JOURNAL OF THE AMERICAN WATER RESOURCES ASSOCIATION
VOL. 36, NO. 2 AMERICAN WATER RESOURCES ASSOCIATION APRIL 2000

POTENTIAL EFFECTS OF CLIMATE CHANGE ON SURFACE-WATER QUALITY IN NORTH AMERICA[1]

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ABSTRACT: Data from long-term ecosystem monitoring and research stations in North America and results of simulations made with interpretive models indicate that changes in climate (precipitation and temperature) can have a significant effect on the quality of surface waters. Changes in water quality during storms, snowmelt, and periods of elevated air temperature or drought can cause conditions that exceed thresholds of ecosystem tolerance and, thus, lead to water-quality degradation. If warming and changes in available moisture occur, water-quality changes will likely first occur during episodes of climate-induced stress, and in ecosystems where the factors controlling water quality are sensitive to climate variability. Continued climate stress would increase the frequency with which ecosystem thresholds are exceeded and thus lead to chronic water-quality changes. Management strategies in a warmer climate will therefore be needed that are based on local ecological thresholds rather than annual median condition. Changes in land use alter biological, physical, and chemical processes in watersheds and thus significantly alter the quality of adjacent surface waters; these direct human-caused changes complicate the interpretation of water-quality changes resulting from changes in climate, and can be both mitigated and exacerbated by climate change. A rigorous strategy for integrated, long-term monitoring of the ecological and human factors that control water quality is necessary to differentiate between actual and perceived climate effects, and to track the effectiveness of our environmental policies.
(KEY TERMS: global water change; water quality; ecological thresholds; droughts; floods.)

INTRODUCTION

Freshwater and estuarine ecosystems throughout the world are vital to the socioeconomic status, public health, quality of life, and environmental sustainability of nations. Aquatic ecosystems not only provide food and water for human consumption; they maintain and improve water quality by filtering out, storing, and converting contaminants, provide transportation corridors, replenish farmland nutrients through periodic flooding, support habitats and food for wildlife, and provide recreational opportunities.

Greater than one-third of the Earth's accessible freshwater that drains from land to ocean is currently used for human needs, resulting in a profound influence on water quality (Postel *et al.*, 1996). Point-source discharges (pollutants entering surface waters at a definable location) and non-point source discharges (pollutants entering surface waters in a dispersed manner) generated from human activities add large quantities of nutrients, pathogens, and toxins to freshwaters (Mueller *et al.*, 1995). The resulting eutrophication and toxicity in surface waters has produced both undesirable ecological consequences and increased costs for treating the water for certain uses (Carpenter *et al.*, 1998). These problems are often exacerbated by consumptive water uses such as irrigation and domestic water supplies, as the consumption reduces in-stream flows and thus concentrates pollutants introduced by urban, domestic, agricultural, and industrial sources (Jacoby, 1990).

Global warming, changes in precipitation patterns, and changes in resource use by humans in response to climate change could also significantly alter the quality of surface waters. While there is abundant evidence for dramatic and rapid climate warming in the past from lake-sediment, paleo-hydrology, and tree-ring records (Fritz, 1996; Ely *et al.*, 1993; Woodhouse and Overpeck, 1998), never before have climate changes coincided with large-scale landscape fragmentation and alteration as is occurring today (Dale, 1997). Adverse changes to aquatic ecosystems

[1]Paper No. 99097 of the *Journal of the American Water Resources Association.* **Discussions are open until December 1, 2000.**
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predicted to result from changes in climate include increases in floods and droughts, degradation of drinking-water quality, loss of biodiversity, increased transmittal of infectious diseases, loss of food supplies, and costly remedial actions (Nielson and Lee, 1987; Watson *et al.,* 1996; McKnight *et al.,* 1996; Cushing, 1997). While these climate-induced changes in water quality may be minor in comparison to land-use effects in many regions of North America (Howarth *et al.,* 1996), significant changes in water quality could occur as a result of the superposition of climate change on the effects of human resource use.

The large and increasing effects of local land- and water-use on surface and ground waters make deciphering the potential effects of climate change on water quality difficult. Where land- or water-use changes are the result of changes in climatic condition, they also represent a significant indirect effect of climate change on water quality. In some locations, changes in climatic conditions may well mitigate existing water-quality problems; in other locations, the combined effects of climate change and human resource use will make matters worse (Dale, 1997). Predicting the ecological response to changes in climate is therefore complex, and is currently hampered by a paucity of quality-controlled, well-distributed, long-term data on water-quality trends. Resource managers charged with maintaining water quality for human uses must determine which of the change scenarios offered in the literature are relevant to their resource area, and how to prepare for the most likely changes with achievable management strategies. This paper will (1) summarize the major water-quality changes associated with specific climate-change scenarios that have been suggested in the recent literature, and (2) present important patterns of water-quality response to climate change that warrant consideration in the development of resource-management plans and monitoring strategies.

POTENTIAL CLIMATE SCENARIOS AND ASSOCIATED EFFECTS ON WATER QUALITY

Most recent studies that consider the effects of climate change on water quality in North America are based on the prediction that global warming will lead to less surface water volume, particularly during the summer (Watson *et al.,* 1996; Schindler, 1997; Mulholland *et al.,* 1997). This prediction is derived from interpretation of climate records that indicate a 0.3-0.6°C increase in global air temperatures over the past century (Bridgham *et al.,* 1995), and model results that indicate milder winters, longer growing seasons, and increased evaporation and transpiration (McKnight *et al.,* 1996; Cushing, 1997). Estimated rates of evapotranspiration (ET) in these studies are predicted to exceed increases in precipitation (Schindler, 1997), resulting in lower streamflow, declining lake levels, shrinking wetlands, and decreased rates of groundwater recharge (Watson *et al.,* 1996; Schindler, 1997; Magnuson *et al.,* 1997; Poiani *et al.,* 1995; Poiani and Johnson, 1991). In contrast, some regions of North America are predicted to have increased runoff because projected precipitation increases exceed projected increases in ET (e.g., Florida and other Gulf states) (Mulholland *et al.,* 1997; Justic *et al.,* 1996). A recent study of streamflow trends in the United States indicates that a wetter, but less extreme climate with fewer floods or droughts has been developing during the 20th century in North America (Lins and Slack, 1999a,b). They attributed an observed increase in the proportion of annual surface-water discharge at low- to medium-flow rates and a decrease in peak flows to decreased ET caused by increased cloud cover. These regional models of future climate conditions may be of limited use to managers of specific resources. Local climates at specific resource areas can exhibit long-term trends that are inconsistent with regional patterns (Williams, *et al.,* 1996). The uncertainty associated with predicting local climate change from global climate models means that management plans for addressing potential climate effects on the quality of specific water resources must consider the effects of dryer, wetter, more variable, less variable, or simply warmer conditions depending on the interactions of several site-specific environmental factors.

Factors Controlling Climate-Induced Changes in Water-Quality

The water quality of a given surface-water body is a direct reflection of the chemical inputs received from the air and surrounding landscape, and the biogeochemical processes that transform those inputs within the water body itself. Inputs such as atmospheric deposition to the water surface or point source pollutant discharge represent direct chemical injections into the aquatic system. Water that falls on the adjacent watershed must take a more indirect route through variable portions of the vegetation, soil, and deep soil ecosystems, each of which may contribute to, extract from, and transform the chemistry of that water before it reaches a stream. The hydrologic flowpath that water takes through the watershed, and the resulting transformations that take place along that route, will determine its chemical characteristics when it enters the surface-water system. Studies of paleo-climate changes recorded in lake sediments show that the hydrological setting of a lake was a significant control on how the lake responded to climate fluctuation (Fritz, 1996). Likewise, current changes in climate will influence water quality by altering the balance between the interwoven atmospheric, terrestrial, and aquatic processes in a watershed, and the effects of human resource use on these processes. Water-resource managers wishing to anticipate, and either mitigate or take advantage of climate change in their management plans, must therefore have a working understanding of the local factors that control water quality and volume, and the sensitivity of those factors to change in climate and resource-use.

The effects of specific climate change scenarios on several of these controlling factors, and the associated changes in water quality predicted in the recent literature, are summarized in Figure 1. The figure assumes two broad types of climate change – increases in air temperature, and changes in moisture (wetter and dryer) in addition to the temperature increases. The information provided in these charts is not comprehensive, but provides the reader with a general picture of the major changes in water quality predicted in the literature to date for each of the climate scenarios presented. A brief discussion of the anticipated changes associated with each climate scenario is presented below.

Changes in Water Quality Anticipated From Increased Global Temperature

Global increases in air temperature, and the associated increases in water temperature, can cause measurable changes in water quality that are independent of changes in moisture (Jacoby, 1990; Schindler, 1997). Temperature alone can decrease the volume of surface waters through increases in ET. The southeastern United States has little seasonal variability in precipitation, but significant variability in streamflow because of seasonal changes in rates of ET (Mulholland *et al.*, 1997). Increases in water temperature decreases the oxygen-holding capacity of surface waters, which can decrease productivity in surface waters already stressed by biological oxygen demand (BOD) (Jacoby, 1990). Increased water temperature is anticipated to result in increased anoxia of already eutrophied waters of the southeastern United States if global warming occurs (Mulholland *et al.*, 1997).

The duration and intensity of stratification in surface waters is a major factor in determining seasonal changes in surface-water quality. The interaction of several factors including the initial temperature of surface and ground water inputs to a lake, lake trophic state, and the physical geometry and volume of the lake basin will determine the magnitude of stratification change in a specific lake, and the associated oxygen depletion that would result from increased water temperatures (Hostetler and Small, 1999; Stefan and Fang, 1999). For example, temperate lakes that completely lose their winter ice cover would begin to stratify only in summer, while more northern lakes that currently mix only in summer could stratify in summer and mix only in the fall and spring (Magnuson *et al.*, 1997). Models of thermal characteristics in northern Wisconsin lakes in a warmer climate indicate an earlier and more stable onset of stratification with a warmer epilimnion (DeStasio *et al.*, 1996). Each of these scenarios change oxygen availability and productivity in the lake hypolimnion.

Warming of air and water temperatures and increased atmospheric CO_2 have been shown to increase biological productivity and decomposition by increasing rates of metabolism, the duration of the growing season, and the volume of the lake epilimnion that is biologically active, and by altering aquatic and terrestrial species composition in the watershed (Hauer *et al.*, 1997; Covich *et al.*, 1997; Mulholland *et al.*, 1997; Band *et al.*, 1996). This increase in productivity will in turn increase nutrient cycling and accelerate eutrophication in aquatic systems with sufficient nutrient and oxygen supplies (Mulholland *et al.*, 1997). The increase in productivity from warmer temperatures may not, however, reach maximum potential levels in strongly oligotrophic or oxygen depleted waters. In oxygen-poor waters, the increased productivity could lead to anoxia and subsequent decreased productivity (Mulholland *et al.*, 1997), and in oligotrophic waters it could deplete nutrient concentrations to the point at which further growth is limited (Skjelkvale and Wright, 1998). Increased decomposition will decrease the proportion of biologically-available carbon in surface waters, thus potentially limiting productivity in systems with low or recalcitrant carbon pools (Clair *et al.*, 1996; Sun *et al.*, 1997).

Increases in the rates of chemical transformations in surface waters and lengthened periods of biological activity could result in increased bioaccumulation of toxins in aquatic organisms (Moore *et al.*, 1997). The toxicity of metals in aquatic ecosystems is enhanced by increased water temperature, through chemical interactions with dissolved organic carbon in low-pH waters (Wiener and Spry, 1996). This bioaccumulation of contaminants and toxins will decrease concentrations in the water column and thus may improve water quality. Likewise, warming will lead to increased chemical cycling between the water column and bed sediments, resulting in changes such as increased dissolution of iron phosphates from sediments in anoxic bottom waters or increased transfer

A. Effects of Increased Air Temperature on Factors that Control Water Quality

HYDROLOGIC FACTORS

Increased water temperature

- Decreased oxygen-carrying capacity
  - *Increased anoxia in eutrophic waters (Jacoby, 1990; Mulholland et al., 1997)*
- Earlier, more intense shift in lake stratification, warmer epilimnion (DeStasio et al., 1996)
  - *Temperate dimictic lakes become monomictic (more productive), cold monomictic lakes become stratified (less productive) (Schindler, 1997; Hostetler and Small, 1999; Magnuson et al.,1997)*
- Decreased volume (increased ET) for dilution of chemical inputs
  - *Increased concentrations of nutrients and pollutants (Cruise et al., 1999)*
  - *Increased salinity in arid-region lakes (Evans and Prepas, 1996; Covich et al., 1997; Grimm et al., 1997)*
- Decreased ice cover, ice-jam flooding, depth of lake freezing (Anderson et al., 1996; Rouse et al., 1997)
  - *Increased nutrient and chemical cycling (Magnuson et al., 1997; Rouse et al., 1997)*
  - *Desiccation of delta lakes in arctic (Marsh and Lesack, 1996; Rouse et al.,1997)*

Increased rates of productivity, decomposition, and chemical reactions

- Longer growing seasons, faster metabolic rates
  - *Decreased bioavailable carbon (Clair et al. 1996; Sun et al., 1997)*
  - *Increased nutrient and mineral cycling (Rouse et al.1997; Mulholland et al., 1997)*
- Larger epilimnion volume (deeper lake thermoclines)
  - *Increased biologically active zone; increased nutrient cycling (Fee et al., 1996; Covich et al., 1997)*
- Warmer hypolimnion water and sediments
  - *Increased chemical reactions between water and sediments (Magnusen et al., 1997)*
- Increased biological processing of toxins and other contaminants
  - *Decreased concentration of toxins and contaminants (Moore et al., 1997)*
  - *Increased nutrient uptake and bioaccumulation in sediments (Mulholland et al., 1997)*

Decreased volume (increased ET) for dilution of chemical inputs

- *Increased concentrations of nutrients and pollutants (Cruise et al., 1999)*
- *Increased salinity in arid-region lakes (Evans and Prepas, 1996; Covich et al.,1997; Grimm et al., 1997)*

Invasion by temperature-sensitive exotic species

- *Increased algal blooms, macrophyte nutrient cycling (Mulholland et al.,1997; Hauer et al.,1997)*

Increased runoff and erosion in permafrost and glaciated watersheds

- *Increased DOC, POC, and sediment (Wang et al.,1999; Rouse et al., 1997)*
- *Decreased water temperatures (Melack et al.,1997)*

TERRESTRIAL FACTORS

Vegetation change

- Species distribution changes
  - *Changes in nutrient leaching rates (Lovett and Rueth,1999)*
- Encroachment of vegetation above treeline in alpine regions
  - *Increased nutrient cycling in alpine lakes (Hauer et al., 1997; Campbell et al., 2000)*
- Invasion by temperature sensitive exotic species, pests
  - *Shifts in nutrient cycling, carbon storage (Mulholland et al., 1997; Eshleman and Morgan, 1998; Jenkins et al., 1999)*

Soil change: Increased microbial processing rates in soils

- *Increased leaching of nitrate to surface waters (Murdoch et al., 1998)*

Melting of permafrost and glaciers, reduction in peatland extent in Arctic and alpine regions

- *Increased DOC/POC, sediment (Wang et al.,1999; Rouse et al., 1997; Hauer et al., 1997)*
- *Decreased light (UV-B) penetration in areas affected by melting (Hauer et al., 1997; Rouse et al., 1997)*
- *Decreased southern extent of peatlands- increased carbon export (Rouse et al., 1997)*

RESOURCE-USE FACTORS

Increased energy use for cooling

- *Increased fluctuation in flow/temperature below reservoirs (Baldwin et al.,1999)*

Increased water use (ET compensation)

- *Increased pollutant concentrations (Cruise et al.,1999)*

Reservoir construction for water supply, energy

- *Conversion from stream conditions to reservoir conditions (Mulholland et al., 1997;Schindler, 1997)*

Compensating conservation behavior and technology

- *Genetic engineering of crops for low water requirements (Hatch et al.,1999)*
- *Change in water-use technology and behavior (Solley et al., 1999*

Figure 1A. Commonly Anticipated Effects that Changes in Hydrologic, Terrestrial, and Human-Related Factors Resulting from Increased Air Temperature Would Have on Water Quality.

B. Effects of Increased Air Temperature and Decreased Moisture on Factors that Control Water Quality (increases in ET exceed increases in precipitation)

HYDROLOGIC FACTORS

Increases in temperature caused by decreased flow (Jacoby, 1990)
- *Low dissolved oxygen more extensive (Mulholland et al., 1997; Magnuson et al., 1997)*
- *Enhanced in-stream, in-lake chemical processes (Mulholland et al., 1997; Schindler, 1997)*

Lower ground-water levels and decreased stream discharge (decreased dilution capacity)
- *Decreased export from streams (Murdoch, 1991; Schindler, 1997)*
- *Increased chemical concentrations in streams (Murdoch, 1991; Schindler et al., 1996)*
- *Increased concentration of point-source pollution (Cruise et al., 1999)*
- *Increased flushing time for contaminants (Mulholland et al., 1997)*
- *Increased lake ANC, decreased stream ANC (Schindler, 1997)*
- *Increased salinity in arid-region lakes and soils (Grimm et al., 1997; Melak et al., 1997)*

Decreased surface-water extent (desiccation of wetlands and ephemeral streams)
- *Increased sediment, carbon release (Moore et al., 1997; Rouse et al., 1997)*
- *Shift from reducing to oxidizing processes (LaBaugh et al., 1996)*
- *Decreased nonpoint-source runoff (Mulholland et al., 1997)*

Deeper flowpaths, decreased runoff and erosion, increased clarity of surface waters
- *Decreases in DOC, Si, P, NO3, and sediment transport (Schindler et al., 1996; Magnuson et al., 1997)*
- *Deeper lake thermocline; increased UV-B penetration (Magnuson et al., 1997; Schindler et al., 1996; Fee et al., 1996)*
- *Decreased concentration of weathering products in precipitation-dominated lakes (Webster et al., 1997)*
- *Decreased concentration of nonpoint-source pollutants (Mulholland et al., 1997)*
- *Increased reactions in bottom sediments; increased release of metals (Jacoby, 1990)*
- *Increased concentrations of weathering products (base cations, silica) (Mulholland et al., 1997)*

Increased lake stratification (Schindler, 1997)
- *Hypolimnetic anoxia in lakes more common (Stephan and Fang, 1999)*
- *Greater sediment biomass accumulation and nutrient release (Mulholland et al., 1997)*
- *More oxygen and light at depth and greater productivity in estuaries (Moore et al., 1997)*

Altered hydrologic factors specific to coastal zones
- *Increased estuarine salinity (Melack et al., 1997)*
- *Decreased stratification and hypolimnetic anoxia (Moore et al., 1997)*
- *Saltwater intrusion into coastal aquifers (Jacoby, 1990)*
- *Saltwater intrusion upstream of river-derived water supplies (P.DeVries, USGS, written commun., 1999.)*

TERRESTRIAL FACTORS

Increased fire frequency
- *Short-term increase in nitrate concentrations (Schindler et al., 1996)*

Changes in erosion
- Stabilization of streambanks by vegetation
  - *Decreased sediment and DOC (Poff et al., 1996, Hauer et al., 1997)*
- Decreased infiltration rates, increased runoff flashiness
  - *More concentrated episodes of nonpoint-source pollutants and sediment (Mulholland et al., 1997)*
  - *Temporary increase in DOC from episodic erosion of dry wetlands (LaBaugh, et al., 1996)*

Changes in chemical export from watershed
- Decreased weathering and weathering-product exports
  - *Decreased base-cation and silica concentrations in streams (Schindler et al., 1996)*
- Decreased forest-nutrient cycling
  - *Decreased foliar-N production (Mulholland et al., 1997)*
- Decreased soil flushing
  - *Delayed sulfate decrease and recovery from acid rain (Schndler, 1997)*
  - *Enhanced nitrate and sulfate export following drought periods (Schindler, 1997; this paper)*
- Earlier, smaller snowmelt flux
  - *Decreased nitrate runoff and surface-water acidification (Baron et al., 1998; Hauer et al., 1997; Campbell et al., 2000)*

RESOURCE-USE FACTORS

Increased percentage of annual flow consumed for human use
- *Decreased pollutant-dilution capacity of streams (DeWalle et al., 1999)*

Water-supply reservoir construction
- *Conversion of free-flowing conditions to lake conditions (Mulholland et al., 1997)*

Increased conservation actions and improved technologies
- Shift from flood irrigation to spray or drip irrigation
  - *Decreased sediment and nonpoint-source pollutants (Ebbert and Kim, 1998)*
  - *Increased dilution capacity for other pollutants (Jacoby, 1990)*
  - *Increased soil-water salinity and accumulation of metals in soil and surface water (Fio and Leighton, 1994)*
- Water-use conservation and decrease in per-capita use
  - *Increased dilution capacity of surface waters (Solley and Perlman, 1999)*
  - *Dredging to deepen navigation channels*
  - *Increased suspended solids, turbidity DOC (Jacoby, 1990)*

Figure 1B. Commonly Anticipated Effects that Changes in Hydrologic, Terrestrial, and Human-Related Factors Resulting from Increased Air Temperature and Decreased Moisture Would Have on Water Quality.

C. Effects of Increased Air Temperature and Increased Moisture on Factors that Control Water Quality (increases in precipitation exceed increased in ET)

HYDROLOGIC FACTORS

Increased water volume
- *Dilution of point-source pollutants (Jacoby, 1990)*
- *Increased load of nonpoint-source pollutants, increased eutrophication (Mulholland et al., 1997)*
- *Export of organic matter stored in river channel and wetlands (Mulholland et al., 1997)*

Decreased residence time of surface water and ground water
- *Decreased chemical reactions in streams and lakes (Schindler, 1997)*
- *Decreased flushing time for contaminants (Mulholland et al., 1997)*
- *Increased pollutant export to coastal waters (Jacoby, 1990; Mulholland et al., 1997)*

Inundation of wetlands and peatlands, and extension of perennial streams
- *Increased denitrification and sulfate reduction in saturated soils (Covich et al.1997; LaBaugh et al.,1996; Mulholland et al.,1997)*
- *Mobilization and bioaccumulation of metals (mercury) (Rudd, 1995)*

Increased turbidity
- *Decreased light and UV-B penetration in lakes (Rouse et al., 1997; Magnuson et al., 1997)*
- *Decreased hypolimnetic productivity (Rouse et al., 1997)*

Decreased lake stratification (greater mixing)
- *Increased productivity in hypolimnion (Schindler, 1997)*

Increased stratification in coastal waters (increased freshwater influx)
- *Increased hypoxia in coastal waters (Justic et al.,1996)*

TERRESTRIAL FACTORS

Decreased fire frequency
- *Decreased fire-related nutrient runoff; decreased regrowth-related nutrient retention (Schindler, 1997)*

Higher water tables

Expanded source areas for runoff
- *Increased concentrations and loads of nonpoint-source pollutants (Mulholland et al., 1997; Jacoby,1990)*

Increased runoff, erosion, and flushing of shallow soils
- *Increased concentrations of sediment, phosphorus, and nutrients (Meyer and Likens, 1979)*
- *Increased export of weathering products (Williams et al., 1996; White and Blum, 1995)*
- *Increased nitrate and DOC export from shallow soils (Murdoch and Stoddard, 1992)*

Increased snowpack in sufficiently cold regions
- *Increased snowmelt runoff (Williams et al., 1996)*

RESOURCE-USE FACTORS

Decreased competition between human need and natural ecosystem needs
- *Increased pollutant-dilution capacity (Jacoby, 1990; Eheart, 1988)*

Flood-control dam construction
- *Conversion of free-flowing conditions to lake conditions (Baldwin et al., 1999; Mulholland et al., 1997)*

Pollution cleanup
- *Decreased contaminants, toxins, nutrient loads (Jacoby, 1990)*

Increased runoff from urban areas (DeWalle et al., 1999)
- *Increased contamination of streams near combined-sewer overflows (Jacoby, 1990)*
- *Increased nutrients from stormflows that bypass wastewater treatment (Jacoby, 1990)*

Figure 1C. Commonly Anticipated Effects that Changes in Hydrologic, Terrestrial, and Human-Related Factors Resulting from Increased Air Temperature and Increased Moisture Would Have on Water Quality.

of constituents from the water column to the sediments through enhanced biological activity (Magnuson *et al.*, 1997). Thus, depending on the magnitude of the climate change and the initial water-quality condition, water quality will be improved by warmer temperatures in some circumstances and degraded in others.

The temperature of surface waters draining glaciated and permafrost terrain will decrease if air temperatures rise above thresholds that would stimulate widespread melting (Melack *et al.*, 1997; Rouse *et al.*, 1997). Melting of permafrost already is creating new small lakes and draining others in the Alaskan tundra (Wang *et al.*, 1999). The increasing infiltration that develops as the permafrost melts is raising ground-water levels and increasing lake depth (Rouse *et al.*, 1997; Wang *et al.*, 1999). Receding glaciers add to stream discharge in the glaciated regions, thus lowering stream temperatures until the glaciers completely melt (Melack *et al.*, 1997). Decreased ice cover on Arctic rivers could result in a decrease in the magnitude and frequency of ice-jam induced flooding, resulting in the drying up of delta lakes and wetlands (Marsh and Lesack, 1996; Rouse *et al.*, 1997).

If air temperatures increase, fewer lakes and streams in Arctic regions will freeze to the bottom, and northern-temperate surface waters will have an increased number of ice-free days (Anderson *et al.*, 1996; Rouse *et al.*, 1997, Magnuson *et al.*, 1997). In both cases these changes will increase nutrient cycling and productivity in the affected water body. Shortening of the ice-cover period in northern lakes and streams and shallower depths of freezing in Arctic lakes during warm years has been shown to cause increased productivity through the lengthening of the growing season. If sufficient nutrients sources are available, lakes in arctic and alpine regions will also experience increasing productivity as a result of more frequent mixing and deeper thermoclines withincreases in temperature (Skjelkvale and Wright, 1998; Rouse *et al.*, 1997; Wang *et al.*, 1999). Longer thaw seasons will also enhance decomposition and greenhouse gas releases from northern wetlands and peatlands (Bridgham *et al.*, 1995) This melting of ice and permafrost in arctic and alpine environments, and the associated increase in runoff and erosion, will cause increased concentrations of dissolved (DOC) and particulate (POC) organic carbon, increased sediment, and decreased light and UV-B penetration in surface waters (Melack *et al.*, 1997; Rouse *et al.*, 1997).

Warming-induced changes in terrestrial ecosystems, with resulting changes in water quality, have also been described in the literature. The most commonly mentioned changes are in rates of terrestrial nutrient cycling and delivery of nutrients to surface waters (Vitousek, 1994; Aber *et al.*, 1995). As in the aquatic system, lengthened growing seasons coupled with increased atmospheric CO2 will increase terrestrial productivity (Dixon *et al.*, 1994). Different tree species have different rates of nutrient cycling (Lovett and Rueth, 1999), so changes in species composition as a result of temperature changes will also change rates of nutrient cycling in a watershed. Changes in respiration rates in a watershed can produce significant feedback effects in the atmosphere that could counteract or exacerbate the effects of increased temperature on nutrient export (Band *et al.*, 1996). The encroachment of forests above current timberline elevations in alpine terrain as a result of receding glaciers, or simply due to an expanded elevational range for temperature-sensitive vegetation, will increase nutrient and carbon cycling in alpine watersheds and decrease clarity and UV-B penetration in surface waters (Hauer *et al.*, 1997; Campbell *et al.*, 2000).

Changes in soil productivity and biogeochemical cycling can significantly affect the quality of runoff from terrestrial ecosystems. Nitrification rates in soils of the Catskill Mountains of New York are temperature dependent, and mean annual nitrate concentrations in streamwater are highly correlated with average annual air temperature (Murdoch *et al.*, 1998). Deposition of nitrogen in the Catskills exceeds the nutrient demands of vegetation, and rates of microbial processing of N in soils appear to now exert a large influence on rates of N leaching to surface waters (Murdoch *et al.*, 1998). Evidence from other regions indicate that soil microbes play a significant role in determining rates of N retention and release to surface waters in forested watersheds (Fenn *et al.*, 1998), and a significant correlation has been observed between soil respiration rates and temperature (Stottlemyer and Toczydlowski, 1996; Niklinska *et al.*, 1999; Murdoch *et al.*, 1998). Other studies have shown significant increases in nutrient cycling and nitrogen leaching to surface waters associated with infestations of forest pests (Swank, 1988; Eshleman and Morgan, 1998; Jenkins *et al.*, 1999). These infestations are anticipated to increase with increasing temperatures in the southeastern United States (Mulholland *et al.*, 1997).

The northward invasion of temperature-sensitive exotic species with increasing temperatures, and the resulting changes in nutrient supply, nutrient cycling, and the BOD in aquatic ecosystems is a concern in the southeastern United States (Mulholland *et al.*, 1997). Likewise, the range of temperature-restricted exotic species already established within North America is expected to expand more rapidly northward and to higher elevations if temperatures increase (Watson *et al.*, 1995). The range of native aquatic species could also change as a result of changes in climate (Eaton and Scheller, 1996). If such species changes alter the nutrient or sediment flux through watersheds or the trophic state of surface waters, changes in water quality would likely result. As we will discuss later, the water-quality responses to climate change in a given water body will depend on whether specific temperature thresholds for specific species, biogeochemical processes, and physical factors that control local water quality are surpassed.

Changes in Water Quality Anticipated from a Warmer-Dryer Climate

Drought and associated decreases in stream discharges allow for greater warming of surface waters, and thus further decreases the oxygen-holding capacity while stimulating productivity and rates of chemical reactions (Jacoby, 1990; Mulholland *et al.*, 1997). As a result, resource managers for flow-regulated systems are expected to maintain stream discharges at rates sufficient to maintain oxygen concentrations above stress levels for aquatic organisms (Jacoby, 1990). Decreased stream discharge is typically associated with increases the concentration of chemical constituents, but decreases in total export of chemical constituents because of decreased stream volume (Murdoch, 1991; Mulholland *et al.*, 1997; Schindler, 1997). A warmer, drier climate also increases the residence time of chemical constituents within surface waters and watershed soils, and causes longer flushing times for toxins in lake waters (LaBaugh *et al.*, 1996; Schindler, 1997). These longer residence times will also extend the time available for in-lake bioremediation, which will decrease nutrient and contaminant concentrations (Schindler, 1997). The actual change in concentrations in a given lake will therefore depend on the balance between these concentrating and diluting processes. In general, concentrations of surface-water constituents typically derived from the flushing of surface and shallow-soil solutions during high-runoff periods (DOC, nitrogen, acidity, and non-point-source pollutants) and through erosion (phosphorous, particulate organic matter) will decrease, and the concentrations of constituents derived from deeper flowpaths (base cations, silica), and from point sources (sewage and industrial waste) will increase in a warmer, drier climate (Murdoch, 1991; Schindler, 1997; Mulholland *et al.*, 1997).

Warmer-dryer conditions also would lead to increased desiccation of currently saturated areas and an increase in ephemeral conditions in streams (Poiani *et al.*, 1995). This change would potentially add a significant amount of carbon dioxide and methane to the atmosphere, and episodically increase DOC concentrations during storm runoff from the desiccated areas (Swanson *et al.*, 1988; Khalil and Shearer, 1993; Poiani *et al.*, 1995). If evapotranspiration increases and snowmelt runoff decreases, a large reduction in meadow and shallow marshlands of the northern prairies is predicted (Poiani and Johnson 1991; Poiani *et al.*, 1995; Winter, 1989), resulting in the loss of an important sink for nutrients and carbon. Decreased scouring in near-stream zones as a result of lower flows will allow vegetation to stabilize riverbanks and further decrease sediment and DOC transport (Poff *et al.*, 1996). In arid regions of the U.S. and Mexico, and semiarid areas of central Canada, warmer-dryer conditions has been shown to increase salinity and metals in surface waters and watershed soils (Evans and Prepas, 1996; Grimm *et al.*, 1997).

Decreased runoff from the Experimental Lakes Area (ELA) watersheds in western Ontario during the 1970s and 1980s resulted in lower DOC in lake waters, thus increasing light penetration and exposing more of the lake depth to UV-B radiation (Schindler *et al.*, 1996; Fee *et al.*, 1996). DOC concentration is a determining factor in the depth of penetration of UV-B radiation into surface waters, and may be more significant as a control on UV-B toxicity in aquatic systems than changes in stratospheric ozone concentrations (Williamson *et al.*, 1996). The increased light penetration will also lead to an increase in the photochemical oxidation of DOC, further clarifying lake waters (Magnuson *et al.*, 1997; Skjelkvale and Wright, 1998). The greater light and temperature penetration will result in greater biological activity at depth and in lake-bottom sediments and, thus, could ultimately increase the duration and extent of hypolimnetic anoxia (Jacoby, 1990; Schindler, 1997; Mulholland *et al.*, 1997).

Decreased streamflow is of particular concern downstream of large point sources of nutrients such as sewage treatment facilities or concentrated animal management operations (Jacoby, 1990). Effluent containing large amounts of dissolved or fine-particulate phosphorus (P) will cause higher stream concentrations during low-flow rather than high-flow conditions because of the decreased dilution capability associated with lower surface-water volume (Smith *et al.*, 1982). In contrast, P from fertilizers and soil erosion is transported mostly (95 percent) in particulate form and decreases during low-flow conditions (Meybeck, 1982; Meyer and Likens, 1979). Dissolved P entering surface waters during low flow periods will have a longer in-stream residence time and will be more readily available for biological uptake and chemical reactions than the particulate P that is transported by storm runoff. The relative contribution of P and other nutrients from point sources to surface-waters in a drier-climate scenario would therefore be expected to generally increase relative to the amounts contributed from non-point sources.

As in the warmer climate scenario, eutrophication is predicted to increase in the southeastern United States in a warmer, dryer climate, because of increased nutrient concentrations, warmer water temperatures, and increased water residence times (Mulholland *et al.,* 1997). Nutrients in some southern U.S. lakes became increasingly available during a prolonged drought through nutrient release from sediments caused by increased stratification and development of anoxic bottom waters (Porter *et al.,* 1996; Mulholland *et al.,* 1997). Conversely, a 20-year period of lower-than-average flow and higher- than-average air temperatures in ELA lakes from the early 1970s to the early 1990s resulted in a decreased frequency and magnitude of watershed flushing and decreased concentrations of nutrients available in surface waters (Schindler *et al.,* 1996).

A warmer, drier climate has been shown to alter the patterns and frequency of terrestrial ecosystem disturbance, with significant repercussions in water quality (Naiman *et al.,* 1995; Kurz *et al.,* 1995; Schindler, 1997). For example, regional precipitation patterns and frequency can have a direct effect on the frequency of forest and grass fires (Dale, 1997). Fire frequency increased during the 20-year drought at the ELA, and contributed to accelerated warming of stream and lake waters and short-term increases in nutrient exports in the fire-affected areas (Schindler *et al.,* 1996). Water stress also decreases nitrogen concentrations in foliage, thus decreasing nitrogen sources for runoff to surface waters (Mulholland *et al.,* 1997). The effect of drought on surface water quality will therefore be partly dependent on the extent of drought-induced terrestrial disturbance that occurs in the watershed.

As a result of the Clean Air Act mandated reductions in industrial sulfate emissions, concentrations of sulfate in lakes and streams acidified by acid deposition are anticipated to decline over time through hydrologic flushing of the watershed soils by precipitation with declining sulfate loads (NAPAP, 1998). Decreased runoff will prolong stream and lake recovery from acid deposition because sulfate would be flushed from the soil at a decreased rate (Schindler, 1997). Concurrently, stream acid-neutralizing capacity would further decrease because rates of mineral weathering that regulate base-cation availability are positively correlated with precipitation amount and physical transport (Bluth and Kump, 1994; White and Blum, 1995; Williams *et al.,* 1996).

Rates of permafrost melting are greater in a warm, dry climate than in a climate that is warm and wet (Rouse *et al.,* 1997). With a longer -term shift toward a warmer-dryer climate, surface waters of the Arctic would be expected to exhibit changes similar to those predicted for temperate regions. Until the permafrost and glaciers are melted, however, a warmer-dryer climate in the Arctic will have hydrologic characteristics of a temperate warmer-wetter scenario, with greater runoff and erosion, higher nutrient and DOC concentrations in surface waters, and lower water temperatures because of melting ice (Melack *et al.,* 1997; Rouse *et al.,* 1997).

Changes Specific to Estuaries. Climate change will pose a unique set of water-quality issues for coastal ecosystems. Estuary and coastal water stratification generally decreases with decreased river inflow, which in turn leads to greater mixing throughout the water column, decreased anoxia of bottom waters, and decreased nutrient availability (decreased extent of eutrophication) (Justic *et al.,* 1996; Mulholland *et al.,* 1997). The maximum extent of hypoxia in the Gulf of Mexico is correlated with high river flow which increases density stratification and eutrophication from increased nutrient inputs (Mulholland *et al.,* 1997). However, increased water clarity and residence time during low-flow periods in the Hudson River estuary in eastern New York correlates with increased biological activity and eutrophication, barring a concurrent decrease in available nutrients sufficient to limit growth (R. W. Howarth, Cornell University, written commun., 1999). The effects of a warmer, dryer climate on water quality in estuaries, as in freshwater systems, will therefore be strongly influenced by local controlling factors.

Decreased discharge in rivers draining directly into the coastal zone also can result in above-normal intrusion of salt water into tidal reaches of rivers (Mortch and Quinn, 1996). Such intrusions have extended upstream of municipal water-supply intakes during extremely low flows (Jacoby, 1990). Low-flow conditions in the Hudson River during the mid-1990s resulted in salt-front intrusions beyond the water-supply intake pipe for the city of Poughkeepsie, New York, and temporarily suspended use of the Hudson River as a source of drinking water (P. DeVries, U.S. Geological Survey, written commun., 1999). Saltwater intrusion into the lower Delaware River during the 1960s drought contaminated coastal aquifers where a combination of diminished ground-water recharge and over pumping of wells lowered the water table below levels required for maintaining a hydraulic gradient toward the river (Jacoby, 1990). Obviously, the magnitude and extent of saltwater intrusion issues will depend on the shoreline hydrological gradients, ground-water pumping rates, inland distance of river-water withdrawal points, and drought characteristics, all of which will be unique to specific coastal regions.

Changes in Water Quality Resulting from a Warmer-Wetter Climate

Increased water volume in a wetter climate will have a mitigating effect on many of the water-quality issues described for the dryer climate scenario, but will create other problems for water quality managers. Greater dilution of point-source pollutants resulting from increased water volume will have a positive effect on water quality where human activity is currently degrading water-quality conditions (Jacoby, 1990; Cruise *et al.,* 1999). These pollution sources include point sources and non-point sources from landscapes that are flushed by runoff typical of the current climate conditions. Increased erosion and sediment transport during high runoff, and decreased biological and chemical transformations within surface waters because of decreased residence times, will have a counterbalancing effect on these water-quality improvements. Organic matter stored within stream corridors, wetlands, and lakes could be mobilized by higher flows, thus increasing DOC concentrations and decreasing water clarity (Mulholland *et al.,* 1997). A wetter climate will expand the spatial extent of direct runoff to surface waters, thus significantly increasing pollutant loads from point and non-point pollution sources that are hydrologically isolated or filtered through groundwater aquifers under current flow conditions (Jacoby, 1990; Lins and Slack, 1999; Mulholland *et al.,* 1997). Increased sea level and expanded watershed source areas for runoff could result in leaching of contaminants from the numerous hazardous waste dumps located in the coastal region (Jacoby, 1990). The extent of these expanded variable source areas, and the potential contamination sources within them, is again dependent on the local geology and human resource use, and therefore will be highly location-specific.

Inundation of wetlands, riparian zones, and low-lying soils has been shown to result in increased mobilization of trace metals and organics from soils, increased mobilization and methylation of mercury, and greater anaerobic activity in saturated soils (sulfate reduction, denitrification) (Meyer and Pulliam, 1992; Rudd, 1995; LaBaugh *et al.,* 1996). As with the effects of lowered ground-water tables and stream discharge mentioned earlier, sea-level rise can also contaminate coastal aquifers by shifting the hydrologic gradient toward the land (Hansler and Major, 1999).

Increased moisture is predicted to decrease rates of permafrost melting compared to warm-dry conditions (Rouse *et al.,* 1997) but increased erosion from high runoff associated with a wetter climate may yield greater nutrient, DOC, and sediment loads from melting permafrost terrain. Warmer, wetter conditions will favor increased weathering and erosion, and greater runoff of phosphorous and other weathering products (Meyer and Pulliam, 1992; Bluth and Kump, 1994; White and Blum, 1995; Covich *et al.,* 1997). Shallower watershed flowpaths also favor leaching of organic carbon and acid anions to surface waters (Murdoch and Stoddard, 1992). Increasing the duration or frequency of high-inflow conditions will generally lengthen the annual period of acidification in acid-sensitive lakes in the Northeastern United States (Moore *et al.,* 1997), but shorten the time for recovery of watersheds from acid deposition effects following decreases in sulfur and nitrogen deposition (Schindler, 1997). However, a wetter climate may also diminish the magnitude of episodic acidification events because pools of available acids in the shallow soils are depleted by persistent hydrologic flushing (Murdoch and Stoddard, 1992).

Increased stratification and hypolimnetic anoxia are also predicted to result from increased freshwater and nutrient runoff in other coastal waters of the US and Canada (Moore *et al.,* 1997; Mulholland *et al.,* 1997). Models of hypoxia in the Gulf of Mexico indicate a 30-60 percent decline in summer oxygen levels below the pycnocline with a 20 percent increase in river discharge (Justic *et al.,* 1996). This change was predicted by simulations that increased stratification and nutrient inputs of the Gulf as a result of greater freshwater runoff from the Mississippi River (Justic *et al.,* 1996). Increased flow into saline lakes has been shown to increase stratification, but results in greater mixing at depth (Romero and Melack, 1996).

Changes in Resource Use

Water supply for human use is already highly managed to optimize societal access to sufficient water, with a significant buffer engineered into most systems for handling increased or decreased supply (Watson *et al.,* 1996; Naiman *et al.,* 1995; Lins and Slack, 1999b). Such "buffers" for maintaining surface water quality during a changing climate are less well defined. If streamflow decreases as a result of changes in climate, waste-flow reallocation or flow-augmentation may be necessary to protect water quality in river reaches downstream of human activity (Meyer and Pulliam, 1992). This issue will be exacerbated in areas where expanding urban population or irrigation-supported agriculture is coupled with increased temperature and decreased streamflow (DeWalle *et al.,* 1999; Cruise *et al.,* 1999). For example, managers of the New York City water supply, which provides drinking water from the Delaware and Hudson River watersheds to more than 20 million people, are required by law to maintain specific flow levels downstream of their reservoirs regardless of water demand for human use (Wolock *et al.,* 1993).

Changes in land and water use can be both a cause of (e.g., clearing of forests, filling-in of wetlands, and desertification) and a response to (e.g., altering irrigation practices, demographic shifts, etc.) changes in climate, with significant repercussions in water quality (Dale, 1997; Vitousek, 1994; Mulholland *et al.,* 1997). Simulations of hydrologic conditions in the South Platte River basin under current and pre-agricultural conditions indicate that agricultural land-use has increased ET by 37 percent and primary productivity by 80% above natural rates for the Colorado plains (Baron *et al.,* 1998). A 4°C increase in air temperature predicted by climate change models for the Platte basin increases evaporation 28 percent and decreases plant productivity as a result of decreased moisture (Baron *et al.,* 1998). The changes in agricultural water demand associated with such a change in climate would affect the quality of nearby surface waters through decreased flow and increased pollutant concentrations in runoff.

Decreases in agricultural water use through improved technology can affect surface-water quality in both positive and negative ways (Solley and Perlman, 1999). A shift from flood to drip irrigation methods in the Crab Creek watershed of western Washington state resulted in a 15 percent reduction in the yield of suspended solids in Crab Creek over a 13-year period (Figure 2) (Ebbert and Kim, 1998). Decreased use of water in irrigation has been shown to have a dramatic effect on surface-water and ground-water quality and quantity in the landscape affected, including decreased transport of sediment and associated contaminants to surface waters, and increased salt concentrations in soils, ground water, and surface water (Grimm *et al.,* 1997; Fio and Leighton, 1994).

Urbanization of non-developed landscapes adversely affects water quality through increased flashiness of runoff from impervious surfaces, lowered groundwater levels resulting from human water use, and increased concentration and flux of nutrients and contaminants (DeWalle *et al.,* 1999). Management strategies that increase infiltration of surface runoff, and public awareness programs such as labeling of storm drains, have improved water quality in urban runoff from some areas.

Reservoir construction is anticipated to increase in a drier climate as a means of increasing water supply for human use (Mulholland *et al.,* 1997). Reservoir construction would also occur in response to a climate that shifts the seasonality of runoff events as a means of sustaining water supplies into dry seasons (Baldwin *et al.,* 1999), or to a wetter climate as a means of controlling flooding (Frederick and Schwartz, 1999). Water impoundment and the associated flooding of soils, and ground-water pumping in arid environments can cause metals that were previously sequestered to go into solution and degrade water quality (Rudd, 1995; Fio and Leighton, 1994). For example, the concentration of selenium in wetlands of the Central Valley in California are a result of the leaching of selenium from soils by irrigation water pumped from rivers and ground water aquifers (Fio and Leighton, 1994). The impoundment and consequent slowing of flowing waters will result in (a) a shift from river to lake water-quality conditions within the impounded section of the river; (b) cooler water temperatures immediately downstream of the reservoir because of the release of hypolimnetic reservoir water; (c) warming of river waters further downstream because of decreased flow; (d) decreased variability in flow and sediment downstream of the new impoundments; and (e) increased methane, CO2, and methylated mercury production from the inundation of vegetation and soils (Jacoby, 1990; Schindler, 1997; Baldwin *et al.,* 1999; Rudd,1995).

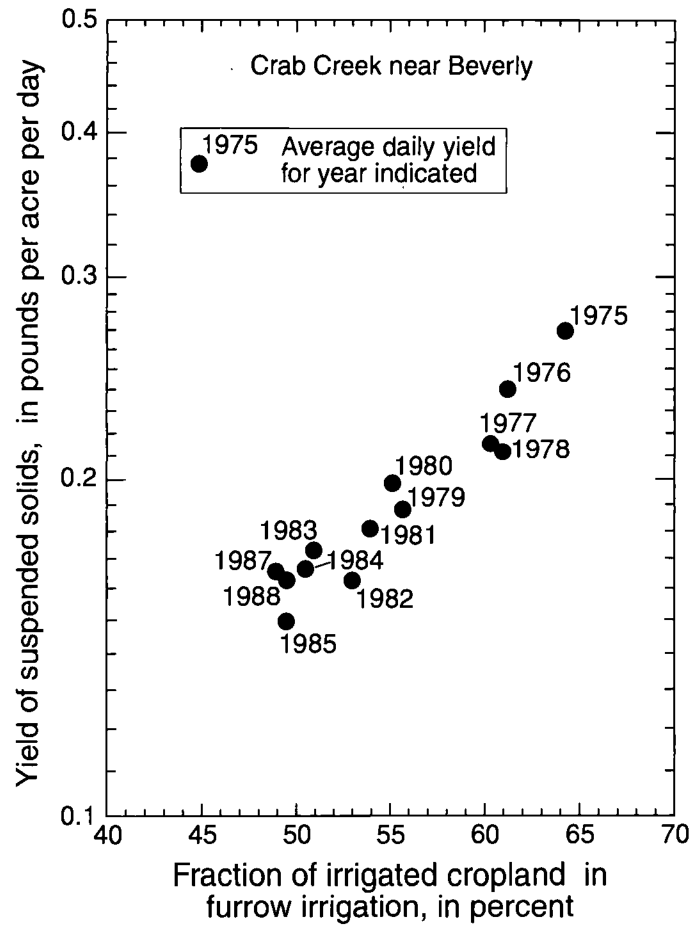

Figure 2. Relation of the Load of Suspended Soils in Streamwater to the Fraction of Cropland Being Irrigated by Furrow Irrigation Methods in Tributaries to the Yakima River in Eastern Washington State (Ebbert and Kim, 1998).

Such changes in land use and human behavior, or technological advances in efficiency that improve water conservation, will therefore have a significant affect on water quality in regions of human resource use. Water use per-capita has steadily declined in the United States since the mid-1980s as a result of new technologies that require less water, improved industrial efficiency, increased water recycling, and changes in water regulations (Solley and Perlman, 1999). Genetic engineering of crops is expected to reduce water requirements for some agricultural products (Hatch *et al.,* 1999). Thermoelectric power generation has also not increased since the 1980s, despite warmer air temperatures. On the other hand, projected increases in population result in increases of water use by 24 billion gallons per day from 1995 to 2040 (Solley and Perlman, 1999). Water use in the Rio Grande, lower Colorado, and Great Basin watersheds of the southeastern U.S. is already having a negative effect on surface-water habitat and water quality (Frederick and Schwartz, 1999). In watersheds influenced by human water use, the balance between improvements in technological-efficiency and water demand for ecological and human needs will likely have a more significant effect on water quality than changes in climate (Solley and Perlman, 1999).

THRESHOLDS OF WATER QUALITY CHANGE IN RESPONSE TO CHANGES IN CLIMATE, AND MANAGEMENT IMPLICATIONS

Aquatic ecosystems have a buffering capacity that provides a variable level of resistance to significant water-quality change from the stresses associated with climate change. For example, temperatures in the northern Arctic are sufficiently cold to be able to withstand some climate warming without significant melting of the permafrost. There are, however, thresholds of stress (e.g., air temperatures above freezing for a certain number of days) beyond which significant and often rapid ecosystem changes that affect water quality can occur. Whether such thresholds are exceeded will depend on the magnitude of the stress and the susceptibility to change of the specific water body and adjacent watershed. If a system is near a stress threshold, a small shift in stress can precipitate a disproportionate change in water quality.

Susceptibility of aquatic systems to water-quality change depends on the hydrology, biogeochemistry, and trophic state of those systems, and therefore will be spatially and temporally variable. The transition zone between major ecoregions roughly represents a spatial susceptibility threshold beyond which the sustainability of one ecoregion is diminished and another ecoregion is enhanced. For example, modeling of forest productivity and soil nitrification rates indicate that the forests of the boreal ecosystem fringe are sensitive to small changes in temperature and moisture (Stottlemyer, 1992). The ecological changes resulting from crossing these thresholds have implications for the quality of water draining the affected landscapes. The clearest paleo records that allow us to document ancient climate change are found within these ecoregion boundaries, and in regions of climatic extremes (Fritz, 1996). Likewise, it is within such transition zones that changes in climate will most likely have their most immediate effect on water quality.

The magnitude of stress factors are also spatially variable. Stress factors such as acid deposition or increased air temperature have spatial gradients of intensity that may be measurable across a region (Ollinger *et al.,* 1993; Rouse *et al.,* 1997). Modeled estimates of air temperature change resulting from increased CO_2 in the atmosphere indicate a gradient of increasing temperature change from the equator to northern latitudes (Felzer and Heard, 1999). Recent assessments of data from multiple locations throughout the globe indicate a synchrony of responses in surface waters to similar changes in climate (Magnuson *et al.,* 1990; Baron and Caine, 1999). Therefore, if the spatial gradient of a specific stress factor that influences water quality is observable across a region, a resource manager may be able to use their position along that gradient to determine the vulnerability of their management area to water-quality degradation.

Both stress factors and susceptibility to change are also temporally variable. As mentioned previously, flushing of nutrients or pollutants from watershed soils during rainstorms can cause episodic water-quality degradation of a few hours' duration. Likewise, seasonal variability in hydrologic conditions or biological activity (e.g., summer lake stratification and associated decreased in hypolimnetic oxygen concentrations) will cause seasonal changes in the susceptibility of a given surface water body to increases in stress factors. This spatial and temporal heterogeneity in the balance between stress factors and the susceptibility of surface waters to water quality change will naturally lead to spatial and temporal variability in the response of water quality to a warming climate. Climate change effects will be first observed in surface waters where and when the added stress is sufficient to overcome the system's resistance to change, and will therefore likely be episodic in it's developing stages. Determining how close specific resource areas are to climate-dependent thresholds of water-quality change is therefore an essential ingredient to developing cost-effective, pre-emptive management strategies.

Episodic Excursions Beyond Ecological Thresholds

The seasonality, intensity, and form of precipitation can affect water quality by increasing the variability in ground-water levels and stream discharge. A commonly predicted result of climate change is that floods and droughts will become more extreme or frequent over large regions of the United States and Canada (Watson *et al.*, 1996; Mulholland *et al.*, 1997; Moore *et al.*, 1997). Historical evidence indicates that even moderate changes in climate can result in large changes in the magnitude of hydrologic events (Knox *et al.*, 1993) and associated water quality (Grimm *et al.*, 1997; Moore *et al.*, 1997). For example, phosphorous export from forested catchments has been shown to be primarily associated with episodes of high discharge and sediment load, and thus would increase in concentration in surface waters with a higher frequency of storm-event runoff (Meyer and Likens, 1979). Increased storm-related discharges, with longer dry periods between storms in which pollutants can accumulate in the watershed, result in pulses of high-concentration pollutant runoff (Murdoch and Stoddard, 1992; Mulholland *et al.*, 1997). These pulses of concentrated acid, nutrient, contaminant, or toxin releases from the landscape can be more damaging to aquatic ecosystems that the slower release of similar amounts of pollutants during more-frequent, smaller runoff events (Baldigo and Murdoch, 1997; Moore *et al.*, 1997).

If changes in water quality occur because of exceedance of stressor thresholds, pollution-reduction targets based on annual median flux rates will be inadequate for protecting against episodic degradation of some aquatic ecosystems as the climate warms (Alexander *et al.*, 1996; Murdoch and Stoddard, 1992). For example, less frequent but larger storms in regions affected by acid deposition would increase the severity of acid pulses in streams and lakes during those storms, as the acids that entered the watershed in dry deposition, snow, or small rainstorms are flushed into the streams during the large storm events (Moore *et al.*, 1997). Duration of these acid pulses is also important; fish that are exposed to toxic forms of aluminum in acidified streams in the Northeastern United States can survive that exposure if the length of time they are exposed is short (Baldigo and Murdoch, 1997). Conditions toxic to fish have been shown to be generated for short periods during rainstorms and snowmelt in some Northeastern U.S. streams, while water quality in those same streams is non-toxic during baseflow periods (Van Sickle *et al.*, 1996; Baldigo and Murdoch, 1997). In the Catskill Mountains of New York, differences in the annual export of hydrogen ion during wet and dry years were driven more by the relative magnitude and duration of these acidic "episodes" than by differences in baseflow acid concentrations (Murdoch, 1991). These acidic episodes were primarily caused by increases in nitric acid. Further, historical records of nitrate concentrations in rivers of the Catskill region indicate an increase since 1950 in the magnitude of nitrate concentrations in surface waters during stormflows, while changes in base-flow nitrate concentrations have been considerably less (Murdoch and Stoddard, 1992). These findings support the hypothesis first postulated by Stoddard (1994) that in ecosystems receiving nitrogen in excess of the biological demand will begin leaking that nitrogen to surface waters during stormflows. Thus the effects of climate on stream acidification in the Catskills is expected to first appear as changes in the chemistry and duration of stormflow.

High-flow periods can cause significant short-term water-quality changes in large rivers and estuaries as well. Nutrient and herbicide flux rates are correlated with stormflows in the Mississippi River (Goolsby, 1994). Alexander *et al.* (1996) analyzed the importance of episodic nutrient flushing to annual nutrient loads at 104 monitoring stations on rivers that drain into the Atlantic from the United States. The study showed that mean annual nutrient flux can vary from year to year by as much as two orders of magnitude in some rivers, primarily as a result of differences in the number or magnitude of stormflows. The largest rivers generally exhibited a greater-than-proportional increase in flux for a unit increase in flow (1.2 percent increase in flux for a 1 percent increase in flow in the Potomac and Hudson Rivers), while most rivers in the study showed a proportional to slightly less-than-proportional change (decreased flux relative to a unit increase in flow, or a net dilution) (Alexander *et al.*, 1996).

These short-term changes in water quality could exceed biologically-relevant thresholds that are not apparent at median flow levels, and thus would not be detected by conventional base-flow sampling. Major ecological events, such the development and expansion of hypoxia zones in the Gulf of Mexico after flooding of the Mississippi River, or fish kills from aluminum-induced toxicity in streams of the Northeastern United States during acid episodes, suggest that short-term climatic fluctuations can have significant long-term effects on ecosystem balance (Baldigo and Murdoch, 1997; Goolsby, 1994; Justic *et al.*, 1996).

These results have significant management implications for regions such as the Chesapeake Bay watershed, where pollution improvement goals have been based on median rather than episodic conditions (Alexander *et al.,* 1996). Knowing the range of potential episodic water-quality changes in a specific resource area, and how the factors controlling those changes could be influenced by a change in climate, will be important in the development of appropriate management strategies.

Drought Related Episodes of Water-Quality Change. Episodic release of pollutants from the landscape can also occur following periods of drought (Schindler *et al.,* 1996). Re-inundation of wetlands or reservoir littoral zones following a period of low water levels results in a short-duration increase in the production of methyl-mercury in surface waters (Rudd, 1995). Nitrate concentrations increased in Schoharie Creek in southeastern New York following extended drought periods that have occurred since the 1920s (Figure 3). Broad periods (seasons to years) of oscillation between drought and normal precipitation conditions can therefore also result in large, short-duration changes water-quality. Such changes in water quality can be anticipated in management plans if historical records of water quality and volume are available. Episodic changes in water quality are therefore potentially significant in two ways: as the early manifestation of a growing chronic change in condition, or as the result of increased climate variability (e.g., longer dry periods punctuated by average or larger runoff events).

Climate Induced Decreases in Episodic Water-Quality Change. Decreases in the seasonality, frequency, or magnitude of climate extremes or runoff episodes could also cause changes in ecosystems and water quality. In several parts of Canada and the United States, increases in air temperatures over the past 20 years have shortened the period of snow accumulation and the number of days in which lakes and streams are ice-covered (Magnuson *et al.,* 1997; Schindler 1997; Hauer *et al.,* 1997). This change lengthened the growing season by shortening the snow accumulation period, increased the frequency of mid-winter snowmelt runoff, and decreased the magnitude of individual snowmelt runoff peaks (Baron *et al.,* 1997; Melack *et al.,* 1997). Acids deposited in the snowpack of the northeastern United States generally migrate to the bottom of the pack and upper soil column during the winter, from where they are flushed

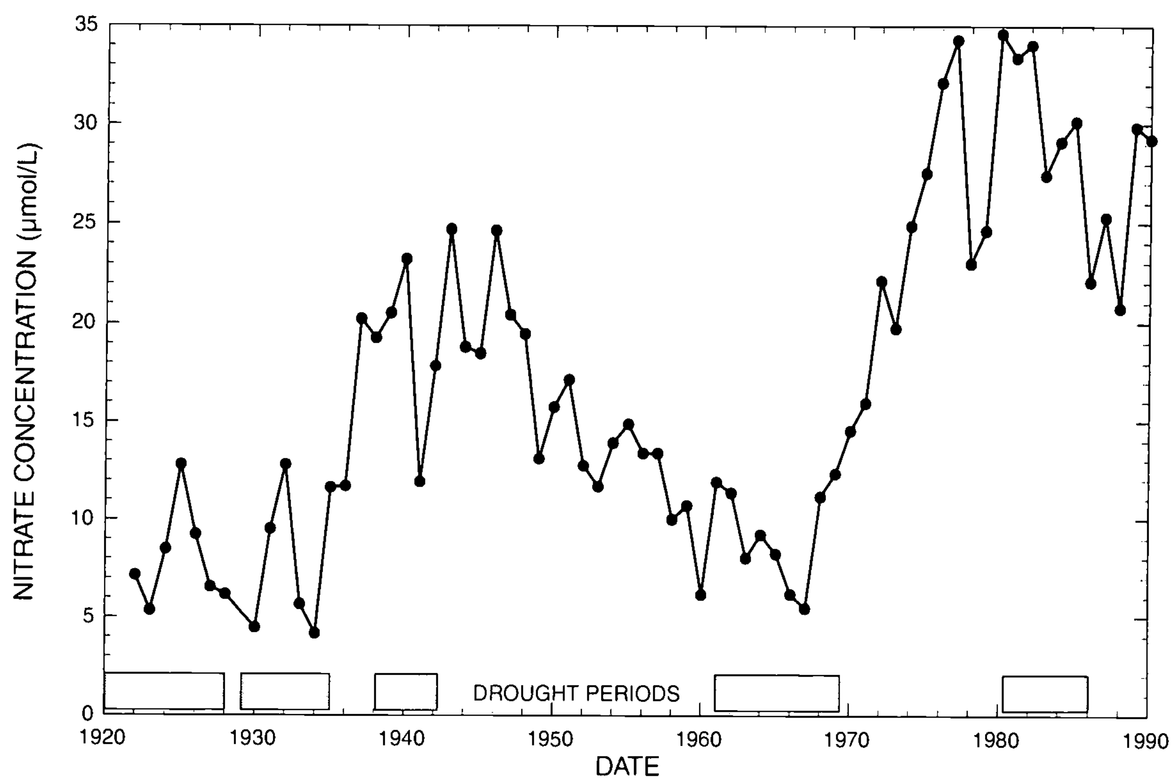

Figure 3. Relation of Nitrate Concentrations in Schoharie Creek, New York, to Periods of Drought During the 1990s.

into the deeper soil and surface waters during the first spring runoff event (Rascher *et al.,* 1987; Murdoch and Stoddard, 1992). Soil microbial processes also continue in unfrozen soils during the winter, and nitrogen-rich byproducts accumulate until snowmelt and rainwater wash it from the soils. Studies using natural isotopic tracers in the Catskill Mountains of New York indicate that very little of the nitrate in melting snow reaches the stream – rather, the nitrate entering the stream has been largely processed by soil microbes (Kendall *et al.,* 1995). In a warmer climate, snowmelt will occur earlier and more frequently, resulting in lower runoff volume and acidity leaving the snowpack during each melt event. Furthermore, a thinning of the snowpack will decrease its capacity to insulate soils from sub-freezing air temperatures, and result in more frequent soil frost (Mitchell *et al.,* 1996). Soil freezing can mobilize nitrogen from soil pools through the frost-induced breakdown of soil organic matter (Mitchell *et al.,* 1996), but will also decrease mid-winter microbial processing of nitrogen. The net effect on water quality during spring snowmelt will depend on the balance between these counteracting factors.

In the Rocky Mountains, mid-winter thaws are less common than in the Northeast, and nitrate is retained throughout a thick, cold snowpack until much later in the spring (Denning *et al.,* 1991; Campbell *et al.,* 1995). A warmer climate in the Rocky Mountains would result in a warmer snowpack which, if warm enough to create mid-winter thaws, would allow concentration of nitrate in the lower snowpack or shallow soils earlier in the spring, and an earlier spring snowmelt (Baron *et al.,* 1995). Snowmelt nitrate in the Rocky Mountains is also partially processed by soil microbes before entering surface waters, although more meltwater directly enters the stream during snowmelt than in the streams described above (Campbell *et al.,* 2000). An earlier snowmelt and longer growing season could lead to increased biological retention of N and decreased annual nitrate export from soils to surface waters. An earlier spring snowmelt in the Colorado Rockies also can have a significant indirect effect on downstream water quality. Spring runoff that fills reservoirs earlier in the year will decrease the flood-control capacity of those reservoirs and decrease the availability of water for irrigation in the following summer (Baldwin *et al.,* 1999). Increased reservoir construction and low streamflows caused by irrigation have been mentioned earlier as causes of water quality decline.

Implications for Water-Quality Management

The results we have reviewed in this paper from the recent literature indicate that the response of water quality to changes in climatic conditions will vary significantly among resource management areas, and types of water bodies (lakes, flowing waters, and estuaries) because of differences in the biological and chemical processes that control water quality. The effect of climate change on water quality also will vary from region to region, and within region among resource types, because of differences in thresholds for change in controlling factors. Aquatic resources that appear to be most vulnerable to water-quality change as a result of a change in climate are those already near their climatic thresholds for chemical change, where competition for water supply between urban, agricultural, and natural uses is high or increasing, or where climate change will act in concert with other existing human-driven stresses to adversely affect water quality. Regions that embody some or all of these characteristics include, but are not restricted to, the southwestern and southeastern regions of the United States, prairie wetlands of the Great Plains, highly productive estuaries and wetlands such as the Chesapeake Bay and Everglades ecosystem, acid-sensitive lakes and streams of the eastern United States, Pacific coastal rivers, and the Arctic and alpine watersheds containing permafrost or glaciers (Naiman *et al.,* 1995; Flather *et al.,* 1984; Kaufman *et al.,* 1988; Wang *et al.,* 1999). The effects of increased temperatures will be critical in some locations where extreme summer temperatures are already near the thermal maxima of some species. The Kansas-Oklahoma-Texas region is one area where the anticipated decline of habitat and water quality will be compounded by increased human demand for water supplies (Covich *et al.,* 1997). If unpolluted freshwaters become increasingly scarce, competing demands for use will become more intense. Increasing human use of water will aggravate water quality problems, both by removal for consumption, and through wastewater returns (Schindler, 1997; Golubev and Biswas, 1985; National Research Council, 1992). Water removal will be particularly detrimental to water quality by reducing dilution of effluents. In the southwestern and southeastern United States, this is already occurring and will worsen if climate warms (Mulholland *et al.,* 1997, Grimm *et al.,* 1997).

However, specific resource areas within any of these regions may be influenced by sub-regional conditions that are inconsistent with regional climatic conditions. For example, empirical evidence of climate change in the Rocky Mountains indicate a cooler, wetter climate since 1951 while the adjacent climate of the Plains has been warming (Williams *et al.*, 1996). While models and empirical data for other regions of the United States and Canada indicate regional synchrony in climate trends (Magnuson *et al.*, 1990), the effect of predicted climate changes on water quality will still largely depend on local factors. From a resource management perspective, the evidence of complexity in ecosystem response to climate change is providing a strong message – that planning for the potential effects of climate on water quality will be most cost-effective and appropriate if it is based on a knowledge of the local biogeochemical and physical processes controlling water quality in the water body being managed.

Monitoring Strategies

As with many environmental issues, the small number of monitoring programs that gather sufficient information to track trends in water quality, hydrology, and associated changes in ecosystem function has limited the discussion of climate-related water-quality changes. Monitoring for trends in water quality and stream discharge has been in decline in recent years, particularly in non-developed landscapes [Committee on Environmental and Natural Resources (CENR), 1997]. To determine climate influences on water quality, a baseline-monitoring infrastructure for tracking trends in discharge, temperature, and chemistry must be defined and maintained. For example, recent studies using baseline water temperature conditions from 1,700 U.S. Geological Survey stream monitoring stations, and extrapolations of published fish temperature tolerances, concluded that stream habitat for cold and cool water fish could decline by 50 percent with a 2-6°C temperature increase (Eaton and Scheller, 1996). Without the long-term data on stream water quality and discharge these analyses could not have been made.

New monitoring programs seldom invest in the methods comparison needed to integrate historical or parallel databases. Methods comparison research is needed to link new and historical databases that were developed by different methods and for different objectives. This linkage could greatly expand our interpretive capabilities across space and time.

Finally, many of the environmental issues facing the Nation today cannot be tracked or solved with our current disaggregated monitoring strategies (ie flow data in one place, forest data in another, fish data in yet a third, etc.). The Nation needs an integrated, cross-media, cross-scale, long-term environmental monitoring strategy, so that the effects of climatic change and other human actions on processes controlling whole-ecosystem function can be studied, understood, and used to inform policy decisions (CENR, 1997). Such a strategy is essential if we are to differentiate between actual and perceived environmental issues, and address them appropriately to avoid both unnecessary regulation and serious environmental problems.

CONCLUSION

The climate research community is largely in agreement that warmer global temperatures and associated shifts in climatic conditions are likely over the next century. While the normal uncertainty in any scientifically-based prediction of the future remains (Mahlman, 1998), basic planning and preparation for such changes has become the conservative track for resource managers to take. The specific effect that climate warming will have on water quality within a manager's jurisdiction will be dependent on the magnitude of the local climate change, local ecosystem characteristics, changes in land use, human behavior and technology, and the new equilibrium achieved among several controlling ecosystem processes. Tracking changes in processes that control water quality within a manager's jurisdiction, and knowing how close the ecosystems are to temperature or moisture-dependent thresholds, will be a necessary part of management programs in the future.

Improved technology for purifying surface waters and management plans that improve water-use efficiency will provide important strategies for mitigating the effects of climate change on water quality in regions where human activity has a direct effect on water quality and volume. In natural landscapes and water bodies, managers have fewer options but may be able to moderate undesirable consequences of climate stress through direct intervention (e.g., chemical additives, removal of exotic species, etc.), or reduction of other stressors such as air pollution.

A review of the scientific evidence presented in the literature to date yields several broad conclusions regarding the potential effect of climate change on water quality:

1. Significant changes in water quality have occurred as a direct result of short-term changes in climate.
2. Episodes of rapidly changing water quality can exceed ecologically significant thresholds more frequently as a result of climate change or variability.
3. Management for the median condition may not always provide adequate protection for sustaining long-term ecosystem health.

4. Water quality in ecological transition zones and areas of natural climate extremes is vulnerable to climate changes that increase or decrease temperature and/or precipitation variability.

5. Changes in land and resource use will have a comparable or greater effect on water quality than changes in temperature and precipitation. Depending on local ecosystem characteristics and human activities, these changes will either exacerbate or mitigate water quality changes directly resulting from changes in climate, and complicate the interpretation of climate change and it's influence on water quality.

As human population and activities on the landscape inevitably increase in the coming decades, tensions between the water resource requirements of humans and natural ecosystems will grow. Establishing long-term monitoring of the ecosystem processes that control water quality, and developing scientifically-rigorous management strategies for protecting water quality during times of climate stress, are worthy goals even before our ability to forecast climate change has fully matured.

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