EPA/600/R 07/085 | March 2008 | www.epa.gov/ncea

United States
Environmental Protection
Agency

Climate Change Effects on Stream and River Biological Indicators: A Preliminary Analysis

National Center for Environmental Assessment
Office of Research and Development, Washington, DC 20460

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Top left: Emerging caddisfly larvae, Oregon State University
Top right: Drought conditions in the Little Medicine Bow River at Boles Spring, R. Swanson, USGS
Bottom left: Dimmer Road East Fork, West Branch, J. Grabarkiewicz
Bottom right: Rainbow trout, W. Davis, http://www.epa.gov/bioindicators

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ABSTRACT

Climate change is projected to affect aquatic ecosystems through changes in water temperature, hydrological cycles, and degree days. These effects will manifest themselves through changes in community composition, phenology, number of reproductive cycles, evolutionary adaptations, and genetic selection. These changes also serve as indicators of climate change effects on ecosystems and could be used to document ecosystem condition. State and tribal water quality agencies use biological indicators to assess ecosystem condition as required by the Clean Water Act. These assessments rely on comparisons of reference and non-reference sites. Climate change, however, will affect organisms at both types of sites, unlike traditional stressors. Therefore, understanding how biological indicators respond to the effects of climate change, what novel indicators may be available to detect effects, how well current sampling schemes may detect climate-driven changes, and how likely it is that current sampling schemes will continue to detect impairment, are important issues in need of discussion. This report is meant to initiate this discussion by providing information on the potential effects of climate change on biological indicators, outlining initial strategies to modify assessment activities to account for climate change effects, and highlighting possible next steps.

Preferred citation:
U.S. Environmental Protection Agency (U.S. EPA). (2008) Climate change effects on stream and river biological indicators: A preliminary analysis. Global Change Research Program, National Center for Environmental Assessment, Washington, DC; EPA/600/R-07/085. Available from the National Technical Information Service, Springfield, VA, and online at http://www.epa.gov/ncea.

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PREFACE

This report was prepared by Tetra Tech, Inc. and the Global Change Research Program (GRCP) in the National Center for Environmental Assessment of the Office of Research and Development at the U.S. Environmental Protection Agency (U.S. EPA). It is intended for managers and scientists working on biological indicators, bioassessment, and biocriteria. The report will provide them with (1) information on the potential effects of climate change on indicator organisms used, (2) initial strategies for adapting their programs to accommodate these environmental changes, and (3) highlight possible next steps. The GCRP established a partnership with the Health and Ecological Criteria Division within the U.S. EPA's Office of Water with State Water Quality Agencies, and with some Tribal Environmental Agencies in an effort to develop a foundation for linking climate change to their monitoring and assessment programs. The background information and research results in this report were presented at a workshop with state and tribal biocriteria managers and scientists from the U.S. EPA and the academic community. The "Introductory Workshop on climate Change Effects on Biological Indicators" was held in Baltimore, MD in March 2007 and focused on climate change effects on river and stream ecosystems. The Workshop provided state and tribal biocriteria managers with information on how climate change may affect their monitoring and assessment programs for water resource protection and restoration. The Workshop included keynote presentations on the current state of scientific understanding of climate change effects on aquatic ecosystems, particularly rivers and streams, climate change trends in the past, present, and future, and models and tools that managers can use to monitor and assess climate change effects. Workshop attendees also participated in breakout sessions in an effort to identify (1) current biological indicators of environmental condition, (2) vulnerabilities of biocriteria programs in water quality agencies, and (3) adaptations of program elements to recognize effects of climate change. Case studies were presented to aid in understanding the technical ramifications of adapting existing biocriteria programs. This report includes background information about climate change effects on rivers and streams and the initial elements of a framework that state and tribal biocriteria managers can use to modify their programs in response to these effects. The framework elements described in this report and presented at the workshop are (1) an approach for identifying biological indicators sensitive to climate change, (2) an analysis for detecting climate change effects, and (3) methods for continuing to detect impairment under climate change. Workshop participants made recommendations about research needs and information gaps. This information is presented in this report and was used in part to derive the proposed next steps that conclude this report.

AUTHORS, CONTRIBUTORS, AND REVIEWERS

The Global Change Research Program, within the National Center for Environmental Assessment, Office of Research and Development, is responsible for publishing this report. This document was prepared by Tetra Tech, Inc. under Contract No.GS-10F-0268K, U.S. EPA Order No. 1106. Dr. Britta Bierwagen served as the Technical Project Officer. Dr. Bierwagen provided overall direction and technical assistance, and she contributed as an author.

AUTHORS

Center for Ecological Sciences, Tetra Tech, Inc., Owings Mills, MD
Anna Hamilton, Michael Barbour, Jeroen Gerritsen, Michael Paul

U.S. EPA
Britta G. Bierwagen

REVIEWERS

This report benefited greatly from the comments and suggestions of the following reviewers.

U.S. EPA Reviewers
Mark Bagley, Evan Hornig*, Sue Norton, Lester Yuan

Other Reviewers
Paul Kazyak (Maryland Department of Natural Resources, Steve Ormerod (Cardiff University, UK), Mark Southerland (Versar)

ACKNOWLEDGMENTS

The authors thank the Global Change Research Program staff in NCEA, particularly Susan Julius, Tom Johnson, Chris Weaver, and Jordan West, for their input and advice throughout the development of this project. We also thank staff in the Office of Science and Technology in the U.S. EPA's Office of Water for their input and assistance. We appreciate all of the comments from workshop participants. Their thoughts and ideas contributed greatly to this report and the formulation of further research.

* Present affiliation with U.S. Geological Survey.

EXECUTIVE SUMMARY

Climate change can have a variety of effects on aquatic species. Changes in air temperature and precipitation patterns are reflected in changes in water temperature, hydrological cycles, and degree days. These alterations, in turn, affect aquatic ecosystems, whose responses can be documented through changes in community composition, phenology, number of reproductive cycles, evolutionary adaptations, and genetic selection. One method for documenting changes in ecosystems is through indicators that are particularly sensitive to the changes or stressor of interest—in this case climate. Some potential indicators of climate change effects include the following: ratios of drought tolerant to intolerant mussel species; ratios of invertebrate response guilds that indicate hydrological status; and changes in community composition. Indicators of changes in composition include shifts from cold- or cool-water fishes to warm-water fishes; shifts from species associated with hydrologically stable to variable conditions; and declines in particularly sensitive species, such as salmon, brook trout, or darter species. The goals of this report are to provide managers and scientists working on biological indicators, bioassessment, and biocriteria with information on the potential effects of climate change on indicator organisms used and initial strategies for adapting their programs to accommodate these environmental changes, and to highlight possible next steps.

Biocriteria programs exist in state and tribal water quality agencies to assess the biological status and health of ecosystems as required by the Clean Water Act. The United States Environmental Protection Agency (U.S. EPA)'s Office of Water has developed guidance documents for states and tribes on bioassessment methods and biocriteria establishment. River and stream ecosystems were among the first for which methods were developed. The general approach of bioassessment includes defining reference conditions so that impaired sites can be defined by comparison with these natural or minimally impacted sites. Currently, there is no mandate or guidance for biocriteria programs to include climate change effects in the design of monitoring programs or assessment of impairment. However, as a stressor, climate change will impact both reference and non-reference sites, unlike the more conventional anthropogenic stressors currently considered in bioassessment.

Because climate change will affect both, reference and non-reference sites, bioassessment programs would benefit from the collection of data on climate change effects in their systems. These data may come from indicators that detect such effects. Biological indicators that are currently used in bioassessment programs have been selected for their sensitivity to certain environmental stressors. Knowledge about their sensitivities allows a general extrapolation of their response to other environmental variables related to climate change. Therefore, most indicators that are sensitive to the conventionally considered anthropogenic stressors are also expected to have some sensitivity to climate change: For example, changes in temperature and precipitation patterns that affect stream flow. Biological indicators that are only (or at least predominantly) sensitive to climate change may be possible to define, but are likely to be novel, at least in terms of their application in state bioassessment programs. Although review of the scientific literature may lead to the identification of novel indicators, these indicators will need to be easy to measure and practical to implement by state programs in order to be widely adopted.

Indicators that are specifically sensitive to climate change effects are only one approach that programs could use to detect effects. The case studies discussed in this report present additional methods and considerations to aid in the detection of climate change effects. The first case study discusses issues of sampling power needed to detect effects using one type of indicator of climate change and the second case study examines how climate change may affect the ability of current monitoring programs to detect impairment due to conventional stressors.

The first case study, Assessing Trends, focuses on sampling power. The power to detect effects depends on the effect size. In this case, that means the species' loss rate due to increases in water temperature. The case study explores a low and high species loss rate and low versus high temperature change scenarios. Using data from one long-term data set and sampling scheme, it would take 15 years to detect effects due to climate change under the high loss rate and high temperature change scenario. The other extreme, low loss rate and low temperature change, would take more than 100 years to detect. As expected, an increasing number of samples will help detect effects sooner.

The second case study, Accounting for Trends, examines the ability of bioassessment programs to continue to detect impairment due to conventional stressors in the face of climate change. The analysis shows that climate change effects will decrease the ability of states to discriminate between reference and impaired sites, particularly if reference sites are already somewhat stressed. These results underscore the importance of monitoring sentinel sites, sites that are revisited during each sampling cycle, in order to detect deterioration of condition at reference sites due to climate change.

The results of these case studies, preliminary analyses of indicator sensitivities, and reviews of the literature of climate change effects on aquatic ecosystems were presented at the Workshop for state biocriteria managers of rivers and streams. Their responses to this information led to recommendations for additional research and a variety of mechanisms for assistance to states from U.S. EPA concerning climate change effects in these ecosystems. The large number of recommendations suggests that it is important to continue this dialogue by conducting further research and other activities leading to more specific recommendations and assistance for state programs. This information could then be used to modify state programs to account for climate change effects and to ensure that management goals continue to be reached.

State biocriteria managers also outlined a number of actions that they could take now in response to the information presented. In particular, their response reflects an understanding that climate change will affect the entire ecosystem, and, therefore, regular and repeated monitoring of reference and sentinel sites to collect biological, hydrological, and temperature information will be particularly valuable to detect and control for climate change effects.

The recommendations for further research also lead to potential next steps. These steps include (1) conducting another workshop for biocriteria managers of other aquatic ecosystems; (2) conducting more in-depth analyses of climate change effects on river and stream bioassessment programs in different regions of the U.S.; (3) disseminating information on regional climate change effects on biological indicators; and (4) coordinating information across U.S. EPA and state agencies to evaluate trends in bioassessment data.

1. INTRODUCTION

Changes in climate affect ecosystems, and, therefore, also their management. State and tribal water quality (WQ) programs fulfilling Clean Water Act of 1972 CWA) requirements will need to take these climate change-induced effects into account. Currently, state and tribal assessment programs include biological monitoring in addition to physical and chemical parameters, because organisms, and the communities they comprise, integrate the effects of physico-chemical changes over time and space and provide any trends detected with ecological relevance. It is for this reason that understanding the biological and ecological responses to climate change, as well as interactions between climate change effects and other environmental stressors, is important for continued operation of bioassessment programs and interpretation of bioassessment results.

Human activities and natural factors have already changed the climate; these trends are likely to continue into the future (IPCC, 2007). There is now high confidence that anthropogenic emissions of greenhouse gases and aerosols have resulted in warming, with evidence of globally increasing air and ocean temperatures, melting of snow and ice, and rising sea levels (IPCC, 2007; Rahmstorf et al., 2007). Global air temperatures have increased about 0.6°C over the last 30 years and 0.8°C over the last century, and global ocean temperatures are probably as warm now as they were during the Holocene maximum (about 5000 to 9000 years ago) (Hansen et al., 2006). Observed increases are greater over land masses than over oceans, and they are greatest at high latitudes in the Northern Hemisphere (Hansen et al., 2006). Extreme cold days, cold nights, and frost have been less frequent; hot days, hot nights and heat waves have been more frequent (IPCC, 2007). The third Intergovernmental Panel on Climate Change (IPCC) report (2001) revealed that the diurnal temperature range was decreasing; however, evaluation of more extensive data in the fourth assessment report shows that daytime and nighttime temperatures are actually increasing at comparable rates (IPCC, 2007). An understanding of the potential consequences of these climatic changes for aquatic ecosystems is an initial step that will assist water resource managers in modifying their programs to ensure that they will continue to meet their management goals.

The goals of this report are to provide managers and scientists working on biological indicators, bioassessment, and biocriteria with information on the potential effects of climate change on indicator organisms used, initial strategies for adapting their programs to accommodate these environmental changes, and highlight possible next steps. This report supports these goals by presenting background information about climate-change effects on rivers and streams (see Section 1), an overview of bioassessment programs (see Section 2), and the initial elements of a framework that state and tribal biocriteria managers can use to modify their programs in response to these effects. The framework elements described in this report are (1) a proposed approach for identifying biological indicators sensitive to climate change (see Section 3), and (2) two preliminary case studies evaluating selected aspects of biological assessment programs (see Sections 4 and 5).

Biological assemblages integrate effects from all impinging sources of stress, including "conventional" anthropogenic stressors, which are commonly the focus of state programs assessing and regulating water quality, and any other significant source of environmental change, including climate change. This integrative characteristic makes biological assemblages effective monitoring tools, but it also means that the analyses must reasonably account for all major sources of stress in order to attribute observed responses to particular sources of stress in a reliable manner. This attribution allows for more effective regulation of the stressor and/or management of the resource. The ongoing success of biological monitoring and assessment programs will require an understanding of *both* an understanding of what climate-associated changes are occurring in monitored aquatic communities and how monitoring programs can account for them. Accounting for climate-change influences will support effective attainment of management goals using monitoring program results as a foundation.

The two case studies included in this report examine how climate-change effects can be taken into account through program design and/or analytical approaches and how climate change may affect the ability of biological monitoring and assessment programs to meet key goals. Climate change can be viewed as a "global stressor" that affects both reference and non-reference locations monitored for the effects of more "conventional" stressors. The ability to account for climate change requires (1) an understanding of how vulnerable monitoring data are to climate-change effects, and (2) how effectively differences that are a result of climate change can be detected within existing monitoring programs.

The first case study (see Section 4) describes several important temporal aspects of change detection. The second case study (see Section 5) examines methods for continuing to detect impairment under climate change. More detailed examinations of these and other important components of biological assessment programs are essential before comprehensive recommendations for adaptation of bioassessment programs to climate change can be made. Preliminary recommendations, derived from these analyses and the participants in the "Introductory Workshop on Climate Change Effects on Biological Indicators," and a summary of proposed next steps conclude this report (see Sections 6 and 7).

1.1. BIOINDICATORS, BIOCRITERIA, AND THE CLEAN WATER ACT

The CWA of 1972 (Federal Water Pollution Control Act, Public Law (P.L.) 92-500) as amended in 2002 (P.L. 107-303, November 27, 2002) has as a stated goal and policy (Section 101(a)):

> "…to restore and maintain the chemical, physical, and biological integrity of the Nation's waters."

The concept of biological integrity has received much attention since passage of the CWA. It is commonly defined as "the ability to support and maintain a balanced, integrated, and adaptive community with a biological diversity, composition, and functional organization comparable to those of natural aquatic ecosystems in the region" (Karr et al., 1986; Karr and Dudley, 1981; Frey, 1977). This wording highlights some key attributes of biological communities that are fundamental to preserving "integrity" (diversity, composition, and functional organization), and it alludes to a basic element of the approach used in bioassessment, which is comparison to existing natural communities, or reference conditions.

The use of biological monitoring and assessment to establish criteria is mandated in Section 303(c)(2)(B) and 304(a)(8) of the CWA. Biological criteria (biocriteria), derived from biological monitoring and assessment, provide narrative and numeric targets that define the desired condition of communities of aquatic organisms inhabiting streams and rivers where water quality is subject to regulation. Biocriteria and biological assessment (bioassessment) thus provide a valuable and direct regulatory mechanism for protecting biological resources at risk from chemical, physical, and biological impacts. The CWA requires that states and tribes designate aquatic life uses (i.e., environmental goals) for their waters that appropriately address biological integrity and adopt biological criteria necessary to protect those uses (Barbour et al., 2000).

Biologists and other natural resource scientists develop biocriteria by using accepted scientific principles to characterize the regional reference conditions for the different water bodies found within a state or tribal nation (Barbour et al., 2000). Effects of climate change are broad and need to be considered in tracking trends in regional reference conditions that form the basis of assessing ecological condition. Biological assessment programs are now widespread throughout the states (U.S. EPA, 2002) and are best served by indicators that can be used for multiple purposes. Water resource agencies in the 50 states and several tribes are in various stages of development and implementation of bioassessment methods (Barbour et al., 2000). Essentially, the multiple purposes of bioassessment can be reduced to two basic questions: (1) asking whether a water body meets, or exceeds, an impairment threshold, and (2) asking whether the biological condition of a water body is degraded or improved compared to an earlier time, an upstream or a nearby site (Barbour and Gerritsen, 2006). Climate change influences both of these questions.

1.2. CLIMATE CHANGE EFFECTS ON AQUATIC ORGANISMS AND ECOSYSTEMS

There is a substantial weight of evidence, summarized in several reviews and meta-analyses, of ecological changes that are linked to existing climate change (Walther et al., 2002; Root et al., 2003; Parmesan, 2006). Figure 1-1 is a conceptual model of how climate changes affect aquatic ecosystems, and some possible ways in which ecological systems respond, which can be measured using indicators. These analyses identify several categories of ecological responses expected from climate change, including (1) changes in range and distribution of species; (2) changes in phenology; and (3) evolutionary effects on morphology, behavior, and genetic frequencies, due to altered selection regimes. These, in turn, are predicted to alter community composition and interactions, as well as ecosystem processes, including production and material cycling. A few of the more common examples are discussed here.

1.2.1. Changes in Ranges, Distributions of Species, and Community Composition

Water temperature drives many biological functions in aquatic invertebrates and fish, including growth and metabolic rates, reproduction, feeding, and survival. Many fish and insect species have fairly narrow temperature range tolerances, and these narrow ranges influence their distribution. Temperature regime determines distributions of species in relation to temperature tolerances and adaptations combined with competitive interactions, effects on food supply, and other factors (e.g., Sweeney and Vannote, 1978; Vannote and Sweeney, 1980; Matthews, 1998).

Changing thermal regimes are expected to shift species ranges to the north (and/or to higher elevations); species at the southern limits of their ranges will migrate or suffer local extinctions. However, in many areas northward or upstream migrations of certain aquatic species may be limited by barriers to dispersal such as habitat fragmentation due to dams and reservoirs, deforestation, and water diversions (Poff et al., 2002; Moore et al., 1997; Covich et al., 1997; Smith, 2004; Hawkins et al., 1997). There is some experimental evidence for dispersal of Ephemeroptera-Plecoptera-Trichoptera (EPT) taxa across watersheds in Wales, such that natural fragmentation of river basins is not directly an impediment to recovery from large-scale disturbance (Masters et al., 2007). However, northward migrations may be limited in regions including the southwest and southern Great Plains of the U.S. where most drainages flow east and west (Poff et al., 2002). Species that are already restricted to headwater streams may be displaced (Poff et al., 2002). In the U.S., from 36% (Mohseni et al., 2003) to 50% (Eaton and Scheller, 1996) of cold-water fish habitat, and up to 15% of cool-water habitat may disappear

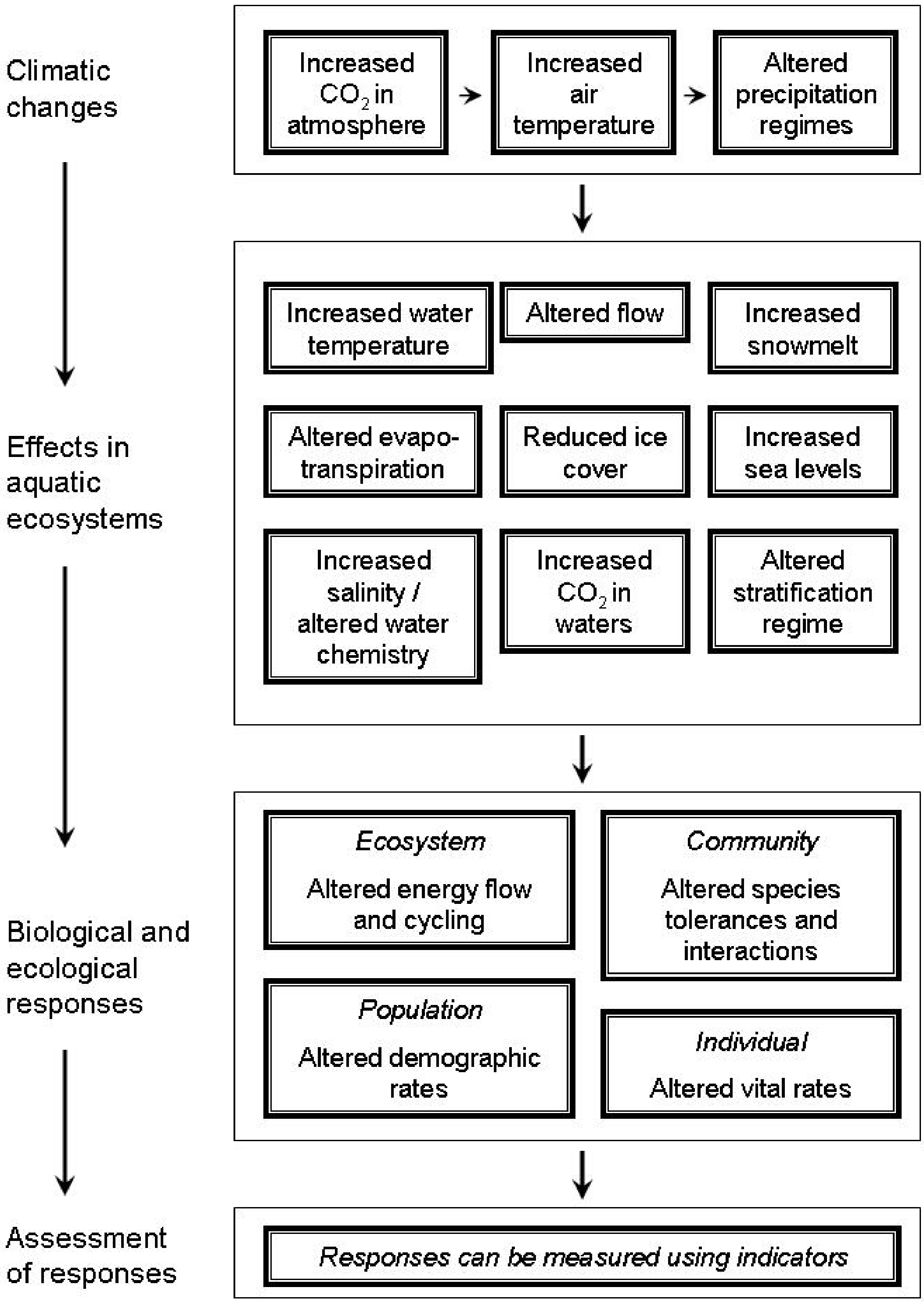

Figure 1-1. Conceptual diagram of how climate change affects aquatic ecosystems and the possible ecosystem responses that can be measured using biological indicators.

(Mohseni et al., 2003) due to the warming projected for a doubling of atmospheric CO_2 concentrations. Fish with the smallest geographic ranges are the most vulnerable. Rahel et al. (1996) estimate habitat losses for cold-water fish species in the Platte River, Wyoming ranging from 7–76% for temperature increases of 1–5ºC. They anticipated potential population fragmentation as cold-water species were progressively limited to colder headwater stream reaches; fragmented populations (based on models for cut-throat trout) may be more susceptible to extirpation if populations are isolated with limited carrying capacity or interactions with other populations (Hilderbrand, 2003). In the Mid-Atlantic Appalachian mountains, cold-water brook trout are near the southern limit of their range, and suitable habitat is mainly found at higher elevations. Projected temperature increases could raise the elevation at which acceptable temperatures occur by 700 m, effectively eliminating most brook trout habitat in this region (Moore et al., 1997).

Daufresne et al. (2003) documented species replacements, range shifts, and variations in community composition for both fish and macroinvertebrates in the Upper Rhone River in France associated with increasing water temperatures from atmospheric warming. Increased fish abundances were associated with increased temperatures and lower flows during the reproductive period (April–June). In moorland and forest streams in Wales, directional climate change (increasing temperatures) decreased spring macroinvertebrate abundances over a 25 year period, yielding an estimated average of 21% reduction in abundance per 1°C of temperature increase, and in combination with the North American Oscillation (NAO) accounted for 70% of interannual variation (Durance and Ormerod, 2007).

There are several other examples of community responses to climate variables in different regions of the U.S. In the Southeast, freshwater mussels are especially vulnerable to drought, along with the corresponding low flows and depressed dissolved oxygen (DO) levels, leading to increased mortalities and local extinctions (Golladay et al., 2004). In the Great Plains, where many fishes already exist at or near their thermal tolerance limits as a result of high temperatures and low flows typical of shallow water habitats, increasing temperatures due to climate change are expected to result in increased extinctions of endemic and local species populations (Covich et al., 1997). Finally, in the Southwest, the stream fauna is typically highly resilient and adapted to disturbance, but nonetheless is vulnerable to habitat losses that could accompany increased runoff variability (Grimm et al., 1997). Biological effects may be manifested as changes in relative abundance, species losses (local extinctions), and reduced diversity.

In addition to temperature effects, projected changes in stream flow from climate change may alter community structure. When considering climate change alone, the Sacramento River could lose 10–18% (low and high climate change scenarios) of its fish species by 2080; the

Colorado River 0–5% of fish species; the Rio Grande River 0–5%; and the Sabine River 11–13% (Xenopoulos et al., 2005). Xenopoulos et al. (2005) predict high risk of species extinctions for subtropical and tropical rivers with a rich endemic fauna and note the vulnerability of fish species that require seasonal floodplain connection for life cycle completion. Using a similar approach, Xenopoulos and Lodge (2006) estimated potential fish richness losses associated with 20–90% reductions in discharge for several rivers in two regions of the U.S., and report a range of 2–30% of fish species lost in rivers of the Lower Ohio-Upper Mississippi Basin, and 3–38% of fish species lost in the Southeastern U.S. Changes in timing of spring flows resulting from climate change may have the greatest effects on spring spawning fishes in the Northeast and may alter the survival of Atlantic salmon by changing migration timing and coincidence with optimal conditions for survival (Hayhoe et al., 2007). It should be noted that climate-driven changes in hydrology are considered potentially difficult to define because inter-annual rainfall variability is large relative to trends predicted from climate change (see Wilby, 2006).

1.2.2. Changes in Phenology

Warmer water may increase the growth rates of aquatic invertebrates and result in earlier maturation (Poff et al., 2002). In a mesocosm experiment using the mayfly *Cloeon dipterum*, temperature increases alone had little effect on nymph abundance, and only small effects on body length, though emergence began earlier in the year (McKee and Atkinson, 2000). McKee and Atkinson (2000) also show that for treatments with both increased temperatures and nutrients, both nymph abundance and size increase. For a Japanese species of mayfly (*Ephoron shigae*), cumulative degree days and time of emergence are significantly correlated, explaining 80–90% of the variation in emergence date, depending on whether the analysis is done for all individuals or separately by sex (Watanabe et al., 1999). For at least this species and most likely for related species, increasing water temperatures associated with climate change will likely result in earlier emergence of mayflies due to an earlier accumulation of degree days.

1.2.3. Evolutionary Effects

Evolutionary changes may play a small role in species' responses to climate change through adaptation (Parmesan, 2006; Berteaux et al., 2004; Hogg and Williams, 1996). These include processes that have been documented for range shifts due to hybridization and novel adaptations, chromosomal inversions that allowed tolerance of warmer temperatures in southern range sub-populations, and body size responses to increasing temperatures due to genetic plasticity (Parmesan, 2006). However, capacity for evolutionary responses of species will be limited by range of genetic diversity and generation time, with species characterized by small, short-lived and abundant individuals more likely to respond adaptively (Bradshaw and

Holzapfel, 2006; Berteaux et al., 2004; Hogg et al., 1995). Extinctions are still expected as a likely consequence of directional climate change even with evolutionary changes: in part because the mean phenotypes lag behind optimal phenotypes, and the rates of environmental change can outpace estimated maximum sustainable rates of evolution (Bradshaw and Holzapfel, 2006; Berteaux et al., 2004; Burger and Lynch, 1995).

Parmesan (2006) points out that while there are local examples of adaptations to changing environmental conditions, there is little evidence in the geologic record of the appearance of novel genotypes in species in response to the larger climate changes associated with glaciations and interglacial periods. It is expected that species' responses to climate change will primarily be through range shifts and extinctions rather than through evolution.

1.2.4. Ecosystem Effects

There is evidence that projected increases in CO_2 will reduce the nutritional quality of leaf litter to macroinvertebrate detritivores. Reduced litter quality would result in lower assimilation and slower growth (Tuchman et al., 2002). While seemingly a secondary climate-change effect, changes in these processes could have food web implications: altered stream productivity that impacts fish and other consumers. In contrast to this, Bale et al. (2002) found little evidence of the direct effects of CO_2 on insect herbivores and instead discuss a range of temperature effects (including interactions with photoperiod cues) on various life history processes that affect ecological relationships.

It is not clear whether changes in nutrient loading due to climate change will have any effects on streams and rivers. Effects of nutrient enrichment in streams are highly variable, due to questions about which primary nutrient (nitrogen or phosphorus) is limiting, shading (light availability), water clarity, flow regime, and available substrate for periphyton growth (e.g., Dodds and Welch, 2000). In general, nutrient enrichment leads to changes in the algal and diatom community composition of a stream, and sometimes, in some streams, to increased production and chlorophyll concentrations, leading to changes in primary invertebrate consumers (e.g., Gafner and Robinson, 2007) which could cascade through the community (Power, 1990; Rosemond et al., 1993).

Changes in the distribution and intensity of precipitation may induce related changes in nutrient loading to streams from runoff. However, it is not clear if total nutrient loading to a stream will change with altered precipitation. For example, increased precipitation does not increase nitrogen available on the land surface to run off. However, changes in precipitation patterns combined with other changes in land use, for example, may affect nutrient loadings.

1.3. ORGANISMAL/BIOLOGICAL INDICATORS OF CLIMATE CHANGE

Since organisms respond to climatic variability and trends, some of these responses may be useful as indicators of climate change (Figure 1-1). This section describes several candidate indicators based on the current literature. Section 3 identifies novel indicators and species traits that may be sensitive to climate change.

Golladay et al. (2004) surveyed mussel species during drought conditions, identifying several low flow-sensitive species (e.g., *Lampsilis straminea claibornensis*, *Villosa villosa*, and *Lampsilis subangulata)*. Other species (*Pleurobema pyriforme*, *Mediunidus penicillatus*) show signs of drought intolerance due to decreased DO concentrations during low flows. Mussel species less affected by drought-induced low flow and low oxygen levels include *Elliptio complanata/icterina*, *Villosa vibex*, and *Villosa lineosa* (Golladay et al., 2004). A comparison of drought intolerant to drought tolerant mussel species may be an indicator of hydrologic variability or drought possibly due to climate change.

Golladay et al. (2004) suggest that wetland invertebrates could be divided into four response guilds to indicate hydrologic status that may be adaptable to river/stream systems: (1) overwintering residents that disperse passively, including snails, mollusks, amphipods, and crayfish; (2) overwintering spring recruits that require water availability for reproduction, including midges and some beetles; (3) overwintering summer recruits that only need saturated sediment for reproduction, including dragonflies, mosquitoes, and phantom midges; and (4) non-wintering spring migrants that generally require surface water for overwintering, including most water bugs and some water beetles. Changes in density-weighted ratios of these response guilds could be used as indicators of climate driven changes in hydrologic conditions over time.

Monitoring changes in community composition, including shifts from cold- and cool-water dominated systems to warm-water communities, may be another good indicator. It is expected that cool-water and warm-water fishes will be able to invade freshwater habitats at higher latitudes, while cold-water fish will disappear from low latitude limits of their distribution where summer temperatures already reach fish maximum thermal tolerances (Carpenter et al., 1992; Tyedmers and Ward, 2001). In east-west drainages fish may not be able to find thermal refuge and may experience local extinctions (Carpenter et al., 1992). However, cold-water fish that do persist at higher altitudes and latitudes may not experience as many winter stresses, and their ranges at may expand with increased duration of optimal temperatures (Carpenter et al., 1992; Melack et al., 1997).

Salmon species are known to prefer cold water temperatures and a number of studies have investigated the impact of potential climate changes on these fish species. Pacific salmon may be particularly sensitive to climatic changes because suitable habitat is projected to decrease due to altered thermal regimes (Schindler et al., 2005). Research has linked increased river temperatures with increased mortality of sockeye salmon, particularly in species which migrate during the summer when river temperatures are at their highest (British Columbia Ministry of Water, Land and Air Protection, 2002). Warmer waters cause increased energy use and bacterial/fungal infections in salmon, decreasing the likelihood that they will survive their migration and be equipped to spawn (British Columbia Ministry of Water, Land and Air Protection, 2002). Melack et al. (1997) suggest that higher temperatures will lead to reduced growth and increased mortality of sockeye salmon in freshwater and marine waters. In freshwater, Melack et al. (1997) suggest that there will be greater inputs of nutrients during the winter season rather than in the spring as well as a longer period of thermal stratification, which would likely lead to lower planktonic productivity and smaller juvenile sockeye salmon. However, a study in southwestern Alaska by Schindler et al. (2005) shows increased juvenile growth rates, because the warmer water temperatures increase the length of the growing season due to earlier ice breakup and increase zooplankton densities, prey for juvenile salmon. In marine waters, Melack et al. (1997) note that all of the growth and gathering of excess energy reserves is done during the time that Fraser River sockeye salmon spend in the ocean. However, general circulation models (GCMs) forecast increases in sea surface temperatures and weaker north-south pressure gradients over the north-east Pacific Ocean, which could weaken ocean upwelling and reduce secondary productivity (Melack et al., 1997; IPCC, 2007). The higher temperatures and reduced zooplankton would likely lead to smaller adult sockeye with fewer and smaller eggs and less energy reserves (Melack et al., 1997). In addition, the Fraser River sockeye salmon that Melack et al. (1997) focused on in their analysis already live at the southern edge of their thermal range. Melack et al. (1997) also reviewed the potential impacts of climate change on salmon species spawning such that increased winter flows and spring peaks may reduce salmonid egg to fry survival. For example, higher spring peaks in flow and warmer water temperatures may cause earlier emergence of fry and migration of pink and chum salmon fry to estuaries at a time when their food sources have not developed adequately (Melack et al., 1997). Similarly, low summer flow could lead to a decrease in available spawning and rearing habitat (Melack et al., 1997). For species that spawn in the fall, including many salmonid species, an increase in scouring resulting from higher precipitation rates in winter could result in the reduced survival of eggs (Tyedmers and Ward, 2001).

Some research has shown that fish species living in streams and rivers in semi-arid regions may be more susceptible to climate impacts than species living in streams and rivers in sub-humid regions. Milewski (2001) found that species richness, number of insectivorous cyprinid (minnow) species, and number of species intolerant of degraded water quality and habitat were lower in the semi-arid region of their study suggesting that fish species rebound from low and high water levels more easily in sub-humid regions than in semi-arid regions. Poff and Allan (1995) also investigated hydrologic variation in streams and the impact of hydrologic variability on fish species. For the sites in their study, fish assemblages that were associated with the hydrologically variable streams had the following characteristics: generalized feeding strategies, association with silt and general substrata, slow velocity, headwater affinity, and tolerance to silt. Fish species occurring at more than 50% of the hydrologically variable sites but less than 50% of the stable sites included *Ameiurus melas* (black bullhead), *Perca flavescens* (yellow perch), *Notemigonus crysoleucas* (golden shiner), *Ameiurus natalis* (yellow bullhead), and *Lepomis gibbosus* (pumkinseed). Fish species occurring only at hydrologically variable sites (often only one or two sites total) include *Fundulus notatus* (blackstripe topminnow), *Lepisosteus osseus* (longnose gar), *Lepisosteus platostomus* (shortnose gar), Amia calva (bowfin), *Anguilla rostrata* (American eel), and *Dorosoma cepedianum* (gizzard shad) (Poff and Allan, 1995). Fish species occurring at more than 50% of the stable sites and less than 50% of the hydrologically variable sites include *Moxostoma macrolepidotum* (shorthead redhorse), *Micropterus dolomieu* (smallmouth bass), *Hypentelium nigricans* (northern hog sucker), *Rhinichthys cataractae* (longnose dace), and *Notropis rubellus* (rosyface shiner).

Cold-water fish species, and salmon species in particular, may be good indicators of climate-change effects in streams and rivers. To use a salmon species or any fish species as an indicator, one must be sure not to count or include fish that may have been stocked rather than occur naturally in a particular stream or river. Native brook trout populations may be a useful climate-change indicator for streams and rivers for certain regions since they often live at the edge of their thermal tolerance; therefore a decline in brook trout numbers in a certain area may be a sign of climate impacts. A decline in brook trout numbers would not always necessarily indicate climate effects, however, because a decline in this species could also be due to other stressors or even species competition. Species with widespread ranges and high thermal tolerance such as largemouth bass, carp, channel catfish, and bluegills would generally not be good indicators of climate impacts since they are relatively insensitive and their ranges extend south into Mexico. Another possible effect of increased water temperatures is to reduce DO levels in stream waters. Darter species are sensitive to benthic oxygen depletion because they feed and reproduce in benthic habitats (U.S. EPA, 1999), making them another potential indicator of climate change.

2. STATE BIOASSESSMENT PROGRAMS—RIVERS AND STREAMS

Aquatic organisms integrate the effects of all sources of stress that impinge on them, including “conventional” anthropogenic stressors, which are commonly the focus of state programs assessing and regulating water quality (e.g., point and non-point sources of pollutants, habitat alterations, landscape-level changes), and any other significant source of environmental change, including climate change. Because organisms reflect all sources of environmental disturbance to which they are exposed over time, assessments of biological communities can provide information that may not be revealed by measurements of concentrations of chemical pollutants or toxicity tests (U.S. EPA, 1999; Rosenberg and Resh, 1993; Resh and Rosenberg, 1984). Bioassessment thus provides a means of assessing not just biological condition or health but also overall ecological integrity of stream and river systems.

Their integrative characteristic makes biological assemblages effective monitoring tools, but it also means that all major sources of stress must be reasonably accounted in order to reliably attribute observed responses to particular sources of stress and to effectively regulate the stress and/or manage the resource. The ongoing success of biological monitoring and assessment programs will require an understanding of what climate-associated changes are occurring in monitored aquatic communities and how monitoring programs can account for them. Accounting for climate-change influences will support effective attainment of management goals using monitoring program results as a foundation.

2.1. BIOASSESSMENTS OF RIVERS AND STREAMS

Since the mid-1980s, the U.S. EPA has worked interactively with national, regional, and state agency biologists and other nationally recognized experts to develop approaches and technical guidance for implementation of biological assessment. Resulting guidance included U.S. EPA’s Rapid Bioassessment Protocols (RBPs) (U.S. EPA, 1989), which provided a technical framework for using benthic macroinvertebrate and fish assemblage data as a direct indicator of ecological health. These were updated with the additional consideration of periphyton communities, in 1999 (U.S. EPA, 1999). As a complement to the bioassessment development, procedures for developing narrative biocriteria were published in 1992 (U.S. EPA, 1992), and for developing biocriteria for streams and rivers in 1996 (U.S. EPA, 1996). Following this initial focus on streams and rivers, bioassessment technical guidance was developed for lakes and reservoirs (U.S. EPA, 1998), estuaries, and coastal marine waters (U.S. EPA, 2000), and wetlands (U.S. EPA, 2002).

Any well designed monitoring and assessment program (in this case, bioassessment) is inherently anticipatory in that it will provide information for present needs and those not yet determined (Yoder and Rankin, 1995). Programs that are adaptable to immediate and future needs are also cost efficient (Barbour et al., 2000). Regardless of approach, all bioassessment programs adhere to some basic technical elements: (1) selection and calibration of appropriate biological indicators, (2) determination of reference condition or benchmarks for assessment, and (3) use of standardized protocols that maximize the information on the indicators, optimize gear efficiency, and minimize variability due to sampling error (Barbour et al., 2000).

Biological indicators are considered the best overall measure of ecological integrity from multiple stressors, because of their continuous exposure to magnitude, frequency, and duration to the synergistic effects of chemical and non-chemical stressors; therefore, these indicators need to be calibrated on a regional basis and possess a range of sensitivity to the various stressors, including climate change. Section 2.2 addresses the more common and relevant components of bioindicators.

Reference conditions are established in various ways (U.S. EPA, 1996). However, the use of actual reference sites in a regional population of minimally disturbed sites is ideal for calibrating a quantitative means of assessing ecological condition. The influence of climate change will affect the maintenance of stable reference conditions. A gradient of degradation of reference sites over time is plausible, and it is an important factor for establishing a credible bioassessment. Bioassessment programs throughout the U.S. have established viable reference conditions for assessment. Many programs also establish sentinel sites that are assessed during each monitoring cycle. The continued monitoring of sentinel sites within the reference population will be important to identify where on the condition gradient a set of reference sites may be for a state or tribal program.

Standardized protocols are a feature of all bioassessment programs. However, these may vary among agencies, and they are not necessarily comparable between jurisdictions. As the effects of climate change upon bioassessment programs are better described, modification of protocols to capture more sensitive indicators or to collect specific attributes of established indicators may be necessary.

2.2. BIOINDICATORS USED IN STATE PROGRAMS—RIVERS AND STREAMS

The choice of bioindicators has some commonality throughout the U.S. Benthic macroinvertebrates are the most common assemblage used for bioassessment in streams and rivers among the states and tribes (U.S. EPA, 2002). Fish assemblages are the second most prevalent assemblage used to assess biological condition. The U.S. EPA recommends the use of multiple assemblages in programs to increase the robustness of the overall bioassessment

(U.S. EPA 1996). Periphyton or algae is of interest to many states as an added assemblage for use in their monitoring and assessment program because of their sensitivities to stressors.

Tables 2-1 and 2-2 list the common metrics, which are measures of change in features or attributes of the structure and/or function of the assemblage due to exposure to stressors, for both benthic macroinvertebrates and fish. These metrics generally respond to various stressors in different manners. The sensitivity to climate change is known, in a general sense, for some of these attributes. Further study is needed to characterize signature responses to climate change for specific use in bioassessment programs around the country. The aggregation of a series of metrics into a biological index provides the primary measure of overall attainment of the desired biological condition. However, certain bioassessment programs (e.g., Maine DEP, Oregon DEQ) use discriminant or predictive models as primary bioindicators, which may provide a different dimension of climate-change sensitivity.

Table 2-1. Table of benthic macroinvertebrate metrics taken from the Rapid Bioassessment Protocols (U.S. EPA, 1999)

Category	Metric	Definition	Predicted response to increasing perturbation
Richness measures	Total number of taxa	Measures the overall variety of the macroinvertebrate assemblage	Decrease
	Number of EPT taxa	Number of taxa in the insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)	Decrease
	Number of Ephemeroptera taxa	Number of mayfly taxa (usually genus or species level)	Decrease
	Number of Plecoptera taxa	Number of stonefly taxa (usually genus of species level)	Decrease
	Number of Trichoptera taxa	Number of caddisfly taxa (usually genus or species level)	Decrease
Composition measures	% EPT	Percent of the composite of mayfly, stonefly, and caddisfly larvae	Decrease
	% Ephemeroptera	Percent of mayfly nymphs	Decrease
Tolerance/ intolerance measures	Number of intolerant taxa	Taxon richness of those organisms considered to be sensitive to perturbation	Decrease
	% Tolerant organisms	Percent of macrobenthos considered to be tolerant of various types of perturbation	Increase
	% Dominant taxon	Measures the dominance of the single most abundant taxon. Can be calculated as dominant 2, 3, 4, or 5 taxa.	Increase
Feeding measures	% Filterers	Percent of the macrobenthos that filter FPOM from either the water column or sediment	Variable
	% Grazers and scrapers	Percent of the macrobenthos that scrape or graze upon periphyton	Decrease
Habit measures	Number of clinger taxa	Number of taxa of insects	Decrease
	% Clingers	Percent of insects having fixed retreats or adaptations for attachment to surfaces in flowing water	Decrease

Table 2-2. Fish metrics used in various bioassessment programs.[a] Table adapted from the Rapid Bioassessment Protocols (U.S EPA, 1999)

Category	Metric	Definition	Predicted response to increasing perturbation
Richness measures	Total number of species	Measures the overall variety of the fish assemblage	Decrease
	Number of native fish species	Those species of fish that are indigenous	Decrease
	Number of salmonid age classes[b]	Measures the life stage representation of particular top predators in coldwater systems	Decrease
	Number of darter species	Diversity of darters, which are typically in fast flowing waters and cobble substrate	Decrease
	Number of sculpin species	Normally coldwater bottom feeders	Decrease
	Number of benthic insectivore species	Those species that depend on aquatic insects for primary food source	Decrease
	Number of darter and sculpin species	Combination of clean-water forms, mostly in coldwater systems	Decrease
	Number of darter, sculpin, and madtom species	Combination of key taxa that represent important structure of fish assemblage in certain systems	Decrease
	Number of salmonid juveniles (individuals)[b]	Density of juvenile salmon intended to evaluate nursery function	Decrease
	% round-bodied suckers	Warm-water species of suckers representative of good quality bottom feeders	Decrease
	Number of benthic species	Diversity of feeders of all benthic fauna, including insects and non-insects	Decrease
	Number of sunfish species	Warm-water pelagic species representative of good water quality and habitat	Decrease
	Number of cyprinid species	Diversity of minnows that include a range of tolerance	Decrease
	Number of water column species	Indicative of good quality pools and migration routes	Decrease
	Number of sunfish and trout species	Combination of species representing good water and habitat quality	Decrease
	Number of salmonid species	Diversity of salmon in coldwater systems able to accommodate a variety of top carnivores	Decrease
	Number of headwater species	Diversity in generally depauperate systems	Decrease

Table 2-2. continued.

Category	Metric	Definition	Predicted response to increasing perturbation
Richness measures (continued)	Number of sucker species	Diversity of all suckers—round-bodied and other	Decrease
	Number of sucker and catfish species	Combination of suckers and catfish in warm-water systems to be indicative of healthy systems	Decrease
Tolerance/ intolerance measures	Number of intolerant/sensitive species	Diversity of sensitive fish species; may be stressor dependent	Decrease
	Presence of brook trout	Indigenous to many areas of the Midwest and threatened by competition of other species	Decrease
	% stenothermal cool and cold water species	Narrow temperature tolerance of coldwater taxa	Decrease
	% of salmonid individuals as brook trout	Compositional dominance of brook trout to other salmonids	Decrease
	Number of green sunfish	Tolerant of warm-water sunfish that becomes dominant as other taxa decline	Increase
	% common carp	Tolerant bottom feeder	Increase
	% white sucker	Tolerant bottom feeder	Increase
	% tolerant species	Compositional dominance of all tolerant species	Increase
	% creek chub	Tolerant minnow species	Increase
	% dace species	Tolerant minnow species	Increase
	% eastern mudminnow	Tolerant minnow species	Increase
Trophic measures	% omnivores	No particular food preference	Increase
	% generalist feeders	Generalist feeders, able to deal with a variable diet	Increase
	% insectivorous cyprinids	Minnows that prefer aquatic insects as primary diet	Decrease
	% insectivores	All fish that prefer aquatic insects	Decrease
	% specialized insectivores	Highly specialized in food preference and easily affected by decrease in food availability	Decrease
	% juvenile trout	Indicative of food source able to support nursery function of juvenile trout	Decrease
	% insectivorous species	Composition of taxa with preference for aquatic insects	Decrease
	% top carnivores	Composition of taxa that prey on other fish and non-fish higher trophic levels	Decrease

Table 2-2. continued.

Category	Metric	Definition	Predicted response to increasing perturbation
Trophic measures (continued)	% pioneering species	Those species that occur early in succession of an ecosystem, and are usually very tolerant	Increase
Effort measures	Number of individuals (or catch per effort)	Relative measure of density of fish in ecosystem related to amount of effort to sample the fish assemblage	Decrease
	Density of individuals	Density regardless of effort	Variable
	% abundance of dominant species	Dominance versus evenness of taxa in fish assemblage	Increase
	Biomass (per m^2)	Relative measure of ability to sustain healthy fish assemblage through food availability and good habitat	Variable
Reproduction measures	% hybrids	Measures breakdown of distinct reproductive guilds usually due to habitat alteration	Increase
	% introduced species	Intentionally or non-intentionally taxa introduced into ecosystem and competitive or predatory upon native taxa	Increase
	% simple lithophills	Composition of individual fish that spawn in clean sand or gravel	Decrease
	% simple lithophills species	Composition of species as lithophills	Decrease
	% native wild individuals	Measure of relative reproductive success for native taxa	Decrease
	% silt-intolerant spawners	Need for clean substrate of larger particles than silt; affected by sedimentation processes	Decrease
Disease measures	% Diseased individuals (deformities, eroded fins, lesions, and tumors)	Chronic exposure to stressors resulting in some form of disease or deformation that may result in lethal conditions	Increase

[a] Data from Karr et al. (1986), Leonard and Orth (1986), Moyle et al. (1986), Fausch and Schrader (1987), Hughes and Gammon (1987), Ohio EPA (1987), Miller et al. (1988), Steedman (1988), Simon (1991), Lyons (1992), Barbour et al. (1995), Simon and Lyons (1995), Hall et al. (1996), Lyons et al. (1996), Roth et al. (1997).
[b] Metric suggested by Moyle et al. (1986) or Hughes and Gammon (1987) as a provisional replacement metric in small western salmonid streams.

Note: X = metrics used in region. Many of these variations are applicable elsewhere.

3. SENSITIVITY TO CLIMATE CHANGE OF BIOLOGICAL INDICATORS USED IN STATE BIOCRITERIA PROGRAMS

The sensitivity of common biological indicators to climate change is not very well known. The review of relevant literature presented in Section 1.2, as well as in this section, helps establish expectations for the significant modes of effect and probable categories of responses. However, important details regarding (1) the sensitivity or robustness of specific metrics or ecological attributes to changing climate parameters over time and among different regions; (2) the mechanisms by which specific responses will interact with other stressors and impact interpretation of effects and their causes; and (3) how such responses might combine to alter biological index responses are recommended as components of needed research (see Section 7).

To understand probable climate-change effects on stream/river biological indicators, the linkage between climate and stream/river ecology must be defined. Anthropogenic increases in greenhouse gases directly affect air temperature and precipitation (considered primary climate drivers). Climate-change projections for the year 2100 include global average air temperature increases of 1.1–2.9°C (for the lowest emissions scenario) to 2.4–6.4°C (for the highest emissions scenario) (IPCC, 2007). Increases in precipitation are predicted for many regions, with a higher percentage of total precipitation occurring in more frequent and intense storms. Other predictions include more precipitation in winter and less precipitation in summer; more winter precipitation as rain instead of snow; earlier snow-melt; earlier ice-off in rivers and lakes; and longer periods of low flow and more frequent droughts in summer (Hayhoe et al., 2007; IPCC, 2007; Barnett et al., 2005; Fisher et al., 1997).

Changes in these primary climate drivers will affect stream/river water and aquatic-life resources mainly through direct and indirect alterations in hydrologic and thermal regimes. Changes in hydrologic regime (including magnitude, timing, duration and frequency of runoff events) will vary regionally (NAST, 2001), but they are expected to include changes in the magnitude of flow ranging from increases of 10–40% in the northeastern U.S. to decreases in annual flow of 10–30% in the South, Midwest, and West (Hayhoe et al., 2007; Milly et al., 2005; Magnuson et al., 1997). Changes in patterns of flow will likely include increases in stream flow occurring mainly in the winter and spring, lower stream flow in the summer and fall, and greater variability and "flashiness" of stream flows (Hayhoe et al., 2007; Moore et al., 1997). These projected alterations in stream flow dynamics are critical in structuring aquatic ecosystems through influence on sediment supply and transport, habitat stability, channel formation and maintenance, and water volume, which, in part, controls habitat availability and water quality (Poff et al., 2002; Richter et al., 1996; Poff et al., 1996; Poff and Allan, 1995). Seasonal patterns of flow and other flow dynamics strongly influence the types of species that can inhabit an area, defining the composition, structure, and functioning of aquatic assemblages (Poff et al., 2002; Richter et al., 1996; Poff and Allan, 1995). As a result, climate-associated changes in stream-flow magnitude are expected to modify habitat, species composition and abundance, and ecological interactions over time.

Stream/river water temperature regimes will be altered by air temperature increases and modified by other influences including variations in flow volume and snow melt, groundwater influence, riparian shading, presence of deep pools, meteorology, river conditions, and geographic setting (Cassie et al., 2006; Mohseni et al., 2003; Daufresne et al., 2003; Hawkins et al., 1997). Thermal regime influences the distribution and abundance of aquatic species in relation to temperature tolerances and evolutionary adaptations combined with competitive interactions, effects on food supply, and other factors; it also drives timing of life cycle events (phenology), biological productivity, and species interactions (e.g., Matthews, 1998; Hawkins et al., 1997; Vannote and Sweeney, 1980; Sweeney and Vannote, 1978).

As discussed in Section 2.2, common metrics monitored as biological indicators in existing bioassessment programs are measures of change in features or attributes of the structure and/or function of the macroinvertebrate or fish assemblages; Tables 2-1 and 2-2 summarize many of the widely applied categories of biological indicators. Additional research will provide information on specific sensitivities of individual biological indicators to climate change. However, taken by category of metric, expectations for probable responses of various biological indicators to climate change can be summarized from literature information and projections of future climate changes (see Sections 1.2, 1.3, and this section). Table 3-1 summarizes expected climate change responses by category (Note: this summary of potential responses represents examples and is not considered comprehensive. Categorization as sensitive or tolerant refers generally to anticipated climate-change sensitivity, in particular to temperature and hydrologic changes).

It is clear that many of the types of responses that can be expected for common categories of biological indicators in response to climate change can be similar to changes caused by other conventional stressors. For biological indicators that are sensitive to both conventional stressors and climate change, the confounding interactions of climate change and other stressor effects will influence the process of attributing cause to particular stressors. This interaction will require the development of an approach to partition observed responses between climate change and other stressors, so that the ability to manage resources and regulate water quality through the process of monitoring and assessing biological indicator data remains viable.

Conceptually, this approach can include adaptations of monitoring methods in order to account for climate change. Preliminary aspects of this component are discussed in Section 4

Table 3-1. Summary of expectations for responses of common categories of stream and river biological indicators to climate change influences on water temperature and hydrologic regime

Category	Expected climate change effects/sensitivities	References
Macroinvertebrates		
Richness and abundance measures	Overall richness generally expected to decline due to temperature sensitivity and hydrologic stresses including increased flashiness, increased instances of summer low flows, drought, etc. However, replacements over time with tolerant forms may ameliorate this in some situations. Abundance or eurytolerant species may increase in some habitats.	Durance and Ormerod, 2007; Bradley and Ormerod, 2001
Community composition, persistence measures	Compositional changes resulting from reductions in temperature and/or flow sensitive taxa (examples potentially include *Chloroperla*, *Protoneumura*, *Neumura*, *Rhyacophila munda*, *Agabus* spp,, Hydrophilidae, and *Drusus annulatus*) and increases in less temperature and/or flow sensitive taxa (examples potentially include *Athricops*, *Potamopyrgus*, *Lepidostoma*, *Baetis niger*, *Tabanidae*, *Hydropsyche instabilis*, *Helodes marginata*, *Caenis* spp.), and/or from shifts in range ; patterns of persistence or community similarity that track climatic patterns; changes may also occur in functional roles of species.	Daufresne et al., 2003; Durance and Ormerod, 2007; Bradley and Ormerod, 2001; Burgmer et al., 2007; Golladay et al., 2004; Parmesan, 2006; Hawkins et al., 1997
Tolerance/ intolerance measures	Climate-change sensitivities related to temperature or flow regime may be documented as decreases (potentially resulting from local extinctions and/or range shifts) in richness (number of taxa) of temperature or flow-regime sensitive groups (see "Composition Measure" for examples). Dominance by tolerant taxa also may increase.	Daufresne et al., 2003; Durance and Ormerod, 2007; Burgmer et al., 2007; Golladay et al., 2004; Parmesan, 2006
Feeding measures	Variable responses expected, driven by interactions between temperature, which may increase phytoplankton and periphyton productivity and thus increase associated feeding type; hydrologic factors which may decrease periphyton if habitat stability is decreased or sedimentation is increased; CO_2 concentrations, which can directly affect leaf litter composition and decomposition; and changes in riparian vegetation.	Gafner and Robinson, 2007; Dodds and Welch, 2000; Tuchman et al., 2002
Habitat measures	Number and percent composition of clingers likely to decrease if hydrologic changes decrease habitat stability, increase embeddedness, or decrease riparian inputs of woody vegetation.	Johnson et al., 2003; Townsend et al., 1997

Table 3-1. continued.

Category	Expected climate change effects/sensitivities	References
	Fish	
Richness and abundance measures	May have initial increase in diversity as more warm-water assemblages replace cool- or cold-water species. Habitat availability is expected to be diminished by altered flow regimes with an associated loss of diversity. If barriers to dispersal limit community replacements, richness also may decline. May also, for example, lose spring spawners (e.g., some salmon species) due to changes in timing of spring flows. Abundance of warm-water species may increase, while coldwater species may decrease.	Xenopoulos and Lodge, 2006; Xenopoulos et al., 2005; Poff et al., 2002; Grimm et al., 1997; Hayhoe et al., 2007; Wehrly et al., 2003
Composition measures	Expect fish community compositional changes resulting from losses of cold- and/or cool-water fishes (e.g., brook trout, dace and bleak), and increases in warm-water fishes (e.g., chub and barbell).	Daufresne et al., 2003; Mohseni et al., 2003; Schindler, 2001; Covich et al., 1997; Moore et al., 1997; Rahel et al., 1996, Eaton and Scheller, 1996
Tolerance/ intolerance measures	Loss of temperature-sensitive cold- and cool-water species will decrease intolerant measures, increase tolerant measures.	Mohseni et al., 2003; Moore et al., 1997; Rahel et al., 1996; Eaton and Scheller, 1996
Feeding measures	Shift in food sources through attrition of lower trophic levels will affect higher trophic levels, including top carnivores.	Schindler et al., 2005; Melack et al., 1997
Habitat measures	Breakdown of habitat features and connectivity fosters hybridization and drift in species gene pool.	Matsubara et al., 2001; Heggenes and Roed, 2006

(below), to the extent that they were addressed in this preliminary case study. Another aspect of program adaptation is restructuring the analytical approach used to evaluate biological monitoring data, detect impairment, and assess cause (see preliminary case study results discussed in Section 5). These monitoring components—sampling strategy and analytical approach—are clearly inter-related, and implicitly include components such as tracking changes at reference locations through both altered sampling design and appropriate analyses.

Another component that is being considered for its potential contribution to tracking and differentiating climate-change effects from other stressors is the categorization of biological indicators based on differing sensitivities to these effects. These indicators include both community metrics and population measures of individual sensitive species (see Table 3-1). In concept, there would be analytical and interpretive advantages if at least some biological indicators could be identified that are especially sensitive to particular conventional stressors but insensitive to climate-change effects. Conversely, community metrics and individual taxa that are specifically sensitive to climate change would be valuable in identifying and defining trends at reference sites. These could be applied analytically to separate monitored biological responses into components related to long-term, climate-change effects and other stressors. Such separation is the major goal of efforts to adapt bioassessment programs to account for climate change.

In practice, evidence gathered from the literature and the professional opinions of many state/tribal bioassessment managers[1] suggests that few, if any, biological indicators currently used in bioassessment programs are likely to be insensitive to climate-change effects. This is largely because climate change affects aquatic communities through the critical ecological drivers of flow dynamics (hydrology) and water temperature. Thus, the modes of action of climate-change effects and effects of other stressors are similar in many cases, and taxa that are sensitive to conventional stressors are likely to be sensitive to climate change as well. Taxa identified in the Workshop as being "potentially insensitive to climate change" were mainly those species already characterized as being broadly tolerant, "weedy," and/or generalist species.

Beyond categorization of existing biological indicators as sensitive/insensitive to climate-change effects, there are biological metrics that could be considered for incorporation into bioassessment programs that are not currently measured on a routine basis in most existing programs. Such "novel" indicators are considered specifically because of their sensitivity to climate-change effects—most have been predicted or observed in the literature as biological responses to directional climate change, especially increases in water temperature. Table 3-2 summarizes examples of such "novel" biological indicators.

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[1] Hamilton, A; Barbour, M; Gerritsen, J; et al. (2007) Introductory workshop on climate change effects on biological indicators: workshop summary report.

Table 3-2. Novel indicators that may be sensitive to climate change

Category	Metric	Comments	References
Phenology	Early emergence of mayfly species (also stonefly and caddis species)	Indirect effects on timing of salmonid feeding regime	Harper and Peckarsky, 2006; Briers et al., 2004; Gregory et al., 2000; McKee and Atkinson, 2000
	Early trout spawning in warmer water		Cooney et al., 2005
	Accelerated development and earlier breeding of the amphipod *Hyallela azteca*		Hogg et al., 1995
Longer growing season	Increased algal productivity	In northern areas a response to decreased ice cover and increased light penetration	Flanagan et al., 2003
	Additional reproductive periods of amphipod species		Hogg et al., 1995
Life stage-specific	Altered sex ratios for certain insects (e.g., trichopteran *Lepidostoma*)		Hogg and Williams, 1996
	Smaller size at maturity and reduced fecundity of plecopteran *Nenoura trispinosa* and amphipod *Hyallela azteca*	From increased temperature	Turner and Williams, 2005; Hogg et al., 1995
	Decreased salmon egg to fry survival	Increased turbidity from eroded sediment due to increased precipitation	Melack et al., 1997
Temperature sensitivity	Reduced size of sockeye salmon	Reduced growth and increased mortality in higher temperatures as well as to lower plankton productivity	Melack et al., 1997
	Increased growth rate of juvenile salmon in Alaska		Schindler et al., 2005
	Decreased growth rate of trout		Jensen et al., 2000

Table 3-2. continued.

Category	Metric	Comments	References
Hydrologic sensitivity	Decreased survival of eggs of autumn-spawning salmon (e.g., dolly varden, brook trout, coho salmon)	Results in decreased abundance of autumn-spawning species, and/or change in relative composition between spring and autumn spawners	Gibson et al., 2005
	Decreased fry survival of pink and chum salmon due to earlier (late winter to early spring) peak flows	Earlier emergence and migration of pink and chum salmon fry to estuaries at a time when their food sources have not developed adequately	Melack et al., 1997
	Differential mortality of drought-intolerant mussel species (e.g., *Lampsilis straminea claibornensis*, *Villosa villosa*, *Lampsilis subangulata)*	Results in changes in relative abundance, extirpation of vulnerable species	Golladay et al., 2004

One consideration that must be taken into account in the ongoing evaluation of potential novel indicators and their role in adaptation of bioassessment programs is that many of these metrics are more difficult or time- and resource-consuming to measure, especially on a routine basis. Some of them also require sampling techniques and timing or frequency of sampling that are quite different from the commonly applied bioassessment approaches. For example, the process of measuring sizes of all individuals of one or more species of mayflies, stoneflies, or caddisflies (representative EPT taxa) to establish size-class composition and evaluate reduction in size of the last instars (i.e., the last nymphal stage just before emergence) and how this changes over time to define climate-change effects; or similarly, the sampling of emerging adult insects that would be needed to evaluate earlier emergence, are not commonly done. Another consideration for future evaluation of novel indicators is their potential sensitivity to other (conventional) stressors, in addition to their responsiveness to climate change. This will affect how they might be incorporated into a monitoring design and analysis approach.

Having given a summary of climate-change effects, an overview of state bioassessment and biocriteria programs, and a framework for considering the sensitivities of established and novel biological indicators to climate change, preliminary consideration can be given to aspects of possible vulnerabilities of biological assessment programs to climate change. This report first addresses the sampling power needed to detect climate-change effects using current indicators. Secondly, the report describes how climate change may affect reference and non-reference sites differently. The report includes case studies using data from one program to illustrate these preliminary results obtained from implementation of the proposed framework.

4. CASE STUDY 1—ASSESSING TRENDS: THE POWER OF BIOLOGICAL ASSESSMENTS TO DETECT CLIMATE CHANGE

The ability to account for climate change requires an understanding of how vulnerable monitoring data are to climate change effects and how effectively differences that are a result of climate change can be detected within existing monitoring programs. This case study describes (1) how much sampling would be needed to distinguish expected levels of climate change effects, and (2) how long it would take to detect climate change effects with a specified probability of detection, given a particular monitoring framework. The information summarized in this section highlights the approach, the key results, and the main conclusions of Case Study 1.

4.1. OBJECTIVES

The main objective of this case study is to evaluate one aspect of the vulnerability of biological monitoring and biocriteria programs to climate change with respect to the effects on ecological communities. This case study focuses on the ability of a typical bioassessment program to detect expected climate change effects on one selected community component, taxon richness. The focus is on two questions:

- How long must monitoring be conducted to have a fixed probability of detecting a change in the mean native taxon richness of the reference site population?
- How long must monitoring be conducted to have a fixed probability of detecting a change in mean native taxon richness for a particular site?

The first question is important because most states use reference populations as the basis for constructing indices and deriving biocriteria. The second is important because many individual sites are tracked for specific regulatory reasons (permitting, restoration, etc.).

4.2. ANALYSIS APPROACH

The questions in this study are approached by evaluating the ability, or power, of a typical biological monitoring program to detect expected levels of change in a particular biological attribute—in this case species richness. Statistical power, the ability to detect a real effect, is a critical issue in designing monitoring programs and is expressed as a probability. The more power a test has, the more likely one is to correctly infer that a real change has actually occurred. In this case study, the power analysis approach is used to evaluate how much of a change in taxon richness (the effect size or minimal detectable difference) can be detected in a typical biomonitoring program. This detectable difference can then be compared to expected taxa loss from climate change to see how long (in years) the program must monitor to ensure detection (at a set probability level) of a taxa loss signal resulting *only* from climate change.

4.2.1. Ability to Detect Change–Power Analysis

Power is defined as the probability of rejecting a false null hypothesis and is related to type II error (β), which is the probability of accepting a null hypothesis (no change) when it is false (there is change). Thus, power = 1-β. Power analysis requires several critical components:

- sample size (N)
- variability in an observed factor (taxon richness in this case) (σ)
- effect size (how much of a change one wants to detect) (δ)
- significance level (type I error, α)

For this application of power analysis, the desired level of power (1 - β) must be fixed as well. This case study demonstrates how changing some of these components can increase or decrease the ability to detect a climate change effect.

The equations for calculating effect size for comparing two paired population means can be found in many statistical textbooks. The basic formula, assuming normally distributed populations, is:

$$\delta = \frac{(Z_{1-\alpha/2} + Z_{1-\beta}) \times \sigma}{\sqrt{n}} \qquad \text{(Eq. 1)}$$

where Z_α and Z_β are the z-scores (probability levels) for the desired type I (α) and type II (β) error rates.[2]

For this case study, variance (σ^2) is estimated using existing monitoring data for sites sampled repeatedly over several years during an index period. Such data give an estimate of the natural variability in biological condition through time, assuming minimal external changes. Knowledge about what taxa changes (effect size) might be expected in response to climate change is also needed. This value was derived from existing literature on taxa loss in relation to temperature changes (see Section 4.2.3).

For each of the two assessment questions (see Section 4.1), different confidence levels were investigated (i.e., α and β were varied). The effects of sample size were also investigated

[2] Source: Snedecor and Cochran (1980).

for both questions. To address the first question of detecting change in a reference population, the number of sampling sites, N, was either fixed per year (at N = 40 for the reference population) or cumulative (N increasing by 40 each year for the reference population) and assumed a comparison of year one versus a cumulative pooled sample size through time. For the second question (detecting a change at a particular site), N was either fixed at one of three different levels (N = 5, 10, and 20) or N was cumulative, increasing by 5, 10, or 20 for the respective analysis runs.

Variance estimates were derived from one particular state bioassessment data set (see Section 4.2.2 below) and were then used for each region evaluated. Z scores were varied by changing the type I and type II error rates to illustrate the effects of these choices on the ability to detect a climate change effect. The outputs from these analyses are time series of taxa loss rates predicted from climate change effects. These outputs are compared with minimal detectable effect sizes to illustrate the length of time required to detect a climate change effect on taxon richness under various conditions (taxa loss rates, temperature scenarios, and error rates).

4.2.2. The Maryland Biological Stream Survey Data Set

For this case study, variance (reflecting natural variability in biological condition over time) is estimated using existing monitoring data from the Maryland Biological Stream Survey (MBSS) (Boward et al., 1999; URL: http://www.dnr.state.md.us/streams/mbss/). The MBSS program includes statewide monitoring of all watersheds using a multi-stage probability based design with a 5-year rotating basin sampling approach. It also includes repeated annual sampling at a series of 28 fixed reference (“sentinel”) sites using the same sampling methods. It is this repeat sampling of benthic macroinvertabrates and fish at sentinel sites that was used to estimate population variance (σ^2) in taxon richness. While this case study compares expected effects across various regions of the U.S., the MBSS derived variance was used for all regions as the estimate of variance associated with biomonitoring.

Relative variability of taxon richness is assumed to be constant over time. However, this is likely not true as both the mean and variation in biological condition of sites may change with warming water temperatures. For simplicity, it is assumed that the variance in taxon richness associated with the MBSS program can be extrapolated in time and across different regions.

MBSS data (Boward et al., 1999; URL: http://www.dnr.state.md.us/streams/mbss/) are available with uniform collection methods for the period 1994–2004, with sentinel site data available from 1998–2004. The 6-year period for which sentinel site data are available fortuitously includes both a dry and a wet climate cycle. Sampling for the MBSS is conducted during index periods—the spring reproduction/recruitment period for benthic macroinvertebrates (March–early May), and the summer-fall low-flow period for fish (June–September). A wide range of physical, chemical, and habitat variables are measured and/or calculated in association with the biological collections. These include water chemistry variables (e.g., temperature, pH, DO concentration, various nutrient concentrations, conductivity); numerous physical habitat variables (e.g., the Maryland physical habitat index (PHI), instream habitat condition, epifaunal substrate, water velocity, water depth, embeddedness, shading, distance to road, riffle quality, etc.); and land use characteristics (17 detailed categories aggregated to the larger categories of agriculture, urban, water, wetland, barren, and forest).

Benthic macroinvertebrates are collected using D-nets, employing a multi-habitat approach over a 75-m reach. A 100-organism subsample is processed for each sample. Identifications are made to the genus level. Fish are collected using quantitative, double-pass electrofishing in 75-meter stream segments, with a blocking net at the end of the segment. Fish are identified to the species level (Boward et al., 1999).

4.2.3. Information on Taxa Loss Rates

This study focuses on climate change effects associated with temperature because (1) the goal is to demonstrate a process for calculating the capacity of a program to detect change and not to predict all the effects of climate change and (2) while few data exist on climate change effects on aquatic assemblages in general, there are more data on temperature effects than hydrologic effects.

Predicted macroinvertebrate taxa loss rates due to temperature increases were derived from the literature based on observed changes in taxon richness associated with temperature increases. For this study, native or expected taxon richness is considered rather than total richness; species replacement is not considered. Total richness may not change if species replacement rates are high. For example, it is possible that stenothermal species (those with a narrow range of temperature tolerance) that are lost will be replaced over time by eurythermal taxa (those with wide temperature tolerances). However, native taxa are expected to be lost from many streams (e.g., Parmesan, 2006; Xenopolous et al., 2005; Moore et al., 1997), and native taxon richness based on current climate will decrease.

Other ecological responses are expected but are not considered in this study: density changes, range shifts, timing changes in important life history stages and phenology, morphological changes, physiological changes, and behavioral changes, and gene frequencies changes (Parmesan, 2006; Root et al., 2003; Walther et al., 2002; Hogg et al., 1998; Schindler, 1997). Taxon richness, a very common component metric evaluated in bioassessment programs and incorporated in multimetric indices, is evaluated for signs of bioassessment program vulnerability.

Macroinvertebrate and fish taxa (i.e., genus or species, reflecting practical taxonomic limitations) loss rates were obtained from literature reporting on climate-change associated temperature effects on taxon richness (Daufresne et al., 2003), and on thermal discharge effects (Gammon, 1973; Lehigh University, 1960) using linear projections (estimating average species loss per unit of temperature change). Given the complexity of biotic interactions as well as temperature effects, this is probably not an accurate representation of reality. In reality, taxa loss rates are likely to occur episodically over time, for instance when particular species thresholds are reached. For example, loss of keystone taxa may precipitate abrupt and dramatic changes on stream communities as well as on stream processes (Flecker, 1996; Pringle et al., 1993; Power, 1990; Power et al., 1985). In addition, the amount of decrease and range of variation around expected decreases may also be affected by differences in the size of the species pool, e.g., species-rich sites could lose more taxa, and/or show greater variation in loss rates. On the other hand, small losses from locations with a naturally poor fauna may be ecologically more significant. However, the simplifying assumption allows us to model changes into the future. Based on the limited information currently available in the literature, the high and low native taxa loss rates used are simply to bracket a range to estimate detection ability. The ecological implications and relative importance of being able to detect changes in taxa will still vary between sites.

Daufresne et al. (2003) observed a loss of 7 macroinvertebrate taxa in streams associated with a 1.5ºC increase over the period 1980–1999. This equals a loss rate of roughly 4.6 taxa per °C and is being considered the high taxa loss rate. A second estimate was derived from the literature associated with thermal discharge studies associated with the CWA 316 program. Most of the 316(a) studies focused on fish effects, and, out of those studies, many were physiological. One study (Lehigh University, 1960) included macroinvertebrate effects and this study found a loss rate of approximately 1 taxon per °C over the range 22–28°C; this is the low taxa loss rate. This represents a fairly high thermal range, but these studies were designed to investigate effects of thermal effluent, not effects of climate change. Nevertheless, the results are considered applicable. There are likely more 316(a) studies with invertebrate data, but these individual studies, for the most part, are not published in standard scientific citation databases and can be hard to locate (but see the Energy Citations Database [http://www.osti.gov/energycitations/index.jsp]).

Predicted fish taxa loss rates were also considered from the thermal discharge literature. A study of thermal effluent on the Wabash River found a loss rate of 3.6 fish taxa per °C increase in temperature (Gammon, 1973). This may be on the high side for loss rates, but it was one of the few data-based values found within a temperature range comparable to climate change projections.

4.2.4. Prediction of Expected Taxa Losses with Projected Temperature Increases

Estimates of taxa loss rates were coupled with projected temperature increases to model the expected rate of taxa loss per year due to climate change. Projected temperature increases due to climate change for each region of the U.S. were taken from the National Assessment Synthesis Team (NAST) summary report (2001). NAST (2001) relied mainly on results from two coupled atmosphere/ocean general circulation models (AO-GCMs) that were used to estimate projected temperature increases for various regions of the U.S. (Table 4-1). Predicted temperature increases by the year 2100 ranged between 2.3°C and 6.5°C for the Hadley and Canadian models, respectively. Although the biological data are from the Mid-Atlantic region, we also investigated how projected climate changes in other regions affected taxa loss rates.

Table 4-1. Average annual temperature increases expected by region of the U.S. (NAST, 2001)

Region	Average Annual Temperature (ºC) Increases by 2100	
	Min	**Max**
Northeast/Mid-Atlantic	2.6 (Hadley)	5 (Canadian)
Southeast	2.3 (Hadley)	5.5 (Canadian)
Midwest	3 (Hadley)	6 (Canadian)
Great Plains	3 (Hadley)	6.5 (Canadian)
West	4 (Hadley)	5.5 (Canadian)
Pacific Northwest	2.7 (by 2050) (Hadley)	3.2 (by 2050) (Canadian)

Reported rates of temperature increase were linked with estimated rates of taxa losses to model taxa losses per year due to climate change, incorporating both the low and high estimates of each.

Linear projections of climate change effects are used in this case study as a basis for estimating ability to detect climate-induced changes after various monitoring periods. It is likely that climate will change in a non-linear fashion with periods of fast change followed by periods of slower changes (IPCC, 2007). There is little way to predict this course, however, so the linear assumption is the more conservative approach and is a common assumption used in the literature (e.g., Najjar et al., 2000).

4.3. KEY FINDINGS

4.3.1. How Long Must Monitoring be Conducted to have a Fixed Probability of Detecting a Change in the Mean Native Taxon Richness of the Reference Site Population?

If a population of reference sites (N = 40) is sampled each year, an average macroinvertebrate taxon richness in reference streams can be calculated. For comparing any two samples of N = 40 sites, there is a fixed difference in mean taxon richness (effect size) at which significance can be detected with a specified power. For $\alpha = \beta = 0.05$ (95% confidence, 95% power), and the Maryland data, the effect size is 4.5 taxa. Thus, to have a 95% probability of detecting a significant ($p < 0.05$) taxa loss between 2 samples of 40 sites, requires a mean difference of 4.5 taxa. At high taxa loss rates and under the higher estimate for warming in the Northeast/Mid-Atlantic region, it will take 15 years to achieve a mean loss of 4.5 taxa (Figure 4-1), assuming that (1) the same 40 sites are sampled each year; (2) samples from a site are not treated as cumulative through time; and (3) the analysis uses type I and type II error rates of 0.05. This value is derived by identifying the point where the effect size line (hatched) crosses the taxa loss rate line (solid) (Figure 4-1).

Figure 4-1 illustrates a variety of scenarios representing different confidence levels and either fixed or cumulative sample sizes. For example, relaxing the confidence level decreases the time to detect a change. Increasing α and β from 0.05 to 0.20 (reducing both confidence and power), reduces the time to achieve 80% probability of detecting a significant climate change effect ($p < 0.2$) to approximately 8 years. If a 1 in 5 (rather than a 1 in 20) chance that statistically significant results are due to random chance alone is acceptable, a taxa change attributable to climate change could be detected in half the time. This is the type of trade-off that is important for programs to consider.

Similarly, if samples taken across the reference population are treated as cumulative estimates of the population condition (replicates of the reference condition), then the projected climate change effect can be detected very quickly under the conditions of a high taxa loss rate and high temperature increase. If this replication is temporal, i.e., if samples from consecutive years are grouped, this also would increase the ability to detect the climate change effect. However, there is an associated assumption that interannual variation is constant (i.e., that successive years are comparable and can be grouped for analysis). This may be true on short time scales, but it might be faulty over longer periods given climate change, which is progressive. Combining samples into decadal (or shorter) groups (N = 400) may be more defensible and would also result in detecting the climate change effect more quickly.

Using the same assumptions but altering the taxa loss rate to the lower taxa loss rate (1 taxon per °C), the time to detect a climate change effect increases dramatically (Figure 4-2). This is reasonable, given that subtle effects will be much harder to detect than strong effects.

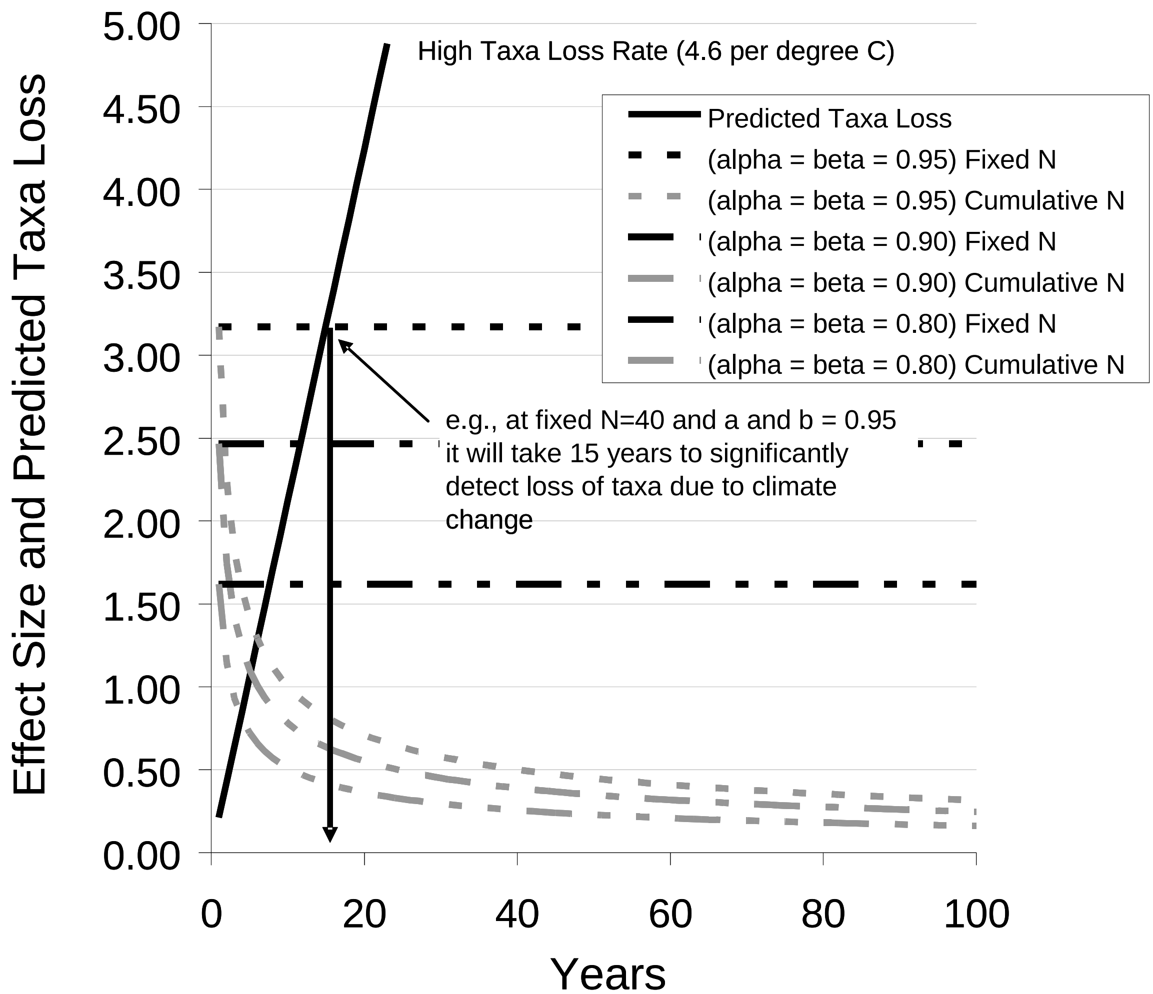

Figure 4-1. Effects of confidence level (α [a] and β [b]) on time to detect a climate effect on macroinvertebrate taxa loss due to climatic warming at high taxa loss rates in the Northeast/Mid-Atlantic U.S. Sample size (N) is either fixed at 40 per year or is cumulative. This analysis was based on a high estimate of global warming (5ºC by 2100).

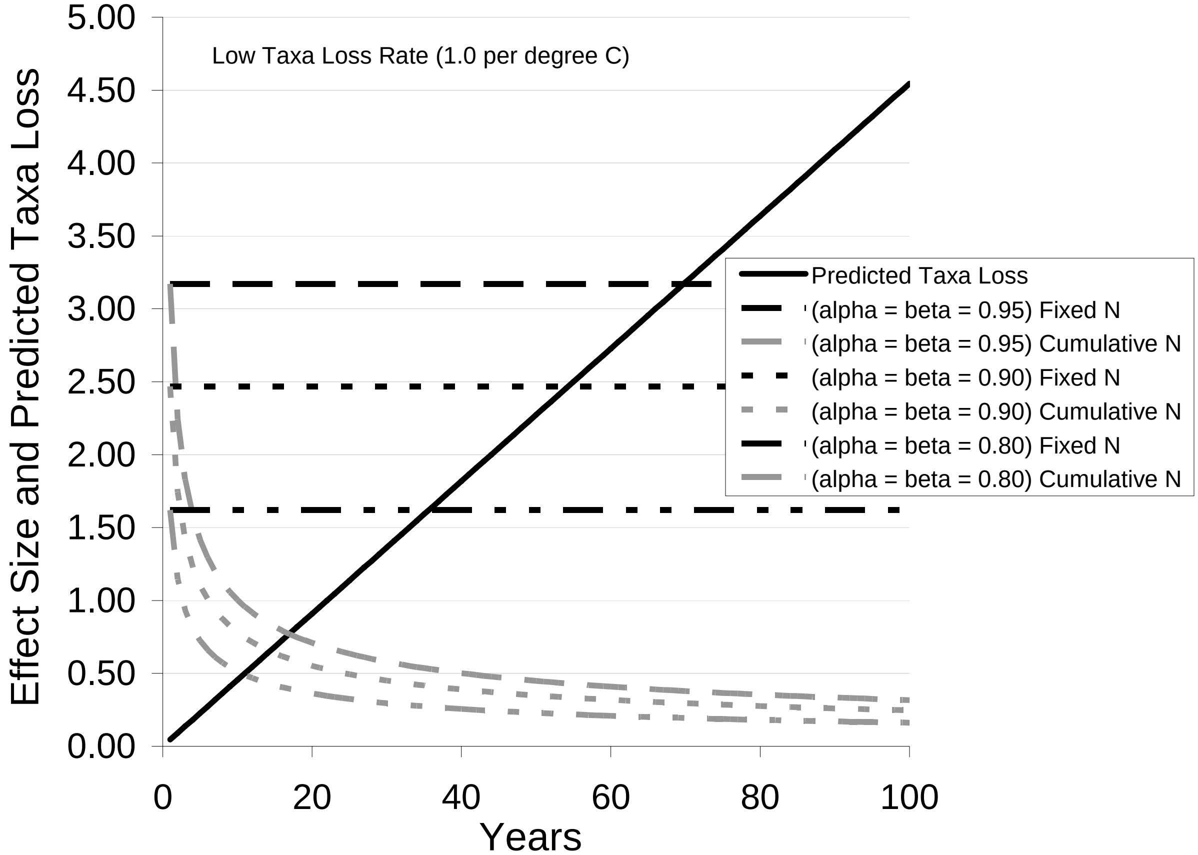

Figure 4-2. Effects of confidence level (α [a] and β [b]) on time to detect a climate effect on macroinvertebrate taxa loss due to climatic warming at low taxa loss rates in the Northeast/Mid-Atlantic U.S. Sample size (N) is either fixed at 40 per year or is cumulative. This analysis is based on a high estimate of global warming (5ºC by 2100).

Uncertainty about the effects of warming water temperatures on taxa loss also influences the ability to detect a climate change signal. The effects of confidence level and sample size are the same under the lower taxa loss rates as under the high taxa loss rates. For example, under the same fixed sample size (N = 40) and confidence level (95%), it would take approximately 70 years to detect a climate effect under the low taxa loss rate as opposed to 15 years under the high taxa loss rate.

Lastly, there was only one defensible rate for fish taxa loss (3.6 per °C). This fish taxa loss rate was used in the same models as those for the macroinvertebrate loss rate and indicates similar effects of sample size and confidence level on time to detect a change. Under these assumptions, it would take 10 to 20 years to achieve a fixed probability of detecting the loss of fish taxa due to climate change in the reference site population using confidence levels of 0.80 and 0.95, respectively (see Figure 4-3).

Similar analyses were run for the lower temperature increase scenario for this region (Table 4-1). Not surprisingly, if temperatures warm more slowly, they will have less of an effect on taxa loss and it will take comparatively longer to detect a loss in average taxon richness in the reference population (Table 4-2).

4.3.2. How Long Must Monitoring be Conducted to have a Fixed Probability of Detecting a Change in the Mean Native Taxon Richness for a Particular Site?

This second question focuses on the ability to detect these same effects at a single site, which could be a reach of stream or a watershed. In either case, the assumption is that replicate samples are apportioned probabilistically across the site. The analysis specifically defines the effect of increasing sample size.

For this question, three different sample sizes were investigated (N = 5, 10, or 20) and were treated as either fixed (non-additive over time) or cumulative. Only results for the Northeast/Mid-Atlantic region under the maximum predicted temperature increase are shown, although results are similar across regions, shifting only due to differences in the projected temperature increases.

Whether for a watershed or a specific reach, increasing the sample size will shorten the time required to detect an effect of climate change on taxon richness (see Figure 4-4). Many biomonitoring programs may collect only one sample at a site per year; a means comparison could be applied in this framework (use N = 1 in equation), but the differences would have to be quite large to be significant, and this is not likely over the short term (e.g., between consecutive years). Samples could be combined cumulatively over consecutive years to support testing, but the same problem exists in combining consecutive years over a long time period for analysis: the communities being sampled are probably changing over time due to climate change.

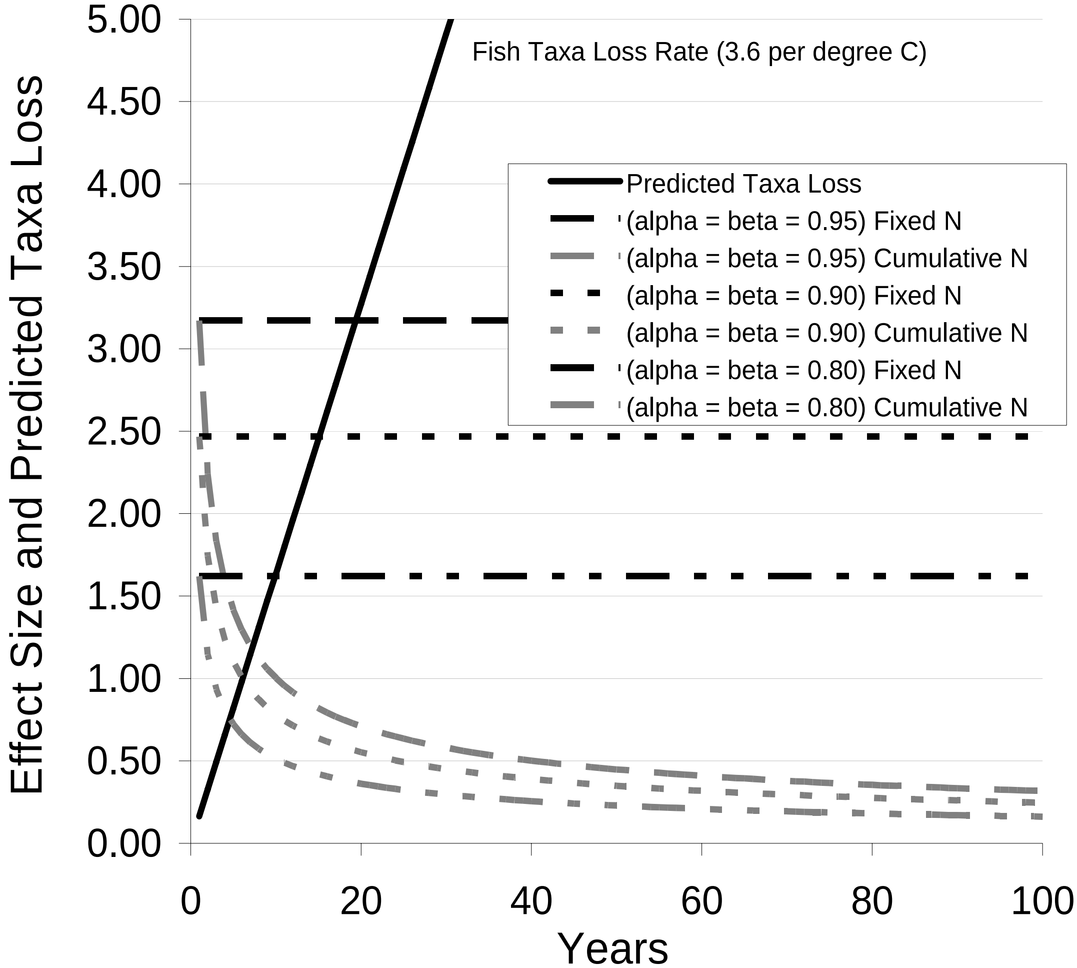

Figure 4-3. Effects of confidence level (α [a] and β [b]) on time to detect a climate effect on fish taxa loss due to climatic warming in the Northeast/Mid-Atlantic U.S. Sample size (N) is either fixed at 40 per year or is cumulative. This analysis is based on a high estimate of global warming (5ºC by 2100).

Table 4-2. The time (years) to achieve a fixed probability of detecting a statistically significant effect of temperature increases on macroinvertebrate and fish taxa loss across different regions under maximum and minimum temperature projections. These data are for question 1 and assume a fixed sample size of N = 40 reference sites sampled each year. Data are shown for different taxa loss rates and for different confidence levels.

		Regions					
		Northeast/ Mid-Atlantic	**Southeast**	**Midwest**	**Great Plains**	**West**	**Pacific Northwest**
Maximum predicted temperature increase by 2100							
Macroinvertebrates—high taxa loss rate (4.6 per °C)							
	$\alpha = \beta = 0.95$	15	14	13	---	13	---
	$\alpha = \beta = 0.8$	8	7	7	---	7	---
Macroinvertebrates—low taxa loss rate (1 per °C)							
	$\alpha = \beta = 0.95$	70	64	58	---	57	---
	$\alpha = \beta = 0.8$	36	33	30	---	29	---
Fish taxa loss rate (3.6 per °C)							
	$\alpha = \beta = 0.95$	20	18	17	---	16	---
	$\alpha = \beta = 0.8$	10	9	9	---	9	---
Minimum predicted temperature increase by 2100							
Macroinvertebrates—high taxa loss rate (4.6 per °C)							
	$\alpha = \beta = 0.95$	29	33	38	19	17	14
	$\alpha = \beta = 0.8$	15	17	19	10	9	7
Macroinvertebrates—low taxa loss rate (1 per °C)							
	$\alpha = \beta = 0.95$	>100	>100	>100	88	79	64
	$\alpha = \beta = 0.8$	69	78	89	45	41	33
Fish taxa loss rate (3.6 per °C)							
	$\alpha = \beta = 0.95$	38	42	49	25	22	18
	$\alpha = \beta = 0.8$	19	22	25	13	12	9

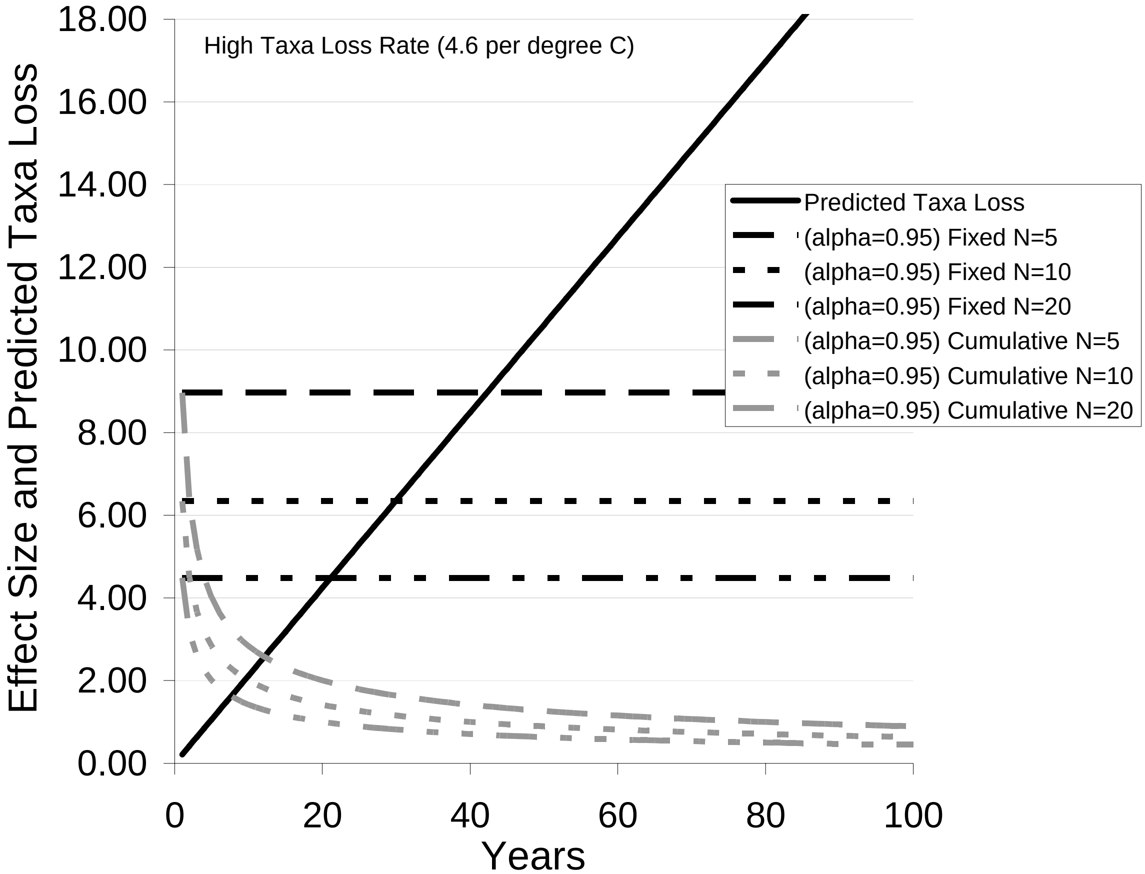

Figure 4-4. Effects of sample size on time to detect a climate effect on macroinvertebrate taxa loss due to climatic warming at high taxa loss rates in the Northeast/Mid-Atlantic U.S. The confidence level is fixed at 0.95. This analysis is based on a high estimate of global warming (5ºC by 2100) and the highest macroinvertebrate taxa loss rate.

As before, relaxing assumptions about required power and confidence levels (e.g., decreasing α and β to 0.8) will decrease the duration of monitoring needed to be able to detect a climate change effect of taxa loss at a particular site. Similarly, if macroinvertebrate taxa loss rates are slower than the high taxa loss rate used for Figure 4-4, it would take much longer to detect an effect (see Figure 4-5), all else being equal. Effects of sample size on fish taxa loss rates are similar to those for macroinvertebrates (see Figure 4-6): Locations with higher rates of climate change-associated temperature increases and/or higher rates of taxa loss responses would require less monitoring time to detect (i.e., statistically demonstrate) an effect; the converse (lower ranges of temperature increase and/or taxa loss) would increase the required monitoring time.

4.4. KEY CONCLUSIONS

Results of this case study highlight considerations for monitoring programs in light of the need to account for climate change. Increasing sample size, either by increasing the number of reference sites sampled each year or increasing the number of samples taken per watershed, or per reach for targeted studies, will increase the ability to discern a climate change effect using biomonitoring data. Regions with lower rates of climate change and/or taxa loss rates will require either a longer monitoring period or a larger sampling effort to detect climate change taxa losses effectively. On the other hand, with lower rates of climate change, effects from other regulated sources of perturbation may be reliably detectable for longer, although increases in variability and degradation of signal-to-noise ratio will degrade ability to detect impairment to some extent (see Section 5). Since greater variability in the data decreases ability to detect differences in taxon richness due to climate change, region-specific estimates of data variance are important for an evaluation of a particular monitoring program. In addition, factors within a monitoring design that can control for predictable sources of variation, such as partitioning by watershed or ecoregion, become important, as they would reduce (account for) natural sources of variation and increase ability to reliably recognize climate change effects.

The choice of a probabilistic or targeted sampling protocol is an important monitoring design issue, and will depend on the questions being asked. It also bears on the ability to detect climate change effects. Probabilistic designs are good for asking questions about, for instance, the average condition of streams or watersheds, including taxon richness, within a region. With regard to climate change effects, probabilistic sampling across reference sites would be ideal for defining condition but would require relatively large sample sizes to detect differences in biological attributes such as taxon richness because of the greater variation in the data. In the context of this case study, sample size and power are based on paired tests, which are much more

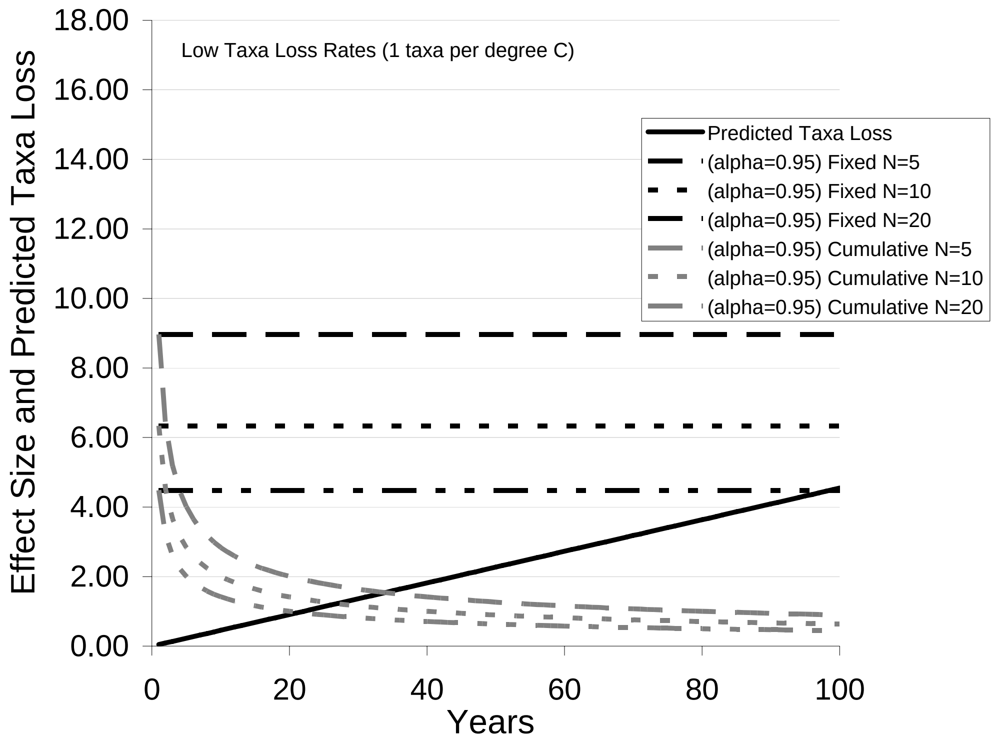

Figure 4-5. Effects of sample size on time to detect a climate effect on macroinvertebrate taxa loss due to climatic warming at low macroinvertebrate taxa loss rates in the Northeast/Mid-Atlantic U.S. The confidence level is fixed at 0.95. This analysis is based on a high estimate of global warming (5°C by 2100) and the highest macroinvertebrate taxa loss rate.

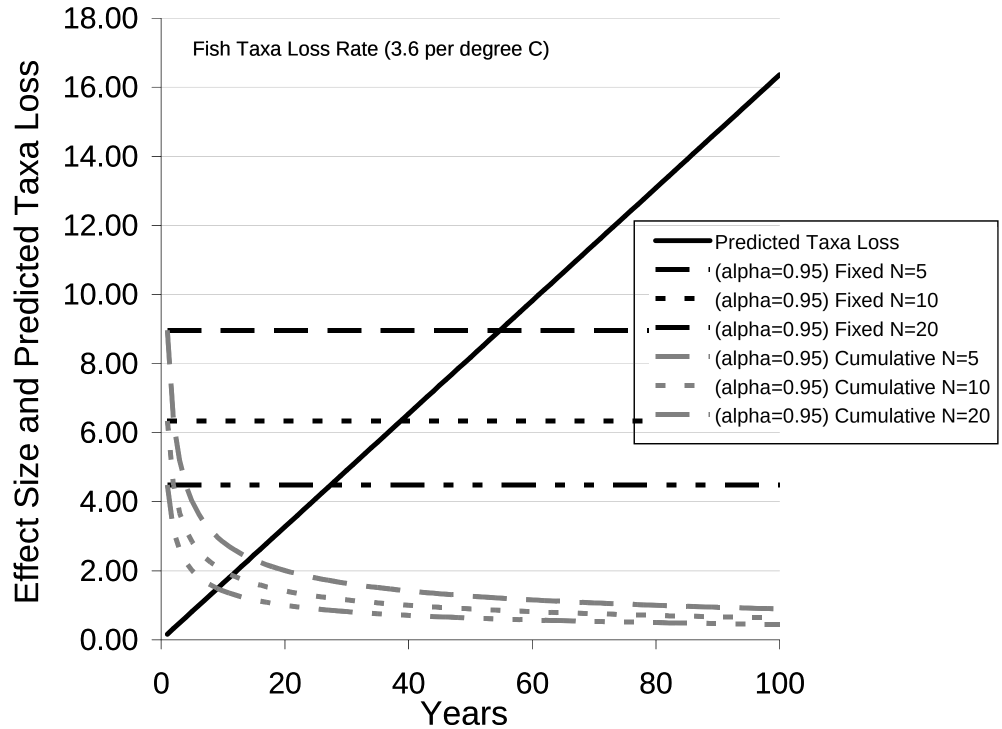

Figure 4-6. Effects of sample size on time to detect a climate effect on fish taxa loss due to climatic warming in the Northeast/Mid-Atlantic U.S. The confidence level is fixed at 0.95. This analysis is based on a high estimate of global warming (5°C by 2100) and the highest macroinvertebrate taxa loss rate.

powerful than drawing new independent samples every year because site-to-site differences are removed from the variance term, leaving only differences over time between sites.

Targeted site selection, however, is often needed to answer specific questions, including site-specific questions such as whether a site is meeting its designated use or permit requirements. Another question that benefits from targeted site selection is what the effect of a specific land use is on stream condition, because of the benefits of targeting sampling locations along a gradient of effects. This may be important for studying how land use will interact with climate change to affect stream condition.

If a bioassessment program is going to encompass climate change monitoring, there are several points to consider:

1. Protection of reference streams emerges as an important concept, especially considering that reference sites will be used to gauge climate change effects as well as the relative effects of climate change on other stressors.
2. Ongoing monitoring of reference sites becomes an even more important aspect of program design, with more sampling sites in reference locations and/or greater frequency of sampling increasing the ability to detect change.
3. The use of rotating designs (rotating sampling among basins over years so that a complete cycle of sampling may take 5 or more years) is often employed by state biomonitoring programs to optimize resources, because crews can stay within defined areas, travel can be limited, and total numbers of samples collected and processed each year is reduced by focusing on a subset of basins. This approach also means that reference sites within any one basin will only be sampled once every several years, increasing the time it will take to obtain replicate samples needed to define climate change-associated trends.

5. CASE STUDY 2—ACCOUNTING FOR TRENDS: BIOLOGICAL ASSESSMENT IN THE PRESENCE OF CLIMATE CHANGE

Detection of biological impairment and identification of its causes are two principal objectives of bioassessment. Climate change will affect these central objectives, especially the ability to discern impairment by comparison to reference locations. The second case study examines how to detect impairment under climate change, particularly the ability to differentiate between reference conditions and locations of reduced biological condition and the ability to assign cause to impaired conditions. This approach is a foundation for defining how monitoring may have to be modified to incorporate climate change and how data can be analyzed to account for climate change and remain viable. Appendix B provides some additional details of this case study, in particular descriptive and supplemental results of correlations examining potential relationships between environmental variables and stressors that might be used to reflect direct climate change effects.

5.1. OBJECTIVES

The case study examines the potential vulnerability of biomonitoring programs and assessment methods to biological changes that result from climate change. This case study addresses the following questions:

- How do we detect impairment under climate change?
- How does climate change affect our ability to identify causes of biological impairment?
- Are there analytical or monitoring design approaches that will allow managers to effectively identify and manage stressors independently of climate change?

Climate change effects will likely drive the attributes of reference sites toward greater similarity with impaired sites (i.e., decreased distance between the condition state of reference and impaired). This decrease in effect (signal) may also be accompanied by increases in variation (noise). A decrease in the signal-to-noise ratio would decrease the ability to detect impairment. In addition to direct effects on site assessment, climate change effects may interact with conventional stressors, further confounding the ability to discriminate stressor effects based on reference/impaired site comparisons.

5.2. ANALYSIS APPROACH

The case study uses existing data, and by examining the associations of biological attributes with proxy attributes of climate change, evaluates the potential effects and vulnerabilities of aquatic biomonitoring programs to climate change. Biological responses of streams to various stressors were examined—with particular emphasis on hydrologic parameters that may be influenced by climate change.

Detectable biological responses to climate change effects in streams that are important in a bioassessment framework include the following:

- Southern taxa expanding their range northward
- Habitat change from increased winter/spring scour
- Loss of taxa sensitive to summer drought periods or higher temperatures (including higher water temperature associated with drought)
- Improved conditions for invasive species, including disturbance regimes favoring invasive species and warmer water temperatures allowing overwintering
- Change in number of reproductive periods leading to changes in timing of peak abundance (possibly also tied to changes in phenology)
- In addition to the direct effects of temperature change, streams are also subject to hydrologic changes from changed precipitation patterns and increased evapotranspiration (e.g., Moore et al., 1997). Extreme stream flows reshape the stream habitat, and summer low-flow events represent bottlenecks of both warm temperature and reduced habitat (Moore et al., 1997; Poff and Ward, 1989). This analysis is focused on changes that might occur in the Mid-Atlantic region, but the results may be generalized to similar changes occurring in other regions

Data were partitioned into subsets defined by wet, normal, and dry periods, and biological indicators of reference and impaired sites were examined. Several stressor-response relationships were evaluated under the different climatic conditions. The intent was to estimate probable minimum and maximum changes.

5.2.1. Datasets Evaluated

The MBSS data set was used to evaluate biologic responses to stressors under different conditions. Section 4.2.2 describes attributes of this data set.

5.2.2. Metrics

Several invertebrate metrics were calculated in the Ecological Data Application System (EDAS) database for the 1320 randomly located benthic samples in the Piedmont and Highlands regions that were collected over the 10-year period (1994-2004) and analyzed as response variables. These included total taxa (taxon richness), number of taxa in the insect families of mayflies (Ephemeroptera), stoneflies (Plecoptera), and caddisflies (Trichoptera) (EPT taxa collectively), and the 2005 version of the Maryland benthic index of biotic integrity (B-IBI). For fish, response variables examined include the Maryland fish IBI, total number (abundance) of fish, and number of species of fish (taxon richness). The main analytical focus was on 2 fish indicators and 2 benthic macroinvertebrate indicators: the Maryland fish IBI score, and fish taxon richness; and the Maryland B-IBI score, and the EPT taxon richness. The selected indicators are all responsive, general indicators of stress but are not diagnostic of any particular stressor.

The Maryland 305(b) evaluation of the status of waters of the state, which uses the MBSS data in addition to other data sources, uses benthic and fish Indices of Biotic Integrity (IBIs) to determine impairment status and attainment of uses (Maryland Department of the Environment, 2004; http://www.mde.state.md.us/assets/document/AppndxC2004-303d_Final.pdf). For a single stream reach assessment, Maryland takes into account population-wide measurement error. The approximate result is that if both indexes are ≥3.3, the stream segment is considered unimpaired, and if either index is ≤2.7, the segment is impaired. Intermediate values are considered to be potentially impaired but are still listed as supporting aquatic life uses.

5.2.3. Regional Data Partitions

The MBSS data were partitioned based on Maryland's classification into four ecoregions (Coastal Plain, Eastern Piedmont, Cold-water Highlands, and Warm-water Highlands; see Figure 5-1), to account for known sources of natural variation in both habitat (physical and chemical) and biological data. The heavily developed Eastern Piedmont region, with a high level of urbanization that represents an existing source of impairment, was targeted for evaluation. Due to the level of development, the Eastern Piedmont region has relatively few reference areas. In the original MBSS index development, the Piedmont and Highlands regions were deemed to be biologically similar (Roth et al., 1998; Stribling et al., 1998). Sampling of more reference sites showed that the Piedmont can be separated from the Highlands region (Southerland et al., 2007, 2005). However, to have sufficient reference locations to support the analyses, the original classification was used, recombining the Piedmont and Highlands sites for analyses requiring identified reference sites.

5.2.4. Climate Data

The National Climatic Data Center (NCDC; www.ncdc.noaa.gov) makes available several average monthly parameters, organized by state climatic regions. Although climate does not follow state boundaries, it was convenient in this case because our biological data did follow the state boundaries. We used data from two NCDC regions of Maryland: the Northern Central Division (primarily Northern Piedmont ecoregion, and the Blue Ridge ecoregion within

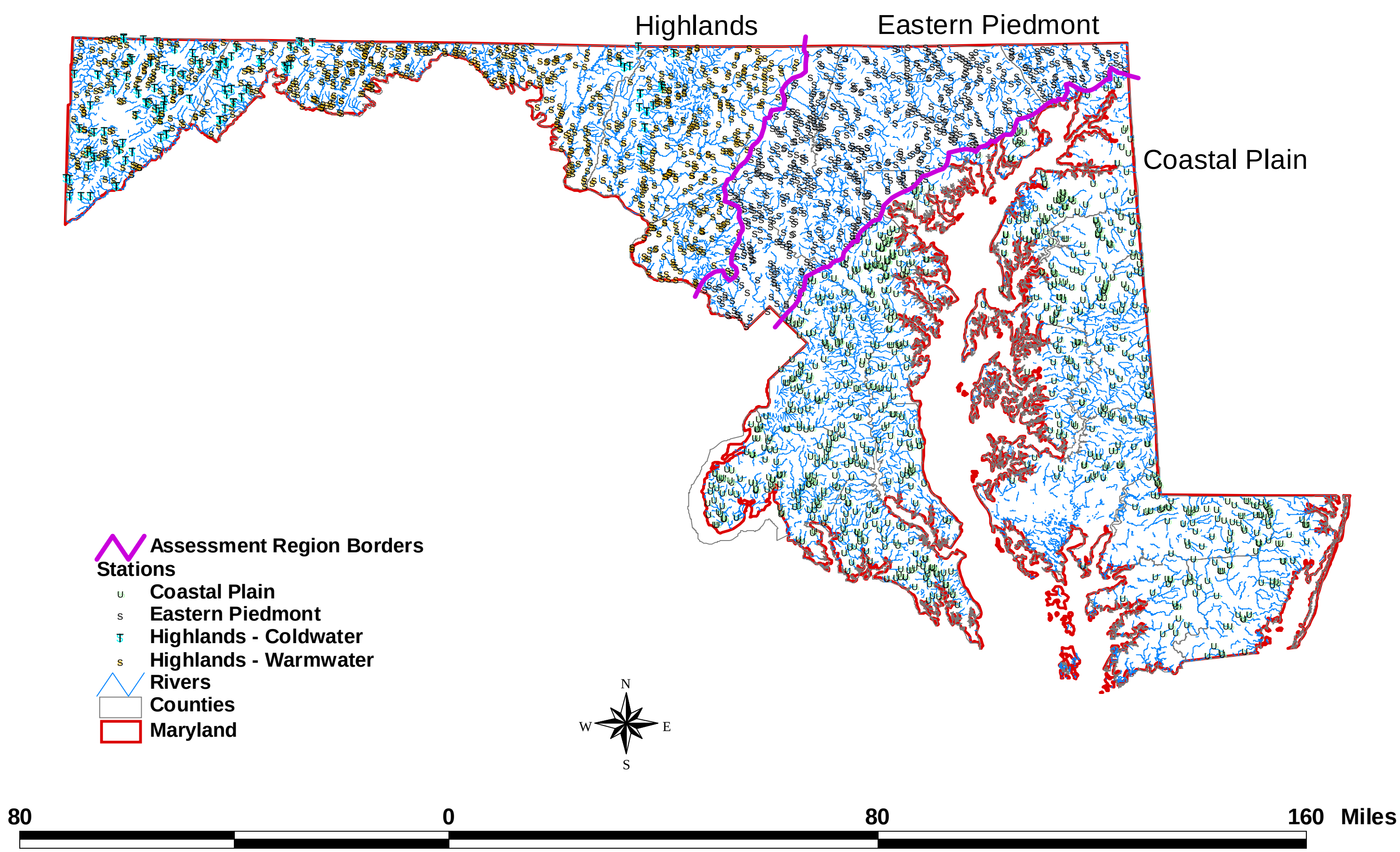

Figure 5-1. Maryland MBSS sampling stations showing regional divisions.

Maryland), and the Appalachian Mountain Division (Central Appalachian Ridge and Valley ecoregion). Maryland's Piedmont and warm-water mountain streams occur primarily in these two climatic divisions. The Northern Central Division data were applied to the Eastern Piedmont streams, and the Appalachian Division data to Highlands streams.

We estimated potential hydrologic effects of climate change by using the Palmer Hydrologic Drought Index (PHDI) as a proxy for estimates of hydrologic changes due to climate change. The PHDI is a monthly hydrological drought index used to assess long-term moisture supply to water bodies (Karl, 1986) and is described in detail on the NCDC website (http://www.ncdc.noaa.gov/oa/climate/onlineprod/drought/ readme.html). The index ranges from −7 to +7, with negative values indicating dry spells, and positive values indicating wet conditions. The PHDI takes into account water storage as soil and groundwater, and therefore is more applicable to stream flow than the PHDI, which uses only temperature and rainfall information (Karl, 1986). Figure 5-2 shows the 30-year distribution of the PHDI for the Maryland Northern Central Division, which includes the Piedmont. The range of the PHDI varies little from month to month, but the 30-year median value is positive during the spring/summer macroinvertebrate sampling index period (>0 in March–May) and markedly lower in late summer-early fall during the last half of the fish sampling index period (<−1 in September–October).

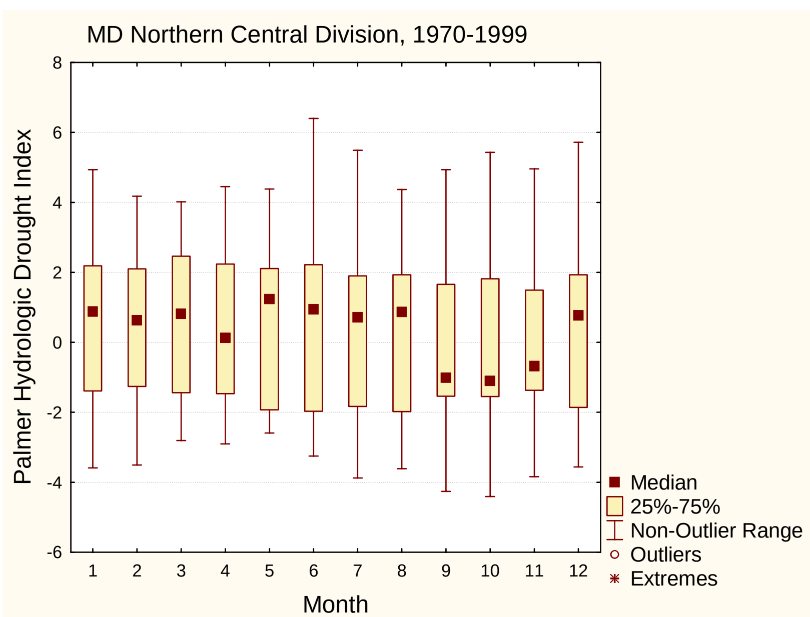

Figure 5-2. Monthly Palmer HDI for the 30-year period 1970-1999 (Source: NCDC; http://www.ncdc.noaa.gov).

In addition to calculating monthly PHDI, a variety of alternative running averages were also calculated to account for possible lags in effects, and specifically for the time lag between when droughts occur and directly impact the biota (summer/early fall) and the sampling period (the spring index period): the previous 6-month average, the previous 12-month average, and the previous summer PHDI. These alternative PHDI averaging methods did not yield results (i.e., in developing wet/dry/average year comparison) different from those developed using the simple monthly average; therefore average monthly PHDI was used in all analyses.

5.2.5. Hydrologic Attributes

Baker's flashiness index (Baker et al., 2004) was estimated for each stream. Flashiness is a component of the hydrologic regime of streams, and, in general, is related to the frequency of short-term changes in runoff associated with rainfall events, and how rapidly each event comes and goes. Flashiness is generally considered to increase with increases in impervious cover associated with urbanization and/or with land clearing for agriculture (Allan, 2004). It is both responsive to urbanization as an existing stressor, and it also is expected to change in the future in response to climate change projections of increased frequency and intensity of storms within many regions of the U.S. Baker's index is calculated as the average of absolute values of daily mean flow change divided by mean flow for the 2-day period. The maximum range is from 0 (absolutely constant flow) to 2 (alternating days of flow and no flow).

Daily flows were simulated for each site using the Flow Time Series Estimation tool (FTSE; Tetra Tech, 2005). The model can be used to estimate daily flows for ungauged streams based on multiple regressions using a smaller set of gauged streams. The main criterion for proper functioning of the model is that there must be gauged stations relatively near to the ungauged streams (e.g., within the same ecoregion) so that a standard is available for calibrating the model. Estimates had been developed for a set of 764 streams in the Piedmont only (Barbour et al., 2006); no set of appropriate gauged streams was available for the Appalachians.

5.2.6. Specific Analyses

5.2.6.1. *Sensitivity of the System to Climate Change*

As discussed in Sections 1 and 3, the primary hydrologic stressors associated with climate change are changes in precipitation patterns combined with changes in temperature regime, which will drive changes in hydrologic regime. The projected extent of changes in temperature and precipitation varies regionally in the U.S.; therefore, so too will changes in the magnitude, frequency, flashiness, and other patterns of runoff. The National Assessment of climate change in the U.S. provides regional summaries of projected changes in the temperature and precipitation regimes of the major regions of the U.S. (NAST, 2001). Table B-1 (in Appendix B) summarizes these projections by region.

5.2.6.2. *Stress-Responses*

We established the ranges and variability of system response variables to climate change, and described the response signatures to stressors. Due to the large number of parameters available in the MBSS database, correlation analyses were used to identify stressor variables that were most strongly related to response variables. Spearman Correlation coefficients were used for this analysis; as a non-parametric test, there is no need to make assumptions about or test for normal distributions for each variable. A locally weighted scatter plot smoothing (LOWESS) line was used to illustrate the pattern of any relationship between the variables being correlated. LOWESS smoothing was done in Systat10 by running along the x values and finding predicted values from a weighted average of nearby y values. The surface is allowed to flex locally to better fit the data. For the LOWESS, the degree to which the line or surface (tension) is allowed to flex locally to fit the data to 0.5 was specified, meaning that half the points are included in the running window. Graphs of key variables (scatter plots with LOWESS line) were used to illustrate the relationships defined by correlation analysis, and to confirm that all relationships reflected consistent data with no errors or false trends introduced by data entry errors, reporting unit errors, or other inconsistencies. The subset of parameters showing the strongest relationships were used for further exploration of stressor-response models.

A conditional probability approach (Paul and MacDonald, 2005) was used to examine changes in the biological community along stressor gradients. A conditional probability statement provides the likelihood (probability) of a predefined response, if the value of a pollutant stressor (condition) is exceeded. Conditional probability is the probability of an event when it is known that some other event has occurred. To estimate conditional probability of impairment, we first define impairment as a specific value for a response variable (e.g., EPT <11 genera). The analysis addresses the following question: for a given threshold of a stressor, what is the cumulative probability of impairment? For example, if total phosphorous concentration is greater than 0.2 mg/L, what is the probability of biological impairment for each site under consideration? All observed stressor values (in this example, all observed values of total phosphorous) are used to develop a curve of conditional probability (Paul and MacDonald, 2005).

5.2.6.3. *Effects of Climate Change*

We used proxy estimates of climate (in the existing data) that are representative of projected climate change, and examined the ability to detect biological impairment and stressor-response relationships. Our proxies of climate change were the estimates of wetter-than-normal and drier-than-normal conditions in the PHDI for each sampling event. The MBSS data were post-stratified into dry, normal, and wet conditions based on the index; selected stressor-response relationships were then reexamined under the wet and dry scenarios. This evaluation was done separately for reference sites (defined *a priori* in the MBSS), impaired sites (defined *a priori* in the MBSS plus sites with 10% or more impervious surface), and intermediate sites (sites not included in the impaired or reference groups).

As part of the analysis, it can be assumed that future biological responses to altered hydrological conditions will be similar to responses to current natural variability, and that future hydrologic changes will be comparable to extremes observed in the past 10 years. The assumptions are probably reasonable in the near-term (i.e., 50 years), but become less reasonable farther into the future.

5.3. KEY FINDINGS

5.3.1. Observed Stressor-Responses

Establishing definitive stressor-response relationships is a critical step in the Stressor Identification (SI) process, and it is fundamental to identifying probable causes of impairment. Numerous relationships were examined; Appendix B summarizes these results. Only a subset of results that show some correlation and/or those that were considered potentially important but showed no significant relationship are presented in this section.

The fish and benthic invertebrate response variables that showed the strongest responses in these correlations were the fish index of biotic integrity (FIBI), fish taxon richness, total number of fish, the Maryland benthic IBI, total benthic taxon richness, and total EPT taxon richness (see Appendix B). From these, 2 fish indicators were selected and 2 benthic macroinvertebrate indicators were used to evaluate the Maryland Fish IBI score, and fish taxon richness; and the Maryland Benthic IBI score (B-IBI), and EPT taxon richness.

5.3.1.1. *Physical Habitat*

Both fish and benthic macroinvertebrate measures were correlated with overall physical habitat, as measured by the Maryland Physical Habitat Index (Paul et al., 2002) (Appendix B). Fish taxon richness was not correlated with the habitat index, but the fish IBI and both invertebrate indicators were strongly correlated, increasing with improved habitat score. Among habitat components, the EPT taxa were also positively correlated with the embeddedness score, reflecting a component of habitat (interstitial spaces in cobble substrate) utilized by these organisms. Fish taxon richness was also very strongly correlated with total flow, but this was a reflection of the effect of stream size.

5.3.1.2. *Hydrology*

Both the fish and the benthic macroinvertebrate indicators were negatively associated with Baker's flashiness index (see Appendix B). Below a flashiness index value of 0.5, biological indicator values could be in the normal range, but above a flashiness of 0.6, most biological values indicated impairment. Flashiness is affected by impervious surface, which in the study area, indicates urban land use. The macroinvertebrate indicators declined with impervious surface in a catchment, but the fish indicators did not (see Appendix B).

5.3.1.3. *Water Quality*

The invertebrate indicator EPT was associated with dissolved organic carbon (DOC), total phosphorus (TP), and conductivity, with the number of EPT taxa declining as the stressors increased. The strongest association was with conductivity. No other chemical water quality measures were associated with either fish or benthos (DO was uniformly moderate to high in the dataset, and there were too few observations of low DO to show any relationship).

5.3.1.4. *Temperature*

The associations of both the fish and benthic macroinvertebrate communities to water temperature were examined. Fish observations in the data set had already been classified according to expected warm-water and cold-water communities, using current and likely sustainable distributions of brook trout to define cold-water streams in the region west of Evitts Creek in western Maryland (Southerland et al., 2005). It is important to note that temperature was measured in late summer and fall, at the same time that the fish assemblage was sampled. Macroinvertebrates were sampled in spring, and temperature was not measured at that time.

Fish taxon richness increased with temperature in warm-water streams in both the Piedmont and in the Appalachians, but there was no detectable relationship in the cold-water streams (see Figure 5-3). EPT and total macroinvertebrate taxon richness (measured in early spring) were reduced in the cold-water Highland streams where late summer temperatures exceeded 18-20ºC (see Figures 5-4a, b). There was no detectable relationship between temperature and benthic macroinvertebrates in Piedmont streams (see Figures 5-4c, d).

5.3.2. Estimates of Climate Change Effects

5.3.2.1. *Temperature*

Increases in average regional temperature might have the result that some fraction of cold- or cool-water streams change to warm-water conditions and biota. Global average air temperatures are expected to increase by at least 2ºC by 2100 (likely range 2ºC to 4.5ºC, likely

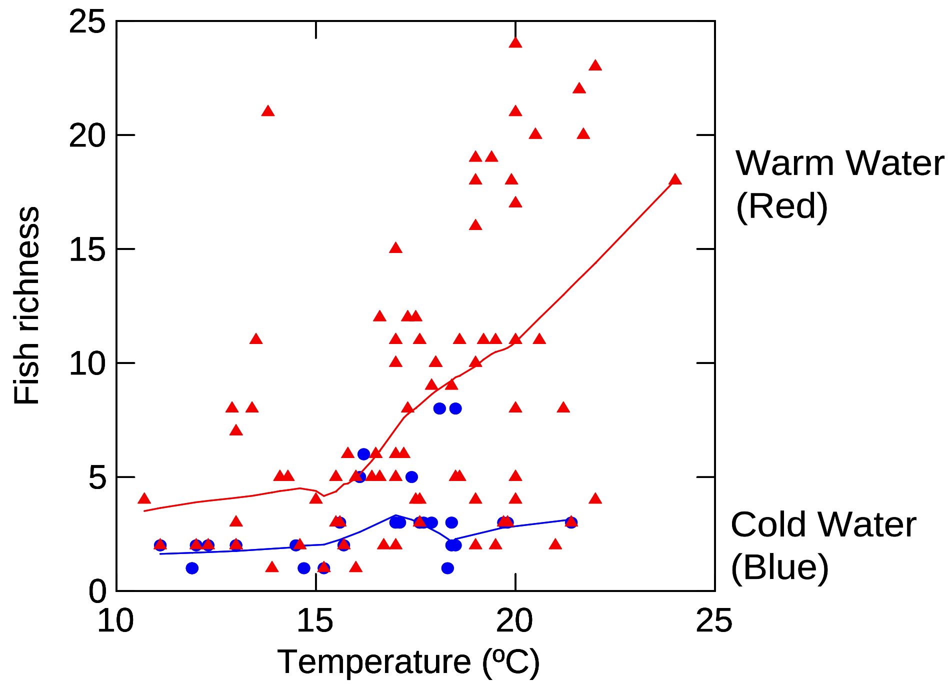

Figure 5-3. Fish richness vs. temperature in Highland reference streams. Lines are LOWESS estimates.

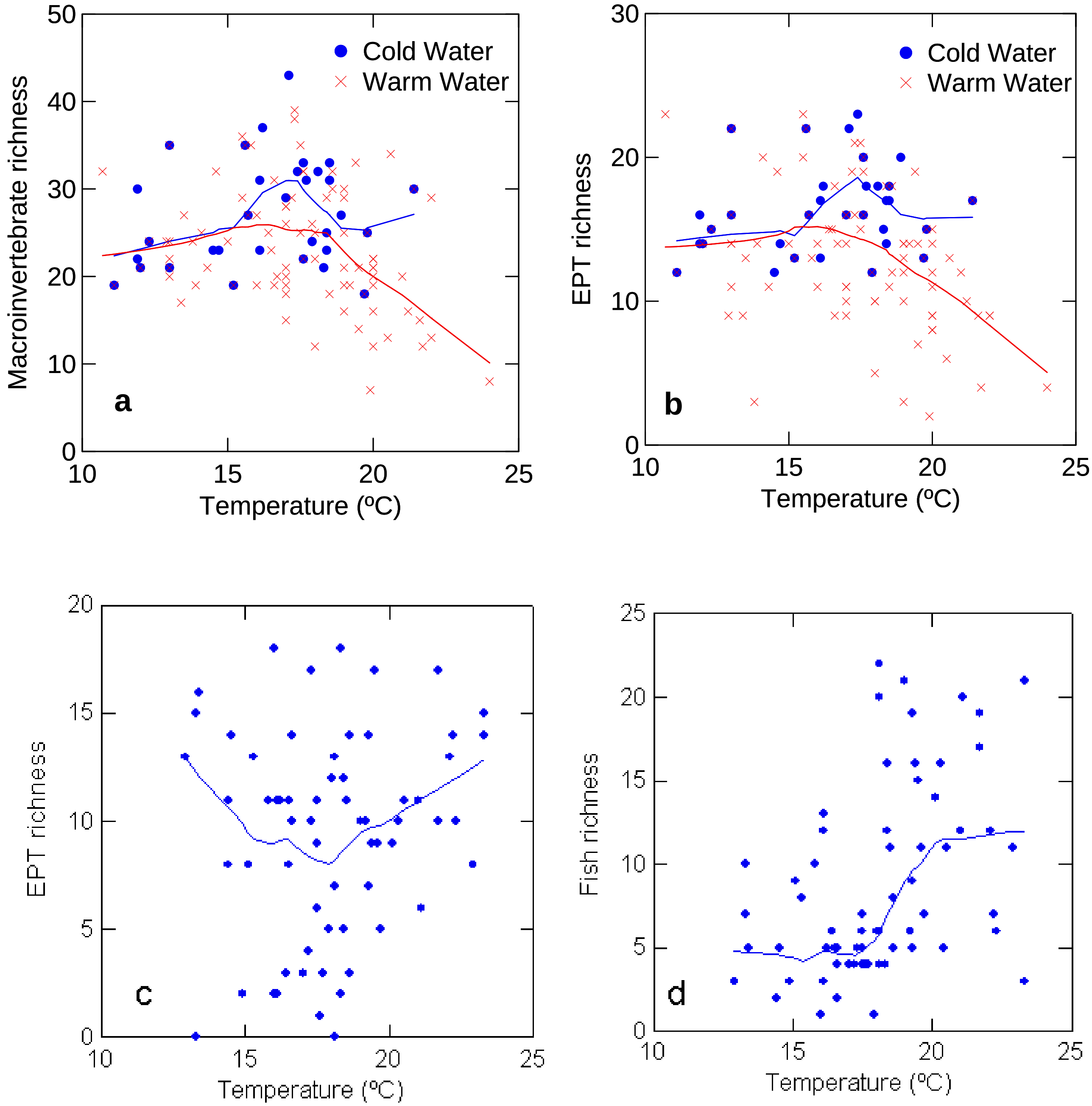

Figure 5-4. (a) Macroinvertebrate richness vs. temperature; (b) EPT richness vs. temperature in Highland reference streams; (c) EPT vs. temperature relation; and (d) fish richness vs. temperature relation in reference sites in Piedmont streams. Lines are LOWESS estimates.

average 3ºC; IPCC, 2007). On the average, summertime air temperature increases are projected to be less than wintertime increases (MacCracken et al., 2001), and the late-summer stream water temperature increases are expected to be less than the average increase (Mohseni et al., 2003). Based on these results for fish and invertebrate taxa in Mid-Atlantic streams, a net increase in site-specific fish richness can be expected, as individual streams change from cold- or cool-water conditions to warm-water. Fish taxon richness has previously been found to be higher in warm-water habitats (Wehrly et al., 2003). In contrast, invertebrate taxa per site may decrease in Highland streams that exceed 18ºC due to climate change, but with no change in streams that remain well below 18ºC in late summer, suggesting that Highland streams macroinvertebrate communities may be sensitive to climate change according to (future) temperature regime.

5.3.2.2. ***Hydrology***

To examine the potential effects of changed rainfall and evapotranspiration patterns, the existing data was divided into three groups: samples taken in relatively dry conditions, samples taken in approximately normal conditions, and samples taken in relatively wet conditions. "Dry," "Normal," and "Wet' were defined according to the distribution of the PHDI in the data set, thus, the range of conditions from the recent past (from the month of sampling to the preceding year) was used to obtain some insight into consequences of climate change. The range of PHDI was from −4.24 to +4.75, with a median of +1.8. Although the total range was symmetrical from extreme drought (< −4) to extreme wetness (> +4), there were more wet months than dry months in the 10-year period. We defined 3 climatic groupings: Dry: PHDI < -2.5 (N = 264); Normal: −1.1 < PHDI < 1 (N = 176); and Wet: PHDI > 3.5 (N = 353). These groupings were selected to get substantial differences between wet and dry conditions, i.e., to eliminate confounding effects of "moderately dry" and "moderately wet" conditions, and yet have sufficient sample size in each of the hydrologic groups.

Figure 5-5 shows the Benthic IBI (B-IBI) scores of the three stream classes under the three climatic conditions. Dry conditions are associated with greater variability of reference sites, and a net degradation of median B-IBI score in both reference and intermediate sites. Wet conditions are similarly associated with increased variability and a net decline in median B-IBI score, but less so than in dry conditions. A comparison of reference sentinel site B-IBI (as well as F-IBI) scores for the period 2000–2004, which included a wet year (2003) and a drought period (2002–early 2003) did not show a notable variation between years among the Piedmont and Highland sentinel sites (Prochaska, 2005). However, fewer sites and years are included in that analysis, and variation around the mean is not evaluated. The EPT taxa metric showed the same overall pattern (Figure 5-6): a slight net loss of median number of taxa in reference and intermediate sites, and increased variability in reference sites. The macroinvertebrate

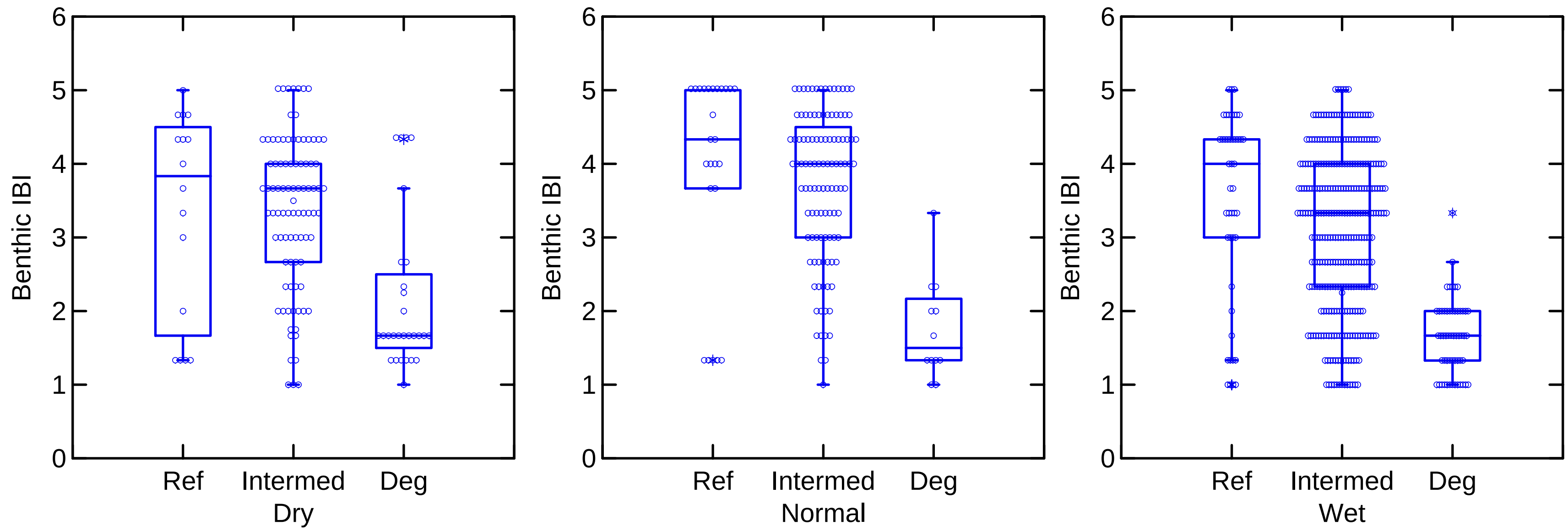

Figure 5-5. Benthic IBI performance and climatic condition. The dry, normal and wet designations under each of the three graphs refers to categorizations based on the PHDI.

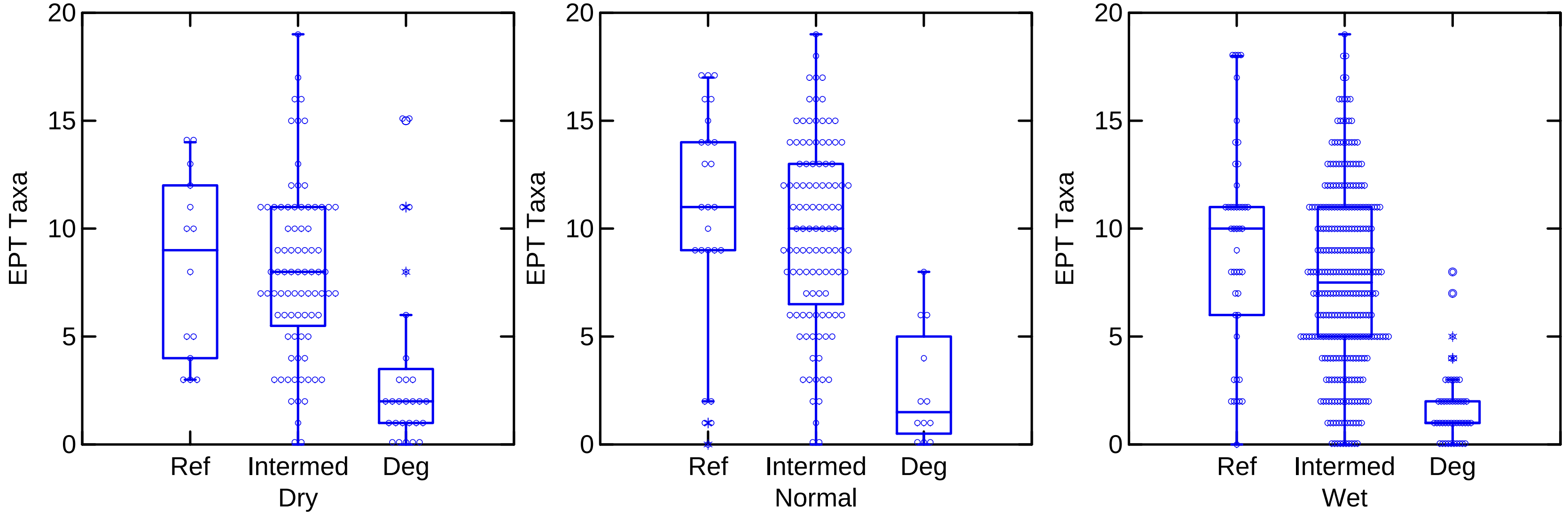

Figure 5-6. EPT performance and climatic condition.

communities at degraded sites were low in EPT taxa and IBI scores, so changes of hydrological condition did not affect them much. The Fish IBI was also similar (Figure 5-7) but showed slightly greater effects under wet conditions than did macroinvertebrates: larger decline in median reference score and larger reference variability. It should be noted that there are differences in sample sizes between categories of climate condition, and, in particular, there are fewer “dry” years represented. This is a consequence of limited (10 years) data combined with the rotating-basin sampling scheme (i.e., only a subset of basins are sampled each year). Having more data would improve these comparisons.

A quantitative measure of the efficacy of an index in discriminating between reference and stressed sites is the Discrimination Efficiency (DE), which is calculated as the percent of stressed sites with scores less than the 25th percentile of the reference sites (U.S. EPA, 1999). DE is influenced both by the absolute difference between the reference and stressed site mean scores, and the variability or spread of the scores. DEs under the scenarios described above are given in Table 5-1. From this analysis, it appears that increased drought degrades reference sites enough to reduce the ability to discriminate impaired from reference conditions for both the benthic IBI and EPT taxon richness. Interestingly, the median value under both dry and wet conditions was reduced compared to normal conditions in the intermediate sites, indicating a net impairment from normal conditions. Also, the overall spread or variability of reference IBI scores increased in both the wet and dry scenarios.

Benthic macroinvertebrates were sampled in spring, and fish were sampled in late summer and fall. In wet years, the fish IBI showed much higher variability in reference sites, reducing the discrimination efficiency (see Figure 5-7, Table 5-1). Late summer and fall are slightly drier than other times of the year: the 30-year median of the PHDI during the fish sampling index period is less than −1 (Figure 5-2). Thus, wet conditions during the fish sampling period may represent a greater departure from a median expectation than do dry conditions during the invertebrate sampling period. This may explain the increased variability of the reference site fish IBI values under wet conditions than under dry conditions, and the reduction of discrimination efficiency.

The pattern of extreme (wet or dry) hydrologic conditions both decreasing mean index values at reference stations and increasing variability demonstrates a tendency for these surrogate estimates of hydrologic changes associated with climate change to drive reference locations to be more like impaired locations, and thus decrease the ability to discriminate between the two based on biological indicator data.

The association of climatic condition on the relationships between EPT taxa and two environmental stressors, conductivity and impervious surface, which had shown good stressor-

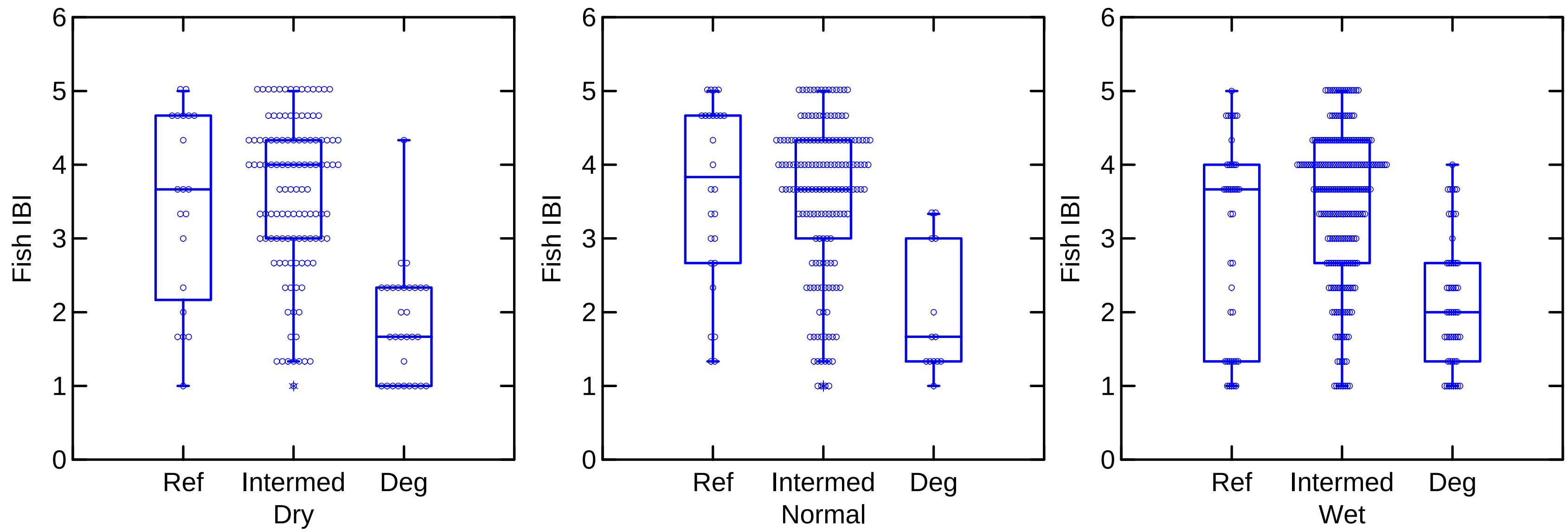

Figure 5-7. Fish IBI performance in three climatic conditions. The dry, normal and wet designations under each of the three graphs refers to categorizations based on the PHDI.

Table 5-1. Discrimination efficiencies of IBIs and EPT taxa under 3 climatic conditions

Climatic condition	Benthic IBI	EPT Taxa	Fish IBI
Base (current normal year)	100%	100%	69%
Dry year	64%	78%	60%
Wet year	98%	95%	16%

response relationships, was examined further (Appendix Figure B-6). The full data set (both Piedmont and Highland warm-water streams) was included in these analyses.

The plots of the stressor-response relationships from all three climate scenarios were overlaid to determine whether changes in the response curves might be associated with the climate change scenario, namely increasing drought or increasing storm events. Figure 5-8 shows the stress-response relationships (with linear regressions) between EPT taxon richness and conductivity for the Piedmont region, and the conditional probability analysis. First, mean number of EPT taxa is generally higher in the base condition, and reduced under wet conditions, with little difference between base and dry conditions.

The conditional probability analysis (see Figure 5-8b) examined the probability of impairment along the stressor gradient. EPT taxa <8 was defined as the threshold of impairment, consistent with the threshold used by Maryland DNR in the Piedmont (Southerland et al., 2005). Conditional probabilities of EPT impairment under base, wet and dry conditions, show that the probability of impairment is higher under the wet scenario than under baseline conditions This is not merely the result of reduced conductivity in wet years because the overall distribution of conductivity in wet and normal years is almost identical (see Figure 5-9; CDF of conductivity). Under dry conditions, the probability of impairment was greater at low conductivities, and less at high conductivities, though the actual difference in numbers of EPT taxa were small.

The natural conductivities of streams in the region are generally low due to low buffering capacities of the parent rocks and soils, with the exception of limestone-influenced streams in the Great Valley, in smaller limestone valleys of the Ridge and Valley ecoregion, and marble formations in the Piedmont ecoregion (Woods et al., 1999). Increased conductivity is consistently and reliably associated with reduced stream biological condition throughout the Appalachian region (Gerritsen and Zheng, unpublished data).

Figure 5-10 shows the relationship with impervious surface for the climate scenarios. One of the consequences of urbanization is an increase in impervious area from roads, parking

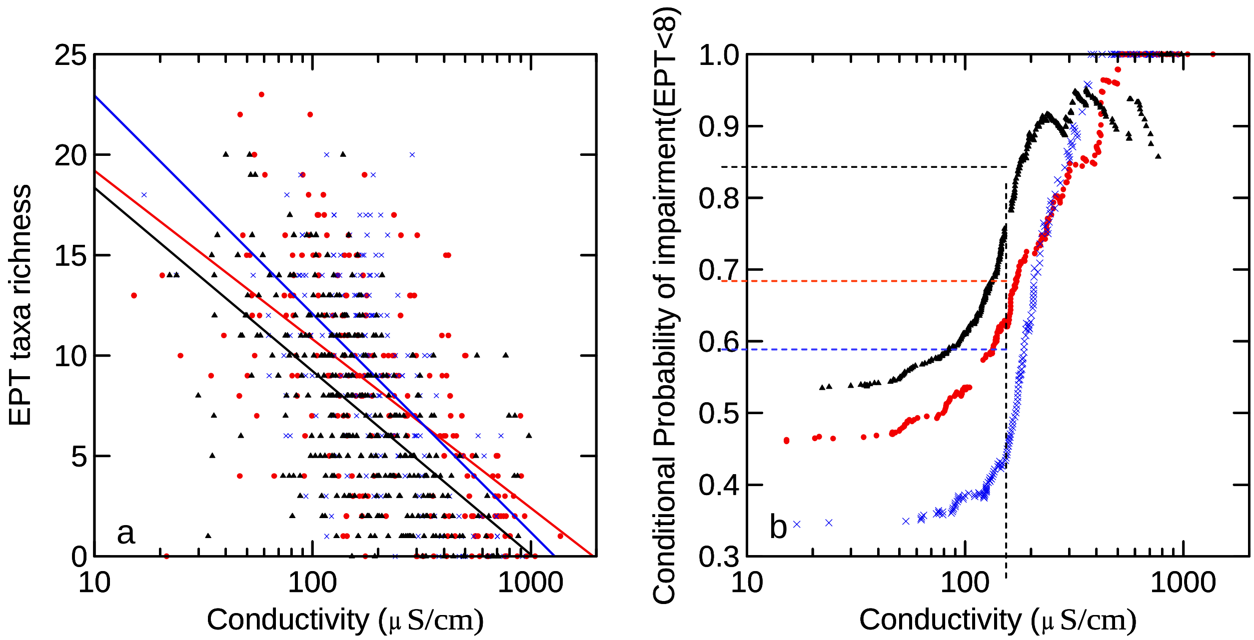

Figure 5-8. (a) Relationship of EPT richness to conductivity under drought (red), base (blue), and wet (black) conditions and (b) conditional probability in impairment for the same three relationships.

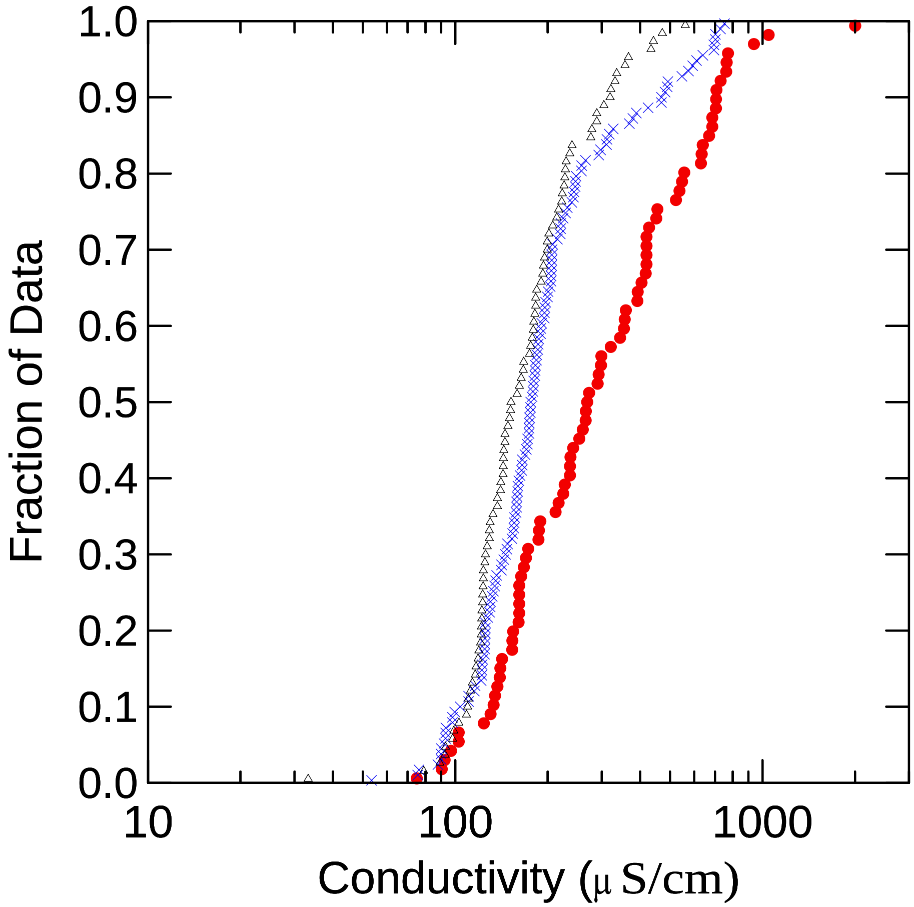

Figure 5-9. Conductivity CDFs—Piedmont. Drought—red, base—blue, wet—black.

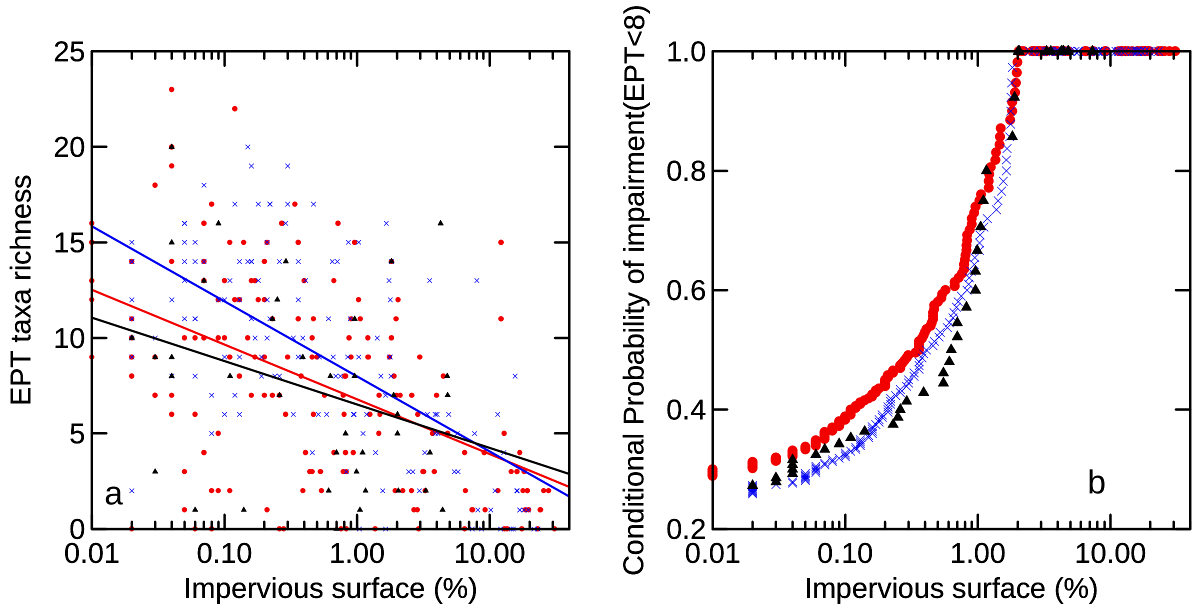

Figure 5-10. (a) Relationship of EPT richness to impervious surface under drought (red), base (blue), and wet (black) conditions, and (b) conditional probability of impairment for the same three relationships.

lots, and rooftops. Impervious surface increases the "flashiness" of streams, as well as being a conduit for urban contaminants and pollutants. Overall, the base group (i.e., average hydrologic conditions) had higher levels of EPT taxa than either the drought or the storm groups, but the differences were subtle, a difference of approximately 1–2 taxa, and the differences were not consistent. Drought conditions yield a higher risk of impairment with impervious surface, but the change is marginal.

5.4. DISCUSSION AND KEY CONCLUSIONS

Several biological indicators and their associations with stressors have been examined under scenarios of normal, relatively dry, and relatively wet conditions. These scenarios were derived by partitioning a long-term data set from the Mid-Atlantic Piedmont and Appalachians by moisture conditions estimated by the PHDI. Some caveats regarding the sampling design and the partitioning include the following:

- Grab-sample temperature and water chemistry measurements did not coincide with the benthic macroinvertebrate samples. Macroinvertebrates as well as nutrients were sampled in spring (March–early May), coincident with the spring freshet; fish and water quality (temperature, DO, conductivity, habitat, etc.) were sampled in late summer (June–September), coincident with annual low water. Although the MBSS deploys continuous temperature loggers during the summer index period (June 1–September 30), these data were not available at the time the analyses for this report were conducted, and the deployment still does not cover the benthic sampling index period. Different index periods for the organisms would have resulted in different drought index estimates.
- The PHDI applied to the month and year a site was visited; all sites sampled in the same month (e.g., March 1999) and NCDC district (e.g., Piedmont) would have the same PHDI value.
- The climatic conditions we examined are all recent, from the period 1995–2005. Future climate is expected to show a greater frequency of extreme conditions, but they have not been linked to the frequency and magnitude of climate projections and models.

Differences in median values and distributions of several biological indicators associated with dry, normal, or wet conditions were not observed; however, the associations may have been due to an "unlucky" random sample and can not be ruled out—especially at the basin level. All samples from a particular basin-year sampling would fall in the same dry-normal-wet category, and there is no assurance that basins sampled in any one year are representative of the range of stressor conditions throughout the region, especially with respect to urbanization.

In spite of these caveats, the results indicate the potential consequences of climate change on bioassessment indicators. In dry and wet years, indicator variability increased markedly in reference sites and there were slight reductions in median indicator values. Consequently, there was reduced ability to discriminate between reference and stressed sites under dry conditions (especially for macroinvertebrates) and under wet conditions (especially for fish). Associations of the indicators with stressors, which are used to develop stress-response relationships for SI (Suter et al., 2002; Norton et al., 2002), may also change as evidenced by the apparent response to conductivity; though there was little change in response to impervious surface.

5.4.1. Reference Conditions

These results illustrate the potential sensitivity of reference sites to climate change. Reference sites in many regions of the country are not pristine, but are merely the "best available" in the region. This is especially true for the eastern Piedmont ecoregion, which has been settled, farmed, and industrialized since Colonial times. It is unlikely that there are any sampled watersheds in the Piedmont of Maryland that are free of suburban development; the average population density of HUC-8 accounting units in the Maryland Piedmont ranges from 111 to >400 persons per square kilometer (1990 census; U.S. EPA, 1997).

Moderately stressed reference sites may be more sensitive to slight increases in additional stress due to climate change than truly minimally stressed reference sites (Stoddard et al., 2006). Therefore, it would be important to identify minimally stressed reference sites if they exist, to document reference site selection criteria, whether minimally stressed or not, and to monitor reference sites to document changes over time.

5.4.2. Importance of Monitoring

To be able to account for the effects of climate change on biological indicators and on stressor-response relationships, it will be necessary to monitor a set of fixed sites over time ("sentinel" sites), such that the same sites are revisited. Systematic changes in biological attributes can only be attributed to climate change if other potential causes are eliminated or accounted for, hence the need to have sentinel sites that span a wide range of other potential stressors, and not just least-stressed reference sites.

Because climate change effects are pervasive, components of trends that are common to all sentinel sites can be assumed to reflect climate change effects. If no other degradation was occurring at reference sites, then the magnitude and variation in trends at reference sites could be used directly to characterize the climate change component and account for that component within trends observed at non-reference sites. However, assumptions of continued "pristine" (or even steady) conditions at reference sites are unlikely over time, given population growth, invasion of non-native species, expected encroachment of suburban and other land uses, increased water withdrawals for human use, and other landscape-scale effects. Even if recommendations to protect reference sites are adopted, lack of contribution from landscape-scale stressors would have to be verified in the process of estimating climate change-associated trends.

Once trends common to all sentinel monitoring sites are defined, different components of trends at non-reference sites can be considered potentially due to other stressors and evaluated through the SI approach.

5.4.3. Analytical Methods

A question that arises is whether there are more robust or more powerful analytical methods that can overcome the projected degradation in signal quality and discrimination ability. Unfortunately, it is the quality of the information (signal to noise) that will degrade, and not the analytical methods. If the information is degraded, then no amount of statistics can recover something that no longer exists. Nevertheless, tracking time trends at both reference and non-reference "sentinel" locations over time provides a framework for defining climate change-associated trends and differentiating these from the effects of conventional stressors that are of regulatory interest.

In view of the likelihood of ubiquitous biological degradation due to climate change, it becomes increasingly important to protect reference sites from degradation. Application of the Biological Condition Gradient (BCG) (a kind of universal measurement yardstick that will be explain in Section 5.4.6) and Tiered Aquatic Life Uses (TALU) would establish a framework for such protection (U.S. EPA, 2005) (see also Section 5.4.6). For example, one expected outcome of defining TALUs is that states would adopt "high" and "exceptional" quality use classes along the BCG, which would be above their current action threshold for "fishable/swimmable." Each aquatic life use class would have biological criteria associated with it, which would allow detection of degradation at reference sites at a stage substantially before the reference site would be "impaired" under current definitions. Such a formalized process also provides for implementation of particular management actions, such as identification of the cause of impairment and implementation of corrective actions.

5.4.4. Stressor Identification

At least some associations of the indicators with stressors, which are used to develop stressor-response relationships for SI (Suter et al., 2002; Norton et al., 2002), are expected to change as hydrological conditions are altered by climate change. There was a marked change in the stressor-response relationship between macroinvertebrates and conductivity under wetter than usual conditions, which was associated with an increased probability of impairment. However, almost no response was observed for impervious surface. SI may be similarly hampered by pervasive degradation and increased variability of all sites. If the conductivity stressor-response in the wet condition is considered a typical scenario, then conductivity is implicated in a smaller fraction of impairment (because the baseline frequency of impairment is higher), yet the threshold water quality criterion for conductivity would also be lower. That is, protection from degradation by conductivity may need to be tighter and set at a lower conductivity than before the climate changed.

5.4.5. Biocriteria

Increased variability of reference sites as a consequence of climate change could decrease the ability of states to detect impairment, if impairment thresholds are determined by a statistical percentile of the indicator distribution in reference sites. Many states use a lower percentile of the reference distribution as a numerical biocriterion for 305(b) assessment, for example, the 25th percentile (Ohio EPA), or the 10th percentile (Maryland), or the 5th percentile (West Virginia). If climate change causes the percentiles to drift downward, and the state reevaluates its water quality criteria with new data, then the new criteria may set a lower bar, i.e., permit more degradation to take place, before any kind of management is implemented (e.g., total maximum daily load (TMDL) calculations). The potential drift of reference site condition due to climate change illustrates the importance of establishing a universal measurement scale of biological condition (e.g., the BCG) so that reference site drift can be identified as such.

5.4.6. Universal Scale to Measure Biological Condition

Acceptable biological condition is determined in many states from statistical properties of a numerical index. Index values and criteria vary widely from state to state because of differences among data sets used to develop the respective indexes. Furthermore, the criteria "action level" often reflects substantial biological degradation from relatively undisturbed conditions, such that the highest quality waters are not adequately protected. Results of this case study demonstrate that biological responses to climate change may further confound assessment and criteria for water management. To resolve these issues, panels of state and academic aquatic biologists have proposed a conceptual model for a universal measurement scale of aquatic biological condition—the BCG (Davies and Jackson, 2006).

The conceptual BCG model describes ecological changes that take place in flowing waters with increased anthropogenic degradation, from pristine to degraded (Davies and Jackson, 2006). The BCG promotes consistency among agencies in the application of the CWA by identifying tiers, or condition classes, that can be operationally defined in a consistent manner. The model is intended to be broadly applicable to any kind of stream; the tiers are independent of actual monitoring methods. Although the model promotes conceptual unification, it recognizes regional natural variability, and is <u>not</u> applied as a one-size-fits-all approach. The BCG is a general description of change in aquatic communities, it is consistent with ecological theory, and the BCG approach has been verified by aquatic biologists throughout the U.S. (Davies and Jackson, 2006).

Calibration of the BCG to local conditions, and on a nationwide basis, would help establish two baselines that would reduce the effects of confounding by climate change. The first baseline is the description of pristine or nearly pristine conditions, Tier 1 of the BCG. In many regions, the description of Tier 1 must rely on historical descriptions of fauna and historical ranges of organisms (these may be available for fish, but rarely aquatic invertebrates), on modeling approaches, on best professional judgment, or on sites available across political boundaries (Stoddard et al., 2006). The second baseline is the description of the present-day reference, or least stressed condition, before large-scale effects of climate change have occurred.

6. RECOMMENDATIONS FOR U.S. EPA TO IMPLEMENT A FOUNDATION FOR STATE/TRIBAL BIOASSESSMENT/BIOCRITERIA PROGRAMS TO CONSIDER CLIMATE CHANGE

6.1. RECOMMENDATIONS FOR U.S. EPA

Results of the case study analyses to date, continued development and review of indicator sensitivity classification, and discussion and input from state/tribal biocriteria managers at the Workshop in March 2007 provide the basis for recommending the focus of ongoing and future efforts to continue development and implementation of a framework for biological assessment programs to account for climate change effects. Recommendations can be categorized as technical requirements and resource requirements. Technical requirements focus on information needed to better understand the interactions between expected effects of climate change and biomonitoring program endpoints, additional technological support, and general policy support. During the Workshop, some of these activities were identified as falling within the purview of the U.S. EPA's ORD, and some in the purview of U.S. EPA's Office of Water (OW). These identifications are made after each recommendation.

The ORD can

- Conduct further research through pilot studies (see Section 7) to determine the best hydrologic and biological response indicators, to define biologically sensitive measures to hydrologic changes, and to identify species traits responsive to climate change (temperature, flow, sediment).
- Investigate how taxa replacement will affect biological indices used in state programs. Determine the extent of change in the biological indices if specific metrics are changed.
- Develop and provide technical guidance regarding program adaptations and other approaches needed to account for climate change in biological assessment programs, including categorization of indicators (metrics), modification of monitoring designs, data analysis approaches, etc., through guidance documents and/or website support .
- Fill gaps in knowledge and available modeling tools and outputs between regional climate, hydrologic, and ecological models.
- Develop tools to make climate data available to other models (e.g., CADDIS).

The OW can

- Include language in U.S. EPA grants, policies, etc. on climate change as a stressor for monitoring and assessment programs, to establish climate change as an important program focus.

- Provide assistance to state bioassessment and resource management programs to integrate the concept of climate change as a significant issue that should be accounted for in assessing the condition of aquatic resources.
- Evaluate Water Quality Standards to be protective in the face of a changing condition paradigm.
- Provide funding support for state/tribal water quality programs to assist in adaptations to existing programs.
- Provide support for identification and sampling of reference sites, re-sampling of reference sites, and more intensive characterization of reference and sentinel sites.

Together, the ORD and the OW can

- Conduct additional workshops to begin the process of evaluation and development of recommendations for other aquatic ecosystems (e.g., large rivers, lakes, wetlands, coral reefs, estuaries).
- Develop a nationwide database of state biological monitoring and assessment data to support evaluation of national/ecoregional climate change trends and effects.
- Transfer technology for use of equipment, such as in situ temperature monitors, that could be used to extend and enhance the value of monitoring data collected by state programs with limited resources, including incorporation of processes and guidance.
- Provide technical support for data management tools (e.g., R code) to manage temperature logger data and reduce it to useable metrics.
- Form partnerships across the U.S. EPA and other federal agencies on a comprehensive climate change strategy to address mandates of CWA.
- Provide a summary of this meeting to U.S. EPA top management for information and support for making informed decision-making.

States and tribes attending the Workshop also discussed resource limitations that impact their ability to implement new and/or additional efforts related to revising and adapting their existing programs to account for climate change. These limitations could be addressed through

- funding support
- personnel support
- priority setting for management actions
- assistance in developing and supporting a structure for sharing resources among agencies to expand capacity

6.2. RECOMMENDATIONS FOR STATES AND TRIBES

Even with constraints of limited resources, state and tribes participating in the Workshop identified several potential program adaptations they considered feasible with resource and technical assistance from the U.S. EPA. These actions include the following:

- Conduct regular and repeat reference site sampling.
- Consider strategies for maintenance and protection of reference sites and areas, including identification of water bodies in the best condition.
- Evaluate the need to shift the sampling index period and/or expand sampling seasons.
- Establish sentinel sites for trend monitoring.
- Improve hydrological and temperature data collection.
- Retrieve historical data records to establish a basis for evaluating climate change.
- Incorporate traditional ecological knowledge, citizen monitoring, and phenological knowledge in assessment of biomonitoring data.
- Continue the refinement of biocriteria programs to incorporate the Tiered Aquatic Life Use (TALU) strategy.
- Accept moving target paradigm versus steady state model and adapt accordingly.
- Perform critical elements reviews of individual programs to identify relevant refinements.
- Engage in collaborative data and resource sharing to maximize limited resources.

7. POTENTIAL NEXT STEPS

Several components of work, with somewhat different time frames, should be contemplated to expand understanding of climate change effects on bioassessment programs and develop a toolbox of appropriate responses.

1. Plan and conduct a national workshop for state and tribal WQ agency managers and biologists on the next most appropriate ecosystem. There would be benefits to developing and conducting a workshop for any of the remaining ecosystems of interest with regard to bioassessment and biocriteria programs (large rivers, lakes, freshwater wetlands, coastal wetlands, and estuaries). There are compelling reasons to consider lakes and freshwater wetlands, including:
    - Lakes have the next most well developed bioassessment and biocriteria programs, and, together with flowing systems, they have the most TMDL issues with biologically impaired waters; thus, integrating aspects of climate change with the TMDL process is important here.
    - Lakes are the next system of focus in the U.S. EPA’s national assessment of ecological condition of the Nation’s aquatic resources.
    - There is substantial overlap in state/tribal bioassessment/biocriteria scientists and/or managers dealing with streams/rivers and lakes (often the same individuals), providing an immediate opportunity to involve states and tribes that were unable to participate in the first stream/river-oriented workshop, while still expanding outreach to another ecosystem.
    - Climate change effects in lakes, and freshwater wetlands will probably be similar to effects in streams/rivers, but also would have different ecological responses with different levels of importance (e.g., wetlands may be particularly susceptible to droughts). Thus, this effort would build upon the experience and knowledge developed in the Workshop and expand both the knowledge base as well as consideration of ecosystem responses.
    - Combining lakes and freshwater wetlands offers some efficiencies in summarizing and discussing current information on climate change projections and evidences that are most pertinent to inland freshwater non-flowing ecosystems, despite ecological differences that certainly will have differences in ecological processes and in responses that are most important.
    - Inclusion of freshwater wetlands with lakes provides an opportunity to consider a system much less advanced in the process of bioassessment/biocriteria program development (wetlands) in an earlier time frame.
2. Implement a more in-depth assessment of climate change effects on stream and river bioassessment programs in a detailed pilot study that would include selected states. It is recommended that states in different parts of the U.S. be targeted to serve as regionally distinct pilot studies. Ideally, as many as four states distributed regionally should be

included, and, at minimum, two states should be included to account both for regional (ecological) variations and for differences between bioassessment programs to at least some extent. Some important considerations for including states in this pilot study are

- Regional distribution, preferably representing a spectrum of very different geographical and ecological areas.
- Continuity and temporal duration of data sets available (ideally at least 20 years) with comparable collection and analytical methods that would support rigorous long-term analyses.
- Willingness of state personnel to be involved and interactive throughout the analysis process—to maximize effective consideration of state- and location-specific issues.
- Information from multiple basins or watersheds is typically needed to characterize the breadth of variation in stressors and responses. An analysis approach should be developed that includes several major aspects.
- Evaluation of all the specific metrics and the composite indices used in the state.
- Consideration and incorporation of the ecological traits of the species included in the state database (classification by ecological traits and sensitivities may already exist for the state databases likely to be utilized, especially if they completed development of biological indices).
- Use of the long-term data sets to investigate and document existing evidence of climate change.
- Compilation of thermal tolerance information for fish and invertebrates as a resource to support predictions of probable climate change effects.
- Evaluation of the sensitivity of component metrics and biological indices to climate change effects, possibly including recalibration of indices (and/or the index development process) to identify components that may be more robust. Analyze how biological indices can be modified to detect or exclude climate change effects; investigate how taxa loss, replacement, and other predicted responses will affect multimetric and other biological indices.
- Evaluation of index sampling periods, including the possible need to shift or expand recommended sampling periods to better account for climate change effects.
- Incorporation of the BCG and TALU into the analysis framework, to evaluate, for example, how climate change degrades reference sites over time between tiers above the CWA “fishable/swimmable” threshold, how this progressively impacts detection of impairment and identification of stressors, and how reference locations can be classified and protected.
- Use of ecological, habitat, and climatological data to characterize climate changes and resulting changes in biological structure and function, especially in reference sites or other benchmark for assessment of condition. May introduce targeted

species/communities changes to the data to mimic climate change responses for "future" analyses, based on documented projections for local/regional climate effects and knowledge of species traits and sensitivities. Relate findings to state WQ standards and designated uses as an example of confounding factors for assessing and determining impairment.

- If scope of effort allows, evaluation of some novel indicators/metrics identified in the framework based on extant research reported in the literature. Consideration should be given to the feasibility of long-term, spatially distributed measurements that could be made within the framework of a monitoring program; and to robustness and interpretability of results with regard to climate change effects and other stressors.

- Beyond the physical and chemical habitat data and biological data typically collected in bioassessment programs, it will be very important to have comprehensive climatological data corresponding to the regions being analyzed. Projection of precipitation data to all sampling locations may be important. More specifically, it may be important to be able to develop site-specific hydrologic projections. In the preliminary case studies, the PHDI was used to project possible effects of dry years and wet years to establish a proxy for projected climate effects of increased summer droughts and increased precipitation. An alternative would be to develop site-specific hydrological estimates to correspond to sampled biological data. The calibrated FTSE model can be used to estimate high and low flow conditions for a specific site and a specific time period, to estimate hydrologic conditions associated with a given sampling event. Such hydrologic projections produced by the model could be informative in estimating the effects of dry periods, or of numerous storm events, and in projecting future climate changes.

- Having sufficiently detailed climate change projections for the states that will be evaluated also is of great importance. It is clear from the Workshop just conducted that detailed regional downscaling from GCMs are possible, and that the technical approaches for developing these are improving. It was also clear that such regionally specific modeling is not accomplished for all areas. An effort will be needed to determine the nature of modeling results available for each state/region considered in the pilot study, and to interact with the appropriate climate modeling scientists to understand the status of these results and obtain needed outputs.

3. Plan a special JNABS issue and special workshop/session at the ASLO/NABS conference in 2010 on the effects of climate change on biological indicators. This would provide a scientific forum to articulate the known science of the effects of climate change on biological indicators. This publication/special session is a follow-on to an earlier ASLO/NABS collaboration held in 1998.

    - Special publication series in the Journal of the North American Benthological Society would bring together international scientists working on the concept of climate change upon aquatic ecosystems, particularly biological indicators.

Ideally, the papers would be published prior to the joint congress of the two societies in June 2010.

- Special session devoted to climate change would be held at the joint congress and would be highlighted as a key theme of the congress. The international scientists in the publication series would be the featured speakers in the session at the ASLO/NABS conference to be held in Santa Fe, New Mexico in June 2010.

4. Work across U.S. EPA and state programs to develop a national database compiling all available state/tribal bioassessment data to support regional and national-scale evaluation of climate change status and trends. To consider this strategy, it is suggested that development of a national database compiling all available state bioassessment data be considered to support regional and national-scale evaluation of climate change status and trends. At least two frameworks exist, which should be considered for adaptation to this purpose.

    - Oracle-based Ecological Data Application System (EDAS), an extension and improvement over Access-based EDAS that is already used by many states. This is a purpose-tailored database for bioassessment data, which accommodates physical, chemical and habitat data, and biological data for multiple assemblages including detailed taxonomic review and manipulation. In addition (and importantly), it includes built-in analyses that support all the steps in bioassessment, metric evaluation and index calculations and development.

    - WQX, the replacement for STORET, is being designed to accommodate existing state bioassessment data, but is not quite ready to house the volumes of state ecological data. The existing accessibility to all states is an advantage of this option. A disadvantage is the lack of associated bioassessment-specific analytical capability. This lack could be addressed relatively easily by developing an analysis front-end (from existing resources to a great extent). The handling of taxonomic data in WQX is potentially another disadvantage that may be more difficult to address.

    - The effort to establish a national data base with acceptable quality control, comparable data (considering taxonomy, reporting units, collection and analytical methods, sampling index periods, and many other factors) would be substantial. Analyses would be relatively simple once this was accomplished. It may be (and perhaps is likely) that not all state data would be adequate for inclusion, and certainly there will be large differences in spatial coverage, and especially in chronological longevity of the data sets.

8. CONCLUSIONS

The review of the literature on climate change effects on aquatic ecosystems shows that it is likely that changes are already occurring (see Section 1). Although current sampling schemes used by bioassessment programs are not explicitly designed to detect climate change effects, it is possible to use the data for this purpose. The case studies presented in this report demonstrate this capability. While the first case study focuses on the length of time it would take to detect a specific effect due to climate change under a variety of scenarios, it is important to remember that the aquatic systems being surveyed are probably already somewhere on the trajectory toward a detectable effect. Recent climate change reports underscore this point that systems are not at time zero with respect to effects (IPCC, 2007).

Existing and ongoing climate change effects have impacts within bioassessment programs that affect how benchmarks are set and how expectations for acceptable conditions are anchored. Monitored reference conditions now reflect temporally changing conditions, and the results from case study two underscore the importance of monitoring reference sites. Characterizing climate change as an additional but global stressor must be accounted for within monitoring designs, analytical approaches, and assessment frameworks. Ultimately, efficacy of the current programmatic approach to definition of acceptable and/or desirable conditions and assessment of the need for regulatory intervention in the management of water resources requires an understanding of all significant influences on the systems being assessed and regulated. It is critically important to be able to distinguish between multiple stressors, and this is done through the acquisition of high-quality bioassessment and other ecological data. This, in part, guarantees the integrity of regulatory decisions through appropriate program adaptations.

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APPENDIX A

REGIONAL PATTERNS OF CLIMATE CHANGE PROJECTIONS AND CONSEQUENCES FOR RIVERS AND STREAMS

A.1. CLIMATE CHANGE AND FUTURE PROJECTIONS

The rate of global warming has increased over the last century; the linear average over the last 50 years (0.13°C per decade) has almost doubled the linear rate over the last hundred years (0.074 °C per decade) (IPCC, 2007). The current rate is estimated at about 0.2°C per decade (IPCC, 2007; Rahmstorf et al., 2007; Hansen et al., 2006) and may increase in the future. There also have been widespread changes in precipitation. Frequency of heavy precipitation events is also predicted to increase over most areas, as is the frequency of droughts (IPCC, 2007).

Climate change will continue and temperatures will potentially increase in the future (IPCC, 2007). General projections for the year 2100 include global average temperature increases of 1.1–2.9°C (from the lowest emissions scenario) to 2.4–6.4°C (from the highest emissions scenario). Increases in precipitation are predicted, with a higher percentage of total precipitation occurring in more frequent and intense storms. Other projections include more precipitation in winter and less precipitation in summer; more winter precipitation as rain instead of snow; earlier snow-melt; earlier ice-off in rivers and lakes; and longer periods of low flow and more frequent droughts in summer (Hayhoe et al., 2007; IPCC, 2007; Barnett et al., 2005; Fisher et al., 1997). Changes in temperature and precipitation will have regional differences that will be important for assessing ecological effects.

A.1.1. REGIONAL PATTERNS

Several points must be considered to understand how the existing and future climate changes are likely to affect aquatic ecosystems, specifically streams and rivers. Ecosystems do not respond to global averages but to regional and local patterns (Walther et al., 2002). Regional patterns of climate change are affected by factors that include atmospheric circulation patterns, topography, land use, and region-specific feedbacks (Hayhoe et al., 2007), and they are more difficult to project because of these localized factors. Within the U.S., regional projections for future temperature increases are variable among models, but almost all regions project greater temperature increases in winter than summer (NAST, 2001). Table A-1 presents a summary of regional climate change projections from the National Assessment and Synthesis Team (NAST) (2001). Increased frequencies of extreme hot days (and decreases in extreme cold days) are also projected throughout the U.S., with the greatest increase projected for the Southwest (Seager et al., 2007). Other notable increases are projected for high-elevation areas of California, central Utah, central Idaho, and the Appalachian Mountains (Diffenbaugh et al., 2005).

Table A-1. Summary of regional climate projections (NAST, 2001). Averages and/or ranges for Hadley (H) and Canadian (C) model projections to 2100 compared to 1961–90.

		Northeast[a]	Southeast[b]	Midwest[c]	Great Plains[d]	West[e]	Pacific Northwest[f]
Temperature	Average annual	2.6–5°C increase	2.3–5.5°C increase	2–6°C increase (mins increase more than max)	Increase >3°C (both models)	4.5–6°C increase	3°C increase by 2050
	Winter min	4 to 2–7°C			5–7°C (winter average)	4.9–7.1°C (winter average)	4.7–5.9°C (winter average)
	Winter max	1.6–3 to 3–6.6°C					
	Summer min	3°C			4–5°C (summer average)	4.3–4.6°C (summer average)	4.1–4.6°C (summer average)
	Summer max	1.3 to 4–6°C					
Precipitation	Average annual	Small increase to 5–10% decrease (C); up to 25% increase (H)	10% decrease (C) to 20% increase (H)	Increase 20–40% by 2100 in upper Midwest, decrease up to 20% Ohio Valley (C); average precipitation increases everywhere, up to 20–40% (H)	13% increase (both models)	Double winter precipitation in CA, decrease in parts of Rockies (both models); summer precipitation no change (C) to decrease (H).	Increase of a few to 20% (average 10%) (H); increase 0–50% (average 30%) (C)
	Extreme events	Small increase in count and strength of storms (H); decrease in count for most of region except mid–Atlantic (C)		Increase in mean associated with increase in frequency and intensity of heavy events	Likely increase, esp. in southern GPs	More extreme wet and dry years (both models)	Winters wetter and warmer, increase in precipitation in heavy storms

Table A-1. continued.

		Northeast[a]	Southeast[b]	Midwest[c]	Great Plains[d]	West[e]	Pacific Northwest[f]
Droughts		Less (H) to more (C) drought	Slightly drier (C) to more precipitation in long term (H)	Small increase in soil moisture (H); small decrease (C)	Decrease in soil moisture in large part of region		
Runoff	Magnitude	More high flow events in winter (+80% in northern areas under higher emissions). Lowest weekly flow projected to decrease ~10% by 2100. For a 2.5°C increase in average temperature and 17.5% (17.5 cm) increase in precipitation, flow at the mouth of the Susquehanna will increase 24% $\pm$ 13% (11.8 $\pm$ 6.7 cm) (Najjar et al., 2000). Neff et al. (2000) projects decrease of 4% to increase of 24%. Moore et al. (1997) estimates decrease of 21–31% by 2100		Overall decrease in river levels (i.e., decrease in runoff?)	Great Lakes area runoff decrease up to 32% (Magnuson et al., 1999)	Summer runoff less by 20% of annual total (Knowles and Cayan, 2002)	
	Timing	Peak (spring) flows advancing 10 to >14 days by 2100. Low summer flows extended almost 1-month under higher emissions					

There are substantial regional and model differences in projections for precipitation changes. The biggest average increases are projected for the Pacific Northwest and Midwest (10–30% by 2100) and the Northeast (up to 25% by 2100), with smaller increases in the Great Plains (13% by 2100), and variable projections for the Southeast (10% decrease to 20% increase by 2100) and the West (e.g., doubling of winter precipitation over California, but decreased precipitation over some parts of the Rockies) (NAST, 2001). Most of the increase in precipitation is projected to occur during the winter, with more frequent and/or more intense storm events, and with more winter precipitation as rain instead of snow (e.g., Polsky et al., 2000; Magnuson et al., 1997).

Most models project increases in evapotranspiration—due to increased temperature rather than increased summer precipitation—leading to a net decrease in soil moisture and a greater likelihood of late-summer drought (NAST, 2001).

A.1.2. HYDROLOGY

There are also secondary drivers that are important in structuring aquatic ecosystems that will be altered by climatic changes, especially hydrologic regimes (e.g., Poff and Ward, 1989; Richter et al., 1996). Projected hydrologic changes, driven by climate-associated changes in temperature and precipitation, include changes in the magnitude, timing, frequency, and duration of various flow events. Together these projected hydrologic changes will result in redistributing stream flow (Hayhoe et al., 2007).

In North America, projected changes in average stream flow range from an increase of 10–40% (at high latitudes) to a decrease of about 10–30% (in mid-latitude western North America) by 2050 (Milly et al., 2005). Consistent with this large-scale pattern, Hayhoe et al. (2007) predict an increase in stream flow in the northeastern U.S., ranging from and increase of 9–18% in the southwest part of the region to an increase of 11–27% in the northeast part of the region. In an earlier study, in the U.S. Northeast, however, average stream flow was projected to decrease by an average of 21–31%, reflecting a temperature-associated increase in evapotranspiration that will exceed small net changes in precipitation (increases in winter and spring and decreases in summer and fall) (Moore et al., 1997). Hayhoe et al. (2007) attributes these differences to their use of updated forcing scenarios that include projected increases in winter precipitation. Patterns of stream flow also are projected to change in the Northeast, with increases in stream flow occurring mainly in the winter and spring, but with lower stream flow in the summer and fall (Hayhoe et al., 2007). Climate changes in the Northeast may also include more intense thunderstorms, especially in the summer. If this happens, it could result in greater variability and “flashiness” of stream flows (Moore et al., 1997). In the Great Lakes Basin, average basin runoff is projected mainly to decrease up to 32% in response to increased temperatures, despite precipitation increases (Magnuson et al., 1997).

In snow-pack dominated regions, the combination of warming temperatures, a shift toward less winter precipitation as snow, and snow-melt occurring earlier will transition the peak runoff from spring to late-winter/early spring (Barnett et al., 2005). Predicted shifts in peak runoff (anywhere from about two weeks to one month earlier) by the end of the century are anticipated (Dettinger et al., 2004; Hayhoe et al., 2007).

Rain-dominated streams are expected to be especially responsive to altered precipitation patterns, with runoff, flow variability, and flood frequency responding directly to changes in precipitation. These streams may also respond to increased variability in precipitation, impacting flood frequency and variation in flow; and to increasing temperatures, causing decreases in runoff (Poff at al., 1996). In a Mid-Atlantic perennial flow (rain-dominated) stream, with a 25% modeled increases in precipitation, Poff et al. (1996) predicts mean flow (runoff) to increase about 10–15%, flow variability to increase about 20–25%, and flood frequency to more than double (from 1.1 to 2.5 floods per year). Doubling the coefficient of variation of precipitation had an even more dramatic effect on these hydrologic characteristics.

In the Mid-Atlantic region, using the Susquehanna River as a model, Neff et al. (2000) estimated annual changes in stream flow by 2100. Estimated changes range from –4%, based on the Canadian Climate Center (CCC) model (Flato et al., 2000; McFarlane et al., 1992) to +24%, based on the Hadley model (Johns et al., 1997). Both models project increased stream flows during the winter, with peak stream flow occurring up to 1 month earlier.

Even with net projected increases in annual precipitation for many regions of the U.S., increased durations of low flows and increased frequency of summer droughts are also projected. In California's Sacramento/San Joaquin Basin, Knowles and Cayan (2002) predict a 20% loss in the amount of annual stream flow occurring during the summer (April–July) by 2100. Dettinger et al. (2004) made a similar prediction for the Merced, American, and Carson Rivers in California, including a decrease in summertime low flows and reduced soil moisture.

Finally, the increased temperatures will lengthen the growing season, as well as increase evapotranspiration during the warm months. These could have the net effect of reducing groundwater recharge of streams, which would increase the severity of summer dry periods independent of rainfall. This could, however, be mediated to some extent by the increased CO_2 concentrations, which reduce evapotranspiration (e.g., Gedney et al., 2006).

A.1.3. WATER TEMPERATURE

In addition to hydrologic alterations, changes in stream temperature are also expected. The IPCC report projects a continued increase in global air temperature at approximately 0.2ºC each decade (IPCC, 2007). Although these are global averages, regional models also support these projections. For example, Mid-Atlantic regional models project that average air temperature will increase 2.6–5.0ºC by 2100 (Polsky et al., 2000; Barron, 2001). These temperature changes will increase the maximum, average, and minimum stream temperatures as well as the number of degree days and the rate of degree day accrual (Note: degree days are the cumulative sum of average daily temperatures above a baseline. For example, if the baseline is 10ºC, then one day with an average temperature of 12ºC contributes 2ºC days).

Though a relationship between increasing air and water temperatures is expected, the magnitude and seasonal patterns of changes in stream and river water temperatures are likely to vary regionally. These regional differences will be due to water source influences (surface versus ground water), watershed characteristics, and season. Stephan and Preudhomme (1993) estimated weekly average water temperatures in ºC (excluding the ice cover period) to be a factor of 0.86 times the weekly average air temperatures for 11 streams in the Mississippi River Basin. Eaton and Scheller (1996) used this same linear relationship between air temperature and stream water temperature to estimate probable loss of fish habitat due to global warming projections. However, Mohseni et al. (2003) claim that the relationship between air and water temperatures is better explained by an S-curve such that at higher air temperatures, stream temperature increases level off due to evaporative cooling. In the Upper Rhone River, Daufresne et al. (2003) showed a clear, though non-linear, correspondence between long-term increases in annual average air temperature (increase of about 1.0ºC, 1979–1999) and average annual water temperature (increase of about 0.6ºC, 1979–1999). Annual patterns were similar, but average annual water temperature did not correspond with average air temperature perfectly, suggesting possible influences of other factors such as annual variations in flow conditions or snow melt. In a review of the thermal regime of rivers, Caissie (2006) showed that thermal regime is strongly influenced by meteorology, river conditions, and geographic setting.

A.1.4. HABITAT

Stream hydrologic patterns control habitat stability, channel formation and maintenance (Poff et al., 1996), and they define composition, structure, and functioning of aquatic assemblages (Richter et al., 1996, Poff and Allan, 1995). Changes in hydrologic pattern, especially flood frequency, episodic runoff events frequency and intensity (including "flashiness"), peak runoff magnitude, and flow total, will alter stream habitat and its dynamics. Flow dynamics not only influence sediment supply and transport and, therefore, channel form, but water volume also influences the amount of available habitat and water quality. Seasonal patterns of flow magnitude, duration and frequency of runoff events, and other parameters strongly influence the types of species that can inhabit an area (Poff et al., 2002). As a result, over time, regional changes in hydrologic regime are expected to modify habitat, species composition, and ecological interactions.

In the Southwest, Grimm et al. (1997) concluded that neither hydrologic nor climate models were sufficiently developed at that time to predict magnitude and direction of future climate changes. Instead, they identified stream and river surface flows in the arid Southwest as being particularly vulnerable to even small changes in precipitation, with even modest decreases in precipitation potentially causing large decreases in stream flow. In addition, increased temperatures could increase the likelihood of winter/early spring precipitation as rain instead of snow and could increase the likelihood of severe episodic flooding, while increased temperatures may also be associated with increased drought conditions during the summer. Associated habitat changes could include changes in riparian vegetation, shifts from perennial to intermittent flow, loss of aquatic habitat, and alterations in nutrient retention and instream production (Grimm et al., 1997). More recent evaluation of the numerous climate model outputs available provide consistent projections for existing and future decreases in annual precipitation minus evaporation (increased aridity), especially during the winter (Seager et al., 2007), which will have substantial negative impacts on availability and condition of stream and river aquatic habitat.

A.1.5. POLLUTANT BEHAVIOR

Stream water quality is expected to respond to changes in runoff magnitude and timing. Reduced flow in summer combined with increased temperatures will likely decrease dissolved oxygen (DO) concentrations, while increased storm frequency and magnitude are likely to increase introduction of silt and pollutants (Poff et al., 2002). In the Mid-Atlantic region, increased stream flow in the winter and the spring is expected to degrade water quality due to increased inputs of nutrients, sediments, and toxicants (Neff et al., 2000; Rogers and McCarty, 2000). For example, nitrate loads have a high, positive correlation with stream flow in this area, $r^2 = 0.8$, (Neff et al., 2000). However, nitrate loads are projected to decrease in July and August associated with projected decreases in stream flow, which could, in part, ameliorate low (DO) conditions. In boreal streams in northwestern Ontario, Schindler et al. (1996) reported substantially reduced runoff during warm, dry periods in the 1970s and ‘80s that coincided in magnitude with similar increases in temperature and decreases in precipitation that are projected for future climate change in that region. There was also a substantial reduction in stream export of phosphorus in association with these periods, though not in nitrogen export. In the Northeast, overall drier conditions and reduced stream flow are expected to increase watershed retention of non-point source nutrients, to reduce nutrient runoff, and to reduce erosion; increased thunderstorm intensity could increase episodic erosion and nutrient loading (Moore et al., 1997). These region-specific examples help define general expectations (i.e., for all regions) for pollutant loading and other water quality (e.g., DO) changes that should be expected in association with climate-driven changes in temperature and runoff.

APPENDIX B

DESCRIPTIVE AND SUPPLEMENTAL RESULTS FOR CASE STUDY 2: BIOLOGICAL ASSESSMENT IN THE PRESENCE OF CLIMATE CHANGE

B.1. RESULTS OF CORRELATION ANALYSES

The fish and benthic invertebrate response variables that showed the strongest responses in these correlations were the 2005 version of Maryland's fish index of biotic integrity (IBI), fish taxa richness, total number of fish, the 2005 version of Maryland's B-IBI, total benthic taxa richness, and total EPT taxa richness (Southerland et al., 2005). A series of these stressor response relationships are shown below, and are used for further exploration of stressor-response models.

Relationships between flashiness and a variety of environmental variables were investigated to identify variables that are potential stressors related to flashiness. Flashiness is a hydrologic characteristic that can be reflective of stream and surrounding watershed alterations, and might also be responsive to climate change. Figure B-1 shows the relationships between Baker's flashiness index scores (Baker et al., 2004) and eight other environmental parameters tested (dissolved oxygen, phosphate concentration, physical habitat index, conductivity, embeddedness, percent urban, impervious surface, and total phosphorus). In this and subsequent figures, the solid line is the locally weighted scatter plot smoothing line (LOWESS). LOWESS smoothing was done in Systat10 by running along the x values and finding predicted values from a weighted average of nearby y values. The surface is allowed to flex locally to better fit the data. For the LOWESS, the degree to which the line or surface (tension) is allowed to flex locally to fit the data to 0.5 was specified, meaning that half the points are included in the running window. The strongest relationships were between percent urban land use and Baker's flashiness index score, and between impervious surface and flashiness. Both of these are factors contribute to alterations in watershed runoff that result in greater "flashiness." Some other variables that are closely associated with runoff, such as nutrient concentrations, had relatively weak relationships with flashiness (e.g., total phosphorus and total organic carbon).

Figure B-2 shows the relationships between physical habitat index (PHI) and macroinvertebrate and fish IBI scores in the MBSS data. Overall, as habitat condition improved (PHI increased), the fish and benthic IBIs increased.

Comparisons were made between three fish response variables and a suite of environmental parameters. Figure B-3 shows the relationships between fish IBI scores and dissolved oxygen (DO), instream habitat, temperature, channel flow, flashiness, and impervious surface. Fish IBI scores increased with increasing DO, habitat score, and channel flow. Fish IBI declined with increased impervious surface and flashiness. The lack of relationship between fish IBI and temperature is at least, in part, related to the factors that collections are made only during a seasonal (summer/fall) index period, and that these analyses are being conducted on a combination of only two very closely related ecoregions.

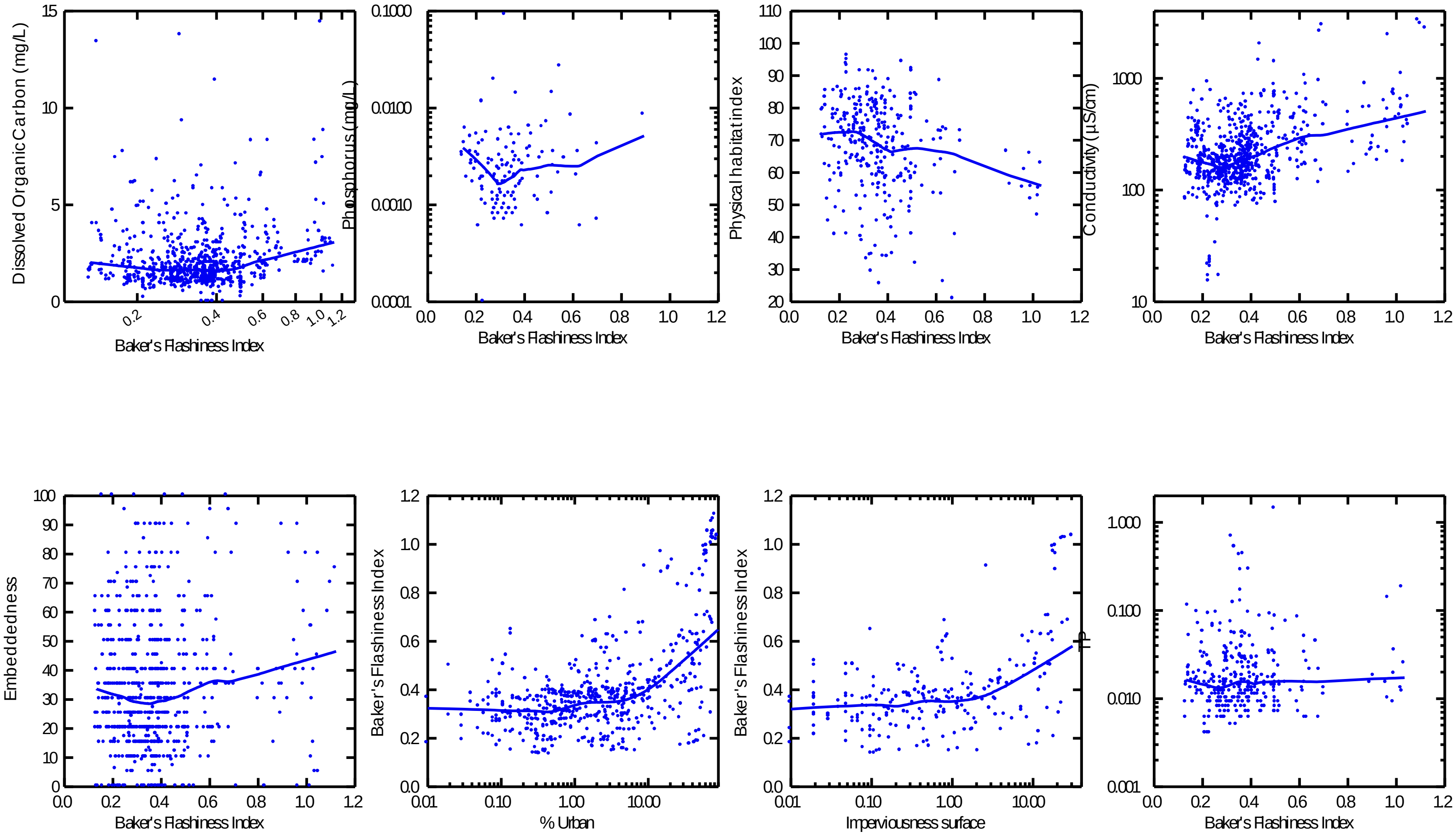

Figure B-1. Flashiness vs. environmental variables; the solid line is the locally weighted scatter plot smoothing line (LOWESS).

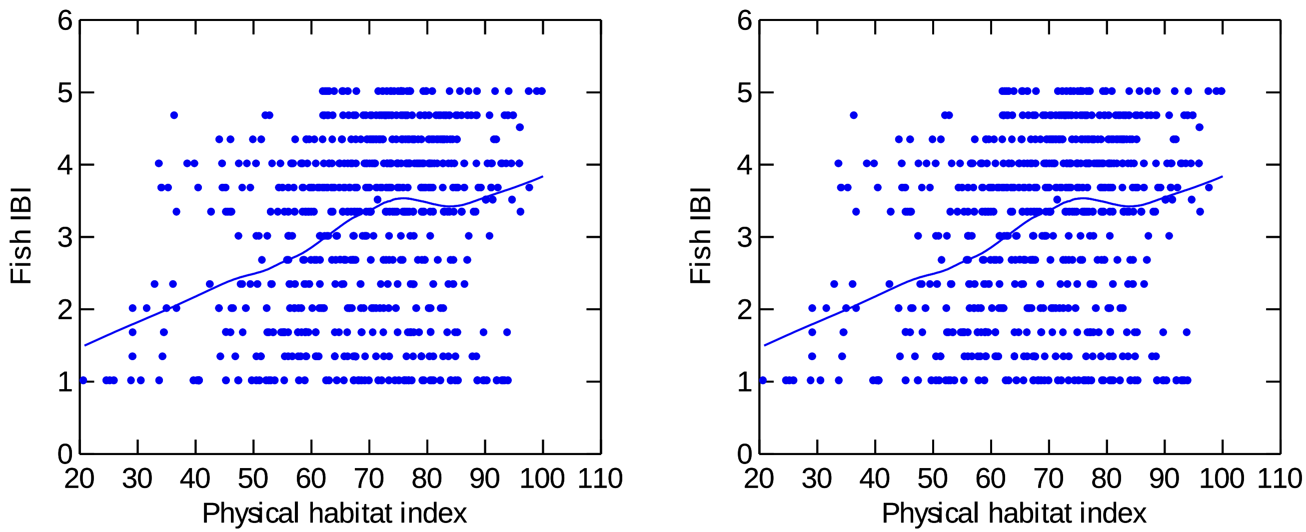

Figure B-2. Index comparison; the solid line is the locally weighted scatter plot smoothing line (LOWESS).

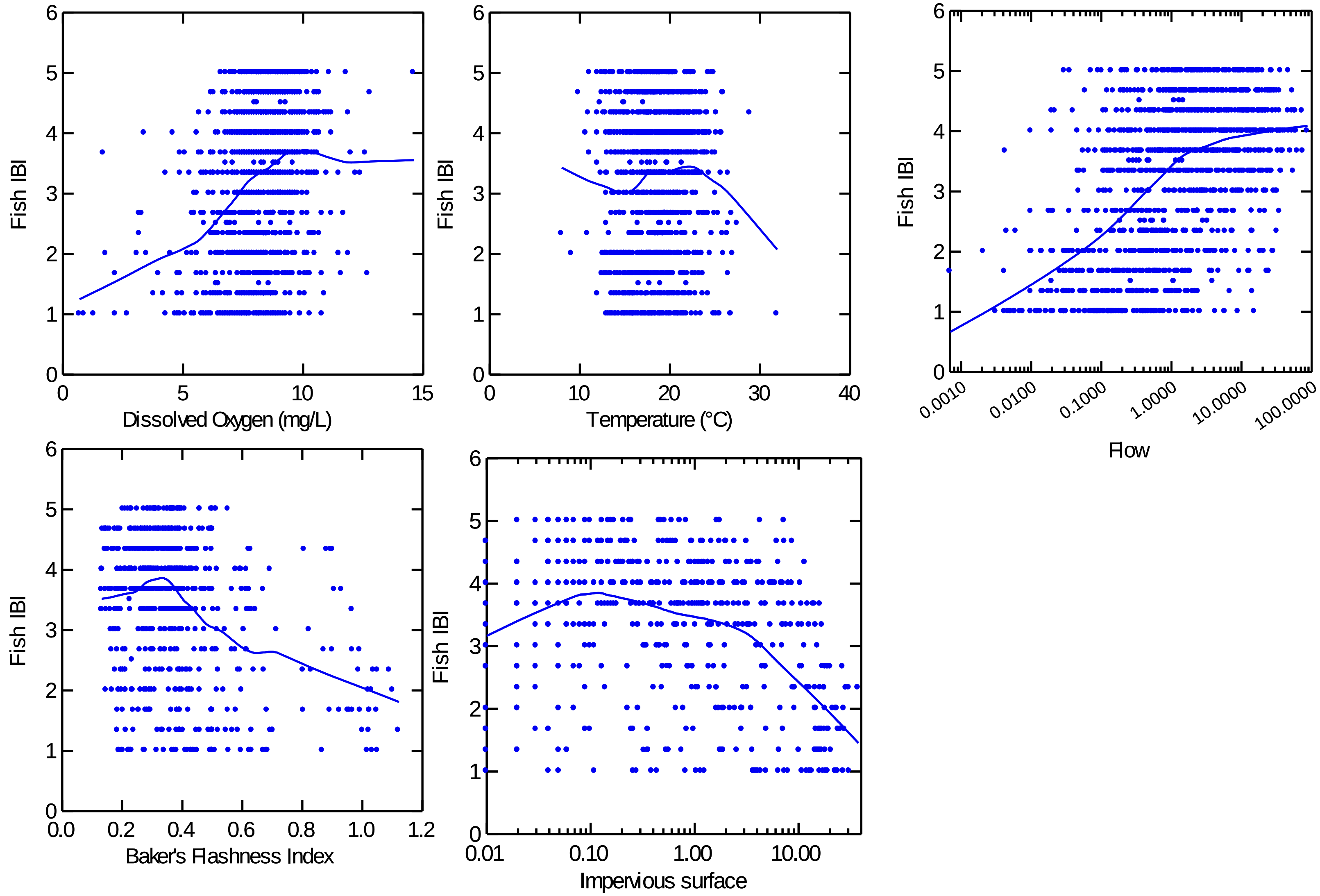

Figure B-3. Fish vs. environment; the solid line is the locally weighted scatter plot smoothing line (LOWESS).

Figure B-4 shows the relationships between abundance of fish and the same suite of environmental variables (DO, habitat score, temperature, channel flow, flashiness, impervious surface). Fish abundance increased with increasing DO, though the relationship is very weak. Fish abundance also increased with increasing habitat score, PHI, and channel flow. Fish abundance declined with increasing impervious surface, though again in a very weak relationship. There was no meaningful relationship between fish abundance and flashiness.

The number of fish species present (richness) also increased with increasing instream habitat score and with flow (Figure B-5). Fish richness increased with increasing DO, but with a weaker relationship. Fish species richness declined with increased impervious surface, but showed no real relationship with flashiness.

For the benthic macroinvertebrate community, species richness increased very slightly with increasing PHI, and decreased slightly with increasing flashiness and impervious surface (Figure B-6). However, it appeared that total benthic taxa richness was not a strong response variable with most parameters. In comparison, EPT taxa richness showed reasonable stressor-response relationship with several environmental parameters (Figure B-7). The number of EPT taxa increased with increasing PHI, and decreased with increasing DOC, phosphorus, conductivity, embeddedness, Baker's flashiness index score, and impervious surface. EPT taxa had strong relationships with all these variables and were, therefore, selected as a primary response variable to examine potential climate change effects.

Table B-1 summarizes the hydrologic parameters of greatest interest in the analysis. Low flow events, high flow events, and Baker's flashiness were estimated by the FTSE model. Low flow and high flow events are the number of events during a year below the 25^{th} and above the 75^{th} percentiles, respectively, of the area-weighted mean discharge of all streams.

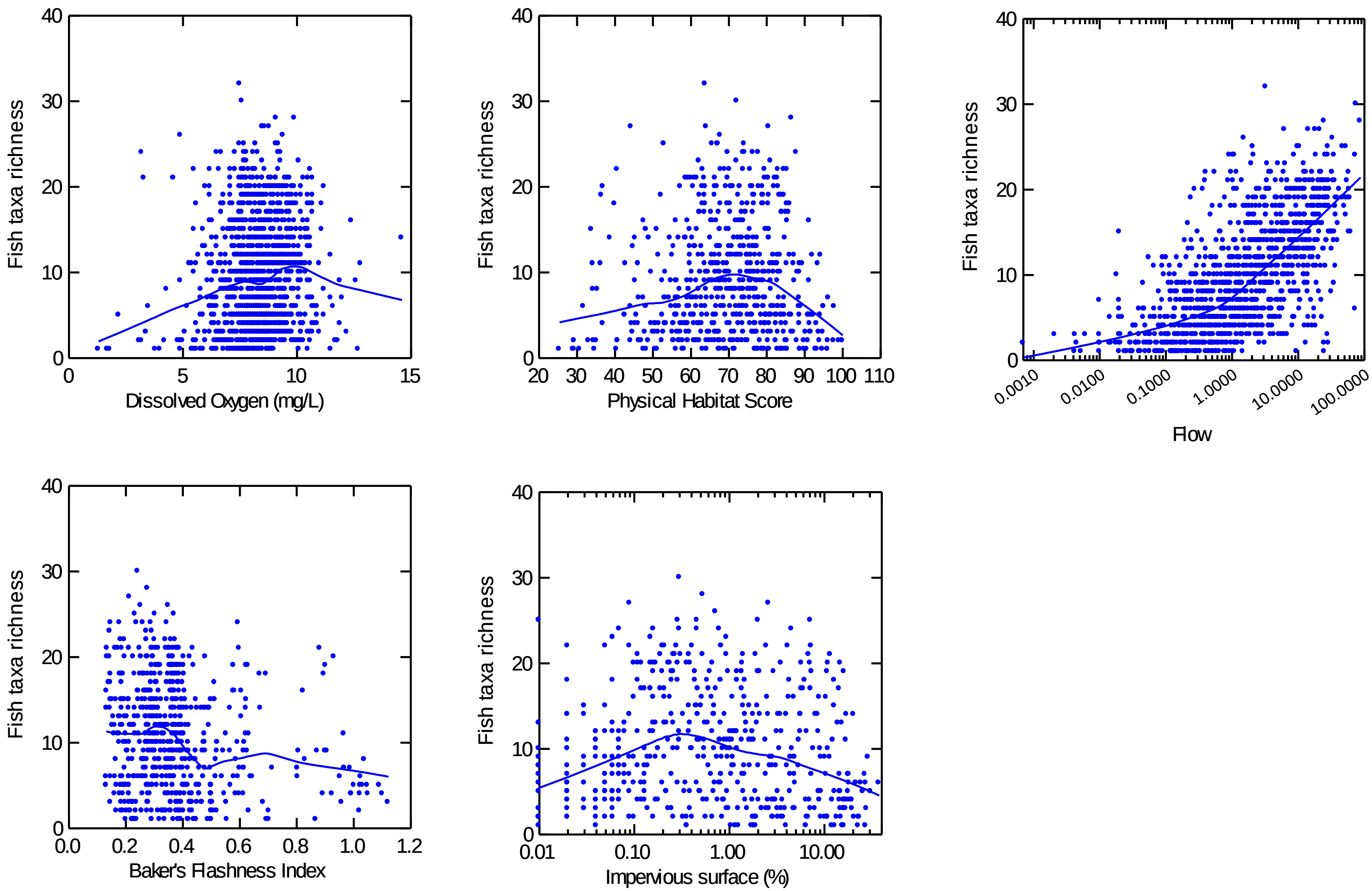

Figure B-4. Fish richness vs. others; the solid line is the locally weighted scatter plot smoothing line (LOWESS).

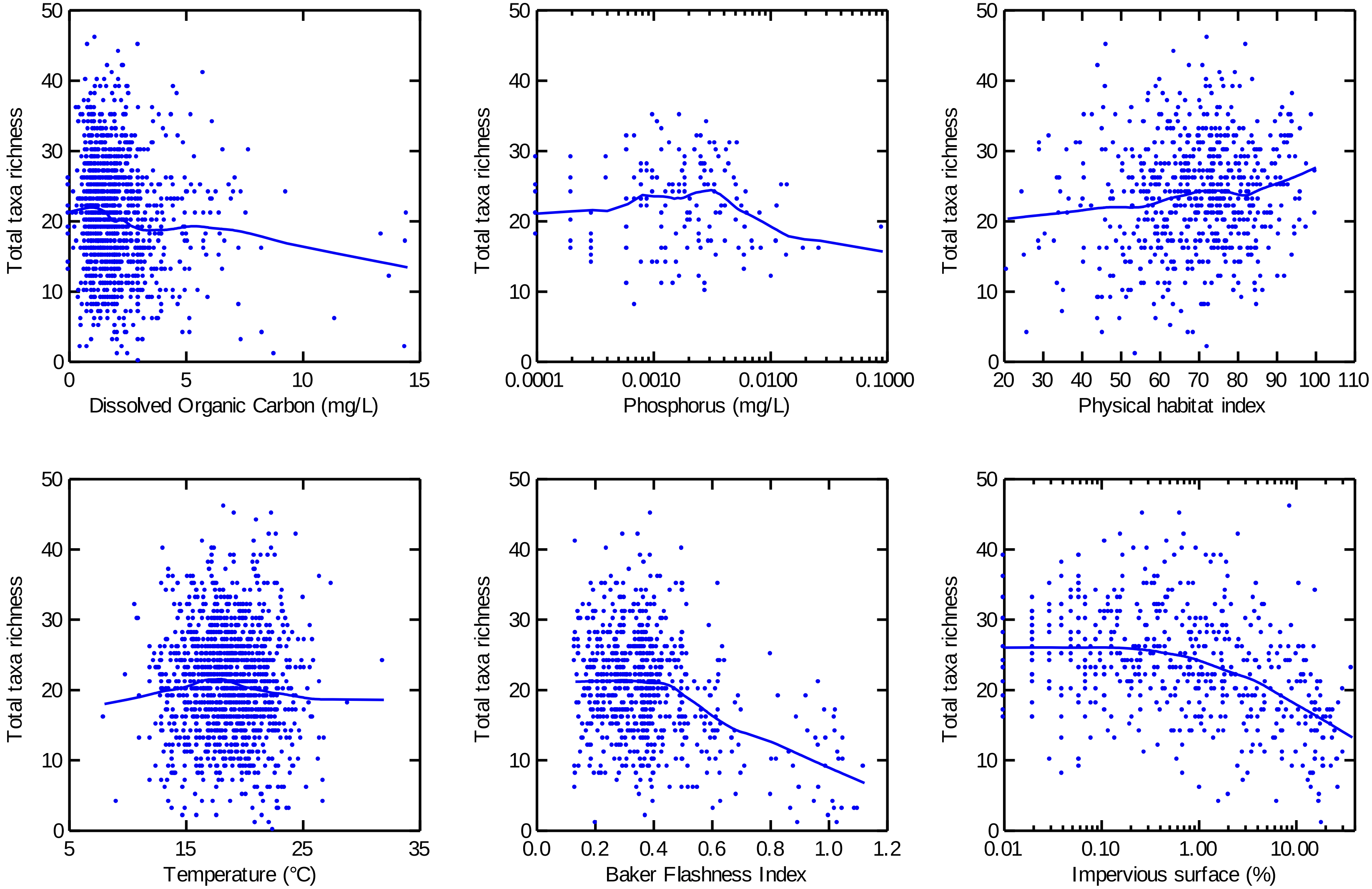

Figure B-5. EPT taxa vs. environmental 2; the solid line is the locally weighted scatter plot smoothing line (LOWESS).

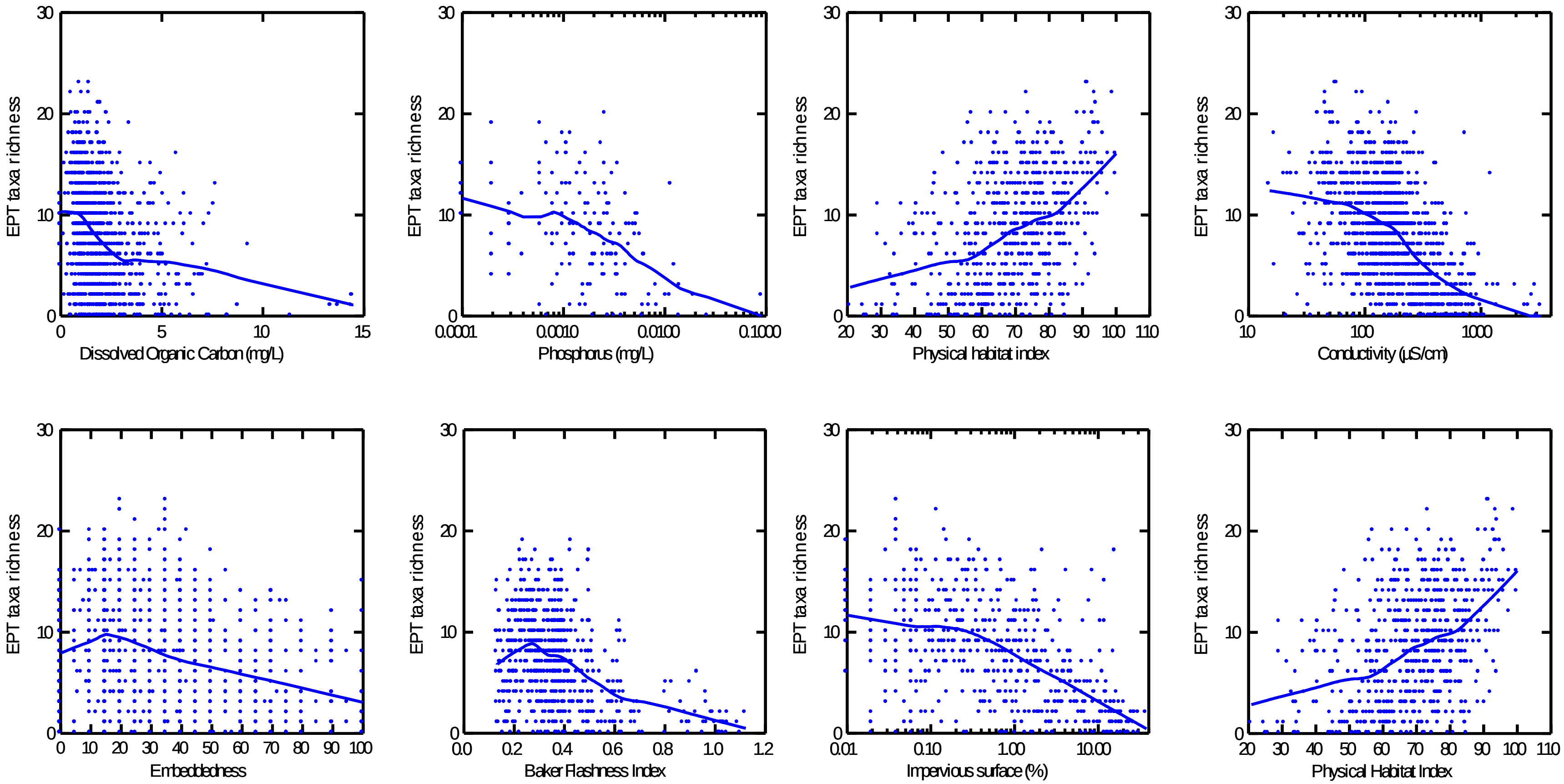

Figure B-6. EPT and environmental variables; the solid line is the locally weighted scatter plot smoothing line (LOWESS).

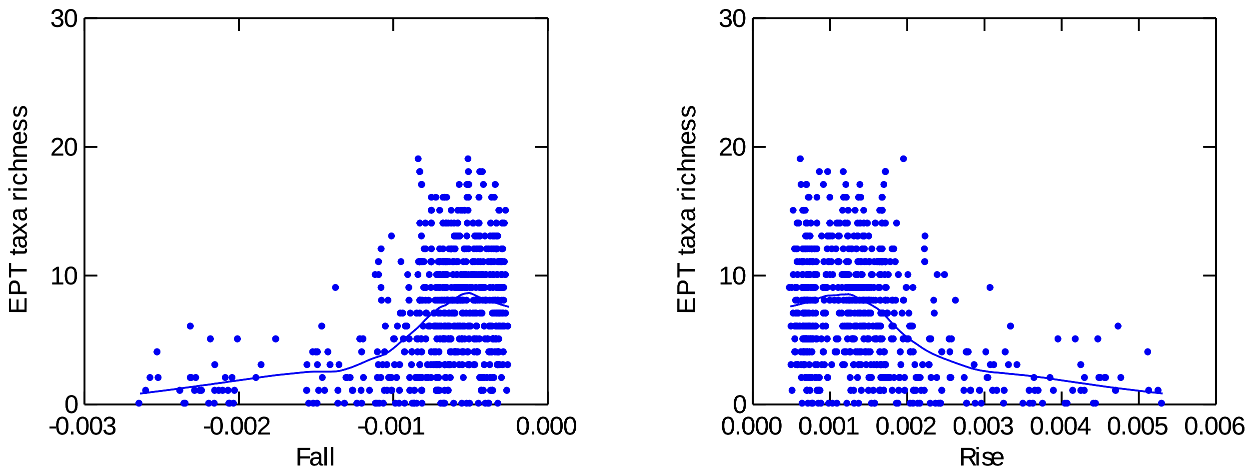

Figure B-7. EPT and hydrology; the solid line is the locally weighted scatter plot smoothing line (LOWESS).

Table B-1. Summary of hydrologic parameters used in analyses

	Baker's Flashiness Index	Palmer Hydrologic Drought Index
Number of cases	764 (streams)	15 (months)
Minimum	0.132	-4.24
Maximum	1.121	4.75
Median	0.362	1.02
Mean	0.385	0.819
Standard Deviation	0.179	2.56

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