environment

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Four Case Studies in the United States

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PEW CENTER ON Global CLIMATE CHANGE

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Foreword *Eileen Claussen, President, Pew Center on Global Climate Change*

In 2007, the science of climate change achieved an unfortunate milestone: the Intergovernmental Panel on Climate Change reached a consensus position that human-induced global warming is already causing physical and biological impacts worldwide. The most recent scientific work demonstrates that changes in the climate system are occurring in the patterns that scientists had predicted, but the observed changes are happening earlier and faster than expected—again, unfortunate. Although serious reductions in manmade greenhouse gas emissions must be undertaken to reduce the extent of future impacts, climate change is already here and some impacts are clearly unavoidable. It is imperative, therefore, that we take stock of current and projected impacts so that we may begin to prepare for a future unlike the past we have known.

The Pew Center has published a dozen previous reports on the environmental effects of climate change in various sectors across the United States. However, because climate impacts occur locally and can take many different forms in different places, *Regional Impacts of Climate Change: Four Case Studies in the United States* examines impacts of particular interest to different regions of the country. Although sections of the report examine different aspects of current and projected impacts, a look across the sections reveals common issues that decision makers and planners are likely to face in learning to cope with climate change.

Kristie Ebi and Gerald Meehl find that Midwestern cities are very likely to experience more frequent, longer, and hotter heatwaves. According to Dominique Bachelet and her coauthors, wildfires are likely to increase in the West, continuing a dramatic trend already in progress. Robert Twilley explains that Gulf Coast wetlands provide critical ecosystems services to humanity, but sustaining these already fragile ecosystems will be increasingly difficult in the face of climate change. Finally, Donald Boesch and his colleagues warn that the Chesapeake Bay may respond to climate change with more frequent and larger low-oxygen “dead zone” events that damage fisheries and diminish tourist appeal. These authors are leading thinkers and practitioners in their respective fields and provide authoritative views on what must be done to adapt to climate change and diminish the threats to our environmental support systems.

A key theme emerges from these four case studies: pre-existing problems caused by human activities are exacerbated by climate change, itself mostly a human-induced phenomenon. Fortunately, manmade problems are amenable to manmade solutions. Climate change cannot be stopped entirely, but it can be limited significantly through national and international action to reduce the amount of greenhouse gases emitted to the atmosphere over the next several decades and thereafter, thus limiting climate change impacts. Managing those impacts requires that we adapt other human activities so that crucial resources, such as Gulf Coast wetlands or public emergency systems, continue to function effectively. The papers in this volume offer insights into how we can adapt to a variety of major impacts that we can expect to face now and in decades to come.

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I. Introduction

The Pew Center on Global Climate Change has published many reports that address the impacts of climate change in a number of sectors and ecosystems across the United States, including agriculture, forests, coastal resources, water resources, and others. Results of previous studies in this series are summarized in a synthesis report (Smith, 2004).

But differences in climate, topography, land use, and infrastructure result in different climate change impacts at the regional and local scales. As a complement to earlier Pew Center reports focusing on the United States in general, this report presents four case studies of specific climate change impacts in different regions of the country:

- ***The Heat is On: Climate Change & Heatwaves in the Midwest*** by Kristie L. Ebi of ESS and Gerald A. Meehl of the National Center for Atmospheric Research;
- ***The Importance of Climate Change for Future Wildfire Scenarios in the Western United States*** by Dominique Bachelet of Oregon State University, and James M. Lenihan and Ronald P. Neilson of the U.S. Forest Service;
- ***Gulf Coast Wetland Sustainability in a Changing Climate*** by Robert R. Twilley of Louisiana State University; and
- ***Ramifications of Climate Change for Chesapeake Bay Hypoxia*** by Donald F. Boesch, Victoria J. Coles, David G. Kimmel and W. David Miller of the University of Maryland Center for Environmental Science.

Each case study focuses on a specific type of impact that is of particular concern for a region, but is not unique to that region. Each study also considers non-climatic factors, such as development and management practices, that are likely to interact with climate change. Consequently, cross-cutting themes emerge that are relevant to a wide array of regional and local climate change impacts beyond those examined here.

A. Individual Case Studies

Midwestern heatwaves. In coming decades heatwaves in the Midwest are likely to become more frequent, longer, and hotter than cities in the region have experienced in the past. This trend will result from a combination of general warming, which will raise temperatures more frequently above thresholds to which people have adapted, and more frequent and intense weather patterns that produce heatwaves. Studies projecting future mortality from heat foresee a substantial increase in health risks from heatwaves. Several factors contribute to increasing risk in Midwestern cities, including demographic shifts to more vulnerable populations and an infrastructure originally designed to withstand the less severe heat extremes of the past. The elderly living in inner cities are particularly vulnerable to stronger heatwaves; other groups, including children and the infirmed, are vulnerable as well. Adaptations of infrastructure and public health systems will be required to cope with increased heat stress in a warmer climate.

Fire in the West. Wildfire is a natural part of the western landscape and is very sensitive to climate variability. In recent decades, a trend toward earlier spring snowmelt and hotter, drier summers has already increased the number and duration of large wildfires in the West (Westerling et al., 2006). Although total annual precipitation may increase in the Northwest, climate projections generally foresee less precipitation throughout the West during the summer when risk of fire is greatest. In Alaska and Canada, warming has accelerated the reproduction and increased the winter survival and geographic range of insect pests that may make forests more vulnerable to fire by killing more trees (Berg et al., 2006; Volney and Fleming, 2000). Development in the West has placed more people and assets in fire-prone areas, increasing the need to suppress wildfires (McKinley and Johnson, 2007). Ironically, suppression increases the risk of catastrophic fire by allowing vegetation to build up, providing more fuel for fires when they ignite. Humans have also introduced invasive plant species that consume limited soil moisture and burn readily. Careful attention to development decisions and human-induced ecosystem stressors may help with adapting to increased risk from fire in the West resulting from climate change.

Gulf Coast wetlands. The coastline of the Gulf of Mexico offers a prototypical example of how human development patterns and climate change can interact to create high risks to human and natural systems. The combination of intense development in low-lying coastal areas, building levees along major rivers such as the Mississippi, high pollution levels, and extreme weather events, have degraded economically and culturally valuable coastal wetlands and made many human settlements in the Gulf region more vulnerable to rising seas and coastal storms. Accelerated sea-level rise and more intense hurricanes resulting from climate change would increase these risks. Therefore, plans to restore Gulf Coast wetlands and make them resilient to human activities and climate variability require careful consideration of how future climate change and human activities will degrade or enhance the natural processes that build and maintain coastal wetlands.

Chesapeake Bay hypoxia. Hypoxia (inadequate levels of oxygen that can lead to dead zones) in the Chesapeake Bay is another example of a natural phenomenon made substantially worse by human development and that could also be exacerbated by climate change. Hypoxia occurs when nutrient runoff from land stimulates biological oxygen demand, reducing oxygen levels in the Chesapeake Bay. This condition adversely affects the bay ecosystem, including its fisheries, and recreational opportunities in the bay. Development within the Chesapeake Bay watershed has resulted in runoff of nutrients from farms and settlements, increasing the incidence and intensity of hypoxia in the bay. Increased regional rainfall, which washes nutrients into the bay, and higher summer temperatures, which accelerate oxygen depletion, are likely to increase the incidence and intensity of hypoxia in the Chesapeake Bay. These changes could alter the current assessment of nutrient reductions needed to meet water quality objectives.

B. Cross-cutting Themes

The case studies provide but a few diverse examples of potential climate change impacts. Many other impacts will occur far and wide and will affect many sectors in all regions of the country and the world in different ways. However, several key themes emerge from these studies that are likely to cut across many distinct impacts in many different regions:

Impacts from climate change are already apparent. In all four of the case studies, there is growing evidence that climate change may already be increasing risks. To be sure, attribution of particular events either wholly or partially to climate change is a difficult process that can be controversial. But the literature linking climate change with the events discussed in this report is growing. Westerling et al. (2006) found that climate change over the 20^{th} century is a key factor explaining the increase in fires in the American West after accounting for human settlements and fire management. Extreme heat events in the United States are on the rise. DeGaetano and Allen (2002) found that minimum and maximum temperatures increased in the latter half of the 20th century, with particularly large increases in urban areas. Multi-day extreme heat events are also increasing. Global sea levels have been rising for centuries, but recently the rate of sea-level rise has accelerated (IPCC, 2007). This rise is likely contributing to some loss of wetlands in places such as the Gulf of Mexico and the Chesapeake Bay. Finally, there is growing evidence that the intensity and possibly the number of hurricanes in the Atlantic have increased in recent decades as a result of rising sea surface temperatures (Emanuel, 2005; Hoyos et al., 2006).

Multiple stressors exacerbate climate change impacts on natural systems. Enlarged pest populations, invasive species, and fire suppression all increase the vulnerability of ecosystems to fire. Nutrient inputs from farms and settlements increase the potential for hypoxia in coastal estuaries. Canals, flood-control structures, and pollution decrease the resilience of wetlands to rising sea levels and powerful storms. In many cases stressors that limit the ability of natural systems to resist stress from climate change are under human control, either directly (e.g., development) or indirectly (e.g., invasive species). Successful adaptation to climate change will likely require close attention to the many ways that human activities can be altered to increase ecosystem resilience to climate change.

Development patterns affect vulnerability to climate change impacts. In the four studies presented here, development and associated planning decisions and management practices exacerbate the impacts of climate change. The concentration of infrastructure and housing along with dense populations of the poor and elderly make inner cities more vulnerable to heatwaves than less developed areas. Increased population, building of impervious surfaces, and agriculture in the Chesapeake Bay watershed increase runoff of nutrients and risk of hypoxia. Development in low-lying coastal areas of the Gulf and Atlantic coasts places more people and property along the coastline and degrades buffering wetlands, putting people at greater risk from faster sea-level rise and more intense coastal storms. More development in wilderness areas in the West also increases the number of people and amount of property facing wildfire risk, as climate change increases the frequency and intensity of large fires. Adaptation to climate change will require closer attention to the implications of development patterns and land use decisions for climate change impacts.

There are likely to be increasing risks and costs from future climate change. The impacts of future climate change are likely to become greater as climate continues to change. There will likely be more loss of wetlands, higher risk to human life and property from stronger storms and hurricanes in the Gulf of Mexico and the Atlantic, more potential for hypoxia in the Chesapeake Bay and other coastal waters, more frequent and more intense heatwaves with greater risks to human health, and more frequent and intense wildfires. Many impacts not examined here would likely follow similar trends. Droughts and flash floods, for example, will likely increase in the future, presenting greater risks in areas that are already prone to such events (IPCC, 2007).

Climate change could have important consequences for the private insurance industry and for public disaster management and response. Many of the impacts discussed in these studies could affect lives and property, and therefore, are likely to affect insurance claims as well as government response to (and perhaps preparation for) disasters. For example, greater loss of life from more intense heatwaves and property damage from hurricanes and fires could well result in higher insurance payouts and insurance companies refusing coverage to more individuals and businesses. This effect would likely have further consequences for insurance rates, deductibles, and profits, which could affect other parts of the economy. Public disaster management and response will require increased resources and more funding in a future with more frequent and bigger fires, floods, and heatwaves.

Adaptation will be important in determining future vulnerability. The climate is already changing and affecting society and nature. Significant reductions in greenhouse gas emissions leading to lower atmospheric concentrations would reduce the magnitude of climate change and its impacts. Nonetheless, even with the most optimistic emissions reductions, there will still be substantial additional climate change. Thus, adaptation is an important component of a response to climate change. Reducing the level of pollution in the Chesapeake Bay will most likely reduce the risks of hypoxia. Adoption of heatwave early warning systems and other measures such as improving access to air conditioning have been shown to reduce risks from extreme heat events (Ebi et al., 2004). Wisely managing development patterns and vegetation can reduce the risks of fire (Platt et al., 2006). Evacuation planning, adoption of certain building designs, and limiting development in coastal areas can reduce risks from hurricanes. Furthermore, limits on certain types of development can also reduce destruction of wetlands, which are important for their ecosystem services.

C. Final Thoughts

Although climate change is a global problem, its impacts vary widely and are felt locally. With this report, the Pew Center on Global Climate Change endeavors to provide not just useful information about particular impacts in particular regions, but also a more general perspective on the types of challenges decision-makers everywhere will face in developing sustainable responses to varied climate impacts. Historically, risk management strategies have relied on the past as a guide to the future. But with global climate change, the future will no longer resemble the past. As illustrated by the four regional studies that follow, new strategies for developing resilience to climate variability and extreme weather events will be needed. Well-considered assumptions about regional climate change should be incorporated into development and management plans. Studying regions with different vulnerabilities will provide insights and methods for conducting assessments in other regions and sectors.

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References

Berg, E.E., J.D. Henry, C.L. Fastie, A.D. De Volderd, and S.M. Matsuoka. 2006. Spruce beetle outbreaks on the Kenai Peninsula, Alaska, and Kluane National Park and Reserve, Yukon Territory: Relationship to summer temperatures and regional differences in disturbance regimes. *Forest Ecology and Management* 227:219-232.

DeGaetano, A.T. and R.J. Allen. 2002. Trends in twentieth-century temperature extremes across the United States. *Journal of Climate* 15:3188-3205.

Ebi, K.L., T.J. Teisberg, L.S. Kalkstein, L. Robinson, and R.F. Weiher. 2004. Heat watch/warning systems save lives: estimated costs and benefits for Philadelphis 1995-1998. *Bulletin of the American Meteorological Society* 85:1067-1073.

Emanuel, K. 2005. Increasing destructiveness of tropical cyclones over the past 30 years. *Nature* 436:686-688.

Hoyos, C.D., P.A. Aguidelo, P.J. Webster, and J.A. Curry. 2006. Deconvolution of the factors contributing to the increase in global hurricane intensity. *Science* 312:94-97.

IPCC (Intergovernmental Panel on Climate Change). 2007. *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.* S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M.Tignor, and H.L. Miller (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

McKinley, J. and K. Johnson. 2007. On fringe of forests, homes and wildfires meet. *New York Times*, June 26.

Platt, R.V., T.T. Veblen, and R.L. Sherriff. 2006. Are wildfire mitigation and restoration of historic forest structure compatible? A spatial modeling assessment. *Annals of the Association of American Geographers* 96:455-470.

Smith, J.B. 2004. *A Synthesis of the Potential Impacts of Climate Change on the United States.* Pew Center on Global Climate Change, Arlington, VA.

Volney, W.J.A. and R.A. Fleming. 2000. Climate change and impacts of boreal forest insects. *Agriculture, Ecosystems & Environment* 82:283-294.

Westerling, A.L., H.G. Hidalgo, D.R. Cayan, and T.W. Swetnam. 2006. Warming and earlier spring increases western U.S. forest wildfire activity. *Science* 313:940-943.

II. The Heat is On: Climate Change & Heatwaves in the Midwest

Kristie L. Ebi, ESS *and Gerald A. Meehl,* NCAR

A. Introduction

Heatwaves affect human health through heat stress and exacerbate underlying conditions that can lead to an increase in mortality. Over the period 1979–1999, 8,015 deaths in the United States were recorded as being heat-related, 3,829 of which were attributed to weather conditions (Donoghue et al., 2003). Populations in the Midwest are particularly at increased risk for illness and death during heatwaves, as evidenced during events occurring in the 1980s and 1990s. A heatwave in July 1980 caused a 57 percent increase in mortality in St. Louis and a 64 percent increase in Kansas City (Jones et al., 1982). The 1995 Chicago heatwave is perhaps the most widely known; it caused an estimated 696 excess deaths (Whitman et al., 1997; Semenza et al., 1999). A heatwave of similar magnitude in 1999 resulted in 119 deaths in Chicago (Palecki et al., 2001).

B. Heat-Related Illnesses

Illnesses caused by exposure to high temperatures include heat cramps, fainting, heat exhaustion, heatstroke, and death (Kilbourne, 1997). Heat exhaustion is the most common response to prolonged exposure to high outdoor temperature; it is characterized by intense thirst, heavy sweating, dizziness, fatigue, fainting, nausea or vomiting, and headache. If unrecognized and untreated, heat exhaustion can progress to heatstroke, a severe illness with a rapid onset that can result in delirium, convulsions, coma, and death (Lugo-Amador et al., 2004). Heatstroke has a high fatality rate. Non-fatal heatstroke can lead to long-term illness. For example, about one-third of the patients admitted with heatstroke during the 1995 Chicago heatwave exhibited severe impairment and those who survived showed no improvement after one year (Dematte et al., 1998). In addition to heatstroke, many causes of death increase during heatwaves, particularly cardiovascular and respiratory disease (Kilbourne, 1997).

Except for heat cramps, heat-related illnesses are the result of varying degrees of the body's failure to regulate its internal temperature. To keep its internal temperature within healthy limits, the body's responses to hot weather include an increase in blood circulation (to move heat to the body surface) and an increase in perspiration. Heat loss is reduced when air temperature and/or humidity increase. To compensate, the body further increases circulation, but may be limited by its ability to increase heart rate and blood volume (because of loss of body fluids). For less fit subjects, heat illness can occur at low levels of activity, or even in the absence of exercise (Havenith et al., 1995).

C. Populations at Increased Risk

Although the risk of heat illness exists for the entire population, some factors increase the risk:

- ***Older and younger age.*** Older adults are more vulnerable to heatwaves because a natural part of the aging process is a decrease in the body's ability to control its internal temperature. Also, age correlates strongly with reduced fitness and increased illness, disability, and medication use. Most studies have found that heat-related mortality is highest in those over 65 years of age (Kovats and Koppe, 2005).

  Babies and infants also are at increased risk during a heatwave because they are at higher risk of dehydration, due to the relatively higher volume of fluid in their bodies compared with an adult (King et al., 1981). During 1979–2002 in the United States, 6 percent of the 4,780 deaths classified as heat-related occurred in children (LoVecchio et al., 2005).

- ***Use of certain drugs.*** Certain drugs interfere with the body's ability to cope with high temperatures (such as stimulants, beta-blockers, anticholinergics, digitalis, and barbiturates) (Koppe et al., 2004). For many individuals, a side effect of these drugs is that they may not be aware that high outdoor temperatures are making them ill and therefore may not take appropriate actions.

- ***Dehydration.*** Sufficient nonalcoholic fluid intake during a heatwave is a critical factor in reducing illness and death, particularly in those who are more vulnerable (Kilbourne, 1997).

Chronic dehydration, which is common among older adults, can increase susceptibility to heatwaves. The presence of multiple diseases and/or drug treatments also increases the risk of dehydration (Hodgkinson et al., 2003).

- ***Low fitness.*** A low level of fitness, due to reduced physical activity, increases vulnerability due to a reduction in the ability of the body to adjust to high outdoor temperatures (Havenith, 2001). Reduction in fitness can result in a vicious circle, as the increased strain experienced with activity may in itself result in further activity reduction, which may further decrease fitness.

- ***Excessive exertion.*** Excessive exertion during a heatwave is dangerous for everyone, regardless of age or fitness. Outdoor workers and those who maintain a vigorous exercise regimen during a heatwave are particularly at risk.

- ***Overweight.*** Being overweight increases the risk of heat-related illness and death. Fatty tissues are poorer conductors of heat than are other tissues in the body, thus providing an insulative barrier to heat flow (Koppe et al., 2004). Because a higher heart rate is needed to dissipate heat for an obese person, reduced fitness increases their risk further.

- ***Reduced adjustment to high outdoor temperatures.*** Although people physically adjust to the weather in the region in which they live, living in areas with relatively high daily temperature variability increases risk partly because adjustment is more difficult (Chestnut et al., 1998). Short-term adjustment to a change in outdoor temperature usually takes 3–12 days, but complete adjustment may take several years (Babayev, 1986; Frisancho, 1991). Short-term adjustment gradually disappears over a period of several weeks after a heatwave ends. Avoiding heat exposure leads to a reduction in the body's adjustment to higher temperatures, placing individuals at increased risk during a heatwave. This factor may add risk for people living in the Midwest where summer temperatures are highly variable and extreme heat occurs rarely.

- ***Urban populations.*** A number of studies suggest that urban populations suffer more illness and death during heatwaves (e.g., Smoyer et al., 2000; Sheridan, 2003). Urban populations may be more vulnerable because of higher underlying rates of cardio-respiratory disease. Also, a heatwave causes higher daytime and nighttime temperatures in cities than in rural areas because buildings and asphalt absorb more heat than do trees and plants. While rural areas cool after the sun goes down, this additional urban heat keeps temperatures high around the clock. Exposure to heat stress is higher in housing that is not designed to effectively insulate occupants from high outdoor temperatures.

- ***Lower socio-economic status.*** Studies have indicated that lower socio-economic status is a risk factor for heat-related mortality (Kovats and Koppe, 2005). For example, heatwave deaths in St. Louis in 1966 were the highest in inner city areas where population density was higher, open spaces were fewer, and where socio-economic status was lower than in surrounding areas (Henschel et al., 1969; Schuman, 1972). However, it is not clear whether the increased risk is due to differences in housing, neighborhood, access to air conditioning, or the underlying prevalence of chronic disease.

- ***Living alone.*** Studies designed to investigate why some people died during the 1995 and 1999 heatwaves in Chicago found that the strongest risk factor was living alone, particularly for those who did not leave home daily (Semenza et al., 1996 ; Naughton et al., 2002). O'Neill et al. (2003) found a nearly ten-fold increase in heat-related deaths for deaths occurring outside of a hospital compared with those in hospital, suggesting that people living alone without someone to check on them regularly are at particular risk. Similar risks were found in the 2003 heatwave in Paris and other regions of Europe, with many deaths of elderly adults occurring outside of a hospital (Kosatsky, 2005).

D. Projected Changes in the Frequency and Intensity of Heatwaves

There are a number of ways to define an extreme heatwave, most related to some kind of impact. We analyzed results from a global coupled climate model using two definitions of a heatwave.

The first definition comes from analysis of the Chicago heatwave of 1995. Mortality increased dramatically after three consecutive nights of very hot temperatures; in total, nearly 700 more people died than expected (Karl and Knight, 1997). Therefore, one definition of a heatwave is the warmest average minimum temperatures over three consecutive nights in a given year. This definition was used to quantify heatwave intensity for comparing observations and model results to determine how well the model simulates present-day events.

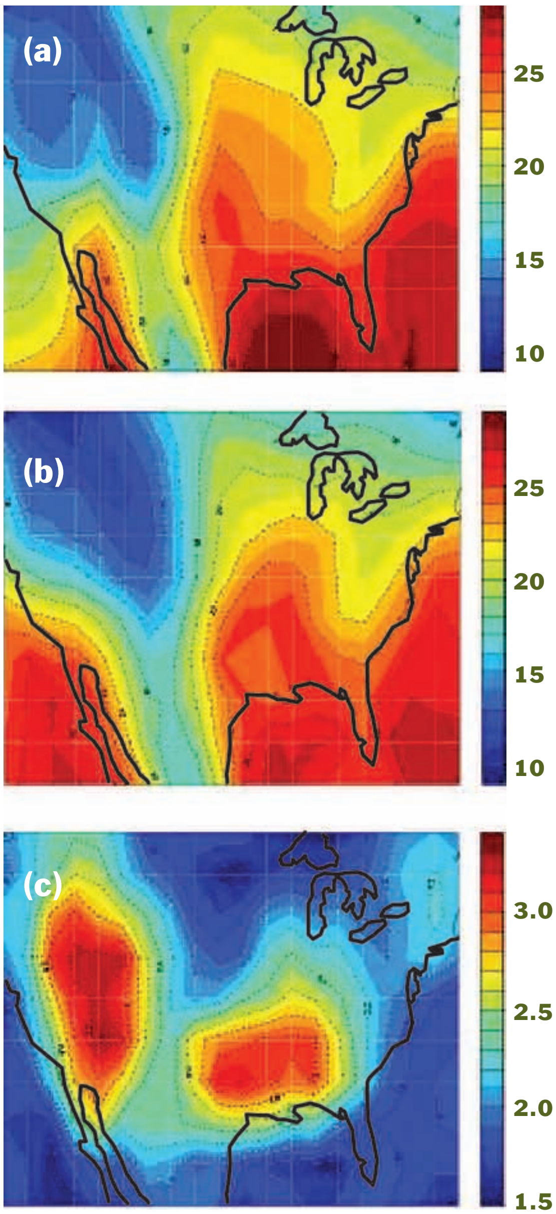
The National Center for Atmospheric Research/Department of Energy Parallel Climate Model (PCM) was used for the analysis. It is a global coupled climate model incorporating atmosphere, ocean, land surface, and sea ice components. Simulations of 20th century climate start in 1870, then run forward with time-evolving factors that affect the climate system, including natural (solar and volcanoes) and anthropogenic (greenhouse gases, sulfate aerosols, and tropospheric and stratospheric ozone) climate drivers (Meehl et al., 2004). The model was run four times from slightly different initial conditions, providing simulations for present-day heatwaves. Observations of past climate were analyzed in a similar fashion and compared to the model results (Figure 1a,b). The model did a good job of simulating the amplitude and the geographic pattern of observed heatwave intensity over North America. Both the model results and the observations show that heatwaves are most severe over the Eastern Seaboard, the southern and upper Midwest, and the southwestern United States. This model simulation of heatwave intensity is similar to a number of other models, as depicted by Tebaldi et al. (2006).

To project changes in future heatwaves, we used a "business as usual" future climate change scenario that assumed little policy intervention to mitigate greenhouse gas emissions in the 21st century (Dai et al., 2001). The model was run five times with slightly different initial conditions. We defined the "present-day" reference period as 1961–1990, and "future" as the time period from 2080–2099, and computed differences between these two periods. Future changes in heatwave intensity show a distinct geographical pattern (Figure 1c). Although differences were projected to be positive in all areas—indicative of the general increase in nighttime minimum temperatures—heatwave severity shows a greater increase in the western, upper midwestern, northeastern, and southern United States. Throughout much of the Midwest, the model projects future increases in nighttime temperatures of more than 2 °C (3.6 °F) during the worst heatwaves (Figure 1c).

Many of the areas most susceptible to heatwaves today (greatest heatwave severity in Figures 1a and 1b) are projected to experience the greatest increase in heatwave intensity in the future. But the model projects that other areas not currently as susceptible, such as northwestern North America, also could experience increased heatwave severity in the 21st century. These patterns of projected future heatwaves suggest different types of impacts. Regions already adapted to heat extremes (e.g., the southern, eastern, and southwestern parts of the United States) could experience negative effects

Figure 1

Heatwave **Severity**



Using the definition of heatwave severity as the average annual 3-day warmest nighttime minimum temperatures (ºC) from Karl and Knight (1997): a) observations for the "present-day" (1961–1990) from NCEP/NCAR reanalysis data for the United States; b) model simulation for the same period; c) model simulation for the "future" (2080–2099) minus model simulation for the "present-day" (1961–1990) for North America. Results in (c) indicate how much hotter on average heatwaves are projected to become by the end of the 21st century. (After Meehl and Tebaldi, 2004).

related to increased power generation to run the greater use of air conditioning. In areas such as the northwestern United States, where heatwaves are not severe at present and where use of air conditioning is less common, future increases in heatwave intensity could result in more heat-related illnesses and deaths. As more people install air conditioning, the health impacts could lessen, but the region may then face an increased strain on power generation. The pattern of projected future changes is therefore important for assessment of vulnerability and adaptation.

Heatwaves also can be defined to occur when weather conditions exceed a particular threshold; this definition identifies changes in heatwave frequency and duration. We examined model outputs for grid cells near three Midwestern cities (Chicago, Cincinnati, and St. Louis), to illustrate future projections of heatwave characteristics using three criteria to define heatwaves: (1) maximum temperature exceeding the 97.5th percentile (i.e., an event happening one out of 25 times) for at least three days, (2) average minimum temperature above the 97.5th percentile for at least three days, and (3) maximum temperature above the 81st percentile for the entire period (Huth et al., 2000).

All model scenarios projected future increases in the average frequency and duration of heatwaves (Figure 2). For all three cities, the observed frequency fell within the range of the present-day (1961–1990) frequency simulated by the model, indicating that the model mimics the observed climate effectively. In contrast, all of the observed values for heatwave frequency fell outside of and below the frequency range simulated by the model for the future (2080–2099), thus projecting future increases in heatwave frequency for all three cities. The model results were similar for average heatwave duration, with the exception that the observed value for Cincinnati fell slightly below the range of the model-simulated present-day values. This one inconsistency suggests the model may overestimate the absolute duration of heatwaves near Cincinnati, but it does not challenge the relative increase in duration projected for the future.

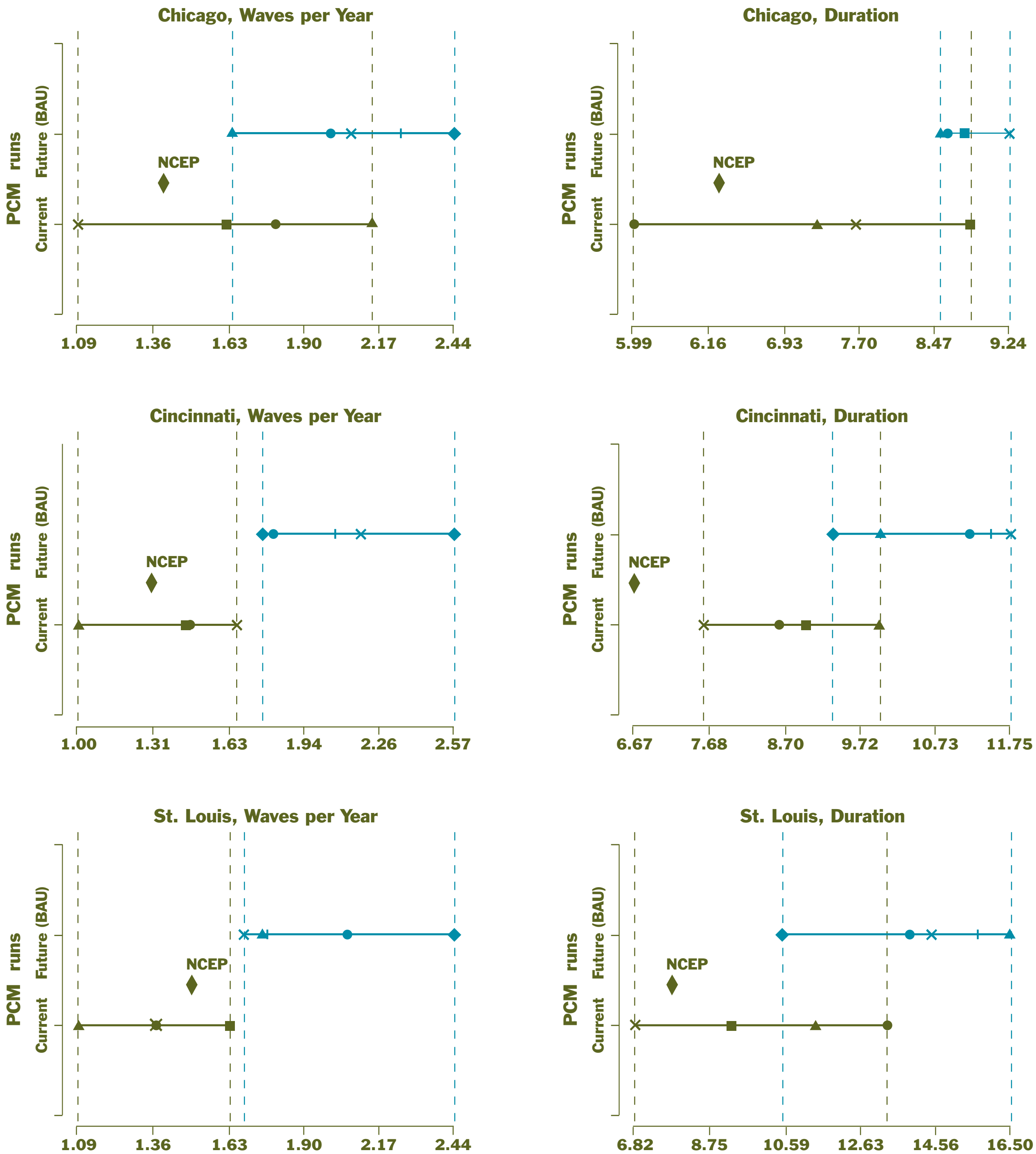
The model projected an increase in the average heatwave frequency of about 24 percent for Chicago—from 1.7 to 2.1 heatwaves per year; 50 percent for Cincinnati—from 1.4 to 2.1 heatwaves per year; and 36 percent for St. Louis—from 1.4 to 1.9 heatwaves per year. The average duration of heatwaves was projected to increase by 21 percent for Chicago—from 7.3 to 8.8 days; by 22 percent for Cincinnati—from 8.8 to 10.7 days; and by 38 percent for St. Louis—from 10.3 to 14.2 days.

These analyses show that the model simulated the present-day number and duration of heatwaves within or near the range of the observations, and that the range of projections lies well beyond the present-day observations (i.e. more and longer-lived heatwaves). On average, the frequency of heatwaves for all three cities increased by 36 percent and the duration of individual heatwaves increased by 27 percent. Combining these two effects implies an overall increase of about 70 percent in the annual number of heatwave days for the Midwestern region by the late 21st century. Moreover, as shown in Figure 1, these extreme days will be hotter on average than at present.

Heatwaves generally are associated with semi-stationary "domes of high pressure" that produce clear skies, light winds, warm air, and prolonged hot conditions at the surface (Kunkel et al., 1996; Palecki et al., 2001). These conditions were present during the 1995 Chicago and 2003 Paris heatwaves (Meehl and Tebaldi, 2004), with significant domes of high pressure over Lake Michigan and over northern France for the duration of the heatwaves. The model projections simulated comparable patterns during heatwaves. One reason for the intensification of future heatwaves is that the high pressure associated with a given heatwave is projected to be amplified due to anthropogenic emissions

Figure 2

Present and Future **Frequency and Duration** of Heatwaves



Based on the threshold definition of heatwaves from Huth et al. (2000), this figure shows the observed and model-simulated average number of heatwaves per year and the average duration of heatwaves for the “present-day” (1961–1990) and the “future” (2080–2099) climate near Chicago, Cincinnati, and St. Louis. In each panel, the green diamond marked “NCEP” indicates the observed value for the “present-day” base period of 1961–1990, computed from NCEP/NCAR reanalysis data. The green segment shows the range of values obtained from the four model runs for the “present-day” (1961–1990) simulation and the blue segment shows the range of values obtained from the five model runs for the “future” (2080–2099) simulation. The values for individual model runs are marked by individual symbols along the green and blue segments. Dashed vertical lines mark the endpoints of the simulated ranges for the “present-day” (green) and “future” (blue) and facilitate comparisons of the simulated ranges and observed values. (Results for Chicago are from Meehl and Tebaldi, 2004; results for Cincinnati and St. Louis are unique to this study).

of greenhouse gases (Meehl and Tebaldi, 2004). The average future climate shows higher pressure over the upper Midwest, which is directly associated with more intense heatwaves.

This pattern of increased high-pressure events results in an increase in summer nighttime minimum and daytime maximum temperatures (Meehl and Tebaldi, 2004), consistent with increased variability of temperature extremes in addition to a shift in the average (Schar et al., 2004). Thus, such events as the 2003 Paris heatwave could become common in the future climate (Stott et al., 2004). A study by Tebaldi et al. (2006) shows that such results are typical, with all models indicating an increase in heatwave intensity in a future warmer climate.

E. Projected Health Impacts of Future Heatwaves

Projections of an increase in the frequency and intensity of heatwaves are insufficient to estimate future illness and death. Projections of the health impacts of future heatwaves need to incorporate a variety of factors, including the degree to which the population is acclimatized to higher temperatures, the characteristics of the vulnerable population, and the extent to which effective adaptation strategies and measures have been implemented. These factors need to be estimated for the geographic region and time scale of interest, acknowledging that estimates of these factors become more uncertain for longer time frames.

A few studies have projected that health impacts of heatwaves could increase under various climate change scenarios (Kalkstein and Greene, 1997; Keatinge et al., 2002; Dessai, 2003; McMichael et al., 2003; Hayhoe et al., 2004). When the model includes assumptions about adjustment to higher outdoor temperatures and adaptation measures, estimates of heat-related deaths attributable to climate change are reduced but not eliminated. Because of incomplete understanding of how future populations might respond to heatwaves, these studies could either over- or under-estimate possible health impacts. Also, studies have not included changes in the frequency or intensity of severe heatwaves, such as occurred in 2003 in Europe and as projected to occur in this study (Figures 1 and 2).

Hayhoe et al. (2004) projected the implications of low and high greenhouse gas emission scenarios (Nakicenovic et al., 2000) for extreme heat and heat-related mortality in California. Taking some acclimatization into account (but no change in the prevalence of air conditioning), assuming a linear increase in heat-related mortality with increasing temperature, and assuming no change in the population, expected heat-related deaths in Los Angles were projected to increase (from a baseline of about 165 excess deaths annually) two- to three-fold under a low emission scenario and five- to seven-fold under a high emission scenario by 2070–2099.

Trends that are likely to increase vulnerability to heat-related morbidity (prevalence of disease) and mortality in the next few decades include an increased number of elderly people (Hobbs and Damon, 1996), increased urbanization, and increased frequency and intensity of heatwaves.

Overall, in the Midwest, the health burden of heatwaves is likely to be relatively small for moderate heatwaves, because most deaths will occur in persons who are already ill and because implementation of effective heatwave early warning systems has increased. Moreover, the prevalence of air conditioning in cities in the Midwest is high and can reasonably to be expected to increase further, which should further reduce population vulnerability. Extreme heatwaves present greater risk and are likely to become more frequent if manmade greenhouse gas emissions continue to rise unabated. Greater adaptation measures will be needed to manage these risks.

F. Adaptation Options

Short-term adaptation options include development of effective heatwave early warning and response plans, increasing appropriate use of air conditioning, and better education. Heatwave early warning systems can be an effective approach to reducing the illnesses and deaths associated with heatwaves (Palecki et al., 2001; Weisskopf et al., 2002; Ebi et al., 2004). Because heatstroke has a fast onset and a poor survival rate, prevention efforts must begin when oppressive weather is forecast, rather than when it arrives. The principal components of an early warning system include identification and forecasting of the event (including consistent, standardized weather criteria guiding the activation and deactivation of warnings), prediction of possible health outcomes that could occur, effective and timely response plans that target high-risk populations, and ongoing evaluation and revision of the system and its components (Ebi and Schmier, 2005; Bernard and McGeehin, 2004). Longer-term adaptation options focus on infrastructure changes, such as establishing building codes designed to reduce urban heat islands.

Considerably more education is needed of the public and of the responsible agencies about the dangers associated with heatwaves and about the appropriate responses. For example, in the review of heatwave response plans in the United States, five of the plans reported fan distribution programs, despite evidence that fans may increase heat stress if used improperly (Bernard and McGeehin, 2004). In addition to general messages detailing ways to lower body temperature to prevent the onset of heat stress (including drinking more fluids, going to an air-conditioned place, wearing light-colored and loose-fitting clothing, and limiting outdoor activity; CDC, 2007), messages should be targeted to vulnerable groups—such as those with low incomes, the elderly, the disabled, children, and ethnic minorities (Ebi and Schmier, 2005). A review of 18 U.S. heatwave response plans found that although people with mental or chronic illnesses and the homeless constitute a significant proportion of the victims in recent heatwaves, only one plan emphasized outreach to disabled persons, and only two addressed the shelter and water needs of the homeless (Bernard and McGeehin, 2004).

Air conditioning is frequently promoted as a key adaptation option to reduce heat-related illness and death. There is evidence that increased air conditioning coverage in the United States has reduced vulnerability (Davis et al., 2003). More than 80 percent of homes in the United States have air conditioning (U.S. Census Bureau, 2002). On the other hand, centralized cooling centers have not proved effective in reaching the most at-risk seniors (Naughton et al., 2002; Palecki et al., 2001). Hence, increased prevalence of air conditioning alone does not necessarily address the needs of those at greatest risk.

A key constraint to reducing the health impacts of heatwaves is that a normal part of the aging process is a reduction in the ability to thermoregulate. Many of the elderly at increased risk during a heatwave have underlying diseases that cause them to feel ill on most days. During a heatwave, feeling hot in addition to feeling ill is insufficient motivation for many of the elderly to take actions to reduce body temperature, such as visiting a cooling center, opening windows, drinking additional water, changing into more appropriate clothing, or turning on an air conditioner.

Better understanding of how to motivate appropriate behavior during a heatwave will reduce current and future vulnerability, no matter what the future climate brings.

References

Babayev, A.B. 1986. Some aspects of man's acclimatization to hot climates. In *Proceedings of the Symposium on Climate and Human Health*. World Meteorological Organization, Leningrad, Russia, Volume 2.

Bernard, S.M. and M.A. McGeehin. 2004. Municipal heatwave response plans. *American Journal of Public Health* 94:1520-1521.

CDC (Centers for Disease Control and Prevention). 2007. *Extreme heat: a prevention guide to promote your personal health and safety.* Centers for Disease Control and Prevention, Atlanta, GA. Available online at http://www.bt.cdc.gov/disasters/extremeheat/heat_guide.asp.

Chestnut, L.G., W.S. Breffle, J.B. Smith, and L.S. Kalkstein. 1998. Analysis of differences in hot-weather-related mortality across 44 U.S. metropolitan areas. *Environmental Science and Policy* 1:59-70.

Dai, A., T.M.L. Wigley, G.A. Meehl, and W.M. Washington. 2001. Effects of stabilizing atmospheric CO_2 on global climate in the next two centuries. *Geophysical Research Letters* 28:4511-4514.

Davis, R.E., P.C. Knappenberger, P.J. Michaels, and W.M. Novicoff. 2003. Changing heat-related mortality in the United States. *Environmental Health Perspectives* 111:1712-1718.

Dematte, J.E., K. O'Mara, J. Buescher, C.G. Whitney, S. Forsythe, T. McNamee, R.B. Adiga, and M. Ndukwu. 1998. Near-fatal heat stroke during the 1995 heatwave in Chicago. *Annals of Internal Medicine* 129:173-181.

Dessai, S. 2003. Heat stress and mortality in Lisbon Part II: An assessment of the potential impacts of climate change. *International Journal of Biometeorology* 48:37-44.

Donaldson, G.C., W.R. Keating, R.S. Kovats, and A. McMichael. 2002. Heat- and cold-related mortality and morbidity and climate change. In *Health Effects of Climate Change in the UK*. Department of Health, London, United Kingdom.

Donoghue, E.R., M. Nelson, G. Rudis, R.I. Sabogal, J.T. Watson, G. Huhn, and G. Luber. 2003. Heat-related deaths—Chicago, Illinois, 1996–2001, and the United States, 1979–1999. *Mortality and Morbidity Weekly Report* 52:610-613.

Ebi, K.L. and J.K. Schmier. 2005. A stitch in time: improving public health early warning systems for extreme weather events. *Epidemiologic Review* 27:115-21.

Ebi, K.L., T.J. Teisberg, L.S. Kalkstein, L. Robinson, and R.F. Weiher. 2004. Heat watch/warning systems save lives: estimated costs and benefits for Philadelphia 1995–1998. *Bulletin of the American Meteorological Society* 85:1067-1073.

Frisancho, R. 1991. *Human Adaptation: A Functional Interpretation*. University of Michigan Press, Ann Arbor, MI.

Havenith, G. 2001. Individualized model of human thermoregulation for the simulation of heat stress response. *Journal of Applied Physiology* 90:1943-1954.

Havenith, G., V.G.M. Luttikholt, and T.G.M. Vrijkotte. 1995. The relative influence of body characteristics on humid heat stress response. *European Journal of Applied Physiology* 70:270-279.

Hayhoe, K., D. Cayan, C.B. Field, P.C. Frumhoff, E.P. Maurer, N.L. Miller, S.C. Mose, S.H. Schneider, K.N. Cahill, E.E. Cleland, L. Dale, R. Drapek, R.M. Hanemann, L.S. Kalkstein, J. Lenihan, C.K. Lunch, R.P. Neilson, S.C. Sheridan, and J.H. Verville. 2004. Emission pathways, climate change, and impacts on California. *Proceedings of the National Academy of Sciences USA* 101:12422-12427.

Henschel, A., L.L. Burton, L. Margolies, and J.E. Smith. 1969. An analysis of the heat deaths in St. Louis during July 1966. *American Journal of Public Health* 59:2232-2242.

Hobbs, F.B. and B.L. Damon. 1996. *65+ in the United States.* Bureau of the Census, U.S. Department of Commerce, Washington, DC.

Hodgkinson, R., D. Evans, and J. Wood. 2003. Maintaining oral hydration in older adults: a systematic review. *International Journal of Nursing Practice* 9:19-28.

Jones, T.S., A.P. Liang, E.M. Kilbourne, M.R. Griffin, P.A. Patriarca, S.G.F. Wassilak, R.J. Mullan, R.F. Herrick, H.D. Donnell, D. Choi, and S.B. Thacker. 1982. Morbidity and mortality associated with the July 1980 heat wave in St. Louis and Kansas City, MO. *Journal of the American Medical Association* 247:3327-3331.

Kalkstein, L.S. and J.D. Greene. 1997. Evaluation of climate/mortality relationships in large US cities and the possible impacts of climate change. *Environmental Health Perspectives* 105:84-93.

Karl, T.R. and R.W. Knight. 1997. The 1995 Chicago heatwave: How likely is a recurrence? *Bulletin of the American Meteorological Society* 78:1107-1119.

Kilbourne, E.M. 1997. Heat waves and hot environments. In *The Public Health Consequences of Disasters*. E.K. Noji (ed.). Oxford University Press, New York, NY.

King, K., K. Negus, and J.C. Vance. 1981. Heat stress in motor vehicles: a problem in infancy. *Pediatrics* 68:579-582.

Koppe, C., G. Jendritzky, R.S. Kovats, and B. Menne. 2004. *Heat waves: risks and responses*. World Health Organization, Copenhagen, Denmark.

Kosatsky, T. 2005. The 2003 European Heat Waves. *Eurosurveillence* 10:148-149.

Kovats, R.S. and C. Koppe. 2005. Heat waves: past and future impacts. In *Integration of Public Health with Adaptation to Climate Change: Lessons Learned and New Directions.* K.L. Ebi, J.B. Smith and I. Burton (eds.). Taylor & Francis, London, United Kingdom, pp. 136-160.

Huth, R., J. Kysely, and L. Pokorna. 2000. A GCM simulation of heat waves, dry spells, and their relationships to circulation. *Climatic Change* 46:29-60.

Kunkel, K.E., S.A. Changnon, B.C. Reinke, and R.W. Arritt. 1996. The July 1995 heat wave in the Midwest: a climatic perspective and critical weather factors. *Bulletin of the American Meteorological Society* 77:1507-1518.

LoVecchio, F., J.S. Stapczynski, J. Hill, and A.F. Haffer. 2005. Heat-related mortality—Arizona, 1993–2002, and the United States, 1979–2002. *Mortality and Morbidity Weekly Report* 54:628-630.

Lugo-Amador, N.M., T. Rothenhaus, and P. Moyer. 2004. Heat-related illness. *Emergency Medicine Clinic of North America* 22:315-327.

McMichael, A., R. Woodruff, P. Whetton, K. Hennessy, N. Nicholls, S. Hales, A. Woodward, and T. Kjellstrom. 2003. *Human Health and Climate Change in Oceania: Risk Assessment 2002*. Commonwealth Department of Health, Canberra, Australia.

Meehl, G.A. and C. Tebaldi. 2004. More intense, more frequent, and longer lasting heatwaves in the 21st century. *Science* 305:994-997.

Meehl, G.A., W.M. Washington, C. Ammann, J.M. Arblaster, T.M.L. Wigley, and C. Tebaldi. 2004. Combinations of natural and anthropogenic forcings and 20th century climate. *Journal of Climate* 17:3721-3727.

Nakicenovic, N. and R. Swart (eds.) 2000. *Special Report on Emissions Scenarios: A Special Report of Working Group III of the Intergovernmental Panel on Climate Change.* Cambridge University Press, Cambridge, United Kingdom.

Naughton, M.P., A. Henderson, M.C. Mirabelli, R. Kaiser, J.L. Wilhem, S.M. Kieszak, C.H. Rubin, and M.A. McGeehin. 2002. Heat related mortality during a 1999 heatwave in Chicago. *American Journal of Preventive Medicine* 22:221-227.

O'Neill, M.S., A. Zanobetti, and J. Schwartz. 2003. Modifiers of the temperature and mortality association in seven U.S. cities. *American Journal of Epidemiology* 157:627-634.

Palecki, M.A., S.A. Changnon, and K.E. Kunkel. 2001. The nature and impacts of the July 1999 heat wave in the midwestern United States: learning from the lessons of 1995. *Bulletin of the American Meteorological Society* 82:1353-1367.

Schar, C., P.L. Vidale, D. Luthi, C. Frei, C. Haberli, M.A. Liniger, and C. Appenzeller. 2004. The role of increasing temperature variability in European summer heatwaves. *Nature* 427:332-336.

Schuman, S.H. 1972. Patterns of urban heatwave deaths and implications for prevention: data from New York and St. Louis during July 1996. *Environmental Research* 5:59-75.

Semenza, J.C., J.E. McCullough, W.D. Flanders, M.A. McGeehin, and J.R. Lumpkin. 1999. Excess hospital admissions during the July 1995 heat wave in Chicago. *American Journal of Preventive Medicine* 16:269-277.

Semenza, J.C., C.H. Rubin, K.H. Falter, J.D. Selanikio, W.D. Flanders, H.L. Howe, and J.L. Wilhelm. 1996. Heat related deaths during the July 1995 heat wave in Chicago. *New England Journal of Medicine 335:84-90.*

Sheridan, S. 2003. Heat, mortality, and level of urbanisation. *Climate Research* 24:255-265.

Smoyer, K.E., D.G. Rainham, and J.N. Hewko. 2000. Heat stress-related mortality in five cities in Southern Ontario: 1980–1996. *International Journal of Biometeorology* 44:190-197.

Stott, P.A., D.A. Stone, and M.R. Allen. 2004. Human contribution to the European heatwave of 2003. *Nature* 432: 610-614.

Tebaldi, C., J.M. Arblaster, K. Hayhoe, and G.A. Meehl. 2006. Going to the extremes: an intercomparison of model-simulated historical and future changes in extreme events. *Climatic Change* 79:185-211.

U.S. Census Bureau. 2002. *American Housing Survey for the United States: 2001.* Current housing reports, series H150/01. U.S. Government Printing Office, Washington, DC.

Weisskopf, M.G., H.A. Anderson, S. Foldy, L.P. Hanrahan, K. Blair, T.J. Torok, and P.D. Rumm. 2002. Heat wave morbidity and mortality, Milwaukee, Wis, 1999 vs 1995: an improved response? *American Journal of Public Health* 92:830-833.

Whitman, S., G. Good, E.R. Donoghue, N. Benbow, W. Shou, and S. Mou. 1997. Mortality in Chicago attributed to the July 1995 heat wave. *American Journal of Public Health* 87:1515-1518.

III. The Importance of Climate Change for Future Wildfire Scenarios in the Western United States

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A. Introduction

Fire has always been an integral part of the ecology of the western United States and has contributed to the diversity of its ecosystems, influencing their carbon and nutrient cycling. Will a warmer future climate increase fires in western forests? If so, how will this change affect the ecosystems and the carbon cycle in the western United States?

To answer these questions, we should first examine the past (Veblen, 2003). During droughts in the 19th century and earlier, fires were as severe as they have been in the last decade. Effective fire suppression since the 1950s has reduced the extent of wildfires in the United States by a factor of eight since the beginning of the 20th century, according to model simulations (Lenihan et al., forthcoming). However, in the past decade, while the number of fires continued to decrease, the size of the fires increased. Recent large fires have captured the headlines in western states—Arizona (Rodeo-Chediski fire in 2000 more than 468,638 acres), Oregon (Biscuit fire in 2002 almost 500,000 acres), and Alaska (6.38 million acres burned in 2004).

Fire suppression leading to the accumulation of vegetation, which fuels fires, was identified as the major culprit leading to larger wildfires, and forest thinning and logging to reduce available fuel was touted as a reliable mitigation option to reduce fires in the West. However, the direct role of humans in causing the current increase in fire activity may have been greatly overstated, as forests in which fire has not been suppressed have seen increases in fire comparable to managed forests. Since 1986, the combination of earlier snowmelt due to warmer springs (resulting in a longer fire season), and warmer summers (resulting in lower soil moisture) have been the major contributors to the increase in fire activity in managed and unmanaged forests, alike (Westerling et al., 2006). Western forests are responding clearly to a strong climate signal rather than simply to mismanagement.

What can we expect from western U.S. forest fires in the 21st century? Projections of future climate change from general circulation models simulate significant increases in temperature across the western United States during the 21st century. Projections of precipitation are more variable, but they generally suggest drier summer conditions in the West (Running, 2006). In fact, a transition to persistently drier conditions has already begun in the Southwest, and mountain snowpack has already declined throughout the West (Mote et al., 2005; Seager et al., 2007). These projections, combined with an increase in population density and the continued expansion of the urban–wildland interface, indicate that fires will continue to be a concern in the West.

An important question now is whether fires will increase across all western forests in the future, or whether more frequent droughts will decrease fuel production and ultimately starve future fires in certain areas. Another question is whether managers can target more sensitive areas for intensive control? This chapter summarizes projections for the 21st century from vegetation models that integrate knowledge of past fire occurrence to simulate changes in fire patterns and their effect on carbon sequestration in western states.

B. Terminology

Some specialized terminology is required to discuss fire trends and their implications for natural and human systems.

- ***Fuel*** is potentially combustible material, and *fuel load* is the amount of fuel available for ignition. Wind, topography, and fire history are among the many local factors that affect fuel load and, in turn, fire frequency and intensity. *Surface fuels* include grasses, dead leaves, and needles; *ladder fuels* include dead branches, mosses, and lichens that carry the fire up into the canopy. *Fine fuels* (such as dead leaves) respond quickly (within hours) to changes in air moisture, while *coarse fuels* (such as tree trunks) respond much more slowly (weeks).

- ***Fire intensity*** describes the amount of energy released by a fire, usually characterized by the maximum temperature of a fire and the height of the flames. It depends on the type and availability of fuel and on weather conditions.

- ***Fire severity*** refers to the degree to which a site has been altered by fire—for example, the amount of fuel burned and the degree of tree mortality. Severity depends on both fire intensity and duration. High-severity or *stand-replacing* fires typically kill trees by burning the tree tops or through very hot surface fires.

- ***Fire return interval*** is the average number of years between consecutive fires at a given location.

- ***Fire frequency*** is the number of fires during a given unit of time and depends on the type of ecosystem, the type of weather and its duration, and the source of ignition.

- ***Fire regime*** refers to fire patterns over long periods of time and their effects on ecosystems. Fire regimes are a function of the frequency of fire occurrence, fire intensity and the amount of fuel consumed. Frequent fires tend to maintain fire-tolerant species such as ponderosa pine, while sporadic fires promote shade-tolerant conifer species such as Douglas firs. Low-severity fires support fire-resistant (thick-barked) vegetation, while high-severity areas are usually dominated by faster-growing, thin-barked trees.

Low-severity fire regimes are characterized by frequent small fires with low intensity. They often occur in lightning-prone areas that are fuel limited, such as most Southwest low-elevation ponderosa pine forests with abundant fine fuels, including grasses and long needles that dry easily. In these forests, average fire return intervals have historically ranged from 4 to 36 years, and fire suppression has been effective in reducing the number of fires, leading to a build-up of fuel (Schoennagel et al., 2004). Due to their accessibility, these forests have also been extensively managed for timber production and livestock grazing (thereby reducing the amount of grass fuel), which has altered tree density and forest composition (Smith and Fischer, 1997). Management activities have often caused widely spaced old-growth trees to be replaced by stands of dense, small-diameter trees that tend to fuel high-intensity fire when fires do occur in these dry ecosystems.

Moderate or mixed-severity fire regimes consist of a combination of frequent low-intensity surface fires and infrequent stand-replacing fires of intermediate size. Middle and lower mountain forests are more likely affected by both mixed- and low-severity fire regimes. Both climate and fuels vary considerably in these forests and fire return intervals vary accordingly, ranging from 25 years to more than 250 years. Forest management activities (primarily logging) and fire exclusion have also affected fire regimes, particularly on sites that once supported open woodlands (Schoennagel et al., 2004).

High-severity fires, typically infrequent fires of high intensity and large size (Agee, 1998), generally affect high-elevation dense forests, where ladder fuels abound. At these cold and wet elevations, fire occurrence is mostly limited by fuel moisture. Fires are often caused by weather events such as droughts accompanied by high winds (Agee, 1997). They typically burn infrequently (50–300 years), although often at a much higher intensity compared with low-elevation drier forests. Because of the paucity of needles and grasses in these forests, only prolonged dry weather conditions create optimal conditions for fires (Schoennagel et al., 2004). However, substantial natural variability exists within each forest type due to both climate and topography (Heyerdahl et al., 2001). The 1988 Yellowstone fire was a good example of a high-severity fire facilitated by a 12-year drought, a low winter snowpack, and a dry, hot, and windy summer (Schoennagel et al., 2004).

C. Fire and Natural Climate Variability in the West

Fire in western ecosystems is determined by climatic variability, local topography, and human intervention. Managers have associated an average fire return interval to the forest types, but these averages mask large year-to-year variability in fire occurrence. When climate alters fuel loads and fuel moisture, forest susceptibility to wildfires changes and contributes to this natural variability (Whitlock et al., 2003). Drought has a major influence on fire in the United States (Siebold and Veblen, 2006; Enfield et al., 2001). Historically, drought patterns in the West are related to large-scale climate patterns in the Pacific and Atlantic oceans (Table 1). In the Pacific, the El Niño–Southern Oscillation (ENSO) varies on a 5–7 year cycle. In the Southwest and Colorado, La Niña years are dry and drought-induced fires are frequent, whereas El Niño years are wet and promote fuel accumulation. Conversely, in the Pacific Northwest El Niño years bring drier conditions and more fires (Swetnam and Baisan, 2003; Westerling and Swetnam, 2003). The Pacific Decadal Oscillation (PDO) varies on a 20–30 year cycle and the Atlantic Multidecadal Oscillation (AMO) varies on a 65–80 year cycle (McCabe et al., 2003; Schoennagel et al., 2004). The 1930s Dust Bowl in the Southwest occurred when both the PDO and AMO were in their warm phases.

As these large-scale ocean climate patterns vary in relation to each other, drought conditions shift from region to region in the United States (Table 1).

Mountainous terrain and the resulting small-scale climatic variation across the landscape also contribute to a mosaic of diverse types of vegetation in the western United States. Plants are well adapted to their environments, and fire has contributed strongly to their distribution. Historically, naturally occurring fire regimes have preserved the boundaries between prairies and forests, and sustained savannas and open forests. Thunderstorms and frequent lightning strikes—common occurrences during summers in the Great Plains, the slopes of the Sierras, and the Rockies, and during the monsoon season in the Southwest—allowed wildfires to maintain natural vegetation boundaries.

Cognizant of the natural relationship between climate, topography, and fire, Native Americans used fire to sustain grazing grounds for buffalo, for hunting and gathering food, and for controlling

Table 1

Historic Drought Patterns in Rocky Mountain Subalpine Forests and in the United States

Rocky Mountain subalpine forests	PDO positive phase (1913-1943, 1977–present)	PDO negative phase (1944-1976)
La Niña (NINO3 negative phase) (e.g., 1890, 1924, 1947, 1976, 2006)	High frequency (36–54%) of extreme drought in southern California and parts of the Southwest	Highest frequency (52–68 percent) of extreme drought in the Southwest
El Niño (NINO3 positive phase) (e.g., 1925, 1946, 1977, 1998, 2005) (Greater precipitation during all seasons in the central and southern Rockies)	high frequency (26–38%) extreme drought in the Pacific Northwest and Northern Rockies,	Least influence on drought

Schoennagel et al., 2005

United States	PDO positive phase (1913-1943, 1977-present)	PDO negative phase (1944-1976)
AMO positive phase (1926-1963, 1995-present)	1926–1943, 1995-present (e.g., 1930s drought; did not affect the Southwest)	1944–1963 (e.g., 1950s drought; mostly in the Midwest, Southwest, Rockies, and Great Basin)
AMO negative phase (1964-1994)	1977–1994 (e.g., Pacific Northwest and Maine drought)	1964–1976 (e.g., Southern California and central High Plains drought)

McCabe et al., 2003.

Historic drought patterns in Rocky Mountain subalpine forests and across the United States are related to large-scale climate patterns in the Pacific and Atlantic Oceans (Schoennagel et al., 2005; McCabe et al., 2003). For each mode of climate variability, the “positive phase” corresponds to the warm period of the oscillation and the “negative phase” corresponds to the cool period. See text for further explanation.

pests. Pyne (1982) observed that "the general consequence of the Indian occupation of the New World was to replace forested land with grassland or savannah, or, where the forest persisted, to open it up and free it from underbrush." Similarly, European descendants used fires to clear the land they settled during the western migration. They also introduced livestock and grazing, which disrupted fire regimes dramatically in the Southwest and the Sierras. These human activities led to extensive changes in land cover, including considerable shrub expansion (Swetnam and Baisan, 2003).

In the past few decades, the western United States has experienced droughts as severe as any on record (NOAA, 2002 as cited by Whitlock et al., 2003). Drought-killed trees have made forests more vulnerable to fires. Sustained drought conditions will make those forests less likely to recover, favoring replacement by grass-dominated semi-arid systems in the future. For example, large-scale drought-related dieback of pinyon pines has been observed recently in the Southwest and could bring large fires to the area in the near future. In the past, similar events caused by natural climate variability promoted vegetation shifts. They may become more prevalent again in a warmer future (Tebaldi et al., 2006; Seager et al., 2007) and will require human adaptation and changes in land use in the more arid parts of the western states.

The close relationship between climate and fire regime and its consequences for ecosystem structure are well recognized. Future changes in the climate, especially with regard to precipitation and drought, are likely to alter fire regimes in the western United States.

D. Fire and Humans

Human activities have added complexity to the western landscape. Fire suppression and prescribed burning, urban expansion, cattle production and grazing, and introduction of exotic plant species, have all conspired to artificially modify the fire regimes of western ecosystems and add another layer of uncertainty to the projections of the future of western forests.

1. Fire suppression

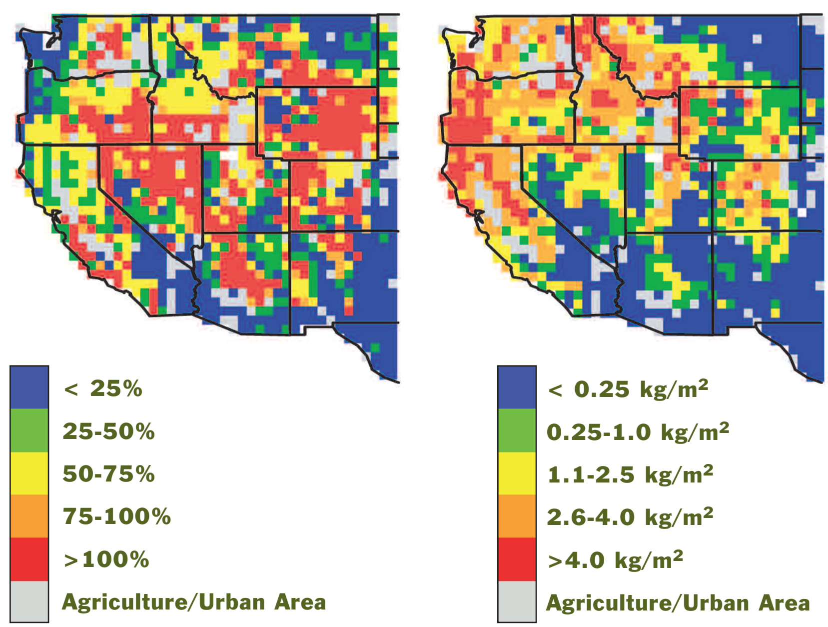
Fires were actively suppressed in the West during the 20th century. Increased government support for fire suppression and rapid mechanization of fire control after World War II contributed to an abrupt decline in the annual area burned after 1950. Fire suppression altered the ecology of plants and animals that depend on fire, sometimes with economic or aesthetic consequences. For example, some commercial tree species need fire to reproduce—lodgepole pine cones only release their seeds after intense heat has melted the waxy resin that keeps their scales closed. Similarly, giant sequoia seeds need bare soil, free from dead plant remains, to germinate. If current fire suppression practices were to continue for centuries, fire would not clean the forest floor, and sequoia groves would be in danger of disappearing as the old trees die and seedlings fail to establish.

Fire suppression in the Sierra Nevada and in the Southwest has resulted in denser forests with high fuel loads, allowing for catastrophic fires in low-elevation dry forests where high-intensity fires were rare historically. Land managers use prescribed burning to mitigate fuel build-up near population centers, but many forests in wilderness areas have accumulated so much fuel that run-away fires have become more common and remain a real danger in dry environments. Past forest management sought to eliminate wildfires but only succeeded in diminishing their extent, as climatic influences have overridden control efforts (Agee, 2003).

In contrast to the dry forests, subalpine forests at higher elevations are considerably wetter and colder. They often have historic fire-return intervals (50–300 years) longer than the period of time in which the current fire-exclusion policies have been in effect. So in general, they have not yet missed fire cycles as the dry forests have, and increased fire in these systems results from a climate signal rather than human intervention.

Figure 1

Simulated Increase in Plant Biomass as a Result of Fire Suppression



Simulated increase in total aboveground plant biomass (percent change on the left; absolute change on the right) due to post-1950 fire suppression by the end of year 2003 (Lenihan et al., forthcoming). Model results are displayed using the VEMAP (Schimel et al., 2000) agricultural and urban mask.

Fire suppression affects the role of forests in carbon sequestration. Scientists have shown that temperate forests of North America have been sequestering carbon over the past several decades. Sequestration in eastern states is thought to be primarily a result of regrowth since forest harvest and agricultural abandonment. In contrast, sequestration in the West is thought to be due primarily to decades of fire suppression coupled with a beneficial climate for growth and possible growth enhancement due to the anthropogenic increase in atmospheric carbon dioxide (CO_2; Caspersen et al., 2000). Several studies calculate the amount of carbon stored due to fire suppression (Table 2). Simulation results using a dynamic vegetation model (Lenihan et al., forthcoming) show the wide-ranging carbon gains across the continental United States (Figure 1). It also illustrates the importance of suppression in the West for total national carbon sequestered. However, future fires enhanced by the accumulation of fuels and climate change could eliminate much of the carbon gains due to suppression.

Table 2

Historical **Effects of Fire Suppression** on Ecosystems and Carbon Sequestration in the United States

Reference	Method	Fire Supression Impacts	Carbon Gains and Losses in the 1980s (Pg C $year^{-1}$)
Houghton (1999, 2003)	Land use statistics	Emissions from wildfire	-0.081
	Bookkeeping model	Re-growth from wildfire	+0.144
		Enhanced growth in western pines due to fire suppression	+0.026 (0.0052)
		Woody encroachment in non forest lands due to grazing and fire suppression	+0.061 (0.122)
Pacala et al. (2001)	Inventory Land Use Change	Woody encroachment in non-forest lands (grazing and fire suppression)	+0.12 to 0.13
Hurtt et al. (2002)	ED model	Woody encroachment (fire suppression)	+0.13
Sohngen and Haynes (1997)	—	Enhanced growth in U.S. forests	+0.0005
Lenihan et al. (forthcoming)	MC1 DGVM	Emissions from wildfires (with suppression)	+0.02 (0.003)
	(USA+Canada)	Enhanced growth due to fire suppression	+0.12
		Enhanced soil respiration due to suppression	-0.07
		Carbon sequestration due to suppression	+0.098

2. Effects of development

Most of the human settlements in the western United States are near dry woodlands and dry forests. As more people have moved into forested areas, public pressure for fire prevention has increased. As a result, the rapidly expanding wildland–urban interface, which was once subject to frequent, low-intensity surface fires, has now become prone to mixed- or high-severity fire regimes and thus more difficult to control, while population pressure and poor zoning regulations heighten the risk of fire damage.

The 20th century population explosion along the West Coast contributed to significant changes to natural fire regimes. Sources of fire ignition in fire-prone but lightning-poor environments, such as the California chaparral (Swetnam and Baisan, 2003), increased as population density rose. Jones (1995) and Stephens (2005) document a significant increase in human-caused ignitions in California. Smokers, campers, and arsonists are likely agents responsible for the high frequency of fires along roadways in southern California (Stephens, 2005).

3. Indirect human impacts

The indirect effects of population growth have also significantly affected fire regimes. Livestock grazing has reduced the amount of fine fuels, thus decreasing ignition potential. Logging has decreased the extent of old-growth forests and the amount of large dead wood critical to fire intensity. However, road cuts and timber harvest sites fragment forests, warming the air below the canopy and leading to the drying of fuels. Timber production has also promoted extensive plantations of even-age trees of the same species, which can easily propagate disturbances, such as fire or insect infestations.

In some areas, the combination of land use and fire exclusion has exacerbated the accumulation of dead fuels resulting from insect and disease outbreaks (Baker, 2003). Spruce budworm in Alaska is now able to successfully complete its life cycle in one year rather than two (Volney and Fleming, 2000). Multi-year droughts reduce tree growth and the production of defensive chemicals by the trees to fend off insect attacks (Logan et al., 2003). The resulting increase in dead fuels from extensive insect damage enhanced by fire suppression is often thought to greatly enhance the probability of catastrophic fires. The extent to which increased insect damage may affect fire regimes remains unclear. Despite an increase in dead fine fuels resulting from a spruce beetle outbreak in high elevation coniferous forests in Colorado, fire susceptibility did not increase (Bebi et al., 2003). Moreover, Bigler et al. (2005) found that the 2002 fires in mountain forests of northern Colorado were attributable primarily to extreme drought conditions rather than to insect damage or past management practices (Veblen, 2003). However, data on insect damage and fire is limited, and land managers would be prudent to perceive large-scale insect damage as a potential fire threat.

People also introduce non-native plant species that can change the fire regime and contribute to the disappearance of native systems. Invasive grasses such as cheatgrass can fill the gaps between native bunchgrass and sparse woody desert vegetation with a dense, continuous vegetation cover that, when dry, presents a major fuel source for brushfires. Frequent fires then allow the displacement of native vegetation by fire-tolerant invasive species, causing a shift from native shrubland to grassland. This invasion process is enhanced by urban development in desert areas, which introduces potential fire hazards and accelerates the invasion of grasses (Brooks et al., 2004). Alternatively, D'Antonio and Mahall (1991) show that ice plants from South Africa successfully compete with native shrubs in coastal California chaparral for critical surface water provided by winter rain or summer fog. The ice plants induce changes in the native shrub rooting profiles and can cause a decline in shrub biomass, lifespan, and reproduction. The conversion from a fire-prone coastal chaparral to a mixture of shrub and succulents with high live fuel moisture reduces fire intensity and fire spread, altering ecosystem properties and reducing the recruitment of native shrub species. By modifying the natural fire regime, exotic species are fostering changes that have not been observed before in these areas, making management of vegetation more difficult.

E. Fire in the 21st Century

Projections of future climate change suggest an increase in growing season length with earlier snowmelt periods as winters become milder and minimum temperatures increase (Mote et al., 2005). These changes will likely cause late-summer drought stress and increase plant susceptibility to pests and pathogens. At the same time, warmer conditions tend to speed up the life cycles of pests and pathogens and allow them to extend their ranges to vulnerable populations thus far protected by colder habitats (Berg et al., 2006). While pest damage can increase the amount of dead fuels in forests, drought stress can slow forest growth and reduce live fuels. These complex interactions make projections of future fire regimes and consequent impacts difficult. Consequently, researchers have designed models that can synthesize current knowledge about ecosystem structure and change and test hypotheses to project possible outcomes following changes in local or regional climate.

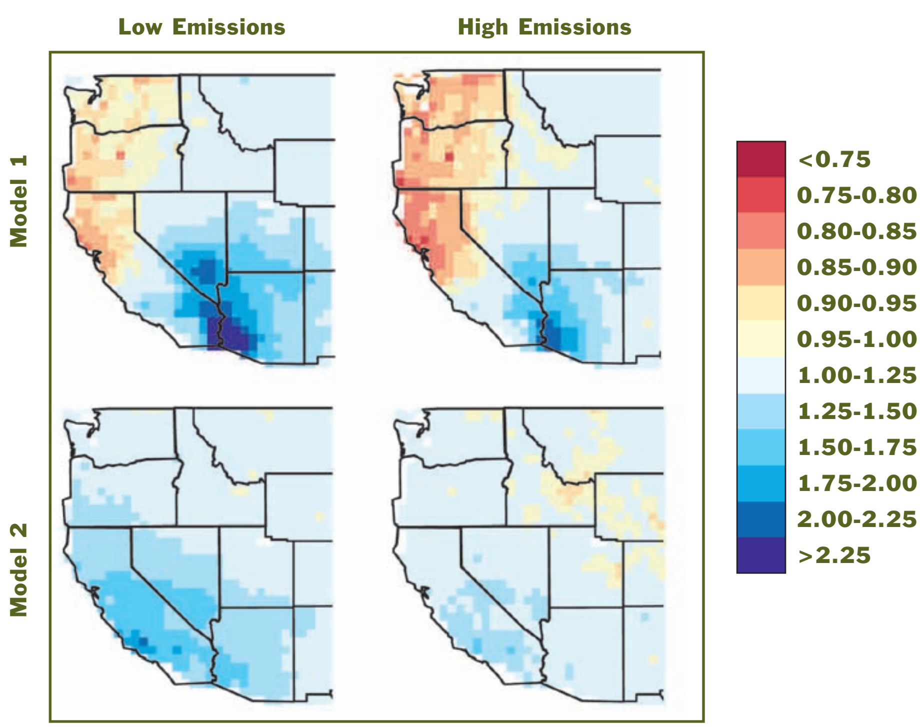
1. Future climate change scenarios

On average, climate models project drying of the western United States as a result of climate change, although the Northwest may be drier only during the summer when fire hazards are greatest (IPCC, 2007; Seager et al., 2007). However, western fire regimes vary on spatial scales smaller than those on which most climate models operate (see Wigley, 1999) and projections vary among individual models (Price et al., 2004), with some predicting wetter conditions in certain parts of the Southwest (e.g., Figure 2).

In particular, general circulation models may disagree in simulating the North American monsoon system, a seasonal precipitation pattern that brings moisture to parts of the southwestern United States during the summer (Collier & Zhang, 2007). Similarly, some climate models project an increase in annual average precipitation in California (Price et al., 2004). Moreover, global climate models have not been designed to simulate regional climate variability

Figure 2

Alternative Projections of Precipitation Change in the Western United States



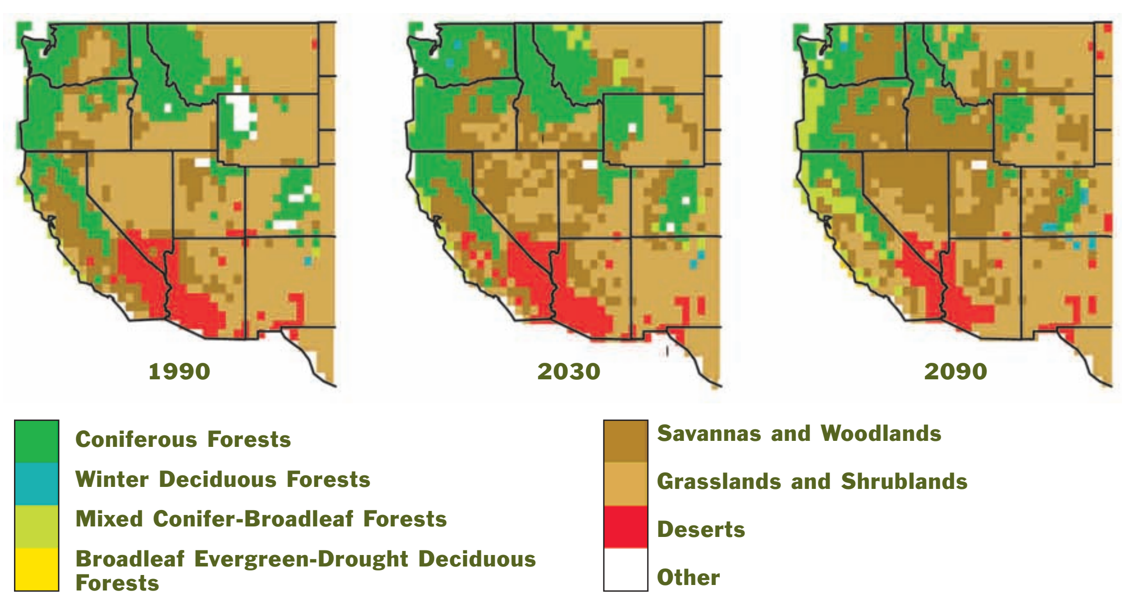
Alternative projections of precipitation change relative to the average for 1961–1990 in the western United States under different climate change scenarios. Two global climate models (Model 1 = HadCM3; Model 2 = CGCM2) and high (SRES A2) and low (SRES B2) greenhouse gas emission scenarios are compared. Values are fractions compared to the present. Values greater than 1.00 are increases; values less than 1.00 are decreases.

and extremes. Consequently, even though models agree on a general drying of the western United States, considerable uncertainty persists about seasonal and regional changes in precipitation patterns on the spatial scales needed to predict wildfires. Therefore, vegetation and fire models need to be driven by a variety of climate scenarios (e.g., Figure 2) to examine how fire is likely to respond to alternative possible climate outcomes (e.g., Figure 3).

When coupled with higher atmospheric CO_2 concentrations and longer growing seasons, wetter conditions promote the expansion of woody vegetation (Figure 3). The build-up of fuels combined with natural climate variability, and the likely occurrence of longer and more intense periodic droughts in the future, increases the likelihood of wildfires (Bachelet et al., 2001). While fuel loads are building, lightning and fire season length are expected to increase (Price and Rind, 1994). Increased use of wildlands by people is also likely to increase human-caused ignitions. Ironically, more frequent human-induced fires could reduce the fuel build-up that has resulted from fire suppression in dry forests, and it could therefore reduce fire danger in the long run. Of course, inadvertent burning is a poor means of land management and increased property damage is likely to result from an increase in accidental fires.

Figure 3

Simulation of **Past and Future Vegetation** Across the Western United States



Simulation of Past and Future Vegetation across the western United States with the CGCM2 global climate model and a high (SRES A2) greenhouse gas emission scenario (Lenihan et al., forthcoming).

The seasonal pattern of future climate change is also important. In general, climate models project higher average winter temperatures with increased winter rainfall (instead of snow) and decreased summer precipitation. Observations of a dwindling snowpack have confirmed that winter and spring temperatures are already getting warmer (Mote et al., 2005). In areas where winters are wet, this increase will not greatly affect fire danger. However, a change in summer moisture could greatly affect the spread of fire. The positive (in terms of carbon sequestration) outcome of a longer growing season resulting from earlier snowmelt could be cancelled out by the early build-up of fuels followed by late-summer droughts; such seasonal changes have already been linked to an increase in wildfires across the West (Westerling et al., 2006).

2. Fire effects on competitive interactions

Fire effects also include conversion of one type of vegetation cover to another as shown by Lenihan et al., (2003). Because of the uncertainty in future precipitation regimes, two types of vegetation changes are possible. A reduction in precipitation would allow drought-tolerant grasses to invade native shrublands and eventually shift the vegetation dominance to grassland. The flammability of grasses promotes greater rates of fire spread. Consequently, more extensive fires would progressively lead to higher fire frequencies, thereby depressing tree or shrub recovery and promoting the dominance of easily ignited grasses (Lenihan et al., 2003).

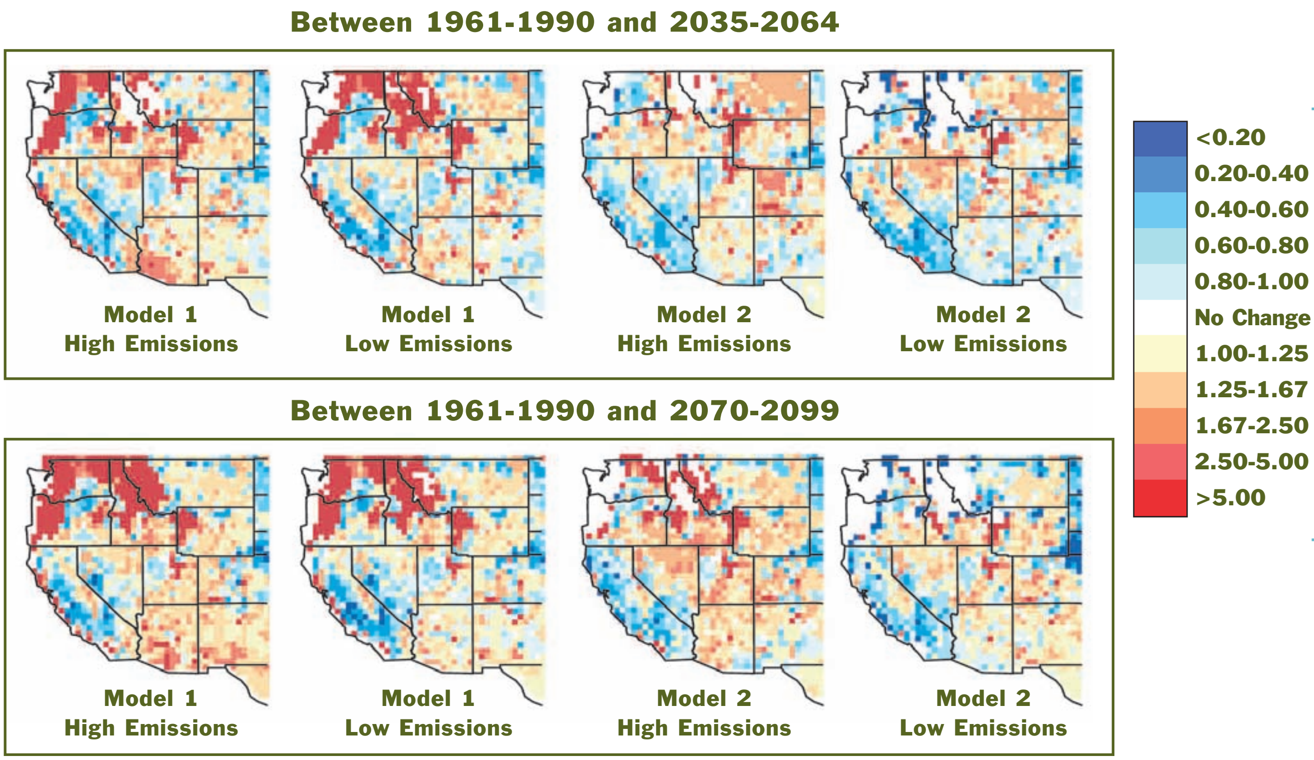
On the other hand, an increase in precipitation would enhance woody plant expansion. Trees and shrubs could provide cool moist shade and create “islands of fertility” unlikely to carry extensive fires because of their patchiness. However, because fire intensity depends on the properties of the total fuel load, a general increase in woody biomass and expansion of dense woodlands could promote more intense fires and more biomass consumption and mortality in the wake of a drought, thus ultimately reducing total tree biomass (Lenihan et al., 2003). In either wetter or drier conditions, models therefore indicate that fire could reduce forest and woody vegetation cover in the West in a future warmer world.

3. Projections

Biomass consumed by wildfire is estimated to at least double in the western United States during the 21st century, when simulated by a dynamic vegetation model under several future climate scenarios (Bachelet et al., 2001). The model shows that global warming may cause significant changes in regional vegetation patterns that would significantly alter the occurrence and distribution of wildfires in forest and grassland areas. Similarly, the average annual acreage and biomass burned across the United States during the 21st century is estimated to increase in comparison with the 20th century average, regardless of whether precipitation increases or decreases (Figure 4 and Table 3; Bachelet et al., 2003). For instance, some climate models project an increase in annual average precipitation in California (Figure 3; Price et al., 2004). Under these circumstances, the vegetation model simulates increased fire intensity and area burned because increased precipitation reduces fire and promotes fuel buildup during relatively wet years, setting the stage for larger, more intense fires during inevitable dry years (Lenihan et al., 2003). This interaction between fuels and year-to-year variability in precipitation produces the somewhat counter-intuitive result of more severe fire years simulated under wetter future climate scenarios.

Figure 4

Projected Change in **Plant Biomass Burned** by Wildfires



Projected change in biomass burned by wildfires from historical conditions based on two different climate models (Model 1 = HADCM3; Model 2 = CGCM2) and using high (SRES A2) and low (SRES B2) greenhouse gas emission scenarios. Values are fractions compared to the present. Values greater than 1.00 are increases; values less than 1.00 are decreases.

Different fire management regimes result in a variety of responses among the western vegetation types (Lenihan et al., forthcoming). For example, the biomass of both maritime conifer forests (that extend from Canada down to northern California) and Southwest temperate arid shrublands decrease under most future climate change scenarios when their natural fire regime is simulated, but they increase under most scenarios when fire suppression is simulated (Lenihan et al., forthcoming).

Several possible future climates and associated ecological responses can be simulated for regional studies (Figures 2 and 4). However, a general warming across the country is consistently projected by the models. Rainfall projections are more variable but point to a general decrease of up to 15 percent from 2040 to 2070 (Running, 2006). Regional droughts and resulting wildfires could significantly distress ecological systems, while wetter climates would benefit most of them. Under the drier scenario (HADCM3), a large increase in biomass consumed by wildfires is simulated for the

Table 3

Impacts of Future Climate Change on Fire and Carbon Sequestration

	Years	United States	Oregon	California
Live vegetation carbon	1901–2000	38 Pg C	2.43 Pg C	1.8 Pg C
	2001–2100	–0.20	–0.08	–0.18
	2031–2060	–0.12	+0.68	–0.23
	2071–2090	–0.37	–0.14	–0.12
Soil carbon	1901–2000	114 Pg C	6.20 Pg C	6.8 Pg C
	2001–2100	–0.01	–0.04	–0.07
	2031–2060	–0.01	–0.05	–0.07
	2071–2090	–0.02	–0.04	–0.08
Area burned by wildfires	1901–2000	104,019 km^2	3,299 km^2	7,352 km^2
	2001–2100	+1.10	+0.21	+0.19
	2031–2060	+1.08	+0.19	+0.14
	2071–2090	+2.61	+0.57	+0.19
Biomass burned by wildfires	1901–2000	140.661 Tg C	13.39 Tg C	22.39 Tg C
	2001–2100	+1.18	+0.08	–0.23
	2031–2060	+3.74	+0.01	–0.26
	2071–2090	+1.39	+0.52	–0.25

Impacts of future climate change on fire and carbon sequestration simulated by vegetation model MC1 (Lenihan et al., 2003; Bachelet et al., 2001). Results are shown for the conterminous United States as a whole and for the individual western states of Oregon and California. Reported are average annual carbon stocks in Pg (billion tons) C, biomass burned in Tg (million tons) C and area burned (km^2), averaged over the historical period, and future fractional changes for the entire 21st century (2001–2100) and the middle (2031–2060) and late (2071–2090) 21st century in the CGCM2 climate model and the SRES A2 greenhouse gas emission scenario.

Pacific Northwest region while a decrease occurs in southern California (Figure 4). It is possible that some areas will see initial gains in carbon sequestration in plant biomass, followed by net losses later in this century as climate change progresses (Table 3).

Given the uncertainty among future scenarios of rainfall, managers should develop contingency plans for alternative futures with specific regional emphases. Monitoring of ecosystem indicators could be configured to provide early warning of changing conditions.

F. Conclusions

Despite imprecise knowledge of future climate and human behavior, it is reasonable to conclude that fires will likely increase in the West. Future climate scenarios project summer temperature increases between 2 and 5°C and precipitation decreases of up to 15 percent (Running, 2006). Such conditions would exacerbate summer drought (Seager et al., 2007) and further promote high-elevation forest fires, releasing stores of carbon and further contributing to the buildup of greenhouse gases. Forest response to increased atmospheric CO_2 concentration—the so-called "fertilization effect"—could also contribute to more tree growth and thus more fuel for future fires, but the effects of CO_2 on mature forests are still largely unknown (Körner et al., 2005). However, high CO_2 should enhance tree recovery after fire and young forest regrowth, as long as sufficient nutrients and soil moisture are available, although the latter is in question for many parts of the western United States because of climate change.

Fire is a natural component of ecosystems in western North America, and its occurrence is strongly correlated with climate variability (Table 1). Paleoclimatic records and future climate scenarios show that regional climate can also shift within centuries and even decades. In addition, fuel conditions may change within years or a few months of major disturbances such as insect infestations, drought, windstorms, or forest thinning. Finally, population expansion and land-use changes can dramatically affect fire regimes. These complex interactions between natural and human pressures on natural systems present a challenge for projecting future fire regimes.

Woody expansion in the western United States may continue under future warming because of natural climate variability, ongoing increases in atmospheric CO_2, and continuing grazing restrictions and fire suppression. This expansion would allow for greater carbon sequestration. However, it would also allow for an increase in coarse fuels likely to carry catastrophic fires if droughts were to occur more frequently in the future. On the other hand, future droughts may also limit tree growth, thus ultimately reducing fuel production and allowing for more open forests with lower fire danger.

Fire suppression has allowed a build-up of fuels in low-elevation tree-dominated systems in the drier regions of the western United States. The 2002 Rodeo-Chedisky fire was the largest Arizona fire in recorded history with an extent of severe burning unprecedented in ponderosa pine forests. This high-severity fire illustrates the danger of fire suppression, which increases ladder fuels and allows contiguous tree crowns to develop in these dry forests. Historically, mild surface fires occurred every 7 to 10 years in these forests, preventing them from developing to this highly susceptible state. Similarly, the introduction of invasive grasses has allowed the build-up of a continuous source of fuel for wildfires in formerly patchy shrublands. Consequently, both fire exclusion and exotic species have tilted dry woody systems towards a greater sensitivity to drought and increased the likelihood of greater fire intensity and spread in the future.

Concurrently, higher elevation forests, which are not limited by the quantity but by the dryness of fuels and have not been greatly affected by suppression activities, are now subject to increasing drought stress. Between 1970 and 1986, the area of forest burned was six times smaller than in the last two decades as longer fire seasons (78 days longer), earlier snowmelt (1 to 4 weeks earlier), and warmer summers (almost 1°C warmer) combined to increase fire activity in the West (Westerling et al., 2006). Western forests, whether they were affected by past management, as in ponderosa pine forests, or not, as in subalpine spruce and fir forests, are now responding as a whole to an overwhelming climate signal. High-severity fires, such as the 1988 Yellowstone fire and the 2002 Hayman fire, happened in response to extreme climate signals, which could become more dominant in a warmer future (Running, 2006; Seager et al., 2007).

Future population pressure will also contribute to an increase in fire danger in the West as human development expands into increasingly fire-prone environments in a more variable climate. New strategies have been delineated to address this problem (Stephens and Ruth, 2005), but it will constitute an increasing challenge for land managers in the 21st century.

References

Agee, J.K. 2003. Historical range of variability in eastern Cascades forests, Washington, USA. *Landscape Ecology* 18:725-740.

Agee, J.K. 1998. The landscape ecology of western forest fire regimes. *Northwest Science* 72:24-34.

Agee, J. K. 1997. The severe weather wildfire - too hot to handle? *Northwest Science* 71:153-156.

Bachelet D., R.P. Neilson, T. Hickler, R.J. Drapek, J. M. Lenihan, M.T. Sykes, B. Smith, S. Sitch, and K. Thonicke. 2003. Simulating past and future dynamics of natural ecosystems in the United States. *Global Biogeochemical Cycles* 17:doi:10.1029/2001GB001508.

Bachelet, D., R.P. Neilson, J.M. Lenihan, and R.J. Drapek. 2001. Climate change effects on vegetation distribution and carbon budget in the United States. *Ecosystems* 4:164-185.

Baker, W.L. 2003. Fires and climate in forested landscapes of the US Rocky Mountains. In *Fire and Climatic Change in Temperate Ecosystems in the Western Americas.* T.T. Veblen, W.L. Baker, G. Montenegro and T.W. Swetnam (eds.). Springer-Verlag, New York, NY.

Bebi, P., D. Kulakowski, and T.T. Veblen. 2003. Interactions between fire and spruce beetles in a subalpine Rocky mountain forest landscape. *Ecology* 84:362-371.

Berg, E.E., J.D. Henry, C.L. Fastie, A.D. De Volderd, and S.M. Matsuoka. 2006. Spruce beetle outbreaks on the Kenai Peninsula, Alaska, and Kluane National Park and Reserve, Yukon Territory: Relationship to summer temperatures and regional differences in disturbance regimes. *Forest Ecology and Management* 227:219-232.

Bigler, C., D. Kulakowski, and T. T. Veblen. 2005. Multiple disturbance interactions and drought influence fire severity in rocky mountain subalpine forests. *Ecology* 86:3018-3029.

Brooks, M.L., C.M. D'Antonio, D.M. Richardon, J.B. Grace, J.E. Keeley, J.M. DiTomaso, R.J. Hobbs, M. Pellant, and D. Pyke. 2004. Effects of Invasive Alien Plants on Fire Regimes. *BioScience* 54:677-688.

Caspersen, J.P., S.W. Pacala, J.C. Jenkins, G.C. Hurtt, P.R. Moorcroft, and R.A. Birdsey. 2000. Contributions of land-use history to carbon accumulation in U.S. forests. *Science* 290:1148-1151.

Christensen, J.H., B. Hewitson, A. Busuioc, A. Chen, X. Gao, I. Held, R. Jones, R.K. Kolli, W.T. Kwon, R. Laprise, V. Magaña Rueda, L. Mearns, C.G. Menéndez, J. Räisänen, A. Rinke, A. Sarr, and P. Whetton. 2007. Regional Climate Projections. In *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*. S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M. Tignor, and H. L. Miller (eds.), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Collier, J. C. and G. J. Zhang, 2007. Effects of increased horizontal resolution on simulation of the North American monsoon in the NCAR CAM3: an evaluation based on surface, satellite, and reanalysis data. *Journal of Climate*, 20, doi:10.1175/JCLI4099.1.

D'Antonio, C.M. and B. Mahall. 1991. Root overlap and interference for soil moisture between an invasive succulent and two native shrub species. *American Journal of Botany* 78:885-894.

Enfield, D.B., A.S. Mesta-Nuñez, and P.J. Trimble. 2001. The Atlantic multidecadal oscillation and its relation to rainfall and river flows in the continental U.S. *Geophysical Research Letters* 28:2077-2080.

Heyerdahl, E., L.B. Brubaker, and J.K. Agee. 2001. Factors controlling spatial variation in historical fire regimes: a multiscale example for the interior West, USA. *Ecology* 82:660-678.

Houghton, R.A. 1999. The annual net flux of carbon to the atmosphere from changes in land use 1850-1990. *Tellus B* 51:298-313.

Houghton, R.A. 2003: Why are estimates of the terrestrial carbon balance so different? *Global Change Biology* 9:500-509.

Hurtt, G.C., S.W. Pacala, P.R. Moorcroft, J. Caspersen, E. Shevliakova, R.A. Houghton, and B. Moore III. 2002. Projecting the future of the U.S. carbon sink. *Proceedings of the National Academy of Sciences USA* 99:1389-1394.

Jones, R. 1995. How will increased population affect wildfire incidence: is ignition frequency in the Sierra Nevada related to population density? Fire and Resource Assessment Program, California Department of Forestry and Fire Protection. Available online at http://frap.cdf.ca.gov/projects/ignition_regression/ignit_pop.html.

Körner, C., R. Asshoff, O. Bignucolo, S. Hättenschwiler, S.G. Keel, S. Peláez-Riedl, S. Pepin, R.T.W. Siegwolf, and G. Zotz. 2005. Carbon flux and growth in mature deciduous forest trees exposed to elevated CO_2. *Science* 309:1360-1362.

Lenihan, J.M., R.J. Drapek, D. Bachelet, and R.P. Neilson. 2003. Climate change effects on vegetation distribution, carbon and fire in California. *Ecological Applications* 13:1667-1681.

Lenihan, J.M., R.J. Drapek, D. Bachelet, and R.P. Neilson (Forthcoming). Simulated response of conterminous United States ecosystems to climate change at different levels of fire suppression, CO_2 emission rate, and growth response to CO_2. *Global and Planetary Change.*

Logan, J.A., J. Regniere, and J.A. Powell. 2003. Assessing the impacts of global warming on forest pest dynamics. *Frontiers in Ecology and the Environment* 1:130-137.

McCabe, G.J., M.A. Palecki, and J.L. Betancourt. 2003. Pacific and Atlantic ocean influences on multidecadal drought frequency in the United States. *Proceedings of the National Academy of Sciences USA* 101:4136-4141.

Mote, P.W., A.F. Hamlet, M.P. Clark, and D.P. Lettenmeir. 2005. Declining Mountain Snowpack in Western North America. *Bulletin of the American Meteorological Society* 86:39-49.

Pacala, S.W., G.C. Hurtt, R.A. Houghton, R.A. Birdsey, L. Heath, E.T. Sundquist, R.F. Stallard, D. Baker, P. Peylin, P. Moorcroft, J. Caspersen, E. Shevliakova, M.E. Harmon, S.M. Fan, J.L. Sarmiento, C. Goodale, C.B. Field, M. Gloor, and D. Schimel. 2001. Consistent Land- and Atmosphere-Based U.S. Carbon Sink Estimates. *Science* 292:2316-2320.

Price, C. and D. Rind. 1994. Possible implications of global climate change on global lightning distributions and frequencies. *Journal of Geophysical Research* 99:10823-10831.

Price, D.T., D.W. McKenney, P. Papadopol, T. Logan, and M.F. Hutchinson. 2004. High resolution future scenario climate data for North America. *Proceedings of the 26th Conference on Agricultural and Forest Meteorology, Vancouver, B.C., 23–26 August 2004*. American Meteorological Society, Boston, MA, 7.7, 13 pp.

Pyne, S.J. 1982. *Fire in America: A Cultural History of Wildland and Rural Fire*. Princeton University Press, Princeton, NJ.

Running, S.W. 2006. Is global warming causing more, larger wildfires? *Science* 313:927-928.

Schimel, D., J. Melillo, H. Tian, A.D. McGuire, D. Kicklighter, T. Kittel, N. Rosenbloom, S. Running, P. Thornton, D. Ojima, W. Parton, R. Kelly, M. Sykes, R. Neilson, and B. Rizzo. 2000. Contribution of increasing CO_2 and climate to carbon storage by ecosystems in the United States. *Science* 287:2004-2006.

Schoennagel, T., T.T. Veblen, and W.H. Romme. 2004. The interaction of fire, fuels and climate across Rocky Mountain Forests. *BioScience* 54:661-676.

Schoennagel, T., T.T. Veblen, W.H. Romme, J.S. Sibold, and E.R. Cook. 2005. ENSO and PDO variability affect drought-induced fire occurrence in Rocky Mounatin subalpine forests. *Ecological Applications* 15:2000-2014.

Seager, R., M. Ting, I. Held, Y. Kushnir, J. Lu, G. Vecchi, H.P. Huang, N. Harnik, A. Leetmaa, N.C. Lau, C. Li, J. Velez, and N. Naik. 2007. Model projections of an imminent transition to a more arid climate in southwestern North America. *Science* 316:1181-1184.

Siebold, J.S. and T.T. Veblen. 2006. Relationships of subalpine forest fires in the Colorado Front Range with interannual and multidecadal-scale climatic variation. *Journal of Biogeography* 33:833-842.

Smith, J.K. and W.C. Fischer. 1997. *Fire ecology of the forest habitat types of northern Idaho.* General Technical Report, INT-GTR-363. USDA Forest Service, Intermountain Forest and Range Experiment Station, Ogden, UT.

Sohngen, B. and R. Haynes. 1997. The potential for increasing carbon storage in United States unreserved timberlands by reducing forest fire frequency: an economic and ecological analysis. *Climatic Change* 35:179-197.

Stephens, S.L. 2005. Forest fire causes and extent on United States Forest Service Lands. *International Journal of Wildland Fire* 14:213-222.

Stephens, S.L. and L.W. Ruth. 2005. Federal forest-fire policy in the United States. *Ecological Applications* 15:532-542.

Swetnam, T.W. and C.H. Baisan. 2003. Tree-ring reconstructions of fire and climate history in the Sierra Nevada and Southwestern United States. In *Fire and Climatic Change in Temperate Ecosystems of the Western Americas.* T.T. Veblen, W. Baker, G. Montenegro and T. W. Swetnam (eds.). Springer-Verlag, New York, NY.

Tebaldi, C., J.M. Arblaster, K. Hayhoe, and G.M. Meehl. 2006. Going to the extremes: an intercomparison of model-simulated historical and future changes in extreme events. *Climatic Change* 79:185-211.

Veblen, T.T. 2003. Historic range of variability of mountain forest ecosystems: concepts and applications. *Forest Chronicle* 79:223-226.

Volney, W.J.A. and R.A. Fleming. 2000. Climate change and impacts of boreal forest insects. *Agriculture, Ecosystems & Environment* 82:283-294.

Westerling, A.L. and T.W. Swetnam. 2003. Interannual to decadal drought and wildfire in the western United States. *Eos, Transactions, American Geophysical Union* 84:545, 554-555.

Westerling, A.L., H.G. Hidalgo, D.R. Cayan, and T.W. Swetnam. 2006. Warming and Earlier Spring Increase Western U.S. Forest Wildfire Activity. *Science* 313:940-943.

Whitlock, C., S.L. Shafer, and J. Marlon. 2003. The role of climate and vegetation change in shaping past and future fire regimes in the northwestern US and the implications for ecosystem management. *Forest Ecology and Management* 178:5-21.

Wigley, T.M.L. 1999. *The Science of Climate Change: Global and U.S. Perspectives.* Pew Center on Global Climate Change, Arlington, VA.

IV. Gulf Coast Wetland Sustainability in a Changing Climate

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A. Introduction

The wetlands of the U.S. Gulf Coast provide services that are significant to the quality of life in the region, help sustain the national economy, and help protect life and property from climate extremes. Fisheries, recreation, and tourism have all thrived in the Gulf Coast region alongside urban development, agriculture, shipping, and the oil and gas industries. However, some regions of the Gulf Coast, such as the Mississippi River Delta and Florida Everglades, are experiencing some of the highest wetland loss rates in the U.S., largely because of engineered modifications to regional watersheds and coastal landscapes. Such modifications increase the vulnerability of these wetlands to future climate variability and change. Sustainable restoration of Gulf Coast wetlands requires planning for a more extreme future climate by returning critical water resources in the coastal landscapes to levels that existed before humans began modifying this region three centuries ago (Day et al., 2007).

Gulf Coast wetlands support economic and ecological productivity as well as quality of life in many ways. Wetlands provide food, refuge, and nurseries for fish and shellfish, and they support the region's large commercial and recreational fishing industries. As a result, Louisiana's commercial fisheries account for about 30 percent of the nation's total fish catch. In addition, Gulf Coast wetlands provide stopover habitat for an estimated 75 percent of the waterfowl migrating along the Central Flyway (Environmental Health Center, 1998). Wetland soils and vegetation naturally store water, filter sediment and pollutants from fresh water supplies, and help stabilize shorelines by reducing erosion and storm surges associated with rising sea levels (Daily et al., 1997; Mitsch and Gosselink, 2000).

Gulf coastal systems also provide diverse natural resources that have been transformed to provide opportunities for economic development. The United States ranks second in worldwide natural gas production and third in oil production (Hetherington et al., 2007). Between 2000 and 2005, one-fifth to one-quarter of U.S. domestic natural gas and crude oil production occurred in the Gulf of Mexico (Energy Information Administration, 2007). In 2004, shipping ports in the Gulf states (Texas, Louisiana, Mississippi, Alabama, and Florida) accounted for 49 percent (by tonnage) of all waterborne cargo entering and leaving the United States (U.S. Army Corps of Engineers, 2006). Texas and Louisiana each handled more waterborne cargo by tonnage than California, New York, and New Jersey combined. The Mississippi delta is heavily impacted by shipping—the ports of New Orleans, South Louisiana, Baton Rouge, and Lake Charles handle more than 20 percent of the nation's foreign waterborne commerce. Economic development in Florida has transformed coastal wetlands through recreational activities and residential development, along with major investments in agriculture. In 2005, Florida had almost 86 million visitors who spent more than $62 billion (Florida Tax Watch, 2006). The vast majority of tourism in Florida is to visit the state's coastal resources. Such heavy use can create pressures on natural coastal ecosystems.

Degradation of coastal wetlands through land development and water management reduces the capacity of wetlands to provide significant ecosystem services that reduce the risks of living and working in coastal landscapes. For example, extensive coastal wetland landscapes, especially forested ecosystems, can reduce storm surge and wind energy during tropical storms and cyclones, minimizing hurricane damage to life and property. In part because of recent hurricanes, local, state, and federal agencies have renewed their emphasis on coastal wetland restoration in the Gulf Coast region (Working Group for Post-Hurricane Planning for the Louisiana Coast, 2006; Day et al., 2007). However, such programs may fail without effective planning for future climate change, including accelerated sea-level rise and the potential intensification and increased frequency of hurricanes.

Human activities intended to reduce damage to life and property from climate extremes have unintentionally increased the vulnerability of coastal areas to climate change by altering the natural hydrologic functions of wetlands (National Research Council, 2005; CPRA, 2007). For coastal wetlands to be sustained in a changing climate, therefore, restoration planning must account for the consequences of both climate change and human engineering of the environment.

B. Gulf Coast Wetlands and Water Management

Two of the most distinctive and extensive wetland landscapes in North America and in the world are located along the Gulf Coast—the Florida Everglades and Louisiana's Mississippi River Delta. These wetland ecosystems depend heavily on water availability, as does the region's economic development. However, the natural capacity of coastal wetlands in the Gulf Coast region to store, distribute, and purify water has been greatly diminished by coastal development and the construction of water management systems.

The highly engineered landscapes of the Everglades and the Mississippi Delta were developed in response to major floods and hurricanes that occurred from 1926 to 1948 (Light and Dineen, 1994; Barry, 1997). Major federal work projects, including the Mississippi River and Tributary Project of 1930 and the Central and Southern Florida Project for Flood Control and Other Purposes of 1948, were authorized by Congress to protect life and property following these major natural disasters. Canals, floodgates, levees, and water control structures were built to reduce flood risks to agriculture, urban development, energy-related industries, and commercial transportation.

Although flood control projects provided temporary relief from flooding, they also interfered with the natural hydrological processes that are necessary to sustain the structure, function, and extent of wetland ecosystems and reduced the natural capacity of the wetlands to mitigate flooding (Boesch et al., 1994; Davis and Ogden, 1997). The loss and degradation of wetlands has resulted in increased risks from coastal storms and tidal surges, leading to unintended consequences for both human and natural systems. Today, Louisiana and Florida, along with Texas, are the top three states in the nation in terms of annual economic losses resulting from hurricanes and floods.

Net wetland elevation is determined by the balance between soil building processes (accretion) and land sinking (soil subsidence) relative to the rate of sea-level rise. Wetland soils develop from and are sustained by mineral sediments carried by rivers and deposited by floods, and from organic material produced by plants within the wetland landscape. These soil-building processes enable wetlands to gain elevation (accrete) as sea-levels rise (Mitsch and Gosselink, 2000). Human activities slow down accretion by regulating water flow and, therefore, sediment and nutrient supply. Humans also cause soil subsidence and erosion through such processes as groundwater extraction, oil and gas withdrawals, and dredging of navigation channels (Morton et al., 2003). Natural compaction processes also contribute to subsidence. In effect, sea-level rise adds to the rate of subsidence as the sea surface rises relative to the land. In order for wetland elevation to remain stable or to rise, therefore, the rate of soil accretion must equal or exceed the combined rates of natural and human-induced soil subsidence plus sea-level rise.

Prior to human modification, Gulf Coast wetlands were sustained because soil formation kept pace with natural compaction and historical sea-level rise. Today and into the future, their sustainability depends on the ability to keep pace with human-induced elevation loss and accelerating sea-level rise resulting from global warming caused by increasing concentrations of manmade greenhouse gases in the atmosphere (IPCC, 2007). If soil formation cannot keep pace with all of these competing processes, coastal wetlands will experience increased flooding from rising seas, reducing plant production and further accelerating wetland loss (DeLaune et al., 1994). Under such conditions, wetlands ultimately will "drown" and convert to open water.

The well-documented history of adaptation by ecological and social systems in both the Everglades and the Mississippi delta provides insights into the increasing challenges to sustainable development faced by coastal communities under the added stress of a changing climate.

1. Coastal Louisiana

Southern Louisiana has a working coast, with extensive human settlements across the landscape. Humans have taken a variety of actions to manage the risks of occupying the extremely dynamic river delta environment (Boesch et al., 1994; Laska et al., 2005; Day et al., 2007). Major landscape changes have occurred over the past century in the nearly 1.3 million square mile watershed of the Mississippi River, including conversion of more than 80 percent of forested wetlands to agriculture and urban areas, river channels, and dams and levees (CENR, 2000).

Under natural conditions, deltaic environments, such as the Mississippi River Delta in southern Louisiana, receive sediment through openings in natural levees (river crevasses) during flood pulses, adding to soil formation (Day et al., 1994; 2007; Perez et al., 2003). The construction of earthen and concrete levees and of a massive structure to regulate the flow of the Mississippi and Red Rivers has restricted the natural supply of sediment and fresh water to the delta's floodplain (Kesel, 1988). In addition, dams on the Mississippi River have decreased sediment delivery to the lower delta by more than 50 percent over the past 150 years. Together, the reduction of sediment delivery from the Mississippi watershed to the lower delta and the inability of sediment to enter wetland basins through river crevasses have caused significant losses of the region's coastal wetlands (Kesel, 1988; Day et al., 2007).

The Mississippi delta also receives sediment from hurricanes. A recent study estimates that hurricanes Rita and Katrina deposited an average of two inches of sediment over a large area of coastal wetlands in Louisiana (Turner et al., 2006). However, sediment from hurricanes alone has been insufficient to maintain the elevation of coastal wetlands in southern Louisiana over the past century relative to regional subsidence (Cahoon et al., 1995), particularly given other changes in regional hydrology caused by extensive construction of canals and other artificial water control features (Boesch et al., 1994; Stokstad, 2006).

Coastal Louisiana experiences the greatest wetland loss in the nation, and delta wetlands are now disappearing at an average rate of 17 square miles per year or about 50 acres per day (Gosselink, 1984; Conner and Day, 1988; Barras et al., 2003). Wetland loss rates over the next 20 years in coastal Louisiana, due to the combination of sea-level rise and disruption of natural coastal processes, will continue to convert land to open water, threatening the region's fisheries, aquaculture and coastal agriculture, as well as commercial shipping and other industries located near the coast (Louisiana Coastal Wetlands Conservation and Restoration Task Force, 1998; Barras et al., 2003; U.S. Army Corps of Engineers, 2004a).

2. Florida Everglades

In contrast to the heavily developed Louisiana coast, South Florida's Everglades are protected by the U.S. National Park Service, with additional international designation as a Biosphere Reserve (MAB, 2007), a World Heritage Site (World Heritage Committee, 2007), and a Wetland of International Importance (Ramsar Convention, 2007). However, the Everglades National Park is also located within a watershed of intensive human settlement, with one of the area's largest urban and agricultural regions to the north of the park boundary (Harwell, 1997).

Before European settlement, the landscape of South Florida was a mosaic of habitats connected by the flow of fresh water across a gently sloping landscape from Lake Okeechobee through the Everglades and south to Florida Bay (Light and Dineen, 1994; Harwell, 1997). The wetland landscape included sawgrass interspersed with tree islands, with mangrove forests extending over an area of three million acres in the estuarine transition zone (Gunderson, 1994). The natural evolution of the region was driven in part by the very slow relative rise in sea level over the past 3,200 years, as well as extreme episodic events—in particular, fires, freezes, hurricanes, floods, and droughts.

To protect human settlement from these natural events, the federal government developed one of the world's most extensive water management systems in South Florida. A series of canals and water control structures unnaturally reduced the flow of fresh water to Florida Bay (Light and Dineen, 1994). The subsequent alterations of wetland habitat and reduction in wading bird populations implicate these fresh water diversions in the reduced sustainability of the region's natural resources. As a result of water engineering, the Everglades is now an endangered ecosystem, the sustainability of which is vulnerable to projected climate changes (Harwell, 1997; 1998).

Mangrove forests dominate the coastal margin of the Everglades. In contrast to wetlands in the Mississippi delta, soil building and elevation in the Everglades is dominated by plant productivity, producing highly organic soils in the absence of significant river sediment deposition (Lynch et al., 1989; Parkinson et al., 1994). Although mangroves situated at the mouths of estuaries in the southwest Everglades experience pulsed inputs of sediment during storm events (Chen and Twilley, 1998; 1999), the Everglades as a whole rely on *in situ* soil production. Hence, the rate of soil building in the Everglades is primarily limited by plant productivity, regulated by water and nutrient delivery. Because subsidence in the Everglades is insignificant, plant vulnerability is related mainly to the rise in sea level relative to the rate of soil formation. As in the Mississippi delta, soil-building processes have been altered by engineered water management systems.

C. Gulf Coast Wetlands in a Changing Climate

Global climate change is expected to affect air and water temperatures, ocean and atmosphere circulation, sea-level rise, the intensity of hurricanes, and the timing, frequency, and magnitude of precipitation (IPCC, 2007). Under natural conditions, coastal wetlands adjust to rising seas and changes in local storm patterns, but climate changes and human activities that alter natural conditions disrupt wetland hydrology, biogeochemical cycling, and other processes that sustain wetlands (Morris et al., 2002). In the Gulf Coast region, the combined effects of water engineering, land development, natural subsidence, and climate change will have tremendous consequences for coastal wetlands in the coming decades (Twilley et al., 2001; Scavia et al., 2002). The Gulf Coast region is considered especially vulnerable to a changing climate because of its relatively flat topography, rapid rates of land subsidence, water engineering systems, extensive shoreline development, and exposure to major storms. In the Mississippi delta, rapid subsidence has already produced accelerated rates of relative sea-level rise (absolute sea-level rise plus land subsidence; Working Group for Post-Hurricane Planning for the Louisiana Coast, 2006; Day et al., 2007).

Recent evidence suggests that human-induced global warming has already increased both the intensity and frequency of hurricanes in the North Atlantic, including the Gulf of Mexico (Emanuel, 2005; Webster et al., 2005; Mann and Emanuel, 2006; Santer et al., 2006; Trenberth, 2006; Trenberth and Shea, 2006). A recent analysis of Atlantic basin hurricane activity by Goldenberg and others (2001) indicated a five-fold increase in hurricanes affecting the Caribbean when comparing 1995–2000 to the previous 24 years (1971–1994). Hurricanes exhibit multi-decadal patterns that appear to be associated with variations in tropical sea-surface temperature patterns and vertical wind shear, and the Atlantic basin is in a period of high-level hurricane activity that could persist for 10–40 years, irrespective of global warming (Goldenberg et al., 2001). Moreover, several ocean-coupled global circulation models project that the intensity of hurricanes will increase as the climate warms during the next 100 years (Knutson and Tuleya, 2004; IPCC, 2007).

Recent Gulf Coast hurricanes demonstrate the damaging effects that intense hurricanes can have on life, property, and natural resources in coastal areas. However, hurricanes can also increase the rate of soil accretion in coastal wetlands, helping to maintain wetland elevation relative to sea-level rise

(Poff et al., 2002; Turner et al., 2006). In addition, the increased runoff resulting from hurricanes transports more water and nutrients to coastal habitats (Poff et al., 2002). These positive effects, however, will depend on local wetland conditions, and in highly altered systems, hurricane-related changes in accretion and runoff patterns may be more damaging than beneficial, especially considering that these processes were insufficient to maintain coastal wetlands during the 20th century.

The frequency and magnitude of the El Niño/Southern Oscillation (ENSO) also has a strong effect on ecological conditions in coastal areas. During ENSO events, large-scale disruptions to global weather patterns occur in the atmosphere and in the tropical Pacific Ocean. In general, El Niño events (the warm ENSO phase associated with unusually warm waters in the tropical Pacific) are correlated with greatly increased winter precipitation in the Gulf Coast region. During La Niña (the cool ENSO phase), fall and winter along the Gulf Coast are warmer and drier than usual. Hurricanes increase during La Niña events, but are less frequent during El Niño events (Bove et al., 1998). El Niño events have occurred more frequently and have persisted longer since the 1970s, a trend that has been linked statistically to global warming (Trenberth and Hoar, 1997), although this linkage remains to be confirmed (McPhadden et al., 2006). Future intensification of El Niño events could alter marine and terrestrial ecosystems in unpredictable ways (McPhadden et al., 2006).

The future hydrology of Gulf Coast watersheds, including peak flows, will depend on the balance of rainfall and evaporation in a warming climate, as modified by human consumption and management of water resources. In major rivers such as the Mississippi, water flows will be determined by rainfall trends in watersheds hundreds of miles upstream from the coast, as well as by the region's massive flood control projects. However, regional predictions of runoff are uncertain because runoff is sensitive to interactions among precipitation, temperature, and evaporation (Poff et al., 2002). The future of local precipitation remains uncertain, but the intensity of rainfall events can increase even if average precipitation decreases (Knutson et al., 1998). Intense rainfall can contribute to marsh flooding. On the other hand, extended periods of drought during La Niña events can lead to marsh dieback. A 25-month drought, interacting with other environmental stresses, is considered the main cause of a severe dieback of 100,000 acres of salt marsh in coastal Louisiana in 2000 (Kennedy et al., 2002; McKee et al., 2004).

Even if storm intensities remain constant, sea-level rise may contribute to increased shoreline erosion, wetland flooding, and higher storm surges. Rising sea levels will generate higher storm surges even from minor storms (Reed, 2002). Flood damage in Gulf Coast states will increase as a result of the combined effect of increased storm surges and the decreased storm surge-reduction capacity of altered wetlands (Working Group for Post-Hurricane Planning for the Louisiana Coast, 2006).

Climate models project sea-level rise along the Gulf Coast to range from one to three feet in the next century (Twilley et al., 2001; Kennedy et al., 2002). The position of wetlands relative to the sea surface will remain constant over time only if the combined effects of land subsidence and rising seas can be balanced by elevation gain from wetland soil formation (Morris et al., 2002; Reed, 2002). With regional subsidence projected to range from 8 to 40 inches in the next century, relative sea-level rise—the combination of absolute sea-level rise and land subsidence—over the next 100 years could range from two feet along most of the Gulf Coast to more than six feet along the Mississippi delta and coastal Louisiana (Penland and Ramsey, 1990; Church, 2001; IPCC, 2007).

Wetland response to sea-level rise depends on local interactions between sediment and organic matter accumulation, hydrology, subsurface processes, and storm events (Reed, 1995; Cahoon et al., 1995). Over the past several decades, the engineering of water management systems and the increased frequency and intensity of storms have altered the timing and amount of sediment delivered to the wetlands of the Mississippi River deltaic plain. Under natural conditions, deltaic environments receive river sediment during pulsed flood events (Day et al., 1994; Perez et al., 2003). However, river management systems, such as levees and flow diversions, have reduced river-pulsed floods and the delivery of sediment to delta wetlands, decreasing their ability to form soil and raise elevation (Baumann et al., 1984; Day et al., 2007).

Some Louisiana marshes have adjusted to this change in the magnitude and source of sediment delivery, and still survive in hydrologic basins where relative rates of sea-level rise measured at tide gauges reach 0.4 inches per year (three feet per century; Penland and Ramsey, 1990). But other wetland areas within these same basins are showing reduced soil build up and a decreased ability to keep pace with the net changes in water levels. Salt marshes with high sediment loading (such as those in Louisiana) are likely to keep pace with a relative sea-level rise of less than four feet per century, based on models for similar marshes along the U.S. Atlantic Coast (Morris et al., 2002).

However, in some parts of the central Gulf region, relative sea-level rise is projected to reach six feet by the end of the century, exceeding the ability of many Louisiana marshes to cope under the present conditions of reduced sediment delivery. If inundation exceeds accretion, and if inland migration is blocked by shoreline development (see below), sea-level rise will flood wetlands and lead to plant death.

Accelerated sea-level rise is also likely to be one of the most critical environmental challenges to the sustainability of mangrove ecosystems along the Florida coast, in spite of the lack of subsidence in these systems (Davis et al., 2005). In regions with little sediment input, the maximum rate of relative sea-level rise that mangroves can sustain is estimated to be 0.75 feet or less over the next century, much lower than estimates for sediment-rich deltaic regions and projected sea-level rise for the Gulf region (reviewed in Twilley, 1997). This estimate assumes stable geologic formations and minimum rates of subsidence, which generally apply for Florida wetlands underlain with limestone (Wanless et al., 1994).

The ability of wetlands to migrate inland to areas of decreasing tidal inundation along undeveloped shores is another way coastal wetlands in south Florida and coastal Louisiana can persist in spite of rising seas (Ross et al., 2000). However, in many areas coastal development just above the extreme high tide line has limited or eliminated opportunities for wetland migration, a phenomenon that has been labeled "coastal squeeze" (Twilley, 1997). The maximum rate that Gulf Coast wetlands can migrate into available inland areas is unknown relative to projected changes in sea level over the next century. Nonetheless, the vulnerability of coastal resources and infrastructure to sea-level rise can be expected to increase as both human development and climate change progress (Twilley et al., 2001).

D. Water Management, Climate Change, and Wetland Restoration

In some areas, Gulf Coast wetlands have adapted thus far to major changes in hydrology and sediment transport resulting from human engineering of river basins. In other areas, wetlands are being lost because they no longer have the natural capacity to adapt to these changes. These observations demonstrate that wetland vulnerability is based on the ability of wetland systems to cope with varying rates of environmental change. With capacity for adaptation already reduced by human activities, additional climatic changes have important implications for wetland sustainability.

Many coastal restoration projects proposed for the Mississippi River Delta and the Everglades are predicated on returning many ecosystem functions to natural wetlands (CERP, 2004; U.S. Army Corps of Engineers, 2004b; Day et al., 2007). One of the hurricane protection opportunities being considered is related to the natural hydrologic functions of coastal wetlands (Working Group for Post-Hurricane Planning for the Louisiana Coast, 2006). Modification of water management systems in both the Everglades and the Mississippi River basin is being considered as a way to increase freshwater and sediment supply, respectively, to promote wetland development (Harwell, 1997; Day et al., 2007). However, wetland vulnerability to present conditions has provided the traditional context for restoration planning; this context is insufficient to assure wetland sustainability over the century-long lifetime of major restoration efforts in the face of projected sea-level rise and hurricane intensification. Forward-looking measures are required to ensure that the necessary water resources will be restored to allow wetlands to build soil sufficient to survive a changing climate.

There is still time to plan and execute large-scale coastal restoration projects for the Everglades and Mississippi delta that would be sustainable against projected climate change through the 21st century (CPRA, 2007). The long-term sustainability of coastal wetlands will have to be re-evaluated over time as coastal systems respond to restoration measures. Ultimately, sea-level rise will continue for centuries after human-induced greenhouse gases are stabilized in the atmosphere (IPCC, 2007). The benefits of coastal wetlands to society can only be secured by accounting for the long-term effects of climate change in the design of near-term restoration projects.

References

Barras, J., S. Beville, D. Britsch, S. Hartley, S. Hawes, J. Johnston, P. Kemp, Q. Kinler, A. Martucci, J. Porthouse, D. Reed, K. Roy, S. Sapkota, and J. Suhayda. 2003. *Historical and Projected Coastal Louisiana Land Changes: 1978–2050.* Open File Report 03-334. U.S. Geological Survey, Reston, VA.

Barry, J. 1997. *Rising Tide: The Great Mississippi Flood of 1927 and How It Changed America.* Simon and Schuster, New York, NY.

Baumann, R.H., J.W. Day, and C.A. Miller. 1984. Mississippi deltaic plain wetland survival: sedimentation versus coastal submergence. *Science* 224:1093-1095.

Boesch, D.F., M.N. Josselyn, A.J. Mehta, J.T. Morris, W.K. Nuttle, C.A. Simenstad, and D.J.P. Swift. 1994. Scientific assessment of coastal wetland loss, restoration and management in Louisiana. *Journal of Coastal Research* 20:1-103.

Bove, M.C., J.J. O'Brien, J.B. Elsner, C.W. Landsea, and X. Niu. 1998. Effect of El Niño on U.S. landfalling hurricanes, revisited. *Bulletin of the American Meteorological Society* 79:2477-2482.

Cahoon, D.R., D.J. Reed, J.W. Day, R.M. Boumans, J.C. Lynch, D. McNally, and N. Latif. 1995. The influence of Hurricane Andrew on sediment distribution in Louisiana coastal marshes. *Journal of Coastal Research* 18:280-294.

CENR (Committee on Environment and Natural Resources). 2000. *Integrated Assessment of Hypoxia in the Northern Gulf of Mexico.* National Science and Technology Council, Washington, DC.

CERP (Comprehensive Everglades Restoration Plan). 2004. *Comprehensive Everglades Restoration Plan Annual Report.* South Florida Water Management District, West Palm Beach, FL and Florida Department of Environmental Protection, Tallahassee, FL.

Chen, R. and R.R. Twilley. 1998. A simulation model of organic matter and nutrient accumulation in mangrove wetland soils. *Biogeochemistry* 92:1-12.

Chen, R. and R.R. Twilley. 1999. Patterns of mangrove forest structure associated with soil nutrient dynamics along the Shark River estuary. *Estuaries* 22:1027-1042.

Church, J.A. 2001. Climate change—how fast are sea levels rising? *Science* 294:802-803.

Conner, W.H. and J.W. Day, Jr. 1988. Rising water levels in coastal Louisiana: implications for two coastal forested wetland areas in Louisiana. *Journal of Coastal Research* 4:589-596.

CPRA (Coastal Protection and Restoration Authority). 2007. *Integrated Ecosystem Restoration and Hurricane Protection: Louisiana's Comprehensive Master Plan for a Sustainable Coast.* Coastal Protection and Restoration Authority of Louisiana, Baton Rouge, LA.

Daily, G.C., P.A. Matson, and P.M. Vitousek. 1997. *Ecosystem services supplied by soil. In Nature's Services: Societal Dependence on Natural Ecosystems*. G.C. Daily (ed.). Island Press, Washington, DC.

Davis, S.E. and J. Ogden. 1997. *Everglades: The Ecosystem and Its Restoration.* St. Lucie Press, Boca Raton, FL.

Davis, S.M., D.L. Childers, J.J. Lorenz, H.R. Wanless, and T.E. Hopkins. 2005. A conceptual model of ecological interactions in the mangrove estuaries of the Florida Everglades. *Wetlands* 25:832-842.

Day, J.W., Jr., C.J. Madden, R.R.Twilley, R.F. Shaw, B.A. McKee, M.J. Dagg, D.L. Childers, R.C. Raynie, and L.J. Rouse. 1994. The influence of Atchafalaya River discharge on Fourleague Bay, Louisiana (USA). In *Changes in Fluxes in Estuaries.* K.R. Dyer and R.J. Orth (eds.). Olsen and Olsen, Fredensborg, Denmark.

Day, J.W., Jr., D.F. Boesch, E.J. Clairain, G.P. Kemp, S.B. Laska, W.J. Mitsch, K. Orth, H. Mashriqui, D.R. Reed, L. Shabman, C.A. Simenstad, B.J. Streever, R.R. Twilley, C.C. Watson, J.T. Wells, and D.F. Whigham. 2007. Restoration of the Mississippi Delta: lessons from hurricanes Katrina and Rita. *Science* 315:1679-1684.

DeLaune, R.D., J.A. Nyman, and W.H. Patrick, Jr. 1994. Peat collapse, ponding, and wetland loss in a rapidly submerging coastal marsh. *Journal of Coastal Research* 10:1021-1030.

Emanuel, K. 2005. Increasing destructiveness of tropical cyclones over the past 30 years. *Nature* 436:686-688.

Energy Information Administration. 2007. Official Energy Statistics from the U.S. Government. Available online at http://www.eia.doe.gov/ (accessed August 23, 2007).

Environmental Health Center. 1998. *Coastal Challenges: A Guide to Coastal and Marine Issues.* National Safety Council, Washington, DC.

Florida Tax Watch. 2006. *The Impact of Tourism on Florida's Economy: Telling a More Complete Story.* Research Report No. 6. Florida Tax Watch, Tallahassee, FL.

Goldenberg, S., C. Landsea, A. Mestas-Nunez, and W. Gray. 2001. The recent increase in Atlantic hurricane activity: causes and implications. *Science* 293:474-479.

Gosselink, J.G. 1984. *The Ecology of Delta Marshes of Coastal Louisiana: A Community Profile.* Biological Services Program Report No. FWS/OBS-84/09. U.S. Fish and Wildlife Service, Washington, DC.

Gunderson, L.H. 1994. Vegetation of the Everglades: determinants of community composition. In *Everglades: The Ecosystem and Its Restoration*. S.M. Davis and J.C. Ogden (eds.). St. Lucie Press, Delray Beach, FL.

Harwell, M.A. 1997. Ecosystem management of South Florida. *BioScience* 47:499-512.

Harwell, M.A. 1998. Science and environmental decision-making in South Florida. *Ecological Applications* 8:580-590.

Hetherington, L.E., T.J. Brown, A.J. Benham, P.A.J. Lusty, and N.E. Idoine. 2007. *World Mineral Production, 2001–05*. British Geological Survey, Keyworth, Nottingham, United Kingdom.

IPCC (Intergovernmental Panel on Climate Change). 2007. *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change*, S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K.B. Averyt, M. Tignor, and H.L. Miller (eds.). Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Kennedy, V.S., R.R. Twilley, J.A. Kleypas, J.H. Cowan, Jr., and S.R. Hare. 2002. *Coastal and Marine Ecosystems and Global Climate Change: Potential Effects on U.S. Resources*. Pew Center on Global Climate Change, Arlington, VA.

Kesel, R.H. 1988. The decline in the suspended sediment load of the Lower Mississippi River and its influence on adjacent wetlands. *Environmental Geology and Water Science* 12:271-281.

Knutson, T.R and R.E. Tuleya. 2004. Impact of CO_2-induced warming on simulated hurricane intensity and precipitation: sensitivity to the choice of climate model and convective parameterization. *Journal of Climate* 17:3477-3495.

Knutson, T.R., R.E. Tuleya, and Y. Kurihara. 1998. Simulated increase of hurricane intensities in a CO_2-warmed climate. *Science* 279:1018-1020.

Laska, S., G. Woodell, R. Hagelman, R. Gramling, and M. Teets Farris. 2005. At risk: the human community and infrastructure resources of coastal Louisiana. *Journal of Coastal Research* 44:90-111.

Light, S.S. and J.W. Dineen. 1994. Water control in the Everglades: a historical perspective. In *Everglades: The Ecosystem and Its Restoration*. S.M. Davis and J.C. Ogden (eds.). St. Lucie Press, Delray Beach, FL.

Louisiana Coastal Wetlands Conservation and Restoration Task Force. 1998. *Coast 2050: Toward a Sustainable Coastal Louisiana.* Louisiana Department of Natural Resources, Baton Rouge, LA.

Lynch, J.C., J.H. Merriwether, B.A. McKee, F. Vera-Herrera, and R.R. Twilley. 1989. Recent accretion in mangrove ecosystems based on ^{137}Cs and ^{210}Pb methods. *Estuaries* 12:284-299.

MAB (Man and the Biosphere). 2007. Biosphere Reserves' Directory Interactive Map. UNESCO Man and the Biosphere Programme. Available online at http://www.unesco.org/mab/BRs/map.shtml.

Mann, M.E. and K. Emanuel. 2006. Atlantic Hurricane Trends Linked to Climate Change. *Eos, Transactions, American Geophysical Union* 87:233, 238, 241.

Mckee, K.L., I.A. Mendelssohn, and M.D. Materne. 2004. Acute salt marsh dieback in the Mississippi River deltaic plain: a drought-induced phenomenon? *Global Ecology and Biogeography* 13:65-73.

McPhadden, M.J., S.E. Zibiak, and M.H. Glantz. 2006. ENSO as an integrating concept in earth science. *Science* 314:1740-1745.

Mitsch W.J. and J.G. Gosselink. 2000. *Wetlands*. John Wiley & Sons, New York, NY.

Morris, J.T., P.V. Sundareshwar, C.T. Nietch, B. Kjerfve, and D.R. Cahoon. 2002. Response of coastal wetlands to rising sea level. *Ecology* 83:2869-2877.

Morton, R.A., G. Tiling, and N.F. Ferina. 2003. Causes of hot-spot wetland loss in the Mississippi delta plain. *Environmental Geosciences* 10:71-80.

National Research Council. 2005. *Drawing Louisiana's New Map: Addressing Land Loss in Coastal Louisiana.* National Academies Press, Washington, DC.

Parkinson, R.W., R.D. DeLaune, and J.R. White. 1994. Holocene sea-level rise and the fate of mangrove forests within the wider Caribbean region. *Journal of Coastal Research* 10:1077-1086.

Penland, S. and K.E. Ramsey. 1990. Relative sea-level rise in Louisiana and the Gulf of Mexico: 1908–1988. *Journal of Coastal Research* 6:323-342.

Perez, B.C., J.W. Day Jr., D. Justiç, and R.R. Twilley. 2003. Nitrogen and phosphorus transport between Fourleague Bay, Louisiana, and the Gulf of Mexico: the role of winter cold fronts and Atchafalaya River discharge. *Estuarine, Coastal and Shelf Science* 57:1065-1078.

Poff, N.L., M.M. Brinson, and J.W. Day, Jr. 2002. *Aquatic ecosystems and global climate change: potential impacts on inland freshwater and coastal wetland ecosystems in the United States.* Pew Center on Global Climate Change, Arlington, VA.

Ramsar Convention. 2007. The Ramsar List of Wetlands of International Importance. Available online at http://www.ramsar.org/key_sitelist.htm.

Reed, D.J. 1995. The response of coastal marshes to sea-level rise: survival or submergence? *Earth Surface Processes and Landforms* 20:39-48.

Reed, D.J. 2002. Sea level rise and coastal marsh sustainability: geological and ecological factors in the Mississippi Delta plain. *Geomorphology* 48:233-243.

Ross, M.S., J.F. Meeder, J.P. Sah, L.P. Ruiz, and G.J. Telesnicki. 2000. The southeast saline Everglades revisited: 50 years of coastal vegetation change. *Journal of Vegetation Science* 11:101-112.

Santer, B.D., T.M.L. Wigley, P.J. Gleckler, C. Bonfils, M.F. Wehner, K. AchutaRao, T.P. Barnett, J.S. Boyle, W. Bruggemann, M. Fiorino, N. Gillett, J.E. Hansen, P.D. Jones, S.A. Klein, G.A. Meehl, S.C.B. Raper, R.W. Reynolds, K.E. Taylor, and W.M. Washington. 2006. Forced and unforced ocean temperature changes in Atlantic and Pacific tropical cyclogenesis regions. *Proceedings of the National Academy of Sciences USA* 103:13905-13910.

Scavia, D., J.C. Field, D.F. Boesch, R.W. Buddemeier, V. Burkett, D.R. Cayan, M. Fogarty, M.A. Harwell, R.W. Howarth, C. Mason, D.J. Reed, T.C. Royer, A.H. Sallenger, and J.G. Titus. 2002. Climate change impacts on U.S. coastal and marine ecosystems. *Estuaries* 25:149-164.

Stokstad, E. 2006. Katrina study stirs debate on coastal restoration. *Science* 313:1713.

Trenberth, K. 2006. Uncertainty in hurricanes and global warming. *Science* 308:1753-1754.

Trenberth, K.E. and T.J. Hoar. 1997. El Niño and climate change. *Geophysical Research Letters* 24:3057-3060.

Trenberth, K.E. and D.J. Shea. 2006. Atlantic hurricanes and natural variability in 2005. *Geophysical Research Letters* 33:L12704

Turner, R.E., J.J. Baustian, E.M. Swenson, and J.S. Spicer. 2006. Wetland sedimentation from hurricanes Katrina and Rita. *Science* 314:449-452.

Twilley, R.R. 1997. *Mangrove wetlands.* In *Southern Forested Wetlands: Ecology and Management.* M. Messina and W. Connor (eds.). CRC Press, Boca Raton, FL, pp. 445-473.

Twilley, R.R., E. Barron, H.L. Gholz, M.A. Harwell, R.L. Miller, D.J. Reed, J.B. Rose, E. Siemann, R.G. Wetzel, and R.J. Zimmerman. 2001. *Confronting Climate Change in the Gulf Coast Region: Prospects for Sustaining Our Ecological Heritage.* Union of Concerned Scientists, Cambridge, MA and Ecological Society of America, Washington, DC.

U.S. Army Corps of Engineers. 2004a. *Louisiana Coastal Area (LCA), Louisiana Ecosystem Restoration Study.* U.S. Army Corps of Engineers, New Orleans, LA. Available online at http://data.lca.gov/Ivan6/main/main_report_all.pdf.

U.S. Army Corps of Engineers. 2004b. *Louisiana Coastal Area (LCA), Louisiana Ecosystem Restoration Study. Appendix A. Science and Technology Program.* U.S. Army Corps of Engineers, New Orleans, LA. Available online: http://data.lca.gov/Ivan6/app/app_a.pdf.

U.S. Army Corps of Engineers. 2006. CY 2004 Waterborne Tonnage by State (In Units of 1000 Tons). U.S. Army Corps of Engineers, Institute for Water Resources, Navigation Data Center, Waterborne Commerce Statistics Center, New Orleans, LA. Available online at http://www.iwr.usace.army.mil/ndc/wcsc/stateton04.htm.

Wanless, H.R., R.W. Parkinson, and L.P. Tedesco. 1994. Sea level control on stability of Everglades wetlands. In *Everglades: The Ecosystem and Its Restoration*. S.M. Davis and J.C. Ogden (eds.). St. Lucie Press, Delray Beach, FL, pp. 199-223.

Webster, P., G.J. Holland, J.A. Curry, and H.R. Chang. 2005. Changes in tropical cyclone number, duration, and intensity in a warming environment. *Science* 309:1844-1846.

Working Group for Post-Hurricane Planning for the Louisiana Coast. 2006. *A New Framework for Planning the Future of Coastal Louisiana after the Hurricanes of 2005.* University of Maryland Center for Environmental Science, Cambridge, MD. Available online at http://www.clear.lsu.edu/clear/web-content/Web_items/NewFramework_Boesch.pdf.

World Heritage Committee. 2007. UNESCO World Heritage Site List. Available online at http://whc.unesco.org/.

V. Ramifications of Climate Change for Chesapeake Bay Hypoxia

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A. Introduction

Climate change is likely to significantly complicate the achievement of environmental management objectives that presently command public attention and significant commitment of resources. This is particularly the case for coastal environments that are subject to numerous societal uses and pressures from human activities but also to concerted efforts to restore their ecological health and productivity. The Chesapeake Bay is a global model for such large-scale ecosystem restoration.

The Chesapeake Bay is the United States' largest and best-studied estuary. The bay is more than 190 miles long and its tidal waters cover more than 4,200 square miles. Its 64,000-square-mile watershed extends over six states and the District of Columbia and includes a population of approximately 16 million people. The Chesapeake is situated along the transition between warm-temperate and cool-temperate regions and is influenced both by freshwater runoff and by the Atlantic Ocean. Consequently, the Chesapeake ecosystem has experienced substantial climatic variability over 4,000 years in its present geographic configuration.

Humans had begun altering the Chesapeake Bay ecosystem even before the arrival of Europeans; however, pervasive human effects became obvious only during the late 20th century. In particular, eutrophication—the increase in organic matter loading due principally to inputs of nitrogen and phosphorus nutrients—has been recognized as the chief cause of degradation of the ecosystem and, consequently, has been the central focus of restoration efforts (Boesch et al., 2001; Kemp et al., 2005).

Eutrophication has manifold consequences in coastal ecosystems, including increased production of phytoplankton, including harmful or noxious algal blooms; decreased water clarity, resulting in loss of seagrasses; altered food chains; and severe depletion of dissolved oxygen in the water column (Cloern, 2001). Particularly during the summer, dissolved oxygen can fall to very low levels in denser bottom waters that are isolated from the warmer surface waters (the source of oxygen replenishment). Depletion of dissolved oxygen (hypoxia) to levels that exclude fish, crustaceans, and mollusks, or even the complete absence of oxygen (anoxia), is a phenomenon that has increased in coastal waters around the world (Diaz and Rosenberg, 1995). Commonly referred to as "dead zones," these expanding hypoxic regions have attracted wide attention from the public and policy-makers (Dybas, 2005).

In the late 1980s, a concerted effort to reduce nutrient pollution in the Chesapeake Bay was initiated through the multistate–federal Chesapeake Bay Program. The present goal of the program is to reduce nutrient inputs sufficiently to restore water quality, including healthy dissolved oxygen conditions, by 2010. While the cost to society of the degradation of the Chesapeake Bay ecosystem is difficult to quantify fully, it is estimated that the cost of restoration, largely driven by stringent requirements to reduce hypoxia, exceeds $15 billion (Chesapeake Bay Watershed Blue Ribbon Finance Panel, 2004). Despite already substantial public and private expenditures, reports of record-sized hypoxic zones in 2003 and 2005 raised public concerns about whether progress is really being made. Hypoxia in the Chesapeake Bay, and in most other regions experiencing this phenomenon, is greatly affected by climate, as well as by nutrient inputs from human activities. Indeed, climatic conditions, including some combination of high river inflows, warm temperatures, and relatively calm summer winds, were major factors in the extensive hypoxia that occurred in 2003 and 2005.

This case study examines how both climate variability and potential climate change can affect hypoxia in the Chesapeake Bay and can present additional challenges to ongoing ecosystem restoration. We use past observations to elucidate the multiple influences of climate on hypoxia and its consequences to the ecosystem. Using this empirical basis, we project how climate change during the rest of the 21st century is likely to affect hypoxia, and how climate change will challenge the achievement of restoration goals.

B. Climate Variability and Hypoxia

Recent variations in the climate of the Chesapeake Bay watershed have included a dry period in the 1960s, a wet period in the 1970s, and a period of unusually large inter-annual variability over the past 25 years (Boesch et al., 2001). Multi-year climate cycles such as the El Niño-Southern Oscillation and the North Atlantic Oscillation influence these regional climate variations (Austin, 2002), as do more localized weather patterns and storms. Variations in precipitation and temperature affect the amount and timing of fresh water flowing into the Chesapeake Bay from the Susquehanna, Potomac, James, and other rivers. These inflows influence hypoxia in the bay by delivering nutrients that stimulate biological production and contribute to the density stratification of the estuarine waters (Hagy et al., 2004). Winds also play a role by forcing denser ocean waters into the bay or by vertically mixing the water column in the estuary. For example, the volume of hypoxic water during the summer of 2005 was particularly large in part because summer winds were weaker than normal, allowing bay waters to remain strongly stratified.

Hypoxia thus has both natural and human causes and has occurred at some level in the Chesapeake Bay for more than 2,500 years (Cooper and Brush, 1993). However, hypoxia in the bottom waters of the mainstem bay has become more frequent, widespread, and severe since the 1960s (Hagy et al., 2004). The natural factors that make the bay susceptible to oxygen depletion include its deep central channel, which acts as a basin to contain the dense, low-oxygen waters; the bay's high ratio of watershed area to volume, leading to large nutrient exports from the watershed into a limited volume of receiving water; and high variability of freshwater flow (Kemp et al., 2005). Anthropogenic causes are largely related to the greatly increased nutrient loading that has occurred since the mid-20th century (Boynton et al., 1995; Harding and Perry, 1997). The higher nutrient levels increase phytoplankton biomass, particularly in the spring. The increase in impervious surface area on the landscape (e.g., from roads and other development) and other land use changes may also affect the volume and timing of freshwater runoff (Jennings and Jarnagin, 2002; Jones et al., 2001). Because nutrient loading to the bay is closely tied to freshwater input, the interaction between climate and anthropogenic nutrient loading will be particularly important in determining future hypoxic events in the Chesapeake.

Freshwater flow into the Chesapeake Bay is typically greatest during the spring. This spring freshet—a freshwater pulse resulting partly from snowmelt—delivers sediment and nutrients that act in concert to control the timing, position, and magnitude of a spring phytoplankton bloom—light limitation controls phytoplankton in the upper estuary (closer to the Susquehanna River) and nutrient stimulation enhances it in the middle to lower estuary (closer to the ocean) (Harding, 1994). The freshet is, to a large degree, controlled by the winter weather (Miller et al., 2006). Drier than normal winters mean very little precipitation is stored in the form of snow in the upper watershed, resulting in less spring runoff. Conversely, wet winters that have high frequencies of storm events result in more snow stored in the watershed and later released to the estuary as snowmelt runoff in the spring (Najjar, 1999). The organic matter produced during the spring bloom is retained particularly in the middle reaches of the bay, as the estuarine circulation produces a net return flow of bottom waters (Boicourt, 1993). The settling material is eventually decomposed in the bottom layer by microbes that consume oxygen in proportion to available organic matter and thus cause hypoxic conditions in waters deeper than about 30 feet (10 m; Kemp et al., 1992). Because fresh water is less dense than salt water, the freshet also increases water-column stratification, preventing the resupply of oxygen from the surface.

The resulting hypoxia affects the cycling of nutrients and other materials in the ecosystem, causes stress and mortality in biota, and changes interactions between predators and their prey, thus impairing normal ecosystem function (Breitburg et al., 1997). Small zooplankton swim upward to avoid low oxygen. They may also be subject to increased predation by jellyfish, which are more tolerant of low oxygen than are other predators. Changes in zooplankton biomass and behavior may reduce key prey for larval fish that use the estuary as a nursery. Benthic (bottom dwelling) organisms are especially vulnerable to hypoxia, as they are unable to flee low-oxygen conditions (Diaz and Rosenberg, 1995). Other animals alter their customary behavior—for example, blue crabs (*Callinectes sapidus*) may alter their migration routes to lower-bay spawning areas. Striped bass (*Morone saxatilis*) experience severe habitat restriction and physiological stress in summer as they try to avoid both the high temperatures of surface waters and low oxygen of bottom waters (Coutant, 1985). Therefore, by driving hypoxia, nutrient pollution—as modulated by climate variability—affects commercially and recreationally important fisheries in the Chesapeake Bay (Houde and Rutherford, 1993).

Changes in nutrient inputs, combined with variability in freshwater flow, have produced large inter-annual variability in the spatial extent and volume of hypoxic water in the Chesapeake Bay, as revealed by records extending from the 1950s (Hagy et al., 2004). Understanding these highly variable records provides insights into possible ecosystem responses to future climate change. A wetter climate would likely result in enhanced phytoplankton production that extends farther down the bay, providing more organic matter to fuel summer hypoxia. A drier climate would likely be characterized by smaller input of nutrients, reduced phytoplankton production, and blooms confined to the upper estuary. The location, timing, and magnitude of the spring bloom, and its subsequent degradation, all combine to affect the severity of summer hypoxia.

C. Modeling of Future Climate Change and Ecosystem Consequences

Projecting the ecosystem response to potential climate change requires an understanding of how precipitation, river runoff, sea level, temperature, and wind will vary and interact with biological processes in the future. These multiple drivers and their relationship to hypoxia can vary considerably (Table 1). Effects of some drivers are relatively direct; for example, increased runoff would likely exacerbate hypoxia. For more complex drivers, it is sometimes difficult to predict the direction, much less the magnitude, of their effect on hypoxia. For example, warmer temperatures could expand agricultural production, increasing nutrient

Table 1

The Influence of Multiple Climate Drivers on the Extent and Severity of Hypoxia in the Chesapeake Bay

Climate Driver	Direct Effect	Secondary Effect	Influence on Hypoxia
Increased temperature	More evapotranspiration	Decreased streamflow	+
		Land-use and cover changes	+/–
	Less snow cover	More nitrogen retention	–
	Warmer bay temperature	Stronger bay stratification	+
		Higher metabolic rates	+
More precipitation	More streamflow	Stronger bay stratification	+
		More nutrient loading	+
	More extreme rainfall	Greater erosion of soil P	+
Less precipitation	Less streamflow	Weaker bay stratification	–
		Less nutrient loading	–
Higher sea level	Greater bay depth/volume	Stronger bay stratification	+
		Greater bottom water volume	–
		Less hydraulic mixing	+
	Less tidal marsh	Diminished nutrient trapping	+
Weaker summer wind	Less water column mixing	More persistent stratification	+
Stronger summer wind	More water column mixing	Less persistent stratification	–

runoff, causing increased algal blooms and greater hypoxia. Conversely, reduced soil moisture during the summer could force abandonment of some crops or increase the use of irrigation. Such complex interactions occur on spatial and temporal scales smaller and shorter than can be resolved by the global climate simulation models used to forecast climate changes. Furthermore, necessary simplifications as well as incompletely understood physical feedbacks contribute to uncertainties in the models used to project future climate variability and change. However, newer models are producing results that are increasingly consistent with observations of recent climate trends (DeGaetano and Allen, 2002), inspiring greater confidence in model results, especially regarding temperature projections.

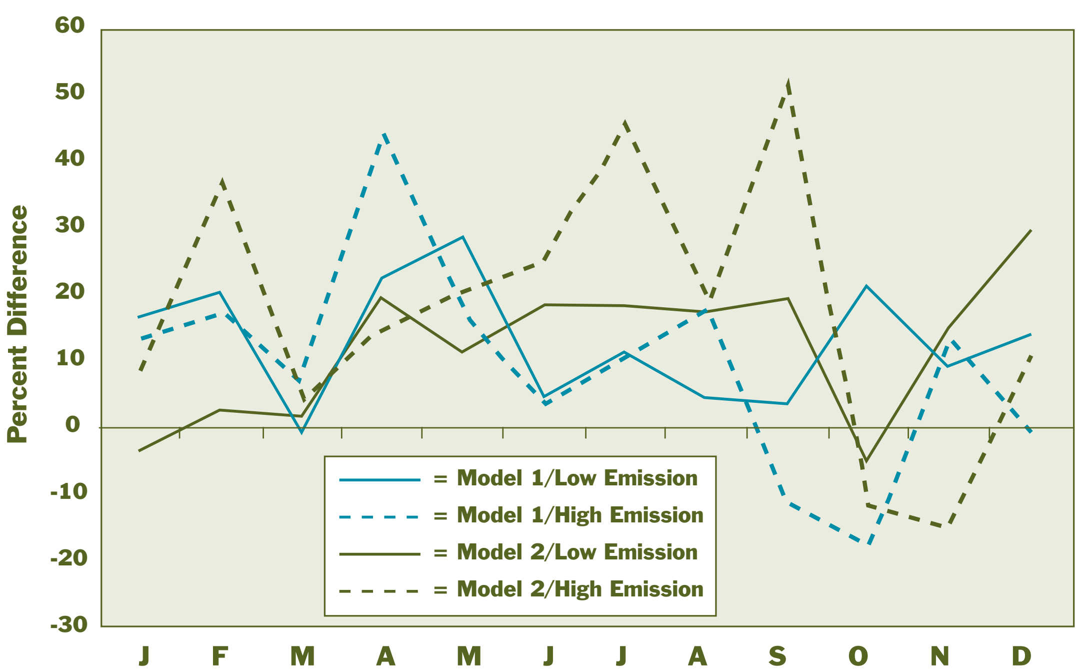
Assessments of climate change impacts in the Mid-Atlantic region, conducted as part of the U.S. National Assessment of Consequences of Climate Variability and Change (Fisher et al., 2000), relied on an earlier generation of coupled ocean-atmosphere general circulation models, specifically the then-available versions of models from the U.K. Hadley and Canadian climate centers. Using these models, Najjar et al. (2000) projected that spring streamflow in the Susquehanna River could change by +12 percent to –4 percent by 2030 and +4 percent to –25 percent by 2095. Based on the 2030 projections, they estimated that average hypoxic volume in the Chesapeake Bay could increase as much as 31 percent or decrease by 10 percent. Earlier, Najjar (1999) used geographically downscaled projections from a version of the GENESIS general circulation model to project an increase in streamflow down the Susquehanna River of 24 ± 13 percent under a doubling of atmospheric carbon dioxide (CO_2).

The newest generation of climate models has improved both spatial resolution and large-scale heat balances. These models no longer require adjustments to match observations as did earlier models (Bader, 2004). On average, the latest models project an increase in annual precipitation for the East Coast of the U.S., but with regional uncertainty (Christensen, 2007). Although applying newer models to project streamflow is beyond the scope of this brief review, it is instructive to examine whether the newer models might change or sharpen earlier projections for future streamflow. A recent high-resolution model covering the continental United States projects only small differences in the degree of change within the Chesapeake Bay watershed for current-generation and earlier models (Diffenbaugh et al., 2005). Thus we examined results for the generalized Chesapeake region from the U.S. Community Climate System Model (CCSM3) and a newer version of the U.K. Hadley Centre for Climate Prediction and Research model (HadCM3) for a range of possible greenhouse gas forcing scenarios.

The projected changes in average monthly precipitation for the Chesapeake Bay region by the end of the 21st century are shown in Figure 1 for two scenarios used in Intergovernmental Panel on Climate Change (IPCC) assessments: the A1B (reversing the growth of greenhouse gases by mid century) and A2 (unrestrained growth in greenhouse concentrations throughout the century). In general, the two models agreed in their projection of more precipitation during most of the year, except during fall, when some modeled scenarios projected decreased precipitation. As would be expected, the more rapidly warming A2 scenario produced wider ranges in precipitation, with increases greater than 30 percent in some months and decreases greater than 10 percent in the fall. One should bear in mind that in a warmer world, increased losses due to evapotranspiration would be expected to moderate the effects of increased precipitation on streamflow, particularly during the summer. Presently, two-thirds of all precipitation returns to the atmosphere via the combined effects of surface evaporation and plant-mediated soil drying (Neff et al., 2000).

Figure 1

Projected **Changes in Average Precipitation** for the Chesapeake Bay Region



Projections of the changes in average monthly precipitation for the Chesapeake Bay region by the end of the twenty-first century, based on two different climate models (Model 1=HadCM3; Model 2=CCSM3) and two IPCC greenhouse gas emission scenarios (Low emissions=SRES A1B; High emissions=SRES A2). In SRES A1B the growth of emissions reverses by mid century, whereas in SRES A2 emissions growth is unrestrained throughout the century. Both scenarios assume that no steps are taken to reduce greenhouse gas emissions in order to limit climate change.

Although these results await detailed hydrologic modeling, their implications for inflows to the Chesapeake Bay can be summarized as (1) increased inflows during winter, due to increased precipitation and less storage as snow; (2) somewhat increased runoff during spring, but without a pronounced freshet from snowmelt, (3) inflows during summer generally similar to the present; and (4) possibly decreased streamflow during fall. In general, moderate increases (in the range of 10–15 percent) in delivery of fresh water and, potentially, nutrients from nonpoint sources should be expected. Najjar et al.'s (2000) results suggest a wider range of possible changes in precipitation and inflows (because of the inclusion of a Canadian Climate Centre model that produced hotter and drier projections). The newer models are more in agreement with the other model they used as well as with the fine-scale model of Diffenbaugh et al. (2005), which also found modest increases in average rainfall and in extreme rainfall frequency in the Chesapeake watershed for both the A2 and A1B scenarios. Similar projections also appear in a more recent assessment for the northeastern U.S. (Hayhoe et al., 2007).

There is generally a greater degree of confidence in projections of temperature than of precipitation. Both the CCSM3 and HadCM3 models project greater warming of air temperature for the Chesapeake Bay region (3–4.5 °C by the end of the century based on the A1B and A2 scenarios) than for the global averages for those models. Both models predict the greatest warming to occur during summer, with maximum increases ranging from 3.5 to 6.5 ºC and an increase in extremely warm days, clustered in the summer months, under conditions of modest winds. The timing of this warming is significant not only because it would increase evapotranspiration and decrease soil moisture, but also because it would result in warmer water temperatures in the bay during the time that hypoxia is most prominent.

Greater and earlier warming of the bay would have multiple effects on hypoxia. First, higher temperatures would reduce the amount of oxygen that can be dissolved in the water, leading to lower overall oxygen content that would be depleted by respiration of biota. Observations from past years with similar freshwater discharges suggest progressively earlier onset of hypoxia when the deep-water column warms early (Hagy et al., 2004). Second, warmer summertime air temperatures would enhance the stratification (and thereby reduce the exchange) between the warmer surface waters and cooler deep waters. Third, both photosynthesis and respiration are temperature-dependent processes and thus the rates of production, decomposition, and nutrient cycling would likely increase under warmer conditions. Although as much as 70 percent of the variance in the extent of hypoxia is explained by springtime runoff, a significant fraction of the remaining variability is due to whether summertime weather conditions are conducive either to stratification or to wind mixing and oxygenation of the water column. Thus, increased summertime temperatures, especially if coincident with reduced winds, would lead to more persistent stratification and the expansion of hypoxia into shallower areas of the bay.

Chesapeake Bay hypoxia may also respond to accelerated sea-level rise resulting from global warming. Locally experienced sea-level rise is also partially due to land subsidence resulting from the post-glacial rebound of regions to the north, as well as other local effects such as groundwater withdrawal. Coupling regional subsidence with IPCC Third Assessment projections of global sea-level rise, Wood et al. (2002) projected an increase of relative sea levels for the Chesapeake Bay region of 38–87 cm (15–34 inches) by the last decade in the 21st century. Assuming a central estimate of approximately 60 cm (24 inches), this increase is twice the locally observed rise in sea level during the 20th century and would increase the volume of the bay by 9 percent, unless counteracted by the increased infilling of the bay with sediment (Cronin et al., 2003).

Sea-level rise would have two potentially competing effects on the volume and duration of hypoxic conditions. As the depth of the Chesapeake Bay increases, the proportional volume of ocean waters filling the bay would also increase without compensatory increases in freshwater flow. This would allow salty bottom waters to penetrate farther up the bay, thus increasing stratification and hypoxia. Assuming that the depth of the discontinuity between the less dense surface water and the denser water below would remain the same, it would also increase the volume of bottom water from which oxygen would have to be depleted to generate hypoxia. Changes in the circulation in the bay could occur, as increasing water depth reduces the effect of the sill that lies off Rappahannock Spit in the lower Bay. This hydraulic control point currently enhances vertical mixing (Chao and Paluskiewicz, 1991). Reduction in mixing would further isolate the salty bottom water from the upper layer and reinforce stratification over a broader region of the bay.

Although some clues to the changes in hypoxia that might occur under climate change can be gleaned from the responses to inter-annual variations discussed earlier, the Chesapeake Bay ecosystem is physically and biologically complex and therefore somewhat unpredictable. Prolonged shifts in climate and its variability, or in the biota inhabiting the bay, may have unprecedented effects that drive the ecosystem to a new state. Such a change in state may have already occurred during the late 20th century—Hagy et al. (2004) could ascribe only part of the increases in hypoxic volume to enhanced nutrient loading, suggesting that some reduction in the resilience of the ecosystem over time, such as a further reduction in filter feeders (e.g., oysters), may have occurred.

D. Management Implications

Although the Chesapeake 2000 Agreement (Chesapeake Bay Program, 1999) expanded the multiple objectives of the Chesapeake Bay Program, the program's central focus remains alleviating hypoxia and other undesirable effects of eutrophication through the significant reduction of nutrient inputs. New nutrient reduction goals for 2010 were based on inverse computer models, essentially "back calculations" that predict the nutrient load reductions necessary to return water quality to levels needed to support living resources. These water quality objectives were determined based on known biological requirements for oxygen and light in various depth zones along the bay and its major tributaries. The Chesapeake Bay Program has estimated that on a bay-wide basis, reductions of 48 percent and 53 percent are required for nitrogen and phosphorus, respectively (derived from a 1985 baseline load; Koroncai et al., 2003). The most demanding requirement for these new targets is the load reductions needed to reduce hypoxia in the central trough of the bay to levels more typical of the mid-20th century.

Estimates of nutrient inputs to the bay indicate that some reduction has been achieved, but confidence in these estimates is low. Watershed models have been used to track estimated load reductions based on the management actions taken and assumptions about their effectiveness. However, the representation of such virtual accounting as a measure of progress has been sharply criticized in a recent Government Accountability Office (2005) report, which emphasizes the need for real-world measurements and integrated assessments of progress. One such measure of progress is the change in loadings of nutrients from the major rivers discharging to the bay. However, the results of such monitoring are also difficult to evaluate, in large part because of the climatic variability that affects the amount and timing of freshwater discharges. Flow-adjustment techniques used to compare concentrations over time show statistically significant downward trends in nutrient concentrations for many of the major rivers, including the Susquehanna (Langland et al., 2004), but these results often do not match well with watershed model projections. Because climatic variations also affect the processes creating, maintaining, and dissipating hypoxia in the bay itself, these watershed and estuarine processes conspire to create variability that has made it difficult to see much improvement in hypoxia in the bay at present (Chesapeake Bay Program, 2006).

Climate change is likely to affect hypoxia in myriad ways, sometimes with opposing results. In addition to changes to the physical drivers of freshwater discharge, temperature, water depth, and winds, processes in the watershed that govern the delivery of nutrients and sediment are likely to change. Climate-influenced changes in forests, land uses, and agricultural practices will surely occur. Reduction in snow cover could result in less runoff of atmospherically deposited nitrogen during snow melt and more retention within forests. Increases in extreme rainfall events may cause more phosphorus delivery as a result of soil erosion. Other important changes in the estuary itself include the probable reduction in tidal wetlands (which serve as important nutrient traps) due to sea-level rise.

While we lack the full understanding needed to integrate all of these factors into a reliable projection of future hypoxic conditions in the Chesapeake Bay, many of the anticipated changes (increased streamflow, warmer temperatures, calmer summer winds, and increased depth due to sea-level rise) would move the ecosystem in the direction of worsening hypoxia. This conclusion is consistent with the simulations of hypoxia in the Gulf of Mexico performed under climate change scenarios (Justiç et al., 2003). If the bay does face these anticipated changes, nutrient loads would have to be reduced further—beyond current targets—to meet the water quality objectives needed to support living resources. Given the long lag times, both in terms of implementation of nutrient control strategies and in ecosystem response, it is not too early to begin assessing the implications of climate change on management objectives for hypoxia and for Chesapeake Bay restoration in general. At a minimum, the linked watershed and estuarine models used to determine nutrient load reduction targets should be run using reasonable assumptions for a range of mid-21st century streamflows, temperatures, and estuarine volume. This update would provide an estimate of the sensitivity of management objectives related to the alleviation of hypoxia to climate change.

References

Austin H.M. 2002. Decadal oscillations and regime shifts, a characterization of the Chesapeake Bay marine climate. *American Fisheries Society Symposium* 32:155-170.

Bader, D. (ed.). 2004. *An Appraisal of Coupled Climate Model Simulations.* Report UCRL-TR-202550. Lawrence Livermore National Laboratory, Livermore, CA.

Boesch, D.F., R.B. Brinsfield, and R.E. Magnien. 2001. Chesapeake Bay eutrophication: scientific understanding, ecosystem restoration, and challenges for agriculture. *Journal of Environmental Quality* 30:303-320.

Boicourt, W.C. 1993. Estuaries—where the river meets the sea. *Oceanus* 36:29-37.

Boynton, W.R., J.H. Garber, R. Summers, and W.M. Kemp. 1995. Inputs, transformations, and transport of nitrogen and phosphorus in Chesapeake Bay and selected tributaries. *Estuaries* 18:285-314.

Breitburg, D.L., T. Loher, C.A. Pacey, and A. Gerstein. 1997. Varying effects of low dissolved oxygen on trophic interactions in an estuarine food web. *Ecological Monographs* 67:489-507.

Chao, S.Y. and T. Paluszkiewicz. 1991. The hydraulics of density currents over estuarine sills. *Journal of Geophysical Research* 96:7065-7076.

Chesapeake Bay Program. 1999. *Chesapeake 2000: A Watershed Partnership Agreement.* U.S. Environmental Protection Agency, Chesapeake Bay Program Office, Annapolis, MD.

Chesapeake Bay Program. 2006. *Chesapeake Bay 2005 Health and Restoration Assessment Part One: Ecosystem Health.* Chesapeake Bay Program, Annapolis, MD.

Chesapeake Bay Watershed Blue Ribbon Finance Panel. 2004. *Saving a National Treasure: Financing the Cleanup of the Chesapeake Bay.* Chesapeake Bay Program, Annapolis, MD.

Christensen, J.H., B. Hewitson, A. Busuioc, A. Chen, X. Gao, I. Held, R. Jones, R.K. Kolli, W.T. Kwon, R. Laprise, V. Magaña Rueda, L. Mearns, C.G. Menéndez, J. Räisänen, A. Rinke, A. Sarr, and P. Whetton. 2007. Regional Climate Projections. In *Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change.* S. Solomon, D. Qin, M. Manning, Z. Chen, M. Marquis, K. B. Averyt, M.Tignor, and H. L. Miller (eds.), Cambridge University Press, Cambridge, United Kingdom and New York, NY, USA.

Cloern, J.E. 2001. Our evolving conceptual model of the coastal eutrophication problem. *Marine Ecology Progress Series* 210:223-253.

Cooper, S.R. and G.S. Brush. 1993. A 2,500-year history of anoxia and eutrophication in Chesapeake Bay. *Estuaries* 16:617-626.

Coutant, C.C. 1985. Striped bass, temperature, and dissolved oxygen: a speculative hypothesis for environmental risk. *Transactions of the American Fisheries Society* 114:31-61.

Cronin, T., L. Sanford, M. Langland, D. Willard, and C. Saenger. 2003. Estuarine sediment transport, deposition and sedimentation. In *A Summary Report of Sedimentary Processes in Chesapeake Bay and Watershed.* M. Langland and T. Cronin (eds.). Water-Resources Investigations Report 03-4123. U.S. Geological Survey, New Cumberland, PA.

DeGaetano, A.T. and R.T. Allen. 2002. Trends in twentieth-century temperature extremes across the United States. *Journal of Climate* 15:3188-3205.

Diaz, R.J. and R. Rosenberg. 1995. Marine benthic hypoxia: a review of its ecological effects and the behavioural responses of benthic macrofauna. *Oceanography and Marine Biology: An Annual Review* 33:245-303.

Diffenbaugh, N.S., J.S. Pal, R.J. Trapp, and F. Giorgi. 2005. Fine-scale processes regulate the response of extreme events to global climate change. *Proceedings of the National Academy of Sciences USA* 102:15774-15778.

Dybas, C.L. 2005. Dead zones spreading in world oceans. *BioScience* 55:552-557.

Fisher, A., R. Neff, and E.J. Barron. 2000. The Mid-Atlantic Regional Assessment: motivation and approach. *Climate Research* 14:153-159.

Government Accountability Office. 2005. *Chesapeake Bay Program: Improved Strategies Are Needed to Better Assess, Report, and Manage Restoration Progress.* Government Accountability Office, Washington, DC.

Hagy, J.D., W.R. Boynton, C.W. Keefe, and K.V. Wood. 2004. Hypoxia in Chesapeake Bay, 1950–2001: long-term change in relation to nutrient loading and river flow. *Estuaries* 27:634-658.

Harding, L.W. 1994. Long-term trends in the distribution of phytoplankton in Chesapeake Bay: roles of light, nutrients and streamflow. *Marine Ecology Progress Series* 104:267-291.

Harding, L.W. and E.S. Perry. 1997. Long-term increase of phytoplankton biomass in Chesapeake Bay, 1950–1994. *Marine Ecology Progress Series* 157:39-52.

Hayhoe, K., C.P. Wake, T.G. Huntington, L. Luo, M.D. Schwartz, J. Sheffield, E. Wood, B. Anderson, J. Bradbury, A. DeGaetano, T.J. Troy, and D. Wolfe. 2007. Past and future changes in climate and hydologic indicators in the US Northeast. *Climate Dynamics* 28:381-407.

Houde, E.D. and E.S. Rutherford. 1993. Recent trends in estuarine fisheries: predictions of fish production and yield. *Estuaries* 16:161-176.

Jennings, D.B. and S.T. Jarnagin. 2002. Changes in anthropogenic impervious surfaces, precipitation and daily streamflow discharge: a historical perspective in a mid-Atlantic subwatershed. *Landscape Ecology* 17:471-489.

Jones, K.B., A.C. Neale, M.S. Nash, R.D. Van Remortel, J.D. Wickham, K.H. Riitters, and R.V. O'Neill. 2001. Predicting nutrient and sediment loadings to streams from landscape metrics. *Landscape Ecology* 16:301-312.

Justiç, D., N.N. Rabalais, and R.E. Turner. 2003. Simulated responses of the Gulf of Mexico hypoxia to variations in climate and anthropogenic nutrient loading. *Journal of Marine Systems* 42:115-126.

Kemp, W.M., P.A. Sampou, J. Garber, J. Tuttle, and W.R. Boynton. 1992. Seasonal depletion of oxygen from bottom waters of Chesapeake Bay—roles of benthic and planktonic respiration and physical exchange processes. *Marine Ecology Progress Series* 85:137-152.

Kemp, W.M., W.R. Boynton, J.E. Adolf, D.F. Boesch, W.C. Boicourt, G. Brush, J.C. Cornwell, T.R. Fisher, P.M. Glibert, J.D. Hagy, L.W. Harding, E.D. Houde, D.G. Kimmel, W.D. Miller, R.I.E. Newell, M.R. Roman, E.M. Smith, and J.C. Stevenson. 2005. Eutrophication of Chesapeake Bay: historical trends and ecological interactions. *Marine Ecology Progress Series* 303:1-29.

Koroncai, R., L. Linker, J. Sweeney, and R. Batiuk. 2003. *Setting and Allocating the Chesapeake Bay Nutrient and Sediment Loads: The Collaborative Process, Technical Tools, and Innovative Approaches.* U.S. Environmental Protection Agency, Chesapeake Bay Program Office, Annapolis, MD.

Langland, M.J., S.W. Phillips, J.R. Raffensperger, and D. Moyer. 2004. *Changes in Streamflow and Water Quality in Selected Nontidal Sites in the Chesapeake Bay Basin, 1985–2003.* Scientific Investigations Report 2004-5259. U.S. Geological Survey, Reston, VA.

Miller, W.D., D.G. Kimmel, and L.W. Harding. 2006. Predicting spring freshwater flow from synoptic-scale weather patterns for the Susquehanna River basin. *Water Resources Research* 42:W05414, doi:101029/2005WR004270.

Najjar, R.G. 1999. The water balance of the Susquehanna River basin and its response to climate change. *Journal of Hydrology* 219:7-19.

Najjar, R.G., H.A. Walker, P.J. Anderson, E.J. Barron, R.J. Bord, J.R. Gibson, V.S. Kennedy, C.G. Knight, J.P. Megonigal, R.E. O'Conner, C.D. Polsky, N.P. Psuty, B.A. Richards, L.G. Sorenson, E.M. Steele, and R.S. Swanson. 2000. The potential impacts of climate change on the mid-Atlantic coastal region. *Climate Research* 14:219-233.

Neff, R., J. Chang, C.G. Knight, R.G. Najjar, B. Yarnal, and H.A. Walker. 2000. Impact of climate variation and change on Mid-Atlantic region hydrology and water resources. *Climate Research* 14:207-218.

Wood, R.J., D.F. Boesch, and V.S. Kennedy. 2002. Future consequences of climate change for the Chesapeake Bay ecosystem and its fisheries. *American Fisheries Society Symposium* 32:171-184.

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