

Land Resources: Forests and Arid Lands

Lead Authors: M.G. Ryan and S.R. Archer

Contributing Authors: R.A. Birdsey, C.N. Dahm, L.S. Heath, J.A. Hicke, D.Y. Hollinger, T.E. Huxman, G.S. Okin, R. Oren, J.T. Randerson, W.H. Schlesinger

3.1 INTRODUCTION

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

In this chapter the focus is on the near-term future. In some cases, key results are reported out to 100 years to provide a larger context but the emphasis is on next 25-50 years. This nearer-term focus is chosen for two reasons. First, for many natural resources, planning and management activities already address these time scales through development of long-lived infrastructure, forest rotations, and other significant investments. Second, climate projections are relatively certain over the next few decades. Emission scenarios for the next few decades do not diverge from each other significantly because of the “inertia” of the energy system.

Most projections of greenhouse gas emissions assume that it will take decades to make major changes in the energy infrastructure, and only begin to diverge rapidly after several decades have passed (30-50 years).

Forests occur in all 50 states but are most common in the humid eastern United States, the West Coast, at higher elevations in the Interior West and Southwest, and along riparian corridors in the plains states (Figure 3.1) (Zhu and Evans 1994). Forested land occupies about 740 million acres, or about one-third of the United States. Forests in the eastern United States cover 380 million acres; most of this land (83 percent) is privately owned, and 74 percent is broadleaf forest. The 360 million acres of forest land in the western United States are 78 percent conifer forests, split between public (57 percent) and private ownership (USDA Forest Service and U.S. Geological Survey 2002).

Forests provide many ecosystem services important to the well-being of the people of the United States: watershed protection, water quality, and flow regulation; wildlife habitat and diversity; recreational opportunities, and aesthetic and spiritual fulfillment; raw material for wood and paper products; climate regulation, carbon storage, and air quality; biodiversity conservation. While all of these services have considerable economic value, some are not easily quantified (Costanza et al. 1997; Daily et al. 2000;

Forested land occupies about 740 million acres, or about one-third of the United States.

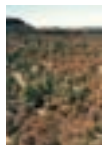




Figure 3.1 Distribution of forest lands in the continental United States by forest type. This map was derived from Advanced Very High Resolution Radiometer (AVHRR) composite images recorded during the 1991 growing season. Each composite covered the United States at a resolution of one kilometer. Field data collected by the Forest Service were used to aid classification of AVHRR composites into forest-cover types. Details on development of the forest cover types dataset are in Zhu and Evans (1994).

and open spaces for expanding urban environments. A changing climate will alter arid lands and their services.

Both forests and arid lands face challenges that can affect their responses to a changing climate: the legacy of historical land use, non-native invasive species, and the slow growth of many species. In forests, for instance, clearing and farming dramatically increased erosion, and the re-established forests are likely less productive as a result. In arid lands, grazing and exurban development can change plant and animal communities. Non-native invasive species are a challenge for all ecosystems, but especially so in arid lands, where non-native invasive grasses encourage fire in ecosystems where fire was historically very rare. The very slow growth of many arid land and dry forest species hinders recovery from disturbance.

Climate strongly influences both forests and arid lands. Climate shapes the broad patterns of ecological communities, the species within them, their productivity, and the ecosystem goods and services they provide. The interaction of vegetation and climate is a fundamental tenet of ecology. Many studies show how vegetation has changed with climate over the past several thousand years, so it is well understood that changes in climate will change vegetation. Given a certain climate and long enough time, resultant ecological communities can generally be predicted. However, predicting the effects of a changing climate on forests and arid lands for the next few decades is challenging, especially with regard to the rates and dynamics of change. Plants in these communities can be long lived; hence, changes in species composition may lag behind changes in climate. Furthermore, seeds and conditions for better-adapted communities are not always present.

Krieger 2001; Millennium Ecosystem Assessment 2005), and many Americans are strongly attached to their forests. A changing climate will alter forests and the services they provide. Sometimes changes will be viewed as beneficial, but often they will be viewed as detrimental. Arid lands are defined by low and highly variable precipitation, and are found in the United States in the subtropical hot deserts of the Southwest and the temperate cold deserts of the Intermountain West (Figure 3.2). Arid lands provide many of the same ecosystem services as forests (with the exception of raw materials for wood and paper products), and support a large ranching industry. These diverse environments are also valued for their wildlife habitat, plant and animal diversity, regulation of water flow and quality, opportunities for outdoor recreation,

Past studies linking climate and vegetation may also provide poor predictions for the future because the same physical climate may not occur in the future and many other factors may be changing as well. CO₂ is increasing in the atmosphere; nitrogen deposition is much greater than in the past, and appears to be increasing; ozone pollution is locally increasing; and species invasions from other ecosystems are widespread. These factors cause important changes themselves, but their interactions are difficult to predict because they represent novel combinations.

Disturbance (such as drought, storms, insect outbreaks, grazing, and fire) is part of the ecological history of most ecosystems and influences ecological communities and landscapes. Climate affects the timing, magnitude, and frequency of many of these disturbances, and a changing climate will bring changes in disturbance regimes to forests and arid lands (Dale et al. 2001). Trees and arid land vegetation can take from decades to centuries to re-establish after a disturbance. Both human-induced and natural disturbances shape ecosystems by influencing species composition, structure, and function (productivity, water yield, erosion, carbon storage, and susceptibility to future disturbance). In forests, more than 55 million acres are currently impacted by disturbance, with the largest agents being insects and pathogens (Dale et al. 2001). These disturbances cause an estimated financial loss of 3.7 billion dollars per year (Dale et al. 2001). In the past several years, scientists have learned that the magnitude and impact of these disturbances and their response to climate rivals that expected from changes in temperature and precipitation (Field et al. 2007).

Disturbance may reset and rejuvenate some ecosystems in some cases and cause enduring change in others. For example, climate may favor the spread of invasive exotic grasses into arid lands where the native vegetation is too sparse to carry a fire. When these areas burn, they typically convert to non-native monocultures and the native vegetation is lost. In another example, drought may weaken trees and make them susceptible to insect attack and death – a pattern that recently occurred in the Southwest. In these forests, drought and insects converted

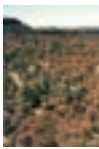


Figure 3.2 The five major North American deserts, outlined on a 2006 map of net primary productivity (NPP). Modeled NPP was produced by the Numerical Terradynamic Simulation Group (<http://www.ntsg.umd.edu/>) using the fraction of absorbed photosynthetically active radiation measured by the Moderate Resolution Imaging Spectroradiometer (MODIS) satellite and land cover-based radiation use efficiency estimates Running et al. (2000). Desert boundaries based on Olson et al. (2001).

large areas of mixed pinyon-juniper forests into juniper forests. However, fire is an integral component of many forest ecosystems, and many tree species (such as the lodgepole pine forests that burned in the Yellowstone fires of 1988) depend on fire for regeneration. Climate effects on disturbance will likely shape future forests and arid lands as much as the effects of climate itself.

Disturbances and changes to the frequency or type of disturbance present challenges to resource managers. Many disturbances command quick action, public attention, and resources. Surprisingly, most resource planning in the United States does not consider disturbance, even though disturbances are common, and preliminary information exists on the frequency and areal extent of disturbances (Dale et al.

Arid lands are defined by low and highly variable precipitation, and are found in the United States in the subtropical hot deserts of the Southwest and the temperate cold deserts of the Intermountain West.



Climate strongly influences both forests and arid lands. Climate shapes the broad patterns of ecological communities, the species within them, their productivity, and the ecosystem goods and services they provide.



2001). Disturbances in the future may be larger and more common than those experienced historically, and planning for disturbances should be encouraged (Dale et al. 2001; Stanturf et al. 2007).

The goal of this chapter is to assess how forests and arid lands will respond to predicted anticipated changes in climate over the next few decades. It will discuss the effects of climate and its components on the structure and function of forest and arid land ecosystems. It will also highlight the effects of climate on disturbance and how these disturbances change ecosystems. Active management may increase the resiliency of forests and arid lands to respond to climate change. For example, forest thinning can reduce fire intensity, increase drought tolerance and reduce susceptibility to insect attack. Grazing management and control of invasive species can promote vegetation cover, reduce fire risk, and reduce erosion. These and other options for managing ecosystems to adapt to climate change are discussed in Synthesis and Assessment Product 4.4 (Preliminary review of adaptation options for climate-sensitive ecosystems and resources, U.S. Climate Change Science Program).

3.2 FORESTS

3.2.1 Brief Summary of Key Points from the Literature

Climate strongly affects forest productivity and species composition. Forest productivity in the United States has increased 2-8 percent in the past two decades, but separating the role of climate from other factors causing the increase is complicated and varies by location. Some factors that act to increase forest growth are 1) observed increases in precipitation in the Midwest and Lake States, 2) observed increases in nitrogen deposition, 3) an observed increase in temperature in the northern United States that lengthens the growing season, 4) changing age structure of forests, and 5) management practices. These factors interact, and identifying the specific cause of a productivity change is complicated by insufficient data. Even in the case of large forest mortality events, such as those associated with fire and insect outbreaks, attributing a specific event to a change in climate may be difficult because of interactions among factors. For example, in the recent widespread

mortality of pinyon pine in the Southwest, intense drought weakened the trees, but generally, the Ips beetle killed them.

In addition to the direct effects of climate on tree growth, climate also affects the frequency and intensity of natural disturbances such as fire, insect outbreaks, ice storms, and windstorms. These disturbances have important consequences for timber production, water yield, carbon storage, species composition, invasive species, and public perception of forest management. Disturbances also draw management attention and resources. Because of observed warmer and drier climate in the West in the past two decades, forest fires have grown larger and more frequent during that period. Several large insect outbreaks have recently occurred or are occurring in the United States. Increased temperature and drought likely influenced these outbreaks. Fire suppression and large areas of susceptible trees (over age 50) may have also contributed.

Rising atmospheric CO₂ will increase forest productivity and carbon storage in forests if sufficient water and nutrients are available. Any increased carbon storage will be primarily in live trees. Average productivity increase for a variety of experiments was 23 percent. The response of tree growth and carbon storage to elevated CO₂ depends on site fertility, water availability, and stand age, with fertile, younger stands responding more strongly.

Forest inventories can detect long-term changes in forest growth and species composition, but they have limited ability to attribute changes to specific factors, including climate. Separating the effects of climate change from other impacts would require a broad network of indicators, coupled with a network of controlled experimental manipulations. Experiments that directly manipulate climate and observe impacts are critical components in understanding climate change impacts and in separating the effects of climate from those caused by other factors. Experiments such as free-air CO₂ enrichment, ecosystem and soil warming, and precipitation manipulation have greatly increased understanding of the direct effects of climate on ecosystems. These experiments have also attracted a large research community and fostered a thorough and integrated understanding because of their

large infrastructure costs, importance and rarity. Monitoring of disturbances affecting forests is currently ineffective, fragmented, and generally unable to attribute disturbances to specific factors, including climate.

3.2.2 Observed Changes or Trends

3.2.2.1 CLIMATE AND ECOSYSTEM CONTEXT

Anyone traveling from the lowlands to the mountains will notice that species composition changes with elevation and with it, the structure and function of these forest ecosystems. Biogeographers have mapped these different vegetation zones and linked them with their characteristic climates. The challenge facing scientists is to understand how these zones and the individual species within them will move with a changing climate, at what rate, and with what effects on ecosystem function.

Temperature, water, and radiation are the primary abiotic factors that affect forest productivity (Figure 3.3). Any response to changing climate will depend on the factors that limit production at a particular site. For example, any site where productivity is currently limited by lack of water or a short growing season will increase productivity if precipitation increases and if the growing season lengthens. Temperature controls the rate of metabolic processes for photosynthesis, respiration, and growth. Generally, plant

metabolism has an optimum temperature. Small departures from this optimum usually do not change metabolism and short-term productivity, although changes in growing season length may change annual productivity. Large departures and extreme events (such as frosts in orange groves) can cause damage or tree mortality. Water controls cell division and expansion, which promote growth and stomatal opening, which regulates water loss and CO₂ uptake in photosynthesis. Productivity will generally increase with water availability in water-limited forests (Knapp et al. 2002). Radiation supplies the energy for photosynthesis, and both the amount of leaf area and incident radiation control the quantity of radiation absorbed by a forest. Nutrition and atmospheric CO₂ also strongly influence forest productivity if other factors are less limiting (Boisvenue and Running 2006), and ozone exposure can lower productivity (Hanson et al. 2005). Human activities have increased nitrogen inputs to forest ecosystems, atmospheric CO₂ concentration, and ozone levels. The effects of CO₂ are everywhere, but ozone and N deposition are common to urban areas, and forests and arid lands downwind from urban areas. The response to changes in any of these factors is likely to be complex and dependent on the other factors.

Forest trees are evolutionarily adapted to thrive in certain climates. Other factors, such as fire and competition from other plants, also regulate species presence, but if climate alone changes enough, species will adjust to suitable conditions

Temperature, water, and radiation are the primary abiotic factors that affect forest productivity.

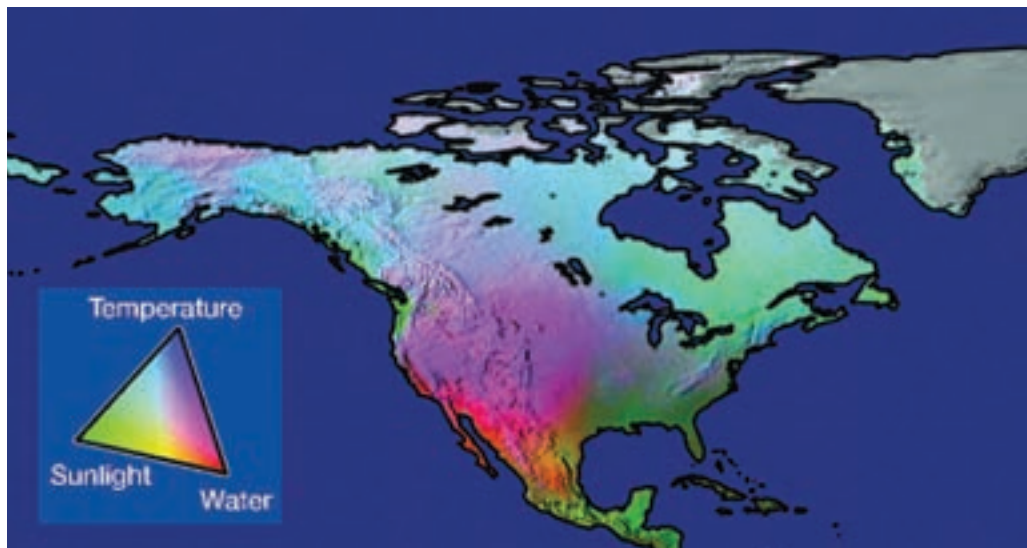


Figure 3.3 Potential limits to vegetation net primary production based on fundamental physiological limits by sunlight, water balance, and temperature. Nutrients are also important and vary locally. From Boisvenue and Running (2006).



or go locally extinct if suitable conditions are unavailable (Woodward 1987). One example of such a species shift is sugar maple in the north-eastern United States. Suitable climate for it may move northward into Canada and the distribution will likely follow (Chuine and Beaubien 2001), assuming the species is able to disperse propagules rapidly enough to keep pace with the shifting climatic zone. Because trees live for decades and centuries, absent disturbance, it is likely that forest species composition will take time to adjust to changes in climate.

Disturbances such as forest fires, insect outbreaks, ice storms, and hurricanes also change forest productivity, carbon cycling, and species composition. Climate influences the frequency and size of disturbances. Many features of ecosystems can be predicted by forest age, and disturbance regulates forest age. After a stand-replacing disturbance, forest productivity increases until the forest fully occupies the site or develops a closed canopy, then declines to near zero in old age (Ryan et al. 1997). Carbon storage after a disturbance generally declines while the decomposition of dead wood exceeds the productivity of the new forest, then increases as the trees grow larger and the dead wood from the disturbance disappears (Kashian et al. 2006). In many forests, species composition also changes with time after disturbance. Susceptibility to fire and insect outbreaks changes with forest age, but the response of forest productivity to climate, N deposition, CO₂, and ozone differs for old and young forests is still not understood because most studies have only considered young trees or forests. Changes in disturbance prompted by climate change are likely as important as the changes in precipitation, temperature, N deposition, CO₂, and ozone for affecting productivity and species composition.

3.2.2.2 TEMPERATURE

Forest productivity in the United States has generally been increasing since the middle of the 20th century (Boisvenue and Running 2006), with an estimated increase of 2-8 percent between 1982 and 1998 (Hicke et al. 2002b), but the causes of this increase (increases in air and surface temperature, increasing CO₂, N deposition, or other factors) are difficult to isolate (Cannell et al. 1998). These effects can

potentially be disentangled by experimentation, analysis of species response to environmental gradients, planting trees from seeds grown in different climates in a common garden, anomaly analysis, and other methods. Increased temperatures will affect forest growth and ecosystem processes through several mechanisms (Hughes 2000; Saxe et al. 2001) including effects on physiological processes such as photosynthesis and respiration, and responses to longer growing seasons triggered by thermal effects on plant phenology (e.g., the timing and duration of foliage growth). Across geographical or local elevational gradients, forest primary productivity has long been known to increase with mean annual temperature and rainfall (Leith 1975). This result also generally holds within a species (Fries et al. 2000) and in provenance trials where trees are found to grow faster in a slightly warmer location than that of the seed source itself (Wells and Wakeley 1966; Schmidting 1994). In Alaska, where temperatures have warmed strongly in recent times, changes in soil processes are similar to those seen in experimental warming studies (Hinzman et al. 2005). In addition, permafrost is melting, exposing organic material to decomposition and drying soils (Hinzman et al. 2005).

Along with a general trend in warming, the length of the northern hemisphere growing season has been increasing in recent decades (Menzel and Fabian 1999; Tucker et al. 2001). Forest growth correlates with growing season length (Baldocchi et al. 2001), with longer growing seasons (earlier spring) leading to enhanced net carbon uptake and storage (Black et al. 2000; Hollinger et al. 2004). The ability to complete phenological development within the growing season is a major determinant of tree species range limits (Chuine and Beaubien 2001). However, Sakai and Weiser (1973) have also related range limits to the ability to tolerate minimum winter temperatures.

3.2.2.3 FIRE AND INSECT OUTBREAKS

Westerling et al. (2006) analyzed trends in wildfire and climate in the western United States from 1974–2004. They show that both the frequency of large wildfires and fire season length increased substantially after 1985, and that these changes were closely linked with advances in the timing of spring snowmelt,



and increases in spring and summer air temperatures. Much of the increase in fire activity occurred in mid-elevation forests in the northern Rocky Mountains and Sierra Nevada mountains. Earlier spring snowmelt probably contributed to greater wildfire frequency in at least two ways, by extending the period during which ignitions could potentially occur, and by reducing water availability to ecosystems in mid-summer, thus enhancing drying of vegetation and surface fuels (Westerling et al. 2006). These trends in increased fire size correspond with the increased cost of fire suppression (Calkin et al. 2005).

In boreal forests across North America, fire activity also has increased in recent decades. Kasischke and Turetsky (2006) combined fire statistics from Canada and Alaska to show that burned area more than doubled between the 1960s/70s and the 1980s/90s. The increasing trend in boreal burned-area appears to be associated with a change in both the size and number of lightning-triggered fires (>1000 km²), which increased during this period. In parallel, the contribution of human-triggered fires to total burned area decreased from the

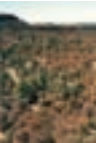
1960s to the 1990s (from 35.8 percent to 6.4 percent) (Kasischke and Turetsky 2006). As in the western U.S., a key predictor of burned area in boreal North America is air temperature, with warmer summer temperatures causing an increase in burned area on both interannual and decadal timescales (Gillett et al. 2004; Duffy et al. 2005; Flannigan et al. 2005). In Alaska, for example, June air temperatures alone explained approximately 38 percent of the variance of the natural log of annual burned area during 1950-2003 (Duffy et al. 2005).

Insects and pathogens are significant disturbances to forest ecosystems in the United States (Figure 3.4), costing \$1.5 billion annually (Dale et al. 2001). Extensive reviews of the effects of climate change on insects and pathogens have reported many cases where climate change has affected and/or will affect forest insect species range and abundance (Ayres and Lombardero 2000; Malmström and Raffa 2000; Bale et al. 2002). This review focuses on forest insect species within the United States that are influenced by climate and attack forests that are ecologically or economically important.



Figure 3.4 Satellite image of the extensive attack by mountain pine beetle in lodgepole pine forests in Colorado. Pre-outbreak image taken October 2002, and post outbreak image taken August 2007. Images courtesy of DigitalGlobe, Inc. (<http://digitalglobe.com/>)

For the projected temperature increases over the next few decades, most studies support the conclusion that a modest warming of a few degrees Celsius will lead to greater tree growth in the United States.



Major outbreaks in recent years include: a mountain pine beetle outbreak affected >10 million hectares (Mha) of forest in British Columbia (Taylor et al. 2006), and 267,000 ha in Colorado (Colorado State Forest Service 2007); more than 1.5 Mha was attacked by spruce beetle in southern Alaska and western Canada (Berg et al. 2006); >1.2 Mha of pinyon pine mortality occurred because of extreme drought, coupled with an *Ips* beetle outbreak in the Southwest (Breshears et al. 2005); and millions of hectares were affected by southern pine beetle, spruce budworm, and western spruce budworm in recent decades in southeastern, northeastern, and western forests, respectively (USDA Forest Service 2005). Ecologically important whitebark pine is being attacked by mountain pine beetle in the northern and central Rockies (Logan and Powell 2001). For example, almost 70,000 ha, or 17 percent, of whitebark pine forest in the Greater Yellowstone Ecosystem is infested by mountain pine beetle (Gibson 2006). Evident from these epidemics is the widespread nature of insect outbreaks in forests throughout the United States.

Climate plays a major role in driving, or at least influencing, infestations of these important forest insect species in the United States (e.g., Holsten et al. 1999; Logan et al. 2003a; Carroll et al. 2004; Tran et al. in press), and these recent large outbreaks are likely influenced by observed increases in temperature. Temperature controls life cycle development rates, influences synchronization of mass attacks required to overcome tree defenses, and determines winter mortality rates (Hansen et al. 2001b; Logan and Powell 2001; Hansen and Bentz 2003; Tran et al. in press). Climate also affects insect populations indirectly through effects on hosts. Drought stress, resulting from decreased precipitation and/or warming, reduces the ability of a tree to mount a defense against insect attack (Carroll et al. 2004; Breshears et al. 2005), though this stress may also cause some host species to become more palatable to some types of insects (Koricheva et al. 1998). Fire suppression and large areas of susceptible trees (a legacy from logging in the late 1800s and early 1900s (Birdsey et al. 2006)), may also play a role.

3.2.3 Possible Future Changes and Impacts

3.2.3.1 WARMING

A review of recent experimental studies found that rising temperatures would generally enhance tree photosynthesis (Saxe et al. 2001), as a result of increased time operating near optimum conditions, and because rising levels of atmospheric CO₂ increase the temperature optimum for photosynthesis (Long 1991). Warming experiments, especially for trees growing near their cold range limits, generally increase growth (Bruhn et al. 2000; Wilmking et al. 2004; Danby and Hik 2007). The experimental warming of soils alone has been found to stimulate nitrogen mineralization and soil respiration (Rustad et al. 2001). An important concern for all experimental manipulations is that the treatments occur long enough to determine the full suite of effects. It appears that the large initial increases in soil respiration observed at some sites decrease with time back toward pretreatment levels (Rustad et al. 2001; Melillo et al. 2002). This result may come about from changes in C pool sizes, substrate quality (Kirschbaum 2004; Fang et al. 2005), or other factors (Davidson and Janssens 2006).

A general response of leaves, roots, or whole trees to short-term increases in plant temperature is an approximate doubling of respiration with a 10°C temperature increase (Ryan et al. 1994; Amthor 2000). Over the longer term, however, there is strong evidence for temperature acclimation (Atkin and Tjoelker 2003; Wythers et al. 2005), which is probably a consequence of the linkage of respiration to the production of photosynthate (Amthor 2000). One negative consequence of warming for trees is that it can increase the production of isoprene and other hydrocarbons in many tree species (Sharkey and Yeh 2001) – compounds that may lead to higher levels of surface ozone and increased plant damage. Physiologically, the overall result of the few degrees of warming expected over the next few decades is likely a modest increase in photosynthesis and tree growth (Hyvonen et al. 2007). However, where increased temperature coincides with decreased precipitation (western Alaska, Interior West, Southwest), forest growth is expected to be lower (Hicke et al. 2002b).

For the projected temperature increases over the next few decades, most studies support the conclusion that a modest warming of a few degrees Celsius will lead to greater tree growth in the United States. There are many causes for this enhancement including direct physiological effects, a longer growing season, and potentially greater mineralization of soil nutrients. Because different species may respond somewhat differently to warming, the competitive balance of species in forests may change. Trees will probably become established in formerly colder habitats (more northerly, higher altitude) than at present.

3.2.3.2 CHANGES IN PRECIPITATION

Relationships between forest productivity and precipitation have been assessed using continental gradients in precipitation (Webb et al. 1983; Knapp and Smith 2001), interannual variability within a site (Hanson et al. 2001), and by manipulating water availability (Hanson et al. 2001). Forest productivity varies with annual precipitation across broad gradients (Webb et al. 1983; Knapp and Smith 2001), and with interannual variability within sites (Hanson et al. 2001). Some of these approaches are more informative than others for discerning climate change effects.

Gradient studies likely poorly predict the response to changes in precipitation, because site-specific factors such as site fertility control the response to precipitation (Gower et al. 1992; Maier et al. 2004). The response of forest productivity to interannual variability also likely poorly predicts response to precipitation changes, because forests have the carbohydrate storage and deep roots to offset drought effects over that time, masking any effects that might be apparent over a longer-term trend.

The effects of precipitation on productivity will vary with air temperature and humidity. Warmer, drier air will evaporate more water and reduce water availability faster than cooler, humid air. Low humidity also promotes the closure of stomata on leaves, which reduces photosynthesis and lowers productivity even where soil water availability is abundant.

Manipulation of water availability in forests allows an assessment of the direct effects of

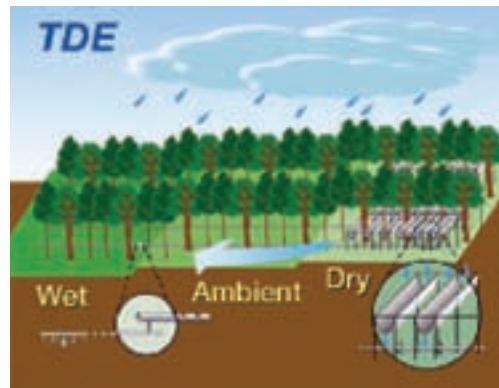


Figure 3.5 Direct manipulation of precipitation in the Throughfall Displacement experiment (TDE) at Walker Branch (Paul Hanson, Oak Ridge National Laboratory).

precipitation (Figure 3.5). Two experiments where water availability was increased through irrigation showed only modest increases in forest production (Gower et al. 1992; Maier et al. 2004), but large increases with a combination of irrigation and nutrients. In contrast, forest productivity did not change when precipitation was increased or reduced 33 percent, but with the same timing as natural precipitation (Hanson et al. 2005). Tree growth in this precipitation manipulation experiment also showed strong interannual variability with differences in annual precipitation. Hanson et al. (2005) conclude that “differences in seasonal patterns of rainfall within and between years have greater impacts on growth than percentage changes in rainfall applied to all rainfall events.”

No experiments have assessed the effect of changes in precipitation on forest tree species composition. Hanson et al. (2005) showed that growth and mortality changed in response to precipitation manipulation for some smaller individuals, but we do not know if these changes will lead to composition changes. However, one of the best understood patterns in ecology is the variation of species with climate and site water balance. So, if precipitation changes substantially, it is highly likely that species composition will change (Breshears et al. 2005). However, limited studies exist with which to predict the rate of change and the relationship with precipitation amount.

Drought is a common feature of all terrestrial ecosystems (Hanson and Weltzin 2000), and generally lowers productivity in trees. Drought



events can have substantial and long-lasting effects on ecosystem structure, species composition, and function by differentially killing certain species or sizes of trees (Hanson and Weltzin 2000; Breshears et al. 2005), weakening trees to make them more susceptible to insect attacks (Waring 1987), or by increasing the incidence and intensity of forest fires (Westerling et al. 2006). Forest management by thinning trees can improve water available to the residual trees. (Donner and Running 1986; Sala et al. 2005).

If existing trends in precipitation continue, forest productivity will likely decrease in the Interior West, the Southwest, eastern portions of the Southeast, and Alaska. Forest productivity will likely increase in the northeastern United States, the Lake States, and in western portions of the Southeast. An increase in drought events will very likely reduce forest productivity wherever these events occur.

3.2.3.3 ELEVATED ATMOSPHERIC CO₂ AND CARBON SEQUESTRATION

The effects of increasing atmospheric CO₂ on carbon cycling in forests are most realistically

observed in Free Air CO₂ Enrichment (FACE) experiments (Figure 3.6). These experiments have recently begun to provide time-series sufficiently long for assessing the effect of CO₂ projected for the mid-21st century on some components of the carbon cycle. The general findings from a number of recent syntheses using data from the three American and one European FACE sites (King et al. 2004; Norby et al. 2005; McCarthy et al. 2006a; Palmroth et al. 2006) show that North American forests will absorb more CO₂ and might retain more carbon as atmospheric CO₂ increases. The increase in the rate of carbon sequestration will be highest (mostly in wood) on nutrient-rich soils with no water limitation and will decrease with decreasing fertility and water supply. Several yet unresolved questions prevent a definitive assessment of the effect of elevated CO₂ on other components of the carbon cycle in forest ecosystems:

- Although total carbon allocation to below-ground increases with CO₂ (King et al. 2004; Palmroth et al. 2006), there is only equivocal evidence of CO₂-induced increase in soil carbon (Jastrow et al. 2005; Lichter et al. 2005).

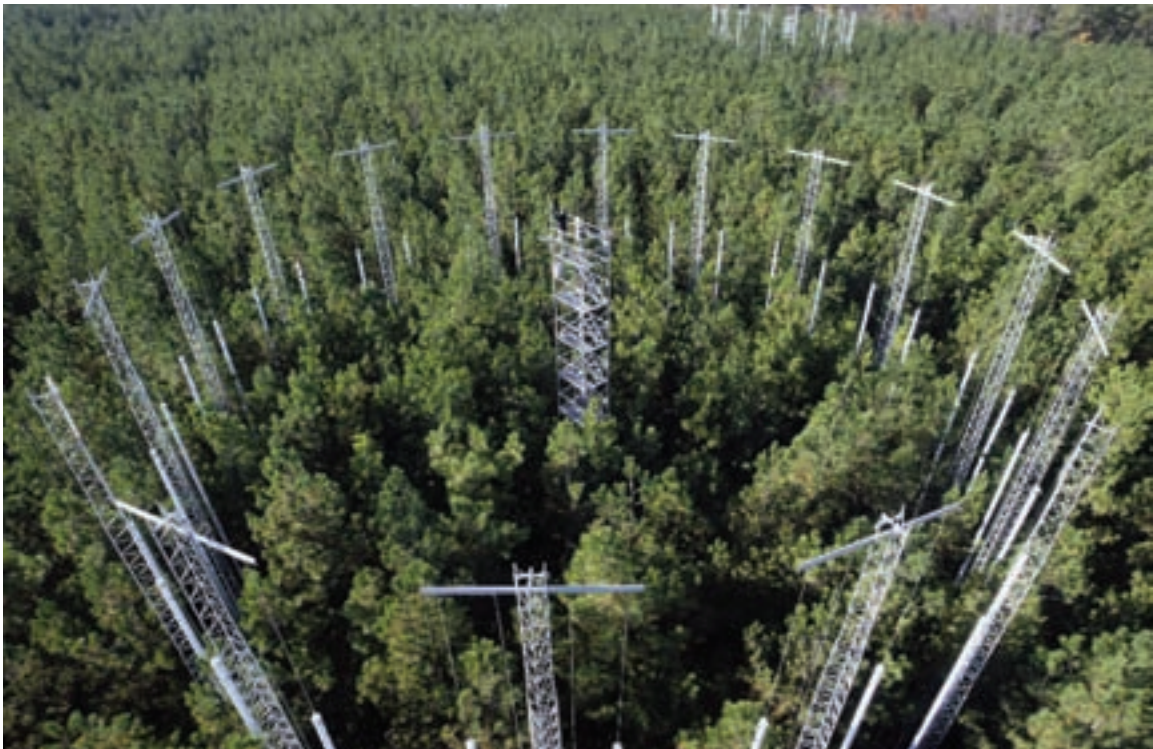


Figure 3.6 FACE ring at the Duke Forest FACE, Durham, North Carolina. (Photo courtesy Duke University.)

- Older forests can be strong carbon sinks (Stoy et al. 2006), and older trees absorb more CO₂ in an elevated CO₂ atmosphere, but wood production of these trees show limited or only transient response to CO₂ (Körner et al. 2005).
- When responding to CO₂, trees require and obtain more nitrogen (and other nutrients) from the soil. Yet, despite appreciable effort, the soil processes supporting such increased uptake have not been identified, leading to the expectation that nitrogen availability may increasingly limit the response to elevated CO₂ (Finzi et al. 2002; Luo et al. 2004; de Graaff et al. 2006; Finzi et al. 2006; Luo et al. 2006).

To understand the complex processes controlling ecosystem carbon cycling under elevated CO₂ and solve these puzzles, longer time series are needed (Walther 2007), yet the three FACE studies in the U.S. forest ecosystems are slated for closure in 2007-2009.

Major findings on specific processes leading to these generalities

Net primary production (NPP) is defined as the balance between canopy photosynthesis and plant respiration. Canopy photosynthesis increases with atmospheric CO₂, but less than expected based on physiological studies because of negative feedbacks in leaves (biochemical down-regulation) and canopies (reduced light, and conductance with increasing leaf area index (LAI); (Saxe et al. 2001; Schäfer et al. 2003; Wittig et al. 2005). On the other hand, plant respiration increases only in proportion to tree growth and amount of living biomass – that is, tissue-specific respiration does not change under elevated CO₂ (Gonzalez-Meler et al. 2004). The balance between these processes, NPP, increases in stands on moderately fertile and fertile soils. The short-term (<10 years), median response among the four “forest” FACE experiments was an increase of 23±2 percent (Norby et al. 2005). Although the average response was similar among these sites that differed in productivity (Norby et al. 2005), the within-site variability in the response to elevated CO₂ can be large (<10 percent to >100 percent). At the Duke FACE site, this within-site variability was related to nitrogen availability (Oren et al.

2001; Finzi et al. 2002; Norby et al. 2005). The absolute magnitude of the additional carbon sink varies greatly among years. At the Duke FACE, much of this variability is caused by droughts and disturbance events (McCarthy et al. 2006a).

The enhancement of NPP at low LAI is largely driven by an enhancement in LAI, whereas at high LAI, the enhancement reflects increased light-use efficiency (Norby et al. 2005; McCarthy et al. 2006a). The sustainability of the NPP response and the partitioning of carbon among plant components may depend on soil fertility (Curtis and Wang 1998; Oren et al. 2001; Finzi et al. 2002). NPP in intermediate fertility sites may undergo several phases of transient response, with CO₂-induced enhancement of stemwood production dominating initially, followed by fine-root production after several years (Oren et al. 2001; Norby et al. 2004). In high fertility plots, the initial response so far appears sustainable (Körner 2006).

Carbon partitioning to pools with different turnover times is highly sensitive to soil nutrient availability. Where nutrient availability is low, increasing soil nutrient supply promotes higher LAI. Under elevated CO₂ and increased nutrient supply, LAI becomes increasingly greater than that of stands under ambient CO₂. This response affects carbon allocation to other pools. Aboveground NPP increases with LAI (McCarthy et al. 2006a) with no additional effects of elevated CO₂. The fraction of Aboveground NPP allocated to wood, a moderately slow turnover pool, increases with LAI in broadleaf FACE experiments (from ~50 percent at low LAI, to a maximum of 70 percent at mid-range LAI), with the effect of elevated CO₂ on allocation entirely accounted for by changes in LAI. In pines, allocation to wood decreased with increasing LAI (from ~65 percent to 55 percent), but was higher (averaging ~68 percent versus 58 percent) under elevated CO₂ (McCarthy et al. 2006a). Despite the increased canopy photosynthesis, there is no evidence of increased wood production in pines growing on very poor, sandy soils (Oren et al. 2001).

Total carbon allocation belowground and CO₂ efflux from the forest floor decrease with increasing LAI, but the enhancement under



elevated CO₂ is approximately constant (~22 percent) over the entire range of LAI (King et al. 2004; Palmroth et al. 2006). About a third of the extra carbon allocated belowground under elevated CO₂ is retained in litter and soil storage at the U.S. FACE sites (Palmroth et al. 2006). At Duke FACE, a third of the incremental carbon sequestration is found in the forest floor. The CO₂-induced enhancement in fine root and mycorrhizal fungi turnover has not translated to a significant net incremental storage of carbon in the mineral soil (Schlesinger and Lichten 2001; Jastrow et al. 2005; Lichten et al. 2005). A recent meta-analysis (Jastrow et al. 2005), incorporating data from a variety of studies in different settings, estimated a median CO₂-induced increase in the rate of soil C sequestration of 5.6 percent (+19 g C m⁻² y⁻¹). Because soil C is highly variable and a large fraction of ecosystem carbon, a long time-series is necessary to statistically detect changes at any one site (McMurtrie et al. 2001).

3.2.3.4 FORESTS AND CARBON SEQUESTRATION

Forest growth and long-lived wood products currently offset about 20 percent of annual U.S. fossil fuel carbon emissions (U.S. Climate Change Science Program Synthesis and Assessment Product 2.2 2007). Because a large forested landscape should be carbon neutral over long periods of time (Kashian et al. 2006), the presence of this large forest carbon sink is either a legacy of past land use (regrowth after harvest or reforestation of land cleared for pasture or crops) or a response to increased CO₂ and nitrogen deposition, or both (Canadell et al. 2007). This carbon sink is an enormous ecosystem service by forests, and its persistence will be important to any future mitigation strategy. If the sink primarily results from past land use, it will diminish through time. If not, it may continue until the effects of CO₂ and N diminish (Canadell et al. 2007).

To understand whether forest growth and long-lived forest products will continue their important role in offsetting a fraction of U.S. carbon emissions, significant unknowns and uncertainties would have to be addressed. The scale of the problem is large: Jackson and Schlesinger (2004) estimate that for afforestation to offset an additional 10 percent of U.S. emis-

sions, immediate conversion of one-third of current croplands to forests would be required. Some of the unknowns and uncertainties are: 1) the economics of sequestration (Richards and Stokes 2004); 2) the timeline for valuing carbon stored in forests – should carbon stored today be worth more than carbon stored later (Fearnside 2002); 3) the permanence of stored carbon and its value if not permanent (Kirschbaum 2006); 4) the ability to permanently increase forest carbon stores in the face of changes in climate that may change species (Bachelet et al. 2001) and increase disturbance (Westerling et al. 2006), and change the process of carbon storage itself (Boisvenue and Running 2006); 5) how much carbon can be counted as “additional” given the self-replacing nature of forests; 6) identification of methods for increasing carbon sequestration in a variety of ecosystems and management goals; 7) how to account for carbon storage “gained” from management or avoided losses in fire; 8) identification of uniform methods and policies for validating carbon storage; 9) vulnerability of sequestered carbon to fire, windthrow or other disturbance; 10) “leakage” or displacement of carbon storage on one component of the landscape to carbon release on another (Murray et al. 2004); 11) will saturation of the carbon sink in North America work against forest C sequestration (Canadell et al. 2007)? 12) the impacts of carbon sequestration on the health of forest ecosystems and the climate system itself; and 13) the impacts of increasing carbon storage on other forest values such as biodiversity and water yield.

3.2.3.5 INTERACTIVE EFFECTS INCLUDING OZONE, NITROGEN DEPOSITION, AND FOREST AGE

Ozone is produced from photochemical reactions of nitrogen oxides and volatile organic compounds. Ozone can damage plants (Ashmore 2002) and lower productivity, and these responses have been documented for U.S. forests (Matyssek and Sandermann 2003; Karlsson et al. 2004). In the United States, controls on emissions of nitrogen oxides and volatile organic compounds are expected to reduce the peak ozone concentrations that currently cause the most plant damage (Ashmore 2005). However, background tropospheric concentrations may be increasing as a result of increased global emissions of nitrogen oxides (Ashmore 2005).



These predicted increases in background ozone concentrations may reduce or negate the effects of policies to reduce ozone concentrations (Ashmore 2005). Ozone pollution will modify the effects of elevated CO₂ and any changes in temperature and precipitation (Hanson et al. 2005), but these interactions are difficult to predict because they have been poorly studied.

Nitrogen deposition in the eastern United States and California can exceed 10 kg N ha⁻¹ yr⁻¹ and likely has increased 10-20 times above pre-industrial levels (Galloway et al. 2004). Forests are generally limited by nitrogen availability, and fertilization studies show that this increased deposition will enhance forest growth and carbon storage in wood (Gower et al. 1992; Albaugh et al. 1998; Adams et al. 2005). There is evidence that chronic nitrogen deposition also increases carbon storage in surface soil over large areas (Pregitzer et al. 2008). Chronic nitrogen inputs over many years could lead to “nitrogen saturation” (a point where the system can no longer use or store nitrogen), a reduction in forest growth, and increased levels of nitrate in streams (Aber et al. 1998; Magill et al. 2004), but observations of forest ecosystems under natural conditions have not detected this effect (Magnani et al. 2007). Experiments and field studies have shown that the positive effect of elevated CO₂ on productivity and carbon storage can be constrained by low nitrogen availability, but in many cases elevated CO₂ causes an increase in nitrogen uptake (Finzi et al. 2006; Johnson 2006; Luo et al. 2006; Reich et al. 2006). For nitrogen-limited ecosystems, increased nitrogen availability from nitrogen deposition enhances the productivity increase from elevated CO₂ (Oren et al. 2001) and the positive effects of changes in temperature and precipitation. Overall, there is strong evidence that the effects of nitrogen deposition on forest growth and carbon storage are positive and might exceed those of elevated CO₂ (Körner 2000; Magnani et al. 2007).

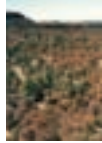
Forest growth changes with forest age (Ryan et al. 1997), likely because of reductions in photosynthesis (Ryan et al. 2004). Because of the link of forest growth with photosynthesis, the response to drought, precipitation, nitrogen availability, ozone, and elevated CO₂ may also change with forest age. Studies of elevated

CO₂ on trees have been done with young trees (which show a positive growth response), but the one study on mature trees showed no growth response (Körner et al. 2005). This is consistent with model results found in an independent study (Kirschbaum 2005). Tree size or age may also affect ozone response and response to drought, with older trees possibly more resistant to both (Grulke and Miller 1994; Irvine et al. 2004).

3.2.3.6 FIRE FREQUENCY AND SEVERITY

Several lines of evidence suggest that large, stand-replacing wildfires will likely increase in frequency over the next several decades because of climate warming (Figure 3.7). Chronologies derived from fire debris in alluvial fans (Pierce et al. 2004) and fire scars in tree rings (Kitzberger et al. 2007) provide a broader temporal context for interpreting contemporary changes in the fire regime. These longer-term records unequivocally show that warmer and drier periods during the last millennium are associated with more frequent and severe wildfires in western forests. GCM projections of future climate during 2010-2029 suggest that the number of low humidity days (and high fire danger days) will increase across much of the western U.S., allowing for more wildfire activity with the assumption that fuel densities and land management strategies remain the same (Flannigan et al. 2000; Brown et al. 2004). Flannigan et al. (2000) used two GCM simulations of future climate to calculate a seasonal severity rating related to fire intensity and difficulty of fire control. Depending on the GCM used, forest fire seasonal severity rating in the Southeast is projected to increase from 10 to 30 percent and 10 to 20 percent in the Northeast by 2060. Other biome models used with a variety of GCM climate projections simulate a larger increase in fire activity and biomass loss in the Southeast, sufficient to convert the southernmost closed-canopy Southeast forests to savannas (Bachelet et al. 2001). Forest management options to reduce fire size and intensity are discussed in Synthesis and Assessment Product 4.4 (Preliminary review of adaptation options for climate-sensitive ecosystems and resources, U.S. Climate Change Science Program).

By combining climate-fire relationships derived from contemporary records with GCM simula-



Future increases in fire emissions across North America will have important consequences for climate forcing agents, air quality, and ecosystem services.



Figure 3.7 Ponderosa pine after the Hayman fire in Colorado, June 2002. While no one fire can be related to climate or changes in climate, research shows that the size and number of Western forest fires has increased substantially since 1985, and that these increases were linked with earlier spring snowmelt and higher spring and summer air temperature. Photo courtesy USDA Forest Service.

tions of future climate, Flannigan et al. (2005) estimated that future fire activity in Canadian boreal forests will approximately double by the end of this century for model simulations in which fossil fuel emissions were allowed to increase linearly at a rate of 1 percent per year. Both Hadley Center and Canadian GCM simulations projected that fuel moisture levels will decrease and air temperatures will increase within the continental interior of North America because of forcing from greenhouse gases and aerosols.

Santa Ana winds and human-triggered ignitions play an important role in shaping the fire regime of Southern California shrublands and forests (Keeley and Fotheringham 2001; Westerling et al. 2004). Santa Ana winds occur primarily during fall and winter and are driven by large-scale patterns of atmospheric circulation (Raphael 2003; Conil and Hall 2006). Using future predictions from GCMs, Miller and Schlegel (2006) assessed that the total number of annual Santa Ana events would not change over the next 30 years. One of the GCM simulations showed a shift in the seasonal cycle, with fewer Santa Ana events occurring in September and more occurring in December. The implication of this change for the fire regime was unknown.

Future increases in fire emissions across North America will have important consequences for climate forcing agents, air quality, and ecosystem services. More frequent fire will increase emissions of greenhouse gases and aerosols (Amiro et al. 2001) and increase deposition of black carbon aerosols on snow and sea ice (Flanner et al. 2007). Even though many forests will regrow and sequester the carbon released in the fire, forests burned in the next few decades can be sources of CO₂ for decades and not recover the carbon lost for centuries (Kashian et al. 2006) – an important consideration

for slowing the increase in atmospheric CO₂. In boreal forests, the warming effects from fire-emitted greenhouse gases may be offset at regional scales by increases in surface albedo caused by a shift in the stand age distribution (Randerson et al. 2006). Any climate driven changes in boreal forest fires in Alaska and Canada will have consequences for air quality in the central and eastern United States because winds often transport carbon monoxide, ozone, and aerosols from boreal fires to the south (McKeen et al. 2002; Morris et al. 2006; Pfister et al. 2006). Increased burning in boreal forests and peatlands also has the potential to release large stocks of mercury currently stored in cold and wet soils (Turetsky et al. 2006). These emissions may exacerbate mercury toxicities in northern hemisphere food chains caused by coal burning.

3.2.3.7 INSECT OUTBREAKS

Rising temperature is the aspect of climate change most influential on forest insect species through changes in insect survival rates, increases in life cycle development rates, facilitation of range expansion, and effects on host plant capacity to resist attack (Ayres and Lombardero 2000; Malmström and Raffa 2000; Bale et al. 2002). Future northward range expansion

attributed to warming temperatures has been predicted for mountain pine beetle (Logan and Powell 2001) and southern pine beetle (Ungerer et al. 1999). Future range expansion of mountain pine beetle has the potential of invading jack pine, a suitable host that extends across the boreal forest of North America (Logan and Powell 2001). Increased probability of spruce beetle outbreak (Logan et al. 2003a) as well as increase in climate suitability for mountain pine beetle attack in high-elevation ecosystems (Hicke et al. 2006) has been projected in response to future warming. The combination of higher temperatures with reduced precipitation in the Southwest has led to enhanced tree stress, and also affected Ips beetle development rates; continued warming, as predicted by climate models, will likely maintain these factors (Breshears et al. 2005).

Indirect effects of future climate change may also influence outbreaks. Increasing atmospheric CO₂ concentrations may lead to increased ability of trees to recover from attack (Kruger et al. 1998). Enhanced tree productivity in response to favorable climate change, including rises in atmospheric CO₂, may lead to faster recovery of forests following outbreaks, and thus a reduction in time to susceptibility to subsequent attack (Fleming 2000). Although eastern spruce budworm life cycles are tightly coupled to host tree phenology even in the presence of climate change, enemy populations that are significant in governing epidemic dynamics are not expected to respond to climate change in a synchronized way (Fleming 2000). Changing fire regimes in response to climate change (Flannigan et al. 2005) will affect landscape-scale forest structure, which influences susceptibility to attack (Shore et al. 2006).

Nonnative invasive species are also significant disturbances to forests in the United States. Although little has been reported on climate influences on these insects, a few studies have illustrated climate control. The hemlock woolly adelgid is rapidly expanding its range in the eastern United States, feeding on several species of hemlock (Box 1). The northern range limit of the insect in the United States is currently limited by low temperatures (Parker et al. 1999), suggesting range expansion in the event of future warming. In addition, the hemlock

woolly adelgid has evolved greater resistance to cold conditions as it has expanded north (Butin et al. 2005). The introduced gypsy moth has defoliated millions of hectares of forest across the eastern United States, with great efforts expended to limit its introduction to other areas (USDA Forest Service 2005). Projections of future climate and gypsy moth simulation modeling reveal substantial increases in probability of establishment in the coming decades (Logan et al. 2003a).

As important disturbances, insect outbreaks affect many forest ecosystem processes. Outbreaks alter tree species composition within stands, and may result in conversion from forest to herbaceous vegetation through lack of regeneration (Holsten et al. 1995). Carbon stocks and fluxes are modified through a large decrease in living biomass and a corresponding large increase in dead biomass, reducing carbon uptake by forests as well as enhancing decomposition fluxes. In addition to effects at smaller scales, widespread outbreaks have significant effects on regional carbon cycling (Kurz and Apps 1999; Hicke et al. 2002a). Other biogeochemical cycles, such as nitrogen, are affected by beetle-caused mortality (Throop et al. 2004). Defoliation, for example as related to gypsy moth outbreaks, facilitates nitrogen movement from forest to aquatic ecosystems, elevating stream nitrogen levels (Townsend et al. 2004).

Significant changes to the hydrologic cycle occur after a widespread insect epidemic, including increases in annual water yield, advances in the annual hydrograph, and increases in low flows (Bethlahmy 1974; Potts 1984). Water quantity is enhanced through reductions in transpiration, in addition to reductions in snow interception, and subsequent redistribution and sublimation. These effects can last for many years following mortality (Bethlahmy 1974).

Interactions of outbreaks and fire likely vary with time, although observational evidence is limited to a few studies (Romme et al. 2006). In central Colorado, number of fires, fire extent, and fire severity were not enhanced following outbreaks of spruce beetle (Bebi et al. 2003; Bigler et al. 2005; Kulakowski and Veblen in press). Other studies of the 1988 Yellowstone



BOX I: The Eastern Hemlock and its Woolly Adelgid

Outbreaks of insects and diseases affect forest structure and composition, leading to changes in carbon, nutrients, biodiversity, and ecosystem services. The hemlock woolly adelgid (HWA), native to Asia, was first recorded in 1951 in Virginia, and has since spread, causing a severe decline in vitality and survival of eastern hemlock in North American forests (Maps 3.1 & 3.2, Stadler et al. 2006). Roads, major trails, and riparian corridors provide for long-distance dispersal of this aphid-like insect, probably by humans or birds (Koch et al. 2006). Although HWA is consumed by some insect predators (Flowers et al. 2006), once it arrives at a site, complete hemlock mortality is inevitable (Orwig et al. 2002; Stadler et al. 2005).

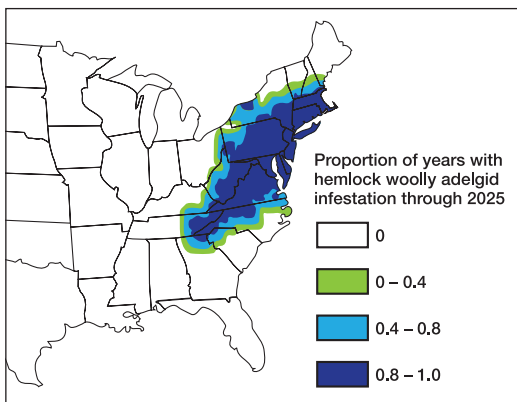
HWA will change biodiversity and species composition. Hemlock seedlings are readily attacked and killed by the HWA, so damaged hemlock stands are replaced by stands of black birch, black oaks, and other hardwoods (Brooks 2004; Small et al. 2005; Sullivan and Ellison 2006). After HWA attack, plant biodiversity increases in the canopy and in the understory; invasive shrubs and woody vines of several species also expand in response to the improved light conditions (Goslee et al. 2005; Small et al. 2005; Eschtruth et al. 2006). Four insectivorous bird species have high affinity for hemlock forest type, and two of these, the blue-headed vireo and Blackburnian warbler, are specialists in the hemlock habitat. Expansion of HWA could negatively impact several million pairs of these birds by eliminating their habitat (Tingley et al. 2002; Ross et al. 2004).

Changes in canopy attributes upon replacement of hemlock with deciduous broadleaf species alter the radiation regime, hydrology, and nutrient cycling (Cobb et al. 2006; Stadler et al. 2006), and result in greater temperature fluctuations and longer periods of times in which streams are dry (Snyder et al. 2002). These conditions reduce habitat quality for certain species of fish. Brook trout and brown trout were two to three times as prevalent in hemlock than hardwood streams (Ross et al. 2003). Low winter temperature is the main factor checking the spread of HWA (Skinner et al. 2003). However, the combination of increasing temperature and the capacity of HWA to evolve greater resistance to cold shock as it has expanded its range northward (Butin et al. 2005) means that stands that have been relatively protected by cold temperatures (Orwig et al. 2002) may fall prey to the insect in the not-so-distant future (Map 3.3).

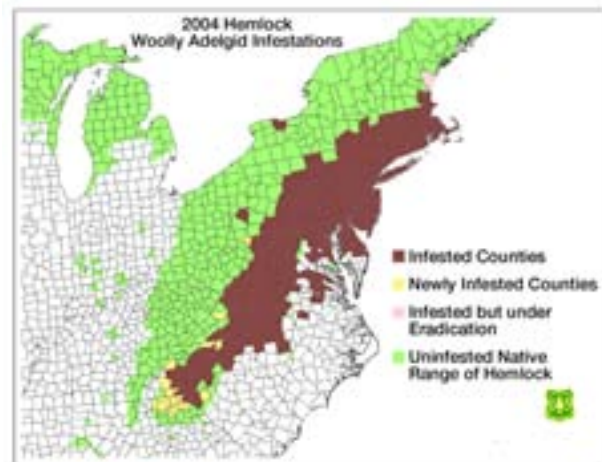


Map 3.1 Sample sites and range expansion of *Adelges tsugae* relative to the native range of eastern hemlock in North America. Map from Butin et al. 2005 (redrawn from USDA Forest Service and Little, 1971).

Map 3.2 Counties in the range of eastern hemlock that are uninfested, newly infested, and infested. From Onken and Reardon.(2005).



Map 3.3 Hemlock woolly adelgid spread map prepared by Randall Marin, Northeastern Research Station, U.S. Forest Service (Souto et al. 1996).



fire that followed two mountain pine beetle epidemics found mixed fire effects, depending on outbreak severity and time since outbreak (Turner et al. 1999; Lynch et al. 2006). A quantitative modeling study of fire behavior found mixed fire effects following an outbreak of western spruce budworm (Hummel and Agee 2003); more modeling studies that incorporate complete effects are needed to explore other situations.

Multiple socioeconomic impacts follow severe insect outbreaks. Timber production and manufacturing and markets may not be able to take advantage of vast numbers of killed trees (Ferguson 2004), and beetle-killed timber has several disadvantages from a manufacturing perspective (Byrne et al. 2006). Perceived enhanced fire risk and views about montane aesthetics following beetle-cause mortality influence public views of insect outbreaks, which could drive future public policy. Threats to ecologically important tree species may have ramifications for charismatic animal species (e.g., influences of whitebark pine mortality on the grizzly bear) (Logan and Powell 2001). Impacts are enhanced as human population, recreation, and tourism increase in forested regions across the nation.

3.2.3.8 STORMS (HURRICANES, ICE STORMS, WINDSTORMS)

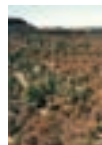
Predictions of forest carbon (C) sequestration account for the effect of fires (e.g., Harden et al. 2000), yet other wide-ranging and frequent disturbances, such as hurricanes (Figure 3.8) and ice storms, are seldom explicitly considered. Both storm types are common in the southeastern United States, with an average return time of six years for ice storms (Bennett 1959) and two years for hurricanes (Smith 1999). These, therefore, have the potential for significant impact on C sequestration in this region, which accounts for ~20 percent of annual C sequestration in conterminous U.S. forests (Birdsey and Lewis 2002; Bragg et al. 2003). Recent analysis demonstrated that a single Category 3 hurricane or severe ice storm could each transfer to the decomposable pool the equivalent of 10 percent of the annual U.S. C sequestration, with subsequent reductions in sequestration caused by direct stand damage (McNulty 2002; McCarthy et al. 2006b). For example, Hurricanes



Figure 3.8 Forest damage from Hurricane Katrina. Dr. Jeffrey Q. Chambers, Tulane University.

Rita and Katrina together damaged a total of 2,200 ha and 63 million m³ of timber volume (Stanturf et al. 2007) which, when decomposed over the next several years, will release a total of 105 teragrams (Tg) of C into the atmosphere, roughly equal to the annual net sink for U.S. forests (Chambers et al. 2007).

Common forest management practices such as fertilization and thinning, forest type, and increasing atmospheric CO₂ levels can change wood and stand properties, and thus may change vulnerability to ice storm damage. A pine plantation experienced a ~250 g C m⁻² reduction in living biomass during a single ice storm, equivalent to ~30 percent of the annual net ecosystem carbon exchange of this ecosystem. In this storm at the Duke FACE, nitrogen fertilization had no effect on storm damage; conifer trees were more than twice as likely to be killed by ice storm damages as leafless deciduous-broadleaf trees; and thinning increased broken limbs or trees threefold. Damage in the elevated CO₂ stand was one third as much as in the ambient CO₂ stand. (McCarthy et al. 2006b). Although this result suggests that forests may suffer less damage in a future ice storm when atmospheric CO₂ is higher, future climate may create conditions leading to greater ice storm frequency, extent and severity (da Silva et al. 2006), which may balance the decreased sensitivity to ice damage under elevated CO₂. All of these predictions are very uncertain (Cohen et al. 2001).



3.2.3.9 CHANGES IN OVERSTORY SPECIES COMPOSITION

Several approaches can predict changes in biomes (major vegetation assemblages such as conifer forests and savanna/woodland) and changes in species composition or overstory species communities (Hansen et al. 2001a). These approaches use either rules that define the water balance, temperature, seasonality, etc. required for a particular biome, or statistically link species distributions or community composition with climate envelopes. These efforts have mostly focused on equilibrium responses to climate changes over the next century (Hansen et al. 2001a), so predictions for the next several decades are unavailable.

Bachelet et al. (2001) used the Mapped Atmosphere-Plant-Soil System (MAPPS) model with the climate predictions generated by seven different global circulation models to predict how biome distributions would change with a doubling of CO₂ by 2100. Mean annual temperature of the United States increased from 3.3 to 5.8 °C for the climate predictions. Predicted forest cover in 2100 declined by an average of 11 percent (range for all climate models was +23 percent to -45 percent). The MAPPS model coupled to the projected future climates predicts that biomes will migrate northward in the East and to higher elevations in the West. For example, mixed conifer and mixed hardwood forests in the Northeast move into Canada, and decline in area by 72 percent (range: -14 to -97 percent), but are replaced by eastern hardwoods. In the Southeast, grasslands or savannas displace forests and move their southern boundaries northward, particularly for the warmer climate scenarios. In the West, forests displace alpine ecosystems, and the wet conifer forests of the Northwest decline in area 9 percent (range: 54 to + 21 percent), while the area of interior western pines changes little. Species predictions for the eastern United States using a statistical approach showed that most species moved north 60-300 miles (Hansen et al. 2001a).

Authors of these studies cautioned that these equilibrium approaches do not reflect the transient and species-specific nature of the community shifts that are projected to occur. Success in moving requires suitable climate, but also dispersal that may lag behind changes

in climate (Hansen et al. 2001a). Some species will be able to move quicker than others, and some biomes and communities may persist until a disturbance allows changes to occur (Hansen et al. 2001a). Because trees are long-lived and may tolerate growing conditions outside of their current climate envelopes, they may be slower to change than modeled (Loehle and LeBlanc 1996). The authors of these studies agreed that while climate is changing, novel ecosystems will arise – novel because some species will persist in place, some species will depart, and new species will arrive.

3.2.4 Indicators and Observing Systems

3.2.4.1 CHARACTERISTICS OF OBSERVING SYSTEMS

Many Earth observing systems (Bechtold and Patterson 2005; Denning 2005) are designed to allow for integration of multiple kinds of observations using a hierarchical approach that takes advantage of the characteristics of each. A typical system uses remote sensing to obtain a continuous measurement over a large area, coupled with statistically-designed field surveys to obtain more detailed data at a finer resolution. Statistically, this approach (known as “multi-phase” sampling) is more efficient than sampling with just a single kind of observation or conducting a complete census (Gregoire and Valentine, in press). Combining observed data with models is also common because often the variable of interest cannot be directly observed, but observation of a closely-related variable may be linked to the variable of interest with a model. Model-data synthesis is often an essential component of Earth observing systems (Raupach et al. 2005).

To be useful, the system must observe a number of indicators more than once over a period, and also cover a large enough spatial scale to detect a change. The length of time required to detect a change with a specified level of precision depends on the variability of the population being sampled, the precision of measurement, and the number of samples (Smith 2004). Non-climatic local factors, such as land use change, tend to dominate vegetation responses at small scales, masking the relationship with climate (Parmesan and Yohe 2003). A climate signal is



therefore more likely to be revealed by analyses that can identify trends across large geographic regions (Walther et al. 2002).

The relationship between biological observations and climate is correlational; thus, it is difficult to separate the effects of climate change from other possible causes of observed effects (Walther et al. 2002). Inference of causation can be determined with carefully controlled experiments that complement the observations. Yet, observation systems can identify some causal relationships and therefore have value in developing climate impact hypotheses. Schreuder and Thomas (1991) determined that if both the potential cause and effect variables were measured at inventory sample plots, a relationship could be established if the variables are measured accurately, estimated properly, and based on a large enough sample. But, in practice, additional information is often needed to strengthen a case – for example, a complementary controlled experiment to verify the relationship.

3.2.4.2 INDICATORS OF CLIMATE CHANGE EFFECTS

The species that comprise communities respond both physiologically and competitively to climate change. One scheme for assessing the impacts of climate change on species and communities is to assess the effects on: (1) the physiology of photosynthesis, respiration, and growth; (2) species distributions; and (3) phenology, particularly life cycle events such as timing of leaf opening. There may also be effects on functions of ecosystems such as hydrologic processes.

Effects on physiology

Net primary productivity is closely related to indices of “greenness” and can be detected by satellite over large regions (Hicke et al. 2002b). Net ecosystem productivity (NEP) can be measured on the ground as changes in carbon stocks in vegetation and soil (Boisvenue and Running 2006). Root respiration and turnover are sensitive to climate variability and may be good indicators of climate change if measured over long enough time periods (Atkin et al. 2000; Gill and Jackson 2000). Gradient studies show variable responses of growth to precipitation changes along elevational gradients (Fagre et al. 2003). Climate effects on growth patterns of

individual trees is confounded by other factors such as increasing CO₂ and N deposition, so response of tree growth is difficult to interpret without good knowledge of the exposure to many possible causal variables. For example, interannual variability in NPP, which can mask long-term trends, can be summarized from long-term ecosystem studies and seems to be related to interactions between precipitation gradients and growth potential of vegetation (Knapp and Smith 2001).

Effects on species distributions

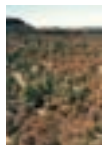
Climate change affects forest composition and geographical distribution, and these changes are observable over time by field inventories, remote sensing, and gradient studies. Both range expansions and retractions are important to monitor (Thomas et al. 2006), and population extinctions or extirpations are also possible. Changes in the range and cover of shrubs and trees have been observed in Alaska by field studies and remote sensing (Hinzman et al. 2005). Detecting range and abundance shifts in wildlife populations may be complicated by changes in habitat from other factors (Warren et al. 2001).

Effects on phenology

Satellite and ground systems can document onset and loss of foliage, with the key being availability of long-term data sets (Penuelas and Filella 2001). Growing season was found significantly longer in Alaska based on satellite normalized difference vegetation index (NDVI) records (Hinzman et al. 2005). Schwartz et al. (2006) integrated weather station observations of climate variables with remote sensing and field observations of phenological changes using Spring Index phenology models. However, Fisher et al. (2007) concluded that species or community compositions must be known to use satellite observations for predicting the phenological response to climate change.

Effects on natural disturbances and mortality

Climate change can affect forests by altering the frequency, intensity, duration, and timing of natural disturbances (Dale et al. 2001). The correlation of observations of changes in fire frequency and severity with changes in climate are well documented (e.g., Flannigan et al.



2000; Westerling et al. 2006), and the inference of causation in these studies is established by in situ studies of fire and vegetation response, and fire behavior models. Similar relationships hold for forest disturbance from herbivores and pathogens (Ayres and Lombardero 2000; Logan et al. 2003b). Tree mortality may be directly caused by climate variability, such as in drought (Gitlin et al. 2006).

Effects on hydrology

Climate change will affect forest water budgets. These changes have been observed over time by long-term stream gauge networks and research sites. Changes in permafrost and streamflow in the Alaskan Arctic region are already apparent (Hinzman et al. 2005). There is some evidence of a global pattern (including in the United States) in response of streamflow to climate from stream-gauge observations (Milly et al. 2005). Inter-annual variation in transpiration of a forest can be observed by sap flow measurements (Phillips and Oren 2001; Wullschlegel et al. 2001).

Causal variables

It is important to have high-quality, spatially-referenced observations of climate, air pollution, deposition, and disturbance variables. These are often derived from observation networks using spatial statistical methods (e.g., Thornton et al. 2000).

3.2.4.3 CURRENT CAPABILITIES AND NEEDS

There are strengths and limitations to each kind of observation system: intensive monitoring sites such as Long Term Ecological Research (LTER) sites and protected areas; extensive observation systems such as Forest Inventory and Analysis (FIA) or the U.S. Geological Survey (USGS) stream gauge network; and remote sensing. A national climate observation system may be improved by integration under an umbrella program such as the National Ecological Observatory Network (NEON), or Global Earth System of Systems (GEOSS) (see Table 3.1). Also, increased focus on “sentinel” sites could help identify early indicators of climate effects on ecosystem processes, and provide observations of structural and species changes (NEON 2006).

Intensive monitoring sites measure many of the indicators that are likely to be affected by climate change, but have mostly been located independently and so do not optimally represent either (1) the full range of forest condition variability, or (2) forest landscapes that are most likely to be affected by climate change (Hargrove et al. 2003). Forest inventories are able to detect long-term changes in composition and growth, but they are limited in ability to attribute observed changes to climate, because they were not designed to do so. Additions to the list of measured variables and compiling potential causal variables would improve the inventory approach (The Heinz Center 2002; USDA 2003). Remote sensing, when coupled with models, can detect changes in vegetation-response to climate variability (Running et al. 2004; Turner et al. 2004). Interpretation of remote sensing observations is greatly improved by associating results with ground data (Pan et al. 2006).

Maintaining continuity of remote sensing observations at appropriate temporal and spatial scales must be a high priority. NASA’s Earth Science division cannot support continued operations of all satellites indefinitely, so it becomes a challenge for the community using the measurements to identify long-term requirements for satellites, and provide a long-term framework for critical Earth science measurements and products (NASA Office of Earth Science 2004).

Another high-priority need is to improve ability to monitor the effects of disturbance on forest composition and structure, and to attribute changes in disturbance regimes to changes in climate. This will involve a more coordinated effort to compile maps of disturbance events from satellite or other observation systems, to follow disturbances with in situ observations of impacts, and to keep track of vegetation changes in disturbed areas over time. There are several existing programs that could be augmented to achieve this result, such as intensifying the permanent sample plot network of the FIA program for specific disturbance events, or working with forest regeneration and restoration programs to install long-term monitoring plots.



3.2.5 How Changes in One Resource can Affect Other Resources

Disturbances in forests such as fire, insect outbreaks, and hurricanes usually kill some or all of the trees and lower leaf area. These reductions in forest cover and leaf area will likely change the hydrology of the disturbed areas. Studies of forest harvesting show that removal of the tree canopy or transpiring surface will increase water yield, with the increase proportional to the amount of tree cover removed (Stednick 1996). The response will vary with climate and species, with wetter climates showing a greater response of water yield to tree removal. For all studies, average water yield increased 2.5 mm for each 1 percent of the tree basal-area removed (Stednick 1996). High-severity forest fires can increase sediment production and water yield as much as 10 to 1000 times, with the largest effects occurring during high-intensity summer storms (see review in Benavides-Solorio and MacDonald 2001). An insect epidemic can increase annual water yield, advance the timing of peak runoff, and increase base flows

(Bethlahmy 1974; Potts 1984). Presumably, the same effects would occur for trees killed in windstorms and hurricanes.

Disturbances can also affect native plant species diversity, by allowing opportunities for establishment of non-native invasives, particularly if the disturbance is outside of the range of variability for the ecosystem (Hobbs and Huenneke 1992). Areas most vulnerable to invasion by non-natives are those areas that support the highest plant diversity and growth (Stohlgren et al. 1999). In the western United States, these are generally the riparian areas (Stohlgren et al. 1998). We expect that disturbances that remove forest litter or expose soil (fire, trees tipped over by wind) will have the highest risk for admitting invasive non-native plants.

Table 3.1 Current and Planned Observation Systems for Climate Effects on Forests

Observation System	Characteristics	Reference
Forest Inventory and Analysis (U.S. Forest Service)	Annual and periodic measurements of forest attributes at a large number (more than 150,000) of sampling locations. Historical data back to 1930s in some areas.	Bechtold and Patterson 2005
AmeriFlux (Department of Energy and other Agencies)	High temporal resolution (minutes) measurements of carbon, water, and energy exchange between land and atmosphere at about 50 forest sites. A decade or more of data available at some of the sites.	http://public.ornl.gov/ameriflux/
Long Term Ecological Research network (National Science Foundation)	The LTER network is a collaborative effort involving more than 1,800 scientists and students investigating ecological processes over long temporal and broad spatial scales. The 26 LTER Sites represent diverse ecosystems and research emphases	http://www.lternet.edu/
Experimental Forest Network (U.S. Forest Service)	A network of 77 protected forest areas where long-term monitoring and experiments have been conducted.	Lugo 2006
National Ecological Observation Network	The NEON observatory is specifically designed to address central scientific questions about the interactions of ecosystems, climate, and land use.	http://www.neoninc.org/
Global Terrestrial Observing System (GTOS)	GTOS is a program for observations, modeling, and analysis of terrestrial ecosystems to support sustainable development.	http://www.fao.org/gtos/



3.3 ARID LANDS

3.3.1 Brief Summary of Key Points from the Literature

Plants and animals in arid lands live near their physiological limits, so slight changes in temperature and precipitation will substantially alter the composition, distribution, and abundance of species, and the products and services that arid lands provide. Observed and projected decreases in the frequency of freezing temperatures, lengthening of the frost-free season, and increased minimum temperatures will alter plant species ranges and shift the geographic and elevational boundaries of the Great Basin, Mojave, Sonoran, and Chihuahuan deserts. The extent of these changes will also depend on changes in precipitation and fire. Increased drought frequency will likely cause major changes in vegetation cover. Losses of vegetative cover coupled with increases in precipitation intensity and climate-induced reductions in soil aggregate stability will dramatically increase potential erosion rates. Transport of eroded sediment to streams coupled with changes in the timing and magnitude of minimum and maximum flows will affect water quality, riparian vegetation, and aquatic fauna. Wind erosion will have continental-scale impacts on downwind ecosystems, air quality, and human populations.

The response of arid lands to climate change will be strongly influenced by interactions with non-climatic factors at local scales. Climate effects should be viewed in the context of these other factors, and simple generalizations should be viewed with caution. Climate will strongly influence the impact of land use on ecosystems and how ecosystems respond. Grazing has traditionally been the most extensive land use in arid regions. However, land use has significantly shifted to exurban development and recreation in recent decades. Arid land response to climate will thus be influenced by environmental pressures related to air pollution and N-deposition, energy development, motorized off-road vehicles, feral pets, and horticultural invasives, in addition to grazing.

Non-native plant invasions will likely have a major impact on how arid land ecosystems respond to climate and climate change. Exotic


grasses generate large fuel loads that predispose arid lands to more frequent and intense fire than historically occurred with sparser native fuels. Such fires can radically transform diverse desert scrub, shrub-steppe, and desert grassland/savanna ecosystems into monocultures of non-native grasses. This process is well underway in the cold desert region, and is in its early stages in hot deserts. Because of their profound impact on the fire regime and hydrology, invasive plants in arid lands may trump direct climate impacts on native vegetation.

Given the concomitant changes in climate, atmospheric CO₂, nitrogen deposition, and species invasions, novel wildland and managed ecosystems will likely develop. In novel ecosystems, species occur in combinations, and relative abundances that have not occurred previously in a given biome. In turn, novel ecosystems present novel challenges for conservation and management.

3.3.2 Observed and Predicted Changes or Trends

3.3.2.1 INTRODUCTION

Arid lands occur in tropical, subtropical, temperate, and polar regions and are defined based on physiographic, climatic, and floristic features. Arid lands are characterized by low (typically <400 mm), highly variable annual precipitation, along with temperature regimes where potential evaporation far exceeds precipitation inputs. In addition, growing season rainfall is often delivered via intense convective storms, such that significant quantities of water run off before infiltrating into soil; precipitation falling as snow in winter may sublimate or run off during snowmelt in spring while soils are frozen. As a result of these combined factors, production per unit of precipitation can be low. Given that many organisms in arid lands are near their physiological limits for temperature and water stress tolerance, slight changes in temperature and precipitation that affect water availability and water requirements could have substantial ramifications. Thus, predicted transitions toward more arid conditions (e.g., higher temperatures that elevate potential evapotranspiration and more intense thunderstorms that generate more run off; Seager et al. 2007) have the real potential to alter species composition and abundance, and the ecosystem goods and



Plants and animals in arid lands live near their physiological limits, so slight changes in temperature and precipitation will substantially alter the composition, distribution, and abundance of species, and the products and services that arid lands provide.

services that arid lands can provide for humans (Field et al. 2007).

The response of arid lands to climate and climate change is contingent upon the net effect of non-climatic factors interacting with climate at local scales (Figure 3.9). Some of these factors may reinforce and accentuate climate effects (e.g., livestock grazing); others may constrain, offset or override climate effects (e.g., soils, atmospheric CO₂ enrichment, fire, non-native species). Climate effects should thus be viewed in the context of other factors, and simple generalizations regarding climate effects should be viewed with caution. A literature review of the relationship between climate change and land use indicate land use change has had a much greater effect on ecosystems than has climate change; and that the vast majority of land use changes have little to do with climate or climate change (Dale 1997). Today’s arid lands reflect a legacy of historic land uses, and future land use practices will arguably have the greatest impact on arid land ecosystems in the next two to five decades. In the near-term, climate fluctuation and change will be important primarily as it influences the impact of land use on ecosystems, and how ecosystems respond to land use.

3.3.2.2 BIO-CLIMATIC SETTING

Arid lands of the continental United States are represented primarily by the subtropical hot deserts of the Southwest, and the temperate cold deserts of the Intermountain West (Figure 3.2). The hot deserts differ primarily with respect to precipitation seasonality (Figure 3.10). The Mojave desert is dominated by winter precipitation (thus biological activity in the cool season), whereas the Chihuahuan desert is dominated by summer precipitation (thus biological activity during hotter conditions). The hottest of the three deserts, the Sonoran, is the intermediate, receiving both winter and summer precipitation. The cold deserts are also dominated by winter precipitation, much of which falls as snow, owing to the more temperate latitudes and higher elevations (West 1983). These arid land formations are characterized by unique plants and animals, and if precipitation seasonality changes, marked changes in species and functional group composition and abundance would be expected.

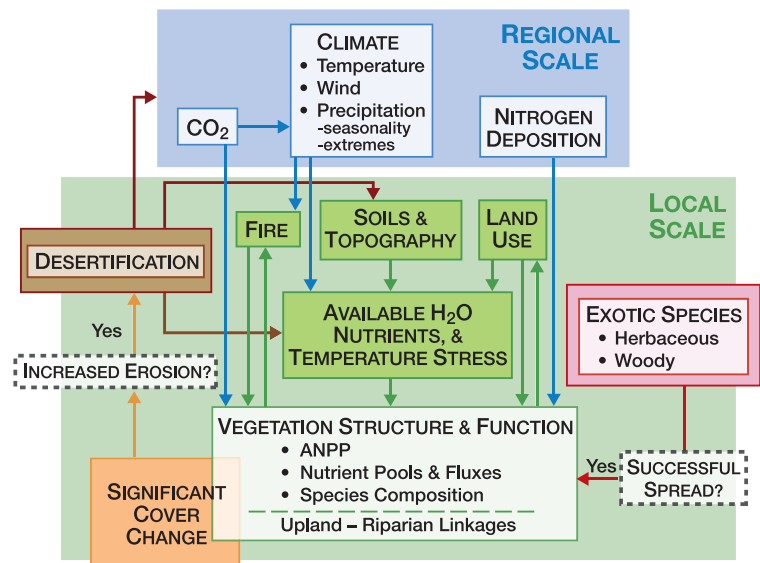


Figure 3.9 Organizational framework for interpreting climate and climate change effects on arid land ecosystems.

There is broad consensus among climate models that the arid regions of the southwestern United States will dry in the 21st century and that a transition to a more arid climate is already underway. In multimodel ensemble means reported by Seager et al. (2007), there is a transition to a sustained drier climate that begins in the late 20th and early 21st centuries. Both precipitation and evaporation are expected to decrease, but precipitation is expected to decrease more than evaporation leading to an overall drier climate. The increasing aridity is primarily reduced in winter, when precipitation decreases and evaporation remains unchanged or slightly reduced. The projected ensemble

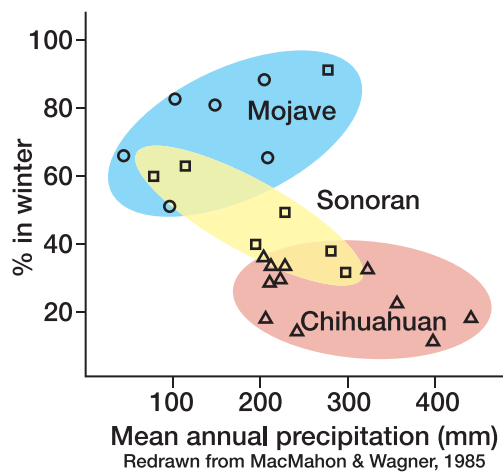
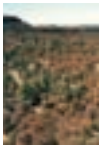


Figure 3.10 Mean annual precipitation and its seasonality in three hot deserts (from MacMahon and Wagner 1985).



median reduction in precipitation reaches 0.1 mm/day in mid-century, though several models show that the decrease could occur in the early 21st century. A substantial portion of the mean circulation contribution, especially in winter, is explained by the change in zonal mean flow alone, indicating that changes in the Hadley Cell and extratropical mean meridional circulation are important to the climate of this region.

The Great Basin is a cold desert characterized by limited water resources and periodic droughts in which a high proportion of the year's precipitation falls as winter snow (Wagner 2003). Snow-derived runoff provides the important water resources to maintain stream and river channels that support riparian areas and human utilization of this region. In the last century, the Great Basin warmed by 0.3° to 0.6°C and is projected to warm by an additional 5° to 10°C in the coming century (Wagner 2003). In the last half-century, total precipitation has increased 6-16 percent and this increase is projected to continue in the future (Baldwin et al. 2003). The increase in total precipitation is offset partially by the decrease in snowpack, which in the Great Basin is among the largest in the nation (Mote et al. 2005). The onset of snow runoff is currently 10–15 days earlier than 50 years ago, with significant impacts on the downstream utilization of this water (Cayan et al. 2001; Baldwin et al. 2003; Stewart et al. 2004). Increased warming is likely to continue to accelerate spring snowmelt. Warmer temperatures are also likely to lead to more precipitation falling as rain which would further reduce overall snowpack and spring peak flow.

Throughout the dry western United States, extreme temperature and precipitation events are expected to change in the next century. Warm extremes will generally follow increases in the mean summertime extremes, while cold extremes will warm faster than warm extremes (Kharin et al. 2007). As a result, what is currently considered an unusually high temperature (e.g., 20-year return interval) will become very frequent in the desert Southwest, occurring every couple of years. On the other hand, unusually low temperatures will become increasingly uncommon. As a result winters will be warmer, leading to higher evapotranspiration and lower snowfall. Changes in precipitation

are also expected. Precipitation events that are currently considered extreme (20-year return interval) are also expected to occur roughly twice as often as they currently do, consistent with general increases in rainstorm intensity (Kharin et al. 2007).

Changes in species and functional group composition might first occur in the geographic regions where biogeographic formations and their major subdivisions interface. Extreme climatic events are major determinants of arid ecosystem structure and function (Holmgren et al. 2006). Thus, while changes in mean temperature will affect levels of physiological stress and water requirements during the growing season, minimum temperatures during winter may be a primary determinant of species composition and distribution. In the Sonoran Desert, warm season rainfall and freezing temperatures strongly influence distributions of many plant species (Turner et al. 1995). The vegetation growing season, as defined by continuous frost-free air temperatures, has increased by on average about two days/decade since 1948 in the conterminous United States, with the largest changes occurring in the West (Easterling 2002; Feng and Hu 2004). A recent analysis of climate trends in the Sonoran Desert (1960-2000) also shows a decrease in the frequency of freezing temperatures, lengthening of the frost-free season, and increased minimum temperatures (Weiss and Overpeck 2005). With warming expected to continue throughout the 21st century, potential ecological responses may include contraction of the overall boundary of the Sonoran Desert in the southeast and expansion northward, eastward, and upward in elevation, and changes to plant species ranges. Realization of these changes will be co-dependent on what happens with precipitation and disturbance regimes (e.g., fire).

The biotic communities that characterize many U.S. arid lands are influenced by basin and range topography. Thus, within a given bioclimatic zone, communities transition from desert scrub and grassland to savanna, woodland, and forest along strong elevation gradients (Figure 3.11). Changes in climate will affect the nature of this zonation, with arid land communities potentially moving up in elevation in response to warmer and drier conditions. Experimental

The vegetation growing season, as defined by continuous frost-free air temperatures, has increased by on average about two days/decade since 1948 in the conterminous United States, with the largest changes occurring in the West.



data suggest shrub recruitment at woodland-grassland ecotones will be favored by increases in summer precipitation, but unaffected by increases in winter precipitation (Weltzin and McPherson 2000). This suggests that increases in summer precipitation would favor the downslope shifts in this ecotone. In the Great Basin, favorable climatic conditions at the turn of the last century enabled expansion of woodlands into sagebrush steppe (Miller and Rose 1999; Miller et al. 2005) and ongoing expansion is significantly increasing fuel loads and creating conditions for catastrophic fire. Plant composition and ecosystem processes (e.g., plant growth, water and nutrient use, herbivory) change along these elevation gradients in a manner that broadly mimics changes in ecosystem structure and function along continental-scale latitudinal gradients (Whittaker 1975). Changes along local elevation gradients may therefore be early indicators of regional responses to climate change (Peters 1992).

3.3.2.3 CLIMATE INFLUENCES AT LOCAL SCALES

Climate and atmospheric CO₂ influence communities at broad spatial scales, but topography, soils, and landform control local variation in ecosystem structure and function within a given elevation zone, making local vegetation very complex. Topography influences water balance (south-facing slopes are drier), air drainage and night temperatures, and routing of precipitation. Soil texture and depth affect water capture, water storage, and fertility (especially nitrogen). These factors strongly interact with precipitation to limit plant production and control species composition. Plants that can access water in deep soil or in groundwater depend less on precipitation for growth and survival, but such plants may be sensitive to precipitation changes that affect the recharge of deep water stores. If the water table increases with increases in rainfall or decreased plant cover, soil salinity may increase and adversely affect vegetation in some bottomland locations (McAuliffe

2003). To predict vegetation response to climate change, it is necessary to understand these complex relationships among topography, soil, soil hydrology, and plant response.

3.3.2.4 CLIMATE AND DISTURBANCE

Disturbances such as fire and grazing are superimposed against the backdrop of climate variability, climate change, and spatial variation in soils and topography. The frequency and intensity of a given type of disturbance will determine the relative abundance of annual, perennial, herbaceous, and woody plants on a site. Extreme climate events such as drought may act as triggers to push arid ecosystems experiencing chronic disturbances, such as grazing, past desertification ‘tipping points’ (CCSP 4.2 2008; Gillson and Hoffman 2007). An increase in the frequency of climate trigger events would make arid systems increasingly susceptible to major changes in vegetation cover. Climate is also a key factor dictating the effectiveness of resource management plans and restoration efforts (Holmgren and Scheffer 2001). Precipitation (and its interaction with temperature) plays a major role in determining how plant communities are impacted by, and how they respond to, a given type and intensity of disturbance. It is generally accepted that effects of grazing in arid lands may be somewhat mitigated in years of good rainfall and accentuated in drought years. However, this generalization is context

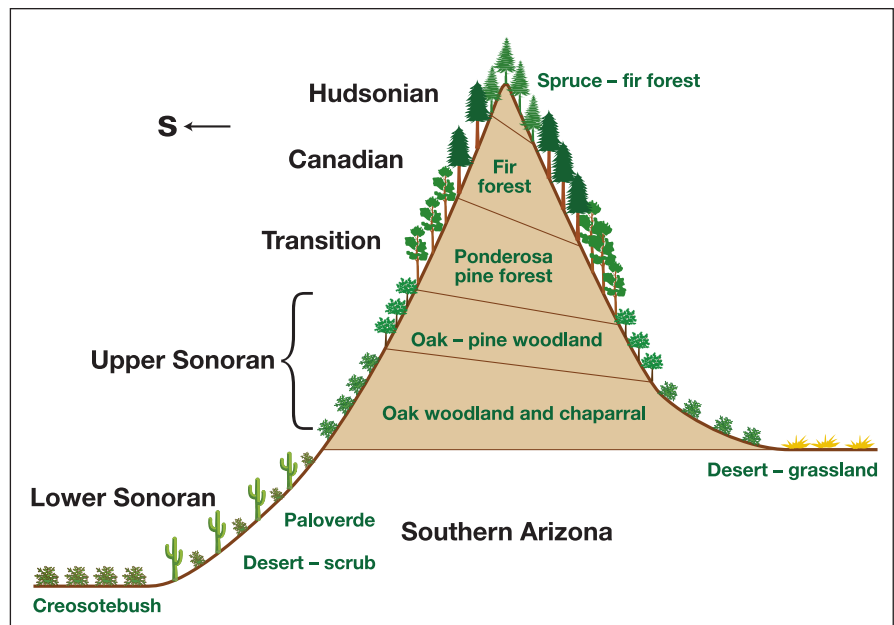


Figure 3.11 Elevation life zones along an arid land elevation gradient (from Brown 1994).



dependent. Landscape-scale factors such as rainfall and stocking rate affect grass cover in pre- and post-drought periods, but grass dynamics before, during, and after drought varies with species-specific responses to local patch-scale factors (e.g., soil texture, micro-topographic redistribution of water) (Yao et al. 2006). As a result, a given species may persist in the face of grazing and drought in some locales and be lost from others. Spatial context should thus be factored in to assessments of how changes in climate will affect ecosystem stability: their ability to maintain function in the face of disturbance (resistance), and the rate and extent to which they recover from disturbance (resilience). Advances in computing power, geographic information systems, and remote sensing now make this feasible.

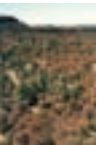
Chronic disturbance will also affect rates of ecosystem change in response to climate change because it reduces vegetation resistance to slow, long-term changes in climate (Cole 1985; Overpeck et al. 1990). Plant communities dominated by long-lived perennials may exhibit considerable biological inertia, and changes in community composition may lag behind significant changes in climate. Conditions required for seed germination are largely independent of conditions required for subsequent plant survival (Miriti 2007). Species established under previous climate regimes may thus persist in novel climates for long periods of time. Indeed, it has been suggested that the desert grasslands of the Southwest were established during the cooler, moister Little Ice Age but have persisted in the warmer, drier climates of the 19th and 20th centuries (Neilson 1986). Disturbances can create opportunities for species better adapted to the current conditions to establish. In the case of desert grasslands, livestock grazing subsequent to Anglo-European settlement may have been a disturbance that created opportunities for desert shrubs such as mesquite and creosote bush to increase in abundance. Rates of ecosystem compositional change in response to climate change may therefore depend on the type and intensity of disturbance, and the extent to which fundamental soil properties (especially depth and fertility) are altered by disturbance.

3.3.2.5 DESERTIFICATION

Precipitation and wind are agents of erosion. Wind and water erosion are primarily controlled by plant cover. Long-term reductions in plant cover by grazing and short-term reductions caused by fire create opportunities for accelerated rates of erosion; loss of soils feeds back to affect species composition in ways that can further reduce plant production and cover. Disturbances in arid lands can thus destabilize sites and quickly reduce their ability to capture and retain precipitation inputs. This is the fundamental basis for desertification, a long-standing concern (Van de Koppel et al. 2002). Desertification involves the expansion of deserts into semi-arid and subhumid regions, and the loss of productivity in arid zones. It typically involves loss of ground cover and soils, replacement of palatable, mesophytic grasses by unpalatable xerophytic shrubs, or both (Figure 3.12). There has been long-standing controversy in determining the relative contribution of climatic and anthropogenic factors as drivers of desertification. Local fence line contrasts argue for the importance of land use (e.g., changes in grazing, fire regimes); vegetation change in areas with no known change in land use argue for climatic drivers.

Grazing has traditionally been the most pervasive and extensive climate-influenced land use in arid lands (with the exception of areas where access to ground or surface water allows agriculture; see Chapter 2). Large-scale, unregulated livestock grazing in the 1800s and early 1900s is widely regarded as contributing to widespread desertification (Fredrickson et al. 1998). Grazing peaked around 1920 on public lands in the West, and by the 1970s had been reduced by approximately 70 percent (Holechek et al. 2003). These declines reflect a combination of losses in carrying capacity (ostensibly associated with soil erosion and reductions in the abundance of palatable species); the creation of federally funded experimental ranges in the early 1900s (e.g., the Santa Rita Experimental Range in Arizona, and the Jornada Experimental Range in New Mexico, which are charged with developing stocking rate guidelines); the advent of the science of range management; federal legislation intended to regulate grazing (Taylor Grazing Act of 1934) and combat soil

There has been long-standing controversy in determining the relative contribution of climatic and anthropogenic factors as drivers of desertification.



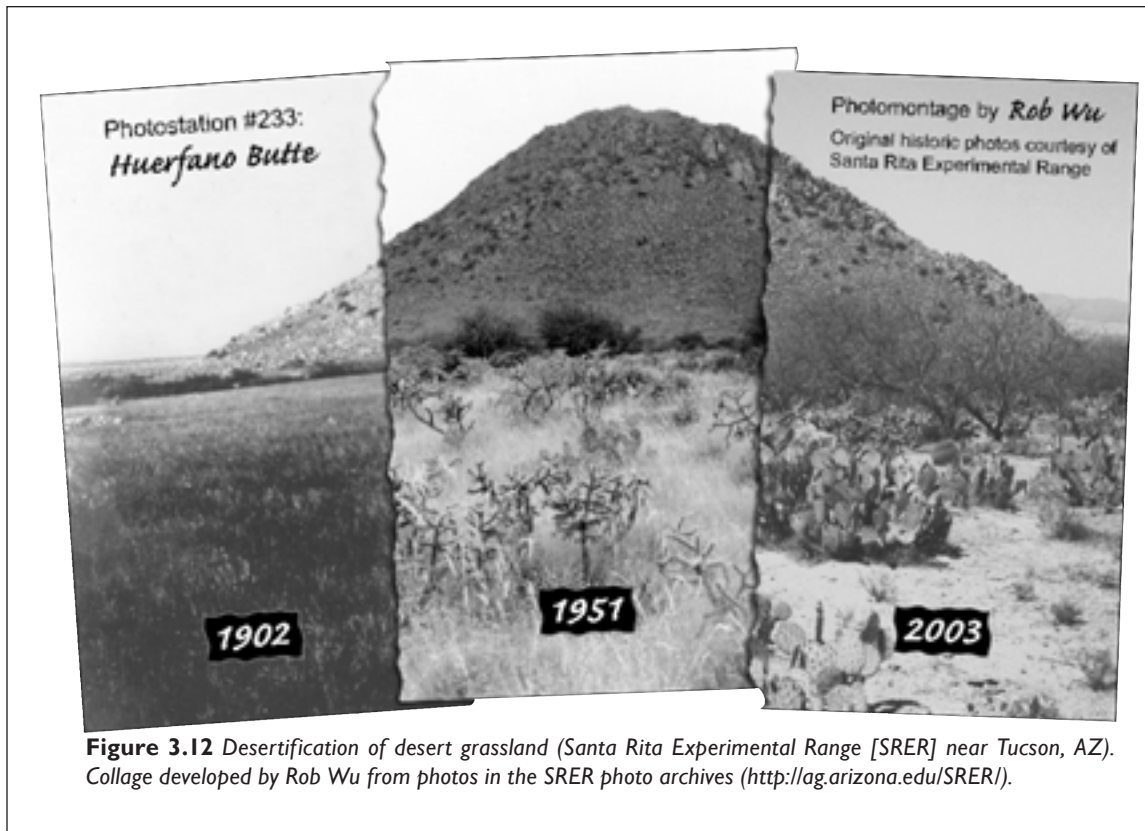


Figure 3.12 Desertification of desert grassland (Santa Rita Experimental Range [SRER] near Tucson, AZ). Collage developed by Rob Wu from photos in the SRER photo archives (<http://ag.arizona.edu/SRER/>).

erosion (Soil Erosion Act 1935); and the shift of livestock production operations to higher rainfall regions.

Arid lands can be slow to recover from livestock grazing impacts. Anderson and Inouye (2001) found that at least 45 years of protection was required for detectable recovery of herbaceous perennial understory cover in cold desert sagebrush steppe. Development of warmer, drier climatic conditions would be expected to further slow rates of recovery. On sites where extensive soil erosion or encroachment of long-lived shrubs occurs, recovery from grazing may not occur over time frames relevant to ecosystem management. While livestock grazing remains an important land use in arid lands, there has been a significant shift to exurban development and recreation, reflecting dramatic increases in human population density since 1950 (Hansen and Brown 2005). Arid land response to future climate will thus be mediated by new suites of environmental pressures such as air pollution and N-deposition, energy development, motorized off-road vehicles, feral pets, and invasion of non-native horticultural plants and grazing.

3.3.2.6 BIOTIC INVASIONS

Arid lands of North America were historically characterized by mixtures of shrublands, grasslands, shrub-steppe, shrub-savanna, and woodlands. Since Anglo-European settlement, shrubs and trees have increased at the expense of grasses (Archer 1994). Causes for this shift in plant-life-form abundance are the topic of active debate, but center around climate change, atmospheric CO₂ enrichment, nitrogen deposition, and changes in grazing and fire regimes (Archer et al. 1995; Van Auken 2000). In many cases, increases in woody plant cover reflect the proliferation of native shrubs or trees (mesquite, creosote bush, pinyon, juniper); in other cases, non-native shrubs have increased in abundance (tamarix). Historically, the displacement of grasses by woody plants in arid lands was of concern due to its perceived adverse impacts on stream flow and ground water recharge (Wilcox 2002; Owens and Moore 2007) and livestock production. More recently, the effects of this change in land cover has been shown to have implications for carbon sequestration, and land surface-atmosphere interactions (Schlesinger et al. 1990; Archer et al. 2001; Wessman et



al. 2004). Warmer, drier climates with more frequent and intense droughts are likely to favor xerophytic shrubs over mesophytic native grasses, especially when native grasses are preferentially grazed by livestock. However, invasions by non-native grasses are markedly changing the fire regime in arid lands and impacting shrub cover.

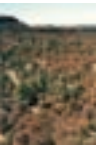
In arid lands of the United States, non-native grasses often act as “transformer species” (Richardson et al. 2000; Grice 2006) in that they change the character, condition, form or nature of a natural ecosystem over substantial areas. Land use and climate markedly influence the probability, rate, and pattern of alien species invasion, and future change for each of these drivers will interact to strongly impact scenarios of plant invasion and ecosystem transformation (Sala et al. 2000; Walther et al. 2002; Hastings et al. 2005). Plant invasions are strongly influenced by seed dispersal and resource availability, but disturbance and abrupt climatic changes also play key roles (Clarke et al. 2005). Changes in ecosystem susceptibility to invasion by non-native plants may be expected with changes in climate (Ibarra et al. 1995; Mau-Crimmins et al. 2006), CO₂ (Smith et al. 2000; Nagel et al. 2004) and nitrogen deposition (Fenn et al. 2003). Invasibility varies across elevation gradients. For cheatgrass, a common exotic annual in the Great Basin, invasibility is related to temperature at higher elevations and soil water availability at lower elevations. Increased variability in soil moisture and reductions in perennial herbaceous cover also increased susceptibility of low elevation sites to cheatgrass invasion (Chambers et al. 2007). In a 45-year study of cold desert sagebrush steppe that included the major drought of the 1950s, abundance of native species was found to be an important factor influencing community resistance to invasion (Anderson and Inouye 2001). Thus, maintenance of richness and cover of native species should be a high management priority in the face of climate change (see also Chapter 5, this report).

Non-native plant invasions, promoted by enhanced nitrogen deposition (Fenn et al. 2003) and increased anthropogenic disturbance (Wisdom et al. 2005), will have a major impact on how arid land ecosystems respond to

climate and climate change. Once established, non-native annual and perennial grasses can generate massive, high-continuity fine-fuel loads that predispose arid lands to fires more frequent and intense than those with which they evolved (Figure 3.13). The result is the potential for desert scrub, shrub-steppe, and desert grassland/savanna biotic communities to be quickly and radically transformed into monocultures of invasive grasses over large areas. This is already well underway in the cold desert region (Knapp 1998) and is in its early stages in hot deserts (Williams and Baruch 2000; Kupfer and Miller 2005; Salo 2005; Mau-Crimmins 2006). By virtue of their profound impact on the fire regime and hydrology, invasive plants in arid lands will very likely trump direct climate impacts on native vegetation where they gain dominance (Clarke et al. 2005). There is a strong climate-wildfire synchrony in forested ecosystems of western North America (Kitzberger et al. 2007); longer fire seasons and more frequent episodes of extreme fire weather are predicted (Westerling et al. 2006). As the areal extent of fire-prone exotic grass communities increases, low elevation arid ecosystems will likely experience similar climate-fire synchronization where none previously existed, and spread of low elevation fires upslope may constitute a new source of ignition for forest fires. Exurban development (Nelson 1992; Daniels 1999) has been and will continue to be a major source for both ignitions (Keeley et al. 1999) and exotic species introductions by escape from horticulture.

3.3.2.7 A SYSTEMS PERSPECTIVE

As reviewed in the preceding sections, the response of arid lands to climate and climate change is contingent upon the net outcome of several interacting factors (Fig 3.9). Some of these factors may reinforce and accentuate climate effects (e.g., soils, grazing); others may constrain, offset or override climate effects (soils, atmospheric CO₂ enrichment, fire, exotic species). Furthermore, extreme climatic events can themselves constitute disturbance (e.g., soil erosion and inundation associated with high intensity rainfall events and flooding; burial abrasion and erosion associated with high winds, mortality caused by drought and extreme temperature stress). Climate effects should thus be viewed in the context of other



factors, and simple generalizations regarding climate effects should be viewed with caution. This is not to say, however, that there is insufficient data and theory to guide prediction of future outcomes. Today's arid lands reflect a legacy of historic land uses, and future land use practices will arguably have the greatest impact on arid land ecosystems in the next two to five decades. In the near-term, climate fluctuation and change will be important primarily as it influences the impact of land use on ecosystems and how ecosystems respond to land use. Given the concomitant changes in climate, disturbance frequency/intensity, atmospheric CO₂, nitrogen deposition, and species invasions, it also seems likely that novel wildland and managed ecosystems will develop (Hobbs et al. 2006). Communities that are compositionally unlike any found today have occurred in the late-glacial past (Williams and Jackson 2007). In climate simulations for the IPCC emission scenarios, novel climates arise by 2100 AD. These future novel climates (which are warmer than any present climates, with spatially variable shifts in precipitation) increase the risk of species reshuffling into future novel communities and other ecological surprises (Williams and Jackson 2007). These novel ecosystems will present novel challenges and opportunities for conservation and management.

The following sections will address specific climate/land use/land cover issues in more detail. Section 3.10 will discuss climate and climate change effects on species distributions and community dynamics. Section 3.11 will review the consequences for ecosystem processes. Section 3.12 will focus on climate change implications for structure and function of riparian and aquatic ecosystems in arid lands. Implications for wind and water erosion will be reviewed in Section 3.13.

3.3.3 Species Distributions and Community Dynamics

3.3.3.1 CLIMATE-FIRE REGIMES

The climate-driven dynamic of the fire cycle is likely to become the single most important feature controlling future plant distributions in U.S. arid lands. Rising temperatures, decreases in precipitation and a shift in seasonality and variability, and increases in atmospheric CO₂



Figure 3.13 Top-down view of native big sagebrush steppe (right) invaded by cheatgrass, an exotic annual grass (left). Photo: Steve Whisenant.

and nitrogen deposition (Sage 1996), coupled with invasions of exotic grasses (Brooks et al. 2004; Brooks and Berry 2006) will accelerate the grass-fire cycle in arid lands and promote development of near monoculture stands of invasive plants (D'Antonio and Vitousek 1992). The frequency of fire in the Mojave Desert has dramatically increased over the past 20 years and effected a dramatic conversion of desert shrubland to degraded annual-plant landscapes (Bradley et al. 2006; Brooks and Berry 2006). Given the episodic nature of desert plant establishment and the high susceptibility of the new community structure to additional fire, it will be exceedingly difficult to recover native plant dominance. A similar conversion has occurred in many Great Basin landscapes (Knapp 1995, 1996), and given the longer period of non-native annual plant presence (Novak and Mack 2001), the pattern is much more advanced and has lowered ecosystem carbon storage (Bradley et al. 2006). Contemporary patterns in natural settings (Wood et al. 2006) and field experiments (Smith et al. 2000) suggest non-native response to climate change will be extremely important in the dynamics of arid land fire cycle, and changes in climate that promote fires will exacerbate land cover change in arid and semi-arid ecosystems.

There is some debate as to how climate contributed to a non-native component of this vegetation-disturbance cycle over the first half of the 20th century. For the upper elevations in the Sonoran Desert, Lehmann lovegrass, a perennial C₄ African grass introduced for



cattle forage and erosion control, has spread aggressively and independently of livestock grazing (McClaran 2003). Its success relative to native grasses appears related to its greater seedling drought tolerance and its ability to more effectively utilize winter moisture. Relatively wet periods associated with the Pacific Decadal Oscillation appear to have been more important than increases in N-deposition or CO₂ concentrations in the spread of these species (Salo 2005).

More recently, warm, summer-wet areas in northern Mexico (Sonora) and the southwestern United States have become incubators for perennial C₄ African grasses such as buffelgrass, purposely introduced to improve cattle forage in the 1940s. These grasses escape plantings on working ranches and, like exotic annual grasses, initiate a grass-fire cycle (Williams and Baruch 2000). In the urbanized, tourism-driven Sonoran Desert of southern Arizona, buffelgrass invasion is converting fireproof and picturesque desert scrub communities into monospecific, flammable grassland. Buffelgrass, like other neotropical exotics, is sensitive to low winter temperatures. The main invasion of buffelgrass in southern Arizona happened with warmer winters beginning in the 1980s, and its range will extend farther north and upslope as minimum temperatures continue to increase (Arriaga et al. 2004). This is complicated further by ongoing germplasm research seeking to breed more drought- and cold-resistant varieties. For example, a cold-resistant

“Frio” variety of buffelgrass recently released by USDA-Agricultural Research Service has been planted 40 km south of the Arizona border near Cananea, Mexico. Escape of “Frio” north of the United States-Mexico border may extend the potential niche of buffelgrass a few hundred meters in elevation and a few hundred kilometers to the north.

3.3.3.2 DROUGHT AND VEGETATION STRUCTURE

Over the past 75 years, the drought of the 1950s and the drought of the early 2000s represent two natural experiments for understanding plant community response to future environmental conditions. While both had similar reductions in precipitation, the 1950s drought was typical of many Holocene period droughts throughout the Southwest, whereas the drought that spanned the beginning of the 21st century was relatively hot (with both greater annual temperatures and greater summer maximum temperatures) (Mueller et al. 2005; Breshears et al. 2005). The 1950s drought caused modest declines in the major shrubs in the Sonoran Desert, whereas the 2000s drought caused much higher mortality rates in numerous species, including the long-lived C₃ creosote bush, which had shown essentially no response to the 1950s drought (Bowers 2005). A similar pattern was seen in comparing the two time periods for perennial species in the Mojave Desert, where dry periods close to the end of the 20th century were associated with reductions in C₃ shrubs and both C₃ and C₄ perennial grass species (Hereford et

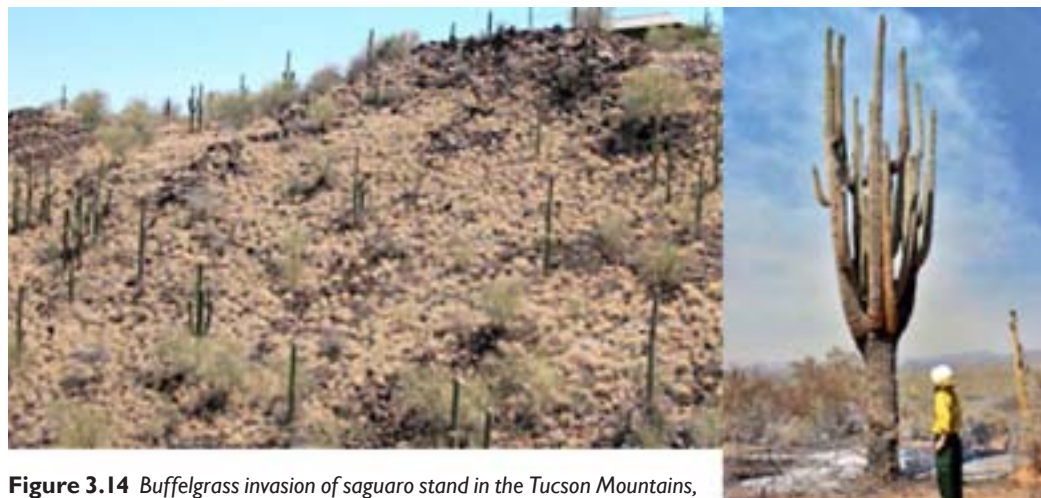


Figure 3.14 Buffelgrass invasion of saguaro stand in the Tucson Mountains, Arizona (left); fire-damaged saguaro (right). Photos courtesy Ben Wilder.

al. 2006). Thus, the greater temperatures and higher rates of evapotranspiration predicted to co-occur with drought portend increased mortality for the dominant woody vegetation typical of North American deserts, and open the door for establishment of non-native annual grasses. These patterns are mostly driven by changes in winter precipitation, but in systems where summer rainfall is abundant, woody plant-grass interactions may also be important. Given the projected increases in the frequency of these “global warming type” droughts (e.g., Breshears et al. 2005), increases in summer active, non-native C_4 grasses (such as buffelgrass in the Sonoran Desert (Franklin et al. 2006)), and the increased probability of fire (Westerling et al. 2006), a similar pattern of a wide-spread woody vegetation conversion to degraded non-native grasslands can be anticipated for the hot deserts of North America – a pattern similar to that already seen in the Great Basin (Bradley et al. 2006).

3.3.3.3 PLANT FUNCTIONAL GROUP RESPONSES

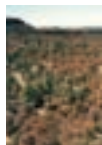
Annual plants are a major source of plant diversity in the North American deserts (Beatley 1967), but exotic annuals are rapidly displacing native annuals. The density of both native and non-native desert annuals in the Sonoran Desert, at Tumamoc Hill in Tucson, AZ, has been reduced by an order of magnitude since 1982 (from ~2,000 plants/m² to ~150/plants m²) (Venable and Pake 1999). Similar reductions have been recorded for the Nevada Test Site (Rundel and Gibson 1996). At the same time, there has been an increase in the number of non-native annual species (Hunter 1991; Salo et al. 2005; Schutzenhofer and Valone 2006). High CO₂ concentrations appear to benefit non-native grasses and “weeds” more so than native species (Huxman and Smith 2001; Ziska 2003; Nagel et al. 2004). Thus, when rainfall is relatively high in the Mojave Desert, non-natives comprise about 6 percent of the flora and ~66 percent of the community biomass, but when rainfall is restricted, they comprise ~27 percent of the flora and >90 percent of the biomass (Brooks and Berry 2006). Competition between annuals and perennials for soil nitrogen during relatively wet periods can be intense (Holzapfel and Mahall 1999). At the western fringe of the Mojave and Sonoran Deserts, nitrogen deposi-

tion is tipping the balance toward the annual plant community (typically non-native) with the resulting loss of woody native species (Wood et al. 2006).

Based on theory and early experiments, rising atmospheric CO₂ and increasing temperature are predicted to increase the competitive ability of C_3 versus C_4 plants in water-limited systems, potentially reducing the current pattern of C_4 dominance in many warm season semi-arid ecosystems (Long 1991; Ehleringer et al. 1997; Poorter and Navas 2003). Photosynthesis and stomatal conductance of leaves of plants in mixed C_3/C_4 communities often show a greater proportional response in C_3 as compared to C_4 species at elevated CO₂ (Polley et al. 2002). However, community composition and productivity do not always reflect leaf level patterns and more sophisticated experiments show complex results. It is likely that whole-system water budgets are significantly altered and more effectively influence competitive interactions between C_3 and C_4 species as compared to any direct CO₂ effects on leaf function (Owensby et al. 1993; Polley et al. 2002). In the Great Basin, which is dominated by C_3 plants, CO₂ enrichment favors non-native annual cheatgrass over native C_3 plants (Smith et al. 2000; Ziska et al. 2005).

Where C_3 species have increased in abundance in elevated CO₂ experiments (Morgan et al. 2007), the photosynthetic pathway variation also reflected differences in herbaceous (C_4) and woody (C_3) life forms. CO₂ enhancement of C_3 woody plant seedling growth, as compared to growth of C_4 grasses, may facilitate woody plant establishment (Polley et al. 2003). Reduced transpiration rates from grasses under higher CO₂ may also allow greater soil water recharge to depth, and favor shrub seedling establishment (Polley et al. 1997). Changes in both plant growth and the ability to escape the seedling-fire-mortality constraint are critical for successful shrub establishment in water-limited grasslands (Bond and Midgley 2000). However, interactions with other facets of global change may constrain growth form and photosynthetic pathway responses to CO₂ fertilization. Increased winter temperatures would lengthen the C_4 growing season. Greater primary production at elevated CO₂ combined with increased

Based on theory and early experiments, rising atmospheric CO₂ and increasing temperature are predicted to increase the competitive ability of C_3 versus C_4 plants in water-limited systems, potentially reducing the current pattern of C_4 dominance in many warm season semi-arid ecosystems.



abundance of non-native grass species may alter fire frequencies (see sections 3.2.2.6 and 3.3.3.1 and 3.3.4.1). Nitrogen deposition may homogenize landscapes, favoring grassland physiognomies over shrublands (Reynolds et al. 1993). Changes in the occurrence of episodic drought may alter the relative performance of these growth forms in unexpected ways (Ward et al. 1999). Predicting changes in C_3 versus C_4 dominance, or changes in grass versus shrub abundance in water-limited ecosystems, will require understanding of multifactor interactions of global change the scientific community does not yet possess.

3.3.3.4 CHARISMATIC MEGA FLORA

Saguaro density is positively associated with high cover of perennial vegetation and mean summer precipitation, but total annual precipitation and total perennial cover are the best predictors of reproductive stem density (Drezner 2006). Because of how these drivers co-vary in the southwestern United States, the drier western regions have lower saguaro densities than the southeastern region where summer rainfall is greater. Additionally, the Northeast and Southeast both have very high reproductive stem densities relative to the West. These patterns reflect the interaction between summer rainfall and the frequency of episodic freezing events that constrain the species' northern range boundary. Despite predicted reductions in the number of freezing events (Weiss and Overpeck 2005), predicted increases in annual temperature, loss of woody plant cover from a greater frequency of "global warming-type" droughts, and increasing fire resulting from non-native grass invasions (Figure 3.14) suggest a restriction of the Saguaro's geographic range and reductions in abundance within its historic range.

The direct effects of rising CO_2 on climatic tolerance and growth of Joshua trees also suggest important shifts in this Mojave Desert species' range (Dole et al. 2003). Growth at elevated CO_2 improves the ability of seedlings to tolerate periods of cold temperature stress (Loik et al. 2000). When applied to downscale climate outputs and included in the rules that define species distribution, this direct CO_2 effect suggests the potential for a slight increase in geographic range. However, like all long-lived,

large-statured species in the North American deserts, the frequency of fire will be a primary determinant of whether this potential will be realized.

3.3.4 Ecosystem Processes

3.3.4.1 NET PRIMARY PRODUCTION AND BIOMASS

Semi-arid and arid ecosystems of the western United States are characterized by low plant growth (NPP), ranging from 20 to 60 $g/m^2/yr$ in the Mojave Desert of Nevada (Rundel and Gibson 1996) to 100 to 200 $g/m^2/yr$ (aboveground) in the Chihuahuan Desert of New Mexico (Huenneke et al. 2002). In most studies, the belowground component of plant growth is poorly characterized, but observations of roots greater than 9 meters deep suggest that root production could be very large and perhaps underestimated in many studies (Canadell et al. 1996).

With water as the primary factor limiting plant growth, it is not surprising that the variation in plant growth among desert ecosystems, or year-to-year variation within arid ecosystems, is related to rainfall. High spatial and interannual variation make it difficult to identify trends in aboveground net primary production (ANPP) over time, especially when disturbances such as livestock grazing co-occur as an additional confounding factor. In their comparison of cold desert sagebrush steppe vegetation structure and production during two 10-year studies from the late 1950s to the late 1960s and three years in the 1990s, West and Yorks (2006) noted high coefficients of variation in aboveground plant production associated with five-fold differences in precipitation at a given locale, sometimes in consecutive years. In the Chihuahuan Desert, shrub encroachment into desert grassland has increased the spatial heterogeneity of ANPP and soil nutrients (Schlesinger and Pilmanis 1998; Huenneke et al. 2002). Although grasslands tended to support higher ANPP than did shrub-dominated systems, grasslands demonstrated higher interannual variation. Projected increases in precipitation variability coupled with changes in species composition would be expected to further increase the already substantial variation in arid land plant production. Other factors, such as soil texture and landscape position, also affect soil moisture availability and determine plant growth in local conditions



(Schlesinger and Jones 1984; Wainwright et al. 2002). Increases in temperature and changes in the amount and seasonal distribution of precipitation in cold deserts (Wagner 2004) and hot deserts (Seager et al. 2007) can be expected to have a dramatic impact on the dominant vegetation, NPP, and carbon storage in arid lands.

Jackson et al. (2002) found that plant biomass and soil organic matter varied systematically in mesquite-dominated ecosystems across west Texas and eastern New Mexico, demonstrating some of the changes that can be expected with future changes in rainfall regimes. The total content of organic matter (plant + soil) in the ecosystem was greatest at the highest rainfall, but losses of soil carbon in the driest sites were compensated by increases in plant biomass, largely mesquite. Despite consistent increases in aboveground carbon storage with woody vegetation encroachment, a survey of published literature revealed no correlation between mean annual rainfall and changes in soil organic carbon pools subsequent to woody plant encroachment (Asner and Archer in press). Differences in soil texture, topography, and historical land use across sites likely confound assessments of precipitation influences on soil organic carbon pool responses to vegetation change.

3.3.4.2 SOIL RESPIRATION

Soil respiration includes the flux of CO₂ from the soil to the atmosphere from the combined activities of plant roots and their associated mycorrhizal fungi and heterotrophic bacteria and fungi in the soil. It is typically measured by placing small chambers over replicated plots of soil or estimated using eddy-covariance measurements of changes in atmospheric properties, particularly at night. Soil respiration is the dominant mechanism that returns plant carbon dioxide to Earth's atmosphere, and it is normally seen to increase with increasing temperature. Mean soil respiration in arid and semi-arid ecosystems is 224 g C/m²/yr (Raich and Schlesinger 1992; Conant et al. 1998), though in individual sites, it can be expected to vary with soil moisture content during and between years.

Intensification of the hydrologic cycle due to atmospheric warming is expected reduce rainfall frequency, but increase the intensity and/or size

of individual precipitation events. A change in the size-class distribution of precipitation has important implications for instantaneous fluxes of carbon dioxide from soils and the potential for ecosystems to sequester carbon (Austin et al. 2004; Huxman et al. 2004a; Jarvis et al. 2007). This is due to differences in the way soil microbial populations and plants respond to moisture entering the soil following rainfall events of different sizes. Larger rainfall events that increase the wetting depth in the soil profile should increase the number of periods within a year where substantial plant activity and carbon storage can occur (Huxman et al. 2004b; Pereira et al. 2007; Kurc and Small 2007; Patrick et al. 2007). However, reducing the frequency of wet-dry cycles in soils will retard microbial activity and nutrient cycling, likely introducing a long-term nitrogen limitation to plant growth (Huxman et al. 2004a). For winter rainfall ecosystems, these shifts in wet-dry cycles can cause reductions in productivity and soil carbon sequestration (Jarvis et al. 2007; Pereira et al. 2007).

3.3.4.3 NET CARBON BALANCE

The net storage or loss of carbon in any ecosystem is the balance between carbon uptake by plants (autotrophic) and the carbon released by plant respiration and heterotrophic processes. Although elegant experiments have attempted to measure these components independently, the difference between input and output is always small and thus measurement errors can be proportionately large. It is usually easier to estimate the accumulation of carbon in vegetation and soils on landscapes of known age. This value, NEP, typically averages about 10 percent of NPP in forested ecosystems. Arid soils contain relatively little soil organic matter, and collectively make only a small contribution to the global pool of carbon in soils (Schlesinger 1977; Jobbagy and Jackson 2002). Given the low NPP of arid lands, they are likely to result in only small amounts of carbon sequestration. Since soil organic matter is inversely related to mean annual temperature in many arid regions (Schlesinger 1982; Nettleton and Mays 2007), anticipated increases in regional temperature will lead to a loss of soil carbon to the atmosphere, exacerbating increases in atmospheric carbon dioxide. Recent measurements of NEP by micrometeorological techniques, such as

Given the low NPP of arid lands, they are likely to result in only small amounts of carbon sequestration.



eddy covariance, across relatively large spatial scales confirm this relatively low carbon uptake for arid lands (Grunzweig et al. 2003), but point to the role of life-form (Unland et al. 1996), seasonal rainfall characteristics (Hastings et al. 2005; Ivans et al. 2006), and potential access to groundwater as important modulators of the process (Scott et al. 2006).

Several scientists have suggested that arid lands might be managed to sequester carbon in soils and mitigate future climate change (Lal 2001). The prospects for such mitigation are limited by the low sequestration rates of organic and inorganic carbon that are seen in arid lands under natural conditions (Schlesinger 1985, 1990), the tendency for warmer soils to store lesser amounts of soil organic matter, and the small increases in net productivity that might be expected in these lands in a warmer, drier future climate. Moreover, when desert lands are irrigated, there can be substantial releases of carbon dioxide from the fossil fuels used to pump irrigation water (Schlesinger 2001). Globally, the greatest potential for soil carbon sequestration is found in soils that are cold and/or wet, not in soils that are hot and dry.

In many areas of desert, the amount of carbon stored in inorganic soil carbonates greatly exceeds the amount of carbon in vegetation and soil organic matter, but the formation of such carbonates is slow and not a significant sink for carbon in its global cycle (Schlesinger 1982; Monger and Martinez-Rios 2000). Some groundwater contains high (supersaturated) concentrations of carbon dioxide, which is released to the atmosphere when this water is brought to the Earth's surface for irrigation, especially when carbonates and other salts precipitate (Schlesinger 2000). Thus, soil carbonates are unlikely to offer significant potential to sequester atmospheric carbon dioxide in future warmer climates.

3.3.4.4 BIOGEOCHEMISTRY

Arid-land soils often have limited supplies of nitrogen, such that nitrogen and water can "co-limit" the growth of vegetation (Hooper and Johnson 1999). These nitrogen limitations normally appear immediately after the receipt of seasonal rainfall. The nitrogen limitations

of arid lands stem from small amounts of N received by atmospheric deposition and nitrogen fixation and rather large losses of N to wind erosion and during microbial transformations of soil N that result in the losses of ammonia (NH_3), nitric oxide (NO), nitrous oxide (N_2O), and nitrogen gas (N_2) to the atmosphere (Schlesinger et al. 2006). These microbial processes are all stimulated by seasonal rainfall, suggesting that changes in the rainfall regime as a result of climate change will alter N availability and plant growth. N deposition is spatially variable, being greater in areas downwind from major urban centers such as Los Angeles, increasing the abundance of herbaceous vegetation and potentially increasing the natural fire regime in the Mojave Desert (Brooks 2003).

In arid lands dominated by shrub vegetation, the plant cycling of N and other nutrients is often heterogeneous, with most of the activity focused in the soils beneath shrubs (Schlesinger et al. 1996). It remains to be seen how these local nutrient hot spots will influence vegetation composition and ecosystem function in future environments. In cold desert shrub steppe, non-native cheatgrass is often most abundant under shrubs, resulting in rapid consumption of the shrub during fire and mortality of native plants and seed banks; the higher available resources on the fertile island enables greater biomass and seed production of cheatgrass in the post-fire period (Chambers et al. 2007). Thus, the rate and extent of invasion of cold desert sagebrush-steppe by cheatgrass may initially be a function of the cover and density of sagebrush plants and the fertile islands they create.

3.3.4.5 TRACE GASES

In addition to significant losses of N trace gases, some of which confer radiative forcing on the atmosphere (e.g., N_2O), deserts are also a minor source of methane, largely resulting from activities of some species of termites, and volatile organic compounds (VOC) and non-methane hydrocarbon gaseous emissions from vegetation and soils (Geron et al. 2006). Isoprene, produced by many woody species and a few herbaceous species, is the dominant VOC released by vegetation; the ability to produce significant amounts of isoprene may or may not be shared by members of the same plant family



or genus (Harley et al. 1999). No phylogenetic pattern is obvious among the angiosperms, with the trait widely scattered and present (and absent) in both primitive and derived taxa, although confined largely to woody species. VOCs can serve as precursors to the formation of tropospheric ozone and organic aerosols, thus influencing air pollution. Emissions of such gases have increased as a result of the invasion of grasslands by desert shrubs during the past 100 years (Guenther et al. 1999), and emissions of isoprene are well known to increase with temperature (Harley et al. 1999). The flux of these gases from arid lands is not well studied, but is known to be sensitive to temperature, precipitation, and drought stress. For example, total annual VOC emissions in deserts may vary threefold between dry and wet years, and slight increases in daily leaf temperatures can increase annual desert isoprene and monoterpene fluxes by 18 percent and 7 percent, respectively (Geron et al. 2006). Thus, changes in VOC emissions from arid lands can be expected to accompany changes in regional and global climate.

3.3.5 ARID LAND RIVERS AND RIPARIAN ZONES

Springs, rivers and floodplain (riparian) ecosystems commonly make up less than 1 percent of the landscape in arid regions of the world. Their importance, however, belies their small areal extent (Fleischner 1994; Sada et al. 2001; Sada and Vinyard 2002). These highly productive ecosystems embedded within much lower productivity upland ecosystems attract human settlement and are subjected to a variety of land uses. They provide essential wildlife habitat for migration and breeding, and these environments are critical for breeding birds, threatened and endangered species, and arid-land vertebrate species. Riparian vegetation in arid lands can occur at scales from isolated springs to ephemeral and intermittent watercourses, to perennial rivers (Webb and Leake 2006). The rivers and riparian zones of arid lands are dynamic ecosystems that react quickly to changing hydrology, geomorphology, human utilization, and climate change. Certain types of springs can also be highly responsive to these changes. As such, spring, river and riparian ecosystems will likely prove to be responsive components of arid landscapes to future climate change.

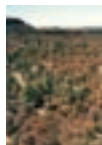
Effects of climate change on aquatic organisms in arid lands are not well known. Introductions of non-native fish and habitat modification have caused the extinction of numerous endemic species, subspecies and populations of fishes, mollusks and insects since the late 1800s. Declines in abundance or distribution have been attributed to (in order of decreasing importance) water flow diversions, competitive or predatory interactions with non-native species, livestock grazing, introductions for sport fisheries management, groundwater pumping, species hybridization, timber harvest, pollution, recreation and habitat urbanization (reviewed by Sada and Vinyard 2002). Most taxa were influenced by multiple factors. It is likely that projected climate changes will exacerbate these existing threats via effects on water temperature, sedimentation, and flows.

Global climate change can potentially impact river and riparian ecosystems in arid regions through a wide variety of mechanisms and pathways (Regab and Prudhomme 2002). Three pathways in which riverine corridors in arid lands are highly likely to be affected are particularly important. The first is the impact of climate change on water budgets. Both sources of water and major depletions will be considered. The second is competition between native and non-native species in a changing climate. The potential importance of thresholds in these interactions will be explicitly considered. The third mechanism pertains to the role of extreme climate events (e.g., flood and droughts) in a changing climate. Extreme events have always shaped ecosystems, but the interactions of a warmer climate with a strengthened and more variable hydrologic cycle are likely to be significant structuring agents for riverine corridors in arid lands.

3.3.5.1 WATER BUDGETS

Analysis of water budgets under a changing climate is one tool for assessing the impact of climate change on arid-land rivers and riparian zones. Christensen et al. (2004) have produced a detailed assessment of the effects of climate change on the hydrology and water resources of the Colorado River basin. Hydrologic and water resources scenarios were evaluated through

Global climate change can potentially impact river and riparian ecosystems in arid regions through a wide variety of mechanisms and pathways.



coupling of climate models, hydrologic models, and projected greenhouse gas scenarios for time periods 2010-2039, 2040-2069, and 2070-2099. Average annual temperature changes for the three periods were 1.0°C, 1.7°C, and 2.4°C, respectively, and basin-average annual precipitation was projected to decrease by 3, 6, and 3 percent for the three periods, respectively. These scenarios produced annual runoff decreases of 14, 18, and 17 percent from historical conditions for the three designated time periods. Such decreases in runoff will have substantial effects on human populations and river and riparian ecosystems, particularly in the lower elevation arid land compartments of this heavily appropriated catchment (e.g., Las Vegas and southern California).

Changing climate also can have a significant effect on major depletions of surface waters in arid regions. Dahm et al. (2002) examined major depletions along a 320-km reach of the Rio Grande in central New Mexico. Major

depletions were reservoir evaporation, riparian zone evapotranspiration, agriculture, groundwater recharge, and urban/suburban use. All of these depletions are sensitive to climate warming. Reservoir evaporation is a function of temperature, wind speed, and atmospheric humidity. Riparian zone evapotranspiration is sensitive to the length of the growing season, and climate warming will lengthen the period of time that riparian plants will be actively respiring (Goodrich et al. 2000; Cleverly et al. 2006), and also increase the growing season for agricultural crops dependent on riparian water. Temperature increases positively affect groundwater recharge rates from surface waters through changes in viscosity (Constantz and Thomas 1997; Constantz et al. 2002). The net result of climate warming is greater depletion of water along the riverine corridor (Figure 3.15). Global warming will place additional pressure on the major depletions of surface water in arid regions, in addition to likely effects on the supply side of the equation.

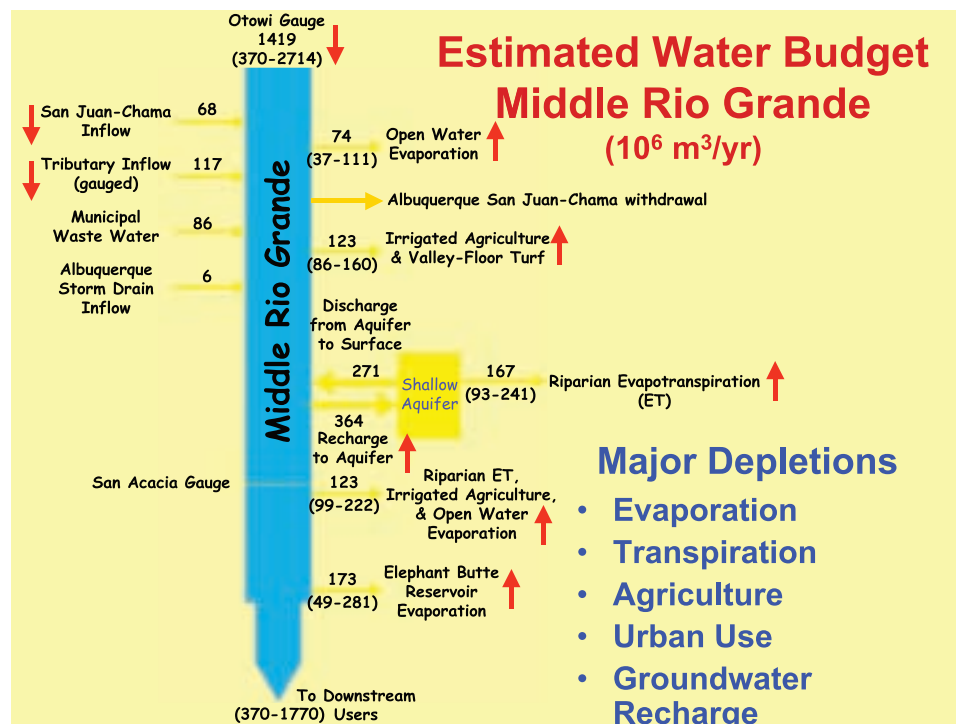


Figure 3.15 A water budget for a 320-km segment of the Middle Rio Grande of New Mexico, USA, with water sources on the left and top, depletions on the right, and downstream output on the bottom (Dahm et al. 2002). The red arrows indicate the direction of change for various water sources and depletions predicted with a warmer climate. Otowi Gauge values are a 26 year mean with range; releases from Elephant Butte dam are ranges only, because releases vary depending on delivery requirements and because releases sometimes include storage water (dam volume is being drawn down) or is much less than inflow (water going into storage). Ranges reflect both interannual variability and measurement uncertainty. The budget balances, but only coarsely, because of the large ranges.

3.3.5.2 NATIVE AND NON-NATIVE PLANT INTERACTIONS

Competition between native and non-native species in a changing climate is a second area where climate change is predicted to have a substantial effect on riparian zones of arid lands. Riparian zones of arid lands worldwide are heavily invaded by non-native species of plants and animals (Prieur-Richard and Lavorel 2000; Tickner et al. 2001). Salt cedar and Russian olive are particularly effective invaders of the arid land riparian zones of the western United States (Figure 3.16) (Brock 1994; Katz and Shafroth 2003). Shallow ground water plays an important role in structuring riparian plant communities (Stromberg et al. 1996) and groundwater level decline, both by human depletions and intensified drought in a changing climate, will alter riparian flora. Stromberg et al. (1996) describe riparian zone “desertification” from a lowered water table whereby herbaceous species and native willows and cottonwoods are negatively impacted. Horton et al. (2001a, b) describe a threshold effect where native canopy dieback occurs when depth to ground water exceeds 2.5-3.0 meters. Non-native salt cedar (*Tamarix chinensis*), however, are more drought tolerant when water tables drop, and readily return to high rates of growth when water availability again increases. Plant responses like these are predicted to shift the competitive balance in favor of non-native plants and promote displacement of native plants in riparian zones under a warmer climate.

Another example of a threshold effect on river and riparian ecosystems in arid lands is the persistence of aquatic refugia in a variable or changing climate. Hamilton et al. (2005) and Bunn et al. (2006) have shown the critical importance of waterhole refugia in the maintenance of biological diversity and ecosystem productivity in arid-land rivers. Arid regions worldwide, including this example from inland Australia, are dependent on the persistence of these waterholes during drought. Human appropriation of these waters or an increase in the duration and intensity of drought due to climate change would dramatically affect aquatic biodiversity and the ability of these ecosystems to respond to periods of enhanced water availability. For example, most waterhole refugia throughout the entire basin would be lost if drought persisted for more than two years in the Cooper Creek basin of Australia, or if surface diversions of flood waters reduced the available water within refugia in the basin (Hamilton et al. 2005; Bunn et al. 2006). Desiccation of waterholes could become more common if climate change increases annual evapotranspiration rates or if future water withdrawals reduce the frequency and intensity of river flows to waterholes. Roshier et al. (2001) pointed out that temporary wetland habitats throughout arid lands in Australia are dependent upon infrequent, heavy rainfalls and are extremely vulnerable to any change in frequency or magnitude. Climate change that induces drying or reduced frequency of large floods would



Figure 3.16 Non-native salt cedar (right) has invaded and displaced native cottonwood and poplar forests (left) in many southwestern riparian corridors. Photo credits: Jim Thibault and James Cleverly.



deleteriously impact biota, particularly water birds that use these temporary arid land habitats at broad spatial scales.

3.3.5.3 EXTREME EVENTS

The role of extreme events (e.g., flood and droughts) in a changing climate is predicted to increase with a warmer climate (IPCC 2007). Extreme climatic events are thought to strongly shape arid and semi-arid ecosystems worldwide (Holmgren et al. 2006). Climate variability, such as associated with the El Niño Southern Oscillation (ENSO) phenomenon, strongly reverberates through food webs in many arid lands worldwide. Fluvial systems and riparian vegetation are especially sensitive to the timing and magnitude of extreme events, particularly the timing and magnitude of minimum and maximum flows (Auble et al. 1994). GCMs do not yet resolve likely future regional precipitation regimes or future temperature regimes. A stronger overall global hydrologic cycle, however, argues for more extreme events in the future (IPCC 2007). The ecohydrology of arid-land rivers and riparian zones will certainly respond to altered precipitation patterns (Newman et al. 2006), and the highly variable climate that characterizes arid lands is likely to become increasingly variable in the future.

3.3.6 Water and Wind Erosion

Due to low and discontinuous cover, there is a strong coupling between vegetation in arid lands and geomorphic processes such as wind and water erosion (Wondzell et al. 1996). Erosion by wind and water has a strong impact on ecosystem processes in arid regions (Valentin et al. 2005; Okin et al. 2006). Erosion impacts the ability of soils to support plants and can deplete nutrient-rich surface soils, thus reducing the probability of plant establishment and recruitment. Although erosion by water has received by far the most attention in the scientific literature, the few studies that have investigated both wind and water erosion have shown that they can be of similar magnitude under some conditions (Breshears et al. 2003).

3.3.6.1 WATER EROSION

Water erosion primarily depends on the erosivity of precipitation events (rainfall rate, storm duration, and drop size) and the erodibility of the surface (infiltration rate, slope, soil, and

vegetation cover). Climate change may impact all of these except slope. For instance, it is well established that the amount of soil that is detached (and hence eroded) by a particular depth of rain is related to the intensity at which this rain falls. Early studies suggest soil splash rate is related to rainfall intensity and raindrop fall velocity (Ellison 1944; Bisal 1960). It is also well established that the rate of runoff depends on soil infiltration rate and rainfall intensity. When rainfall intensity exceeds rates of infiltration, water can run off as inter-rill flow, or be channeled into rills, gullies, arroyos, and streams. The intensity of rainfall is a function of climate, and therefore may be strongly impacted by climate change. The frequency of heavy precipitation events has increased over most land areas, including the United States, which is consistent with warming and observed increases in atmospheric water vapor (IPCC 2007). Climate models predict additional increases in the frequency of heavy precipitation, and thus highly erosive events. Warming climates may also be responsible for changes in surface soils themselves, with important implications for the erodibility of soils by water. In particular, higher temperatures and decreases in soil moisture, such as those predicted in many climate change scenarios, have been shown to decrease the size and stability of soil aggregates, thus increasing their susceptibility to erosion (Lavee et al. 1998).

By far the most significant impact of climate change on water erosion is via its effects on vegetation cover. Vegetation conversion to annual grasses or weedy forbs can result in loss of soil nutrients, siltation of streams and rivers, and increased susceptibility to flooding (Knapp 1996). Although some fireproof shrublands in the Southwest have been invaded by non-native grasses, thus changing the fire ecology and endangering those ecosystems (Knapp 1996; Bradley et al. 2006), many other areas have experienced the loss of native perennial grasses, which have been replaced by shrubs (van Auken 2000; sections 3.9.4 and 3.9.5). This widespread conversion of grasslands to shrublands throughout the desert Southwest has resulted in significantly greater erosion, though research on natural rainfall events to quantify the total amounts of erosion is ongoing. Flow and erosion plots in the Walnut Gulch Experimental



Watershed in Arizona and the Jornada LTER site in New Mexico have demonstrated significant differences in water erosion between grasslands and shrublands (Wainwright et al. 2000). For instance, greater splash detachment rates (Parsons et al. 1991, 1994, 1996), and interrill erosion rates (Abrahams et al. 1988) are observed in shrublands compared to grasslands; shrubland areas are more prone to develop rills, which are responsible for significant increases in overall erosion rates (Luk et al. 1993). Episodes of water erosion are often associated with decadal drought-interdrought cycles because depressed vegetation cover at the end of the drought makes the ecosystem vulnerable to increased erosion when rains return (McAuliffe et al. 2006). No study to date has used climate models to estimate how the periodicity of these cycles might change in the future.

U.S. arid regions have already experienced dramatic increases in erosion rates due to widespread losses of vegetation cover. These changes have created conditions where anticipated increases in precipitation intensity, coupled with reductions in soil aggregate stability due to net warming and drying, will likely increase potential erosion rates dramatically in coming decades.

3.3.6.2 WIND EROSION

As with water erosion, the magnitude of wind erosion is related to both the erosivity of the wind and the erodibility of the surface. However, the impact of increased wind erosion in deserts can have continental-scale impacts because the resulting dust can travel long distances with significant impacts to downwind ecosystems, air quality, and populations. Both hemispheres have experienced strengthening of mid-latitude westerly winds since the 1960s (IPCC 2007). This trend is likely to continue into the near future. Thus, desert regions of the United States are likely to experience more erosive conditions in the near future.

The susceptibility of soil to erosion by wind is determined by both the erodibility of the surface soil and the amount of vegetation present to disrupt wind flows and shelter the surface from erosion. Anticipated net aridification in the desert Southwest (Seager et al. 2007) is likely to lead to a decrease in soil aggregate-

size and stability. Increased temperatures and drought occurrence will result in lower relative humidity in arid lands. Because the top few millimeters of soil are in equilibrium with soil moisture in the overlying air, the decrease in relative humidity may result in soils that require less wind power to initiate erosion (Ravi et al. 2006). Increased drought occurrence throughout the western United States can further lead to lower soil moisture content, which can also increase the erodibility of the soil (Bisal 1960; Cornelis et al. 2004).

Short-term changes in vegetation cause significant changes in the wind erodibility of the surface. For instance, Okin and Reheis (2002) and Reheis (2006) have shown that annual variation in wind erosion on a regional scale is related to variation in precipitation. There appears to be a one-year lag in this effect, with low precipitation one year resulting in significant wind erosion and dust emission the following year. This lag is hypothesized to be due to the fact that the effect of low precipitation must propagate through the system by first affecting vegetation cover. This one-year lag effect has been observed in other arid systems (Zender and Kwon 2005). In addition, dust emission from dry lakes or playas in the desert Southwest also appears to occur after years of particularly intense rainfall. This phenomenon seems to result from the increased delivery of fine-grained sediment to these playas during especially wet years or years with intense rainfall events. Anticipated climatic changes in the coming decades include both increased drought frequency and also increased precipitation intensity during rain events (IPCC 2007). Both of these effects are likely to increase wind erosion and dust emission in arid regions due to, in the first case, suppression of vegetation and, in the second case, greater water erosion resulting in increased delivery of sediment to dry lakes.

Long-term and ongoing vegetation changes in arid regions, namely the conversion of grasslands to shrublands, have dramatically increased wind erosion and dust production due to increased bare areas in shrublands compared to the grasslands they replaced. Measurements of aeolian sediment flux in the Chihuahuan Desert have shown nearly ten-fold greater rates of wind erosion and dust emission in mesquite-

Long-term and ongoing vegetation changes in arid regions, namely the conversion of grasslands to shrublands, have dramatically increased wind erosion and dust production due to increased bare areas in shrublands compared to the grasslands they replaced.



dominated shrublands compared to grasslands on similar soils (Gillette and Pitchford 2004). Large-scale conversion of grasslands to shrublands, coupled with anticipated changes in climate in the coming decades, and increases in wind speed, temperature, drought frequency, and precipitation intensity, contribute to greater wind erosion and dust emission from arid lands.

3.3.6.3 IMPACTS OF WATER AND WIND EROSION

Dust can potentially influence global and regional climate by scattering and absorbing sunlight (Sokolik and Toon 1996) and affecting cloud properties (Wurzler et al. 2000), but the overall effect of mineral dusts in the atmosphere is likely to be small compared to other human impacts on the Earth's climate system (IPCC 2007). Desert dust is thought to play a major role in ocean fertilization and CO₂ uptake (Duce and Tindale 1991; Piketh et al. 2000; Jickells et al. 2005), terrestrial soil formation, and nutrient cycling (Swap et al. 1992; Wells et al. 1995; Chadwick et al. 1999), and public health (Leathers 1981; Griffin et al. 2001). In addition, desert dust deposited on downwind mountain snowpack has been shown to decrease the albedo of the snowpack, thus accelerating melt by as much as 20 days (Painter et al. 2007).

In arid regions, erosion has been shown to increase sediment delivery to large rivers (e.g., the Rio Grande), and can change the flow conditions of those rivers (Jepsen et al. 2003). Transport of eroded sediment to streams can change conditions in waterways, impacting water quality, riparian vegetation, and water fauna (Cowley 2006).

3.3.7 Indicators and Observing Systems

3.3.7.1 EXISTING OBSERVING SYSTEMS

A summary of arid land sites with inventory and monitoring programs is given in Table 3.2. Data from such sites will be important for helping track the consequences of climate change, but unfortunately, most sites do not have this as an explicit part of their mission. Furthermore, there is virtually no coordination among these sites with respect to the variables being monitored,

the processes being studied, the methodologies being used or the spatial and temporal scales over which change is occurring. Lack of coordination and standardization across these existing sites, programs and networks constitutes a missed opportunity.

Repeat photography is a valuable tool for documenting changes in vegetation and erosion. Hart and Laycock (1996) present a bibliography listing 175 publications using repeat photography and information on the ecosystems photographed, where they are located, number of photographs, and dates when the photographs were taken. More recent publications have added to this list (e.g., Webb 1996; McClaran 2003; Webb et al. 2007), and Hall (2002) has published a handbook of procedures. Time-series aerial photographs dating back to the 1930s and 1940s are also a useful source for quantifying landscape-scale changes in land cover (e.g., Archer 1996; Asner et al. 2003b; Bestelmeyer et al. 2006; Browning et al. 2008).

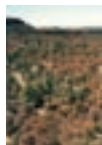
3.3.7.2 OBSERVING SYSTEMS REQUIRED FOR DETECTING CLIMATE CHANGE IMPACTS

While the deserts of North America have been the site of many important ecological studies, there have been relatively few long-term monitoring sites at an appropriate spatial representation that allow us the means to access changes in ecosystem structure and function in response to global change. Coordinated measurements of plant community composition in plots across the North American deserts would enhance our ability to detect change and relate that to aspects of climate. Several important data sets stand as benchmarks – the long-term photographic record at the Santa Rita Experimental Range, the long-term vegetation maps and livestock management records at the Jornada Experimental Range, the long-term perennial plant and winter annual plant studies at Tumamoc Hill, the long-term data collected from large-scale ecosystem manipulations at Portal Arizona, and the new Mojave Desert Climate Change Program. Greater spatial representation of such efforts is important in future assessment of change in these biomes.



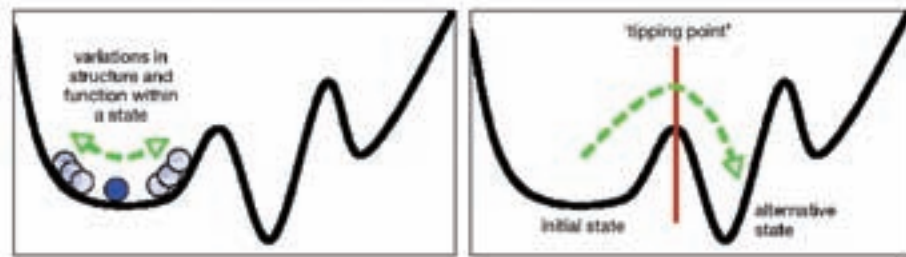
Table 3.2 Arid land sites with research and monitoring systems.

Monitoring system	Site	Location	Source
Free-Air CO ₂ Enrichment (FACE) Site	Nevada Desert FACE	Nevada Test Site, NV	http://www.unlv.edu/Climate_Change_Research
International Biome Project (IBP) Sites	Rock Valley Silverbell	Nevada Test Site, NV Arva Valley, AZ	archived at University of California, Los Angeles, CA archived at University of Arizona, Tucson, AZ
Land-Surface Flux Assessment Sites	Audubon Ranch USDA Agricultural Research Service Flux Tower Network Semi-arid Ecohydrology Array (SECA)	Ameriflux Sites	http://public.ornl.gov/ameriflux/ http://edintl.cr.usgs.gov/carbon_cycle/FluxesResearchActivities.html http://eebweb.arizona.edu/faculty/huxman/seca/
Research Sites (some with long-term data sets)	Desert Experimental Range V Bar-V Ranch Valles Caldera National Preserve Sweeny Granite Mountains Desert Research Center Boyd Deep Canyon Desert Research Center Great Basin Experimental Range Indio Mountains Research Station Jornada Experimental Range USA National Phenology Network The Portal Project Reynolds Creek Experimental Watershed San Juanquin Experimental Range Santa Rita Experimental Range UA Desert Laboratory at Tumamoc Hill Permanent Plots U.S. Sheep Experiment Station Walnut Gulch Experimental Watershed	Pine Valley, UT Rimrock, AZ Jemez Springs, NM Kelso, CA Palm Desert, CA Manti-LaSal National Forest, UT Van Horn, TX Jornada Basin, NM Portal, AZ Boise, ID O'Neals, CA Green Valley, AZ Tucson, AZ Snake River Plain, ID Tombstone, AZ Las Cruces, NM Albuquerque, NM Phoenix, AZ Tucson, AZ Salt Lake City, UT	http://www.fs.fed.us/rmrs/experimental-forests/desert-experimental-range/ http://ag.arizona.edu/aes/vbarv/ http://www.vallescaldera.gov/ http://nrs.ucop.edu/Sweeney-Granite.htm http://deepcanyon.ucnrs.org/ http://www.fs.fed.us/rm/prov/great_basin/great_basin.shtml http://www.utep.edu/indio/ http://usda-ar.s.nmsu.edu/ http://www.usanpn.org http://biology.unm.edu/jhbrown/Portal-LTREB/PortalFront.htm http://ars.usda.gov/main/site_main.htm?modecode=53-42-45-00 http://www.fs.fed.us/psw/ef/san_joaquin http://ag.arizona.edu/SRER/ http://www.pazctn.wr.usgs.gov/home.html http://ars.usda.gov/main/site_main.htm?modecode=53-64-00-00 http://ars.usda.gov/PandP/docs.htm?docid=10978&page=2 http://jornada-www.nmsu.edu/ http://sev.lternet.edu/ http://capiter.asu.edu/ http://www.sahra.arizona.edu/santarita/ http://www.neoninc.org http://science.nature.nps.gov
Long-Term Ecological Research (LTER) Sites	Jornada Basin Sevilleta Central Arizona-Phoenix Santa Rita Experimental Range Onaqui-Benmore		http://jornada-www.nmsu.edu/ http://sev.lternet.edu/ http://capiter.asu.edu/ http://www.sahra.arizona.edu/santarita/ http://www.neoninc.org
National Ecological Observatory Network (NEON)			
National Park Service Inventory & Monitoring Program	The NPS has recently initiated Inventory & Monitoring programs at many of its Parks and Monuments in arid lands		http://science.nature.nps.gov
TRENDS Project	Synthesis of long-term data from 44 research sites		http://fire.lternet.edu/Trends/
Rainfall Manipulations	USDA Agricultural Research Service Rainout Shelter Nevada Global Change Experiment	Burns, OR Nevada Test Site, NV	Svejcar et al. 2003 http://www.unlv.edu/Climate_Change_Research/

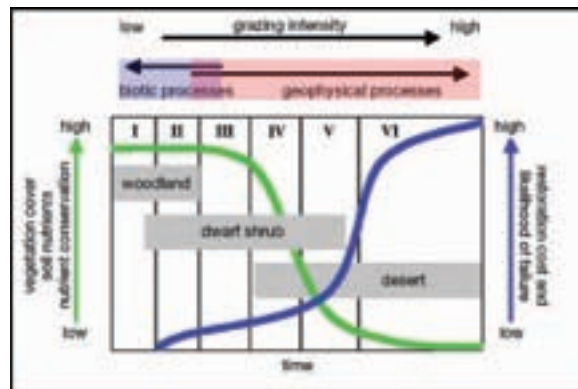


BOX 2: Ecosystem “tipping points.”

There is widespread recognition that ecosystems may exist in “alternate states” (e.g., perennial grassland state vs. annual grassland state; grassland state vs. shrubland state). Within a given state, ecosystems may tolerate a range of climate variability, stress and disturbance and exhibit fluctuation in structure (e.g., species composition) and function (e.g., rates of primary production and erosion). However, there may be “tipping points” that occur where certain levels of stress, resource availability, or disturbance are exceeded, causing the system transition to an alternate state (Archer and Stokes 2000). Thus, change in ecosystem structure and function in response to changes in stress levels or disturbance regimes may be gradual and linear up to a certain point(s), and then change dramatically and profoundly. Once in an alternate state, plant cover, composition and seedbanks, and soil physical properties, nutrient status and water holding capacity, etc. may have been altered to the point that it is difficult for the system to revert to its previous state even if the stresses or disturbance causing the change are relaxed.



In arid lands, threshold examples include shifts from grassland states to shrubland states (Archer 1989) and desertification (Schlesinger et al. 1990). It appears that these state-transitions occur as result of various combinations of vegetation-fire, soil, hydrology, animal, and climate feedbacks (e.g., Thurow 1991; Wainwright et al. 2002; Okin et al. 2006, D’Odorico et al. 2006).



While there is substantial observational evidence for these threshold phenomena, our quantitative understanding is limited and many questions remain:

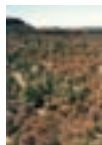
- How far, and under what circumstances, can an ecosystem be pushed before entering into an alternate state?
- What changes in ecosystem properties and feedbacks are involved in these state-transitions?
- What variables could be monitored to predict when a system is nearing a ‘tipping point’?
- How do climate factors influence the risk of exceeding state-transition thresholds?

Soil moisture is a key indicator and integrator of ecological and hydrological processes. However, as noted in the Water Resources chapter (Chapter 4), there is a dearth of information on the long-term patterns and trends in this important variable. Even on well-instrumented watersheds in arid lands (e.g., Lane and Kidwell 2003; NWRC 2007; SWRC 2007) soil moisture records are only erratically collected over time and are limited in their spatial coverage and depth. Thus, there is a pressing need for a distributed network of soil moisture sensors in arid lands that would be a component of a network of monitoring precipitation, evapotranspiration, and temperature. Ideally, such a network would also include collection of plant, soil and precipitation samples for determination of the stable isotope composition of C, O, and H. Such isotope data would provide important clues regarding when and where plants were obtaining soil moisture and how primary production and water use efficiency are being affected by environmental conditions (e.g., Boutton et al. 1999; Roden et al. 2000; Williams and Ehleringer 2000).

Effects of climate change will be most easily observed in relatively few arid land springs. Springs that dry periodically are relatively poor candidates, as long periods of record will be required to determine “baseline” conditions. Similarly, springs supported by large, regional aquifers are also poor candidates, as transmissivity is low and surface discharge is primarily ancient water (Miffllin 1968; Hershey and Mizell 1995; Thomas et al. 2001; Knochenmus et al. 2007). The USGS maintains quantitative historic Web-based records of surface water discharge from springs. These records could provide a “baseline” discharge, but the effect of climate change on such springs will not be evident for decades or much, much longer. Persistent springs fed by aquifers with moderate transmissivity are good candidates to assess effects of climate change. In the arid western United States, these geologically persistent springs are characterized by crenobiontic macroinvertebrates, including aquatic insects and springsnails. They occur on bajadas, at the base of mountains, and sometimes on valley floors (Taylor 1985; Hershler and Sada 2001; Polhemus and Polhemus 2001). While discharge from these springs fluctuates, they

have not dried. Transmissivity through aquifers supporting these springs is relatively high, hence their response to changes in precipitation will be relatively rapid and measurable (Plume and Carlton 1988; Thomas et al. 1996). An existing database, consisting of surveys of >2000 springs (mostly Great Basin and in the northwestern United States) over the past 15 years, includes hundreds of springs that would qualify as potential climate change monitoring sites (Sada and Hershler 2007).

Most land-surface exchange research has focused on forested systems. There is, however, a need for understanding the seasonal carbon dynamics, biomass, annual productivity, canopy structure, and water use in deserts (Asner et al. 2003a, b; Farid et al. 2006; Sims et al. 2006). Studies to date do not yet yield clear generalizations. For example, shifts from grass to shrub domination may show no net effects on evapotranspiration due to offsetting changes in radiant energy absorption and the evaporative fraction in the contrasting cover types (Kurc and Small 2004). However, this may depend upon the type of shrubs (Dugas et al. 1996). Although net changes in evapotranspiration may not occur with this land cover change, ecosystem water use efficiency may be significantly reduced (Emmerich 2007). Part of the challenge in predicting functional ecosystem dynamics in arid lands derives from our relatively poor understanding of non-equilibrium processes driven by highly episodic inputs of precipitation (Huxman et al. 2004). Part derives from the importance of the strong, two-way coupling between vegetation phenology and the water cycle, which is critical for predicting how climate variability influences surface hydrology, water resources, and ecological processes in water-limited landscapes (e.g., Scanlon et al. 2005). Shifts in phenology represent an integrated vegetation response to multiple environmental factors, and understanding of vegetation phenology is prerequisite to inter-annual studies and predictive modeling of land surface responses to climate change (White et al. 2005). Along these lines, the ability to detect ecosystem stress and impacts on vegetation structure will be requisite to understanding regional aspects of drought (Breshears et al. 2005) that result in substantial land use and land cover changes.



In regions where the eroded surfaces are connected to the regional hydraulic systems (i.e., not in closed basins), sediment delivery to streams and streambeds is an excellent indicator of integrated erosion in the catchment when coupled with stream gauging and precipitation data. USGS gauges are few and far between in arid lands and many have been or are being decommissioned due to lack of funds (as is also the case for watersheds on U.S. Forest Service lands). There is currently no integrated monitoring system in place for the measurement of bed-load, but the USGS National Water Information System does collect water quality data that could inform sediment loads. Unfortunately, there are very few sites in the arid United States that are monitored continuously. Additional arid region rivers could be instrumented and sampled to provide further monitoring of stream flow as well as water erosion. In closed basins, or the upland portion of open basins, the development and expansion of rills and gullies is the clearest indicator of water erosion. There is no system in place for the monitoring of these features (Ries and Marzloff 2003), but high-resolution remote sensing (~1-meter resolution) might be used to monitor the largest of these features.

The most important indicator of wind erosion is the dust that it produces. Because dust is transported long distances, even a sparse network of monitoring sites can identify dust outbreaks. For instance, Okin and Reheis (2002) have used meteorological data collected as part of the National Climatic Data Center's network of cooperative meteorological stations (the COOP network) to identify dust events and to correlate them to other meteorological variables. The expansion of this network to include observations in more locations, and especially at locations downwind of areas of concern, would be a significant improvement to monitoring wind in the arid portions of the United States. This existing observation network might also be integrated with data from NASA's Aerosol Robotic Network (AERONET) on aerosol optical depth and radar or lidar systems deployed throughout the region, but particularly near urban centers and airports. In addition, there are several remote sensing techniques that can be used to identify the spatial extent and timing of dust outbreaks (Chomette et al. 1999; Chavez et al.

2002; Miller 2003), though there is no system in place to integrate or track the evolution of dust sources through time.

Novel communities (with a composition unlike any found today) have occurred in the late-glacial past and will develop in the greenhouse world of the future (Williams and Jackson 2007). Most ecological models are at least partially parameterized from modern observations and so may fail to accurately predict ecological responses to novel climates occurring in conjunction with direct plant responses to elevated atmospheric CO₂ and nitrogen deposition. There is a need to test the robustness of ecological models to conditions outside modern experience.

3.4 FINDINGS AND CONCLUSIONS

3.4.1 Forests

Climate strongly influences forest productivity, species composition, and the frequency and magnitude of disturbances that impact forests. The effect of climate change on disturbances such as forest fire, insect outbreaks, storms, and severe drought will command public attention and place increasing demands on management resources. Other effects, such as increases in temperature, the length of the growth season, CO₂, and nitrogen deposition may be more incremental and subtle, but may have equally dramatic long-term effects.

Climate change has very likely increased the size and number of forest fires, insect outbreaks, and tree mortality in the interior west, the Southwest, and Alaska, and will continue to do so. An increased frequency of disturbance is at least as important to ecosystem function as incremental changes in temperature, precipitation, atmospheric CO₂, nitrogen deposition, and ozone pollution. Disturbances partially or completely change forest ecosystem structure and species composition, cause short-term productivity and carbon storage loss, allow better opportunities for invasive alien species to become established, and command more public and management attention and resources.



Rising CO₂ will very likely increase photosynthesis for forests, but the increased photosynthesis will likely only increase wood production in young forests on fertile soils. Where nutrients are not limiting, rising CO₂ increases photosynthesis and wood production. But on infertile soils the extra carbon from increased photosynthesis will be quickly respired. The response of older forests to CO₂ is uncertain, but possibly will be lower than the average of the studied younger forests.

Nitrogen deposition and warmer temperatures have very likely increased forest growth where water is not limiting and will continue to do so in the near future. Nitrogen deposition has likely increased forest growth rates over large areas, and interacts positively to enhance the forest growth response to increasing CO₂. These effects are expected to continue in the future as N deposition and rising CO₂ continue.

The combined effects of expected increased temperature, CO₂, nitrogen deposition, ozone, and forest disturbance on soil processes and soil carbon storage remain unclear. Soils hold an important, long-term store of carbon and nutrients, but change slowly. Long-term experiments are needed to identify the controlling processes to inform ecosystem models.

3.4.2 Arid Lands

Disturbance and land use on arid lands will control their response to climate change. Many plants and animals in arid ecosystems are near their physiological limits for tolerating temperature and water stress. Thus, even slight changes in stress will have significant consequences. Projected climate changes will increase the sensitivity of arid lands to disturbances such as grazing and fire. Invasion of non-native grasses will increase fire frequency. In the near-term, fire effects will trump climate effects on ecosystem structure and function. These factors cause important changes themselves, but the outcome of their interactions are difficult to predict in the context of increased concentrations of atmospheric CO₂ and nitrogen deposition. This is particularly so because these interactions represent novel combinations.

Higher temperatures, increased drought and more intense thunderstorms will very likely increase erosion and promote invasion of exotic grass species. Climate change will create physical conditions conducive to wildfire, and the proliferation of exotic grasses will provide fuel, thus causing fire frequencies to increase in a self-reinforcing fashion (Figure 3.17). In arid regions where ecosystems have not co-evolved with a fire cycle, the probability of loss of iconic, charismatic mega flora such as saguaro cacti and Joshua trees will be greatly increased.

Arid lands very likely do not offer a large capacity to serve as a “sink” for atmospheric CO₂ and will likely lose carbon as climate-induced disturbance increases. Climate-induced changes in vegetation cover and erosion will reduce the availability of nitrogen in dryland soils, which (after water) is an important control of primary productivity and carbon cycling.

Arid land river and riparian ecosystems will very likely be negatively impacted by decreased streamflow, increased water removal, and greater competition from non-native species. Dust deposition on alpine snow pack will accelerate the spring delivery of montane water sources and potentially contribute



Figure 3.17 Mojave Desert scrub near Las Vegas, NV (foreground); and area invaded by the exotic annual grass red brome (background) following a fire that carried from desert floor upslope into pinyon-juniper woodlands. Photo: T.E. Huxman.



to earlier seasonal drought conditions in lower stream reaches. Riparian ecosystems will likely contract, and in the remainder, aquatic ecosystems will be less tolerant of stress. The combination of increased droughts and floods, land use and land cover change, and human water demand will amplify these impacts and promote sedimentation.

Changes in temperature and precipitation will very likely decrease the cover of vegetation that protects the ground surface from wind and water erosion. More intense droughts and floods will accelerate fluvial erosion and higher frequencies of dust storms. Higher intensity rainfall will result in greater sheet erosion. All of these factors will periodically increase the sediment load in water and the atmosphere and decrease air and water quality.

3.4.3 Observing Systems for Forests and Arid Lands

Current observing systems are very likely inadequate to separate the effects of changes in climate from other effects. The major findings in the Land Resources Chapter relied on publications that used data assembled from diverse sources, generally for that specific study. In many cases, finding, standardizing, and assembling the data was the primary task in these studies. This was particularly the case for studies relating climate and disturbance. Findings on the effects of CO₂ and nitrogen deposition were largely based on short-term, small-scale experimental manipulations. Those for the interaction of climate and invasive species with vegetation, riparian ecosystems, and erosion generally came from long-term monitoring and survey studies. The NOAA weather network was invaluable for climate information, but most studies needed to extrapolate weather data to create uniform coverage across the United States. This was and remains a considerable task, and is particularly problematic in arid lands where precipitation is highly localized and varies significantly across short distances. Separating the effects of climate change from other impacts would require a broad network of indicators, coupled with a network of controlled experimental manipulations.

There is no coordinated national network for monitoring changes associated with disturbance and land cover and land use change. Because of the spatial heterogeneity of insect outbreaks and other disturbances, new sampling and monitoring approaches are needed to provide a comprehensive assessment of how climate is affecting the disturbance regime of forest ecosystems and changes in forest soils.

