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THE EFFECTS OF CLIMATE CHANGE ON STREAM INVERTEBRATES IN THEIR ROLE AS BIOLOGICAL INDICATORS AND RESPONDERS TO DISTURBANCE

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Anna T. Hamilton

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M.S., Aquatic Biology, Towson State University, 1983

DISSERTATION

Submitted in Partial Fulfillment of the
Requirements for the Degree of

Doctor of Philosophy
Biology

University of New Mexico
Albuquerque, New Mexico

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ABSTRACT

Global climate models provide estimates of future changes in air temperature and

precipitation patterns, drought, flooding, sea-level rise, and increases in the frequency,

duration, and intensity of extreme heat and storm events. These climate changes will

affect stream invertebrate communities directly, indirectly, and through interactions with

other stressors, resulting in a range of biological responses, including species range shifts,

losses and replacements, novel community compositions, and altered ecosystem

functions and services. Effects will vary regionally and present heretofore unaccounted

influences on biomonitoring, which water-quality agencies use to assess the status and

health of ecosystems as required by the Clean Water Act. Biomonitoring uses biological

indicators and metrics to assess ecosystem condition, and is anchored in comparison of

test sites to regionally established reference benchmarks of ecological condition. Climate

change will affect responses and interpretation of these indicators and metrics at both

reference and non-reference sites and, therefore, has the potential to confound the

diagnosis of ecological condition if reference and non-reference sites do not change in parallel. This dissertation analyzes four regionally distributed state biomonitoring data sets to inform on 1) how biological indicators respond to the effects of climate change, 2) what climate-specific indicators may be available to detect effects, 3) how well current sampling detects climate-driven changes, and 4) how program designs can continue to detect impairment. In addition, responses were examined of stream invertebrates to a particular extreme event, wildfire, which is expected to increase in the future with climate change.

In general, we found that temperature trait groups (cold- and warm-water preference taxa), as well as several taxonomically-defined invertebrate metrics and indicator groups, show responses consistent with expectation to long-term increases in temperature, although the climate responsiveness of these trait groups varied among states and ecoregions. Temperature sensitivity of taxa and their sensitivity to organic pollution were moderately but significantly correlated. Therefore, metrics selected for condition assessments because taxa are sensitive to disturbance or to conventional pollutants also were sensitive to changes in temperature. We explored the feasibility of modifying metrics by partitioning components based on temperature sensitivity to reduce the likelihood that responses to climate change would confound the interpretation of responses to impairment from other causes and to facilitate tracking of climate-change-related taxon losses and replacements. Difficulties arise in categorizing unique indicators of global changes, because of similarities in some of the temperature and hydrologic effects resulting from climate change, land use changes, and water removal. Nevertheless, the utilization of climate-sensitive traits to modify traditional

biomonitoring metrics is promising in the context of using a weight-of-evidence approach for interpreting bioassessment results. Observed invertebrate responses to climate change also impair the condition of reference stations. Combined with our projection that encroachment of developed land uses over time will negatively impact up to a third or more of currently established reference sites by the end of the century, these responses suggest the importance of accounting for reference condition drift through implementation of an objective scale for condition characterization, as well as the need for reference station protection.

These results can be used to identify methods that assist with detecting climate-related effects and highlight steps that can be taken to ensure that programs continue to meet resource protection goals. However, we also must recognize that many aspects of global changes are not tractable at the local to regional scales at which bioassessment is applied in support of water quality management, suggesting the need for a shift in the scale of approach from a narrow focus in the process of water resource quality protection and restoration to one that encompasses broader adaption strategies to address and manage global change impacts.

Overall, we found the response of stream benthic invertebrates to a major wildfire were not devastating, with minimal responses found in total abundance or taxa richness. However, numerous taxa responded to post-fire flow and water quality disturbances either positively, negatively, negatively with recovery, or neutrally, with responses well captured by selected habit and feeding type traits. Post-fire benthic responses reflected three categories, vulnerability, resistance, and resilience, with different groups of organisms and different trait characteristics comprising each. Vulnerability was observed

to both direct physical disturbance, mediated by the flow pulses that followed the fire; and to trophic impacts, themselves a response to loss of food resources due to those same flow pulses and associated water quality effects. Resistance to the post-fire physical disturbance of the stream environment was exhibited by a subset of invertebrates with habit traits that conferred the ability to withstand dislodgement and displacement that would otherwise be expected from the post-fire flow pulses. Resilience, or ability to recover in the short term following cessation of the most prominent post-fire flow events, was conferred mainly by opportunistic life-history traits. This suite of responses suggest the mechanisms through which benthic communities may be altered in the long-term, through suppression of some vulnerable taxa, partial if not temporary (short-term) suppression of some trophic resources, and possibly incomplete recovery (relative to pre-fire conditions) based on the reproductive life-history characteristics of component taxa.

CHAPTER 3

Implications of global change for the maintenance of water quality and ecological

integrity in the context of current water laws and environmental policies

(Anna T. Hamilton, Michael T. Barbour, and Britta G. Bierwagen)

CHAPTER 4

Short-Term Effects of the Las Conchas Fire on Benthos in the East Fork Jemez River in the Valles Caldera, New Mexico

(Anna Hamilton, Clifford N. Dahm, Rebecca J. Bixby, Gerald Z. Jacobi, Betsy M. Shafer, Lauren Sherson, Virginia F. Thompson, Alexander Clark, Shann M. Stringer)

CHAPTER 5

Summary

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CHAPTER 1

INTRODUCTION

Climate Change Context

The reality of global climate change is well established (IPCC 2007a, b; NCADAC 2013), if not well perceived by the public (Seacrest et al. 2000). Changing patterns of climate forcing are expected to alter spatial and temporal patterns of air temperature and precipitation that in turn will drive changes in sea level rise, ice cover, timing and magnitude of snow melt, evapotranspiration, drought, flooding magnitude and frequency, and other extreme events (IPCC 2007a). Many of these global changes will impact people directly (NCADAC 2013). Climate change impacts to stream and river systems are also a specific concern for people, because of our dependence on surface waters for water supply, recreational uses, and ecosystem services, as well as our concerns with environmental quality, habitat protection, and biodiversity (Palmer et al. 2009).

Projections of climate change patterns for air temperature and precipitation come from general circulation models (GCMs). These models provide good insight into the magnitude and scale of changes that will affect the landscapes drained by streams and thus the structure and function of the streams themselves. Combining the outputs from numerous GCMs over a range of potential future scenarios for greenhouse gas emissions provides an ensemble of projected changes in temperature and precipitation patterns (IPCC 2007a). Global average projections of temperature increase over the current century by values that range from $1.1-2.9^{o}C$ for the lowest emissions scenario to $2.4-6.4^{o}C$ for the highest (IPCC 2007a). This projected rate of increase is ~53% higher (about $0.2^{o}C$ per decade) than the rate observed over the last 50 years ($0.13^{o}C$ per decade), and

further rate increases are considered possible (IPCC 2007b, Ramstorf et al. 2007, Hansen et al. 2006). Projections are more uncertain for precipitation (IPCC 2007a), and vary among major geographic regions of the continental U.S. (e.g., Shoof et al. 2010). However, general projections include increased frequency of heavy precipitation events, 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, longer periods of low flow, and more frequent droughts in summer (IPCC 2007a, Barnett et al. 2005, Hayhoe et al. 2007, Fisher et al. 1997). In some regions, particularly the west and southwest, other extreme events such as major wildfires are projected to increase in connection with higher spring and summer temperatures, earlier spring snowmelt, and increasing aridity (Westerling et al. 2006, Seager et al. 2007).

Changes in air temperature and precipitation are key climate change factors that will impact stream and river ecosystems through direct effects on water temperature and hydrologic regimes, as well as through indirect effects on dissolved oxygen (DO), pH, nutrients, and other dissolved constituents, and changes in the assimilation capacity of pollutants into receiving waters, sediment erosion and deposition, and habitat structure. Just as climate patterns and projections for climate change vary among regions of the U.S. (and globally), effects of climate change on stream systems will also vary geographically. Stream water temperature patterns closely follow air temperature patterns (e.g., Mohseni et al. 2003, Pilgrim et al. 1998, Stephan and Preudhomme 1993), potentially resulting in predictable regional shifts in stream thermal regimes. However, stream water temperatures are directly driven by solar insolation (rather than air temperature) and are affected by smaller scale factors that include variations in flow

volume and snow melt, ground water influence, aspect, riparian shading, presence of deep pools, meteorology, river conditions, and geographic setting (e.g., elevation, gradient, etc.) (Allen and Castillo 2007, Caissie 2006, Mohseni et al. 2003, Daufresne et al. 2003, Hawkins et al. 1997, Ward 1985). Patterns in changing stream temperatures may not be predictable without accounting for at least some of these effects. Recent estimates of 'climate velocity,' which relate future projections of change in a climate variable such as temperature to the spatial gradient in that variable, yield a geographically more complex set of climate change expectations. These expectations reflect, for example, more uniform projections of magnitudes and directions of change in the plains of central and southeastern U.S., and more variable expectations in mountainous regions (Dobrowski et al. 2012). Nevertheless, changes in the thermal regimes of streams and rivers in response to climate change have been documented from long-term river temperature datasets around the country (e.g., Kaushel et al. 2010).

Climate changes also will impact the hydrologic characteristics of streams, with consequences to stream biological communities (e.g., Webb et al. 2009, Dewson et al. 2007, Suren and Jowett 2006, Lind et al. 2006, Poff 2002, Extence et al. 1999, Stanley et al. 1994). The IPCC (2007a) projects average annual runoff to increase by 10-40% at high latitudes and in some tropical areas, but to decrease by 10-30% over some mid-latitudes dry regions and the dry tropics. In North America, projected changes in average stream flow range from an increase of 10–40% at high latitudes to a decrease of about 10–30% in mid-latitude western North America by 2050 (Milly et al. 2005). The hydrologic regime of a stream is not a singular variable, and the range of hydrologic alterations that can result from the combination of increasing magnitude and variability of

temperatures combined with a range of projected changes in precipitation and drought

conditions is great, potentially including longer duration and lower summer low flows,

greater incidence of floods, and greater flashiness. In western/southwestern snow pack

dominated regions, the combination of warming temperatures, a shift toward less winter

precipitation falling as snow, and snow melt occurring earlier will shift the peak runoff

from spring to late-winter/early spring, accompanied by a reduced magnitude of snow

pack (Barnett et al. 2005, Clow 2010). Typical projections are for peak runoff to shift

from about two weeks up to one month earlier by the end of the century (Dettinger et al.

2004, Hayhoe et al. 2007). Stewart et al. (2005) have already found evidence for shifts of

this magnitude (1-4 week earlier timing of snow melt and runoff based on data from 1948

to 2002) for several montane catchments in the western U.S. The effects of water

temperature can also interact with stream flow alterations, with higher temperatures and

higher warming rates during low flow conditions (Vliet and Zwolsman 2008, Zwolsman

and Van Bokhoven 2007, Sinokrot and Gulliver 2000). As a result, influences of stream

temperatures and flow conditions cannot always be separated in terms of their effects on

biota.

Freshwater ecosystems are considered sensitive to climate change impacts because

of their fundamental dependence on hydrology and thermal regimes, their dominance by

poikilotherms, their high degree of isolation and fragmentation within terrestrial systems,

and their vulnerability to impact by humans and further complications via interactions

with other stressors (Isaak and Rieman 2013, Woodward et al. 2010, Durance and

Ormerod 2007). There is growing information on the effects of climate change on aquatic

ecosystems (e.g., Doledec et al. 1996, Durance and Ormerod 2007, Daufresne and Boet

2007, Buisson et al. 2008, Hiddink and Hofstede 2008, Collier 2008, Chessman 2009, Flenner et al. 2010, Britton et al. 2010, Winfield et al. 2010, Floury et al. 2013). Temperature regimes greatly affect the distribution and abundance of aquatic species and communities in relation to temperature tolerances and evolutionary adaptations combined with competitive interactions, effects on food supply, and other factors (e.g., Matthews 1998, Hawkins et al. 1997, Vannote and Sweeney 1980, Sweeney and Vannote 1978). Changes in prevailing temperature regime, as well as increased variability of temperature associated with /induced by climate change, may have various biological effects. Species ranges will shift to the north and/or to higher elevations (Comte and Grenouillet 2013, Comte et al. 2013); to a lesser extent, species at the southern limits of their ranges may migrate or suffer local extinctions. 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); or in some regions including the Southwest and southern Great Plains by the prevalence of east-west drainages (Poff et al. 2002). Species that are already restricted to headwater streams may be displaced (Poff et al. 2002). Changes in stream flow from climate change may alter community structure through alteration of quantity and quality of available habitat, as well as through tolerances or requirements for particular flow conditions. Species replacements, range shifts, and variations in community composition for both fish and macroinvertebrates have been documented and associated with increased temperatures and lower flows (Daufresne et al. 2003, Durance and Ormerod 2007, de Figueroa et al. 2010).

Bioassessment Context

In the U.S. and in many other countries around the world, environmental protection regulations such as the U.S. Clean Water Act (CWA) of 1972 identify the restoration and maintenance of physical, chemical and biological integrity of streams and rivers as a long-term goal (Barbour et al. 2000). Biological assessment, or 'bioassessment', is relied on as a tool applied by resource managers to achieve this goal (Norris and Barbour 2009). Bioassessment has been found to be more effective than sampling only chemical parameters (Karr 2006), largely due to the recognition that biological indicators reflect an integrated response to all environmental conditions to which they are exposed over time (Moog and Chovanec 2000, Barbour et al. 2000). Thus, biological indicators 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). Biological assessment, coupled with multi-metric or predictive modeling analyses, is a strong approach for diagnosing diminished ecological integrity, minimizing or preventing degradation of river systems (Karr and Chu 2000), and measuring success of stream and river restoration and mitigation efforts (Palmer et al. 2010).

In the U.S., biological assessment plays a central role in water quality programs that are components of the CWA, from assessment of water quality and identification of biologically impaired waters to development of total maximum daily load (TMDL) for impaired water bodies. Bioassessment results also are used to help identify causes of observed impairments, based on the assumption that various components of aquatic communities will respond differently to different types of stressors. Other CWA

programs that depend on bioassessment data include permit evaluation and issuance, tracking responses to restoration actions, and other components of watershed management and restoration.

Bioassessments rely on characterization of biological indicators and metrics to assess ecosystem condition, and are grounded in comparisons between reference and non-reference sites. Climate change, through effects on both reference and non-reference sites, will impact the responses and interpretation of traditional indicators and metrics, and has the potential to confound diagnosis of cause for impaired sites not attaining their aquatic life use. Climate change can confound the interpretation of bioassessment data if climate changes drive responses in biota that cannot be linked effectively to their climate change cause, which will be the case in the absence of specific diagnostic efforts which until recently have been lacking. These could include, for example, development of climate-sensitive indicators or metrics, or incorporation of long-term sampling for examination of climate-related trends. Without such focused diagnostic approaches, any biological responses observed might be attributed to the range of traditional stressors commonly evaluated. Climate change can further confound the diagnosis of impairment through climate change-associated degradation of reference sites, making reference sites more similar to degraded sites and thus more difficult to differentiated statistically between reference and test conditions, and more difficult to determine degree of impairment (U.S. EPA 2008).

A variety of biological metrics and indices have been developed as ecological indicators used to gauge the condition of aquatic ecosystems relative to reference conditions and to judge causes of degradation (Niemi and McDonald 2004). Indicators

are used as early warnings of degradation, as they often simplify complex environmental data. Various factors govern the selection of biological indicators, including appropriate spatial and temporal scales, incorporation of natural variability, and sensitivity to the range of stressors expected in a system (Niemi and McDonald 2004). The concept of linkage between biological indicators and the stressors on a system is crucial to the interpretation of bioassessment results. Stream benthic invertebrates are the most common assemblage used for biomonitoring in this respect (USEPA 2002). Benthic invertebrates are considered sensitive indicators of their natural environment as well as of a variety of perturbations. Their integrative characteristics make benthic 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.

Given the evidence of how climate is changing and potentially affecting stream population and community responses, it is clear the changing influences of such a major environmental driver should be accounted for in terms of its effects on the components of bioassessment (Figure 1), all of which can be affected by climate change. This dissertation and the larger project of which it was a part are among the first to address many of these concerns, focusing primarily on several of the elements that comprise assessment design (Figure 1), and applying results to consideration of environmental management with regard to their implications for decision making. The goals of this work are to contribute to the foundation for understanding how climate changes affect bioassessment indicators and for advancing the development of specific strategies to ensure the long-term effectiveness of monitoring and management plans.

Bioassessment Program Activities

Assessment Design
- Select reference sites   - Select indicators
- Determine reference   - Create indices,
  condition   predictive models

Implementation
- Collect samples   - Analyze data

Environmental Management
- Develop biocriteria   - Determine impairment
- Determine water   - Develop TMDLs
  quality standards

(Left vertical labels: Climate Change Effects; Life History, Distribution, and Abundance of Macroinvertebrate Species)

Figure 1. Climate change can affect many bioassessment program activities from the initial assessment design, to collecting and analyzing data, and to developing responses to assessment outcomes (TMDLs = total maximum daily loads).

Vulnerability of Metrics and Indices to Climate Change

The first section of this dissertation (Chapter 2, Hamilton et al. 2010a,

'Vulnerability of biological metrics and multimetric indices to effects of climate change')

used several long-term state biomonitoring data sets to investigate 1) whether biological

response signals to climate change are discernible within existing bioassessment datasets;

2) how responses of various biological indicators can be categorized and interpreted with

regard to apparent climate sensitivity or robustness, using both trend analysis and

partitioning of hot and cold years as a proxy for long-term changes in temperature; 3)

whether climate change-sensitive indicators could be developed based on

environmentally realized temperature preferences, and applied to metrics to enhance the

ability to differentiate climate change-related influences from the responses to other stressors; and 4) the vulnerabilities to climate change effects of the metrics and multimetric indices used to assess impairment to the types of climate change responses found. Results can be used to identify methods that assist with detecting climate-related effects and highlight steps that can be taken to ensure that programs continue to meet resource protection goals.

This chapter of my dissertation is a 'data mining' study. The study uses analyses of existing, long-term biomonitoring datasets, which were collected for another purpose (i.e., to monitor the status of stream biota using reference-based comparisons) to address a new question for which the original collection programs were not designed. Global climate change effects on stream habitats can be seen as long term, progressive changes overlain on other natural sources of variability, including other climate drivers. While there are certainly some questions about climate change effects that can be addressed using spatial comparisons, climate change for the most part is a long-term temporal issue. Trend analysis is a logical approach to investigate long-term patterns in temperature, precipitation, flow, other habitat variables, and biological response variables. Trend analysis forms a foundation for examining evidence that long-term, progressive global changes are contributing to the trends, and for considering other possible contributions. Given that in some of the states in the USA there exists biomonitoring programs that have been in place for long periods of time (e.g., 2+ decades), and that outside of this arena long-term biological datasets are relatively rare (Jackson and Fureder 2006), it is an attractive opportunity to apply these long-term biological datasets to the climate change-related questions that are the focus of this study. This type of *post-facto* analysis of

historical datasets has been used by others to determine whether climate change effects are already discernible in ecosystem responses (e.g., Daufresne et al. 2003, Durance and Ormerod 2007, Burgmer et al. 2007, Murphy et al. 2007). However, this data mining approach has several pitfalls, including the lack of broad spatial coverage by stations with continuous long-term data, and the lack of long-term data at both reference sites that are minimally affected by other major anthropogenic stressors as well as stressed sites to help differentiate related responses. As is often the case with the opportunistic use of mined data, the existing biomonitoring datasets available for analysis in this study did not always meet all the criteria that would have allowed the most rigorous evaluation of the study questions.

Biological responses to climate change are likely to include interactions with climate cycles such as the Pacific Decadal Oscillation (PDO) or North Atlantic Oscillation (NAO), which can act synergistically or antagonistically with climate change, depending on their phases (e.g., Seager and Vecchi 2010). These interactions can potentially enhance or obscure the types and magnitudes of biological responses that might be expected over the long term. 'Climate change' can be considered the long-term, average directional changes that span multiple climate cycle oscillations. Nevertheless, the types, directions, and with caution, the magnitudes of biological responses to climate change can be inferred from shorter-term (one to two decade) patterns, even in the absence of the ability to partition long-term direction and cyclic climate patterns. Indeed, the occurrence of cyclical climate changes provides a set of 'natural experiments' that we can take advantage of to gain insights into what kinds of biological responses can be expected, and how these will play out with respect to bioassessment. Inference would be

based on linkages between changes in climate factors, changes in stream conditions, and

associated changes in biological metrics. This will yield valuable information on the

identification of biological metrics that are sensitive to such climate-associated changes.

It is common practice to infer probable sources of cause by clear associations between

types and sources of stressors present and responses of biota whose autecology

characteristics are known (Norris and Barbour 2009, Cormier and Suter 2008).

Linkages to Water Law, Stream Management, and Regulation

The second section of this dissertation (Chapter 3, Hamilton et al. 2010b,

'Implications of global change for the maintenance of water quality and ecological

integrity in the context of current water laws and environmental policies') takes the

results from the studies conducted in Chapter 2 and expands on the evaluation of the

implications of those metric and index responses to the outcomes of decision making for

water resource management. This chapter of the dissertation considers the interactions of

other global change parameters, particularly land use and population growth, in addition

to climate change, with respect to their potential influences on bioassessment metrics and

indices. The chapter also assesses the sensitivities of reference station conditions and

associated impacts to reference-based comparisons that are the foundation of

bioassessment. It compares bioassessment approaches around the world to examine

programmatic vulnerabilities to the combination of global change parameters, and

discerns from this potential programmatic adaptations that could preserve our ultimate

regulatory goals of preserving good water quality and ecological integrity. Awareness

that climate change and other landscape or global-scale stressors can have widespread

effects on biological communities introduces additional uncertainty into a system that assumes there are interpretable patterns of biological indicator responses to "conventional" stressors. This has the potential to cast doubt on assertions of stressor-response relationships that are being evaluated within a regulatory context, and also highlights the potential value of tailoring biomonitoring tools to the types of stressors expected, as well as accounting for the influences of stressors operating at different scales. The study also addresses limitations in this objective related to similarities in biological responses to the sometimes common effects that can arise from different sources. With increasing knowledge of the types of global change effects that are appearing to different degrees in regions around the country and the world, and of the categories of organisms that are showing the most predictable responses, it becomes more realistic to consider adjusting assessment tools and approaches to account for landscape- to global-scale stressors through adaptations to existing programs.

Linkages to an Extreme Event - Fire

Stream biological communities change not only in response to long-term shifts in climate, land use, and other environmental drivers, but also due to the occurrence of extreme events. Large wildfires are one type of extreme event that not only restructure terrestrial ecosystems (e.g., Johnson 1992), but also impact the stream ecosystems within these forested biomes (see Minshall 2003 for a review). Both the frequency (Westerling et al. 2006) and severity (Miller et al. 2009) of forest fires in the western and southwestern U.S. are expected to increase with climate change, due to higher spring and

summer temperatures and earlier spring snowmelt (Westerling et al. 2006) and the increasing frequency of drought (Allen 2002).

A study of changes in benthic macroinvertebrates associated with fire provides a contrast between short-term acute and long-term chronic disturbances, both associated with human-influenced climate change. Bioassessment approaches are generally attuned to accounting for short-term influences with identifiable causes. But periodic, though unpredictable, disturbances will punctuate observed long-term changes, and could alter the long-term trajectories expected in species and community responses. This component of the dissertation provides an initial opportunity to characterize the nature of stream invertebrate responses to an acute, watershed-scale disturbance in terms, such as an evaluation of responsive traits, which will allow comparison to long-term climate change responses.

A long-term, well-instrumented study site was established on the East Fork Jemez River in the Valles Caldera National Preserve in the Jemez Mountains of northern New Mexico, with the long-term goal of studying climate variability and climate change impacts on water quantity and quality in a snowmelt-dominated system. It was also a goal to investigate various ecological linkages with the seasonal dynamics and long-term variations in water quantity and quality characteristics that were being documented through sampling of a variety of biological processes and communities including stream benthic invertebrates. An unexpected but highly valuable opportunity to study the short-term impact and recovery responses of invertebrates to major wildfire was provided by the Las Conchas fire, which burned a vast region in the Valles Caldera and the Jemez Mountains in 2011. The third section of this dissertation (Chapter 4, 'Short-Term Effects

of the Las Conchas Fire on Benthos in the East Fork Jemez River in the Valles Caldera, New Mexico') reports on this spontaneous experiment. We used the assessment of various functional trait groups to identify and differentiate among invertebrate responses to physical disturbances, which we hypothesized were more likely in the short term, and to trophic disturbances, and made linkages between the direct fire effects on stream flow and water quality conditions, and the indirect effects on stream benthos. Implications were considered of the effects of this type of acute, watershed-scale disturbance that often generates a greater magnitude of response than that of a long-term chronic influence such as climate change, on long-term community invertebrate composition.

CHAPTER 2

Vulnerability of biological metrics and MMIs to effects of climate change

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Abstract

Aquatic ecosystems and their fauna are vulnerable to a variety of climate-related changes.

Benthic macroinvertebrates are used frequently by water-quality agencies to monitor the

status of aquatic resources. We used several regionally distributed state bioassessment

data sets to analyze how climate change might influence metrics used to define ecological

condition of streams. Many widely used, taxonomically based metrics were composed of both cold- and warm-water-preference taxa, and differing responses of these temperature-preference groups to climate-induced changes in stream temperatures could undermine assessment of stream condition. Climate responsiveness of these trait groups varied among states and ecoregions, but the groups generally were sensitive to changing temperature conditions. Temperature sensitivity of taxa and their sensitivity to organic pollution were moderately but significantly correlated. Therefore, metrics selected for condition assessments because taxa are sensitive to disturbance or to conventional pollutants also were sensitive to changes in temperature. We explored the feasibility of modifying metrics by partitioning components based on temperature sensitivity to reduce the likelihood that responses to climate change would confound responses to impairment from other causes and to facilitate tracking of climate-change-related taxon losses and replacements.

Key words: climate change, biological indicators, biological metrics, multi-metric indices, vulnerability, biomonitoring, macroinvertebrates

Introduction

Water-quality agencies measure responses of biological indicators to assess the status and health of ecosystems and to establish biological criteria for defining acceptable condition of communities in rivers and streams regulated under the 1972 US Clean Water Act (CWA; section 303[c][2][B]) and 304[a][8]). Stream benthic invertebrates are used frequently for biomonitoring in the US (USEPA 2002). Climate change has the potential to alter benthic invertebrate communities, and therefore, their use as the basis for assessments of stream condition and CWA-related management decisions. Thus, climate-related shifts in benthic community structure are relevant to bioassessment efforts (Dolédec et al. 1996, Daufresne et al. 2003, 2007, Mouthon and Daufresne 2006, Bêche and Resh 2007, Burgmer et al. 2007, Durance and Ormerod 2007, Collier 2008, Chessman 2009). However, the vulnerabilities of bioassessment/biomonitoring to climate-related shifts in community structure have not been evaluated.

Assessment of stream status requires distillation of data on macroinvertebrates, fish, or other stream assemblages into a format that reflects biological responses to environmental conditions. Multimetric indices (MMIs) and predictive modeling are 2 approaches frequently used to distill biomonitoring data. Both are grounded in the assumption that environmental conditions, both natural (e.g., climate, physiography, geology, soil type) and anthropogenic (e.g., land use, pollutant discharges), drive the structure and functioning of biological communities (e.g., Poff and Ward 1990, Allen 1995), so that expectations for reference community composition and responses of disturbed communities can be compared as indicators of degradation (e.g., Barbour et al. 1999, Hawkins et al. 2010). Any metrics or indices of community condition must be

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readily compared between reference and test locations (Hering et al. 2006a, b). We focused on evaluating shifts in some commonly used metrics and in reference community composition and assessed their potential effects on site-condition classifications.

MMIs generally are structured as composites of biological metrics selected to capture ecologically important community structural or functional characteristics and have been applied to fish and benthic macroinvertebrate communities (Karr 1991, DeShonn 1995, Barbour et al. 1995, Yoder and Rankin 1998, Sandin and Johnson 2000, Böhmer et al. 2004, Norris and Barbour 2009). Component metrics are selected based on their responsiveness to the environmental effects most often evaluated (Barbour et al. 1999, Hering et al. 2006c, Johnson et al. 2006). Sites are assessed by comparing the MMI score for the test site to values at comparable reference sites. Predictive models use regional reference conditions to develop relationships between environmental predictor variables and macroinvertabrate taxon occurrence from which predictions for an "expected" (E) community are based. A commonly applied model for macroinvertebrate communities is the River InVertebrate Prediction And Classification System (RIVPACS) (Wright 2000). An important assumption is that the predictor variables are minimally affected by human disturbance and are relatively invariant over ecologically-relevant time (Tetra Tech 2008, Hawkins et al. 2000, Wright 2000, Wright et al. 1984). The E community is then compared to various "observed" (O) communities at non-reference locations. A basis for comparison is that any differences between O and E communities reflect biological responses to the range of environmental pollutants or alterations that are intended to be evaluated. For both approaches, the underlying assumption of site comparisons is that degradation in metrics or scores reflects responses of the aquatic

community to stressors.

Climate change is a stressor that is likely to affect MMI scores. Thus, MMIs must be evaluated to determine: 1) their responsiveness to climate change, 2) whether responses to climate change can be differentiated from responses to conventional stressors, and 3) whether MMIs will continue to be useful tools for attributing likely causes of degradation.

The International Panel on Climate Change (IPCC; IPCC 2001) defined vulnerability as the extent of susceptibility of a system to sustaining damage from climate change, including variability in climate (see also Hurd et al. 1999). Vulnerability is affected by degree of exposure and by sensitivity. Vulnerability of biological indices and metrics can be judged on the basis of existing evidence of biological responses to climate change (exposure), the range of metric responses to climate-related changes in temperature (sensitivity), and the effect of observed changes in metrics on site-condition classifications. We examined bioassessment data sets from 3 US states (Maine, North Carolina, Utah) to assess the vulnerability of biological metrics and indices to climate change. Bioassessment of wadeable streams is based on MMIs in Maine and North Carolina and on predictive modeling in Utah. These states are representative of major ecoregions of the US, and the data sets encompass large-scale variations in current and future climatic conditions, geography, topography, geology, and hydrology. Thus, our results provide a regional view of climate-change implications for commonly used MMIs and predictive models.

Methods

State biomonitoring data sets

We used biomonitoring data sets from Maine, North Carolina, and Utah for our analyses because they are relatively long-term data sets of high quality. Macroinvertebrate collection methods and assessment techniques differ among these states.

Utah.—The protocol used by Utah Division of Water Quality (DWQ) calls for quantitative samples collected from riffle habitats with the US Environmental Protection Agency (EPA) Environmental Monitoring and Assessment Program (EMAP) kick method (UTDWQ 2006). Samples are collected during an autumn index period (typically September/October), and a River InVertebrate Prediction and Classification System (RIVPACS; Wright 2000) model is used as a basis for site-condition classification. The model has 15 predictor variables, and 7 are related to climate (e.g., temperature, precipitation, freeze dates).

Maine.—The protocol used by Maine Department of Environmental Protection (DEP) calls for use of artificial substrates (rock bags or baskets) to collect quantitative samples during late-summer, low-flow periods (July 1–September 30). Site condition is rated with a set of 4 linear discriminant models that incorporate 30 input metrics or indices, and sites are assigned to 1 of 4 classes (A, B, C, and NA, where A is best condition and NA is nonattainment). The same criteria are applied to all sites (Davies and Tsomides 2002).

North Carolina.—The collection method used by North Carolina Department of Environment and Natural Resources (NC DENR) depends on the location and type of

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habitat. We limited our analyses to samples collected between June and September with the NC DENR full-scale collection method, which calls for 2 kick samples, 3 sweep samples, 1 leaf-pack sample, 2 rock- or log-wash samples collected in a fine-mesh sieve, 1 sand sample, and visual collections (NCDENR 2006). Macroinvertebrate abundance is rated as rare, common, or abundant. Site condition is rated based on EPT taxon richness and the HBI (Hilsenhoff 1987) modified for application in North Carolina (Lenat 1993). Typically, taxa are assigned pollution-tolerance values ranging from 1 (most sensitive) to 10 (most tolerant). Sites in North Carolina are assigned to 1 of 5 condition classes: excellent (5), good (4), good/fair (3), fair (2), or poor (1). Different scoring criteria are applied in each major ecoregion (Blue Ridge Mountain, Piedmont, Mid-Atlantic Coastal Plain).

Sites used for analyses

From each state database, we selected reference sites with the longest-term (≥ 9 y) biological data for analysis of long-term trends and temperature–year patterns. Our data set included 2 sites in the Wasatch and Uinta Mountain ecoregion in Utah (UT-1 and UT-4) and 2 sites in the Colorado Plateau ecoregion in Utah (UT-2 and UT-3), 3 sites in the Laurentian Plains and Hills ecoregion in Maine (ME-1, ME-2, and ME-3), and 1 site in the Blue Ridge Mountain ecoregion in North Carolina (NC-1) (Table 1). We used 3 additional reference sites in North Carolina (NC-2 to 4, Table 1) with slightly shorter data records (7 y) to assess the potential effects of climate responses on station condition assessments. These sites were designated by the respective state agencies as reference (least-disturbed, best-available) sites. We focused on reference sites to minimize possible

Table 1. Characteristics of long-term reference sites in Maine, Utah, and North Carolina. Percent urban and agricultural land use was calculated for a 1-km-wide buffer around each site (NLCD 2001). Years of data was based on the subset of years for which samples were collected in the same season with similar methods at a site. W. B. = West Branch, Mnts = mountains.

State	Site code	Water body	Latitude (°N)	Longitude (°W)	Level III ecoregion	Stream order	Elevation (m)	Drainage area (km^2)	Years of data	Land use % urban	Land use % agriculture
Maine	ME-1	Sheepscot	44.22319	69.59334	Laurentian Plains and Hills	4	31.6	362.8	22	16.4	23.0
	ME-2	W. B. Sheepscot	44.36791	69.53129	Laurentian Plains and Hills	3	70.1	38.1	12	9.1	18.5
	ME-3	Duck Brook	44.39340	68.23461	Laurentian Plains and Hills	1	54.6	12.8	9	15.9	0.0
Utah	UT-1	Weber	40.75294	111.37358	Wasatch and Uinta Mtns	5	1846.6	740.7	17	4.5	21.1
	UT-2	Virgin	37.28483	112.94808	Colorado Plateau	4	1369.2	756.3	14	3.4	0.5
	UT-3	Duchesne	40.46139	110.83000	Colorado Plateau	4	2123.5	489.5	12	4.8	10.3
	UT-4	Beaver	38.28000	112.56711	Wasatch and Uinta Mtns	4	1904.8	236.2	9	3.9	0.0
North Carolina	NC-1	New River	36.55220	81.18330	Blue Ridge Mtns	5	713.6	835.0	11	25	13.4

influence of other anthropogenic stressors. However, the distribution of land uses within a 1-km buffer zone around each site suggested that anthropogenic influences, indicated by % urban and % agricultural land use, sometimes exceeded what might be ideal for a reference characterization (Table 1). Land use was ~16% urban at 2 sites in Maine and ~23% agricultural at 1 of these sites. Land use was ~3 to 5% urban at the 4 Utah sites, but was 21% agricultural at 1 site. Land use was 12 to 25% urban at the North Carolina reference sites, with ~13% agricultural at one of these, but only 0 to 3% agricultural at the other 2 North Carolina sites.

We also used data from sites in Maine and North Carolina as case studies with which to explore the potential effects of climate change on commonly used bioassessment metrics and assessment outcomes. We used 3 additional reference sites in North Carolina (1 in the Blue Ridge Mountain ecoregion, and 2 in the Piedmont ecoregion) to analyze effects of potential range shifts of taxa in response to climate change. In Maine, we used all bioassessment stations to describe the average and range of each metric among the four station condition classes.

Data management

We screened and corrected data sets to reflect changes during the period of record in collection methods, sample processing/subsampling methods, taxonomists, and taxonomic protocols. We excluded ambiguous taxa from analyses by developing (as needed) operational taxonomic units (OTUs) (Cuffney et al. 2007). Genus-level OTUs generally were most appropriate, but some exceptions occurred (e.g., a family-level OTU was needed for Chironomidae in Utah to account for inconsistencies among taxonomic

laboratories).

We used weighted averaging or maximum likelihood inferences to assign invertebrates to temperature-preference categories in each biomonitoring database (see Stamp et al. 2010 for details). We ranked organisms based on percentiles of the distribution of temperature optima for all invertebrate taxa in each state data set. We categorized taxa with optima values <40th percentile as cold-water-preference taxa and taxa with optima values >60th percentile as warm-water-preference taxa. We modified these assignments as necessary after considering temperature-preference classifications in traits databases (Poff et al. 2006b, Vieira et al. 2006), weighted-averaging results based on data from other states in the same region, taxon distributions among warmer and colder streams in the states analyzed (USEPA 2010), literature reviews, and best professional judgment from the regional advisory groups.

Temperature and year trend analyses

Annual point measurements of temperature made in conjunction with biological sample collections are inadequate to characterize annual average temperature regime, categorize hottest and coldest years, or analyze long-term temperature trends. We used Parameter-elevation Regressions on Independent Slopes Model (PRISM) annual average maximum and minimum air-temperature data (PRISM Climate Group, Oregon State University, Corvallis, Oregon; http://www.prismclimate.org) to supplement the limited water-temperature data available in the state data sets. The PRISM model uses a digital elevation model and point measurements of climate data to generate estimates of annual, monthly, and event-based climatic variables. We used geographical information system

(GIS) software (ArcGIS 9.2) to obtain minimum and maximum annual site-specific air-temperature values from 1975 to 2006 (USEPA 2010). We used mean (average of maximum and minimum) annual air temperatures to analyze long-term temperature trends and to categorize years in terms of relative temperatures. Air and stream temperatures are correlated, but the magnitude and seasonal patterns of changes in stream water temperatures are likely to vary regionally because of factors such as the influence of water sources, watershed characteristics, and season (Daufresne et al. 2003, Caissie 2006). We assumed that mean air temperature was an acceptable surrogate for mean water temperature for comparison of relative temperature among years and grouped years as coldest, normal, or hottest based on PRISM annual average air temperature values for years during which the biological samples were collected (Stamp et al. 2010). Coldest years had mean annual air temperatures <25th percentile of the overall data set, normal years had temperatures between the 25th and 75th percentiles, and hottest years had temperatures >75th percentile values.

Responses of commonly used metrics

The HBI and EPT metrics (e.g., relative abundance or richness of EPT taxa, relative abundance or richness of taxa within the EPT) are used commonly in bioassessment indices. For example, in Maine, 8 of the input metrics used in the discriminant models are related to EPT taxa and 1 is the HBI. In North Carolina, only EPT richness and the HBI are used in an MMI to classify site condition. Utah recently adopted use of a RIVPACS predictive model to assess site condition, but most other southwestern states currently use MMIs. Several southwestern states, including Idaho,

New Mexico, Colorado, Nevada, Wyoming, Montana, and Arizona, incorporate richness or relative abundance of EPT taxa, Ephemeroptera taxa, Plecoptera taxa, or Trichoptera taxa in their MMIs. The HBI also is used in several southwestern states.

We used 1-way ANOVA to compare various EPT metrics and the HBI among hottest-, normal-, and coldest-year groups. We used Pearson product–moment correlations to test relationships among biological metrics (e.g., various EPT richness and abundance metrics, HBI values) and mean annual temperature or year. We examined correlations between HBI pollution-tolerance rankings and taxon temperature-preference optima (see Stamp et al. 2010 for details) to investigate potential vulnerability of the HBI metric to climate-change effects. We used Statistica software (version 8.0; StatSoft, Tulsa, Oklahoma) for all analyses.

MMI vulnerabilities

Maine.—Vulnerabilities of linear discriminant models to long-term temperature changes were difficult to evaluate because discriminant models test multiple variables simultaneously. Therefore, extrapolating the effect of climate-change on an individual input variable to assessments of site condition is problematic. Moreover, no firm thresholds or values of individual metrics can be identified at which an assessment of condition will change. We used ANOVA to identify component metrics that were particularly influential in differentiating between site-condition classes (see USEPA 2010 for detailed results) in conjunction with tests of climate-related sensitivities of these metrics (see *Responses of commonly used metrics* above) to infer vulnerabilities of the models to climate change.

North Carolina.—Observed biological responses to climate change include shifts in geographical ranges of sensitive taxa. These shifts often involve movements to higher latitudes or elevations. One consequence of such movements is that communities at higher latitudes or altitudes tend to become more similar to communities at lower latitudes or elevations (Bonada et al. 2007a). We used the North Carolina MMI to assess potential consequences of this type of climate-change effect on site-condition classifications. In one scenario, we removed all cold-water-preference taxa from the annual data set for sites in the Blue Ridge Mountain ecoregion (on average, cold-water-preference taxa are more abundant in Blue Ridge Mountain sites than in Piedmont or Mid-Atlantic Coastal Plain sites; Table 2) and recalculated the HBI, EPT richness, and site-condition scores. In another scenario, we applied Blue Ridge Mountain scoring criteria to data from 2 Piedmont sites and evaluating the degree to which site-condition scores changed.

Modified metrics using temperature-preference traits

We modified 2 common invertebrate metrics to assess their ability to account for climate-related trends in cold- or warm-water-preference taxa separately from other stressors. We examined the ratio of cold- or warm-water-preference taxa to total invertebrate taxa richness (cold-to-total, warm-to-total) as an addition to the commonly used total invertebrate community richness metric. We also examined the ratio of cold- or warm-water-preference EPT taxa to total EPT taxa (cold-to-total EPT, warm-to-total EPT). We applied these modified metrics to the reference-site data sets from Utah,

Table 2. Differences in elevation, Parameter-elevation Regressions on Independent Slopes Model (PRISM) mean annual air temperature, and mean (±1 SD) richness and relative abundance of cold- and warm-water-preference taxa among selected level III ecoregions in each state.

State	Ecoregion	Elevation (m)	Air temperature (°C)	Richness		Relative abundance	
				Cold-water-preference	Warm-water-preference	Cold-water-preference	Warm-water-preference
Maine	Laurentian Plains and Hills	65.2	6.5	1.1 ± 1.4	4.7 ± 3.3	2.8 ± 6.6	22.4 ± 22.0
Utah	Colorado Plateau	1729.4	9.1	3.8 ± 2.8	1.2 ± 1.2	9.8 ± 11.5	6.1 ± 11.6
	Wasatch and Uinta Mtns	2131.1	5.4	5.5 ± 4.0	1.0 ± 1.3	13.1 ± 15.4	3.8 ± 11.0
North Carolina	Piedmont	183.5	15.0	1.5 ± 2.0	5.2 ± 3.1	1.8 ± 2.7	6.7 ± 4.7
	Blue Ridge Mountains	714.5	12.1	8.0 ± 4.5	2.8 ± 2.4	11.4 ± 7.9	3.1 ± 3.7

Maine, and North Carolina. We used 1-way ANOVA to compare these modified metrics among hottest-, normal-, and coldest-year groups.

Results

Temperature and year trend analyses

At sites UT-1 and UT-2, richness of total, EPT, Ephemeroptera, and Plecoptera taxa was significantly lower in the hottest- than in the coldest-year group (Table 3; USEPA 2010). The linear relationship between EPT richness and temperature can be used to infer a loss rate of ~3 EPT taxa for every 1.0°C increase in air temperature in the Wasatch and Uinta Mountain ecoregion (Fig. 1A). The median number of EPT taxa at site UT-1 was ~13 to 14 taxa. Based on a projected temperature increase of 2°C over the next 40 y (i.e., by 2050; National Center for Atmospheric Research website: http://rcpm.ucar.edu), an average of 6 taxa could be lost (>40% of total EPT richness). The inferred loss rate (~1.5 EPT taxa/1.0°C) was lower at site UT-2, which is at a lower elevation than site UT-1 (Fig. 1B). At site ME-1, total richness and EPT richness did not differ among hottest-, coldest-, or normal-year groups. This site is in the Laurentian Hills and Plains, with a relatively low elevation and has few cold-water-preference taxa. At the shorter duration reference station in the Maine Northeast Highlands (ME-2), EPT taxa richness was significantly positively correlated with temperature; however, the trend with year was not significant (USEPA 2010). The remaining bioassessment data records did not show significant trends in EPT taxa over time or with temperature (USEPA 2010).

Table 3. Results of analyses of variance testing for differences in standard and modified metrics among hottest-, coldest-, and normal-temperature years. Results are presented as values for hottest-year groups relative to values in coldest- or normal-year groups. + = significantly higher ($p < 0.05$) and – = significantly lower ($p < 0.05$) in hottest- than in coldest- or normal-year groups. NS = no significant difference among groups, * indicates no analysis because no warm-water-preference taxa occurred at that site. EPT = Ephemeroptera, Plecoptera, Trichoptera, cold-to-total = ratio of cold-water-preference taxa to total taxa richness, warm-to-total = ratio of warm-water-preference taxa to total taxa richness.

Site	Cold-water-preference taxa		Warm-water-preference taxa		Standard metrics		Modified richness metric		Modified EPT metric	
	Richness	Relative abundance	Richness	Relative abundance	Total taxa	Total EPT taxa	Cold-to-total	Warm-to-total	Cold-to-total	Warm-to-total
ME-1	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
ME-2	NS	–	+	NS	NS	+	NS	NS	NS	–
ME-3	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
UT-1	–	NS	NS	NS	–	–	–	NS	NS	NS
UT-2	–	–	+	NS	–	–	–	+	–	+
UT-3	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
UT-4	NS	NS	*	*	NS	NS	NS	*	NS	*
NC-1	NS	NS	NS	NS	NS	NS	NS	NS	NS	NS
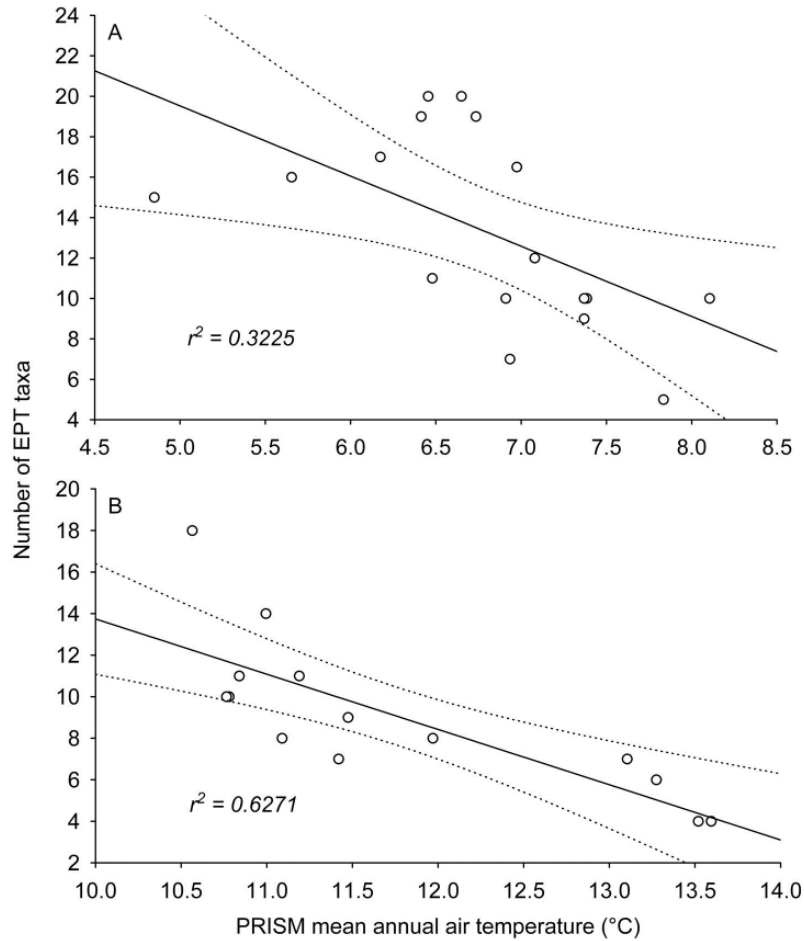
Fig. 1. Correlations of Ephemeroptera, Plecoptera, Trichoptera (EPT) taxa richness (EPT taxa) with mean annual Parameter-elevation Regressions on Independent Slopes Model (PRISM) air temperature at Wasatch and Uinta Mountain long-term reference site UT-1 ($r = 0.5679$, r2 = 0.3225, p = 0.0174) (A) and Colorado Plateau long-term reference site UT-2 ($r = 0.7919$, r2 = 0.6271, p = 0.0007) (B). Dashed curves indicate 95% confidence intervals.

The correlations between temperature-preference optima and HBI tolerance values were statistically significant but weak (Maine: $r = 0.29$, $p = 0.0013$; North Carolina: $r = 0.53$, $p = 0.000$; Utah: $r = 0.2851$, $p = 0.0034$). Except for the chironomids *Larsia* and *Natarsia*, most cold-water-preference taxa in Maine had low ($\leq$3) HBI tolerance values. However, warm-water-preference taxa in Maine had a mix of HBI tolerance values (9 had values $\geq$7, 10 had values $\leq$3). In North Carolina, most (22 of 30)

of the cold-water-preference taxa had low tolerance values (<3). Only one cold-water-preference taxon (the chironomid *Diamesa*) had a tolerance value >7. In contrast, 12 of the warm-water-preference taxa had tolerance values >7, and only one warm-water-preference taxa, *Chimarra*, had a tolerance value <3.

Based on this information alone, a loss of cold-water-preference taxa and an increase in warm-water-preference taxa probably would result in higher HBI scores, which would contribute to lower site-condition classifications. For example, in North Carolina, an increase in the HBI score of 0.1 would reduce the classification of an excellent site from 5 to 4. At lower-quality sites (score ≤ 4), an increase in the NCBI score of 0.6 would reduce the classification 1 full level.

Responses of commonly used metrics and MMI vulnerabilities

Maine.—Many of the discriminant model input metrics were related to EPT taxa and were influential in defining site-condition classifications. On average, higher values for the EPT richness metric occurred at A-quality sites than at sites in other condition classes (Fig. 2A). We explored the mix of cold- and warm-water-preference taxa within these EPT and related metric groups to understand potential vulnerability of these metrics to climate change. In Maine, 28 of 39 cold-water-preference taxa are EPT taxa, whereas 18 of 40 warm-water-preference taxa are EPT taxa (Appendix). Richness of EPT taxa was higher during warm years (Fig. 2B), in part because a large number of EPT taxa were warm-water-preference taxa. The difference between A-quality and B-quality sites in the mean number of EPT taxa (2–3 taxa) was well within the range of difference between coldest and hottest years in the mean number of EPT taxa (Fig. 2A, B). Thus, increases in

the number of warm-water-preference EPT taxa as temperature increases with climate change could result in an apparent improvement in site-condition classifications.
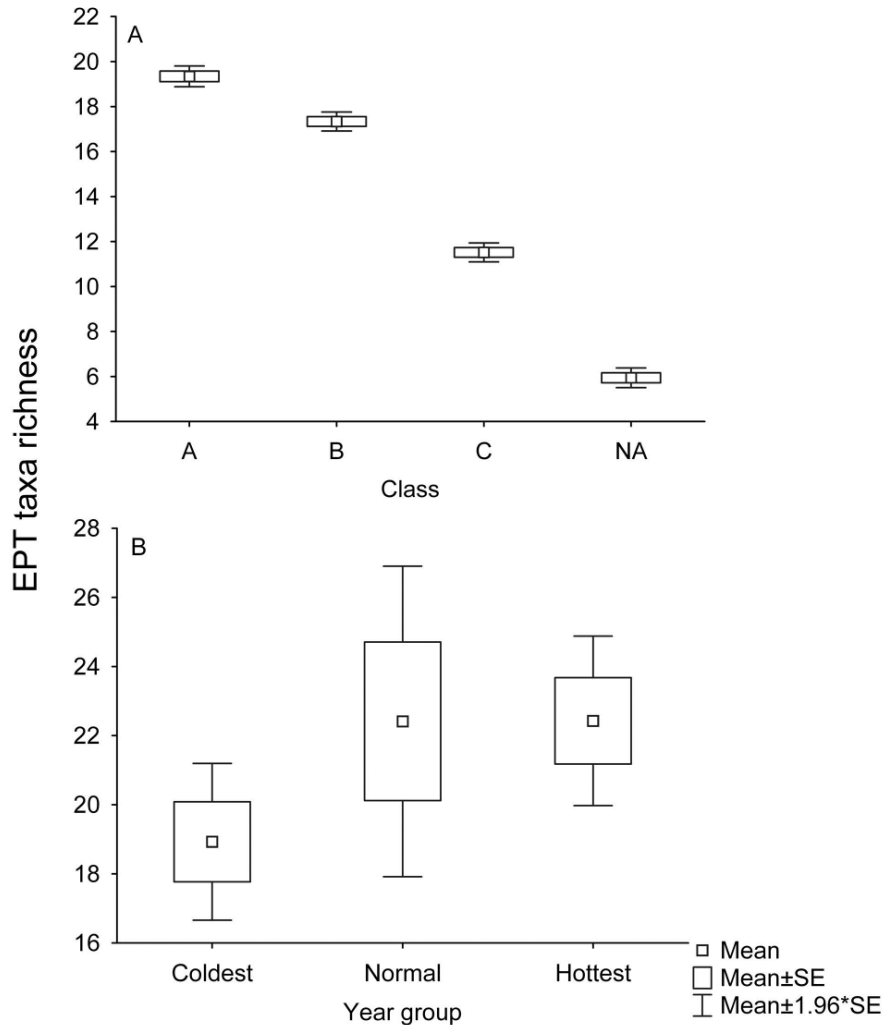
Fig. 2. Box plots of the Maine Ephemeroptera, Plecoptera, Trichoptera (EPT) generic taxon richness metric averaged among site-condition classes (A is best, NA is nonattainment) (A) and among coldest-, normal-, and hottest-year groups at site ME-1 (B).

Two of the model input metrics used by Maine were related specifically to Ephemeroptera (abundance and relative abundance). More warm-water-preference Ephemeroptera taxa (9) than cold-water-preference Ephemeroptera (3) occurred in Maine. Mean values of the Ephemeroptera abundance metric were highest at B-quality

sites and lower at both A- and C-quality sites (Fig. 3A). Thus, the site-condition

classification of a C-quality site might improve, whereas that of an A-quality site might

degrade if the abundance of warm-water-preference Ephemeroptera increased consequent

to climate change. However, the relative abundance (percentage composition) of

Ephemeroptera was greatest at A-quality sites and decreased with decreasing site

condition (Fig. 3B). The influence of increases in warm-water-preference Ephemeroptera

due to climate change on this metric will depend on the net responses of other warm- and

cold-water-preference taxa.
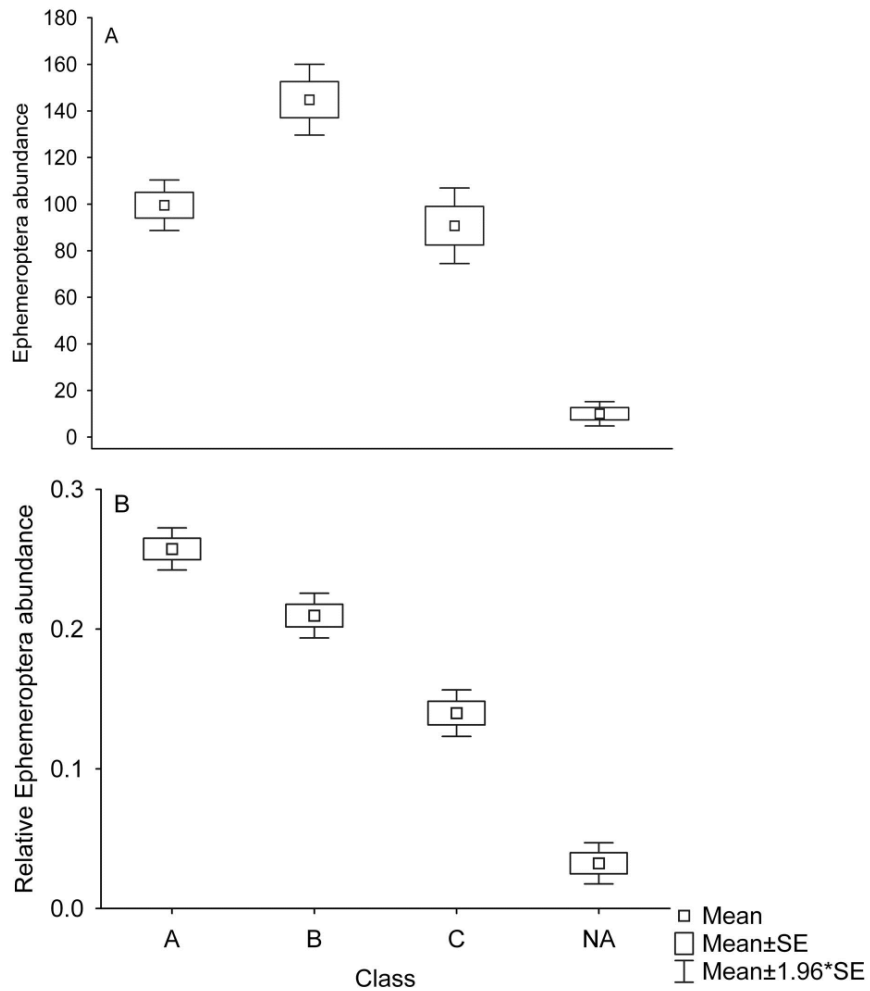
Fig. 3. Box plots of Ephemeroptera abundance (A) and relative abundance (B) averaged among site-condition classes (A is best, NA is nonattainment) in Maine.

Cold-water-preference taxa like Plecoptera also were expected to be sensitive to climate change. Three Plecoptera metrics (Plecoptera abundance, Perlidae abundance, and relative Plecoptera richness) were used as inputs to the Maine discriminant model. Highest Plecoptera abundances or richness occur at A-quality sites, and site-condition classifications decreased as values of Plecoptera metrics decreased (USEPA 2010). Many more Plecoptera taxa are cold- than warm-water-preference taxa in Maine (Appendix), but Plecoptera metrics were not correlated with temperature and did not differ among hottest-, normal-, or coldest-year groups (USEPA 2010. Thus, changes are not expected in Plecoptera metrics in response to climate change.

Two model input metrics related to Trichoptera, *Hydropsyche* abundance and *Cheumatopsyche* abundance, were not correlated with temperature and did not differ among hottest-, normal-, or coldest-year groups (USEPA 2010). Neither taxon was a cold- or warm-water-preference taxa in Maine and neither is viewed as particularly sensitive to temperature. Thus, these taxa are likely to be resilient to climate change.

Many Diptera occurred on both cold-water- and warm-water-preference lists (Appendix). Seven of 39 cold-water-preference taxa were Diptera (Chironomidae), and 10 of 40 warm-water-preference taxa are Diptera (Appendix). In the discriminant model, high abundance or richness of Diptera taxa tends to cause a low site-condition classification, even though several Diptera are classified as cold-water-preference taxa (USEPA 2010). Cold- and warm-water-preference Diptera are expected to respond differently to climate change. Thus, the effects on Maine model outcomes are likely to be variable and unpredictable and might depend on whether cold-water-preference taxa are

replaced by warm-water-preference taxa.

North Carolina.—The North Carolina MMI is composed of an EPT richness metric and the North Carolina HBI. Twenty of 32 cold-water-preference taxa (genus-level OTUs) in North Carolina are EPT taxa, whereas only 5 EPT taxa in North Carolina are warm-water-preference taxa (Appendix). Removal of cold-water-preference EPT taxa from the data set for 1 Blue Ridge Mountain reference site (NC-1) resulted in the loss of 1 to 4 EPT taxa and a reduction in the EPT richness score of up to 0.6 (Fig. 4A), where each unit score represents the difference between a site-condition classification (e.g., excellent to good, good to fair). At a 2nd Blue Ridge Mountain reference site (NC-2), 10 to 14 EPT taxa were lost when cold-water-preference taxa were removed, and the EPT richness score decreased by 0.4 to 1.2 (Fig. 4B). A loss of 3 (Mid-Atlantic Coastal Plain ecoregion) or 4 (Blue Ridge Mountain or Piedmont sites) EPT species from high-quality sites would lower the EPT richness score from 5 (excellent) to 4 (good) (NCDENR 2006). A loss of 10 EPT taxa at Blue Ridge Mountain sites, 8 taxa at Piedmont sites, or 7 at Mid-Atlantic Coastal Plain sites would be needed to decrease EPT richness scores by 1 level at sites currently rated good or lower (NCDENR 2006).
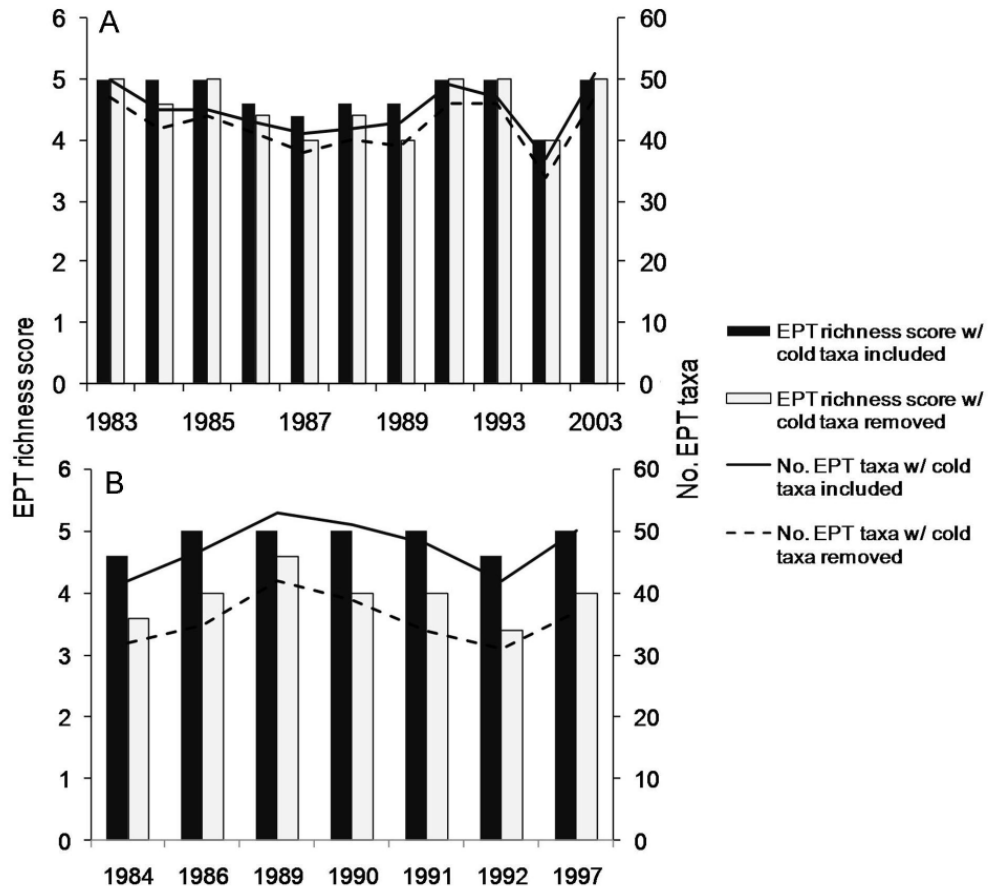
Fig. 4. Ephemeroptera, Plecoptera, Trichoptera (EPT) richness scores before and after all cold-water-preference taxa (cold taxa) were removed from the data sets for reference sites NC-1 (A) and NC-2 (B). w/ = with.

Removal of cold-water-preference taxa from the data sets for these 2 North Carolina Blue Ridge Mountain reference sites resulted in increases in HBI values from 0.03 to 0.24 at NC-1, and from 0.58 to 0.86 at NC-2; this corresponded to decreases in HBI scores of up to 0.2 and 1, respectively (Fig. 5A, B). Cold-water-preference taxa were less abundant at NC-1 than at NC-2. In combination, the changes in EPT richness and HBI metric scores that we calculated from removal of cold-water-preference taxa resulted in no net change in site-condition classification in some years, up to a change in one level (from excellent to good) in 3 of 11 y at NC-1, and in 5 of 7 y at NC-2 (Fig. 6A, B).
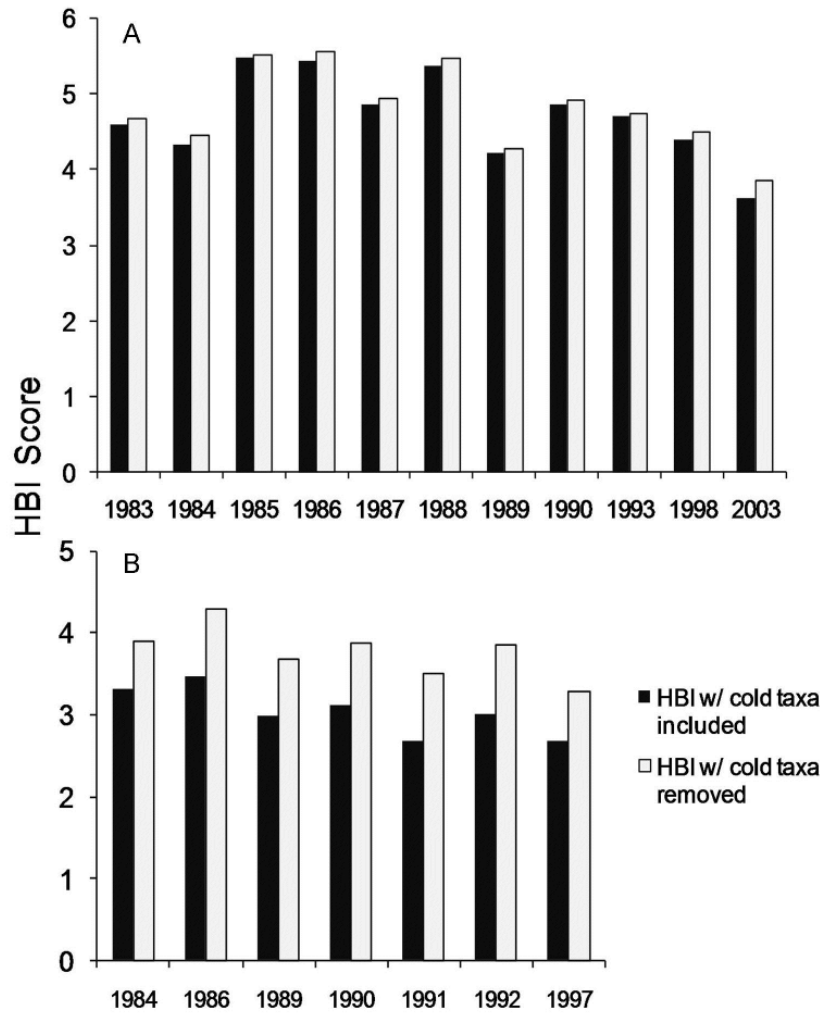
Fig. 5. Hilsenhoff Biotic Index (HBI) scores before and after all cold-water-preference taxa (cold taxa) were removed from the data sets for reference sites NC-1 (A) and NC-2 (B). w/ = with.
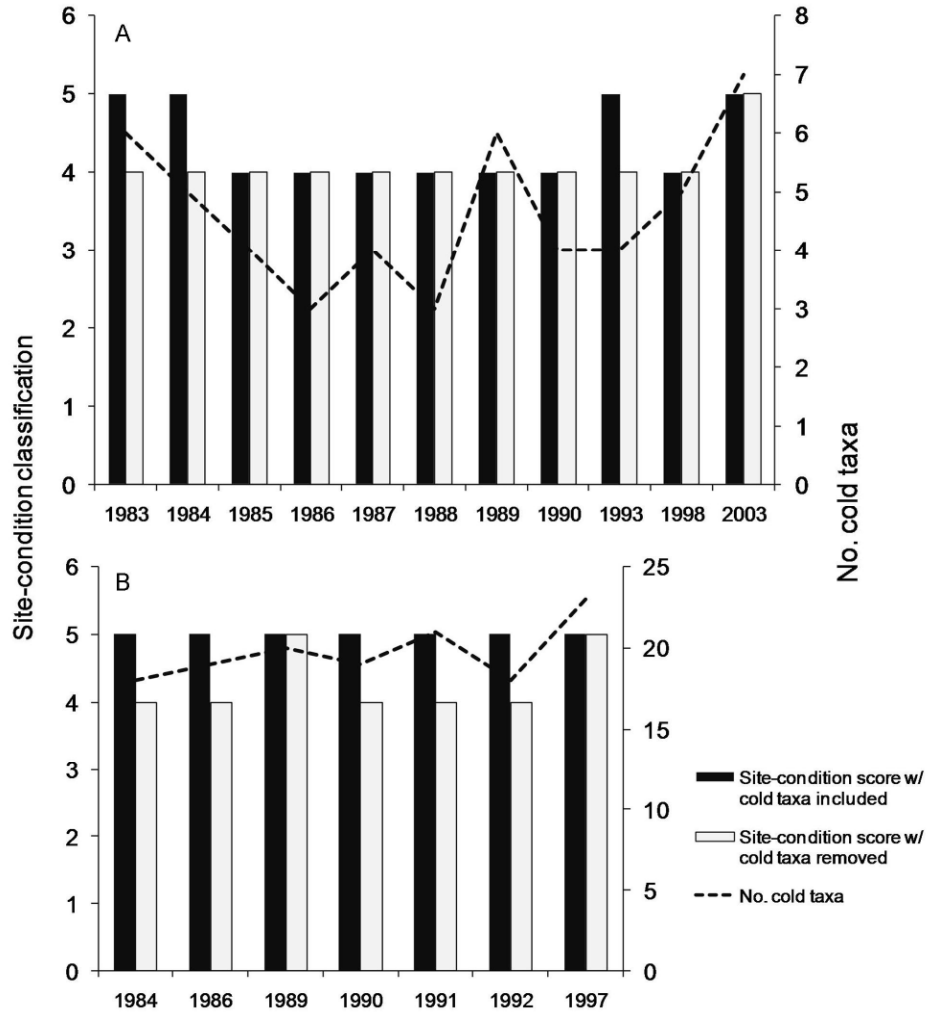
Fig. 6. Final site-condition classification scores before and after all cold-water-preference taxa (cold taxa) were removed from the data sets for reference sites NC-1 (A) and NC-2 (B). w/ = with, Mountain = Blue Ridge Mountain.

Blue Ridge Mountain-ecoregion scoring criteria were applied to Piedmont-ecoregion reference sites (equivalent to replacing the Mountain ecoregion taxa with the Piedmont ecoregion taxa) as an approximation of the outcome of range shifts of sensitive taxa. In the most extreme case (i.e., complete community replacement), site-condition classification of 1 Blue Ridge Mountain site decreased 1 level (from 5 to 4; Fig. 7A, B).
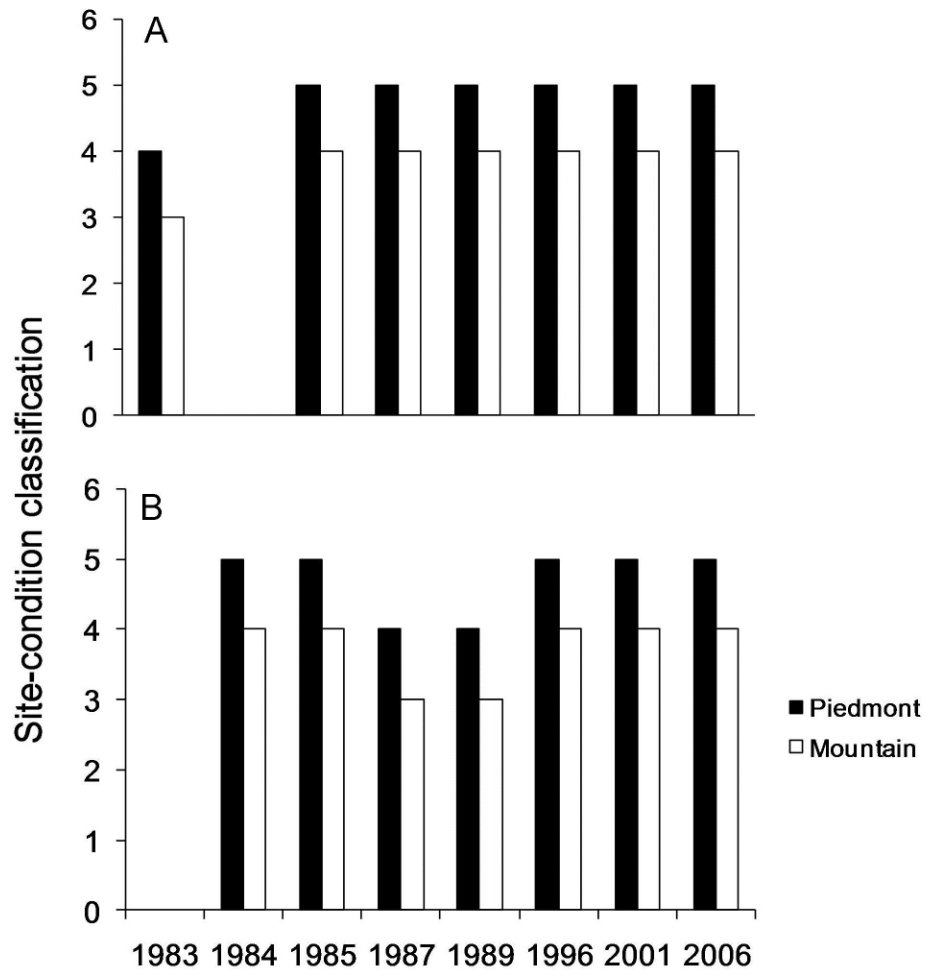
Fig. 7. Final site-condition classifications at 2 reference sites, NC0075 (A) and NC0248 (B), in the Piedmont ecoregion of North Carolina when Piedmont and Blue Ridge Mountain (Mountain) scoring criteria were applied to the site data sets.

Modified metrics using temperature-preference traits

At site ME-1, the cold-to-total and warm-to-total ratios did not differ significantly among coldest, hottest, and normal-temperature years for reference-site (Table 3). The cold-to-total ratio appeared to be slightly higher during the hottest years. However, the number of cold-water-preference taxa at this site was low, and the cold-to-total ratio was so low that any apparent trend is misleading. The number of cold-water-preference EPT

taxa at this site, was too low to permit calculation of the cold-to-total EPT ratio.

At site UT-1, the cold-to-total ratio was significantly lower for the hottest-year group than for the other year groups, but the warm-to-total ratio did not differ among year groups (Table 3). The cold-to-total EPT ratio was lower and the warm-to-total EPT ratio was higher for the hottest-year group than for other year groups, but the number of cold- and warm-water-preference EPT was low, so these trends were not statistically significant. At UT-2, the cold-to-total and cold-to-total EPT ratios were significantly lower and warm-to-total and warm-to-total EPT was significantly higher for the hottest-year group than for the other year groups (Table 3).

At site NC-1, none of the modified metrics differed among year groups (Table 3).

To visualize how much of the variation in the traditional EPT richness metric could be explained by the temperature-preference modified metrics, we plotted all 3 metrics (EPT taxa richness, cold-to-total EPT, and warm-to-total EPT) by year for UT-1 and UT-2 (Fig. 8A, B). At site UT-1, EPT richness declined significantly over time (Fig. 8A). The warm-to-total EPT ratio did not change significantly over time. However, the cold-to-total EPT ratio also decreased significantly over time, and the regression slope was similar to the slope for total EPT richness. At site UT-2, total EPT richness and cold-to-total EPT declined significantly over time, but the slope of the regression for cold-to-total EPT was steeper (Fig. 8B). Moreover, the warm-to-total EPT ratio increased over time (Fig. 8B).
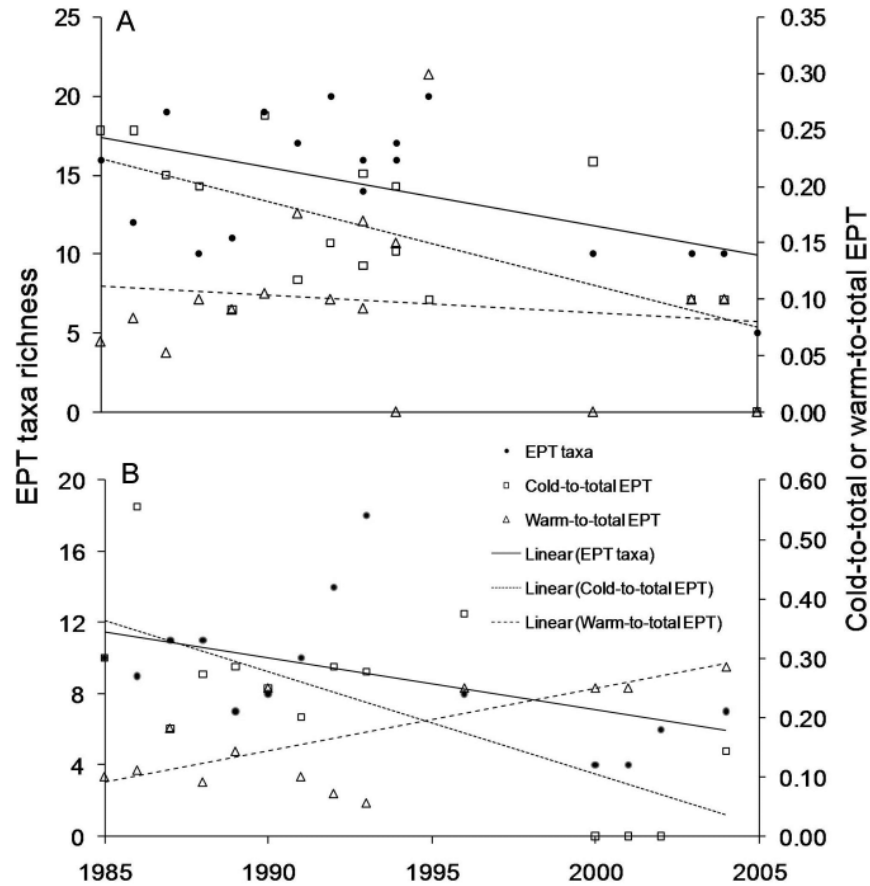
Fig. 8. Linear regressions for Ephemeroptera, Plecoptera, Trichoptera (EPT) taxon richness and the ratio of cold- or warmwater- preference EPT taxon richness to total EPT taxon richness vs year for reference sites UT-1 in the Uinta and Wasatch Mountains ecoregion (A) and UT-2 in the Colorado Plateau ecoregion (B) in Utah over the period 1985 to 2005.

Discussion

Interactive effects of ecoregional characteristics and climate change on bioassessment metrics

Evidence of invertebrate and other aquatic community responses to climate change is accumulating, but the magnitude of effects from climate change is often low compared the magnitude of effects of other large-scale spatial (e.g., land use) and temporal (e.g., the North American Oscillation [NAO]) influences (Bradley and Ormerod

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2001, Collier 2008, Chessman 2009, Sandin 2009). We found temperature-related responses of benthic indicators and metrics that are consistent with long-term climate-change effects and that could be used to establish future expectations for responses and to understand implications of these responses to bioassessment-based decisions. Responses were variable among sites and ecoregions, but they included decreases in richness or relative abundance of cold-water-preference taxa, and in some areas, increases in warm-water-preference taxa with increasing temperatures. These results are consistent with those of other studies that have reported significant increasing or decreasing trends in macroinvertebrates based on the thermophilic characteristics of the taxonomic group (Daufresne et al. 2003, Durance and Ormerod 2007, Chessman 2009). Other widespread responses among commonly used, taxonomically based metrics were declining richness with temperature or over time of Ephemeroptera, Plecoptera, EPT, and total taxa. These responses can alter reference communities to a degree that would affect decisions about site condition. Site condition could be deemed degraded or even improved simply because of taxon-specific responses to climate change.

Temperature-preference groups generally were sensitive to changing temperature conditions, but responsiveness varied among ecoregions. Regional variations in projected climate-change effects indicate that not all ecoregions are equally vulnerable to climate change (USEPA 2007, NCAR 2008, Schoof et al. 2010). Many factors can influence susceptibility to changing water temperature or hydrologic regime from climate change. These factors include elevation (Cereghino et al. 2003, Diaz et al. 2008, Chessman 2009), stream order (Minshall et al. 1985, Cereghino et al. 2003), degree of groundwater influence, or factors that affect water depth and flow rate, such as water withdrawals

(Poff 1997, Poff et al. 2006a, Chessman 2009).

Many taxonomic metrics are based on mixtures of cold- and warm-water-preference taxa, and the degree of mixing is related, in part, to ecoregional characteristics, notably elevation. In all 3 states evaluated, a greater proportion of cold-water-preference taxa occurred in higher-elevation ecoregions and a greater proportion of warm-water-preference taxa occurred in low-elevation ecoregions (Table 2). Our results suggested that elevation is one factor that drives the temperature-trait composition of regional benthic communities. In turn, the temperature-trait composition of a community affects the vulnerability of metrics and MMIs. State or tribal bioassessment managers should consider focusing efforts to evaluate and modify MMIs first in the most vulnerable, higher-elevation ecoregions.

Responses of commonly used metrics and MMI vulnerabilities

Utah.—Fairly predictable losses in EPT taxa richness (especially cold-water-preference taxa) with increasing temperatures have occurred at high- and intermediate-elevation sites in Utah. Projected EPT losses are as high as 40% by 2050. The potential effect of this loss on bioassessment capabilities is too high to ignore. Moreover, this projection of future losses was based on a linear estimate over time, but we have no reason to assume that the actual rate of taxon losses will be linear, especially given year-to-year and decadal-scale climatic variations. Thus, a linear estimate might be a poor predictor of when southwestern regional MMIs or predictive models might become incapable of differentiating reference from impaired sites.

Maine.—The Maine bioassessment protocol is based on a series of discriminant

models that require inputs of ~30 bioindicators. Therefore, other components of the process of making decisions regarding site condition are vulnerable to climate change. For example, Maine uses a group of Class-A indicator taxa as one metric for separating Class-A from Class-B condition ratings. Class-A indicator taxa are evenly divided between cold- and warm-water-preference taxa. As temperature increases, application of this metric could confound results of the models because some of the Class A indicators could decrease with increasing temperatures, whereas others could increase (USEPA 2010). In contrast, North Carolina uses only the HBI and EPT richness to classify site condition. Both of these are vulnerable to effects of climate change on temperature-sensitive taxa, but their effects on the MMI are direct and easy to understand. Thus, size of the effect ultimately realized in MMI-based site-condition classifications will, in some part, be modified by the complexity of the MMI used.

North Carolina.—The HBI is vulnerable to expected increases in water temperature because changes in the temperature-trait composition of the community will be confounded by the relationship between temperature preferences and pollution tolerance. Decreases in cold-water-preference taxa with low HBI tolerance values or increases warm-water-preference taxa with higher tolerance values will cause an increase in the HBI. Because higher HBI values impart a more-impaired site-condition classification, an increase in HBI driven by shifts in temperature-trait composition would result in a concomitant decrease in site-condition classification. This vulnerability will be stronger in regions like North Carolina where the correlation between pollution tolerance and temperature preferences of taxa was strong and consistent. In other regions, such as Maine, a more variable relationship, especially between warm-water-preference taxa and

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HBI tolerances, could lead to variability in HBI vulnerability caused by spatial differences in community composition of warm-water-preference taxa.

Modified metrics using temperature-preference traits

Cause cannot be determined from field observations or retrospective correlative analyses. Moreover, causal assessment, like risk assessment, is retrospective rather than prospective. However, causes must be considered to formulate corrective actions or to make management or regulatory decisions. Therefore, interpretation of bioassessment results often includes a process of inferring likely causes from environmental information on the area being evaluated (e.g., chemistry, land use, watershed conditions, discharges), species autecological information, and toxicological information (e.g., Beyers 1998, Suter et al. 2002). The USEPA has structured this approach into a stressor-identification process (USEPA 2000). In general, biological indicators, which are combined into MMIs, are used for their diagnostic value (Verdonschot and Moog 2006). Sensitivity to climate change and diagnostic capabilities of invertebrate indicators for this stressor have received little consideration because climate change was not considered a stressor of concern until recently. Thus, the effects of progressive changes in temperature and hydrological regimes on existing metrics and MMIs are untested.

We focused on the relative contribution of cold- and warm-water-preference taxa to particular component metrics with the intent of tracking of climate-related taxon losses or replacements. Our preliminary evaluation indicates that a temperature-modified EPT richness metric shows promise as a way to achieve this goal. Separate tracking of cold-to-total EPT and warm-to-total EPT richness metrics successfully accounted for trends in

total EPT richness over time regardless of whether changes in total EPT richness were caused by losses of cold-water-preference taxa (UT-1; Fig. 8A) or by losses of cold-water-preference taxa plus gains of warm-water-preference taxa (taxon replacements; UT-2, Fig. 8B).

Long-term increases in temperature have the potential to confuse the diagnosis of altered conditions as defined by many bioassessment metrics. For example, decreases in total richness or EPT richness, as observed at Utah reference sites, would be evaluated in relation to a conventional stressor, such as organic pollution. However, we found that decreases in cold-water-preference taxa and perhaps increases in warm-water-preference taxa caused by increasing temperatures changed total and EPT richness metrics. The magnitude of these changes was similar to the magnitude of changes caused by conventional stressors that would lead to classification of a site as impaired. The additional information provided by a temperature-modified metric could alter the way in which site condition and probable causes are interpreted. In this example, the additional information would support a shift from a presumption of pollution as the cause of the reduction in the EPT richness metric to consideration of a temperature-related effect. Without some attempt to modify traditional metrics to help characterize the contribution of climate change to changes in metrics, erroneous conclusions might be drawn, and conclusions of pollution effects or habitat degradation will be difficult to support.

We found a consistent moderate but significant relationship between temperature sensitivity and sensitivity to organic pollution, defined in the context of the HBI. A similar relationship was reported for the Ohio state biomonitoring data set (Rankin and Yoder 2009, USEPA 2010). Thus, metrics selected because their component taxa are

generally sensitive or respond to conventional pollutants (Hilsenhoff 1987, Lenat and

Penrose 1996), also will be sensitive to climate-related changes in temperature and flow

conditions. Increasing organic pollution could alter the richness or relative abundances of

cold- or warm-water-preference taxa. Partitioning the HBI metric according to the

temperature-preference classification of component taxa could provide evidence to

distinguish probable effects of climate change from effects of conventional pollution.

This approach could be used in a weight-of-evidence context, supported by

documentation of both temperature trends and pollution status at a site.

Potential effects of losses of cold-water-preference taxa on MMI-based assessments

Estimated quantitative effects on site-condition classifications of changes in

community composition caused by changes in temperature-trait composition varied. In

some cases, changes in metric values were not sufficient to affect site-condition

classifications. In other cases, they changed by 1 level (e.g., from excellent to good, good

to fair). For example, in North Carolina, simulated loss of all cold-water-preference EPT

taxa because of increasing temperatures or community replacement (mimicking

migration of a warmer-water Piedmont community into the Blue Ridge Mountain

ecoregion) reduced site-condition classifications by 1 level. Full realization of either of

these 2 scenarios is unlikely and certainly would not occur in the near term. However,

they represent an upper bound on expected vulnerability of MMIs in the near future and

illustrate the immediate importance of testing and adopting temperature-modified metrics

into bioassessment analysis frameworks.

We evaluated preliminarily the ability of a modified metric to track temperature-

related species replacements, but we have not fully explored its ability to quantify the proportion of changes caused by climate change from the proportion caused by other stressors. Such an analysis would require investigation of modified metrics at a wide variety of nonreference and reference sites, i.e., along a gradient of stressed conditions, to examine combined responses to other stressors and to climate change. Proportional changes in temperature-trait groups (using modified metrics) could be compared between nonreference (conventional stressors plus climate change) and reference (climate change only) sites to differentiate contributing causes.

The most valuable approach for incorporating modified metrics into an analytical approach might be to continue calculating the traditional metric (e.g., EPT richness, HBI), while adding new cold- and warm-water-preference metrics. In this way, proportional changes in cold- and warm-water-preference taxa could be used to assess how much of the difference in the total metric can be accounted for by changes in temperature-trait groups. This comparison could be made over time or among locations or groups of sites (e.g., reference and nonreference). This traits-based approach for detecting and tracking effects of climate change is promising (Poff et al. 2010, Stamp et al. 2010), given that few taxa (genera or species) in our study showed consistent climate-related trends across the multiple sites and states analyzed.

We tested only a temperature-modified EPT richness metric, but other climate-vulnerable and influential metrics (such as the HBI), metrics related to Ephemeroptera, Plecoptera, or Trichoptera taxa, and community diversity metrics also should be modified into new metrics that account for temperature preferences and tested within the bioassessment framework. For example, a climate-tolerant metric could be used to help

separate responses to climate change and conventional stressors. Odonata, Coleoptera, and Hemiptera (OCH) taxa have been used as a high-temperature/low-flow tolerant indicator because of their prevalence in summer conditions, higher temperatures, and lower flows (Bonada et al. 2007b). OCH taxa increased over time, with increasing temperatures, or with lower precipitation at some sites in a study related to ours (USEPA 2010), results that support potential use of OCH as a climate-tolerant metric.

Limitations

Lack of information on temperature preferences for many taxa currently limits development of climate-sensitive or climate-tolerant metrics. We used biomonitoring data to develop temperature-preference and tolerance information for many taxa common to Maine, Utah, and North Carolina (Stamp et al. 2010). Our approach could be used more broadly to support development of temperature-modified metrics in other states.

Data limitations prevent differentiation among interannual, cyclical, and long-term, directional climate changes. However, the larger issue might be how to use biomonitoring data to distinguish effects of climate change from effects of other natural and anthropogenic stressors. Durance and Ormerod (2008) discounted climate as the cause of changes in stream benthic assemblages that were correlated with long-term (18 y) temperature increases at sites in southern England. They argued that some of the faunal changes included taxa with traits (e.g., preferences for high flows and high dissolved O_2) that were contrary to expected responses to climate-driven increases in stream temperatures. However, the observed biological responses used to project biological-indicator and MMI vulnerabilities in our study were based on temperature

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traits that are mechanistically linked to expected increases in water temperature consequent to climate change. Our expectation of increasing temperature was corroborated by significant ($p < 0.05$) long-term increasing trends in air or water temperatures in several of the ecoregions evaluated and in numerous stream in the US (Kaushal et al. 2010, USEPA 2010). Many observed metric and MMI responses were consistent with temperature increases.

We used data from reference sites to minimize effects from conventional stressors so that responses of benthic indicators to climatic variables could be evaluated independently. However, several of our study sites potentially were affected by human land uses (agricultural and urban). Landuse changes in the watersheds of reference sites could have affect our long-term ability to separate climate change from other landscape-scale stressors. Intensity of development-related land use at some of our long-term reference sites has been relatively stable over the period of record (USEPA 2010). Thus, the influence of urban and agricultural land use at those sites probably has been consistent over the period of record. Nevertheless, the interactions of climate change and development pose a substantial concern for biomonitoring programs in terms of data interpretation and protection of sites in reference condition.

Few state biomonitoring programs have adopted landuse criteria for defining and selecting references sites, and no widely accepted criteria exist. However, some southeastern states (e.g., Georgia, Alabama, and South Carolina) apply landuse criteria for selection of reference sites. These criteria are <15% urban plus <20% agricultural for high-gradient streams, and <15% urban plus <30% agricultural for low-gradient streams (Barbour and Gerritsen 2006). A broad spatial analysis of the relationships among

population density, land uses, water-chemistry constituents, and benthic community characteristics in New England states led Snook et al. (2007) to associate best-available reference conditions with ≤5% urban and ≤10% agricultural land use, a result that supports an lower threshold for defining reference sites. Several of the reference sites used in our study (2 in Maine and 1 in North Carolina) had >15% urban land uses, and only a few reference sites available for our study, mostly in Utah, met the more stringent criteria. Additional study and objective documentation to support specific landuse criteria for definition of reference condition will be important for developing and adapting biomonitoring programs to incorporate climate-change detection.

The period of record of a data set might determine our ability to use it to assess effects of climate change. For example, in the Wasatch and Uinta Mountains and Colorado Plateau ecoregions in Utah, detection of significant trends appeared to be determined by the period of record of the data sets (trends detected at >14 y but not at ≤12 y). In the higher-elevation Northeast Highlands ecoregion in Maine, a period of record of 11 y was apparently too short to define significant trends despite a predominance of cold-water-preference taxa (USEPA 2010).

We rarely had >1 or 2 reference sites within an ecoregion with sufficient data to conduct long-term trends analyses, even within the relatively extensive biomonitoring data sets used in our study. For example, 105 reference sites were in the North Carolina biomonitoring data set, but long-term data was available at only 3 of them. One site was in the Blue Ridge Mountain ecoregion (11 y; Table 1), and 2 others had only 5 to 9 y of data (USEPA 2010). Regional consistency of observed responses is difficult to determine when spatial coverage and temporal replication are limited. We will need to understand

how many sites and how long a period of record are needed to detect effects of climate change before we can modify and adapt biomonitoring programs to account for climate change.

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Appendix. Number of cold- and warm-water-preference taxa in each order in each state bioassessment data set. CWP = cold-water preference, WWP = warm-water-preference.

Order	Maine		North Carolina		Utah	
	CWP	WWP	CWP	WWP	CWP	WWP
Arhynchobdellida		1		1		
Basommatophora		4		1		
Coleoptera	1	1	1	2	1	3
Decapoda		1		1		
Diptera	7	10	10	5	8	2
Dorylaimida					1	
Ephemeroptera	4	9	6	1	6	2
Haplotaxida		1				
Hemiptera				1		1
Hoplonemertea		1				
Hydroida		1				
Isopoda				1		1
Megaloptera	1			7		
Mesogastropoda		1				
Odonata	2	1	1			1
Plecoptera	14	3	8		10	1
Rhynchobdellida				2		
Trichoptera	10	6	6	4	7	5
Unionoida				1		

CHAPTER 3

Implications of global change for the maintenance of water quality and ecological integrity in the context of current water laws and environmental policies

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Abstract

There is both a fundamental and applied need to define expectations of changes in aquatic ecosystems due to global changes. It is clear that programs using biological indicators and reference-based comparisons as the foundation for assessments are likely to make increasingly erroneous decisions if the impacts of global change are ignored. Global changes influence all aspects of water resource management decisions based on comparisons to reference conditions with impacts making it increasingly problematic to

find an "undisturbed" water body to define acceptable conditions of ecological integrity. Using a more objective scale for characterizing reference conditions that is anchored in expectations for what would be attainable under undisturbed conditions, such as the Biological Condition Gradient (BCG) is one approach that maintains consistent definitions for ecosystem conditions. In addition, protection of reference stations and of unique or undisturbed aquatic resources is imperative, though the scope of protection options is limited. Projections indicate that encroaching land use will affect 36-48% of current reference surface waters by the year 2100. The interpretation of biological indicators is also at risk from global changes. Distinguishing taxonomic attributes based on temperature or hydrologic preferences can be used to enhance the ability to make inferences about global change effects compared to other stressors. Difficulties arise in categorizing unique indicators of global changes, because of similarities in some of the temperature and hydrologic effects resulting from climate change, land use changes, and water removal. In the quest for biological indicators that might be uniquely sensitive to one global stressor as an aid in recognizing probable causes of ecosystem damage, the potential similarities in indicator responses among global and landscape-scale changes needs to be recognized as a limiting factor. Many aspects of global changes are not tractable at the local to regional scales at which water quality regulations are typically managed. Our ability to implement water policies through bioassessment will require a shift in the scale of assessment, planning, and adaptations in order to fulfill our ultimate regulatory goals of preserving good water quality and ecological integrity. Providing clear expectations of effects due to global change for key species and communities in

freshwater ecosystems will help water quality programs achieve their goals under changing environmental conditions.

Key words: Global change, climate change, water law, environmental policy, bioassessment

The Underpinnings of Environmental Protection of Aquatic Resources

The expression of water laws and environmental policies is to protect the health of aquatic ecosystems. The maintenance of natural ecological processes and ecosystem function is paramount to addressing this goal. This is reflected in the U.S. Clean Water Act (CWA, 1972) long-term goal of maintaining and restoring the ecological integrity of aquatic systems, defined as comprising physical, chemical and biological integrity (CWA 1972). Similarly, the goal of the European Union Water Framework Directive (WFD), Directive 2000), is to achieve "good status" of aquatic systems, including both "good ecological status" and "good chemical status" (see Noges et al. 2009).

In principle, assessment of status is based on a combination of biological, physical, and chemical indicators; however there is recognition that biological indicators are an integrating response to all environmental conditions (Barbour et al. 2000, Moog & Chovanec 2000). Based on this, in the U.S., biological assessment plays a central role in numerous water quality programs that are components of the CWA. Bioassessment data is used to assess water quality, identify biologically impaired waters, and develop National Water Quality Inventory reports. It is used to develop biocriteria and set aquatic

life use categories, which represent different protection standards. Bioassessment data are used to determine whether conditions of the waterbody support designated uses, and if not, to develop total maximum daily load (TMDL) limitations for the pollutant(s) contributing to the impairment. Bioassessment results are used to help identify causes of observed impairments, based on the assumption that various components of aquatic communities will respond differently to different types of stressors. Bioassessment is used to determine the impacts of point source discharges as well as of episodic spills, defining the extent of damage, responses to remediations, and supporting enforcement actions. Other CWA programs that depend on bioassessment data include permit evaluation and issuance, tracking responses to restoration actions, and other components of watershed management. In Europe, the WFD emphasizes biological assessment methods supported by the evaluation of hydromorphological and chemical parameters, definition of strategies against pollution (combined approach for point and diffuse sources), and establishment of river basin management plans (Chovanec et al. 2000).

Assessing the ecological health of rivers and streams is a fundamental and increasingly important water management issue worldwide (Norris and Barbour 2009, Bunn and Davies 2000) and relates directly to the preservation of ecosystem services of aquatic resources. The concept of ecosystem services embraces those processes by which the environment produces resources that we often take for granted, such as clean water, and other services, such as mitigation of drought and floods, the cycling and transport of nutrients, the maintenance of biodiversity, and detoxification and decomposition of waste (Daily 1997, Postel & Carpenter 1997). Efforts such as the UN Millennium Ecosystem

Assessment (MEA 2005) are raising public awareness that high quality aquatic resources provide society with ecological benefits and should be valued.

On a global scale, different countries are at varying levels of success in the endeavor of protection and restoration of their water resources (Jungwirth et al. 2000). Those that have had programs in place for many years have accomplished much in determining the status of their resources and what must be done to restore degrading waters (Barbour & Paul, 2010). However, few have considered how to effectively account for large-scale, global changes, such as progressive climate change, increasing encroachment of developed and agricultural land uses, and increasing human demands for freshwater, all with ramifications to aquatic resources.

Water laws have historically focused more on point-source impacts than on landscape or larger scales of pollution or habitat degradation. During the 20th century, the focus of environmental problems has changed from local or regional issues like sewage discharge in the first decennia towards global issues like climate change today (Verdonschot 2000). This means that now, water resource management requires consideration of stressors that operate at very different scales, such as climate change, which operates on a global scale. It will alter air and water temperatures; alter flow and other hydrological parameters; reduce ice and snow cover; alter the timing of snowmelt; alter stratification regimes in lakes; increase sea levels; and increase salinity in some coastal areas (IPCC 2007). Global changes in land use and water abstraction are landscape-scale stressors that are directly related to human population increases.

An increase in the spatial scale of stressors implies a need to increase the spatial and temporal scales of management and assessment (Verdonschot 2000). This change in scale complicates the already complex management and restoration of impaired water resources. While ecosystem management will be most effective when all stressors impacting a resource are considered, certain aspects of global change impacts are not tractable for control at a local, watershed, or state agency scale.

The environmental research community is engaged in understanding the effects of global changes at a variety of spatial scales. Much of the research in the US on global change and its effects on aquatic ecosystems are being supported by federal agencies, such as the National Science Foundation (NSF), US Environmental Protection Agency (U.S. EPA) and the National Oceanic and Atmospheric Administration (NOAA). The European Union (EU) supports a series of research projects being conducted by several institutions throughout Europe. The Australian government, through the Department of Climate Change the Australian Greenhouse Office, and the Australian Climate Change Science Program, also supports substantial research on climate change effects and adaptations. As the effects of global changes become more evident and attention is drawn to this complicating factor in maintaining good ecological status, scientists and environmental managers should work together to ensure that the goals of environmental protection are met. Research partnerships and information sharing may lead to more effective resource management and decision-making tools that account for global change effects.

Global Changes and Ecosystem Health

The concept of ecological health can be vague, but is generally borrowed from the human health/medical paradigm, such that a "healthy" ecosystem, i.e., one with "integrity", is conceptually one that is functioning within "normal" ranges. We assume this is true for pristine systems, but emphasize the question of whether an acceptable level of ecological integrity exists in systems with some degradation, often termed "minimally disturbed" or "leased disturbed" (Bailey et al. 2004, Stoddard et al. 2006). Conditions observed in "minimally" or "least" disturbed streams or rivers often become the basis for defining restoration or remediation goals, or for determining impairment, in similar systems that are impacted by human uses and development.

The effects of land use alterations on river ecosystem condition have been reasonably well studied (see for instance Paul & Meyer 2001, Allan 2004, Helms et al. 2009). However, taking a global view of the future impacts of progressive land use changes on ecosystem integrity is a more recent concern. There have been many studies of a variety of water use and hydrological alteration impacts on stream ecological condition, including flow regulation (e.g., Nilsson et al. 2005), increasing human uses (e.g., Bunn & Arthington 2002), and hydrologic alteration in general (e.g., Richter et al. 2003, Konrad et al. 2008). Now there is recognition that global changes in climate combined with a growing world population and changes in global patterns of land use, socio-economics, and technical changes in water resource utilization and management will combine to increase water stress (i.e., decrease water availability and increase demand, Bates et al. 2008). Increased water stress will be particularly notable in regions where river runoff is projected to decrease in the future, as well as in regions where water resources are

dependent on snow pack (e.g., the western U.S., the Mediterranean Basin (including southern Europe, Northern Africa, and southwestern Asia), southeastern Australia, southern Africa, and the west coast of South America) (Bates et al. 2008). Progressive increases in regional water stress will put increasing demands on freshwater in likely competition with ecological uses. This will bring conflicts in water use to a head, and put greater pressure on management to consider all sources of impacts to water resources (Brekke 2009).

Increasing attention has been given to whether ecosystem responses to climate change are sufficient to cause concern and whether that concern should extend to our ability to preserve and restore ecological integrity. Freshwater ecosystems are considered sensitive to climate change impacts, owing to their fundamental dependence on hydrology and thermal regimes, their dominance by poikilotherms, and the risks of interactions with other stressors (Durance & Ormerod 2007). However, documentation of aquatic biological responses to climate change on a basis that is meaningful to water quality and resource managers has been slow in coming, with much early attention focused on terrestrial ecosystems (e.g., Root et al. 2003, Thuiller 2004, Walther et al. 2002, 2005; Parmesan 2006, Tobin et al. 2008, Suding et al. 2008, Zuckerberg et al. 2009). Climate change effects in aquatic systems will manifest at all ecological levels (Fig. 1). Examples of shifts in aquatic community structure that are relevant in a bioassessment framework, documented in 10 relevant references, are presented in Table 1. Many of these effects will be species and region specific; therefore, there is substantial variation in aquatic biological responses that may impact the evaluation of ecological status. There will be

potentially major consequences both for ecosystem function and for the interpretation of

biomonitoring results relative to assessment of ecosystem health.
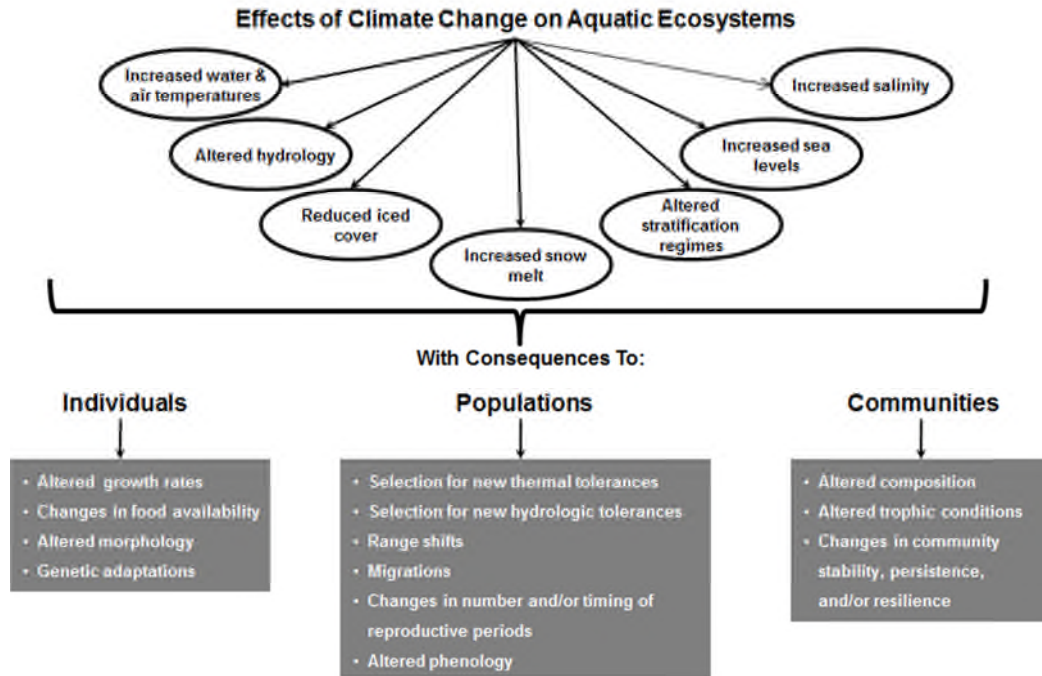
Fig. 1. Illustration of climate change effects on aquatic ecosystems and the subsequent individual-, population-, and community-level responses to these changes. Effective indicators are those that respond to the consequences of climate change.

Table 1. Documented examples of shifts in aquatic community structure due to climate change.

Examples of aquatic community changes	Reference
increases in abundance, species richness, and proportion of southern and of warm-water species of fish in large rivers	Daufresne & Boet (2007)
loss of cold-water fishes from headwater streams, but also extension of more tolerant, thermophilic fishes from larger streams and rivers into newly suitable habitat	Buisson et al. 2008
increases in fish species richness with increasing temperatures at higher latitudes	Hiddink & Hofstede (2008)
displacement of upstream, cold-water invertebrate taxa with downsteam, warm-water taxa	Daufresne et al. (2004)
an increase in lentic and thermophilic invertebrates with increasing temperature	Doledec et al. (1996)
reductions of spring abundance of dominant taxa, shifts in invertebrate assemblage composition from cooler to warmer water taxa, and possible losses (local extinctions) of more scarce taxa with increasing temperatures	Durance & Ormerod (2007)
significant long-term trends related to the thermophily and rheophily of benthic taxa, with groups preferring cold waters and higher flows declining	Chessman (2009)
changes in stability and persistence	Collier (2008)
changes in species composition in lakes	Burgmer et al. (2007)
changes in structure and diversity of riverine mollusk communities with reduction in community resilience during hot years	Mouthon & Daufresne (2006)

There are some similar and confounding effects among global changes that have ramifications to the implementation of water policy. For example, stream temperatures can be increased by climate change, and also by land uses (e.g., urbanization, deforestation) (Table 2). Base flows can be reduced by climate change, and also by water abstraction. Though the scale of these effects can be different, their similarities can make it difficult to separate climate change impacts from those of other landscape-scale stressors within management-relevant time frames. Climate change effects are also compounded by other large-scale climate drivers such as the North Atlantic Oscillation (NAO) (Bradley & Ormerod 2001). The presence of development and water abstraction or diversion infrastructure can be documented independently, e.g., by quantification of watershed land cover or water withdrawal data, which can help define contributing

factors in a weight of evidence framework. However, climate change effects are

pervasive, and if other global change influences are present, quantitative partitioning of

biological responses among these factors will be problematic in many circumstances. In

the quest for biological indicators that might be uniquely sensitive to one global stressor

as an aid in recognizing probable causes of ecosystem damage, the potential similarities

in indicator responses among global and landscape-scale changes should be recognized as

a limiting factor.

Table 2. Examples of temperature and hydrologic effects resulting from climate change, land use changes, and water removal.

Effects on:	Climate change	Land Use change	Water abstraction/ diversion
Water temperature	Increases on a global scale with regional differences, modified to some extent by local to regional variations in vulnerability; superimposed on year-to-year variations.	Local to watershed scale increases due to altered land cover (e.g., runoff from impervious surfaces) and altered channel morphology; possible decreases due to reforestation, restoration of riparian cover.	Local to watershed scale increases due to reduced flows, altered channel morphology; variable temperature effects related to dam releases.
Hydrology	Regionally variable, often including reduced annual discharge, reduced summer flows and baseflow, increased flow variability, increased flooding episodes, increased flashiness; some streams changing from perennial to intermittent flows.	Decreased infiltration, increased surface runoff, altered flood runoff patterns, higher peak flood flows, reduced groundwater recharge and reduced baseflow.	Reduced discharge, flow displacement, altered timing and magnitude of peak and base flows, altered flow variability.

Susceptible Aspects of Assessment Programs to Global Changes

In the framework of water laws and environmental policy aimed at protecting and

restoring the condition of aquatic ecosystems, bioassessments result in information on the

ecological health of a water body (Norris and Barbour 2009). Ecological indicators serve

as easily interpretable surrogates to gauge condition of an aquatic ecosystem (Niemi and

McDonald 2004). They can also be early warnings of degradation that link appropriately to the stressors expected in the system, which now include global changes in climate, land use, and water use.

The effects of global change on bioassessment programs will vary regionally. Land and water use effects are largely driven by locations of and projected future changes in major population and agricultural centers. Differences in climate change threats and responsiveness are instead driven by regional variability in climate, as well as regional differences in the vulnerability of aquatic ecosystems. Differences in regional climate and disturbance regimes are important contributors to species sensitivities to environmental changes (Helmuth et al. 2006). Many factors can influence susceptibility to changing water temperature or hydrologic regime due to climate change, such as elevation (Cereghino et al. 2003, Diaz et al. 2008, Chessman 2009), stream order (Minshall et al. 1985, Cereghino et al. 2003), degree of ground water influence, or factors that affect water depth and flow rate, such as water withdrawals (Poff 1997, Poff et al. 2006a, Chessman 2009).

Essential components of the assessment and management of ecological health include the reference condition paradigm and the use of biological indicators. In a regulatory context, impairment represents a level of departure from defined reference conditions considered unacceptable for maintenance of ecological integrity. However, the continued good status of reference locations, and therefore their use as a basis for comparison, is significantly threatened by global change. Both climate change effects and encroachment of development can be expected to impact streams that were previously categorized as minimally altered, probably more so than previously conceived or documented. We have

found that in some examples from state biomonitoring data in the U.S. losses of cold-preference taxa and replacement by warm-preference taxa had the potential to degrade the condition classification of reference stations by a full level (e.g., from "excellent" to "good", or "good" to "fair"), as defined by the particular state bioassessment schema (Fig. 2) (USEPA 2010).
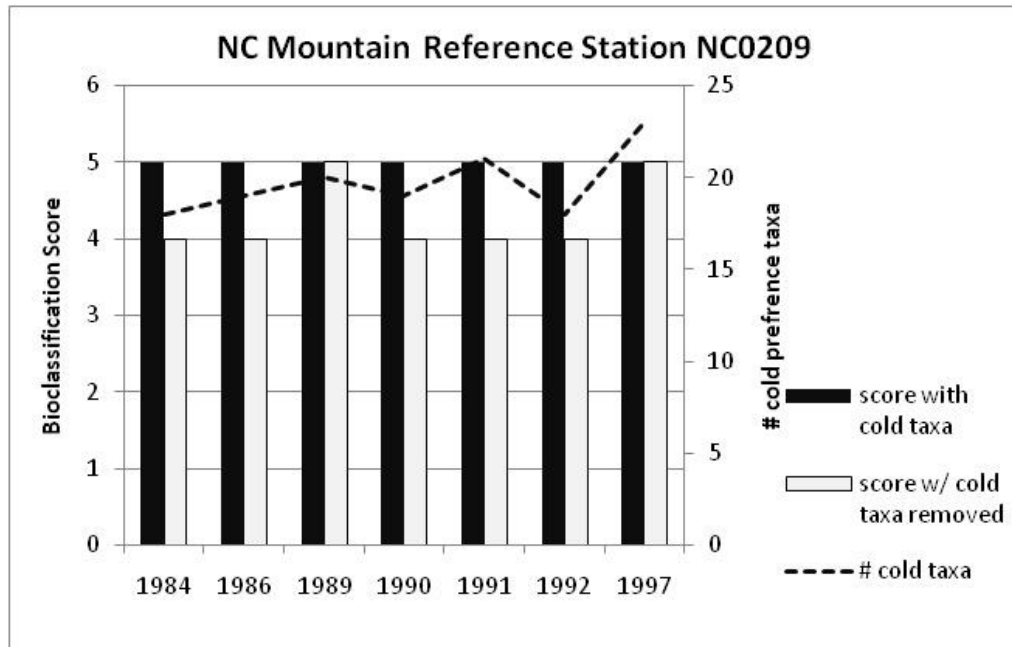
Fig. 2. Final bioclassification (station quality) scores at a North Carolina reference site in the Blue Ridge Mountain ecoregion (NC0209) before and after all cold-preference taxa are eliminated from the observed benthic invertebrate community.

Land use impacts on reference stations have long been considered a factor that can be controlled in the process of selecting sampling stations. However, finding undisturbed reference conditions is already a challenge in the U.S. and Europe (e.g., Herlihy et al. 2008, Noges et al. 2009), and often the only comparisons for assessment are to the "best of what is left", or least disturbed conditions (Stoddard et al. 2006). For state

biomonitoring programs in the U.S., it is often the case that the extent of developed land

uses affecting established reference locations is not quantitatively documented, nor

updated over time. We have found that the influence of urban/suburban land uses on

established reference locations can be greater than previously assumed. In one example,

20-25% of reference sites classified as high quality in Florida were actually surrounded

by >20% urban/suburban land uses (Fig. 3). Furthermore, using spatially explicit

projections of population growth, it was estimated that by the year 2100, from 36% to

48% of current reference locations could be compromised by encroachment of developed

land uses (Fig. 3) (USEPA 2010).
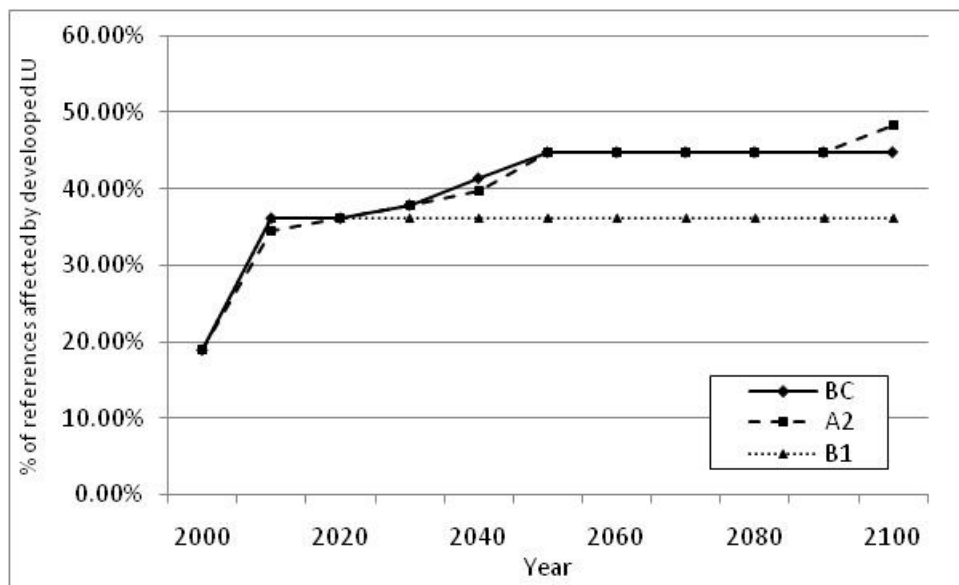
Fig. 3. Percent of existing Florida reference stations (N=58, classified as "exceptional"), that have >20% developed land use (with 25 houses per square mile or more) within a 1-km buffer surrounding the station; for current land use conditions, and for projected developed land use distributions consistent with IPCC (2000) base case, A2 and B1 scenarios, projected for decadal time periods through 2100 (from U.S.EPA 2010).

The combined effects of climate change and land use encroachment are thus reducing availability of reference locations for comparative analyses, and definition of the reference baseline will continue to shift toward more degraded conditions ("reference station drift", Fig. 4). Noges et al. (2007) suggest that in Europe, climate change will alter reference conditions enough to potentially alter typologies (classifications) and impact restoration targets. They point to the need for periodic re-evaluation and adjustment of reference conditions and associated standards to accommodate climate change effects that cannot be mitigated. With regard to the Clean Water Act (CWA), this might be interpreted as a lowering of standards in conflict with anti-degradation policies.
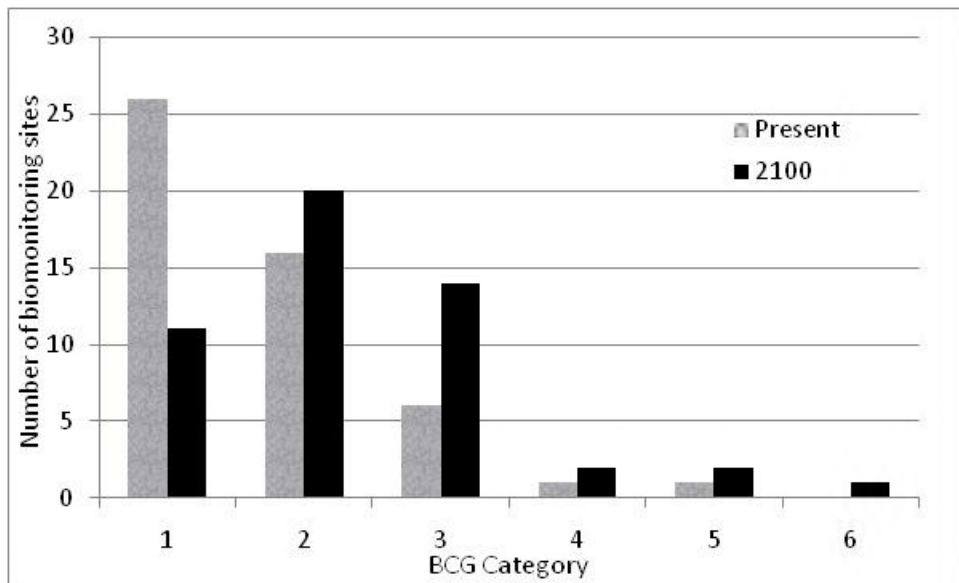
Fig. 4. Hypothetical example of reference station drift over time as climate change degrades the condition of sites, based on classification of stations along a Biological Condition Gradient (BCG).

The influence of global change thus poses a serious risk to management decisions regarding environmental protection and restoration. Unaccounted climate change

impacts, especially when further compromised by increasing influence of developed or

agricultural land uses, are likely to lead to fewer determinations of impairment and

listings of impaired stream reaches. This is expected because biological responses to

climate change, such as decreases in mean abundances or species richness of cold-

preference or other sensitive taxa and trait groups, increases in warm-preference or other

tolerant taxa and groups (U.S.EPA 2010), and increases in the variability of these

indicators, drive reference sites to greater similarity with non-reference areas, as well as

greater difficulty in establishing statistical differentiation (USEPA, 2008). Overall,

progressive under-protection of water resources can be expected.

The interpretation of biological indicators and metrics is also at risk from global

changes. Several traditional, taxonomically-based biological metrics, including total

species richness; Ephemeroptera (mayflies)- Plecoptera (stoneflies)- Trichoptera

(caddisflies)- (EPT) abundance and richness metrics; and Hilsenhoff's biotic index

(HBI), are shown to be composed of both cold- and warm-water preference taxa (U.S.

EPA 2010). The relative contribution from cold- and warm-preference taxa varies

regionally and among metrics (U.S.EPA 2010). This mixed composition of temperature

traits leads to differential responses to climate-associated increases in water temperature

over time, and therefore, the vulnerability of that metric to climate change effects. If a

metric like EPT richness is evaluated in a vulnerable location where cold-preference taxa

are declining over time, the EPT metric would decline with the potential to alter the

ecological status rating of that location. This could happen due to climate change in the

absence of any other impairment to the water resource conditions at that location. In

another example, we have found a widespread, moderate but significant relationship

between temperature sensitivity and sensitivity to organic pollution (U.S. EPA 2010). Thus, metrics such as Hilsenhoff's Biotic Index (HBI), originally adopted to represent responses to organic pollution, are susceptible to changes in index values due only to climate change effects. The magnitude of biological responses due to climate change-mediated increases in temperature may be sufficient to cross thresholds of impairment and alter management decisions (U.S. EPA 2010). Without accounting for these climate responses, the traditional interpretation of changes in EPT, HBI, or other metrics could be in error and increase the uncertainty of currently employed processes of impairment characterization and interpretation of causes.

Tracking long-term responses in trait groups defined by temperature preferences is a means to determine biological responses to global changes in temperature (Poff 1997). Categorization by traits reduces variation across geographical areas that can be associated with taxonomic composition, and thus supports analyses of regional or larger-scale data sets. Such regional-scale analyses are consistent with the larger scale of global change effects. Evaluation of temperature trait groups that are subsets of traditional taxonomically-based metrics also illustrates the vulnerabilities of current bioassessment metrics.

Predictive models such as the River Invertebrate Prediction and Assessment Classification System (RIVPACS) used to assess condition and make compliance or other regulatory decisions are susceptible to responses to climate change, predominantly through the relative composition of cold- and warm-preference trait groups in the "observed communities" (O in the O/E [observed/expected] index) (U.S. EPA 2010). The predictive models for expected communities may be more robust to climate change

than individual indices, given periodic model recalibration, due to the relative stability of using long-term averages in estimating predictor variables (U.S. EPA 2010). The process of periodic recalibration may still have the undesirable effect of altering the reference baseline of comparison over the long term.

Integration of Monitoring and Assessment for Global Change into Environmental Policy

It is clear that programs using biological indicators and reference-based comparisons as the foundation for assessments are likely to make increasingly erroneous decisions, especially in the most climatically vulnerable or in increasingly populated regions and watersheds, if the impacts of global change are ignored. Conversely, programs that adapt their biological assessment framework by characterizing global change vulnerabilities and sensitive ecological traits will be in a better position to make informed decisions regarding the synergism of multiple stressors in the context of global changes. We have highlighted several critical components of the typical water quality management process that are demonstrably vulnerable to global change impacts. These include impacts to biological indicators that dissociate their responses from conventionally interpreted causes; degradation of reference conditions that will rapidly make it difficult or impossible to define desirable levels of ecological integrity; and the synergism of global changes that will alter water distribution and availability and increase the uncertainty under which flows needed to maintain ecological functioning will be maintained. If assessment approaches are not modified, the increasing loss of aquatic habitat to global

shifts in abundance and distribution of flowing waters will become institutionalized because of lack of action to deal with the implications of global change. This will introduce additional uncertainty into a system that requires knowledge of relatively predictable biological indicator responses to different types of "conventional" stressors.

The framework for a new approach needs to address these major vulnerabilities in a manner that accommodates the inherent scale differences between "global change" stressors and many conventional stressors. We recognize that many aspects of on-the-ground implementation of water policy and water quality management will continue to be at a local (e.g., stream reach) level. Local actions that are augmented with a regional (or larger) scale focus will allow bioassessment and implementation of water law and policy to incorporate considerations of global change stressors. Any supplementary monitoring or analyses at a regional scale should also provide inputs that are meaningful at the local scale.

What do we do about loss of reference conditions? Global changes influence all aspects of making water resource manage decisions that are based on comparisons to reference conditions. Clearly it will become increasingly problematic to find an "undisturbed" water body of a particular type as a way to define acceptable conditions of ecological integrity. The baseline described from existing reference locations will increasingly reflect only the best available or least disturbed among available locations, and so will reflect increasingly degraded conditions. As stated earlier, the option of periodic recalibration of reference conditions based on what is left leads directly to acceptance of widespread and institutionalized degradation of water quality conditions and calls into question how anti-degradation policies, typically included in water quality

regulations in the U.S., can be managed considering the additional influences of climate change. There needs to be a basic understanding that recalibration of reference in a downward direction should never be done. Use of a biological condition gradient approach will aid in anchoring the reference condition founded on an achievable schema (Davies and Jackson 2006). Special attention of management and regulatory agencies will be needed to consider implementation of anti-degradation policies with respect to global changes.

On the other hand, there will have to be consideration given in some circumstances to the possibility that global changes may lead to irreversible changes in habitat conditions over the long term that may irretrievably alter attainable ecological conditions and uses. For example, some cold water streams could take on cool water characteristics, with declining abundances or richness of sensitive cold water taxa and possible increases in warm-water taxa. Regulated parameters such as temperature, dissolved oxygen, and ammonia, may also be sensitive to climate change effects, and their values may need to be adjusted relative to revised designated uses (Table 3). Refinement of aquatic life uses can be applied to guard against lowering of water quality protective standards. More refined aquatic uses could create more narrowly defined categories, which could accommodate potentially "irreversible" changes, but with sufficient scope to maintain protection and support anti-degradation from regulated causes. In addition, the application of use attainability analyses (UAA) on vulnerable water bodies may be pertinent for characterizing climate impacts.

Table 3. Variables addressed in criteria and pathways through which they may
be affected by climate change (from Hamilton et al. 2009).

Criteria	Climate change impacts
Pathogens	Increased heavy precipitation and warming water temperatures may require the evaluation of potential pathogen viability, growth, and migration.
Sediments	Changing runoff patterns and more intense precipitation events will alter sediment transport by potentially increasing erosion and runoff.
Temperature	Warming water temperatures from warming air temperatures may directly threaten the thermal tolerances of temperature-sensitive aquatic life and result in the emergence of harmful algal blooms (HABs), invasion of exotic species, and habitat alteration.
Nutrients	Warming temperatures may enhance the deleterious effects of nutrients by decreasing oxygen levels (hypoxia) through eutrophication, intensified stratification, and extended growing seasons.
Chemical	Some pollutants (e.g., ammonia) are made more toxic by higher temperatures.
Biological	Climate changes such as temperature increases may impact species distribution and population abundance, especially of sensitive and cold-water species in favor of warm-tolerant species including invasive species. This could have cascading effects throughout the ecosystem.
Flow	Changing flow patterns from altered precipitation regimes is projected to increase erosion, sediment and nutrient loads, pathogen transport, and stress infrastructure. Depending on region it is also projected to change flood patterns and/or drought and associated habitat disturbance.
Salinity	Sea level rise will inundate natural and manmade systems resulting in alteration and/or loss of coastal and estuarine wetland, decreased storm buffering capacity, greater shoreline erosion, and loss of habitat of high value aquatic resources such as coral reefs and barrier islands. Salt water intrusion may also affect groundwater.
pH	Ocean pH levels have risen from increased atmospheric CO_2, resulting in deleterious effects on calcium formation of marine organisms and dependent communities and may also reverse calcification of coral skeletons.

Detecting and monitoring shifting conditions in the reference population should be
part of any water quality program. As described earlier, a more objective scale for
defining ecological condition, and thus for characterizing reference conditions, that is
anchored in expectations for what would be attainable under undisturbed conditions will
be integral to such a program. While seemingly a tall order, developing a scale of
condition that reflects the full range of biological potential for a region would have two
values as an adaptation of the process of bioassessment in the face of global changes.
First, it would allow existing reference locations to be ranked in terms of ecological
status, where such ranking could be corroborated with documentation of existing levels

of land use encroachment, water withdrawals and flow alterations. Second, it would

define the scale against which future reference station degradation from combined global

change impacts could be tracked and quantified. Characterization of conditions at test

locations would utilize the same scale, and relative changes between reference and test

conditions would provide one piece of evidence in the evaluation of contributions of

global and conventional stressors to impairment.

An example of an objective scale of condition is the Biological Condition Gradient

(BCG), which has been under research and development by the U.S. EPA (Davies and

Jackson 2006). Its application in general is justified as supporting a more uniform

interpretation of condition. This approach would also establish a best estimate of

undisturbed biological potential, and therefore a reference a baseline against which to

track future global changes. This would be a valuable adaptation of the existing

bioassessment approach, and in the long term would help meet the objectives of

maintaining and restoring good levels of water quality and ecological integrity. It is

highly recommended that an objective scaling of biological condition, such as the BCG,

be given much greater focus and support at the national and regional scale.

Development of an objective scale for characterizing ecological status and condition

would have to be regional and be supported by classification of river system types

(development of typologies). This follows the need to define reference conditions

regionally within types, an approach used in the US (e.g., Gerritsen et al. 2000, Barbour

and Gerritsen 2006), Europe (e.g., Verdonschot 2006, Noges et al. 2007), and Australia

(e.g., Kennard et al. 2006). BCGs have been developed for some regions in the US (e.g.,

Gerritsen and Leppo 2005). However, there is still a legacy of conducting site-specific

condition assessments based on comparison to an "upstream" reference site or using a paired-watershed approach. Site-specific comparisons cannot capture or account for effects of large scale, pervasive global changes outside of a regional context that defines the range of variation of conditions that exist for that water body type.

Implicit in tracking and accounting for global changes as a part of bioassessment is a time component to sampling and analyses. Despite the relatively large number of reference stations that may be sampled within a jurisdiction or ecoregion, there are typically few stations that are sampled repeatedly and have long-term data. Preservation of adequate long-term data records, with ongoing, regular (at least annual) monitoring is desirable to increase the robustness of water program assessments to the confounding effects of climate change. Adequate long-term monitoring can be a burden for local to regional management agencies. Considering the desirability of repeated temporal monitoring with the recommendation that reference conditions be established on a regional basis, it is recommended that a network of comprehensive monitoring locations be established and maintained. A regional level of implementation would be consistent with the scale of controlling factors in climate change patterns and vulnerabilities, such as climatic type, geology, topography, elevation, ground water influence, latitude, vegetation, etc. Such conditions often cross state, tribal or other jurisdictional boundaries. Collaboration would be best suited in a modest initial focus of monitoring vulnerable areas and watershed types.

Environment agencies are charged with embracing the concept of protecting remaining high quality stream reaches that define reference conditions, mostly at the watershed management scale (e.g., Palmer et al. 2009). However, the range of possible

protective actions is limited, and for the most part, only addresses climate change indirectly. Protection options encompass minimization, mitigation, and/or buffering from land-use impacts of non-point source runoff, erosion, and hydrologic changes, as well as consideration of water withdrawal impacts. Categories can include socio-political action at the municipal to state scale, such as zoning restrictions, green building incentives, riparian buffer zones, designation of conservation zones or protected areas (e.g., preserves, national parks or forests, wilderness areas), implementation of environmental flow regulations, or other limitations on water withdrawals or diversions from particular stream reaches. Protection actions can be valuable on a local to regional scale, and can be targeted at high quality or unique aquatic resources (Palmer et al. 2009). However, there are often disconnects between the agency with interest in protection of high quality reference locations (e.g., state environment or natural resource departments) and the agencies with the authority to implement such actions. In addition, implementation of comprehensive protection measures takes time, which is in short supply relative to impacts from global changes which are happening now. Given the existing and future projected levels of global change impacts to aquatic resources, targeted protection cannot be solely relied on to maintain the future viability of bioassessment and implementation of water policy.

How do we Resolve Mixed Messages from Existing Biological Indicators?
Appropriate indicators for assessing ecological status and integrity depend on the vulnerability of potential indicators to the range of stressors being tracked, their ease of measurement and interpretability, and the applicability of the information they provide at the spatial or temporal scale of interest. Indicators can be drawn from a wide range of

ecosystem properties, including functional and structural components, production and metabolism, nutrient use and cycling, energy supply, and species composition and feeding types represented in its biotic assemblages (Allan and Castillo 2007). There is increasing emphasis on using measures that reflect functional processes (Paul 1997, Fellows et al., 2006), or that incorporate both ecosystem structure and function (Udy et al. 2006). Moss (2008) argues that true properties of ecological quality include efficiency of nutrient use, habitat connectivity, mechanisms of resilience, and characteristic biological structures and functions, but not necessarily a characteristic species list; and that biomonitoring should not rely mainly on taxonomic indices. Still, it is widely accepted and applied in bioassessment that characteristics of biological assemblages, including both taxonomic and trait groups, strongly reflect ecosystem status and health (Bonada et al. 2007a, b, Johnson and Hering 2009, Norris and Barbour 2009).

Difficulties arise in categorizing unique indicators of global changes, because of similarities in some of the temperature and hydrologic effects resulting from climate change, land use changes, and water removal (Table 2). In addition, climate change is a global influence in the environment, which differentiates it from most conventional stressors, including land use and water demand changes, which are landscape-scale stressors with global implications. Biomonitoring tools should be tailored to the types and scales of stressors expected. If it is a fundamentally sound concept that larger scale (global change) stressors require a comparably large scale of assessment and management, then both the selection of potential indicators and the way they are used in a bioassessment framework must be appropriate for regional (or larger) scale application. This will require a conscientious refocusing of water policy implementation from the

local scale fostered by the current stream reach-specific listing process, to a watershed

scale or larger scale, integrated approach to evaluation and management.

With increasing knowledge of the types of biological responses to climate change

evident around the US, Europe, and elsewhere, as well as the categories of organisms that

are showing relatively predictable responses, it may become possible to adjust

bioassessment metrics to enable a clearer interpretation of stressor identification and

causal analysis. In a management or regulatory context, biological indicators likely to

express the strongest, most regionally consistent and interpretable responses to climate

variables will be most effective to document effects early and establish credible links

between causes (i.e., the stressor of climate change) and observed biological responses.

Groupings of macroinvertebrates (or other organisms used in bioassessment) based

on ecological traits related to temperature or hydrologic preferences provide a link to

temperature- and flow-related global change effects. They are interpretable with regard

to causal relationships, though with limitations, and offer predictive ability and

transferability among regions (Lamouroux et al. 2004, Poff et al. 2006b, Horrigan &

Baird 2008, Verbeck et al. 2008a, 2008b). Trait-based analysis may increase the

diagnostic power of taxon-based bioindicators (Dolédec et al. 2000) and contribute to a

broader understanding of environmental stressor responses (Dolédec & Statzner 2008).

We have already indicated that complete separation of climate change and other global

changes will often be difficult and is unlikely to be accomplished through a single

biological indicator unique to a specific global stressor. Nevertheless, we propose that

separating taxonomic metrics (e.g., EPT taxa, HBI) into sub-categories based on

temperature preferences can be used to enhance the ability to make inferences about

global change effects compared to other stressors. Tracking metrics by temperature preference would provide a mechanism for documenting climate change-related taxa losses or replacements; this could be used in conjunction with other stressor data and trends at reference locations to help differentiate among global and conventional stressor contributions to observed responses.

Preliminary evaluations have been undertaken of the efficacy of separating taxonomic metrics, such as metrics related to EPT taxa, and the HBI, into new metrics that account for temperature preferences of the component taxa (e.g., a 'cold-EPT richness' metric, etc) (U.S. EPA 2010). Additional testing of potential climate-revised metrics would be needed on a regional basis.

Continued success in associating biological responses with probable causes of impairment will require redefinition and recalibration of many traditional metrics. In terms of policy implications, it is likely that such efforts would be viewed by local resource managers as extensive. 'Federal' agencies are already providing support in the form of substantial trait data bases (e.g., Schmidt-Kloiber et al. 2006, U.S.EPA 2010). Additional regional efforts to define temperature traits will be needed to provide an important component in the foundation for evaluating climate change affects on aquatic ecosystems. If the relative contributions from the major global changes can not be reliably separated, this weakens specific impairment or other management decisions with the potential effect of losing the confidence of the regulated community.

Why Embrace A Management Paradigm Shift? The legacy of focusing on point sources of pollution carries an additional legacy of justifying remedial action based on

cause. 'Punishment of the guilty' describes the assessment and management of point sources of pollution, in which the imposition of some regulatory action (e.g., permit limits, mitigations, restoration requirements) is justified based on specific attribution of the cause of impairment. This approach gives a narrow focus to the process of water resource quality protection and restoration and does not encompass adaption strategies to address and manage global change impacts. We recognize there will always be local pollution problems that require local solutions; thus, examination of local causes will continue to be applicable in the context of defining impairment from point source dischargers, supporting issuance and re-issuance of discharge permits, tracking responses to specific restoration actions, and developing total maximum daily load (TMDL) limits. However, local management for preservation of ecological integrity also is impacted by stressors at non-local (global) scales; hence, the need to utilize multiple scales of assessment.

Recognize the management implications of the discontinuities between the scale of global stressors, the scale of the indicators employed in the assessment process (local to regional), and the scale of management options (local to regional) is a first step in developing an appropriate response. The causes of climate change cannot be altered at the scale of most water resource management activities. Even if maximum reductions of greenhouse gas emissions were achieved immediately on a national to global scale, "committed" climate responses would continue for decades. Similarly, population growth along with increases in developed and agricultural land uses will continue, as will demands for human uses of fresh water, outside of the purview of local water quality

agencies. But water resource managers are still faced with protection or restoration of good water quality and ecological integrity.

In the context of climate change it is increasingly important to consider adaptation strategies or other management actions for remediation or restoration that are not necessarily directly related to the proximal cause of the problem. This is a diverse concept that has been touched on by the IPCC ("no regrets" adaptations, IPCC 2001) and the U.S. Climate Change Science Programs (CCSP 2008). We suggest that managers be encouraged to consider actions that can improve or ameliorate an impaired condition in an aquatic system, considering broad factors such as feasibility of implementation, possibility of multiple environmental benefits, and direct or indirect impacts on conditions of concern. The actions may not be directly related to the causes of concern, but given the increasing likelihood that multiple causes may be contributing to a particular condition at multiple scales, this is inevitable. Adaptations can be selected to have recognizable environmental benefits, whether projected global changes ultimately impact the site or not ("no regrets" adaptations). The need to take whatever management actions that may be feasible is starting to be recommended in response to climate change in particular (Palmer et al. 2009).

In the US, the U.S. EPA develops the scientific basis for water law and environmental policy actions and provides recommendations and guidance to state water quality agencies for implementation. In Europe, the guiding principal is the Water Framework Directive and the oversight organization that includes representatives of the partner countries. Each country is then charged with the mandate to implement the Directive.

Regardless, the agency with the responsibility of assessing and protecting its water resources should help advance a revised assessment process, including inputs on:

- Methods and criteria for evaluating the relative vulnerabilities of regions or watersheds to global change effects, focusing on alterations to thermal and hydrologic regimes, and future projections of land use patterns. Assessments should use objective criteria to define vulnerabilities. Providing clear expectations of effects due to global change for key species and communities in freshwater ecosystems will help these programs achieve their goals under changing environmental conditions.

- Support of additional research needs through agency and academic channels, including focus on how to implement program modifications, BCG implementation, traits and biotic response data base expansion, and metrics modifications.

- Development and implementation of more sophisticated watershed modeling to bridge the scale gap between General Circulation Models (GCMs) and watershed responses. The EuroLimpacs project of the EU has done much in this regard (e.g., Whitehead et al. 1998a, 1998b, 2002; Wade et al. 2002). In the US, EPA's Global Change Research Program is supporting research to evaluate the impacts of climate change and urbanization on hydrology and water quality in twenty major river basins throughout the US.

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CHAPTER 4

Short-Term Effects of the Las Conchas Fire on Benthos in the East Fork Jemez River in the Valles Caldera, New Mexico

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Abstract

This study evaluates short-term responses of stream benthic invertebrates to the direct
and indirect effects of a wildfire, examining the traits that best reflect these short-term
responses to understand the life history strategies that contribute to invertebrate
vulnerability, resistance, or resilience to fire impacts on streams. The East Fork Jemez
River is a high-elevation, low-gradient headwater stream in the Valles Caldera of New
Mexico. The study site is instrumented for continuous measurements of a variety of water
quality, water chemistry, and surface and groundwater parameters with the long-term
goal of studying climate variability and climate change impacts on water quantity and
quality. Biological processes and communities, including benthic macroinvertebrates,
have been sampled periodically since March 2011. In late June and July of 2011 the Las
Conchas wildfire burned headwaters of the study area, providing an opportunity to
examine short-term impacts of extreme fire in the watershed on water quality, stream
chemistry, and biological responses. The benthic community was compared at riffle and
pool sites during five collection periods from March to October. Differences between
pool and riffle communities were clear for several taxonomic and trait groups. Results
show the extent to which a fire of the severity of the Las Conchas burn can impact stream
water quality, linking severe fire impacts in the upper forested watershed to downstream
water quality impacts through pulsed monsoonal flow events affecting turbidity,
dissolved oxygen (DO), pH, and nutrients. Post-fire benthic responses were not
devastating, with minimal responses found in total abundance or taxa richness. However,
numerous taxa responded to the post-fire flow and water quality disturbance either
positively, negatively, negatively with recovery, or neutrally, with responses well

represented by selected habitat and feeding type traits. These patterns were interpreted in terms of their evidence of vulnerability to both physical disturbance and trophic impacts, resistance to disturbance, and resilience following disturbance. These are mechanisms through which an increasing frequency of fire disturbances with climate change, and the resulting flow and water quality responses, could lead to persistent invertebrate community alteration through the maintenance of disequilibria.

Introduction

Wildfires are projected to increase in frequency and duration as a result of climate change, particularly in the western U.S., varying year to year in relationship to higher spring and summer temperatures and earlier spring snowmelt (Westerling et al. 2006). There is also evidence that the severity of wildfires in the western U.S. is increasing (Miller et al. 2009) with corresponding changes in landscape patchiness and in physical, chemical, and biological processes. Wildfires are a natural phenomenon, and in some regions are an important agent in structuring ecosystems and stimulating or resetting processes (Scholl and Taylor 2010, Johnson 1992). As such, they can influence species composition and recurrent successional patterns (Franklin and Hemstorm 1981). However, increases in fire frequency and intensity can have a profound effect on watersheds and associated stream ecosystems, through changes to soil chemistry, nutrient and sediment runoff, cover of riparian vegetation, and trophic dynamics (Beche et al. 2005, Gallaher and Koch 2005, Earl and Blinn 2003, Spencer et al. 2003). Fire-driven changes in soil permeability also lead to dramatic increases in rapid surface runoff from post-fire storm events (Robichaud et al. 2000). The degree of disturbance following fire

can include extreme physical disturbance and habitat restructuring from debris flows and sedimentation (Cannon et al. 2010, Allen 2007) or can be less catastrophically linked to increases in light penetration and pulsed inputs of nutrients and turbidity that can alter the trophic base of the stream food web (Spencer et al. 2003). The level of expected effects depends on the interaction of factors that include the severity and extent of the fire, stream size, slope and other topographic features of the watershed, geology and soil properties, and storm rainfall intensity (Canon et al. 2010, Minshall 2003, 2001a).

Much of the work on stream invertebrate responses to fire has been done in forested areas, where woody riparian vegetation contributes both shade and allochthonous organic matter (leaf litter, woody debris) to the stream. Fires affect trophic inputs to streams by burning through riparian and surrounding watershed forests, thus removing the major source of organic matter inputs. At the same time, opening of the forest canopy increases light penetration, allowing greater *in situ* primary production, often augmented by increases in nutrient inputs (Minshall 2003, Spencer et al. 2003, Tronstad et al. 2011). Because stream invertebrate communities are structured by habitat conditions, or 'templates', that prevail (Southwood 1977, 1988), invertebrate life history strategies are expected to reflect the trophic resources, substrate types, and flow patterns that define the habitat (Poff and Ward 1990). Accordingly, traits comprising functional feeding types (FFGs) that were defined to capture the specialization of invertebrates to exploit particular categories of food resources (Cummins and Klug 1979) are expected to respond to fire-induced trophic alterations (Minshall 2003).

The removal of riparian canopy by wildfire is immediate, and will drive trophic alterations in the stream; however, corresponding trophic responses of invertebrates may

take time to manifest. But other short-term effects of wildfires on stream invertebrates can be expected, associated with the physical disturbances from post-fire runoff pulses (Minshall 2003). In the southwestern U.S., increased flow pulse events following a fire can be associated with spring snowmelt or with summer monsoonal rainfall events (Rinne 1996), and can be substantially greater in magnitude compared to pre-fire conditions (Woodhouse 2004, Veenhuis 2002). Physical impacts to invertebrates can include dislodgement of organisms and habitat scour from increased flow velocities, smothering of riffle habitat and in-filling of pools from sediment transport and deposition, alteration of channel morphology, and mass habitat restructuring from debris flows (Minshall et al. 1997, 2001b; Rinne 1996). Thus the types of life history traits that might impart vulnerability or resistance to such impacts, or resilience reflected in the ability to recover quickly after disturbances, should be considered separately. Streams are, by their nature, variable and typified by disturbance (Resh et al. 1988), with greater heterogeneity and more frequent disturbance selecting for species with more opportunistic life history traits (Poff and Ward 1988). Opportunistic taxa are generally found to be better adapted to post-fire conditions (Minshall 1997, 2003; Mihuc et al. 1996), and the presence of many opportunistic taxa in stream communities may increase their resilience to fire disturbance and account for observations of relatively rapid recovery in some studies (e.g., Minshall et al. 2001a). But rapid recovery to pre-fire conditions with no lasting impacts to the stream communities is not always the case. Recovery may be more likely for taxa with certain opportunistic characteristics, such as ability to persist in a wide range of habitat types (Mihuc et al. 1996), which could lead to selection for communities with greater dominance by such opportunistic life histories. With sufficient extent of physical habitat

alteration and organism loss, recovery may not be apparent for years after a fire (Rinne 1996), and convergence to pre-fire benthic community characteristics can often take 10-15 years (Minshall 2003). If the frequency of fire disturbance increases in the landscape, more frequent community restructuring could lead to predictable, long-term shifts to more opportunistic-dominated and therefore less diverse communities.

Previous fire studies suggest that invertebrate traits that are well adapted to post-fire conditions include dispersal in drift, multi-voltine reproduction, high reproductive success, general tolerance, broad habitat preferences, and mixed herbivore-detritivore food habits (Minshall 2003, Minshall et al. 1997, Mihuc et al. 1996). In contrast, invertebrates particularly adapted to stable riffle habitats or to the slower flows of pool habitats have been found to decline in post-fire conditions (Mihuc et al. 1996). Despite the value of tracking different types of invertebrate traits for responses to post-fire conditions, some studies have reported invertebrate responses to fire to be individualistic (Mihuc et al. 1996). This study presents the opportunity to evaluate separately by broad habitat type, whether certain life history traits or particular taxa are more responsive to post-fire stream disturbances.

Since wildfires are unpredictable in the time and place of their occurrence, our ability to study short-term fire effects using before- and after-fire samples in a single stream is fortuitous. Our study site was established early in 2010, as part of the New Mexico Experimental Program to Stimulate Competitive Research (NM EPSCoR), with the goals of examining climate variability and climate change impacts on water quantity and quality. The Las Conchas fire then burned through the study site in late June to early August of 2011. The site was instrumented for continuous measurements of a variety of

water quality, water chemistry, and surface and groundwater data, and included sampling of stream metabolism, periphyton, biofilms, macrophytes, and macroinvertebrates, with the long-term goal of studying climate variability and climate change impacts on water quantity and quality. The opportunistic goals of this study are to evaluate short-term responses of stream benthic invertebrates common to the two different habitat types (riffles and pools) to the direct and indirect effects of this wildfire, and examine the categories of traits that appear to capture short-term responses, and use this to understand the life history strategies that contribute to invertebrate vulnerability, resistance, and resilience to fire impacts on streams.

Methods

Study Area

Our study area is in the East Fork of the Jemez River (EFJR), located near the southern margin of the Valles Caldera National Preserve (VCNP) (Figure 1). The VCNP encompasses 360 km^2 (89,000 acres) in north-central New Mexico (Figure 1; VCNP 2012) that falls within the Southern Rockies EPA Level III ecoregion (Griffith et al. 2006). The study area is a mosaic of three Level IV ecoregions, including Volcanic Subalpine Forests and Volcanic Mid-Elevation Forests at higher elevations that surround a meadow that is part of the Grassland Parks ecoregion, through which the EFJR flows. The EFJR originates as a group of springs in the southeast corner of the VCNP and flows 34.5 km to its confluence with Rio San Antonio, which then together form the Jemez River (Simino 2002). In the vicinity of our study area, the EFJR is a 3rd order stream.

Downstream of our study reach, the lower 17.7 km (11 miles) of the EFJR was

designated as a Wild and Scenic River in 1990.
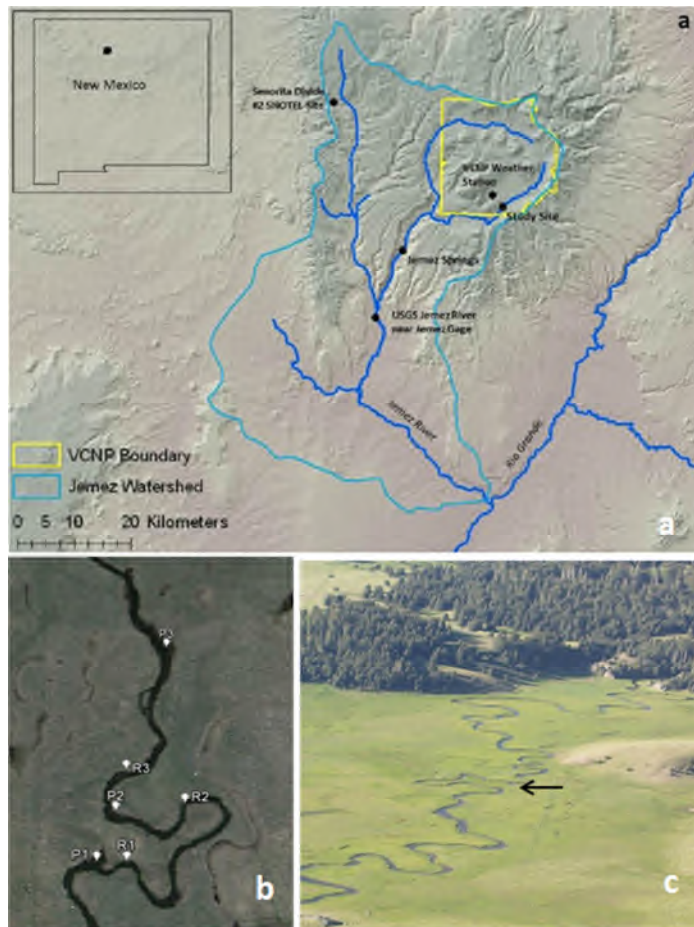
Figure 1. a) Regional map showing the East Fork Jemez River study site, located within the Valles Caldera National Preserve (VCNP) (yellow line) and the Jemez Watershed (light blue line) boundaries, (b) site map showing the 3 riffle and 3 pool sampling locations within the study reach, and (c) photo of study site.

Our EFJR study reach is at high elevation (~2,650 m) but is a low gradient stream,

often with a near 0% gradient, whereas other reaches of the EFJR, including the

headwater reach just upstream of the study area, as well as downstream reaches, can have

a much higher gradient, up to 7% (Simino 2002). The upper reaches of the EFJR within

the Grassland Parks ecoregion have no large woody riparian vegetation, and has not had

woody vegetation in recent history (Allen 1989, Simino 2002, Anschuetz and Merlan 2007). The EFJR study reach itself is 200 m long, and located entirely within a grazing exclosure (a 160 m X 160 m area enclosed by a 2.5-m high fence to exclude elk) that was installed on June 1, 2004 (Van Horn et al. 2012).

Precipitation in the study area ranges from 43-61 cm per year in the grasslands to 61-97 cm in the forested ecoregions at higher elevations, where substantial snowpack also can occur. Snowpack melt dominates the spring runoff pulse, which historically occurred in May or early June. On average, about one third (36%) of the annual precipitation occurs as summer monsoonal events (Muldavin and Tonne 2003). Runoff from monsoon storms contributes to the flashy flow characteristics of the EFJR in summer (Simino 2002).

Water Quality, Chemistry, and Streamflow Measurements

The study site is instrumented for continuous measurements of a variety of water quality, water chemistry, and surface and groundwater data. *In situ* surface water measurements of temperature ($\degree$C), specific conductance (SC, temperature corrected (25$\degree$C), mS/cm), dissolved oxygen (DO, mg/l), pH (standard units), and turbidity (NTU) were made using a Yellow Springs Instruments (YSI) model 6920 V2 sonde. Measurements were made at 15-min intervals from 15 March to 02 November 2011. Measurements of dissolved nitrate (NO_3^-) were made at 15-min or 30-min intervals from 15 March - 13 October 2011 using a Satlantic Submersible Ultraviolet Nitrate Analyzer (SUNA). Measurements of dissolved phosphate (PO_4^{3-}) were made at 1-hr intervals from 16 May to 9 September 2011 using a WETLabs Cycle-PO4 analyzer. Details of

instrumentation methods, deployment, and associated sample analyses can be found in Sherson (2012).

Continuous river stage data were measured from March to October 2011 using a pressure transducer co-located with the nutrient instruments, with data corrected using local barometric pressure. Air temperature, photosynthetically active radiation (PAR), and barometric pressure were obtained from flux towers located in the Valles Caldera National Preserve. Precipitation data were obtained from the Valles Caldera National Preserve Headquarters meteorological station. In addition to study site-specific stage data, streamflow data from the East Fork Jemez ~0.5 km downstream of the study reach were obtained from the Valles Caldera National Preserve at the Hidden Valley Gage.

For each location within the study reach where biological sampling was conducted, discrete measurements of water depth (m) and flow velocity (m/s) were made on each sampling date for each biological sample taken from a pool or riffle. Velocity was measured using a Marsh-McBirney Flo-Mate 2000 portable flow meter.

Biological Sampling

Benthic invertebrates were sampled in a semi-quantitative manner separately from each of three pools and three run/riffles (hereafter referred to as riffles) along the study reach. For each benthic sample from a given pool or riffle, a D-ring dipnet with 0.5 mm mesh was used to collect 5 (.09 m^2) 'jabs', holding the net at the downstream end of the square and disturbing the substrate in the 0.09 m^2 area upstream of the net. Within any one pool or riffle, the five sampling spots were selected arbitrarily but to be distributed within the habitat and composited to obtain a representative habitat sample. Thus, for a

given sampling date, approximately 0.46 m^2 of stream bottom was sampled in each of the three pool and riffle habitats along the study reach. Benthic invertebrates were collected during five months in 2011 (March 9, June 2, July 15, September 5, and October 13). All samples were preserved in the field using 95% ethyl alcohol so that final ethanol concentration was about 70%.

In the laboratory, samples were sorted with the aid of a dissecting microscope. Sub-sampling was used if organisms were very abundant; however, a minimum of 400 organisms were sorted from each sample. Subsampling was accomplished using a 25 cm x 46 cm flat plastic pan that was marked to define 24 equal-area squares. A sample was distributed evenly across the pan, numbered squares were randomly selected, and all material in each selected square was sorted. This process was continued until the minimum of 400 organisms were sorted; however, material from the final square selected was always sorted completely. The fraction of the original sample sorted was recorded. Counts by taxon were corrected for sub-sampling, and converted to number per square meter for analysis.

Most benthic invertebrates were identified to the lowest practical taxon, with most aquatic insects identified to genus, including the Chironomidae (non-biting midges). Chironomids were slide-mounted for identification using a compound microscope. Oligochaeta were only identified to family, and Acari (water mites), Copepoda, Cladocera, and Ostracoda were left at those higher groupings.

Functional feeding group (FFG, Cummins and Klug 1979) as well as habit, rheophily, and macrophyte-association (Merritt et al. 2008) traits for the macroinvertebrate taxa found at the EFJR were assigned using U.S. EPA's Freshwater Biological Traits

Database (http://www.epa.gov/ncea/global/traits) (U.S. EPA 2012). To the extent

possible, missing trait information was then filled in using Merritt et al. (2008).

Other concurrent biological studies included sampling of biomass and species

identification of aquatic macrophytes, and sampling of epiphytic and rock/sediment-

associated diatoms at the same pool and riffle sites sampled for benthic invertebrates. In

addition, stream respiration and primary production were estimated (Schafer 2013).

Data Analyses

We tested the null hypothesis that biological and abiotic environmental data were

normally distributed based on the Shapiro-Wilk test (Shapiro and Wilk 1965), using

Statistica software (Version 10.0, Copyright StatSoft, Inc., 1984-2007). The null

hypothesis was rejected for all but depth data. As a result, all benthic abundance data, as

well as all abiotic data except depth, were log $(x + 1)$ transformed for analysis. Retesting

of transformed data using the Shapiro-Wilk test showed transformation of most traits

groups [all habit and all but shredder functional feeding groups (FFGs)] resulted in

distributions not different from normal. However, only some of the more abundant taxa

(*Microtendipes*, *Dubiraphia*, *Hyalella*, Acari, *Pisidium*, *Physella*, Planorbidae,

Tubificidae) had transformed abundances not different from a normal distribution.

Therefore, testing for differences between groups was accomplished using the non-

parametric Multi-Response Permutation Procedure (MRPP) (McCune and Grace 2002),

using PC-ORD Software (Version 6.11). MRPP was used to test differences among

habitat types, pre- and post-fire periods, sites, and months in taxonomic and trait

community composition, using the Bray-Curtis (Sorensen) distance metric (see below).

The Las Conchas fire burned from June 16 through August 3, 2011; therefore, the July 2011 samples were collected during the fire, but before the start of monsoon rains. Examination of water quality patterns, stream flows, and benthic community similarity based on cluster analyses (see below) suggested that minimal water quality or flow effects had occurred at the time of the July collections. Therefore, the July 2011 samples were evaluated with the pre-fire samples. Thus, in subsequent reporting, March, June, and July are classified as pre-fire, and September and October 2011 samples are classified as post-fire.

Classification techniques

We used Non-Metric Multi-Dimensional Scaling (NMDS) (McCune and Grace 2002) to explore benthic community relationships and grouping patterns related to habitat type (riffles and pools), pre- and post-fire time periods, months, and sites. We performed NMDS based on taxonomic community composition, and also based on functional composition described by functional feeding groups as well as habit traits. In each case, we used the correlations of NMDS axes with a suite of environmental variables, as well as with taxon or trait abundances, to associate community structure with predominant site characteristics and identify which variables were driving observed benthic community groupings. NMDS was performed in PC-ORD Software (Version 6.11, McCune and Mefford 2006).

We also used cluster analysis to determine similarities among samples based on taxonomic or trait composition, and assigned sampling sites and dates to groupings using the clustering routine in PC-ORD (Version 6.11, McCune and Mefford 2006). For both

clustering and NMDS, we used the Bray-Curtis (=Sorensen; Bray and Curtis 1957)

similarity coefficient, which compares samples in a pair-wise fashion based on the sum of

the proportions of the minimum shared abundances for each taxon to total abundance for

each taxon, and is robust for use with taxonomic data. To compare the influence of rare

taxa on results, these analyses were performed on the complete data set and after

removing taxa occurring in <5% of samples. Excluding rare taxa in cluster analysis

resulted in no difference in sequence or composition of groupings, or in clustering

distances. Therefore, analyses using the entire community are presented. We used a

Flexible Beta clustering method with beta set at -0.25 (Boesch 1977, McCune and Grace

2002).

Indicator Species

Indicator species analysis (ISA, Dufrêne and Legendre 1997) identifies taxa that are

strongly characteristic of a particular group defined in a cluster analysis based on how

strongly the abundance of the taxon is concentrated in the group compared to other

groups, and the fidelity of its occurrence in the group compared to other groups (McCune

and Grace 2002). The formula for the Indicator Value (IV) is:

IV = (abundance in group/sum of average abundances across all groups) * (# of

samples of taxon present/total # samples).

This comparison of relative abundance and frequency of occurrence of each taxon

within and among groups is used to calculate an Indicator Value (IV). Monte Carlo

permutations are used to derive the probability that each taxon is indicating a group with

statistical significance. Only taxa with IV's significant at p<0.05 are reported. Definition

of indicator species was used to characterize habitat type (pool or riffle) and pre- and post-fire groupings.

Las Conchas Fire

The Las Conchas fire started near Highway 4, just south of the VCNP, on June 26, 2011. The fire spread rapidly, burning 17,400 hectares during the first 14 hours. The fire was contained 5 weeks later, on August 1, having burned 63,370 hectares (about 156,600 acres). Approximately 12,140 hectares of the VCNP were burned, including headwaters of the study area, mainly during the first week of the fire (Parmenter 2011). Location and relative severity of the burn are shown in Figure 2.

Figure 2. Estimated burn severity of the Las Conchas fire (June 26-August 1, 2011) within the Valles Caldera National Preserve (from Parmenter 2011); study site shown by yellow star.

Results

General EFJR Habitats and Benthic Communities Characteristics

Riffles and pools in the EFJR study area were distinct in terms of geomorphic and hydrologic characteristics. Mean and maximum water depths were lower, and flow velocity was higher in EFJR riffles compared to pools (Table 1). These differences are not large, and some locations, such as pool sampling site P1, show characteristics that are transitional between pool and riffle depth and velocity characteristics during some months. Expected seasonal variation in stream discharge influences the depth and velocity characteristics of the pools and riffles in the EFJR, but correction of depth and velocity for discharge at the time of measurement shows relatively consistent riffle and pool characteristics (Figure 3). Accordingly, benthic community differences and responses to fire were evaluated separately by habitat type, with no *post-facto* reclassification.

Table 1. Riffle and pool habitat characteristics.

Parameter		Riffles	Pools
Depth (m)	mean	0.16 ± 0.08	0.33 ± 0.08
	maximum[1]	0.27 ± 0.09	0.47 ± 0.09
Velocity (m/s)	mean	0.22 ± 0.10	0.08 ± 0.07
	maximum[1]	0.45 ± 0.11	0.23 ± 0.14
Substrate		Gravel to cobble embedded in sand and finer substrate, often packed (hard)	Unconsolidated fine sediment, some gravel overlaying harder substrate
Macrophytes (mean g/m^2)	*Elodea*	18 ± 38	127 ± 168
	Potamogeton	111 ± 304	<1
	Ranunculus	48 ± 47	1 ± 3

1 - average of the maximum values for all pool sites and dates
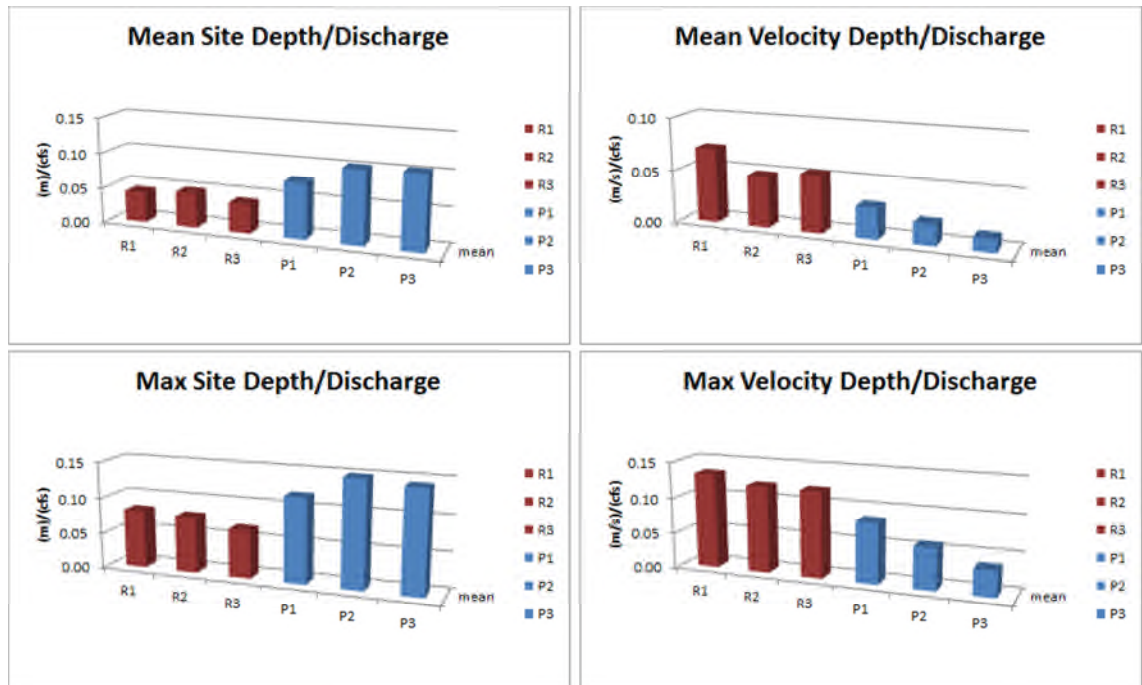
Figure 3. Mean and maximum depth (m) and velocity (m/s) proportioned to mean discharge (cfs) at the time of measurement, of riffle and pool sites in the EFJR, averaged over five collection months (March – October) in 2011.

Pools and riffles also showed predictable differences in substrate type and associated macrophyte communities. The substrate in pools included soft, unconsolidated fine sediments that overlaid a harder packed base, although the coverage and thickness of these soft sediments was variable. Riffle substrate included a range of gravel and cobble, although this was usually embedded in a mix of coarse to fine sand that was often relatively hard. Starting in the spring and continuing through the remainder of the summer and fall sampling period, macrophytes were common in both habitat types, with *Elodea* widely distributed but most abundant in pools, and *Potamogeton* and *Ranunculus* typical of riffles (Table 1).

A total of 69 invertebrate taxa were identified, including 57 in riffles and 60 in pools. Many taxa occurred in both habitats, although many were more abundant and frequent in

occurrence in one habitat or the other. Table 2 shows the top ten dominant taxa found in

each habitat type, averaged over the study period. The mayfly *Tricorythodes* was the

dominant taxon in EFJR riffles, while the chironomid *Microtendipes* was the overall

dominant taxon in pools. *Tricorythodes* was second dominant in pools, although its

abundance in pools was one third of that found on average in riffles (Table 2). Further

examination reveals that among the top 10 dominants in each habitat type, seven taxa

were common to both. In both riffles and pools, the top 10 dominant taxa comprised on

average 82% of the total abundance of benthic invertebrates. No stoneflies were collected

from this study site.

Table 2. Top 10 dominant taxa in riffles and pools of the EFJR, averaged over the 2011
study period.

Riffles		Pools	
Taxon	#/m^2	Taxon	#/m^2
Tricorythodes	691	*Microtendipes*	391
Hyalella	274	*Tricorythodes*	240
Heterlimnius	209	*Hyalella*	148
Microtendipes	161	Planorbidae	147
Acari	86	*Physella*	124
Pisidium	85	*Dubiraphia*	87
Planorbidae	70	*Fallceon*	53
Dubiraphia	65	*Enallagma*	34
Helicopsyche	65	*Ameletus*	33
Fallceon	43	*Pisidium*	33
Percent of total abundance	1749, 82%	Percent of total abundance	1290, 82%

The trait composition of invertebrates differed between riffles and pools. For FFGs,

both habitat types were dominated by collector-gatherers; however, gatherers were more

abundant in riffles compared to pools, while scrapers and filterers were more abundant in

pools than riffles (Figure 4). The greater abundance of filterers in pools was largely

driven by the very high abundance of the chironomid *Microtendipes* in pools. Although a

filter feeder, *Microtendipes* is also a burrower, common in depositional habitats (Merritt et al. 2008). With regard to habit traits, sprawlers also were the dominant trait in both habitats, although these as well as clingers were more abundant in riffles, while burrowers and climbers were more abundant in pool habitats (Figure 4).
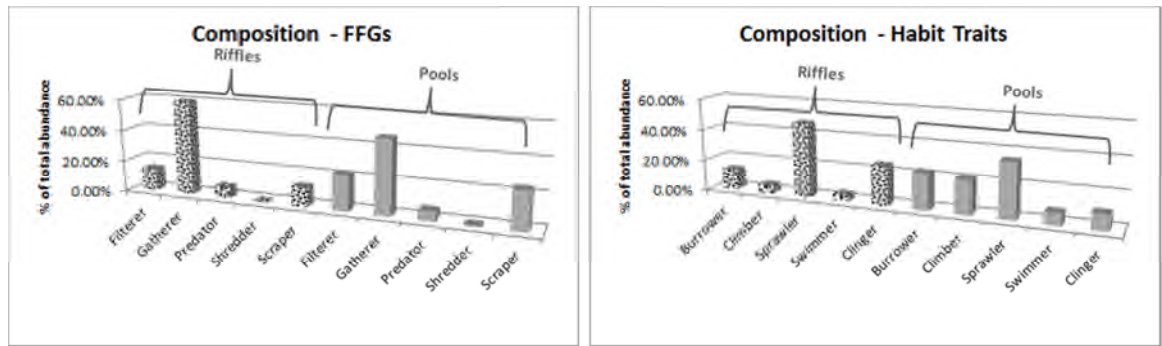
Figure 4. Average benthic community composition in riffles and pools based on functional feeding groups (FFGs) (left panel) and habit traits (right panel).

Based on clustering distances, similarity among benthic invertebrate samples was determined first by habitat type and secondarily by influence of the fire (Figure 5). In each of the pre- and post-fire time periods, habitat was a stronger driver than month except in July, when pool and riffle samples were more similar to each other than to samples from the same habitat in other pre-fire months. All groupings (habitat, fire period, month and site) tested using the non-parametric MRPP were significantly different based on taxonomic composition and abundance (Table 3).
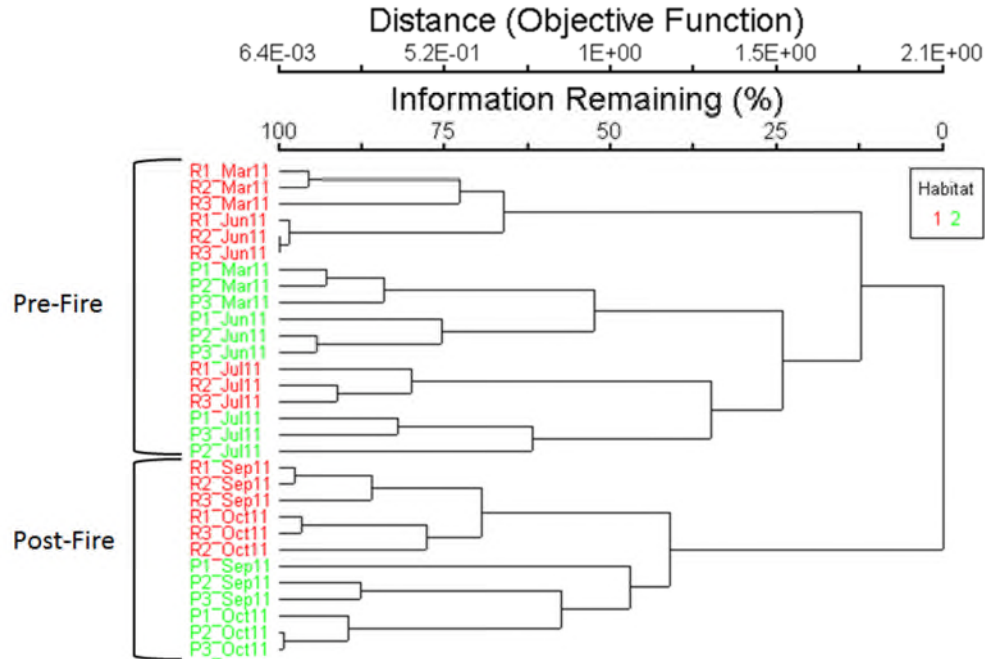
Figure 5. Cluster analysis of benthic invertebrate samples from the East Fork Jemez River, March to October 2011, color coded by habitat type (1=riffles, 2=pools). (Bray-Curtis distance metric, log (x + 1) abundance transformation, flexible beta = -0.25, percent chaining = 1.90).

Table 3. Multi-Response Permutation Procedure (MRPP) results comparing sample groupings based on taxonomic composition.

Grouping	T statistic	A*	p
Habitat	-9.78	0.087	0.0000
Fire	-9.22	0.082	0.0000
Month	-10.27	0.193	0.0000
Site	-2.57	0.055	0.0114

*chance-corrected group agreement

Indicator species analysis (ISA), used to investigate the taxa that were statistically influential in determining habitat or pre- and post-fire community differences, showed distinct groups of taxa that characterized riffles and pools, and a largely different group of taxa that showed pre- and post-fire responses (Table 4).

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Table 4. Indicator Values (IV) showing taxa significantly associated with cluster analysis groupings: A. Habitat groupings of riffles (R) and pools (P); and B. Pre- and post-fire groupings.

A. Habitat (Riffle and Pool) Grouping

Taxon	IV R (P)	P	Taxon	IV P (R)	p
Hydropsyche	87 (1)	0.0002	*Enallagma*	73 (5)	0.0008
Heterolimnus	77 (17)	0.0002	Ephydridae	40 (0)	0.0164
Helicopsyche	74 (8)	0.0004	*Orthocladius*	46 (6)	0.0296
Acari	62 (35)	0.0010	*Ameletus*	45 (5)	0.0380
Oecetis	63 (3)	0.0016	*Physella*	57 (40)	0.0496
Tricorythodes	56 (44)	0.0016			
Bezzia	61 (10)	0.0068			
Hyalella	55 (45)	0.0104			
Simulium	48 (1)	0.0118			
Ophiogomphus	55 (15)	0.0306			

B. Pre- and Post-Fire Grouping

Taxon	IV Pre (Post)	P	Taxon	IV Post (Pre)	p
Microtendipes	63 (34)	0.0002	*Pentaneura*	51 (5)	0.0122
Orthocladius-Cricototopus	72 (0)	0.0002	Ostracoda	48 (6)	0.0200
Eukieferiella	72 (0)	0.0002	*Tricorythodes*	54 (46)	0.0242
Hydrobaenus	61 (0)	0.0012	*Microcylloepus*	44 (2)	0.0212
Fallceon	63 (13)	0.0044	*Dubiraphia*	59 (39)	0.0322
Thiennemaniella	44 (0)	0.0090	*Baetis*	54 (17)	0.0464
Hydroptilla	55 (6)	0.0178	Dytiscidae	25 (0)	0.0484
Procladius	39 (0)	0.0260			
Ephydridae	33 (0)	0.0496			

Even though the mayfly *Tricorythodes* is a sprawler, typical of depositional habitats, and it occurs in both habitat types in the EFJR, it occurs in greater abundance and frequency in riffles in the EFJR, and as a result has a high Indicator Value (IV) score at riffle sites. While under natural conditions, *Tricorythodes* would be expected to increase in abundance over the summer, its abundance remained relatively low through the period of the fire until late in the fall (October), and therefore had a high indicator value for the post-fire period.

The caddisflies *Hydropsyche*, *Helicopsyche*, *Oecetis*, and *Hydroptilla*, the elmid

beetle *Heterlimnius*, and the black fly *Simulium*, all clingers, were strongly associated

with riffle habitat (Table 4A). The amphipod *Hyalella*, another sprawler, and the

burrowers *Bezzia* (a dipteran) and *Ophigomphus* (a dragonfly) were also indicator species

for riffles.

Only a few taxa were significant indicator species of pools (Table 4A). These

included the damselfly *Enallagma* (a climber), brine flies in the family Ephidridae

(burrowers), the midge *Orthocladius* (a sprawler), the mayfly *Ameletus* (a swimmer), and

the snail *Physella* (a climber).

Indicator species values also showed which taxa were most characteristic of the pre-

and post-fire periods. Table 4B shows that six midge taxa (five sprawlers and one

burrower) and one mayfly (*Fallceon*, a swimmer), as well as *Hydroptilla* and Ephidridae,

were all common before the fire, and significantly less common after the fire. Several

taxa, including the sprawling midge *Pentaneura*, the small, shelled crustacean Ostracoda,

the mayfly *Tricorythodes*, the clinging elmid beetles *Microcylloepus* and *Dubiraphia*, the

swimming mayfly *Baetis*, and the predaceous diving beetles (Dytiscidae, swimmers),

were clearly more abundant and widely distributed in the post-fire period (Table 4B).

NMDS results corroborate the grouping of benthic samples based on habitat type and

fire period, and give insight into the environmental and habitat characteristics that drive

these groupings. The NMDS ordination based on taxonomic composition gave a

recommended 3-dimensional solution, with a final stress value of 12.64, well within the

optimal range of 20 or less (McCune and Grace 2002). Riffle and pool habitats separated

mainly on Axis 1, while pre- and post-fire periods separated on Axes 2 and 3 (Figure 6).

Axis 1 is correlated with water depth (negative) and velocity (positive), as well as with

the biomass and frequency of occurrence of the aquatic macrophytes *Elodea* (negative)

and *Ranunculus* (positive) (Table 5). Thus the composition of the benthic assemblages at

riffle sites were, in all months sampled, associated with higher velocities, shallower

depths, and the presence of *Ranunculus*, while pool assemblages were associated with the

deeper slower water and *Elodea*. The taxa that were significantly associated with the

riffle group (correlated positively with axis 1) included *Heterlimnius*, *Tricorythodes*,

Acari, and *Helicopsyche* (Table 6). These are consistent with the riffle indicator species

identified in association with cluster analysis, although it is not as extensive a list. Only

the damselfly *Enallagma* was significantly associated with pool sites in the NMDS

(Table 6).

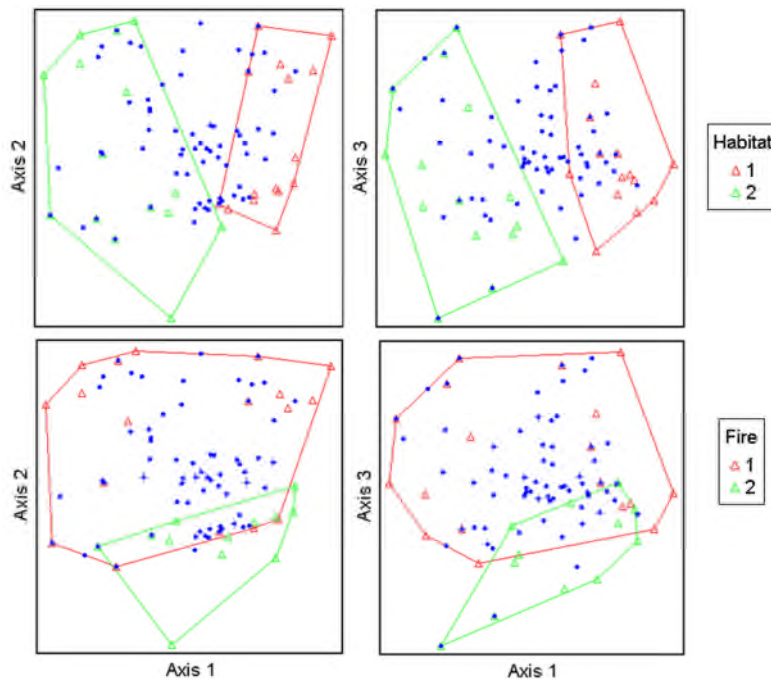

Figure 6. Axes 1 and 2 from NMDS ordination using benthic invertebrate community data (3-dimensional solution recommended, final stress = 12.64), showing samples highlighted according to habitat (top 2 panels, 1=riffles, 2=pools), and association with the Las Conchas fire (bottom 2 panels, 1=pre-fire, 2=post-fire). See Tables 5 and 6 for habitat characteristics and taxa that define each axis.

Table 5. Correlation of habitat characteristic variables on each defining axis in NMDS ordination (see Figure 6).

Loading	Axis 1 Variable	r^2	Axis 2 Variable	r^2	Axis 3 Variable	r^2
Positive	Velocity, max	.57	DO, avg, interval	.46	pH, min, interval	.55
	Velocity, avg	.40	PAR	.68	DO, min, 2 wk	.55
	Ranunculus, biomass	.37				
	Ranunculus, freq	.28				
Negative	Depth, max	.52	NO_3, max, 2 wk	.64	SC, avg, interval	.47
	Depth, avg	.50	PO_4, avg, interval	.52	Stage, max, interval	.69
	Elodea, freq	.43	Temp, avg, interval	.60	Flow, max, interval	.43
	Elodea, biomass	.30	pH, avg, interval	.59	Turbidity, avg, 2 wk	.40
			pH, range, interval	.52		
			Turbidity, max, interval	.59		
			DO, range, interval	.63		
			Precip, max, interval	.83		

Table 6. Correlation of benthic invertebrate taxa on each defining axis in NMDS ordination (see Figure 6).

Loading	Axis 1 Variable	r^2	Axis 2 Variable	r^2	Axis 3 Variable	r^2
Positive	*Heterlimnius*	.82	*Ortho-Cricot*	.56	*Hydroptila*	.60
	Hydropsyche	.73	*Thienemaniella*	.49	*Tventenia*	.45
	Tricorythodes	.47	*Eukieferiella*	.38	*Fallceon*	.43
	Acari	.37	*Tipula*	.30	*Microtendipes*	.40
	Helicopsyche	.32	*Graptocorixa*	.30	*Psectrocladius*	.35
Negative	*Enallagma*	.37	*Dubiraphia*	.54	*Enallagma*	.36
			Ostracoda	.31	*Baetis*	.31

Axes 2 and 3 were correlated with variables that tracked precipitation inputs and flow responses (stage, flow), as well as water quality and chemistry variables that changed in association with these spates (Table 5). The benthic assemblages typical of the pre-fire period grouped in the upper half of Axes 2 and 3, and were associated with high average DO in the interval prior to sample collection, high PAR, higher minimum pH and DO, as well as low precipitation and stage. The assemblages typical of the post-fire period grouped in the lower half of Axes 2 and 3, and were correlated with high

maximum precipitation, stage and flow during the interval prior to benthic sample

collection, as well as peak values of NO_3, PO_4, turbidity, specific conductance (SC),

lower DO and pH minimums, and increased range (variability) in DO and pH (Table 5).

Benthic taxa that in the NMDS analysis were significantly associated with (more

abundant during) pre-fire conditions included the midges *Orthocladius-Cricotopus*,

Thienemaniella, *Eukieferiella*, *Tventenia*, *Microtendipes*, and *Psectrocladius*; the crane

fly *Tipula*, the water boatman *Graptocorixa*, the caddisfly *Hydroptilla*, and the mayfly

Fallceon (Table 6). This result is highly consistent with results of the ISA analysis (Table

4B). Species that were correlated with the post-fire period were *Dubiraphia*, Ostracoda,

Enallagma, and *Baetis* (Table 6). These taxa are again largely consistent with the post-

fire indicator species identified, although the ISA defined a slightly more extensive

group.

Benthic Invertebrate Trait Responses

    Cluster analysis of the EFJR benthic communities based on composition of

Functional Feeding Groups (FFGs) was less effective than taxonomic composition in

differentiating riffle and pool habitats, although the MRPP showed habitats to be

significantly different based on FFGs (Table 7). Based on FFGs, riffles and pools mixed

at an intermediate distance (Figure 7). FFGs separated the benthic communities according

to the pre- and post-fire periods more strongly than habitat (Figure 7).

Table 7. Multi-Response Permutation Procedure (MRPP) results comparing sample groupings based on Functional Feeding Group (FFG) composition.

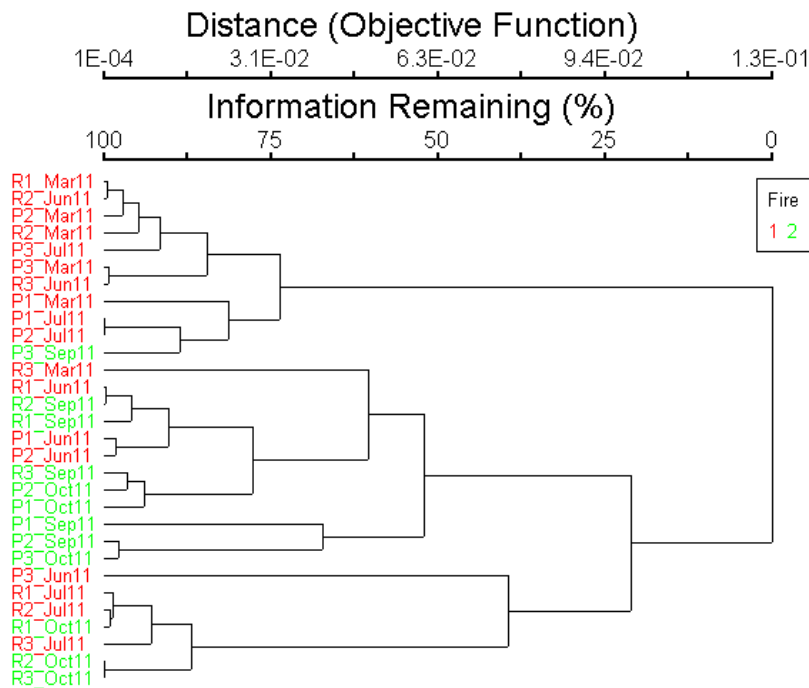| Grouping	T statistic	A*	p
Habitat	-2.60	0.039	0.0230
Fire	-5.81	0.088	0.0004
Month	-6.29	0.201	0.0000
Site	0.19	-0.007	0.5429

*chance-corrected group agreement

Figure 7. Cluster analysis of benthic invertebrate samples from the East Fork Jemez River based on composition of Function Feeding Groups (FFGs), March to October 2011, color coded by fire period (1=pre-fire, 2=post-fire). (Bray-Curtis distance metric, log (x + 1) abundance transformation, flexible beta = -0.25, percent chaining = 6.01).

Indicator values for FFGs showed that filterers and shredders were significantly associated with the pre-fire period, while gathers were indicators of the post-fire period (Table 8). Scrapers and predators showed no significant difference between periods based on this indicator species analysis, which combines abundance and frequency of occurrence metrics. However, when abundances were examined separately by habitat

type, scrapers, which were only slightly less abundant in riffles than pools before the fire, only showed a substantial post-fire decline in abundance in the riffle habitat (Figure 8). Although shredders were more abundant in pools pre-fire, a post-fire decline in abundance occurred in both habitat types (Figure 8). This occurred despite the low overall abundance of shredders in the EFJR, likely reflecting the lack of woody riparian vegetation and associated litter input. Gatherers were more abundant after the fire in both habitats, so the ISA analysis adequately documented their response. Filterers were generally more abundant in pools, and showed a much bigger decline after the fire in pools (Figure 8). Separating the analysis of fire responses by habitat may be important, especially for the more specialized feeders (e.g., scrapers, filterers).

Table 8. Indicator Values (IV) showing Functional Feeding Groups (FFGs; collector-filterers, collector-gatherers, predators, shredders, scrapers) significantly associated with pre- and post-fire cluster groupings.

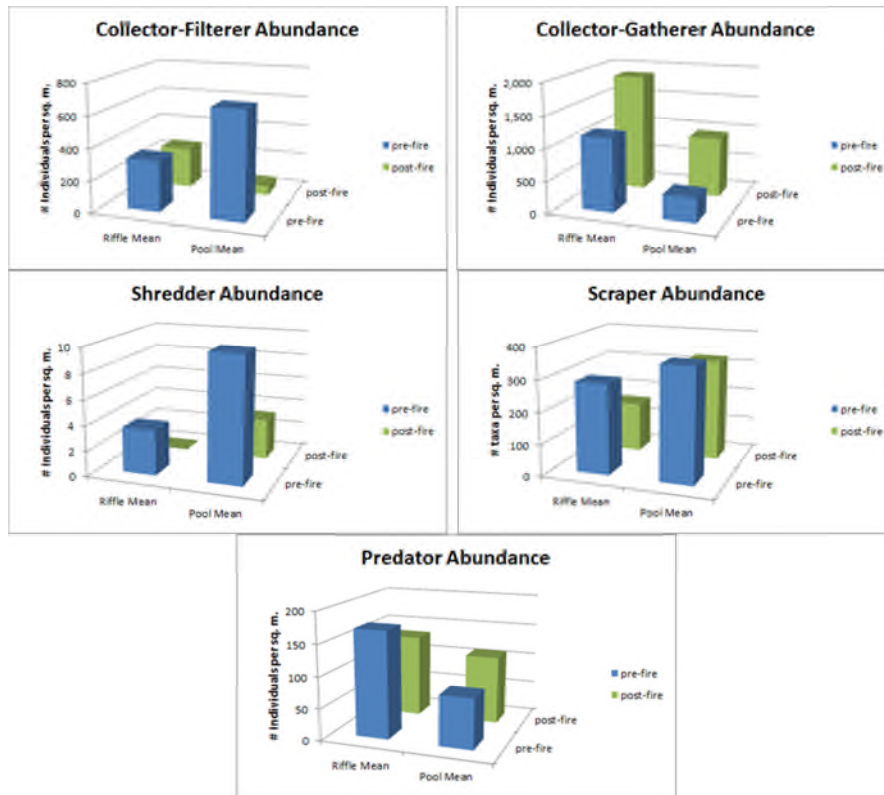| Pre-Fire			Post-Fire		
FFG	IV Pre (Post)	P	FFG	IV Post (Pre)	p
Filterers	55 (45)	0.0036	Gatherers	52 (48)	0.0418
Shredders	52 (1)	0.0134			

Figure 8. Average pre- and post-fire abundances of Functional Feeding Groups (FFGs) averaged separately by habitat type.

Cluster analysis of communities based on habit trait composition separated habitat types (Figure 9), and this habitat grouping was significant based on MRPP (Table 9), but the separation of habitat based on habit is not quite as strong as that based on taxonomic composition. Habit traits do not separate the pre- and post-fire periods fire very well (although the difference is significant in MRPP, Table 9).

Table 9. Multi-Response Permutation Procedure (MRPP) results comparing sample groupings based on habit trait (burrowers, climbers, sprawlers, swimmers, clingers, divers, skaters) composition.

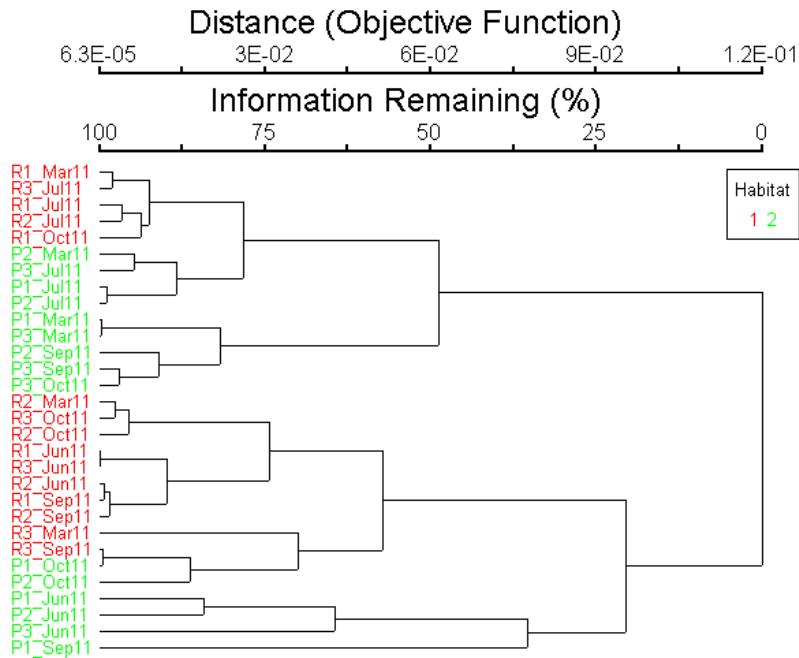| Grouping	T statistic	A*	p
Habitat	-7.76	0.106	0.0000
Fire	-2.16	0.295	0.0375
Month	-4.10	0.118	0.0007
Site	-3.08	0.101	0.0045

*chance-corrected group agreement

Figure 9. Cluster analysis of benthic invertebrate samples from the East Fork Jemez River based on composition of habit traits, March to October 2011, color coded by habitat period (1=riffle, 2=pool). (Bray-Curtis distance metric, log (x + 1) abundance transformation, flexible beta = -0.25, percent chaining = 2.53).

Both clingers and sprawlers had significantly higher Indicator Values for riffles than pools, while climbers were more strongly characteristic of pools than riffles (Table 10). Only burrowers were a significant indicator group with regard to the fire (not shown), with burrowers more abundant pre-fire than post-fire. A weakness of using ISA on trait groups is that all five habit trait groups occur at all sites and sampling periods, so that the frequency of occurrence is always 100%. Thus in the calculation of the IV, the meaningful patterns in the habit types is in the abundance component of the metrics. Overall, habit groups are stronger at separating habitat types than feeding groups, but FFGs are better at distinguishing pre and post fire periods.

Table 10. Indicator Values (IV) showing habit traits (burrowers, climbers, sprawlers, swimmers, clingers, divers, skaters) significantly associated with habitat (R=riffle, P=pool) cluster groupings.

Riffles			Pools		
Habit	IV* R (P)	P	Habit	IV* P (R)	p
Clingers	56 (44)	0.0002	Climbers	55 (45)	0.0080
Sprawlers	52 (48)	0.0072			

* Indicator Value = (abundance in group/sum of average abundances across all groups) * (# of samples of taxon present/total # samples)

If the average abundances of clingers and climbers, both strong indicators of habitat types (riffles and pools, respectively), are separated by habitat type to compare pre- and post-fire periods, it is clear there is no strong response to the fire by either of these trait groups (Figure 10). Sprawlers and burrowers were both responsive to fire effects, increasing in abundance post-fire in both habitat types (Figure 10). The magnitude of response for the burrowers was much larger in pool habitats. Finally, swimmers decreased in abundance post-fire, but only in the riffle habitat, even though the abundance of swimmers was similar between habitat types before the fire (Figure 10).
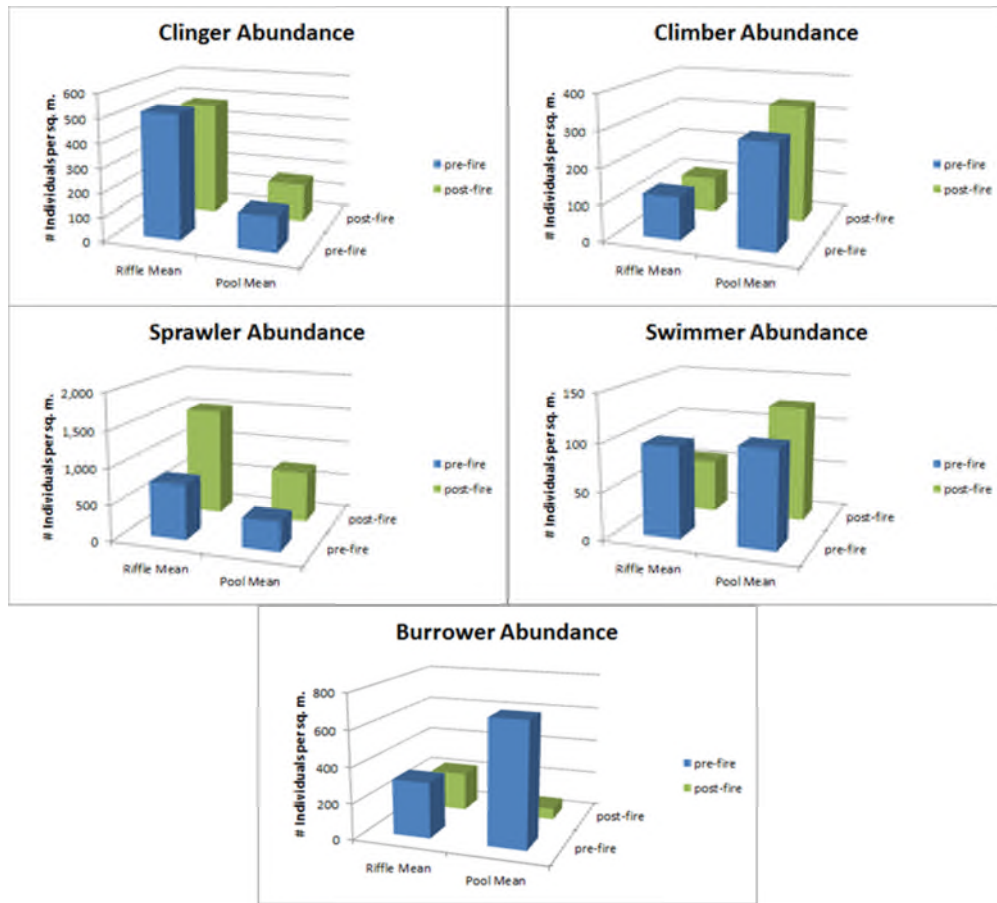
Figure 10. Average pre- and post-fire abundances of habit trait groups averaged separately by habitat type.

Given the evidence that scouring by post-fire flow pulses was influential in reducing macrophyte biomass and potentially determining benthic invertebrate responses, the responses of invertebrates classified based on their rheophily, or flow preference, traits was examined. Depositional taxa were only slightly more abundant in pools compared to riffles before the fire, which may be attributed to the embedding of riffles with finer sediments found in the EFJR (Figure 11). After the fire, depositional taxa were less abundant in pools than before the fire, but more abundant in riffles, suggesting redistribution and deposition of fine sediments during post-fire flow events. Erosional

taxa, those that do well in higher flows, were more abundant in both habitat types after the fire (Figure 11).
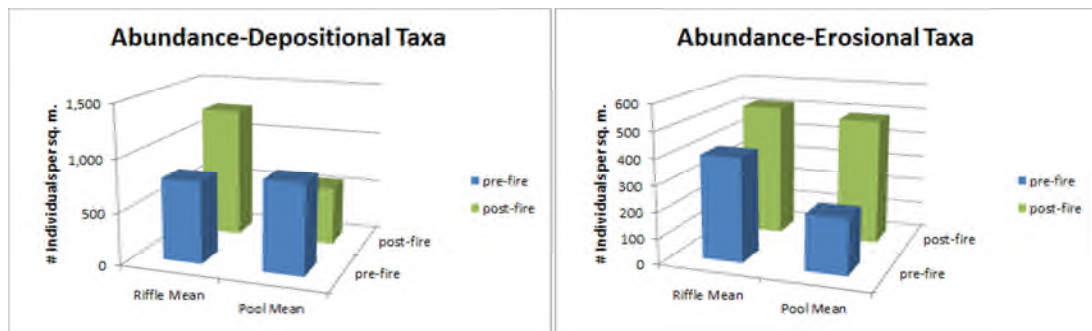

Figure 11. Average pre- and post-fire abundances of rheophily trait groups (preference for erosional or depositional conditions) averaged separately by habitat type.

Key Species Patterns

The abundance patterns of several key taxa show variable response patterns. EPT taxa (Ephemeroptera-Plecoptera-Trichoptera) are often grouped to use as an indicator of habitat condition, because they are considered sensitive to pollution and disturbance (note that no Plecoptera were collected from our study area, so this group is only represented by Ephemeroptera and Trichoptera). Riffle and pool differences in both abundance and richness of EPT taxa are apparent (Figure 12). In both habitat types, EPT taxa increased in abundance but decreased in richness after the fire. However, the abundance response pattern was largely the result of a single taxon, the Ephemeropteran (mayfly) *Tricorythodes*, which increased in abundance late in the year (October), after the fire, and was a community dominant. Many other EPT taxa decreased in abundance in response to the fire, in part reflected in the decreased number of EPT taxa post-fire. These species differences in response are lost when examining EPT as a single indicator metric.
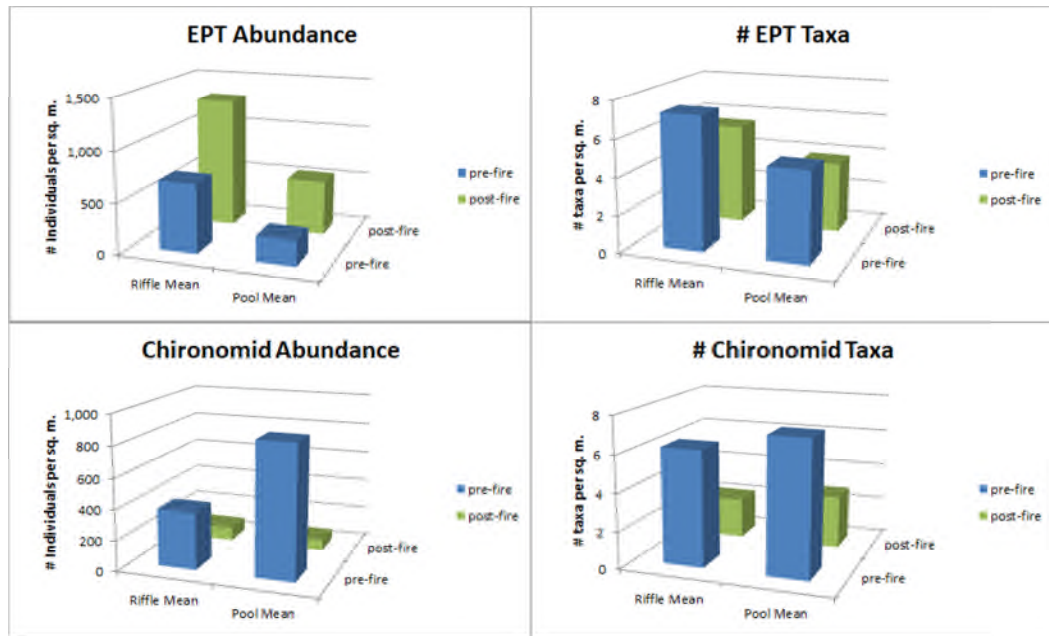
Figure 12. Average pre- and post-fire abundances and richness (number of taxa) of EPT (Ephemeroptera-Plecoptera-Trichoptera) and Chironomidae taxon groups, averaged separately by habitat type.

The Chironomidae (non-biting midges) also are often examined as a group, considered generally tolerant of organic pollution. However in this study, the Chironomidae decreased in both abundance and richness in both habitats post-fire (Figure 12). All but one of the 21 genera identified also decreased in abundance post-fire (*Pentaneura* increased slightly in abundance post-fire).

Discussion

The Las Conchas fire was a very large (63,000 hectare), moderate to severe wildfire that burned much of the East Fork Jemez River watershed upstream and to the east of our study site, without burning directly through the grassland site. Accordingly, although we sampled in July during the burn period but before the onset of the summer monsoons, we neither expected nor observed direct fire effects during that time. Even in streams with

riparian forests that are burned, direct effects from temperature (Mahlum et al. 2011), ash-fall and the like, have been found to be minimal compared to subsequent indirect effects (Minshall 2003, Minshall et al. 2001b, Rinne 1996). Our main focus was thus on indirect, short-term effects of the fire on the EFJR benthic communities based on before-after comparisons.

Unlike most of the forested streams studied for fire effects, the EFJR has an open riparian canopy and good light penetration that supports strong macrophyte and associated algal growth. It is seasonally dominated by aquatic macrophytes, with distinct differences between pools and riffles in dominant species, with typically greater standing stock biomass in pools (Table 1). There is substantial diatom and biofilm growth on riffle substrates, as well as epiphytic diatoms on the macrophytes. The greater abundance of epiphytic diatoms available in pools corresponds well with the greater occurrence of invertebrate scrapers (=grazers) in pools compared to riffles. In addition, the nature of trophic impacts of the fire can be expected to differ from those commonly found in other forested streams with a well-developed riparian overstory and inputs of allochthonous litter.

Water Quality Responses of the EFJR to the Fire

There were distinctive effects of the Las Conchas fire on the water quality of the EFJR that we could document and link to post-fire runoff events at a fine temporal scale not commonly possible due to the deployment of continuous real-time sensors. In-stream responses in turbidity, DO, and pH are clearly associated with monsoon flow pulses that followed the fire (Figure 13). The long period of record of continuously recorded data

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during the pre-fire period documents the typically low turbidity of the EFJR, with a characteristically low range of variability. Although both DO and pH showed expected seasonal patterns during the first half of 2011 (e.g. increase in mean daily DO following ice-off, steadily increasing pH from winter through the spring and early summer reflecting increasing biological activity), the ranges of variation in daily mean DO and pH were small and fairly consistent during that pre-fire period (Figure 13). Interestingly, there was a significant rainstorm on June 24, just prior to the fire start, that had little corresponding effect on runoff, and was associated with only a minor increase in turbidity (<95 NTU). Over the period that the Las Conchas fire burned through the Valles Caldera (early in the fire period shown by the red rectangle in Figure 13), during which there were no major rainfall events, there were minimal if any water quality responses observed, with the possible exception of a very small increase in the range of turbidity variation that may reflect inputs of ashfall (though all daily means remained <90 NTU). However, with the first major monsoon rainfall event following the fire (actually two closely spaced events on August 3rd and August 6th), we observed distinct and rapid responses (by August 4th and 7th respectively) in runoff and water quality. On these first paired events, the increase in daily mean flow was modest (from a daily mean on August 3rd of 0.09 m^3/s to 0.11 m^3/s on August 4th, with similar magnitudes on August 6th and 7th). However, the increase in turbidity was distinct (from 22 to 340 NTU on the first pulse, 23 to 228 NTU on the second pulse). There were associated pulse depressions in mean pH (from about 9.3 to 7.4 on the first pulse, with a lesser change of 8.8 to 7.8 on the second pulse), which also greatly reduced the daily range in pH, and in DO (to a daily mean of 3 mg/l).
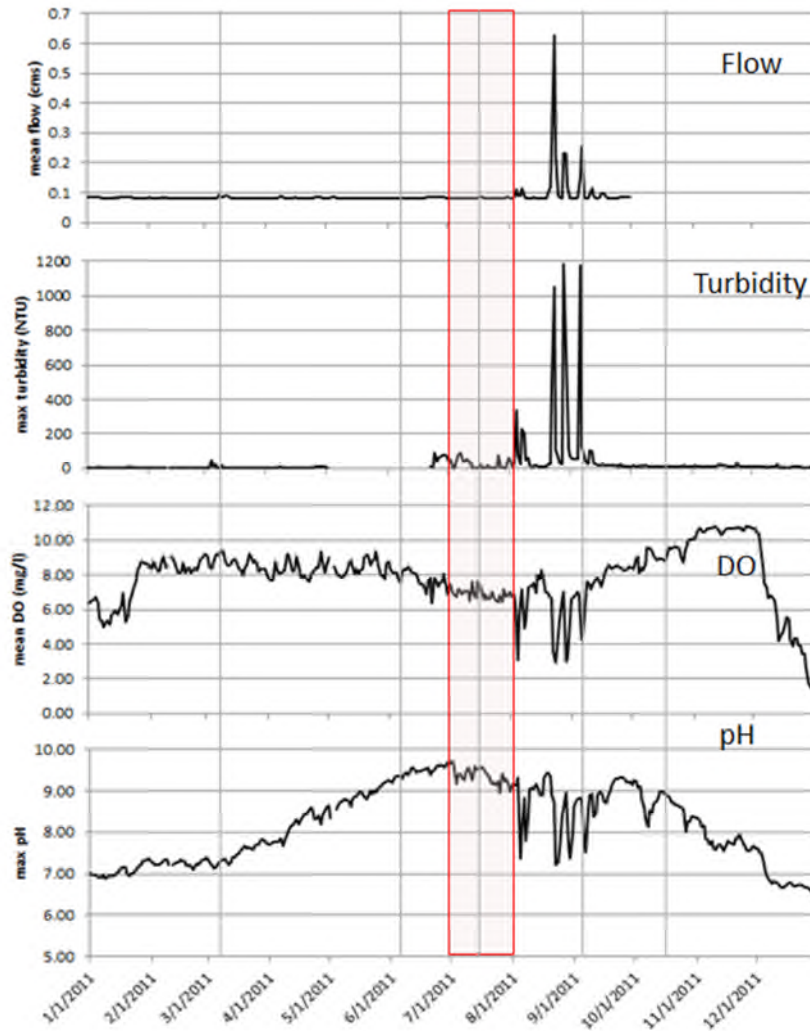
Figure 13. Daily values of mean flow, maximum turbidity, mean dissolved oxygen (DO), and maximum pH in the East Fork Jemez River during 2011. Sampling events are indicated by grey vertical lines. The red rectangle highlights the period of the Las Conchas fire.

There were several subsequent significant monsoon rainfall events that were rapidly followed by very high runoff pulses (e.g., an August 23rd flow pulse of about 0.63 m^3/s [approximately 7.6 times the pre-fire average flow rate], an August 28th and 29th pulse of 0.23 m^3/s, and a September 6th pulse of 0.34 m^3/s). Each of these runoff pulses were associated with sharp increases in turbidity, to levels >1,000 NTU (Figure 13). Turbidity is actually measured based on light scattering, and so is not always reliably associated

with suspended sediment load. Nevertheless, we believe the high turbidities are indicative

of the high levels of ash, suspended sediment, and charcoal that were eroded from the

watershed in these post-fire rainfalls and transported by each of these high-pulse flow

events in the EFJR both upstream of and within the study area (see Figure 14). Rapid

revegetation of the burned grasslands was observed within the VCNP, as illustrated in

Figure 15, suggesting that these eroded materials were not coming primarily from the

surrounding grasslands, but rather from the upstream, severely burned forested portions

of the EFJR watershed.

Figure 14. Photo upstream of the East Fork Jemez River study reach, taken in August
2011 immediately following the Las Conchas fire, during a flow pulse resulting from a
monsoon precipitation event, showing high flows with high loads of debris, ash, and
sediment.

Figure 15. Photos of a Valles Caldera grassland area immediately after the Las Conchas fire burned through (July 6, 2011) and one month later (August 3, 2011) showing rapid recovery of the grasslands.

Responses in other water quality parameters that are biologically significant were also observed to be linked to these post-fire flow pulses. Sags in dissolved oxygen (DO) occurred with each flow pulse, reducing DO to daily averages of about 3 mg/l that persisted for two to three days (Figure 13). These sags included values below 6 mg/l, the NMED standard for high-quality cold waters, for three to four days. During the August 22-24 event, all 15-minute interval DO measurements remained below 4 mg/l for the entire 3-day period, with periods during the peak flows of DO below 3 mg/l, and often down to 2.1 mg/l. Such extended periods of DO sags are likely to be biologically stressful. Maximum pH values during this (and subsequent) pulse events were depressed from about 9.4 to 7.4 (Figure 13). These lower pHs are still circumneutral, so it is not clear whether this relatively large pH change would be directly stressful to macroinvertebrates, or whether the pH response is an important indicator of pulse-associated alterations in stream metabolism processes. Similar responses (not shown) in major nutrients, respiration, and productivity also were observed (Sherson 2012, Shafer 2013). Many of these parameters were significantly associated with invertebrate taxonomic and trait community responses to the fire. Several other studies have documented a substantial increase in post-fire runoff associated with rain events (Veenhuis 2002, Minshall et al. 1997, 2001a and b), post-fire changes in channel morphology resulting from flow alterations (Benda et al. 2003), and impacts to benthic and other aquatic communities that are mainly associated with post-fire flows (Minshall 2003, Earl and Blinn 2003, Rinne 1996). Few, however, have documented the close

linkages among rainfall and runoff pulse events and water quality parameters, nor the

temporal duration and severity of the water quality alterations. This study shows the

extent to which a fire of the severity of the Las Conchas burn can impact stream water

quality, linking severe fire impacts in the upper forested watershed to downstream water

quality impacts through pulsed monsoonal flow events affecting turbidity, DO, pH, and

nutrients.

Characteristics of the EFJR Benthic Communities

Our study catchment includes substantial areas of subalpine and mid-elevation

coniferous forests. About 63% of the EFJR catchment is forested (NMED 2009), but the

study stream reach itself runs through a broad, high-elevation meadow, and has no

natural woody riparian vegetation. Correspondingly, shredders comprise only a small

proportion of the community (0%-1.4%) in both habitats and all months. This is

somewhat low for a high elevation, second to third-order stream, but not inconsistent

with expectations for reduced shredder abundance in the transition from an allochthonous

to autochthonous energy base (Vannote et al. 1980, Cummins 1974). The dominance that

we observed in the EFJR by collectors (considering both gatherers and filterers),

averaging 70% of the fauna in pools and 80% in riffles (see Figure 4), is perhaps high

compared to the general expectations set by the River Continuum Concept (Vannote et al.

1980), and correspondingly, there are somewhat fewer scrapers (=grazers), averaging

about 24% of the fauna in pools and 12% in riffles, than would be expected in a system

like the EFJR with high autotrophic production. These patterns are difficult to explain,

and could reflect variation or uncertainty in assigning taxa to feeding types, or switching

in feeding mode with age, as well as the possibility that despite the high autochthonous

production, a substantial proportion of the energy base in this stream reach may come

from detritus washed into the stream from the surrounding grasslands, or from in-stream

processing of decomposing macrophyte material.

We found no stoneflies in our study reach of the EFJR, although stoneflies are a

common faunal component of the Jemez Mountains, as well as other mountain streams in

the west (e.g., Ward et al. 2002) and southwest (e.g., Molles 1985) and have been

collected from upstream reaches of the EFJR. Many stoneflies are shredders, and many

are considered cold-preference taxa, so their exclusion from the study reach could reflect

responses to the condition of these factors in the EFJR. The EFJR in the vicinity of the

study area was listed in 2001 as exceeding water quality standards for its designated use

(high quality cold water aquatic life) for temperature (temperature of 20°C not to be

exceeded for four or more consecutive hours in a 24-hour period on more than three

consecutive days or a maximum temperature of 23$^{\text{o}}$C), as well as for pH (range of 6.6 to

8.8), DO (minimum of 6.0 mg/l), turbidity (no increase >10 NTU when background is

≤50 NTU), and coliforms (the monthly geometric mean of *E. coli* bacteria 126 cfu/100

mL or less, single sample 235 cfu/100 mL or less) by the New Mexico Environment

Department (NMED) (Simino 2002). Based on monitoring in 2005 by NMED, the EFJR

was again listed as impaired for temperature as well as arsenic (NMED 2009). Simino

(2002) specifically sites the legacy of grazing impacts as contributing to higher

temperatures, because undercut banks that are lost due to trampling and increased erosion

would normally contribute shaded habitat and lower water temperatures. Flow velocities

also decrease with grazing and erosion, which contributes to increasing water

temperatures, while protection from grazing results in recovery of the stream profile and increasing flow velocities (Van Horn et al. 2012). NMED indirectly recognizes the lack of shading and increased stream width as contributing to higher temperatures (NMED 2009). These existing water quality degradations, as well as the very low gradient of the stream, may contribute to the absence of stoneflies in the study area. There are other faunal differences between our EFJR study reach and other high-elevation streams in New Mexico. For example, there is essentially no overlap in the genera of Trichoptera (caddisflies) collected in the EFJR and those reported by Molles (1982) for two forested, first to second order streams in the southern Sangre de Cristo Mountains. This could reflect zoogeographic faunal differences between the Jemez and Sangre de Cristo mountain ranges, in addition to differences expected between forested and open canopy streams.

Some evidence suggests that the riffles and pools in the EFJR may not be differentiated to the extent that would be expected under undisturbed conditions. The EFJR has a history of grazing (as well as logging) disturbance within the Valles Caldera (Maldavin and Tonne 2003, Anschuetz and Merlan 2007) that results in widening and shallowing of the stream profile, erosion of stream banks including loss of undercut banks, and increased sediment runoff that fills in pools and embeds the gravel to cobble riffle substrates. In a 2001 survey of the EFJR, Simino (2002) found conditions in the upper reaches of the EFJR to reflect this type of disturbance, with eight times more riffle than pool habitat due to sediment infilling, stream widening, and loss of sinuosity. Simino (2002) also attributes the loss of pool habitat to the lack of large woody debris, which contributes to pool formation through induction of local scour. Large woody debris is

absent in the study reach of the EFJR within the Valles Caldera, given the lack of riparian forest in this area. Despite these reported conditions, we were able to differentiate riffles and pool in the EFJR study area, based on geomorphic and hydrologic characteristics, to a degree that proved useful in assessing fire effects on the benthic communities in these habitats. Few studies on the effects of fires on stream benthic communities have examined responses separately by riffle and pool habitat types (but see Earl and Blinn 2003).

Before and After Sampling of Invertebrates in a Fire-Impacted Stream

Most studies of fire effects on stream invertebrates, with the exception of prescribed burn studies (e.g., Beche et al. 2005), rely on comparisons between burn-area streams and reference streams (e.g., Minshall 2001a and b, 2003, Spencer et al. 2003). In contrast, we had established a suite of *in situ*, continuous monitoring instruments examining hydrology, water quality, and water chemistry, as well as regular biological monitoring during the year, when the Las Conchas fire burned through and created the opportunity to examine fire effects and responses within the same stream based on pre- and post-fire comparisons. Another study in Yellowstone Park (Tronstad et al. 2011, 2013) also fortuitously sampled benthos in Cub Creek in 2003, just before the East Fire in 2003 and then one, two, and nine years thereafter. Rinne (1996) also sampled benthos (and fish) in three streams in Arizona before and after the Dude Fire in 1990. In both of these cases, the pre-fire sampling occurred a year before the fire incident, while in this study, pre- and post-fire sampling represented a regular series of approximately 6-week interval samples within one year.

After several pulse flow events that occurred in the EFJR immediately following the Las Conchas fire, many vulnerable taxa were reduced in abundance, altering the post-fire composition of invertebrate communities. However, there were no significant reductions in the total abundance of benthic invertebrates in pools, and density increased somewhat in riffles (Figure 16). In contrast to this, Tronstad et al. (2011, 2013) found the total density of invertebrates decreased in Cub Creek the year following the East Fire, with recovery to pre-fire densities in two years. However, they found that biomass was similar before and after the fire, and increased in subsequent years. In the smaller Little Cub Creek watershed, both density and biomass of invertebrates increased following the fire. In three streams in the Gila watershed in Arizona, Rinne (1996) found no differences in invertebrate density between pre-fire and immediate post-fire conditions prior to flooding events, while after storm-related post-fire flood flows, densities of invertebrates were significantly reduced (by up to 95%). The common theme is the linkage through the timing of storm-associated runoff and fire severity, combined with factors that further modify responses, such as land slope and stream gradient. When there were no major pulse flows following the fire (e.g., Little Cub Creek, Tronstad et al. 2013), or before such flows occurred (e.g., the Gila streams, Rinne 1996), minimal responses from the invertebrate communities occurred. When flow pulses occurred after a fire, the magnitude of benthic responses appeared to be greatest when flows transformed the channels, or flows transport, scoured, and deposited large amounts of materials, including toxic ash in transport (Rinne 1996). Earl and Blinn (2003) also observed reductions in abundance of >95% following a large fire in the West Fork Gila River, attributed largely to substantial ash inputs from the burned watershed.
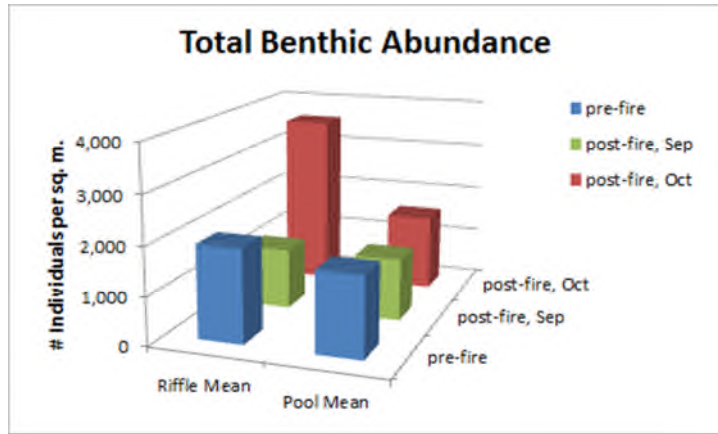
Figure 16. Total abundance (number/m^2) of benthic invertebrates averaged for riffle and pool habitats over the pre- and post-fire sampling periods.

The magnitude of the benthic response in total density in our study was relatively mild, even though the Las Conchas fire was large and severe, storm-driven pulse flows occurred immediately following the fire, and distinct flow pulses with high turbidity and severe water quality effects were documented. The combination of low stream gradient with the displaced exposure to the fire (burning the upper watershed but not the immediate vicinity of the stream itself) combined with the low gradient of the study reach of the EFJR likely contributed to the apparent resistance of the benthos to catastrophic losses. While watershed impacts of the fire were severe enough to cause mass flow of ash and sediment into the upper reach of the EFJR that resulted in burial of benthic habitat with more than 30 cm (one foot) of new sediment, the massive sediment flows did not extend as far downstream as our study area. Slope (basin gradient), along with soil type, area burned, and magnitude of the rainfall event, are predictors of the probabilities of Las Conchas fire debris flow hazards in the Jemez Mountains (Tillery et al. 2011). These data also highlight the often patchy nature of fire impacts to even proximal reaches of streams

(recognized in Minshall 2003), based on factors such as locally variable topographic features, and spatially variable rainfall patterns within a region.

Even though the response in total density of invertebrates was moderate in the EFJR, there were compositional changes in response to the fire in both pools and riffles. Tronstad et al. (2011, 2013) found variable changes in composition among taxonomic groups – ephemeropterans (mayflies) showed no net before-after change, trichopterans decreased in density, and plecopterans (stoneflies) increased. But within these groups, they found species differences (e.g., the mayflies *Rhithrogenia* and *Plauditis* increased following the fire, while *Serratella* decreased). We also found species differences in responses among the EPT taxa of the EFJR. The dominant mayflies *Tricorythodes* and *Baetis* both decreased initially following the fire, and then increased later in the fall, showing post-fire recovery. Other mayflies such as *Fallceon* decreased in abundance post-fire in both habitat types, although they also started to show recovery in October; and *Ameletus* increased in abundance post-fire. Among the caddisflies in the EFJR, *Helicopsyche* and *Hydroptila* were significantly associated with (more abundant during) the pre-fire period, while *Hydropsyche* increased post-fire (though not significantly).

Tronstad et al. (2011) suggested that the variable EPT response was evidence that the responses were not related to water quality, but rather to resistance to floods and tolerance to increased levels of fine sediments. Based on our observations of the recovery trajectories of the mayflies that responded to fire effects (initial large abundance decreases in September immediately following the post-fire pulse flows, followed by increases in abundance one month later), we would suggest that their opportunistic life history characteristics provide a mechanism for post-fire resilience. For example,

Tricorythodes, which has extended reproduction and recruitment throughout the summer rather than a highly synchronized period of reproduction, and *Baetis*, which is bivoltine, both began recovering in abundance in the fall immediately after the monsoon season. In contrast, *Helicopsyche*, which is univoltine with emergence in spring and larval growth and highest densities in summer, remained reduced in abundance throughout the fall. As a clinger on rock surfaces in riffles (erosional environments), *Helicopsyche* was vulnerable to scouring during the post-wildfire pulse-flow episodes, and since these occurred after their main recruitment period, there was little opportunity for immediate recovery.

Tronstad et al. (2013) found that the true flies in the family Chironomidae (midges) increased in Little Cub Creek after the East Fire, and attributed this to their characterization as tolerant taxa. In contrast, we found all genera of Chironomidae that occurred in the EFJR, with the exception of the predaceous tanypod *Pentaneura*, to decrease following the Las Conchas fire, with no recovery yet apparent in the fall of 2011. Given the diverse combination of midges that occurred in the EFJR (20 genera, multiple feeding and life history types), it is not likely that strategies adaptive to the types of fire disturbances observed are common among all of them. As a result, we suggest that tracking chironomids as a singular group would not be informative for assessing the physical and water quality disruptions that the Las Conchas and other fires generate.

Functional Trait Responses to the Las Conchas Fire

Although Mihuc et al. (1996) found that benthic invertebrate responses were individualistic, there is value in investigating whether some combinations of species traits

and habitat conditions may increase the vulnerability of taxa to post-fire effects, and provide predictive power with regard to future and long-term effects. The effects of the Las Conchas fire on the EFJR can be grouped into two different though inter-related categories of disturbance, direct physical disruption, in which we could include direct water quality and trophic impacts. In general, we expected the scouring and sediment deposition associated with post-fire flow pulses to be most likely to generate observable effects within the immediate post-fire timeframe of our study. We hypothesized that life history traits that imparted resistance to dislodgement in strong flows, such as clinging or sprawling habits, or erosional versus depositional rheophily preference, would provide predictive categories to track these physical disturbances. On the other hand, we expected trophic effects of the fire would likely take longer for associated invertebrate changes to manifest in terms of community composition. Although different from most other fire study cases because of the pre-fire open canopy of the EFJR, we expected trophic impacts of the fire to be potentially important but in the longer term. For instance, the loss in macrophyte biomass due to the immediate post-fire scouring pulse flows, the associated loss of epiphytes, and the additional scouring of other (e.g., episammic) diatoms represent a substantial loss of food resources for invertebrates, which could contribute to restructuring of the benthic community. We tested these by tracking the responses of functional feeding types.

Despite expectations, our traits-based community analyses combined with pre- and post-fire abundance patterns by trait groups showed that habit traits were not as effective as functional feeding groups at separating the pre-and post-fire periods. Community composition by habit traits effectively separated riffle and pool habitats, but functional

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feeding groups were more responsive to fire effects within habitat type. This suggests that within the three month post-fire period during which sampling was conducted (August to October), fire-related alterations of EFJR trophic resources influenced the composition of the benthic community.

Tronstad et al. (2011) also postulated that benthic food webs in streams would be altered by fire-induced changes to light levels, nutrient concentrations, and hydrology. It should be noted, however, that the expectation for most streams impacted by severe, stand-replacing fires is for a shift from heterotrophy toward a greater contribution to the energy base by autotrophy. This results from a combination of changes, including the opening of the forest canopy, increased light penetration to support primary production, loss of leaf litter inputs, and increased nutrient inputs. This leads to an expectation for a decrease in shredder abundance and an increase in scraper (herbivore) abundance as a response to decreased leaf litter input and increased algal production following a fire in the near (Minshall 2003) or long-term (Molles 1982). In forested stream case studies, the expected response of feeding types was not consistently observed (Minshall 2003). For example, Tronstad et al. (2011) found that shredders increased in abundance and biomass after the East Fire in Yellowstone, while scraper abundance decreased slightly in the year following the fire, and recovered in the second post-fire year. However, scraper biomass did increase in the year following the fire, consistent with expectations. In contrast, the EFJR already had a substantial autotrophic energy base. Many of the watershed inputs are similar to other forest fire-impacted streams, including altered hydrology, erosion, and nutrient inputs. However, the short-term trophic changes are at first related to the loss of

autotrophs and depression of production rather than to a shift from a heterotrophic resource base.

When examined separately by habitat type, benthic responses to the fire based on functional feeding groups were apparent. Scrapers were strongly impacted in riffles, although not in pools, indicating greater vulnerability to post-fire scouring effects in riffles. While this was consistent with the reduction in abundance of their food resource, we explored whether the immediate post-fire loss of scrapers was due to the loss of their trophic resource (epiphytes) or to dislodgment in scouring flows plus the loss of the habitat structure by comparing pre- and post-fire responses of habit traits instead of feeding types. In the EFJR, scraper taxa are represented by a variety of habit traits including climbers, clingers, swimmers, and one sprawler. However, neither the abundance of climbers (often associated with macrophytes) or of clingers (on rocks, macrophytes, other substrates) declined post-fire. Gatherers are a more general feeding type and are often considered opportunistic, and their greater abundance post-fire is consistent with other fire study results that found opportunists to respond well after fire impacts on streams (Mihuc et al. 1996, Minshall 2003).

Evaluation of habit (i.e., mode of habitat use) traits did highlight invertebrate vulnerabilities that were consistent with the main physical disturbances created by the fire. Within the riffle habitat, swimmers were vulnerable and declined in abundance, presumably reflecting the greater exposure to scour that might be expected in riffles. Burrowers were particularly impacted in pools, although not in riffles, highlighting the potential for peak flow events to scour soft sediments from the pools, and subsequently to deposit transported sediments during recession. However, short-term recovery potential

was not defined by these habitat traits, but rather by the possession of opportunistic reproductive habits that allowed recruitment to occur rapidly following cessation of the large, flashy pulse flows with high turbidity. Species characterized as opportunistic are found to respond favorably after fires (Minshall 2003).

Short-Term Effects, Long-Term Expectations

Historical (pre-1900) fire return intervals averaged 5 to 25 years in the Jemez Mountains, or a watershed-wide fire every 16 years on average (Allen 2002a). Fires were most frequent in mid-elevation ponderosa pines, and periods of fire activity were found to be associated with periods of drought (Allen 2002a). During the twentieth century, fire suppression practices as well as increased grazing altered fire frequency, and resulted in shifts from open forests to dense thickets of smaller trees in the Jemez Mountains, increasing the likelihood and intensity of crown fires, which often result in replacement of the forest with shrublands (Allen 2002a, Allen 1989). The shift to more susceptible landscapes, combined with climate change effects that are increasing aridity in the Southwest (Seager et al. 2007, Gutzler and Robbins 2010), has increased the frequency of stand-replacing fires in the Southwest (e.g., Swetnan and Betancourt 1998, Bachelet et al. 2007). This trend is expected to continue (Running 2006, Westerling et al. 2006). Burned watersheds have altered runoff, erosion, and in-stream flow characteristics that can persist (Robichaud et al. 2000) and cause moderate to traumatic alteration of stream invertebrate communities in the short-term (this study, Minshall 2003, Earl and Blinn 2003, many others). A more frequent recurrence of such traumatic disturbance of montane stream communities can thus be expected in the near term. Disturbance is

recognized as an important factor organizing the structure of stream communities (Resh et al. 1988), but the responses to such restructuring will depend on factors such as the frequency and intensity of the disturbance, the timing of the disturbance relative to the life histories of the resident organisms, and hydrologic characteristics of the stream driven by factors such as regional climate, topography, and geomorphology (Resh et al. 1988). If stream invertebrate community attributes of abundance, species richness, and diversity in burned streams take 10-15 years to return to 'normal' and converge with the values of community attributes from reference (unburned) streams (see Minshall 2003 for summary of pertinent evidence), then major fire disturbances more frequent than an average of once every 16 years should allow disequilibria to persist. Our study results suggest these could include, for example, repeated suppression of certain feeding types such as scrapers. In the absence of fire disturbance, scrapers are sub-dominants in the EFJR, exploiting the abundant autotrophic resources (epiphytic and episammic diatoms). As fire disturbances occur at intervals shorter than observed community recovery times, scrapers may not return to their pre-fire sub-dominant status, despite the availability of exploitable trophic resources. Similarly, we would postulate that taxa with more extended reproduction and recruitment periods or other opportunistic reproduction characteristics would be favored by a higher return rate of fire disturbance. Thus, an increasing fire and associated flow disturbance frequency could result in persistent shifts in benthic community composition and function. While less severe fire disturbances can have minimal effects on benthic communities, with recovery to pre-fire or reference conditions within periods as little as one year (Beche et al. 2005, Earl and Blinn 2003), severe fire disturbance to the watershed that results in traumatic flow disturbance may result in

alterations to the benthic community that are not rapidly recovered and persistent (Arkle et al. 2010).

Overall, we found that responses of stream benthic invertebrates to a major fire disturbance in the EFJR reflected three categories, vulnerability, resistance, and resilience, with different groups of organisms and different trait characteristics comprising each. Even in the short term, vulnerability was observed to both direct physical disturbance, mediated by the flow pulses that followed the fire; and to trophic impacts, that were secondary responses to loss of food resources due to those same flow pulses and associated water quality effects. Resistance to the post-fire physical disturbance of the stream environment was exhibited by a subset of invertebrates with habit traits such as climbers and clingers, that conferred the ability to withstand dislodgement and displacement that would otherwise be expected from the post-fire flow pulses. Resilience, or ability to recover in the short term following cessation of the most prominent post-fire flow events, was conferred mainly by opportunistic life-history traits. This suite of responses suggest the mechanisms through which benthic communities may be altered in the long-term, through suppression of some vulnerable taxa, partial if not temporary (short-term) suppression of some trophic resources, and possibly incomplete recovery (relative to pre-fire conditions) based on the reproductive life-history characteristics of component taxa.

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CHAPTER 5

Summary

This dissertation is about increasing our understanding of how climate changes and extreme events affect stream invertebrate communities, with the goal of linking this understanding to how stream invertebrates are used as bioassessment indicators and responders to disturbance. Invertebrates are widely used as tools to assess conditions within stream ecosystems, and as well, the process of bioassessment is widely applied in the U.S. and around the world for resource management to determine the ecological consequences of environmental stressors. Woodward et al. (2009) expressed concern about the ability of biomonitoring to discern and account for degradation due to novel stressors using procedures and indices developed under a system that focused on documentation of organic pollution. This dissertation sought to further our comprehension of how a novel stressor – climate change – and associated extreme events influence stream benthic communities and the metrics and indicators we derive from them, in a way that helps us advance the development of specific strategies to ensure the long-term effectiveness of stream monitoring and management plans.

Chapter 1 introduced expectations for climate changes and how they relate to stream ecosystems, as well as the basic concepts of bioassessment, as background for why the influences of such a major environmental driver should be accounted for in terms of its effects on the components of bioassessment. It related changing climate conditions in the southwestern U.S. to increased frequency of severe fires, and framed the question of how the short-term effects of such extreme events may overlay on expectations for long-term climate change-related stream responses.

Chapter 2 examined the vulnerabilities to climate change of stream invertebrate metrics and associated multimetric indices. We found that temperature-related responses of benthic indicators were variable among sites and ecoregions, but in general, cold-preference taxa decreased, and in some areas, warm-preference taxa increased with increasing temperatures. Several traditional taxonomically based bioassessment metrics are composed of both cold- and warm-water-preference taxa. The opposing trends in these temperature preference groups with climate-induced changes in stream temperatures create the potential to miss-associate indicator change with stressor source, and thereby undermine the assessment of stream condition based on those metrics, if uncorrected. Our results suggested that elevation is one factor that drives the temperature-trait composition of regional benthic communities and associated metrics, affecting the vulnerability of metrics and indices to climate change. Moderate but significant correlations between temperature sensitivity and sensitivity to organic pollution meant that at least some metrics commonly utilized as indicators of conventional pollutants also are likely to be sensitive to changes in temperature, making it potentially difficult to distinguish these stressors. The magnitude of responses of various taxa and metrics to long-term changes in temperatures was sufficient to potentially alter reference communities to a degree that would affect decisions about site condition. Site condition could be deemed degraded or even improved simply because of taxon-specific responses to climate change. It may be feasibility to modify metrics by partitioning components based on temperature sensitivity to reduce the likelihood that responses to climate change would confound the interpretation of responses to impairment from other causes and to facilitate tracking of climate-change-related taxon losses and replacements.

Chapter 3 addressed the implications of climate and other global changes for resource management and decision making in the regulatory context of the maintenance of water quality and ecological integrity. It is clear that ignoring the impacts of global changes on biological indicators and reference-based comparisons will likely lead to increasing erroneous decisions. Impacts of climate change combined with other global stressors (increasing developed land uses and population) negatively impact reference locations and induce 'reference station drift', making it increasingly problematic to find an ''undisturbed'' water body to define acceptable conditions of ecological integrity. This recommended the use of a more objective scale for characterizing reference conditions, as well as consideration of strategies to protect existing reference locations. Distinguishing taxonomic attributes based on temperature or hydrologic preferences can be used to enhance the ability to make inferences about global change effects compared to other stressors. However, it is difficult to categorize invertebrate indicators to uniquely reflect climate change influences, due to the several common types of organism responses arising from different types of conventional and global stressors. Our ability to implement water policies through bioassessment will require a shift in the scale of assessment, planning, and adaptations in order to fulfill our ultimate regulatory goals of preserving good water quality and ecological integrity.

Chapter 4 examined the short-term effects of a major wildfire on stream invertebrates. In the overall context of this dissertation, wildfires represent an extreme event that is projected to increase in frequency and severity due to climate change, with implications to structural and functional responses of stream invertebrate communities. Our results provided evidence of the extent to which a fire of the severity of the Las

Conchas burn can impact stream water quality, and established a linkage between the severe direct fire impacts in the upper forested watershed to downstream water quality impacts through pulsed monsoonal flow events affecting turbidity, DO, pH, and nutrients. These indirect fire effects link to patterns of benthic responses discerned in feeding type and habitat traits. These responses reflect components of vulnerability to both physical and trophic disturbances, resistance to disturbance, and resilience following disturbance. These are mechanisms through which an increasing frequency of fire disturbances with climate change, and the resulting flow and water quality responses, could lead to persistent invertebrate community alteration through the maintenance of disequilibria.

While such a short-term study is disconnected in temporal scale from the types of long-term effects of climate change on stream invertebrates considered in earlier chapters, it is clear that the long-term, chronic effects of climate change on stream communities will be punctuated by this type of acute, potentially community-restructuring disturbance. At present, this and many other studies of long-term climate change effects consider the more predictable (e.g., through long-term linear trend analysis like that used in Chapter 2) patterns of population and community changes that are expected in response to climate-change driven changes in water temperature and stream flow patterns. Acute episodes such as the fire disturbance studied here are expected to have more dramatic impacts on stream communities in the short term compared to climate change. Such episodes may alter the expected trajectories of climate change-related responses by, for example, altering the types or traits of species in the short term that remain to respond.

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