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

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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 effects on ecosystems and could be used in assessment programs relying on biological indicators 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 to begin to discuss. The results and recommendations from the preliminary analysis presented in this report are an initial step towards helping biocriteria programs modify assessment activities to account for climate change effects and ensure that management goals continue to be met.

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FOREWORD

(To be included for external peer review)

PREFACE

This report was prepared by Tetra Tech, Inc. and the Global Change Research Program (GRCP) in the National Center for Environmental Assessment (NCEA) of the Office of Research and Development at the U.S. Environmental Protection Agency (EPA). It is intended for managers and scientists working on biological indicators, bioassessment, and biocriteria to provide them 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. The GCRP established a partnership with the Health and Ecological Criteria Division within the EPA's Office of Water, and with State Water Quality Agencies and some Tribal Environmental Agencies to develop a foundation for linking climate change to their monitoring and assessment programs. As a part of the information gathering for this report, EPA convened a workshop with state and tribal biocriteria managers and scientists from EPA and academia. The workshop was held in Baltimore, MD in March 2007 and focused on climate change effects on river and stream ecosystems. The goal of the workshop was to provide state and tribal biocriteria managers with information on how climate change may affect their monitoring and assessment programs for protecting and restoring their water resources. 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 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 also presented to aid in understanding the technical ramifications of adapting existing biocriteria programs. This report presents 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 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. Recommendations from workshop participants and a summary of proposed next steps conclude this report.

AUTHORS AND REVIEWERS

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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 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.

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 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, climate change as a stressor 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, in particular changes in temperature and precipitation patterns that affect stream flow. Biological indicators that are sensitive only or predominantly 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 focuses on sampling power. The power to detect effects depends on the effect size, in this case, 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 dataset 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 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, i.e., 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 a 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 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 US; (3) disseminating information on regional climate change effects on biological indicators; and (4) coordinating information across EPA and state agencies to evaluate trends in bioassessment data.

1. INTRODUCTION

Changes in climate are expected to affect ecosystems, and therefore also their management. Water quality programs managed by state and tribal water resource agencies in response to the Clean Water Act of the US, will need to take these effects into account. Human activities as well as natural factors have already changed the climate, and these trends are likely to continue into the future (Alley et al., 2007). There is now high confidence that anthropogenic emissions of greenhouse gases and aerosols has resulted in warming, with evidence of globally increasing air and ocean temperatures, melting of snow and ice, and rising sea levels (Alley et al., 2007; Rahmstorf et al., 2007). Global air temperatures have increased about 0.6 ^{o}C over the last 30 years and 0.8 ^{o}C over the last century, and global ocean temperatures are probably as warm now as they were during the Holocene maximum (about 5,000 to 9,000 years ago) (Hansen et al., 2006). Observed increases are greater over land masses than over the ocean, and 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 (Alley et al., 2007). The third Intergovernmental Panel on Climate Change (IPCC) report (Houghton et al., 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 (Alley et al., 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 (Section 1), an overview of bioassessment programs, (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) an approach for identifying biological indicators sensitive to climate change (Section 3), (2) an analysis for detecting climate change effects (Section 4), and (3) methods for continuing to detect impairment under climate change (Section 5). Recommendations from workshop participants and a summary of proposed next steps conclude this report (Sections 6 and 7).

1.1. BIOINDICATORS, BIOCRITERIA, AND THE CLEAN WATER ACT

The Clean Water Act (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." (Frey, 1977; Karr and Dudley, 1981; Karr et al., 1986). This wording highlights some key attributes of biological communities that are fundamental to preserving "integrity" (diversity, composition, functional organization), and 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 Clean Water Act. Biological criteria (biocriteria), derived from biological monitoring and assessment, provide narrative and numeric targets defining 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. It is national policy 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).

Biocriteria are developed by biologists and other natural resource scientists 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 all-encompassing 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 (USEPA, 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 waterbody meets, or exceeds, an impairment threshold, and (2) asking whether the biological condition of a waterbody 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). These 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 affects 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 stream 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). Northward migrations may also be limited in regions including the southwest and southern Great Plains 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 US, 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 (Mohseni et al., 2003) due to the warming projected for a doubling of atmospheric CO_2 concentrations. Fishes with the smallest geographic ranges are the most vulnerable. Rahel et al. (1996) estimated habitat losses for cold-water fish species in the Platte River, Wyoming ranging from 7–76% for temperature increases of 1–5 ºC. They anticipated potential population fragmentation as cold-water species were progressively limited to colder headwater stream reaches. 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 ^{o}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 US. In the Southeast, freshwater mussels are especially vulnerable to drought, along with the corresponding low flows and depressed dissolved oxygen levels, and respond with 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 both 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) predicted high risk of species extinctions for subtropical and tropical rivers with a rich endemic fauna and noted the vulnerability of fish species that require seasonal floodplain connection for life cycle completion. Using a similar approach, Xenopoulos and Lodge (2006) estimate potential fish richness losses associated with 20–90% reductions in discharge for several rivers in two regions of the US, and reported 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 US. 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).

1.2.2. Changes in Phenology

Warmer water may increase 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 showed that treatments which received both increased temperatures and nutrients, both nymph abundance and size were increased. For a Japanese species of mayfly (*Ephoron shigae*), cumulative degree days and time of emergence were significantly correlated, explaining 80–90% of the variation in emergence date, depending on whether the analysis was 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 small role in species' responses to climate change through adaptation (Parmesan, 2006; Berteaux et al., 2004; Hogg and Williams, 1996), including 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 response 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 mean phenotypes lag behind optimal phenotypes, and 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 be expected to have food web implications, altering stream productivity and potentially impacting fishes 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 discussed 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 would have any effects on streams and rivers. Effects of nutrient enrichment in streams are highly variable, due to questions about which primary nutrient (N or P) 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; Rosmand 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 would 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. This section examines several candidate indicators based on the current literature. One common theme of these potential indicators is comparisons of responses of organisms within an ecological community.

Golladay et al. (2004) surveyed mussel species during drought conditions and found that many vulnerable species declined in abundance or were extirpated from many of the non-flowing streams in the study, including *Lampsilis straminea claibornensis*, *Villosa villosa*, and *Lampsilis subangulata*. Previous research on mussel mortality due to drought in the lower Flint River Basin found that *Pleurobema pyriforme* and *Mediunidus penicillatus* also showed signs of drought intolerance due to decreased dissolved oxygen concentrations. However, other mussel species may be less affected by drought and subsequent low oxygen levels, including *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.

Water level is often linked to important life cycle stages in wetland organisms, and any changes in the timing and amount of water may influence these stages. Golladay et al. (2004) suggest that wetland invertebrates could be divided into four response guilds to indicate hydrologic status: (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. This approach may be adaptable to river/stream systems.

Monitoring changes in community composition and any shifts from cold- and coolwater dominated systems to warmwater fish systems within an ecoregion may be another good indicator. Assessments of impacts on ecological resources from projected climate changes have led to hypotheses about fish community composition; it is expected that coolwater and warmwater fishes will be able to invade freshwater habitats at higher latitudes, while coldwater fishes will disappear from low latitude limits of their distribution where summer temperatures already reach fish maximum thermal tolerances (Carpenter et al., 1992; Tyedemers and Ward, 2001). However, coldwater fishes may not experience as many winter stresses such as osmoregulation at extremely low temperatures and physical damage from ice (Carpenter et al., 1992; Melack et al., 1997). In addition, their ranges at higher altitudes and latitudes may expand with increased duration of optimal temperatures (Carpenter et al., 1992). It is also important to note that fishes in east-west drainage river systems may not be able to find a thermal refuge (Carpenter et al., 1992).

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 (Ministry of Water BC, 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 (Ministry of Water BC, 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) has shown 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). 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) focus on in their analysis already live at the southern edge of their thermal range. Melack et al. (1997) also review potential impacts of climate change on the spawning of salmon species 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 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, the 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).

Coldwater fish species and salmon species in particular may be good indicators of climate change impacts 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 impacts, however, since 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 impact of increased water temperatures is to reduce dissolved oxygen levels in stream waters. Darter species are sensitive to benthic oxygen depletion because they feed and reproduce in benthic habitats (Barbour et al,. 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, assessment of biological communities can provide information that may not be revealed by measurement of concentrations of chemical pollutants or toxicity tests (Barbour et al., 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 for 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-1980'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 EPA's Rapid Bioassessment Protocols (RBPs) (USEPA, 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 (USEPA, 1999). As a complement to the bioassessment development, procedures for developing narrative biocriteria were published in 1992 (USEPA, 1992), and for developing biocriteria for streams and rivers in 1996 (USEPA, 1996). Following this initial focus on streams and rivers, bioassessment technical guidance was developed for lakes and reservoirs (USEPA, 1998), estuaries and coastal marine waters (USEPA, 2000), and wetlands (USEPA, 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 effect of chemical and non-chemical stressors; therefore, these indicators need to be well 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 (USEPA 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 has a dramatic effect on maintaining stable reference conditions. A gradient of degradation of reference sites over time is plausible and is an important factor in establishing a credible bioassessment, or other monitoring, program. Bioassessment programs throughout the US 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 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 US. Benthic macroinvertebrates are the most common assemblage used for bioassessment in streams and rivers among the states and tribes (USEPA 2002). Fish assemblage is the second most prevalent assemblage used to assess biological condition. EPA recommends the use of multiple assemblages in programs to increase the robustness of the overall bioassessment (USEPA 1996). Periphyton or algae is of interest to many states as an added assemblage for use in their monitoring and assessment program.

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 are listed in Tables 2-1 and 2-2. All of 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 (Barbour et al., 1999).

Category	Metric	Definition	Predicted response to increasing perturbation
Richness measures	Total No. taxa	Measures the overall variety of the macroinvertebrate assemblage	Decrease
	No. EPT taxa	Number of taxa in the insect orders Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies)	Decrease
	No. Ephemeroptera Taxa	Number of mayfly taxa (usually genus or species level)	Decrease
	No. Plecoptera Taxa	Number of stonefly taxa (usually genus of species level)	Decrease
	No. 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	No. of Intolerant Taxa	Taxa 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 (Barbour et al., 1999).

Category	Metric	Definition	Predicted response to increasing perturbation
Richness measures	Total No. of species	Measures the overall variety of the fish assemblage	Decrease
	No. native fish species	Those species of fish that are indigenous	Decrease
	No. salmonid age classes[b]	Measures the life stage representation of particular top predators in coldwater systems	Decrease
	No. of Darter species	Diversity of darters, which are typically in fast flowing waters and cobble substrate	Decrease`
	No. sculpin species	Normally cold-water bottom feeders	Decrease
	No. benthic insectivore species	Those species that depend on aquatic insects for primary food source	Decrease
	No. darter and sculpin species	Combination of clean-water forms, mostly in coldwater systems	Decrease
	No. darter, sculpin, and madtom species	Combination of key taxa that represent important structure of fish assemblage in certain systems.	Decrease
	No. 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
	No. benthic species	Diversity of feeders of all benthic fauna, including insects and non-insects	Decrease
	No. of Sunfish species	Warm-water pelagic species representative of good water quality and habitat	Decrease
	No. cyprinid species	Diversity of minnows that include a range of tolerance	Decrease
	No. water column species	Indicative of good quality pools and migration routes	Decrease
	No. sunfish and trout species	Combination of species representing good water and habitat quality	Decrease
	No. salmonid species	Diversity of salmon in coldwater systems able to accommodate a variety of top carnivores	Decrease
	No. headwater species	Diversity in generally depauperate systems	Decrease

Table 2-2. Continued.

Category	Metric	Definition	Predicted response to increasing perturbation
Richness measures (cont'd)	No. of Sucker species	Diversity of all suckers – round-bodied and other	Decrease
	No. sucker and catfish species	Combination of suckers and catfish in warm-water systems to be indicative of healthy systems.	Decrease
Tolerance/Intolerance measures	No. of Intolerant/sensitive species	Diversity of sensitive fish species; may be stressor dependent	Decrease
	No. amphibian species	Use of amphibians in systems where sensitivity to perturbation is measured by non-fish taxa	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 ind. as brook trout	Compositional dominance of brook trout to other salmonids	Decrease
	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 preferance	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

Table 2-2. Continued.

Category	Metric	Definition	Predicted response to increasing perturbation
Trophic measures (cont'd)	% 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
	% 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

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

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 (1992a), 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

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 ^{o}C for the lowest emissions scenario to 2.4–6.4 ^{o}C for the highest emissions scenario (Alley et al., 2007). Increases in precipitation are predicted, 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 (Alley et al., 2007; Barnett et al., 2007; Hayhoe et al., 2007; 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 are expected to include changes in magnitude of flow ranging from increases of 10-40% in the northeastern US 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., 2001). 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 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; Sweeney and Vannote, 1978; Vannote and Sweeney, 1980).

As discussed in Section 2.2, the 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; widely applied categories of biological indicators were summarized in Section 2.2 (see Tables 2-1 and 2-2). The variability among states and regions in the specific metrics within these categories that are used limits their usefulness. 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 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). Expected climate change responses by category are summarized in Table 3-1 (Note: this summary of potential responses represent 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).

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 measures	Overall richness generally expected to decline due to temperature sensitivity and hydrologic stresses including increased flashiness, increased summer low flows, drought, etc. However, replacements over time with tolerant forms may ameliorate this in some situations.	(e.g.,Durance and Ormerod, 2007)
Composition 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 more temperature and/or flow sensitive taxa (examples potentially include *Athricops*, *Potamopyrgus*, *Lepidostoma*, *Baetis niger*, Tabanidae, *Hydropsyche instabilis*, *Helodes marginata*, *Caenis* spp.), and/or from range shifts.	(Daufresne et al., 2003; Durance and Ormerod, 2007 ; Burgmer et al., 2007 ; Golladay et al., 2004 ; Parmesan, 2006 ; Hawkins et al., 1997)
Tolerance/intolerance measures	Focusing climate change sensitivities related to temperature and flow regime, expect 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; CO2 concentrations, which can directly affect for instance leaf litter composition and decomposition; and changes in riparian vegetation.	(e.g., 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.	
Fish		
Richness measures	May have initial increase in diversity as more diverse warm-water assemblages replace cool- or cold-water forms. Also, habitat availability is expected to 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.	(Xenopoulos and Lodge, 2006; Xenopoulos et al., 2005; Poff et al., 2002; Grimm et al., 1997; Hayhoe et al., 2007)
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.	(e.g., 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.	

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 causes 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 impact the process of attributing cause to particular stressors. It will essentially require the development of an approach for partitioning 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 approach and design in order to account for climate change. Preliminary aspects of this component are discussed in Section 4 (below), to the extent that they were addressed in a preliminary case study. Another main aspect of program adaptation is restructuring of 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 include implicitly associated components such as tracking of changes at reference locations through both sampling design and analyses.

Another component that is being considered for its potential contribution to tracking and differentiating climate change impacts from other stressors is the categorization of biological indicators based on differing sensitivities to climate change effects. These indicators include both community metrics and population measures of individual sensitive species. 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 (see Workshop Summary Report, 2007) suggests that few if any biological indicators currently being evaluated in bioassessment programs are likely to be insensitive to climate change effects. This is largely because climate change effects impact aquatic communities mainly 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 first climate change/biological indicators 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. Examples of such "novel" biological indicators are summarized in Table 3-2.

One consideration that must be taken into account in 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. Consider, 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. 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.

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*)		
	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)
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)

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

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 is given to aspects of possible vulnerabilities of biological assessment programs to 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. Given the preliminary nature of these case studies, only two aspects of typical biological assessment programs were selected for consideration. The approach in the first case study (this chapter) examines a couple of important temporal aspects of detection of change. The second case study (Chapter 5) examines selected spatially-related questions, particularly ability to detect impairment based on comparison to reference conditions, and ability to assign cause of the impairment. More detailed examination of these and other important components of biological assessment programs is essential before comprehensive recommendations for adaptation of bioassessment programs to climate change can be made.

This case study explores, from several points of view, (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, key results, and main conclusions of Case Study 1. Details of this case study, including detailed methods (description of data sets used, sampling methods, locations, and dates, etc.) and detailed results are provided in Appendix C

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

The first question is important since most states use reference populations as the basis of constructing indices and deriving biocriteria. The second is important since 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 is a critical issue in designing monitoring programs to detect meaningful effects that are unknown at the present, and it is expressed as a probability. Power is the ability to detect a real effect that has occurred, defined as the probability of rejecting a false null hypothesis. The more power a test has, the more likely one is to correctly infer that a real change has actually occurred.

This study focuses on climate change effects associated with temperature, as one of several expected climate change effects for which sufficient related information exists to make appropriate estimates of expected effects. Predicted macroinvertebrate taxa loss rates due to temperature increases were derived from the literature. For this study, native or expected taxa richness is considered rather than total richness; species replacement is not being considered. Native taxa are expected to be lost from many streams (e.g., Moore et al., 1997; Xenopolous et al., 2005; Parmesan, 2006), and native taxa richness based on current climate will decrease. Other ecological responses are expected but are not being considered in this study, such as changes in density, range shifts, changes in timing of important life history stages and phenology, morphological, physiological, and behavioral changes, and changes in gene frequencies (Schindler, 1997; Hogg et al., 1998; Walther et al., 2002; Root et al., 2003; Parmesan, 2006). Taxa richness, a very common component metric evaluated in bioassessment programs and incorporated in multimetric indices, is evaluated for signs of bioassessment program vulnerability.

For this case study, variance (reflecting natural variability in biological condition over time) is estimated using existing monitoring data for sites sampled repeatedly over several years during an index period. The Maryland Biological Stream Survey (MBSS) dataset was used to estimate population variance (σ^2). The MBSS biological monitoring program approach and sampling methods are described in Appendix B.

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 taxa richness (Daufresne et al., 2003), and on thermal discharge effects (Lehigh University, 1960; Gammon, 1973). From these sources three temperature-associated taxa loss rates were derived: a high-end loss rate for macroinvertebrate taxa of roughly 4.6 taxa per °C, a low estimate for macroinvertebrates of approximately 1 taxon per °C, and a loss rate for fish of 3.6 fish taxa per °C. These calculations implicitly assume a linear loss rate, which is not perceived as reflecting the actual temporal pattern of species losses over time due to climate change, but does allow projection of further losses in the future.

Projected temperature increases due to climate change for each region of the US were taken from the National Assessment Synthesis Team (NAST) summary report (2001). Predicted temperature increases by the year 2100 ranged, depending on region, between 2.3 °C and 6.5 °C for the Hadley and Canadian models, respectively (see Table C-1, Appendix C). Reported rates of temperature increase were linked with estimated rates of taxa losses to model taxa losses per year due to climate change, considering both the low and high estimates of each.

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 taxa richness of the reference site population?

Using the variance associated with macroinvertebrate data from the MBSS, and assumptions about number of reference stations sampled (n=40) and desired levels of confidence and power (95%, or $\alpha=\beta=0.05$), the smallest difference in number of taxa that could be detected between any two stations (the effect size) is 4.5 taxa. In a "typical" bioassessment program, this is the smallest difference in number of taxa due entirely to climate change effects on temperature that could reliably be discerned. At the high taxa loss rate for macroinvertebrates and the higher estimate for warming in the Northeast/Mid-Atlantic region, it will take 15 years to reach this level of difference (i.e., a loss of 4.5 taxa) and therefore to detect this climate change response (Figure 4-1).

Various scenarios would reduce the estimate of time to detect the taxa-loss climate change effect, including relaxing the desired power and confidence levels, or sampling multiple locations (replicates) that represent the same population of sites (e.g., reference sites). 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 assumption would be faulty given climate change, which is progressive, particularly if long time periods are grouped together. Factors that would increase the required monitoring duration to detect the climate-caused taxa loss include a lesser rate of temperature increase and/or a lower taxa loss rate (see Appendix C). For example, the low taxa loss rate (1 taxon per °C), low temperature scenario in the same Northeast/Mid-Atlantic region would be greater than 100 years at a 95% confidence level (Table C-2).

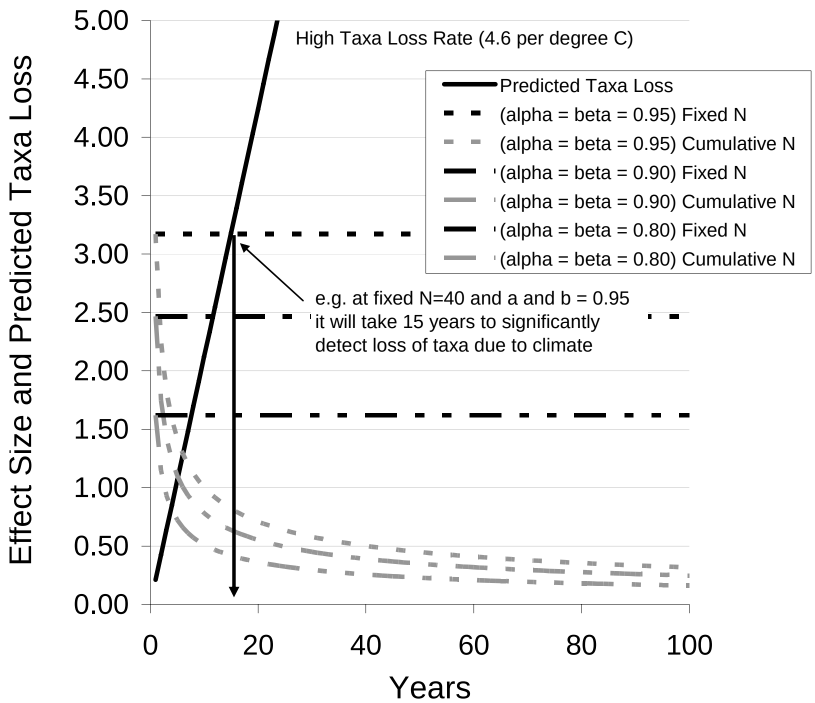

Figure 4-1 - Effects of confidence level (α (a) and β (b)) on time to detect a climate effect on macroinvertebrate taxa loss due to climatic warming at high taxa loss rates in the Northeast/Mid-Atlantic US. Sample size (N) is either fixed at 40 per year or is cumulative. This analysis was based on a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

4.3.2. How long must monitoring be conducted to have a fixed probability of detecting a change in the mean native taxa richness for a particular site?

The 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. 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 taxa richness (Figure 4-2).

Many biomonitoring programs may collect only one sample at a site per year; a means comparison could be applied in this framework, 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. As before, relaxing assumptions about required power and confidence levels will decrease the duration of monitoring needed to be able to detect a climate change effect of taxa loss at a particular site. Locations with higher rates of climate change associated temperature increases and/or higher rates of taxa loss responses would require a lesser duration of monitoring to detect (i.e., statistically demonstrate) the climate change effects, while the converse (lower ranges of temperature increase and/or taxa loss) would increase the required monitoring time.

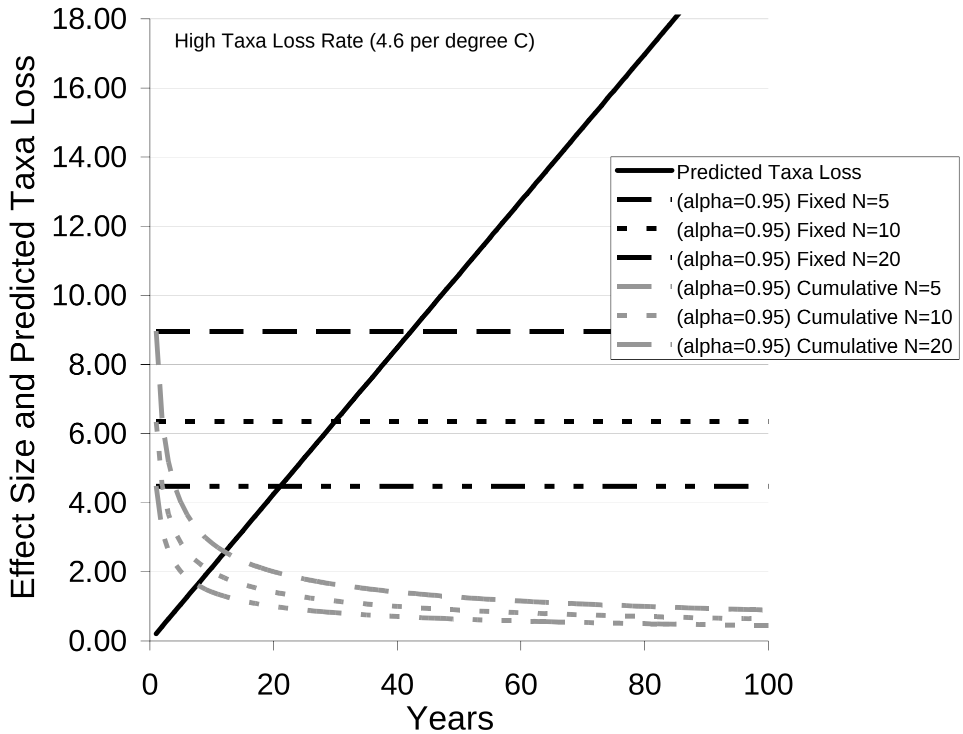

Figure 4-2 - Effects of sample size on time to detect a climate effect on macroinvertebrate taxa loss due to climatic warming at high taxa loss rates in the Northeast/Mid-Atlantic US. The confidence level is fixed at 0.95. This analysis was based on a high estimate of global warming (5ºC by 2100) and the highest macroinvertebrate taxa loss rate.

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 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 time frame or a larger sampling effort to effectively detect climate change taxa losses. 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 impair degrade ability to detect impairment to some extent (see Chapter 5). Since greater variability in the data decreases ability to detect differences in taxa 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 (i.e. "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, including taxa 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 taxa 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 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.

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. Ongoing and more thorough 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. Both of these factors may be constrained by availability of resources (money and manpower). 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. 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 the ability to differentiate between reference conditions and locations of reduced biological condition and the ability to assign cause to impaired conditions, using existing monitoring data and proxy estimates of expected climate changes. This approach is a foundation for defining how monitoring may have to be modified in the face of climate change and how data can be analyzed to account for climate change and remain viable. Details of this case study, including detailed methods (description of data sets used, sampling methods, locations, and dates, etc.) and detailed results are provided in Appendix D.

5.1. OBJECTIVES

The objective of this case study is to examine the potential vulnerability of biomonitoring programs and assessment methods to biological changes that result from climate change. The vulnerabilities examined include

- detection of reduced biological condition, and
- ability to assign cause to impaired condition.

Climate change effects might 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 signal-to-noise ratio could 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. Numerous environmental indices and parameters were applied in this case study that are listed here, and described in detail in Appendix D. Biological responses of streams to various stressors were examined, but with particular emphasis on hydrologic parameters that may be influenced by climate change. 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.

The Maryland Biological Stream Survey (MBSS) dataset was used to evaluate biologic responses to stressors under different conditions. The MBSS biological monitoring program approach and sampling methods are described in Appendix B. Several invertebrate metrics were calculated in the Ecological Data Application System (EDAS) database for the 1320 randomly located benthic samples that were collected over the 10 year period (1994-2004) and analyzed as response variables. Rather than examining all possible biological indicators, we selected 2 fish indicators and 2 benthic macroinvertebrate indicators: the Maryland Fish IBI score, and fish taxon richness; and the Maryland Benthic IBI score (B-IBI), and mayfly-stonefly-caddisfly (EPT) taxon richness. The selected indicators are all responsive general indicators of stress, but are not diagnostic of any particular stressor.

The MBSS data were partitioned based on Maryland's classification into four ecoregions (coastal plain, Eastern Piedmont, Cold Water Highlands, and Warm Water Highlands; 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 has relatively few reference areas. The Warm Water Highlands ecoregion was also analyzed to provide sufficient reference sites (see Appendix D for discussion of the comparability of these two ecoregions).

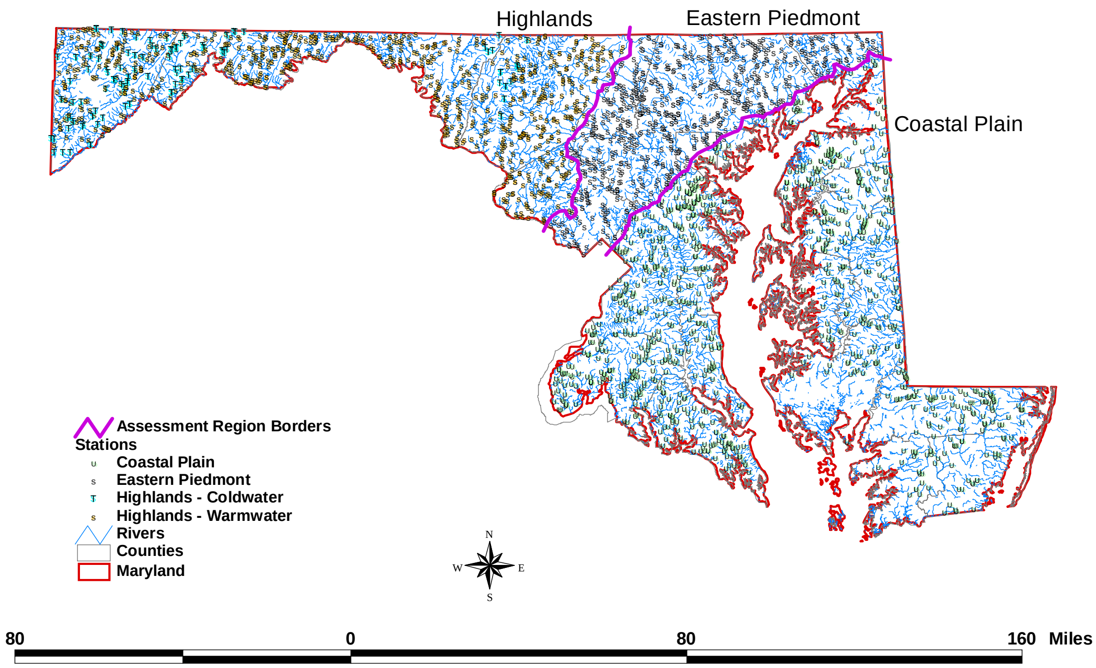

Figure 5-1 – Maryland MBSS sampling stations showing regional divisions

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 Palmer Hydrologic Drought Index (PHDI) for each sampling event. The MBSS data were post-stratified into dry, normal, and wet conditions based on the index, and ability to detect impairment; 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).

5.3. KEY FINDINGS

Detailed results of the basic stressor-response correlations are presented in Appendix D, and not summarized here.

5.3.1. Temperature

Fish taxa richness increased with temperature in warm-water streams in both the Piedmont and in the Highland (Figures 5-2, see also Appendix D), but not in the cold-water streams. EPT and total macroinvertebrate taxa richness were reduced in the cold-water Highland streams where late summer temperatures exceeded 18-20º C (Figure 5-3).

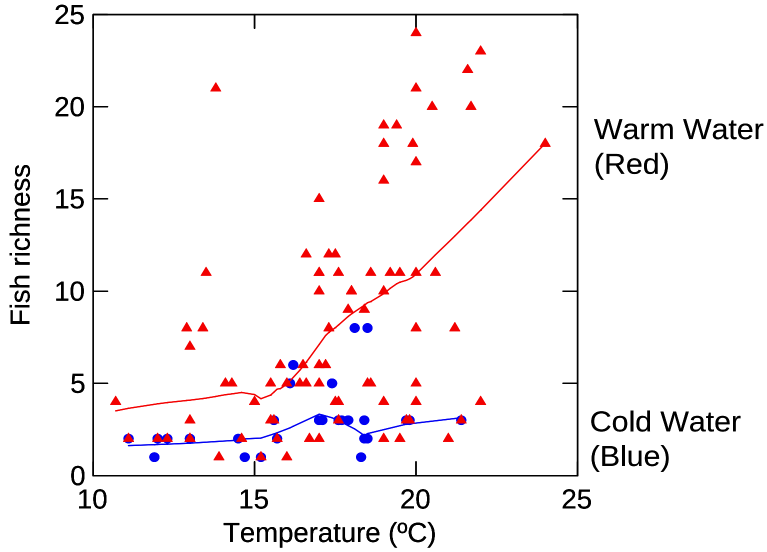

Figure 5-2 – Fish richness vs. temperature in Highland reference streams. Lines are LOWESS estimates.

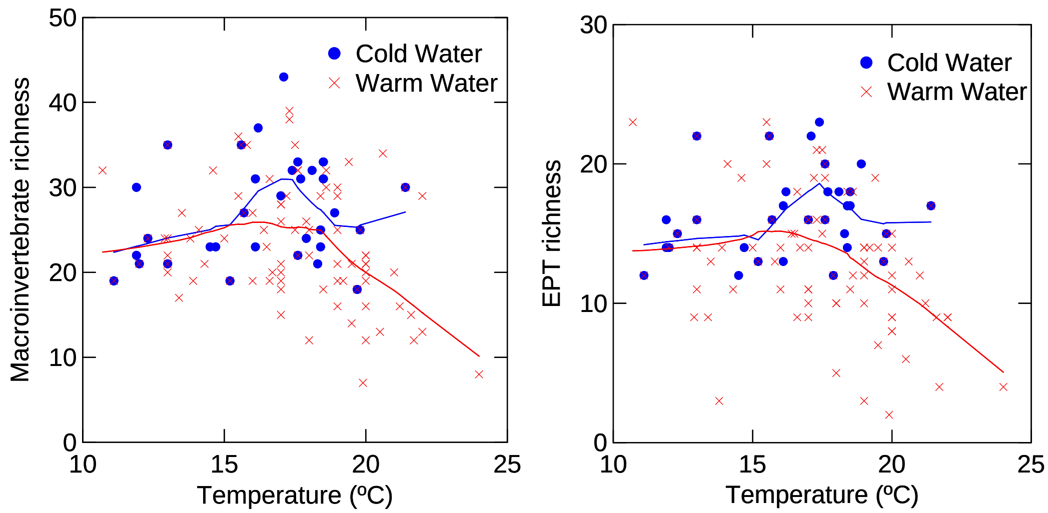

Figure 5-3 – a) Macroinvertebrate richness vs. temperature; and b) EPT richness vs. temperature in Highland reference streams. Lines are LOWESS estimates.

Increases in average regional temperature may have the result that some fraction of cold- or cool-water streams change to warm-water conditions and biota. 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 taxa 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 18 ºC due to climate change, but with no change in streams that remain well below 18 ºC in late summer, suggesting that Highland streams macroinvertebrate communities may be sensitive to climate change according to temperature regime.

5.3.2. Hydrology

Figure 5-4 shows the Benthic IBI (B-IBI) scores of the reference, impaired, and intermediate sites 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. Similar patterns of change were found for the EPT taxa metric and for fish, although fish showed a greater response under wet conditions (see Appendix D). The macroinvertebrate communities at degraded sites were low in EPT taxa and IBI scores, so changes of hydrological condition did not affect them much. It should be noted, as discussed in more detail in Appendix D, 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.

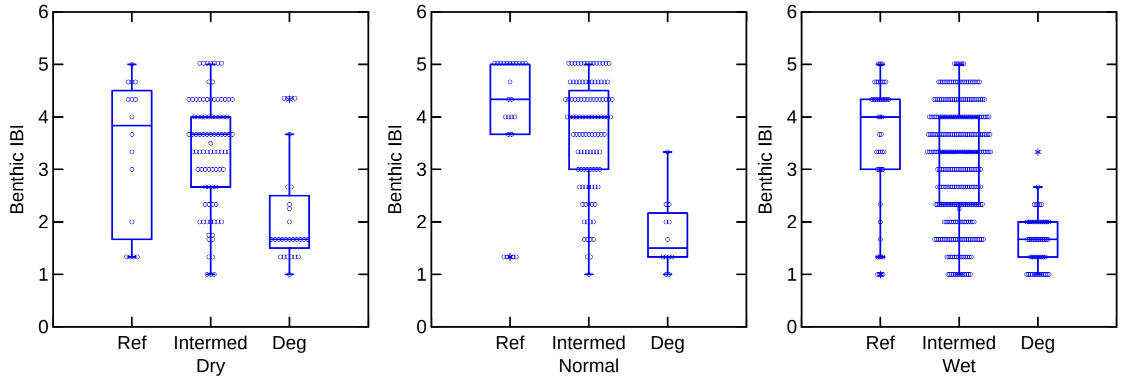

Figure 5-4 – Benthic IBI performance and climatic condition. The dry, normal and wet designations under each of the three graphs refers to categorizations based on the PHDI.

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. Discrimination Efficiency (DE), calculated as the percent of stressed sites with scores less than the 25th percentile of the reference sites (Barbour et al., 1999), shows that increased drought degrades reference sites enough to reduce the ability to discriminate impaired from reference conditions for both the benthic IBI and EPT taxa richness (Table 5-1).

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%

Figure 5-5 shows the stressor-response relationships (with linear regressions) between EPT taxa richness and conductivity for the Piedmont region and the conditional probability analysis, separated by hydrologic condition (base, wet and dry years). The average number of EPT taxa is higher in the base condition and reduced under wet conditions, with little difference between base and dry conditions. To conduct the conditional probability analysis, 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). 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 Appendix D). 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. In a similar analysis comparing benthic invertebrate response and impervious surface, drought conditions yield a higher risk of impairment with impervious surface, but the change is marginal (see Appendix D).

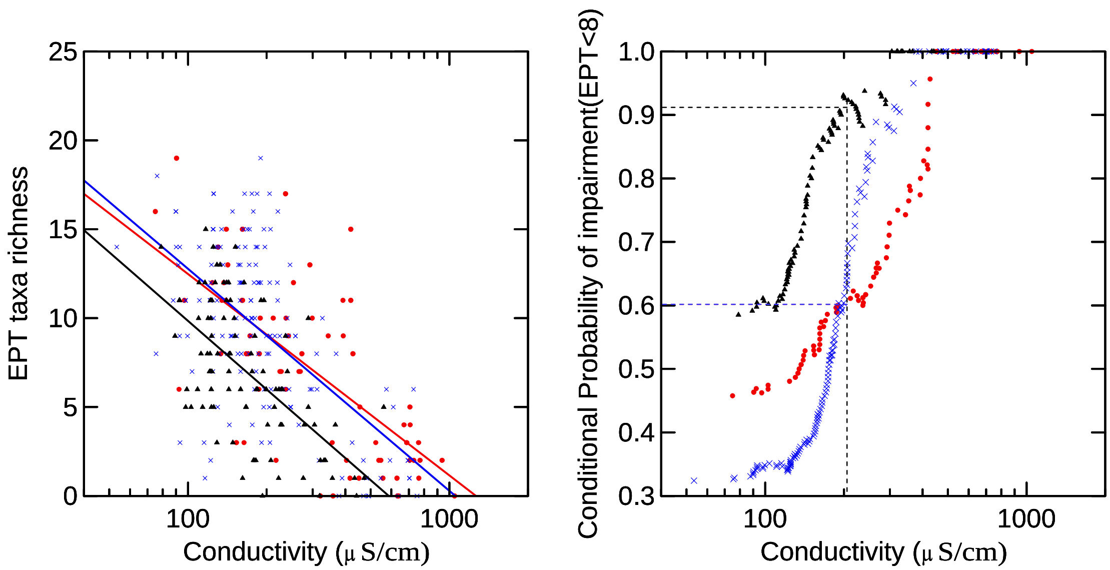

Figure 5-5 – a) Relationship of EPT richness to conductivity under drought (red), base (blue), and wet (black) conditions; and b) conditional probability of impairment for the same three relationships.

5.4. 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 dataset from the Mid-Atlantic Piedmont and Appalachians by moisture conditions estimated by the Palmer Hydrologic Drought Index. 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).

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, which has been settled, farmed, and industrialized since Colonial times. 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 stressor identification 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 of 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) and Tiered Aquatic Life Uses (TALU) would establish a framework for such protection (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 Stressor Identification (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. Stressor Identification 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., TMDL). 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 also 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, called the Biological Condition Gradient (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 Clean Water Act 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, is consistent with ecological theory, and the approach has been verified by aquatic biologists throughout the US (Davies and Jackson, 2006).

Calibration of the BCG to local conditions, and on a nationwide basis, would help establish two baselines that will 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 EPA TO IMPLEMENT A FOUNDATION FOR STATE/TRIBAL BIOASSESSMENT/BIOCRITERIA PROGRAMS TO CONSIDER CLIMATE CHANGE

6.1. RECOMMENDATIONS FOR EPA

Results of the case study analyses conducted 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 are used as a 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 EPA/ORD, and some in the purview of EPA/OW. These identifications are made after each recommendation.

- 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). (ORD)
- Investigate how taxa replacement will affect biological indices used in state programs. Will there be little or no change in biological indices if specific metrics change? (ORD)
- Based on additional research in pilot studies (Section 7), and ongoing interactions with bioassessment managers, 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. (ORD)
- Fill gaps in knowledge and available modeling tools and outputs between regional, hydrologic, and ecological models. (ORD)
- Develop tools to make climate data available to other models (e.g., CADDIS). (ORD)
- 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). (ORD and OW)

- Possibly develop a nationwide database of state biological monitoring and assessment data to support evaluation of national/ecoregional climate change trends and effects. (ORD and OW)
- 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. (ORD and OW)
- Provide technical support for data management tools (e.g., R code) to manage temperature logger data and reduce it to useable metrics. (ORD and OW)
- Form partnerships across EPA and other federal agencies on a comprehensive climate change strategy to address mandates of CWA. (ORD and OW)
- Provide a summary of this meeting to EPA top management for information and support for making informed decision-making. (ORD and OW)
- Include language in EPA grants, policies, etc. on climate change as a stressor for monitoring and assessment programs, to establish climate change as an important program focus. (OW)
- Provide assistance to state bioassessment and resource management programs to integrate the concept of climate change as a significant issue that must be accounted for in assessing the condition of aquatic resources. (OW)
- Evaluate Water Quality Standards to be protective in the face of a changing condition paradigm. (OW)
- Provide funding support for state/tribal water quality programs to assist in adaptations to existing programs. (OW)
- Provide support for identification and sampling of reference sites, re-sampling of reference sites, more intensive characterization of reference and sentinel sites. (OW)

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; and
- Help 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 EPA. These actions include:

- Conducting regular and repeat reference site sampling;
- Considering strategies for maintenance and protection of reference sites and areas, including identification of water bodies in the best condition;
- Evaluating the need to shift the sampling index period and/or expand sampling seasons;
- Establishing sentinel sites for trend monitoring;
- Improving hydrological and temperature data collection;
- Mining historical data records to establish a basis for evaluating climate change;
- Incorporating traditional ecological knowledge, citizen monitoring, phenological knowledge in assessment of biomonitoring data;
- Continuing the refinement of biocriteria programs to incorporate the Tiered Aquatic Life Use (TALU) strategy;
- Accepting moving target paradigm versus steady state model and adapting accordingly;
- Performing critical elements reviews of individual programs to identify relevant refinements; and
- Engaging 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 on the next most appropriate ecosystem for state and tribal WQ agency managers and biologists. 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, 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 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 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.
    - As freshwater ecosystems, lakes and freshwater wetlands would be subject to very similar climate change effects as streams/rivers, but would have, in many cases, different ecological responses with different levels of importance (e.g., wetlands may be particularly susceptible to droughts). Thus, this effort would build effectively upon the experience and knowledge developed in the first workshop, but expand both the knowledge base as well as consideration of ecosystem responses.
    - Combining lakes and freshwater wetlands offers some obvious 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 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 US 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 data base (classification by ecological traits and sensitivities may already exist for the state data bases 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 Biological Condition Gradient (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 Palmer Hydrologic Drought Index 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 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 data base for bioassessment data, which accommodates physical, chemical and habitat data, and biological data for multiple assembages including detailed taxomonic 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. 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. 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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