensitive to extreme high tides resulting from an increase in storm frequency or magnitude; these high tides can carry salts inland to salt-intolerant vegetation and soils, and could lead to the displacement of freshwater flora and fauna by salt-tolerant species (Michener et al., 1997). Rising sea levels could destroy coastal freshwater wetland communities as saline water invades, especially if these communities cannot shift inland due to development or dikes (Tyedmers and Ward, 2001). Salt water inundation of coastal freshwater wetlands is expected to cause significant loss of wetland habitat in Australia and elsewhere (Gitay et al., 2001).

BIOLOGICAL EFFECTS ON WETLANDS

Ephemeral, depressional wetlands, especially those in arid areas, often harbor rare species that would be lost if these areas dry up. For example, several endemic species of fairy shrimp in California (USA) that are already severely threatened by habitat loss (Belk and Fugate, 2000) could disappear if reduced precipitation and increased evaporation eliminates their shallow, vernal pool habitats.

Small, temporary wetlands are the most numerous types of wetlands in many landscapes, and are often used by more species than permanent ponds (Gibbs, 1993; Semlitsch et al., 1996; Semlitsch and Bodie, 1998). The drying and loss of wetlands would reduce not only the number and size of available ponds, but also increase inter-pond distance (Gibbs, 1993; Semlitsch and Bodie, 1998), lowering the chances of amphibian recolonization, since adult frogs are generally only capable of traveling 200-300 m (Sjogren, 1991; Skelly et al., 1999). Drying and loss of wetlands would also reduce habitat connectivity on a regional scale, endangering migrating birds that depend upon a network of wetlands along their migration route (National Research Council, 1995).

Wetlands in areas with increased precipitation might suffer fewer negative effects, and may even benefit from increased wetland area and connectivity. However, some rare species that are adapted to drier, ephemeral wetlands may not be able to compete with invading species adapted to wetter habitats (National Research Council, 1995), and wading birds that require shallow water to feed may experience reduced access to feeding areas (Butler and Vennesland, 2000). Wetter, more permanent wetlands would support more fish, which prey on vulnerable tadpoles and invertebrates that usually inhabit seasonal wetlands with less predation pressure (Semlitsch and Bodie, 1998).

General Considerations for Designing and Managing Freshwater Reserves to Withstand Climate Change

The likelihood that individual species or communities will be able to persist in the face of global climate change depends to a large degree on how resistant (able to withstand change) and resilient (able to recover from change) they are (Noss, 2001). Because there is still a great deal of uncertainty associated with climate change predictions (especially predictions of precipitation changes), and interactions between physical and biological features of freshwater ecosystems can be complex and non-intuitive, focusing on increasing system resistance and resilience is a far better approach than trying to plan for a specific set of predicted changes. Many common considerations in designing and managing ecosystem reserves, such as preserving biodiversity and minimizing outside stresses, will also help increase the resistance and resilience of communities to climate change. Additional considerations that are unique to freshwater ecosystems will become increasingly important in buffering systems against growing climatic and water extraction pressures.

PRESERVE HABITAT HETEROGENEITY AND BIODIVERSITY

Both species and habitat diversity increase resistance and resilience to climate change, as diversity provides a greater range of stress tolerances and adaptive options (Chapin et al., 1997). Diverse communities that have redundant species within functional groups should be more resistant to climate change because there are likely to be differences in environmental sensitivity among members within each group; functional group richness also appears to increase resistance to environmental change (Noss, 2001). High biodiversity areas may also become important as sources for re-colonizing damaged sites or colonizing new ones as the effects of climate change become more severe (De Groot and Ketner, 1994).

In aquatic systems, high biodiversity is often found in older or isolated habitats, in sinkholes, caves or underground habitats, and in areas with high habitat heterogeneity—especially dynamic habitats with seasonal changes in water level (i.e. river floodplains of seasonal wetlands; Abell et al., 2002). Many of these areas also harbor rare species, such as endemic species that have evolved in and remain restricted to a particular habitat (i.e. communities of endemic cichlid fish in African rift lakes), relict species that were restricted to isolated habitats after previous range contractions (i.e. cold stenothermic fish that were isolated to high latitude lakes after the last ice age), or species that are highly adapted to unusual environments (i.e. cave-dwelling fish and invertebrates). In protecting some of these high biodiversity sites, rare or vulnerable species may also be protected (often a primary consideration in reserve design). Protecting rare species, especially those that are charismatic, can assist in drawing public attention and funding to conservation efforts, but strategies aimed solely at protecting one species may detract from the more desirable goals of protecting ecosystem function (Junk, 2002) and increasing resistance and resilience to climate change.

Areas where natural physical barriers separate biota (i.e. impassable waterfalls), and transition zones between different habitats or ecosystems may also harbor high biodiversity (Abell et al., 2002). Protecting transitional zones has the added benefit of accommodating possible range shifts due to climate change, and can help preserve diverse habitat types. Protecting a variety of potential habitats may help increase resistance and resilience in vulnerable species; for example, protecting an array of natural ponds with a wide range of sizes and hydroperiods will help ensure that amphibians have access to suitable breeding sites regardless of climatic variation (Semlitsch, 2002). If possible, replicate sites of a particular habitat type should be protected to safeguard against the complete loss of critical species or communities if one site is damaged beyond repair by an extreme climatic event (Markham and Malcolm, 1996; Roberts et al., 2003).

Although diverse communities may be more resistant and resilient to climate change, it is important that high biodiversity not be used as the sole criterion in selecting sites for conservation. An equally important goal is protecting communities that perform valuable “ecosystem services” (Kareiva and Marvier, 2003), such as relatively low-diversity wetlands that provide flood protection, water filtration and other services that are likely to become increasingly important as climate variability and extreme events increase. Protecting sites with lower biodiversity also helps maintain functioning ecosystems over broader regions of the globe, and preserves distinct evolutionary lineages that can provide fuel for future evolutionary innovation (Kareiva and Marvier, 2003). Preserving sites harboring high biodiversity can still be a valid conservation goal in many areas, and diversity indices can be extremely useful in choosing between different sites *within* a habitat type (rather than using biodiversity indices as an absolute guide in choosing *between* different habitats). However, considering ecosystem services and other potentially beneficial features of low diversity sites is an important aspect of planning for climate change.

PROTECT PHYSICAL FEATURES RATHER THAN INDIVIDUAL SPECIES

Aquatic ecosystems differ from many other ecosystems in that they are usually governed by “bottom-up” rather than “top-down” dynamics—in other words, much of ecosystem function is determined by basic physical features such as water flow, channel morphology, and nutrient balance, rather than by species assemblages (Moss, 2000). Protecting flow patterns, water quality, and water quantity will go a long way towards protecting biodiversity in freshwater habitats (Abell et al., 2002), whereas conservation efforts that focus solely on preserving particular species or groups of species without considering wider physical features of the system may be doomed to failure. In many cases, the function of a species in a freshwater ecosystem is actually more important than its identity; for example, plants are essential components of some aquatic habitats (i.e. floodplain vegetation and aquatic plants in shallow lakes), but the exact species of plant may be less important than the physical features it provides (Moss, 2000).

The physical features of rivers, lakes and wetlands are expected to undergo a number of changes as a result of climate warming and precipitation variability. Removing barriers to water flow, maintaining healthy, forested river basins, and reducing the input of nutrients and toxic substances will increase the likelihood that freshwater ecosystems will be able to adjust to climate change. For example, removing levees and other barriers to the lateral expansion of rivers could prevent the loss of critical edge habitats and the species that depend on them, by allowing new floodplains to be established if average river flows increase or extreme precipitation events become common.

PRESERVE HABITAT CONNECTIVITY TO ALLOW ACCESS TO MIGRATION ROUTES AND THERMAL REFUGIA

Connectivity is an important feature of many freshwater ecosystems, as it can help preserve flow regimes, promote ecological integrity, and allow migrating animals to move between different habitats at various life history stages. Connectivity is important not only between different freshwater habitats (i.e. between rivers, lakes, and wetlands), but also along the length of rivers, and between freshwater habitats and subterranean systems or groundwater sources (Abell et al., 2002). Maintaining connectivity will become even more important in some areas as the effects of climate change increase, because connectivity may provide animals with access to thermal refugia, or allow them to migrate to more suitable habitats.

Although some species may be able to adapt to climate change in their current habitats, warmer waters will force other species to move into cool, thermal refugia, where temperatures remain below their thermal tolerance limits and metabolic demands are lower. Many species (i.e. coldwater fish) already rely on thermal refugia at certain times of the year, and these species are likely to become even more dependent on these refuges for year-round survival. Headwaters of rivers and any areas where temperature-buffered groundwater enters a system should be protected, and vegetation over bodies of water should be maintained to provide cooling. Maintaining or increasing connectivity between cool refugia and the rest of an ecosystem should be considered, as this may help provide additional species with access to these areas.

In many cases, where thermal refugia do not exist or other climate-related changes make an animal’s environment uninhabitable, the only option may be migration to more suitable habitats. Because migration of aquatic animals is already severely limited by the direction of connections between water bodies, preserving or improving the possible migration corridors that do exist may be important in some cases. Rivers and other aquatic ecosystems that allow movement in the north-south direction, as well as freshwater systems spanning altitudinal gradients are particularly valuable migration corridors. In addition, areas near the current range limit of species should be protected (in the direction of expected migration; Markham and Malcolm, 1996). Consideration should be given to protecting areas that are currently not of interest, but that species are likely to migrate into, or marginal habitats (i.e. areas that are too cold for most species) that are likely to be improved by climate warming.

Although maintaining connectivity between habitats and ecosystems may have a number of beneficial effects in terms of allowing native species to adapt to climate change, the increasing threat of invasion by exotic species makes connectivity in aquatic systems a more complicated issue. Because invasion by exotic species can potentially have devastating effects on ecosystems, the risks of invasion should be weighed on a site-by-site basis against the vulnerability of native populations to climate change, and the necessity and feasibility of migration to other habitats. In cases where species are particularly sensitive to temperature changes and are likely to experience local extinctions due to thermal stress, and where migration corridors leading to more suitable habitats exist, the benefits of connectivity may outweigh the risks of invasion. In other areas, however, the risks of invasion may be so severe that allowing one sensitive species to be lost would be preferable to endangering the entire community.

PROTECT SITES FROM HUMAN PRESSURES AND EXOTIC SPECIES

Protecting reserves from outside stresses (particularly stresses that tend to reduce diversity) will become increasingly important as local climates become more variable, because stressed systems display reduced resistance and resilience to change (Noss, 2001). Human stresses, such as overexploitation and poor land use practices, should be reduced as much as possible. In choosing between a group of candidate sites for new reserves, it may sometimes be advisable to avoid sites that are already severely degraded or likely to be subject to intense human pressure (Roberts et al., 2003), although restoration of some ecosystem functions is possible.

It is also critical to increase efforts to prevent access of invading species to reserves, and to eliminate or control harmful non-native species already present. Many systems are likely to become more vulnerable to invasions, as thermal barriers that previously excluded invaders will be removed, and communities that are already stressed by climate change are invaded by warmer-adapted species (Carpenter et al., 1992; Schindler, 2001). Even in sites that are well-protected from all other stresses, non-native species can wreak havoc on biological communities; for example, native fish populations in the pristine Blindekloof River in South Africa have been nearly devastated by invading largemouth bass (Skelton et al., 1995).

Unfortunately, in some systems, preventing access of motile, invasive species may conflict with the goal of maintaining connectivity to allow seasonal or climate-induced migrations. Isolating vulnerable habitats from other freshwater ecosystems may be feasible in some cases, but building barriers that disrupt flow to prevent access of exotic species may do more harm than good. In cases where the risk of invasion is relatively low and migration is important to resident species, maintaining current levels of connectivity while enacting careful monitoring of ecosystems (to allow early, aggressive management responses to invaders) may be a suitable approach.

MANAGE ENTIRE WATERSHEDS AND REGULATE EXTRACTIVE WATER USE

Freshwater ecosystems are intricately connected to their drainage basins, and downstream or lowland rivers, lakes and wetlands can be extremely sensitive to distant, upstream disturbances. Deforestation, agriculture, and other pressures on terrestrial communities that drain into water bodies can alter the quality, quantity, and temperature of water in all freshwater systems downstream of the disturbance.

Because freshwater ecosystems are so intricately connected to one another and to the terrestrial systems that surround them, common reserve considerations such as reserve size and buffer zones are less applicable. For example, Kruger National Park in South Africa is a very large reserve that protects significant downstream portions of the rivers that flow though it; however, populations of several species of fish that are protected within the park have declined as a result of upstream activities outside of the reserve (Skelton et al., 1995). Although reserve size and buffer zones may be applicable in limited, small-scale cases (i.e. no-take fishing areas and buffer zones to protect seasonal breeding grounds in Lake Tanganyika; Abell et al., 2002), protected habitat patches will generally provide only short-term solutions. Freshwater reserves will not be secure unless upstream threats are removed by managing (or in rare cases protecting) the entire drainage basin (Moss, 2000).

Managing entire watersheds rather than simply protecting aquatic elements or habitat patches will become increasingly important as the effects of climate change intensify and are magnified by interactions with human stresses. Human population growth is likely to lead to increased deforestation, agriculture, industrial development, and urbanization within watersheds. Models of land use and climate change in South Africa predict that abrupt, future changes in local land use will have a far greater impact on freshwater hydrology than gradual effects of climate change (Schulze, 2000). Importantly, the stresses caused by these habitat alterations often exacerbate the effects of climate change. For example, deforestation near freshwater ecosystems eliminates cooling, vegetative cover over streams and reduces the input of large, woody debris. The loss of shading further increases water temperatures that are already rising due to global warming, and the lack of large, woody debris eliminate in-stream refugia. In addition, the loss of forest cover may lead to increased terrestrial and aquatic evaporation, and reduced soil moisture and water infiltration. During heavy precipitation events, streams already experiencing unusually high flows would receive increased runoff due to low soil infiltration, as well as increased inputs of sediments and pollutants.

Perhaps even more important than projected increases in habitat alteration are the dramatic projected increases in human water extraction. Population growth will lead to increased water extraction for agricultural irrigation, direct consumption, and industrial development. Increasing human water extraction is likely to be one of the primary stresses on freshwater ecosystems in coming years, and is expected to greatly outweigh global warming in affecting water supply through at least 2025 (Vörösmarty et al., 2000). Increasing human water demands on freshwater ecosystems that are particularly sensitive to climate change are likely to lead to more frequent water shortages and conflicts over water use. For example, Egypt relies on the Nile river basin for over 95% of its national water budget. Based on one scenario of global climate change, models predict that a modest decline in precipitation (20%) and rise in air temperature (4ºC) would nearly halt flow of the Nile (98% flow reduction; Strzepek et al., 1996), and human demands would be likely to outweigh ecosystem needs in determining the fate of the remaining water. Clearly, management schemes aimed at protecting freshwater ecosystems from climate change must take human water needs into account, and attempt to manage the timing and magnitude of water extractions.

RESTORE DEGRADED SITES AND CONSIDER PERFORMING ACTIVE INTERVENTIONS IN RESPONSE TO CLIMATE-INDUCED THREATS

Because freshwater ecosystems are strongly influenced by activities in the surrounding landscape and increasing amounts of water are being extracted for human needs, most drainage basins have already experienced some degree of degradation. Restoration of degraded sites holds great promise for freshwater ecosystems, in terms of both improving the ecological integrity of damaged systems (thus increasing resistance and resilience to future change), and providing tools to smooth the recovery of ecosystems from future damage caused by climate change (McCarty and Zedler, 2002). Successful restoration techniques include neutralizing acidified lakes and rivers by applying lime (Schindler, 1997; Appleberg, 1998), and restoring the hydrology of wetlands by removing impediments to flow (Gilbert and Anderson, 1998). Exotic species such as predatory game fish have been successfully removed from some freshwater systems (McNaught et al., 1999), but in many cases, controlling aquatic community composition has proven difficult (McCarty and Zedler, 2002). All of the ecosystem problems mentioned above could potentially be caused or exacerbated by climate change (i.e. lake acidification, changes in hydrology, invasions of exotic species), and lessons learned from current restoration projects can help guide future responses to climate change. However, it is important to keep in mind that restoration efforts are unlikely to be successful after certain thresholds of damage have been crossed, such as the collapse of fisheries or permanent cultural eutrophication of lakes (McCarty and Zedler, 2002).

Rather than attempting to restore systems after they have already been degraded by climate change, it may be possible to perform active interventions to ameliorate the effects of climate change or to directly protect vulnerable species. Adding to water tables below drying wetlands or draining excess water from flooded habitats has been proposed (De Groot and Ketner, 1994), and inter-basin transfers of water have been suggested as a possible adaptation to climate change in Africa and South America (Pringle, 2000; Smith and Lenhart, 1996). However, interbasin transfers result in the intermixing of diverse faunal communities that were previously isolated, and may have unforeseen effects on native communities (Davies et al., 2000). In addition, tropical rivers and lakes (for which many of these projects have been proposed) are particularly vulnerable to the negative effects of interbasin transfers (Pringle, 2000). Flow conditions below dams could possibly be manipulated to relieve some of the negative effects of climate change, for example by releasing bursts of water to simulate flooding events (Middleton, 1999; Vaselaar, 1997) or releasing water from multiple reservoir layers to control temperature (McCarty and Zedler, 2002). However, although these actions may help alleviate some of the negative effects of climate change on systems already impacted by dams, erecting dams solely to control water flows would cause far more damage to the ecosystem than the original alteration in flow regime caused by climate change.

Some of the more controversial suggestions put forward for helping species and communities adapt to climate change involve transplantation of animals. Although protecting severely threatened species *ex situ* (i.e. in zoos or aquaria) until a suitable habitat for reintroduction becomes available (Noss, 2001) is a sound idea, using artificially-aided dispersal to move aquatic creatures over watershed boundaries to cooler waters (Gitay et al., 2001) could lead to the extinction of existing fauna, as the transplanted species would be pre-adapted to warming waters and could have a range of unanticipated effects on the native community. Similarly, the often-devastating effects that non-native fisheries stocking can have on native fauna would seem to make introducing better-adapted non-native species to replace waning stocks (Gitay et al., 2001) a bad idea. Transplanting vulnerable species to new, created habitats where no native community exists (i.e. uninhabited, enclosed basins) would eliminate the risk of damaging existing biotic communities (McCarty and Zedler, 2002), but creating functioning, self-sustaining ecosystems from scratch is likely to prove difficult. Re-introduction of species to their native habitat may be a useful approach in cases where an extreme weather event destroys existing populations (De Groot and Ketner, 1994).

USE ADAPTIVE MANAGEMENT STRATEGIES TO MAINTAIN FLEXIBILITY

Because many aspects of climate change are unpredictable and climatic variability is likely to increase, it is critical to maintain flexible conservation goals and strategies. Rather than striving to maintain current distributions of species and preserve current ecosystems, managers should allow for and even assist in the adaptation of species and communities. Static management practices with fixed policies designed to protect particular species should be changed to adaptive management strategies aimed at protecting ecosystem processes (Markham and Malcolm, 1996).

Adaptive management is based upon the recognition that uncertainty is inherent in all natural processes, and the expectation that management practices will change over time (Parma et al., 1998). Passive adaptive management involves adjusting management practices based on what is learned from the results of past practices, but learning about the underlying system is not an explicit goal. In contrast, active adaptive management is somewhat like performing a (hopefully) well-designed experiment; it involves forming multiple hypotheses about how the system will respond (to climate change and/or management practices), choosing strategies to systematically test and learn about the underlying hypotheses, monitoring systems closely to evaluate responses, and adapting future management practices based on what was learned about the system. The rationale behind this approach is that we cannot hope to understand the complex systems we are supposed to be managing without experimenting on them (Parma et al., 1998). Management practices in which a single (often arbitrary) strategy is chosen teach us little about the underlying system, and it is often impossible to know exactly what aspects of the strategy led to its ultimate failure or success.

Obviously, there are some cases where experimenting with the response of a system is not an option; some strategies may be irreversible, or may be too risky if populations are already endangered (Parma et al., 1998). However, adopting active adaptive management practices now, while the effects of global climate change are only beginning to be felt in many areas, may provide enough time to learn about the underlying processes governing how a particular system responds to change, and provide an understanding of how to best manage the system in the face of global climate change.

Selecting and Implementing Adaptation Strategies for Your Particular Location

The previous section illustrates some of the conflicts that may arise in choosing a management strategy to sustain freshwater ecosystems in the face of global climate change. For example, should the focus be on protecting high-diversity sites that are more likely to be resistant to climate change, or on protecting low-diversity sites that perform ecosystem services that will become more valuable as climate change continues? Should connectivity between habitats be maintained or even increased to allow migration, or should habitats be isolated to prevent invasion by exotic species? Applying a general list of conservation goals blindly to a freshwater ecosystem ignores the specific strengths and weaknesses of the system, and does not account for the relative importance of various threats. Performing a careful, site-specific analysis that takes all known factors into account and evaluates how to best meet conflicting goals is likely to be the best approach to designing a successful adaptation strategy.

CHARACTERIZE AND MONITOR SPECIES/SYSTEMS THAT ARE MOST VULNERABLE TO CLIMATE CHANGE TO DETERMINE THE SCALE OF CONSERVATION NEEDS

One of the most important steps in selecting an adaptation strategy for climate change is to characterize which life history stages, species, communities, or physical features in your location are most vulnerable to changes in average climate, changes in climate variability, or extreme events (Solomon, 1994). This can help determine the scale of management that is necessary. For example, if only one species of coldwater fish in a series of lakes is declining and the rest of the community appears to be resistant to climate change, a small-scale conservation strategy such as protecting thermal refugia and maintaining potential migration corridors to pristine lakes at higher altitude may be sufficient. However, if increased precipitation and runoff from deforested, agricultural lands is causing a dramatic increase in sediment load and nutrient content, along with large-scale changes in species composition, distribution and abundance, a more ambitious basin-wide management plan may be called for.

Attempting to predict which species will be most vulnerable to climate change before large effects are observed can be very useful, as initiating careful monitoring before effects are noticeable will provide a baseline against which future changes can be compared. While predicting exactly how species will respond to change can be difficult, numerical methods that combine assessments of climatic sensitivity (thermal tolerance, etc.) with general vulnerability (life history traits, current knowledge of the species, taxonomic uniqueness, overexploitation) may help highlight species for further consideration (Herman and Scott, 1994). Identifying and protecting keystone species is also important in order to prevent cascading effects that would alter the entire community (Noss, 2001).

Performing careful and continuous monitoring of both biological and physical indicators of change is critical, as reactions to climate change can be delayed in many plants and animals. For example, directly monitoring the water table and hydrology of temperate wetlands may be critical in preventing habitat loss, because by the time the vegetation shows a response to environmental change it may already be too late to save the system (Solomon, 1994).

DEVELOP STRATEGIES FOR PROTECTING FRESHWATER ECOSYSTEMS WHILE DEALING WITH INCREASING HUMAN WATER NEEDS

A serious potential threat, and one unique to aquatic ecosystems, is the anticipated rise in human water needs, mainly due to population growth and increasing development. Pressures caused by water extraction and climate change will almost certainly interact, exacerbating the effects of climate change on ecosystems and possibly increasing human water needs further due to increased temperatures and evaporation. Proposed adaptation strategies for human water resource management include “demand-side” adaptations, such as price incentives for conserving water, enforceable water efficiency standards, and increased irrigation efficiency, as well as less environmentally-friendly “supply-side” adaptations, such as building more dams (Arnell et al., 2001). The Intergovernmental Panel on Climate Change recommends using Integrated Water Resource Management (IWRM) to adapt to increasing water resource demands (Arnell et al., 2001). In this process, all stakeholders are included in potential considerations of supply- and demand-side actions before a decision is made, and the situation is continuously monitored and re-evaluated . Unfortunately, IWRM does not consider maintaining aquatic ecosystem function as one of the goals of water management (i.e. aquatic ecosystems are not considered a “stakeholder”); environmental damage is only included as a potential negative side effect of some actions.

A vast improvement over this water management strategy in terms of maintaining ecosystem health is ecologically sustainable water management, which strives to protect the ecological integrity of freshwater ecosystems while meeting current and future human needs for water (Richter et al., 2003). This strategy has been applied primarily in cases where one or more large dams already exist on a river and there are conflicts between increasing human water extraction and ecosystem flow needs, a problem that is likely to become more common in the future due to climate change. Methods of addressing water conflicts commonly include altering patterns of surface and groundwater extraction, increasing efforts to improve water efficiency, and changing temporal patterns of water release from dams. All stakeholders are involved in the process of estimating ecosystem flow requirements (seasonal base flows, high and low flows, rates of rise and fall), determining current and future human water needs, identifying incompatibilities (i.e. in seasonal or regional needs), and collaboratively searching for solutions. A critical component of this management strategy in terms of adapting to climate change is the inclusion of short-term management experiments (i.e. injecting treated wastewater back into groundwater reserves) to test the effectiveness of various management options as climate change continues to alter water flows, ecosystem needs, and human water demands. The method also calls for the development of an on-going adaptive management strategy to continuously monitor and respond to ecosystem changes.

This method has proven effective in many cases, such as in the management of the Green River Dam in Kentucky, USA (Richter et al., 2003). This dam was managed to provide recreational opportunities in the reservoir during the summer and water storage capacity to protect against flooding during the winter. As a result, large quantities of water were released from the dam during the fall, which biologists believed were disrupting downstream prey aggregations and dispersing mussel larvae during the fall breeding season. After examining ecosystem and human needs and identifying the incompatibilities (mainly related to the large outflow of water in the fall), the management of the dam was altered to release a steady, low volume of water throughout the fall, along with big bursts in November coinciding with natural storm events. The reservoir level was still lowered before the winter, when large storms could cause flooding, and the new schedule provided a more natural flow regime for downstream wildlife (Richter et al., 2003). This case provides an especially “tidy” example of ecologically sustainable water management (particularly because human water extraction was not a major issue), but the method has also proven effective in far more complicated cases where multiple governing bodies were involved and the goals of various stakeholders differed widely (Richter et al., 2003).

PERFORM INTEGRATED RIVER BASIN MANAGEMENT TO BUFFER SYSTEMS AGAINST CLIMATE CHANGE AND PROVIDE A BASIS FOR LONG-TERM CONSERVATION

Although a primary goal of ecologically sustainable water management is maintaining the ecological integrity of freshwater systems, the main focus is on managing the water itself. As described previously, freshwater ecosystems depend on more than just the water they contain—rivers, lakes and wetlands are intricately connected to all of the terrestrial systems within the drainage basin surrounding them. To protect freshwater ecosystems that are showing large-scale changes in response to climate change, or systems that are likely to be vulnerable (due to land alteration or other anthropogenic stresses), the entire drainage basin must be managed.

One-sided decisions to adopt fixed river basin management plans are likely to fail if unexpected climatic changes occur. For example, the tri-nation development authority in charge of the Senegal River basin in the Sahel region of West Africa initiated a plan several decades ago to convert large areas of the river basin to irrigated rice production, in an effort to reduce the countries’ dependence on foreign imports. Two dams were constructed to provide irrigation water, but during the 1960’s, the Sahel began to experience a severe and long-term drought that current climate models suggest may be permanent (Venema et al., 1997), constraining the amount of water available for flooded rice paddy agriculture. Over the last several decades, the basin has suffered increasing desertification, due in part to climate change and in part to the abandonment of cleared rice paddies. As a result of water shortages and river basin degradation, large numbers of local residents have migrated to urban areas. Proposed changes in river basin use include switching from industrial rice production back to local, village-based agriculture utilizing cereal and grain crops with low irrigation needs, as well as a more ambitious plan to convert areas of the basin to agro-forestry, which involves mixed vegetable production within plots of reforested land; this reforestation would help lower air and water temperatures, and halt desertification. The reforestation plan also recruits unemployed urban residents in an attempt to reverse recent patterns of massive demographic shifts to urban areas (Venema et al., 1997).

In order to avoid the necessity of restoring river basins that have been severely degraded by climate change and failed management, the basin-wide needs of ecosystems and local communities should be considered, as well as the potential impacts of future climate change, before enacting any management plan. Integrated River Basin Management (IRBM) is a method of balancing basin-wide ecosystem needs with human water resource needs to achieve economic, social, and environmental goals. One of the major problems with basin-wide management is that many river basins cross national boundaries; worldwide, there are 261 major transboundary rivers that drain 45% of the Earth's surface, account for 80% of the planet's river flow by volume, and are home to 40% of the world's population (WWF, 2002). Transboundary rivers can make water management more difficult, but management of these systems is essential to future economic and political stability.

With IRBM, the needs and expectations of all "water stakeholders" (local community members, civil authorities, water and fishery resource managers, scientists and conservationists, and representatives of the private sector) from all countries are assessed jointly, a basin-wide authority is created, monitoring methods are developed, and an adaptive management plan is initiated (WWF, 2002). Decisions can be made locally, but must be in accordance with basin-wide strategy. Key aspects in determining the success or failure of IRBM are actively promoting public involvement (through appropriately-scaled local discussions suited to the target audience) and ensuring sustainable funding for the basin-wide authority (often through water taxes, reductions in water subsidies, or international funding programs). Because it employs adaptive management strategies, IRBM is able to continuously respond to the effects of climate change. In addition, because management is basin-wide, IRBM provides the ability to protect climatic refugia and areas that are particularly vulnerable to climate change, as well as to minimize damaging land use practices and other human stresses that are known to interact with and exacerbate the effects of climate change.

Although system-wide conservation can present difficulties in terms of reaching consensus, and can be costly and slower to achieve noticeable results than local campaigns to preserve endangered species, system-wide approaches ultimately conserve the functional value of the river basin and provide a sustainable basis for future conservation (Moss, 2000). Successful IRBM in Costa Rica, in which fossil fuel taxes and payments from a private hydro-electric plant compensate upstream landowners for the maintenance and restoration of forest cover, has improved water quality and quantity for downstream towns, farms, and industries, and reduced sediment accumulation at the hydroelectric plant (WWF, 2002). In South Africa, IRBM led to the creation of the Working for Water Programme, a massive project that protects biodiversity while creating jobs for 18,000 people clearing invasive, water-hungry plants from several river basins.

Within a managed river basin, priority should be given to protecting upstream subbasins and headwaters; these areas serve as thermal refugia, providing cool, oxygen-rich groundwater to many species not found downstream, and are often the destination of migratory fish and other animals (WWF, 2002). In addition, upstream areas affect the flow regime and water quality of all areas further downstream, and are often less degraded and easier to protect (Skelton et al., 1995). It is also important to protect side channels and backwaters, which serve as refugia and spawning grounds for a range of animals, and provide corridors to important floodplains (Sedell et al., 1990). Finally, other downstream areas should also be protected, as these areas are necessary for the migration of diadromous species, possess abundant water resources, and are often more productive and species-rich than upstream areas. However, downstream areas are also more frequently subjected to human pressures such as water extraction and land use, so protecting these areas may involve restoration or creative approaches to resource management (WWF, 2002).

PROTECT ENTIRE FRESHWATER ECOREGIONS TO PRESERVE THE ECOLOGICAL AND EVOLUTIONARY DRIVING FORCES OF BIODIVERSITY

Freshwater ecoregion conservation (ERC) involves protecting relatively large units of water and surrounding land that contain distinct assemblages of natural communities sharing many of the same conditions (Abell et al., 2002). These ecoregions are defined by the similarity of the communities and conditions contained within them, and can include one or more drainage basins. Many freshwater ecoregions encompass a number of basins and cover vast tracts of land, such as the Amazon River and Flooded Forests ecoregion and the Yangtze Rivers and Lakes ecoregion (Abell et al., 2002).

The main difference between freshwater ecoregion conservation and IRBM is that ecoregion conservation focuses primarily on preserving biodiversity, while IRBM emphasizes balancing environmental, economic, and social needs. In addition, ecoregion conservation is often applied to areas of high-biodiversity importance, in order to preserve these exceptional areas while addressing and analyzing the ecological and evolutionary driving forces of biodiversity. Because of its focus on protecting high-diversity communities and ecosystems (which are likely to be more resistant and resilient), and because the large scale of ecoregion conservation allows managers to control most non-climatic stresses affecting freshwater ecosystems, ERC is particularly well-suited to buffer systems against the effects of climate change. Additional benefits of an ecoregion approach include the ability to address the conservation needs of wide-ranging species or species that require particularly large areas of habitat, and the ability to address threats that operate across an entire ecoregion with a single, coherent approach (Abell et al., 2002).

In relation to climate change, freshwater ecoregion conservation may provide the most comprehensive and holistic approach to providing adaptation options (thermal refugia, migration routes, etc.) and minimizing anthropogenic stresses, thereby allowing regions of paramount importance in biodiversity to resist, recover from, and/or adapt to global climate change. However, because many river basins throughout the world are heavily populated, a more balanced approach (such as IRBM) that considers socioeconomic factors in adaptive management of river basins may be more likely to provide long-term protection; as human water demands increase and human populations experience more direct effects of climate change and extreme climatic events, interest in protecting ecosystems is likely to wane and management schemes with the sole goal of protecting freshwater ecosystems and biodiversity may lose support. Ideally, IRBM could be applied to more heavily populated or degraded basins within a freshwater ecoregion, but overall management of the region could be guided by ecological principles aimed at increasing resistance and resilience to climate change, thus combining the benefits of the two approaches. Ultimately, however, any management strategy for dealing with global climate change will simply buy time until either rapid, anthropogenic climate change ceases or species are no longer able to adapt and massive extinctions result. Management strategies can only provide long-term protection of freshwater ecosystems if the root causes of climate change are addressed and solutions are enacted on a global scale.

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CHAPTER 8
Protected Areas

Ecological and Socio-economic Benefits of Protected Areas in Dealing with Climate Change

Nigel Dudley and Sue Stolton
Equilibrium

WELL-DESIGNED AND MANAGED protected areas may ameliorate some of the problems caused by climate change, with benefits well beyond the park boundaries. These potential benefits are predicated on a number of assumptions about the way in which natural ecosystems will respond to changing climate—some of which still need to be thoroughly tested—and on further assumptions about the design and management of protected area systems. But we already know enough about the predicted and observed impacts of climate change, and about the role of protected areas, to draw some initial conclusions about how to get the best out of protected area networks in terms of buffering against climate change.

This chapter outlines some of the known and expected benefits that protected areas offer national or regional strategies to increase resilience against climate change and suggests ways in which protected area authorities and managers can maximise these benefits. Given the uncertainty inherent in climate models, this advice is still preliminary. Most contemporary research on the links between climate change and protected areas not surprisingly focuses on the threats posed to parks and reserves. Their potential role in buffering ecosystems and human societies has been less carefully studied and at least some of the claims that have gained attention are based more on hearsay and received wisdom than on quantitative data or real-life examples. Information has often not progressed from hurriedly prepared technical reports into the scientific literature, making judgements more difficult. In the following account we try to sort out the myths from the reality and to provide protected area managers with a guide to maximizing the potential benefits that protected areas can offer in efforts to resist the impacts of climate change.

There is a growing recognition that natural ecosystems, both large and small, could provide a suite of ecosystem services related to climate change, ranging from protection against immediate physical impacts such as rising temperatures, unstable climates and rising seas, to providing additional insurance against the predicted instability of agriculture, fisheries and water resources. We look first at the significance of particular impacts and then at how protected areas might relieve symptoms, in each case referring to real-life examples of protected areas. Wherever possible, we have drawn on protected area management interventions that have been developed specifically as a response to climate change; in one or two cases we use examples which have relevance to climate change but that have been initiated for other reasons.

The main issues examined are: disaster mitigation covering hurricanes, droughts, flooding, avalanches and coastal erosion; fires; biodiversity; water security particularly with respect to drinking water; the need for rapid crop adaptation; and food security, including the potential of crop and fish stock failure. In some cases, the role of protected areas in mitigating or relieving effects is already clear, while in others the jury is still out and we need to learn more before management recommendations can achieve any degree of certainty.

Managing to Relieve the Impacts of Climate Change

Many protected area managers already wrestle with a range of problems relating to climate change effects within their parks or reserves. Here we suggest that carefully designed and managed protected areas can help relieve problems *beyond* the park boundary. In some cases protected areas are simultaneously a potential buffer against a particular climate-related problem and at risk from the same problem, creating a tension and the need for some tough decisions from managers. We have already made clear that there is a lot to learn about many of these impacts and therefore research and monitoring must be important components of any response. In addition, opportunities to use protected areas as buffers against climate change require two additional management responses:

PLANNING TO MAXIMIZE THE BENEFITS OF PROTECTED AREAS IN RELIEVING THE SYMPTOMS OF CLIMATE CHANGE: the importance of assessing protected area management effectiveness is increasingly recognized; adding climate change elements into such assessments would be relatively easy and enhance their usefulness dramatically. This

Figure 1: Possible matrix for conducting a local overview of climate change effects

Issue	Current situation regarding the issue	Value of PA in addressing threat (descriptive or quantitative)	Trend under climate change	Site modification to maximise benefits	System modification to maximise benefits	Potential economic value
Disaster mitigation						
Fire						
Biodiversity conservation						
Food security						
Water security						

could be helpful for individual protected areas and, in the case of a system assessment, aid in planning future protected areas. Such assessments could include primary research or a thorough literature review, or begin more simply, with a local survey to overview the situation. A draft set of questions for such an overview is suggested in Figure 1.

Ideally, such an analysis could be carried out on the system level (ecosystem or landscape), so that both protected area agencies and individual managers can put management actions into a national or regional context.

TESTING HYPOTHESES THAT PROTECTED AREAS CAN HELP MITIGATE THE EFFECTS OF CLIMATE CHANGE: the cases described below show managers already thinking about using land and water resources to buffer against climate change (or sometimes making interventions that could help mitigate climate change without recognizing the connection). However, these are rare examples. More systematic work is needed to quantify beneficial impacts and to refine management interventions to maximize these benefits.

In the following sections, we review the evidence, suggest what protected areas can do and, in each case, provide at the end a brief summary of management responses.

DISASTERS: DROUGHTS, FLOODS, AVALANCHES AND COASTAL EROSION

Like the four horsemen of the apocalypse, a whole suite of disasters are predicted to increase in severity and frequency as a result of changing climate: changing rainfall patterns causing floods and droughts; extreme weather events such as hurricanes; a combination of climate changes that could increase avalanches; and the inexorable rise in sea level along with increased storm damage to coasts. Available evidence, from modeling studies and observations, supports some but not all of these popular predictions. The Intergovernmental Panel on Climate Change (IPCC) concludes that rainfall scenarios suggest an increase in both flooding and drought frequency in many parts of the world (McCarthy et al., 2001), and these predictions are echoed by both the United Nations (United Nations World Water Assessment Programme, 2001) and the World Bank (World Bank, undated), while the impacts of climate change on frequency and intensity of hurricanes remain more uncertain. Variations in annual temperatures may not increase total avalanches, but certainly make it more difficult to predict when and where they will occur. Sea level rise is predicted to have enormous impacts on many low-lying coastal communities; as this chapter is written the government of the Pacific island nation of Tuvalu is lobbying the Australian government for space for the country’s 9,300 inhabitants, currently living on land only 3 metres above the sea and considering a mass evacuation (Fickling, 2003).

FLOOD

The presence of natural vegetation generally reduces storm-related erosion and landslip/landslide and can ameliorate local flooding, but is normally not enough to stop major floods. Research in New Zealand after Hurricane Bola in 1988 found that shallow landslips/landslides only occurred in 1% of the area with forests older than 5 years versus 30% of the area of cleared lands (Trustrum and Page, 1992). However, a review carried out for the UN Food and Agriculture Organisation concluded that forests were only likely to reduce flooding in relatively minor storms (Calder, 2000), and claims that deforestation in the Himalaya has led to flooding in the Ganges are now believed to be incorrect (Hamilton, 1987). A widespread belief that Hurricane Mitch in Central America had less impact on areas still rich in natural forest, such as Nicaragua, has been challenged by the Center for International Forestry Research (Kaimowitz, 2002). Yet in other cases, forests do appear to have an important role to play in regulating water flow, including particularly swamp forests such as the huge Varzea forests in the Amazon. Forests also provide benefits at a smaller scale, in the areas immediately surrounding the forest, so that integration of protected forests with working landscapes, as in IUCN Category V protected areas, offers benefits to local communities. A review in North America concluded that for many reasons, including likely temperature changes and more frequent and intense storm events, climate change increased the need for forest protected areas to be as large as possible, including the ability to withstand extreme weather events (Noss, 2001).

DROUGHT

Maintaining natural vegetation can provide an important insurance policy in areas prone to drought and there is a growing consensus that social impacts of drought in the Horn of Africa have been exacerbated by prior forest loss. In arid Djibouti, the Foret du Day National Park is the only large protected area in the country (IUCN category II, 10,000 ha) and is also the only significant forest area. Being at a higher elevation and subject to mists, it retains more vegetation and thus provides a source of forage during drought: the protected area is thus a natural buffer against starvation. Currently traditional management patterns are breaking down and over-grazing is resulting in replacement with drought resistant trees, pointing to the need for negotiation and changes in management (Barrow, pers. comm.).

AVALANCHE

At a smaller scale, protection zones, which may or may not be officially protected areas but fulfill the same functions, are a key feature in avalanche control and small-scale flood control throughout the mountainous areas of Europe. Forest management in the Tatra National Park, a transboundary protected area between Poland and Slovakia, has long emphasised avalanche control. In the Swiss national forest, 8% of the total area is managed primarily for avalanche protection and as long ago as the late 1980’s this was calculated to have an annual value of 3-4 billion Swiss francs (Küchli et al., 1998).

COASTAL IMPACTS

Protected areas play an even clearer role in disaster mitigation in marine and coastal areas. Sea level rise and increased storm damage puts coastal communities and small islands at particular risk. Building physical barriers against the rising sea is technically difficult and colossally expensive for countries with large, low-lying coastal regions, stimulating a new approach to integrated management, relying on natural features such as coral reefs and mangroves. A comprehensive review of marine ecosystem services suggests that natural features like coral reefs and mangroves are the most cost-effective option for maintaining coastal integrity and cannot easily be substituted by artificial reefs and seawalls or by aquaculture (Moberg and Rönnbäck, 2003)—making responses such as restoration of mangroves even more important (Field, 1999).

Paradoxically, these natural features are also amongst those most acutely at threat from climate change. In these cases, managers will be faced with a series of hard decisions, balancing the benefits from a feature like a reef or mangrove, the chances of it surviving climate change and the costs and benefits of protection. In some cases trade-offs may be necessary. For example choosing to concentrate money and resources into key areas and abandoning others. Looking beyond spatial biodiversity conservation to include other factors, including the wider benefits of protected areas in terms of coastal protection, may influence the choice of where to focus the most effort and could also help attract additional support for those protected areas that are also helping to mitigate climate change impacts.

The Sundarbans region of Bangladesh provides an almost text book example of the link between climate change and protected areas. This huge area of mangrove in the Ganges floodplain is both at risk from climate change impacts and a potential tool in resisting the impacts of such change. A recent review summarised seven national and international analyses of climate change impacts on Bangladesh (Huq et al., 2003) and concluded that the Sundarbans were at risk and increased disasters were likely in the coastal zones covering 30% of the country. Impacts would result from sea-level rise and associated factors including high evapotranspiration and low flow in winter resulting in increased soil salinity. The IPCC predicts that in worse case scenarios up to 75% of the mangroves could disappear.

The area is heavily affected by tropical storms: about 10% of the world’s tropical cyclones occur in the Bay of Bengal and of those 17% hit land in Bangladesh. The mangroves fringing the delta have been a traditional barrier, providing socially and economically valuable services. The Southwest Area Water Resources Management Project calculated that the absence of the Sunderbans mangroves would mean building 2,200 kilometres of cyclone/flood embankments requiring a capital investment of Taka 16 billion (US$ 294 million) with a yearly maintenance budget of Taka 320 million (US$ 6 million) (UNDP et al., 1995). The Sundarbans are protected in three connected wildlife sanctuaries (IUCN category IV), although they continue to be degraded and in the past have been heavily logged (Scott, 1989). This has led to a requirement for mangrove restoration, and some experience is already being developed in the Sundarbans, for example on Sagar Island (Saha and Choudhury, 1996).

Coral reefs also provide barriers to wave activity for low-lying coastal regions and islands but, like mangroves, are themselves one of the first habitats to suffer the impacts of climate change, mainly through coral bleaching. Many coral communities are also suffering intense damage from, over-exploitation, shipping and pollution. Marine protected areas can relieve some pressure from corals, thus giving them the best chance of maintaining themselves and thus also providing a protective function. In American

Samoa WWF scientists are working with local communities to look at options for managing change. They are examining options for reducing the impact of coral bleaching in the National Park of American Samoa and Fagatele National Marine Sanctuary by, for example controlling high nutrient terrestrial run-off into coastal waters.

FIRE

Climate change will add an additional element to the already complex relationship between fires and natural ecosystems. Hotter, drier conditions tend to increase fire frequency, resulting in changes in vegetation as more fire-tolerant species become common in both the tropics (Goldamer and Price, 1998) and in temperate and boreal regions: in northern Minnesota, USA, forest fires were most common in the warm, dry periods of the fifteenth and sixteenth centuries (Clark, 1988) and research in Canada suggests that a doubling of atmospheric CO_2 could lead to a 46% increase in seasonal severity rating for forest fires, and possibly a similar increase for their incidence (Flannigan and Van Wagner, 1991).

In areas where fires have recently burned at unnatural and ecologically damaging levels – particularly in tropical forests where under normal conditions fires are a rarity – there is strong evidence of a link with the climate phenomenon called ENSO (El Niño-Southern Oscillation), which causes periodic droughts (Leighton and Wirawan, 1986), including the drought that affected much of the forests which caught fire in 1997 and 1998. The frequency and intensity of El Niño could itself be increasing as a result of climate change (Trenberth and Hoar, 1996, 1997), which means the world faces warmer, more violent weather, and more forest fires. More generally, increased droughts and average temperatures in fire prone areas may also increase the frequency of fires. However, these impacts are likely to be outweighed by the influences of management: the fires in Brazil and Indonesia in 1997 and 1998 may have been intensified because of El Niño, but they were begun by plantation owners and slash and burn farmers (Holdsworth and Uhl, 1997) and conditions were made more suitable for fire through earlier management changes (Uhl et al., 1988).

Forest fires can directly affect protected areas – for example Kutai National Park in Borneo was virtually destroyed by fire (Goldammer and Seibert, 1992) – but in other parts of the world unnatural levels of fire suppression are a significant conservation problem (Noss et al., 1995) and it therefore follows that fire management responses to climate change must be tailored to local conditions.

In areas where fire is an expected part of the ecosystem, some balance between management and non-intervention will probably be optimal, with prescribed burning usually a more successful management intervention than outright suppression (Stephenson, 1999) (which allows build-up of fuel and less frequent, but more catastrophic, fires). In fire-dominant landscapes, leaving forests to attain old-growth status can increase the risk of fire and thus protected areas close to centers of population are sometimes regarded as an added fire hazard. In the aftermath of unusually intense fires in Australia in

February 2003, there were calls from some quarters for logging in national parks to reduce fire risk (Victoria Parks Service, pers. comm.) and some measure of fire management may be necessary. Conversely, in areas where fire is generally a factor of human intervention, such as in tropical moist forests, frequent forest fires cause long-term damage (Woods, 1989) and protected areas can provide important buffers to prevent the further spread of fires from poorly managed swidden agriculture or fires used in pasture creation or clearance for plantations.

Training in community fire management is in these cases an important part of management in those protected areas with resident human communities (Karki, 2002). For example, working with local communities to manage and reduce fires is a major focus of management for the Pha Taem protected areas complex between Thailand, Laos and Cambodia, and the presence of the protected area has opened up the possibility of cross-border cooperation in a way that would not otherwise be possible (Trisurat, 2003). In other protected areas with grassland or savannah habitats, such as the Serengeti in Tanzania, controlled use of fire is an integral part of management, usually to maintain particular ecosystems.

Biodiversity

Ever since Edward Wilson published his classic text on biodiversity (Wilson, 1988), it has been recognized that we are undergoing rapid loss of species and it has long been argued that climate change will increase this trend (Markham, 1996). Recent studies suggest that a significant impact of global warming is already discernible in plant and animal species (Root et al., 2003). Researchers have also found that other factors such as the presence of invasive species and imbalances in hydrological cycles increase these threats (Crumpacker et al., 2000).

Most protected areas are primarily aimed at protecting biodiversity, a role that is stressed in the IUCN definition of a protected area: *An area of land and/or sea especially dedicated to the protection and maintenance of biological diversity, and of natural and associated cultural resources, and managed through legal or other effective means* (Anon, 1994). Such protection can range from providing a secure base for species that spreads far beyond the border of protected area networks, such as the giant panda that is protected through a reserve network but also lives in unprotected forests and mountains, to protecting the entire known population of endemic species with a strictly limited distribution. The increased stress on biodiversity created by climate change therefore increases the arguments for protected areas. To some extent their role here is mainly to reduce other threats by maximizing the resilience of the ecosystem and to provide large, unbroken blocks of habitat that will be relatively resistant to environmental change. But managers are also looking at other actions to address climate change at a site and network level, including active restoration to help speed up changes in vegetation patterns created by temperature difference; translocation programs for threatened species; and active management of patch dynamics. Experience with translocation at a habitat scale is not encouraging (Hodder and Bullock, 1997) although translocation of individual species may be more realistic. Using protected areas to conserve biodiversity in the face of climate change often means cooperating between protected areas and also often between countries (or other geographical/managerial jurisdictions), adding weight to arguments for transboundary protected areas.

The practical implementation of these challenges is starting to be worked out by protected area agencies, researchers and managers, often working in cooperation. In the Netherlands, a new national conservation plan looks specifically at increasing options for migration between protected areas and a process of "de-fragmentation" pursued with a target that: "by 2020 migration within and between nature areas will be secured by removal of physical barriers" (Anon, 2000). A review undertaken for the UK government identified three policy responses: the possible need to redraw boundaries of protected areas and re-designate sites as habitat for particular species shifts; managed changes particularly inshore from seawater defenses to recreate coastal habitat before this is flooded, including restoration and the use of holding reserves for species; and translocation of species with a narrow distribution, such as the Snowdon Lily *(Lloydia serotina)* (Hossel et al., 2000). In the high Arctic, scientists working in a series of UNESCO biosphere reserves including Abisko (Sweden), Zackenberg (Greenland), Taimyrsky (Russian Federation) and Niwot Ridge (USA) are measuring responses of certain plants to changing conditions and developing management responses (Arft et al., 2000). In North America, cooperation between the Mapini Biosphere Reserve in Mexico and the Big Bend Biosphere Reserve in Texas, USA, is examining options for translocation of the endangered Bolsón tortoise *(Gopherus flavomarginatus)* further north as a response to climate change (Hadley, 2002).

Water Security

In the past 100 years the world population tripled, but water use for human purposes multiplied six-fold (World Water Council, 2000). Water security is already an important and sometimes politically explosive issue and this is likely to intensify under climate change. After the 1992 Earth Summit, *Agenda 21* noted that: "Higher temperatures and decreased precipitation would lead to decreased water-supplies and increased water demands; they might cause deterioration in the quality of freshwater bodies, putting strains on the already fragile balance between supply and demand in many countries". The IPCC has repeatedly identified potential problems with water supply, particularly in arid regions.

Protected areas are increasingly being used to guarantee the quantity and to an even greater extent the quality of the water that we use, including particularly urban drinking water. As with some of the other links discussed above, this relationship is not without controversy.

There seems to be no constant relationship between presence of forests and *quantity* of water, and simplistic claims that forests increase or stabilize water supply are often incorrect. Exceptions to this appear to be cloud forests that do intercept water and may increase flow. In addition, some old forests also increase net water flow. Studies of rainfall and runoff data in Australia collected from large forested catchments in the Melbourne area that were completely or partially burnt by a large-scale wildfire in 1939, concluded that the amount of water yield from forested catchments is related to the forest age. It was found that forest disturbance can reduce the mean annual runoff by up to 50% compared to that of a mature forest, and can take as long as 150 years to fully recover (Kuczera, 1987).

In addition, and more importantly, natural forests in catchments do often appear to have a beneficial impact on water *quality,* thus reducing the very high costs of purifying drinking water. Forested catchments can therefore play an extremely important economic role in reducing costs of purifying water and in situations where clean water is not universally available they can also help improve public health.

Drinking water supplies provide a clear example of how the link between climate change, hydrology and water supply might relate to protected areas. Currently a third of the world's top hundred cities rely on protected forests for some or all of their drinking water and some, such as Melbourne, Dar Es Salaam and New York have made well-publicized links between forest protection and purity of water supply. Others, like Istanbul and Bangkok, suffer from water supply problems in part because of environmental deterioration in their watersheds (Dudley et al., forthcoming). In most cases, protected areas are important principally because of increased water purity while in some cities they are also believed to increase net quantity of water available. Many municipal authorities have pre-empted the debates amongst hydrologists and protected catchments as insurance for their water supplies, particularly in cloud forest areas.

Water supply links also exist between, for example, Mumbai (Bombay) and Sanjay Ghandi National Park in India; Bogota and Chingaza and Sumapaz National Parks in Colombia; Rio de Janeiro and Tijuca National Park in Brazil; Vienna and Donau-Auen National Park in Austria; and Abidjan and the Parc National du Banco in Côte d'Ivoire. Economic studies point to the importance of these protected areas in terms of reducing purification costs. Recent studies calculated that the presence of Mount Kenya forest alone, which is within a 70,000 ha IUCN category II protected area, saved Kenya's economy more than US$ 20 million by safeguarding the catchment for two of the country's main river systems, the Tana and the Ewaso Ngiro, from deforestation (Emerton, 2001). On a smaller scale, payment for maintaining forests to insure the future of water supplies has been introduced for hydroelectric schemes, such as in Costa Rica (Rojas and Aylward, 2002), for pure industrial water, such as payments negotiated with Pepsi Cola in Guatemala (Gretzinger, pers. comm.) and for irrigation. The World Bank is currently supporting the development or implementation of Payment for Environmental Services systems in Costa Rica, Guatemala, Venezuela, Mexico, Colombia, Nicaragua, Dominican Republic, Ecuador, El Salvador and South Africa. Many of these look specifically at the impacts of protected areas on water quality. For example a project financed by the Global Environmental Facility is under preparation, focusing on Canaima National Park, with significant co-financing from hydropower producer CVG-EDELCA (Pagiola et al., 2002). As climate models suggest that water stress is likely to increase in the future, the added benefits and security provided by forest protected areas are particularly important.

The cloud forests in La Tigra National Park in Honduras are a typical example, sustaining a well-regulated, high quality water flow throughout the year, which provides over 40% of the water supply to the 850,000 people in the capital city Tegucigalpa (Dudley et al., 2003). As with mangroves and coral reefs, there is a tension here because cloud forests are also particularly prone to climate change (Bruijnzeel, 2001; Calder, 2002). Although some of the stresses on cloud forests are outside the control of managers on the ground, research suggests that climate change stress will be exacerbated by other factors, including fire, drought and plant invasions (Foster, 2001). Additionally forest loss nearby can reduce clouds and thus adversely affect cloud forests (Lawton et al., 2001). The close link between cloud forests and water resources is therefore an additional argument for devoting resources to cloud forest protection and for those responsible for developing national or ecoregional protected area networks to include provision of water amongst the arguments for protection of tropical montane cloud forests.

Crop Genetic Stress

Changing climate will increase stress on both new and traditional crop varieties and the IPCC has repeatedly referred to the need for crop breeding to respond to these changes (McCarthy et al., 2001). Changes in agriculture have tended to increase risks by radically reducing the number of crop varieties, for example research by the Rural Advancement Fund International found that 97% of the varieties given on old United States Department of Agriculture lists are now extinct (Fowler and Mooney, 1990). Rice production provides an example of extreme cultivar uniformity, with 75% of rice varieties grown in Sri Lanka descending from one maternal parent, a uniformity mirrored in Bangladesh and Indonesia (Groombridge, 1992). Wild relatives of modern crops therefore have a particularly important role to play in kick-starting the breeding of new strains under time pressure. Unfortunately many of these have virtually disappeared from their original centers of diversity in areas of Europe, the Middle East, Africa and Asia.

Protected areas are one specific response to this loss of agricultural biodiversity and an increasing number are being designed specifically to protect sources of crop genetic material. The potato provides a good example. Modeling suggests that there will be increased potato yield variability for Europe (Wolf, 2000), increasing risk of crop failure. In response to these and other threats, potato conservation is being highlighted in the Andes, where the wild ancestors originate. At Pisac Cusco, in Peru, seven Quechua communities are planning to establish a “Potato Park”; a community-based conservation area focused on agri-biodiversity, managed through an integrated landscape model. The area is a recognised centre of crop diversity for potatoes and other important Andean crops (e.g. Quinoa, Kiwicha, Mashua). It lies at the heart of the ancient Inca Empire (and it is suspected that the Incas brought potato varieties from other parts of the continent). The aim is to protect and conserve native plant genetic resources as well as associated traditional knowledge and local cultural heritage. The initiative addresses local concerns regarding food security, poverty alleviation, gender equality, intellectual property and the right to self-determination for indigenous peoples. It aims to include ecotourism, the marketing of native crops and capacity building in sustainable agriculture. Peruvian authorities and institutions such as the International Potato Centre recognise the value of the scheme and a committee of government and non-governmental organizations is studying legal options for formal recognition. Indigenous peoples are learning of their rights to biological resources and of the potential benefits derived from their use (Phillips, 2002).

Food Security

Climate change will also disrupt agricultural systems and fisheries. While there will be both winners and losers in the resulting shifts in centers of food production, all the signs are that the poor—especially those living in marginal environments—will be the most vulnerable (Downing et al., 1996). The IPCC estimates that by the 2080s about 80 million extra people will be at risk of hunger. Yields are likely to fall particularly in the tropics, where some crops are already near their maximum temperature tolerance (McCarthy et al., 2001). Protected landscapes and seascapes provide a potential buffer in three ways: by providing breeding grounds for commercial species, by providing wild foods to the poorest members of society in time of agricultural failure and by preserving genetic resources needed for adaptive breeding.

SUSTAINING FISHERIES: There is still considerable debate about the likely impact of climate change on fishery resources, although an increasing consensus is developing that climate change will have both positive and negative impacts on the abundance and distribution of marine fish. There are concerns that where climate change impacts reduce fish stocks, this loss combined with over-fishing could lead to serious impacts for coastal subsistence communities and on countries' economic performance. Strategies to address climate change effects must therefore minimize pressures on fish stocks and provide the best conditions for their maintenance. The IPCC identifies an important role for marine reserves: “Adaptation measures that are relevant to the fishing industry may include the following: …Organization of marine biosphere reserves and protected areas for the habitat of marine mammals” (McCarthy et al., 2001).

Unlike terrestrial protected areas, where the initial incentive has usually been wildlife conservation, many marine protected areas start with an aim of maintaining fish stocks. The presence of no-take zones and strictly protected zones often deliberately coincide with spawning or nursery areas for fish. Because marine protected areas are generally newer (less than 1% of the ocean is in a protected area compared with over 10% of the land), they have often included stakeholder approaches from the beginning, allowing their planning and demarcation to be subject to far more negotiation and buy-in than with early land-based protected areas.

Until recently, links between marine protected areas and increased fish stocks were based largely on un-quantified observations by villagers and protected area managers. A series of research projects has now provided more concrete evidence that marine protected areas increase the numbers, biomass and variability of fish and that they have a significant export to surrounding waters, so that fishing communities operating nearby see an increase in catch (Gell and Roberts, 2003). Significantly, they show that MPAs provide insurance against environmental variability and year-to-year fluctuations: the conditions that are predicted to increase under climate change. A review of 80 marine protected areas found that on average reserves doubled abundance, tripled biomass and increased both size and diversity of fish by a third (Halpern, 2003) and the same data showed that increases usually became obvious within five years of protection (Halpern and Warner, 2002). Research projects in Kenya around the Mombassa Marine Park (McClanahan and Mangi, 2000) and in St Lucia in the Caribbean (Roberts et al., 2001) are both typical in finding increased catches around MPAs as compared to other nearby fishing areas, and local fishing deliberately targeting areas near the reserve ("fishing the line"). This evidence is building public support for marine protected areas as a long-term insurance policy. The 1500 km^2 marine component of the new Quirambas National Park in Mozambique has been developed and agreed with local communities in part to protect fish breeding grounds (WWF Endangered Seas Campaign, pers. comm.).

Many current reserves are too small to have a significant impact on fisheries. A recent review (Roberts and Hawkins, 2000) and theoretical modeling exercise (Pezzey et al., 2000) both suggested that larger reserves were often required, with ideally between 20-40% of fishing grounds being set aside, with the amount of protected area increasing with fishing intensity. The presence of reserved areas to help buffer against fluctuations caused by ENSO events and other fluctuations is likely to become increasingly important. However, enforcement effectiveness has been highlighted as a major problem for marine protected areas (Jameson et al., 2002) and their role in maintaining fish stocks depends on the rigour with which governments and local communities are prepared to enforce their protection.

NON-TIMBER FOREST PRODUCTS AND GAME: Throughout history, poor people have collected wild foods in times of shortage, including game and non-timber forest products. This pattern continues, both in the developing world where, for instance, wild game forms 70-90% of protein intake in parts of Africa (Sayer and Ruiz-Perez, 1994) and in Europe and North America, where recent immigrants are reviving non-timber forest product (NTFP) harvests (Hansis, 1996). Natural forests are acknowledged as serving as "safety nets" for the rural poor (Wunder, 2001), and this role is likely to become more important under climate change because sources of agricultural food will become increasingly stressed and liable to seasonal failure under fluctuating climate conditions.

The extent to which protected areas can be included in this safety net is the subject of practical research projects around the world. In places where large-scale land conversion has occurred, they may be the only source of some wild foods. The collection of NTFPs and game from protected areas has traditionally been seen as a management "problem", and indeed unregulated collection can cause serious biodiversity losses (Barnett, 2000), even for subsistence purposes (Noss, 1998). However, protected area managers are increasingly working with local communities, not to ban collection and hunting but to manage it on a sustainable basis. This has immediate social benefits and if local people are allowed controlled access to resources they have more incentive to work with rangers in controlling organized poaching.

Programs to implement sustainable use of wild species have been introduced in many protected areas, so far mainly for reasons other than climate change: the following two examples are from Africa. Bwindi Impenetrable Forest Reserve in Uganda (IUCN category II, 32,000 ha) is home for half the world's remaining mountain gorillas along with important populations of chimpanzees and elephants. It is surrounded by agricultural land, up to the park borders in some places: although soil is fertile, agriculture takes place on steep slopes and in sometimes harsh climatic conditions. To address local resentment at the introduction of protection, agreements have been reached with 21 neighboring parishes to collect NTFPs, including medicinal plants and handicraft species, and to practice beekeeping. Twenty percent of the park has been reserved for this purpose, with collection areas changing over time to maintain populations. Local people collaborate with rangers in collecting information about changes in population of desired species (Makombo, 2001 and information collected on site), thus building experience on how the protected area can be harvested sustainably.

Lobéké National Park in Southeast Cameroon has followed a similar pattern, although here negotiation with communities started as soon as the park was agreed, being run as a WWF project since 1995. Lobéké is in a less heavily populated area than Bwindi and is surrounded by forest, although the latter has generally been logged. The key need of communities is for game. Through a detailed stakeholder consultation, based around the principle of exchange—rights for responsibilities—five community-managed hunting zones have been agreed covering 200,000 ha along with other forms of collection (Hakizumwami, 2000 and information collected on site). It is hoped to avoid the problems of over-exploitation and consequent loss of both biodiversity and meat sources that have plagued other protected areas in Africa.

Conclusions: What Does This Mean for Protected Area Agencies?

This review has shown that, at our present levels of understanding, protected areas have much to offer in terms of helping to maintain food security in times of climate change, benefits in terms of both water security and various forms of protection against disaster on land (although the particular implications under climate change remain obscure in some cases), a key role in controlling coastal erosion and a general role in relieving ecosystem stresses such as fire.

In summary, protected area managers can in many instances contribute to wider efforts at mitigating climate change impacts and incidentally increase support and perhaps resources for management. Key points that have emerged from the following review are:

MAINTAIN AND INCREASE LARGE RESERVED AREAS: Large, unfragmented reserves are likely to be most useful in both resisting climate change and providing resilience within a landscape – increasing core reserve areas, linking reserves and developing effective buffers are therefore crucial first steps.

PLAN PROTECTED AREAS WITH DISASTER MITIGATION IN MIND: protected areas can play an important local role in protecting communities from climate-related floods and landslips and can play a nationally important role in terms of avalanche and coastal protection. As reefs, mangroves and upland forests are also at risk from climate change, choice of where best to put management resources should also be influenced by protective functions.

RECOGNIZE THE ROLE OF PROTECTED AREAS IN MAINTAINING TERRESTRIAL FOOD AND WATER SUPPLIES: the examples given show that small changes in management attitudes and practices can in many cases allow the natural vegetation within protected areas to help supply emergency food and forage and plentiful water, all likely to be in shorter supply in the future.

SITE MARINE PROTECTED AREAS TO MAINTAIN FISHERIES: as fish stocks suffer from multiple problems of over-exploitation, pollution and climate change, ways of maintaining breeding stocks become more important. The beneficial role of MPAs is now proven and the need to maintain fisheries should be a major factor in increasing reserve coverage from the current 1% of the ocean that is protected.

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Regional Biodiversity Impact Assessments for Climate Change: A Guide for Protected Area Managers

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PROTECTED AREAS MANAGERS ARE increasingly concerned about the possible impacts of climate change on the sites they manage. Tools for assessing these impacts are now often in desktop form, including Regional Climate Models (e.g., the Hadley Centre's PRECIS) and species range shift models (e.g., desktop GARP). However, while these tools are now within the computing capacities of some protected areas, they require a long time to run (up to 6 months for a regional climate model like PRECIS), coupled with specialized training and interpretation. For these reasons, protected areas managers will probably remain reliant on studies done by university researchers or assessment agencies for the immediate future. This chapter explores some of the issues protected areas managers need to understand to properly interpret and apply studies of regional biotic effects of climate change.

The chapter opens with a discussion of modeling tools now available for regional analyses. Protected areas managers need to be aware of these tools so that they can judge the quality and appropriateness of regional studies for application at their sites. The second part of the chapter outlines issues of interpreting these studies. Knowing how to interpret studies can help avoid either their being taken too literally or ignored due to the considerable uncertainty they carry. Finally, the chapter examines some modes of collaborative research between protected areas managers and researchers that may help move ahead both understanding of climate change impacts on biodiversity, and the formulation of effective conservation responses.

Assessment Tools

Major regional universities and biological research centers will increasingly have the capacity to generate assessments of biotic impacts of climate change. Regional Climate Models (RCMs) are now available which run on a desktop PC. Such RCMs give climate projections at a scale that is useful for regional impact analysis, in contrast to models of global climate, called General Circulation Models (GCMs), which are generally run at scales too coarse for meaningful regional analysis.

The output of a Regional Climate Model may be used in species range shift (or ‘niche’) models, to provide species-level projections of possible biotic impacts. Such models use the current climatic tolerances of a species to infer possible changes in its distribution due to alterations in climate. Biological impact assessments using these tools are likely to proliferate in the coming years, and not all will be of equal quality. It is therefore important to understand the broader range of assessment tools to put gauge the utility of these studies for protected areas planning.

Several major types of tools are available for assessing the impact of climate change on biodiversity. These include:

- global climate models,
- regional climate models,
- dynamic and equilibrium vegetation models,
- species bioclimatic envelope models (Figure 1), and
- site-specific sensitivity analysis.

Models of global climate, General Circulation Models (GCMs), provide broad resolution projections of future climate changes. A typical protected area occupies just a small fraction of a GCM grid cell, and there are substantial differences in projected climate changes among GCMs. Nonetheless, GCMs are an essential entry point for conservation assessments of climate change, since they represent the only source for estimates of future climate changes due to global greenhouse-gas forcing. Global GCM projections for several models are available on the internet (e.g., http://www.meto.govt.uk/research/hadleycentre/models/modeldata.html). Software is available on CD-ROM for personal computers which allows the comparison of simulated results from several models, which is useful given the considerable inter-GCM uncertainty (see Wigley et al., 2000 for mailing address for CD/software requests).

Most current GCM assessments are transient simulations, that is, they simulated a realistic, gradual buildup of greenhouse gases. Simulations that use an unrealistic, all-at-once increase are called equilibrium simulations and generally should be considered outdated. Equilibrium simulations (i.e., a step increase in CO_2) show increasing temperature change poleward in both hemispheres, while more sophisticated transient simulations show temperature change decreasing with latitude in the southern hemisphere outside of Antarctica. Northern and southern hemisphere climate system dynamics are markedly different and GCM hemispheric coupling is problematic, so models developed with a southern hemisphere focus (e.g. several excellent modeling exercises in Australia) may be more appropriate in southern hemisphere applications. Using inappropriate models or simulations may bias results, especially in the southern hemisphere. GCM relevance to biodiversity assessment is also improved by selecting results from fully-coupled ocean-atmosphere models appropriate to the region in question.

Regional climate models may be imbedded within GCMs to provide higher resolution results for use in regional assessments. Two major regional climate models in wide use are MM5 (Mesoscale Model version 5) and RAMS (Regional Atmospheric Modelling System). These models capture the regional influences that in some settings may be more important than global forcing in determining local climate changes. For instance, conversion of forest to pasture in the Amazon may produce local precipitation effects that overwhelm likely precipitation changes due to global greenhouse gas forcing. Regional models represent both the land-use changes and resultant cloud formation dynamics of this effect in ways impossible in a GCM. Regional models run at national or sub-continental scales useful in conservation planning. Their results are less widely available than those of GCMs and they are not available for all regions. However, the Hadley Centre has released a relatively new RCM called PRECIS, which is relatively simple to run on a personal computer, although it can take 6 months or more of continuous computing for a single model run (thus multiple runs for various emissions scenarios or multiple GCMs might take several computers or several years). Training for PRECIS is available in one-week workshops in many parts of the world. Regional climate modeling is therefore now within the grasp of most major regional universities.

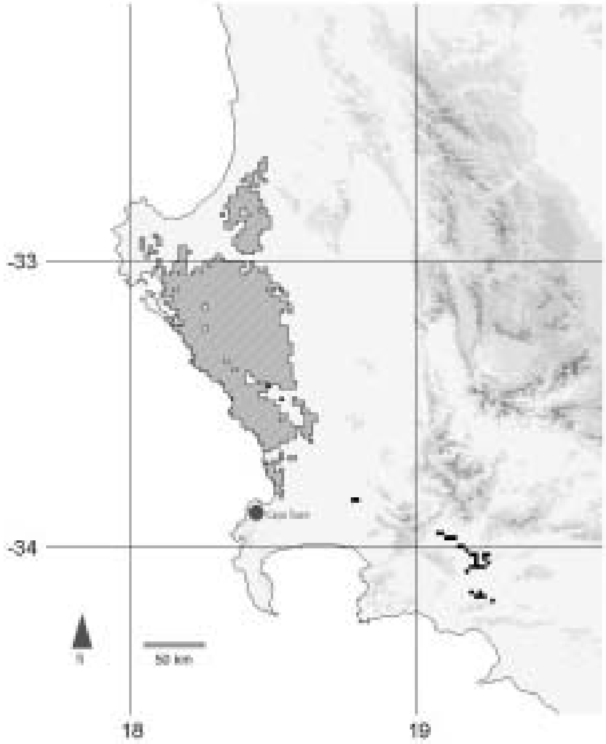
Dynamic vegetation models, forest ‘gap’ models, biome envelope (or ‘correlative’) models and species envelope models all use GCM and regional climate model results to provide insights into different aspects of the biogeography of future climate change. Dynamic Global Vegetation Models (DGVMs) use first principles of photosynthesis, carbon processing and plant physiology to predict plant functional types. Forest ‘gap’ models simulate species-specific succession dynamics at the stand-level (<1 ha), but have limited ability to represent landscape-level changes. They have data requirements that limit their application primarily, but not exclusively, to temperate forests. Global biome models use the climatic boundaries of current vegetation to simulate future distributions in changed climates. Global biome models assume vegetation is in equilibrium with climate and so cannot model dynamic transitions, while DGVMs incorporate dynamics but do not yield species specific results. Forest ‘gap’ models do both, but only for a small area and only for species for which growth and reproductive characteristics have been studied.

Species bioclimatic envelope models are the best available tool for producing the species-specific information necessary in conservation planning (Figure 1). They are similar in principle to biome envelope models, in that the present distribution of a species is used to ‘train’ a model to predict the climatic conditions in which the species may exist in the future. Envelope construction may be done manually on a GIS platform or through rule-based techniques such as genetic algorithms or general additive modeling . Unfortunately, these models currently face numerous limitations, including the inability to model dynamic transitions, the effects of inter-specific competition, herbivory, dispersal, or other factors (e.g. soil type in some models).

To make the results of species bioclimatic envelope models most applicable to real-world conservation problems they must be coupled with land-use projection models. Land use projection models represent the current pattern of habitat fragmentation and model future patterns based on projections of parameters such as population and consumption levels. The potential range shift of a species approximated by bioclimatic models is then reduced to the available habitat as projected by the land use model. For example, a species whose potential climate envelope shifts into an area entirely dominated by agriculture or urban development may be faced with extinction.

Integrative and sensitivity analysis based on the ecology of sites and individual species is an essential supplement to modeling, even if it may lack the attractive spatial specificity of models. Models cannot predict species composition at a landscape scale in a dynamic, competitive environment (dynamic vegetation models lack species-specificity, envelope models lack dynamic and competitive elements, 'gap' models lack spatial resolution). Evidence of paleoecological and paleobiogeographic responses to climate change form a central element of this analysis.

Figure 1



Bioclimatic model of the range shift of *Leucospermum tomentosum* in the Cape region of South Africa in a double CO_2 climate (approximately 2050). The GCM projection used is CSM without sulfates. Present modeled range is indicated by cross-hatching. Future modeled range is indicated by black rectangles. Figure courtesy of the Center for Applied Biodiversity Science at Conservation International.

Sensitivity analysis in a site assessment considers possible cooler climates, as well as anthropogenic warming. A sensitivity analysis asks what would happen if various climate variables changed, systematically testing each variable for changes that are both positive and negative in sign. The end result of this process is a picture of what responses may be expected under a wide range of climatic conditions. Paleoclimatic evidence suggests that global climates may be capable of switching rapidly between states. The possibility of reversal of current warming trends within centuries argues that sound conservation plans should be robust to both warming and possible cooling. An excellent introductory review of climate models, biogeographic models and sensitivity analysis in regional environmental assessment is given by (Sulzman et al., 1995).

Interpreting Modeling

Regional modeling, interpreted carefully, may provide critical input to practical conservation strategies. First, protected areas managers must accept that there are major uncertainties associated with climate change projection and species range shift models. This does not mean that their results are useless or should be ignored. Rather, it means that their results should be viewed as an aid in risk management. Protected areas management is often about identifying threats, weighing future risks and designing management strategies accordingly. Climate change is a threat which should be treated similarly.

An insurance analogy is useful in this context. Insurance companies provide a service that helps their customers manage risk. Their customers want to avoid future scenarios in which low probability but high cost events (such as an auto accident) disrupt their lives. They therefore weigh the likelihood of future high cost events and invest in insurance accordingly.

Protected areas management alternatives are often explored as insurance against future threats having unacceptable impacts on biodiversity. For instance, a park may have a management strategy in place for dealing with a large catastrophic fire, even if none has occurred in recent history. In the insurance analogy, the fire is a low probability event with high costs to biodiversity, so having a plan in place to minimize the damage is a sound investment, just as an insurance policy is often a sound investment.

This type of risk management applies to climate change as well. Any particular projection of the future is unlikely to be exactly correct, but it is wise to plan for a range of possible futures to balance risk and minimize large negative consequences. Certain effects of climate change, such as temperature increase, are now relatively high probability events, while other effects suggested in modeling may be low probability events, some with highly negative consequences. Sound application of modeling results in protected areas planning requires balancing the risk of possible future events and creating management strategies that minimize the probability and impact of scenarios that have large negative consequences for biodiversity.

The output of a typical single-species model is illustrated in Figure 1. While model output is expressed as a map of future species range, it should be recognized that this is only one of many possible alternatives for the future range of the species. Different climate models (both RCM and the GCM in which it is embedded), different emissions scenarios, different species models and different input data might all present different results. A first principle in judging regional modeling or assessments is therefore 'the more scenarios the better'.

Second, assessments can be conducted with coarse-scale GCM climate scenarios or finer-scale regional (RCM) projections. The finer scale of RCM output is much more appropriate to regional analysis, and RCMs capture the effects of regional land use change (such as effects on precipitation due to forest clearing) as well. As second general principle is therefore 'a study done with RCM climatology is more reliable than one that uses GCM climatology'.

However, many assessments in the near future will be limited to a few scenarios and GCM-scale climatology. What use, if any, are these assessments? Here a return to the risk management perspective is useful. If one has no information about the likelihood or severity of a future event, it is extremely difficult to manage the associated risk. Therefore, even one scenario of low probability can be a major help in management. It allows some assessment of risk, and allows other, independent lines of evidence to be explored to refine the estimate of risk.

For example, a range shift projection such as illustrated in Figure 1 provides one assessment of the relative vulnerability of the species to long-distance dislocation due to climate change. This estimate can be refined by examining other relevant factors, such as land use in the direction of shift, dispersal capability of the species, and other factors. The modeling itself may also be examined to determine what climatic factors drove the simulated shift. If these factors seem relevant for the species, the probability of the projection goes up. By adding projections and lines of evidence, uncertainty can be reduced, negative impacts in the future balanced, and management strategies evolved.

Protected Areas Management

Refinement of management practices in response to the results of an impact assessment can be done in four major steps:

- scenario-building;
- enhanced monitoring;
- biological survey; and
- review and revision of management practices.

Scenario-building is an iterative process in which modeling is used to refine management and management revisions suggest further areas of enquiry for modeling. Scenarios are created that span the range of uncertainty in climate change and biotic response modeling, and that capture important management variables. Monitoring and management are tested repeatedly against the scenarios and the scenarios themselves are repeatedly revised as more data becomes available and uncertainties change or decrease.

Scenarios should be created that capture possible major ecological events in the system being conserved. For instance, dynamic vegetation or envelope model results should be examined for biomes or habitats 'on the edge'—systems that are near threshold for conversion to a different growth form, dominant vegetation or disturbance regime. Scenarios should also be constructed for rare, threatened and climate-sensitive species. Rare and threatened species may be vulnerable to further population reductions due to climate change, and these should be considered in management plans for these species. Climate-sensitive species include species with small ranges (even if abundant), species with limited (<500-1000 m) elevational ranges, and upper elevation species whose habitat may be reduced with warming. Finally, scenarios should be constructed that describe the possible impact of climate change on ecosystem processes. Droughts and storms often limit plant functional types or open forest canopies for regeneration. Change in frequency of these events may therefore alter vegetation structure, succession, and species diversity and composition.

An expanded monitoring program is based on the scenarios developed. Testable scenario predictions monitored in the field permit adaptive management responses. Many parameters of enhanced modeling will be biological, including climate-sensitive species and processes. Installation or upgrading of weather-data gathering capability is a physical monitoring step to be considered. Collection of sound weather data has proven important in documenting climate correlates to species range changes, changes in abundance (amphibian decline) and even possible extinctions in the Monteverde cloud forests of Costa Rica. Remote sensing and regional modeling may help in the design of a monitoring system which focuses on variables that may be vulnerable to change, for instance lifting cloud bases in tropical montane settings such as Monteverde (Lawton et al. 2001).

Biological survey work can complement monitoring and scenario refinement by providing key data. Detecting individualistic species range shifts requires data on distribution and abundance generally not available nor previously considered necessary at most protected areas. Survey programs can help fill this data need and provide baseline data for monitoring. For example, scenarios from modeling may show that a species not known from a reserve may find favorable climatic conditions there in the future (Rutherford et al., 1999). Such species may exist in the reserve but have escaped documentation. Survey work can help find outliers of the species or increase confidence that it does not exist in the reserve, information critical to the design of effective monitoring systems. Additional distributional data even on common species may be required for effective monitoring. Inexpensive GPS units make park staff on regular patrol or even tourists on remote trails potential data gathering allies in this effort. Additionally, species range shifts may respond not just to climatological changes, but to changes in community interactions as well (e.g. Harley, 2003). Biological surveys will help refine models to reflect key biotic as well as abiotic variables.

Review and revision of management practices is the final step in an iterative process of revision of management based on modeling, scenarios, monitoring and survey. Modeling results and management scenarios will suggest management practices to be reviewed and revised. Examples of management practices that will often qualify for review are management of fire or other disturbance regimes, classification of ‘sensitive’ areas, and management for ‘representative’ species.

Fire and other disturbance regimes are often intensively managed in protected areas. These management practices will interact with climate change effects in ways that may not be apparent without careful monitoring. Fire may maintain certain vegetation types past their climatic optimum, or, if managed uncritically, suppress new vegetation types that are becoming climatically favored. For example, in Central Canada, long-grass prairie is predicted to be climatically favored over present forest types in future warmer climates (Scott and Suffling, 2000). Fire suppression may retard this transition. Fire management therefore has an effect that must be judged against regional conservation goals—either maintenance of forest or promotion of grassland in newly suitable climate space.

Sensitive areas form an important part of management in many protected areas. Climate change will introduce a new class of sensitive areas. Climate change-driven alterations in range or abundance may render once resilient species sensitive. Rapid range shifts may make robust systems sensitive. Changes in disturbance regime may create new or recovering vegetation sensitive to many types of use. Non-analogue communities may arise with unknown sensitivity requiring conservative management until they are more fully understood. Heavy tourist traffic may facilitate dispersal of invasive species into areas vulnerable because they are in transition to new vegetation types. These and other climate change sensitivities should be considered as sensitive areas are designated and managed.

Many protected areas are established or managed to conserve ‘representative’ ecosystems that may no longer exist in future climates. Minor vegetational elements or even outlier pockets may become dominant vegetation types in the future. Site goals will be difficult to set for changing vegetation without reference to regional trends and conservation goals. In the Central Canada example above, management for ‘representative’ forest is appropriate if the regional management goal is to retard biotic response to climate change, while promoting fire to stimulate transition to long grass prairie is appropriate if the regional goal is to allow natural transitions to take place while maintaining representation goals in a flexible regional protected areas network. Many other management issues will evolve from a systematic modeling, and management scenario analysis.

Finally, almost all protected areas management plans have 3- to 10-year time horizons, which are insufficient to allow for anticipatory management responses to climate change. A minimum appropriate planning time horizon for climate change is 50 years, while a 100-year horizon is necessary to capture many possible climate change effects.

Incorporating sensitivity analysis and climate change management scenarios into a management plan will require that at least part of the management plan has a longer time horizon.

Species range shifts, impacts of extreme events and resource asynchronies often occur on regional scales, so an effective management strategy includes mechanisms for coordinating conservation actions at the regional level. Regional coordination is necessary for conservation goals and management to be coherent on the same scale at which these climate change impacts will operate. Examples above show that managing for 'representative' vegetation is a relative term at the site level when climate is changing. Regional goals for representation can only be maintained in a dynamic climate when management at multiple protected areas is harmonized (Rutherford et al., 1999). This coordinated management may require formal agreements, for instance when national boundaries are crossed, or may simply involve appropriate planning within existing protected areas systems and conservation agencies.

Modeling and monitoring will often be more effective when coordinated within a region. Monitoring must be done in a way that is relevant to management goals, so regional goals require regionally coordinated monitoring. Sharing technical and financial inputs for modeling across multiple users on a regional basis increases cost-effectiveness as well. Regional coordination will become increasingly important as climate change progresses. In the short-term, identifying and establishing these collaborations is a priority. Peace Parks initiatives and other collaborative management efforts are already paving the way for these systems.

Collaborative Research

Creation of a climate change-integrated conservation strategy requires synergy among a novel set of actors and funding sources. Conservation managers, biogeographers, ecologists and climate change scientists are all needed to formulate an effective management strategy. Funding from research sources will be required for modeling and assessment activities with clear connections to applied conservation. Conservation agencies will need to source funding for major new investments in monitoring and revision of management practices. Creation of this synergy will carry a cost, and responding to the new challenges of climate change to the conservation of biodiversity will require major new financial commitments.

In a world filled with conservation challenges, managers will not be able to undertake all of the elements of climate change-integrated conservation strategies described here in the short term. What is important, is that managers, biogeographers and ecologists begin to consider the impacts of climate change in their area, and adopt at least some elements of a management strategy, progressively building capacity at the local level as the challenges posed by climate change mount.

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WWF Climate Change Program

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- to ensure that industrialized nations make substantial reductions in their domestic emissions of carbon dioxide—the main global warming gas—by 2010
- to promote the use of clean renewable energy in the developing world
- to reduce the vulnerability of nature and economies to the impacts of climate change

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