Environmental Flow Analysis for the Marcellus Shale Region

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Forward

Researchers have grappled for decades with how to characterize ecosystem responses to human activities in ways that are useful for conservationists and environmental planners. While lots of ideas have been proposed, most have either languished as untested concepts or have struggled for clear and reproducible strategies for implementation. What makes this work exemplary is the way it innovatively combined disparate empirical datasets of hydrology and ecology with hydrological modeling to generate actionable information about how aquatic ecosystems are likely to respond to water withdrawals from regional streams. I do not want to minimize or "gloss-over" the very real challenges that persist, many having to do with objectively determining spatial scales over which information can be aggregated. But this effort constitutes one of the few successful attempts to bridge theory-to-application in the context of environmental flows. Furthermore, this group of researchers have provided a clear roadmap that allows their approach to be replicated for other regions.

One enormous barrier has been the fact that the empirical data necessary to implement the proposed ideas are, for the most part, simply not available in the tidy formats envisioned by researchers. In the case of this project, which generally tried to follow the Ecological Limits of Flow Alteration (ELOHA) framework proposed by the highly-esteemed aquatic ecologist LeRoy Poff and his colleagues, the researchers looked for sites where both hydrological and ecological data were available. Such data were highly rarified.

In the face of this challenge, researchers have often proposed that simulation models could be used *in lieu* of field measurements and observations; indeed, this has been proposed by ELOHA researchers. Here the researchers, many of whom are accomplished hydrological modelers, have thoroughly tested and shown that this is not currently a viable approach, at least in the geographical context of the Marcellus Gas Shale region of the United States. The reasons they were not able to model their way out of this dilemma are likely due to both limitations of the models they used and the sparse and biased data available to run and test their models.

So what are the lessons learned? First, this research team has demonstrated that there are tremendous opportunities to utilize available, though disparate, environmental data to develop scientifically defensible management rules. Researchers usually develop monitoring strategies that are optimized to answer a specific research question; such data are generally not available for regional-scale, ecosystem issues that we must deal with today and in the future. Second, it is clear that simulation models can play important roles in developing conservation strategies, but they need to be used appropriately and cannot, currently, substitute for empirical observations. Third, this project has shown, not for the first time, that transdisciplinary teams are necessary to address complicated social-environmental problems. Indeed, future phases of this research will need to engage economists and social scientists to fully implement findings from this work. And resources focused on projects, like this one, which intend to fully translate scientific theories into actionable environmental strategies, are likely to provide on-going opportunities to extend the usability of the information generated. I sincerely hope to see the framework and information developed here transferred into user-friendly tools that allow conservationists and environmental planners to utilize the insights from this project to make management decisions.

As a final note, I would like to acknowledge the amazing altruism that made this project possible. Few of the researchers received financial support from the funding organization to participate and lend their expertise. This suggests sincere willingness, maybe desire, for collaboration among environmental scientists as opposed to the too-often perceived competition among scientists. Also, and equally important, was the Appalachian LCC's leadership to directly engage with this gifted group of researchers to evaluate and allow changes in the project direction as new information was revealed. This point cannot be overemphasized. In short, this project exemplifies an ideal collaboration among researchers and between a research team and its funding source.

This project constitutes a model for future similar projects in a myriad of ways.

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Executive Summary

Human alteration of natural flow regimes has long been identified as a leading threat to surface water resources and aquatic ecosystems in the United States (USEPA, 1998). The most conspicuous and commonly studied sources of alteration are typically dams and water withdrawals associated with agricultural operations and industrial consumptive uses. The advent of horizontal hydraulic fracturing has led to a proliferation of natural gas drilling in the Marcellus Shale regions of West Virginia and Pennsylvania, with expansion likely in the neighboring states of Ohio, New York, Virginia and Maryland. The development of a typical gas well requires substantial amounts of water - often obtained via surface water withdrawal from nearby streams. Given the current extent of gas development and the fact that growing energy needs will continue to drive further development, hydraulic fracturing may pose an additional, potentially critical threat to aquatic biota in the region. Despite the serious implications of continued gas development, little guidance currently exists for water resource managers who are tasked with balancing human and ecosystem water demands. This project aims to establish ecologically-defined guidance on allowable water withdrawals for hydraulic fracturing through the application of the Ecological Limits of Hydrologic Alteration (ELOHA) approach.

The project has been divided into two phases. Phase I reviews existing tools and approaches, as well as gathers and formats available and relevant data within the Marcellus Shale Region (MSR); Phase II applies the most appropriate tools to: i) build a hydrologic foundation, ii) estimate flow alteration, iii) develop flow-ecology relationships to provide guidance for establishing quantitative limits to water withdrawals associated with gas development and iv) conduct a pumping scenario analysis to assess the potential hydrologic effects of water withdrawals and to inform a water resource risk assessment in the MSR. This document summarizes progress made towards the completion of both phases.

Phase I

One of the first steps in applying ELOHA involves establishing a hydrologic foundation of daily streamflow data for every stream reach under baseline (natural) and current (altered) conditions for a single time period. We investigated the suitability of two means of constructing a hydrologic foundation: (i) process-based hydrologic models, which simulate the dominant hydrological processes within a watershed through physically-based equations and (ii) an empirical approach which uses statistical relationships between hydrologic metrics calculated at gaged basins and physical basin characteristics (e.g. slope, elevation) to predict natural hydrologic indices across all basins of interest. In this way, a hydrologic foundation is built such that all streams of interest within the MSR will have estimated flow indices under reference and non-reference conditions.

The Marcellus Shale Region is roughly 172,000 km2, has well over 135,000 streams/subbasins (Horizon Version 1; USGS, 2014), and encompasses six physiographic provinces and 79 Ecoregion IV zones. We found that attempting to apply a physically-based model across such a large and hydrologically diverse area at time-steps and periods relevant to both hydraulic fracturing effects and the ecology of interest (e.g. daily time-step and greater than 15 yrs of simulated flow) would prove prohibitively challenging in terms of parameterization and computation requirements, as well as regionalization of calibration parameters. A careful review of existing hydrologic models, as well as consultation with other researchers who have attempted similar regional model applications confirmed that process-based hydrological models may not be the most appropriate means of establishing a hydrologic foundation for the MSR. However, the Soil and Water Assessment Tool (SWAT) model was identified as being suitable for performing case-study analyses, such as assessing the local and cumulative effects of water withdrawal for gas development using a gradient of pumping scenarios in select study catchments. As an alternative to hydrologic modeling, we proposed to develop and apply statistical models which first calculated hydrologic indices (HIs) at all reference and non-reference gages, then use statistical relationships between the HIs and characteristics of their respective basins to extrapolate relevant flow metrics to basins of interest – thereby establishing a suitable hydrologic foundation.

In addition to a review of modeling approaches, we compiled a list of georeferenced stream gage databases for the Marcellus Region. The stream gage database includes both baseline and altered gages which will be used to characterize flows, classify river types, quantify flow alteration, relate ecological responses to flow alteration, and evaluate the status of sites relative to environmental flow standards. Through coordination with representatives of various state and federal agencies and accessing of online databases, we also compiled a georeferenced stream biological database which will be useful for calculating relevant ecological metrics. All datasets are in standardized format and can be made publically available.

Phase II

Using the statistical modeling approach identified in Phase1, along with relevant hydrologic and ecological data, the second phase applies the ELOHA framework to provide guidance for the establishment of quantitative limits to water withdrawals associated with gas development. In addition to detailing the major steps of our application of the ELOHA process, Phase II includes a pumping and risk analysis designed to ascertain the nature and degree of potential hydrologic impacts from gas-related water withdrawals, and predict which streams within the MSR may be at highest risk to flow alteration.

Methods and Findings

The environmental flow assessment for the Marcellus Shale Region involved seven major steps: i) building a hydrologic foundation, ii) estimating flow alteration, iii) selecting flow metrics, iv) calculating ecological metrics, v) stream classification, vi) developing flow-ecology relationships, vii) performing a pumping scenario and preliminary risk analysis. These analyses and associated datasets should assist in the formulation of scientifically and ecologically sound management strategies within the MSR.

Hydrologic Foundation

One hundred and seventy-one hydrologic indices (HIs) were computed for 198 reference and 373 non-reference USGS gaging stations using mean daily discharge data and the USGS HIT program. Following the recommendations outlined in Phase I, we then applied random forest (RF) models to predict natural flow indices at all non-reference flow gages. The RF models performed well, providing reasonable estimates of the expected (natural) magnitude-, duration-, and rate-of-change-related flow metrics. Thus, a hydrologic foundation was built consisting of predicted natural and observed altered flow metrics at all 373 non-reference gages.

Flow Alteration

The percent flow alteration in each flow index was determined as the observed – expected / expected *100. Consequently, positive percent alterations represented an increase in a particular index, while negative alteration indicated a reduced value. We observed that most HIs experienced both negative and positive alteration, and that the degree and direction of the alteration was often influenced by catchment area. For instance, the observed reference and non-reference mean annual flows (MA1) diverged at both smaller and larger basin sizes (Figure ES-1). Anthropogenic influences tended to augment mean annual flows in basins less than ~500 km^2 and decrease MA1 otherwise.

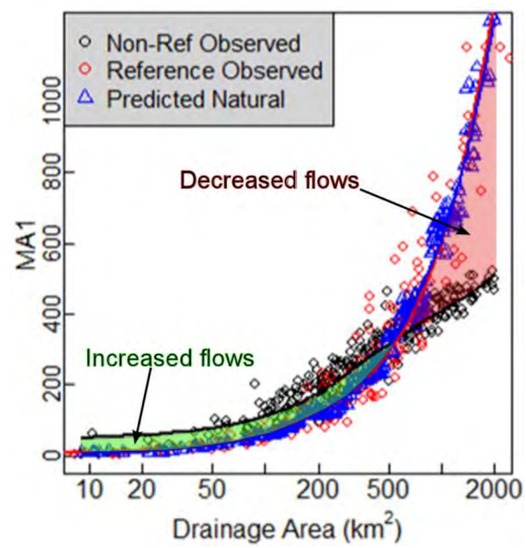

Figure ES-1. Observed non-reference and reference, as well as predcited referecne (natural) mean annual flows as a function of drainage area.

Flow Index Selection

We used the pseudo-R^2 values of the RF models to assess model performance and as a criterion for the selection of flow metrics for further analysis. Using a R^2 threshold of 0.8 reduced the suite of potential metrics from 171 to 60. The remaining 60 HIs covered the major facets of the natural flow regime. However, other than flow constancy and predictability, flow timing-related HIs were not well predicted by the RF models. The remaining HIs were winnowed down to a more tractable set of 28 by considering both the metric's sensitivity to modeled water extraction and its importance according to ecological theory. A subset of monthly flow metrics was retained to characterize spring, summer, fall and winter flows as ecological responses to hydrologic alterations are highly seasonal (DePhilip and Moberg, 2010, 2013).

Ecological Metrics

We focused on fish populations in the MSR as: i) they have been shown to respond more predictably to anthropogenic flow alteration than macroinvertebrates or vegetation (McManamay et al., 2013; Poff and Zimmerman, 2010), ii) data availability and spatial coverage was better in the MSR, iii) fish are a highly valued resource and therefore represent a more charismatic biological endpoint that facilitates meaningful communication with the public and iv) fish encompass a wide range of life history characteristics, which helps reveal long-term disturbances to aquatic ecosystems over broad spatial scales (Karr, 1981, Barbour et al., 1999).

Using 186,518 fish sampling record compiled from various state and federal agencies, we computed 18 different ecological metrics at 11,104 different sampling sites throughout the study region. Ecological metrics covered a range of fish community and assemblage information, including: species richness, composition, tolerance to disturbance, trophic structure, life history strategies and functional guilds. For each ecological metric we provided hypotheses that linked them to changes in flow based on documented and theorized ecological responses to altered flow regimes.

Stream Classification

Classifying streams into hydrologic types is a fundamental step in the ELOHA process as it is thought to reduce natural variation in fish communities, thereby rendering flow-ecology relationships more predictable and statistically significant. All gaged streams and rivers in the MSR were classified into similar hydro-types using a Bayesian mixed-model approach based on principal component scores. We then used the clustering results to predict stream types at all ungaged basins using random forest models. All streams were classified into one of four categories listed and described in Table ES-1.

Table ES-1. Stream class names and narrative description (McManamay et al., 2014b).

Class Name	Description	Hypothesized Sensitivity to Water Withdrawals
Stable High Baseflow	High Baseflow Index, Low Variability, High minimum & low flows, low frequency of high flow events, low rise rates	Low
Perennial Runoff 1	Similar to SHBF but lower baseflows, semi-stable	Low-Moderate
Perennial Runoff 2	Similar to PR1, but lower baseflows and higher runoff than PR1	Moderate
Perennial Flashy	High variability, some intermittency, low minimum & baseflows, high frequency of high flows, high rise rate	High

Based on the hydrologic characteristic of each class we hypothesized the different degrees of sensitivity to flow alteration due to water withdrawals associated with hydraulic fracturing activities (Table ES-1). For example, given the high flow variability, propensity for intermittent flows and low minimum and baseflows, perennial flashy streams were theorized to be the most sensitive to water extraction. Variable importance analysis of all explanatory variables in the RF models revealed that baseflow index, drainage area, average temperature, mean elevation and percent of basin with poorly drained soils were the most influential predictors of stream class.

RF prediction results were mapped to all NHD stream lines in the Marcellus Region (Figure ES-2). The majority of basins were categorized as "Perennial Runoff" which is consistent with McManamay et al. (2014).

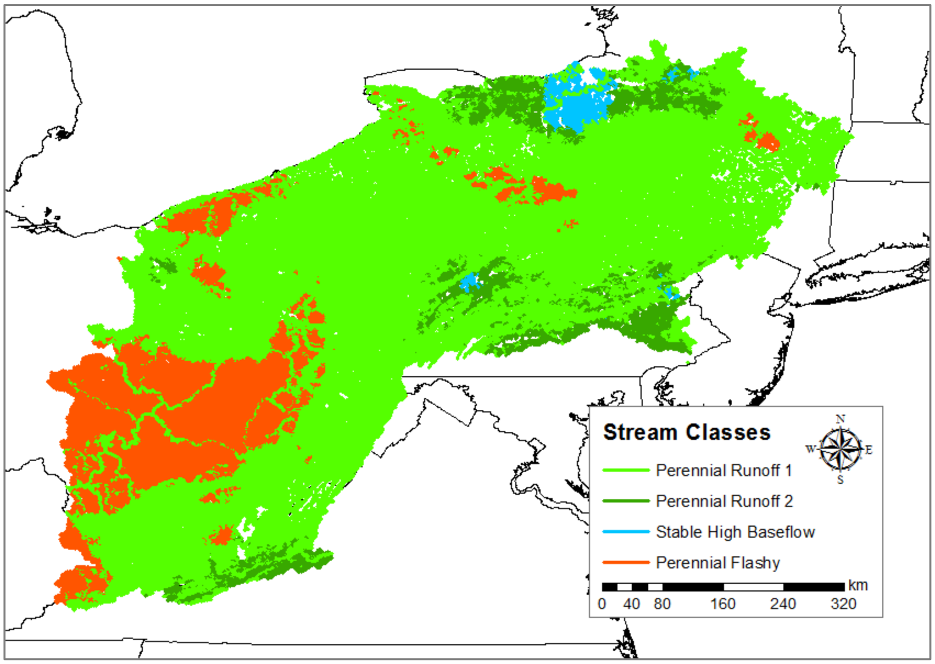

Figure ES-2. Stream classes predicted across all gaged and ungaged basins in the MSR by random forest models.

Flow-Ecology Relationships

Flow-ecology relationships were constructed using multivariate quantile regression. Drainage area was introduced as an explanatory variable in order to control for the potentially confounding effects of stream size on F-E relationships. Significant F-E relationships (90^{th} quantile regressions with a p-value < 0.05) covered a range of fish assemblage and structure metrics, as well as a variety of seasonal and annual flow statistics. The vast majority of statistically significant F-E relationships were associated with negative flow alteration and generally resulted in diminished ecological metrics. Figure ES-3 provides a representative F-E relationship, which illustrates how species richness changes with increasing alteration in median August flow (MA19). The grey lines represent 90^{th} quantile regressions, which provide an indication of how the best possible biological status changes across varying degrees of flow alteration. The red lines represent ordinary least squares regression, indicating of how the mean of the response variable changes with increasing flow alteration.

Water withdrawals associated with hydraulic fracturing activities would result in diminished as opposed to augmented August flows. Figure ES-3 shows that as August flow is reduced (right-to-left from zero along the x-axis), species richness in the MSR declines substantially. More specifically, a 10% reduction in August flow equates to a loss of roughly 3 species at the 90th quantile.

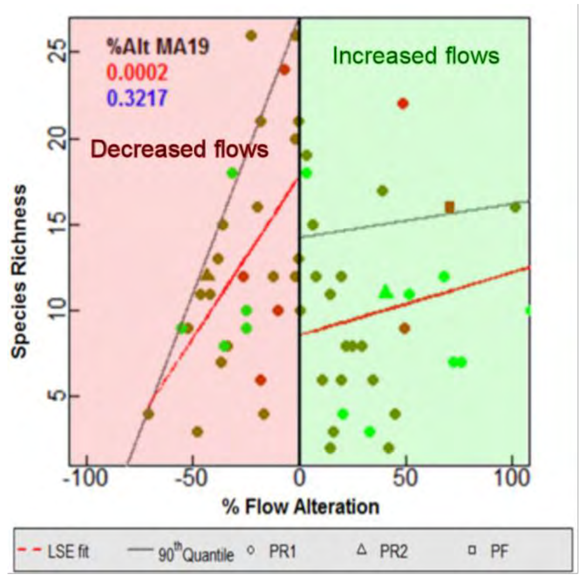

Figure ES-3. Example flow-ecology relationship in the Marcellus Shale Region.

We formatted all F-E points with different colors (Figure ES-4) and shapes (legend in Figure ES-2) to represent the different hydro-types (classes) and physiographic provinces that the streams fell within. We then evaluated the F-E distributions for clustering of points to assess whether F-E relationships were stream-class-specific or differed significantly between physiographic provinces.

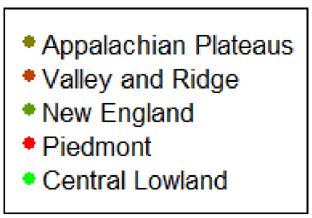

Figure ES-4. Legend for all F-E curves indicating physiographic province of sampling point.

Examination of the F-E points across all ecological endpoints revealed little clustering on the basis of stream class or physiographic province, suggesting that F-E relationships derived in this study can be applied across the entire MSR. However, we should caution that many stream classes and provinces were not well represented in our F-E dataset. The lack of adequate sample size precludes definitive evaluations of class-specific F-E relationships and should be considered as a topic for future research.

We anticipated that reductions in low-flow indices would prove the most significant and consistent predictor of adverse changes in the fish community. However, we found that many average- and high-flow HIs were also significant at the 90th quantile. This suggests that flow standards focused exclusively on low-flows may not provide sufficient protection for riverine ecosystems in the MSR. We also observed that some ecological metrics, such as life history traits and trophic structure displayed inconsistent or insignificant linkages with changes in flow regime. This may indicate that these metrics are not the most responsive to flow alteration or, perhaps, that small sample sizes or error in the F-E computations lead to spurious results.

Pumping and Risk Analysis

We simulated the hydrologic effects of hydraulic fracturing withdrawals by constructing a suite of low/high local (one pumping site per reach) and cumulative (multiple pumping sites within a drainage basin) extraction scenarios. The pumping analysis was implemented by subtracting the pumping rates associated with each scenario from daily flow data from gaged reference basins in the MSR. We then determined hydrologic sensitivity indices (HSI) of the various HIs to pumping as the median percent difference between the natural baseline HI value and the values under the high local and cumulative pumping scenarios across all reference gages. This analysis helped to: i) quantify the relationship between drainage area and a stream's sensitivity to water withdrawal, ii) establish which hydrologic indices and stream classes are most sensitive to withdrawals, and iii) inform a Marcellus-wide risk assessment.

Drainage area – hydrologic sensitivity relationship

Figure ES-5 illustrates how the sensitivity of median August flow is affected by consumptive water extraction under the various pumping scenarios. The percent change in August flow increases with increasing abstraction rates (i.e. low local to high cumulative pumping). A drainage area threshold is evident at approximately 1,000 km^2 – after which pumping has minimal effects (<5% change in MA19). This threshold held across all magnitude-related HIs, providing support for limiting the application of ELOHA in the context of hydraulic fracturing on the basis of drainage area. It also confirms the intuitive notion that water extraction will have a disproportionate effect on smaller streams.

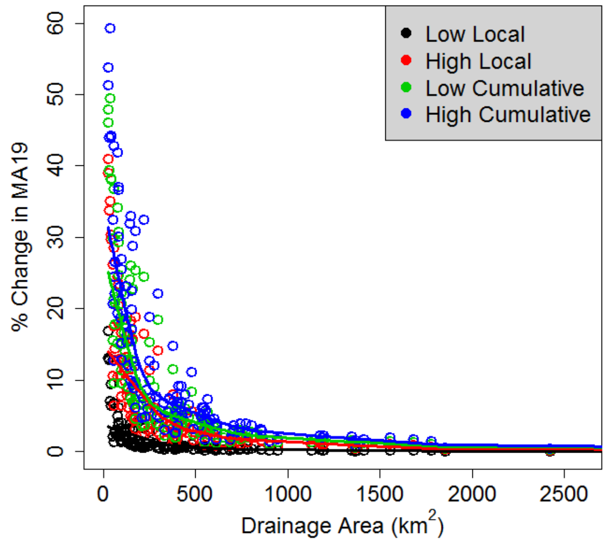

Figure ES-5. Percent change in mean August flow as a function of basin drainage area for the low/high local and cumulative pumping scenarios. Curvilinear lines represent locally weighted regression curves fit to the data to guide the eye.

Sensitivity of hydrologic indices to surface water pumping

By comparing the relative sensitivities of the various HIs to water withdrawals we found that low flow HIs were the most sensitive to pumping – especially 1-, 3- and 7-day low flow durations and seasonal low flows occurring during the summer and fall. In contrast, high flow duration HIs, as well as high flows during the winter and spring months were least sensitive. These results have implications for future studies and stream gaging campaigns designed to assess and monitor the long-term impacts of consumptive water withdrawals.

Sensitivity of stream classes to surface water pumping

The sensitivity of the various stream classes was consistent with their respective hydrologic characteristics. For instance, perennial runoff stream types possessed high baseflows and low flow variability and were shown to be least sensitive to withdrawals. On the other hand, perennial flashy streams were characterized by lower, sometimes intermittent flows, lower baseflows and higher flashiness indices and were showed to be the most sensitive to water withdrawals. This implies that streams possessing flashier flow patterns should be considered for more conservative flow protection standards.

Risk Analysis

We predicted the sensitivity of a select group of HIs to withdrawals across all streams in the MSR using random forest models. In general, we found that smaller catchments with lower temperatures, flatter slopes, shallower seasonal water tables, fewer dams, higher percentages of poorly drained soils and higher percentages of pasture and crop landuses were associated with higher HSI values. HSI predictions were mapped to polyline shapefiles of all NHD streamlines in the MSR, revealing meaningful spatiotemporal patterns (Figure ES-6). For instance, the majority of streams that are sensitive to water extraction (red lines in Figure ES-6) during the summer season are lower order systems located primarily in two areas within the MSR: (i) a southwestern zone (Upper Ohio River, Muskingum and Southern Lake Erie basins) and (ii) in a northern band (Upper Susquehanna River Basin and tributaries of the Upper Hudson River Basin). Streams at lower risk (blue) are generally located in the central MSR (West Branch of the Susquehanna River and Allegheny River Basins) and along the eastern border (Potomac River and eastern portions of the Upper Ohio River Basins). Additionally, there is a marked seasonal difference in HSIs, wherein high-flow periods such as spring (Figure ES-6A) and winter are far less sensitive to withdrawals than summer (Figure ES-6B) or fall.

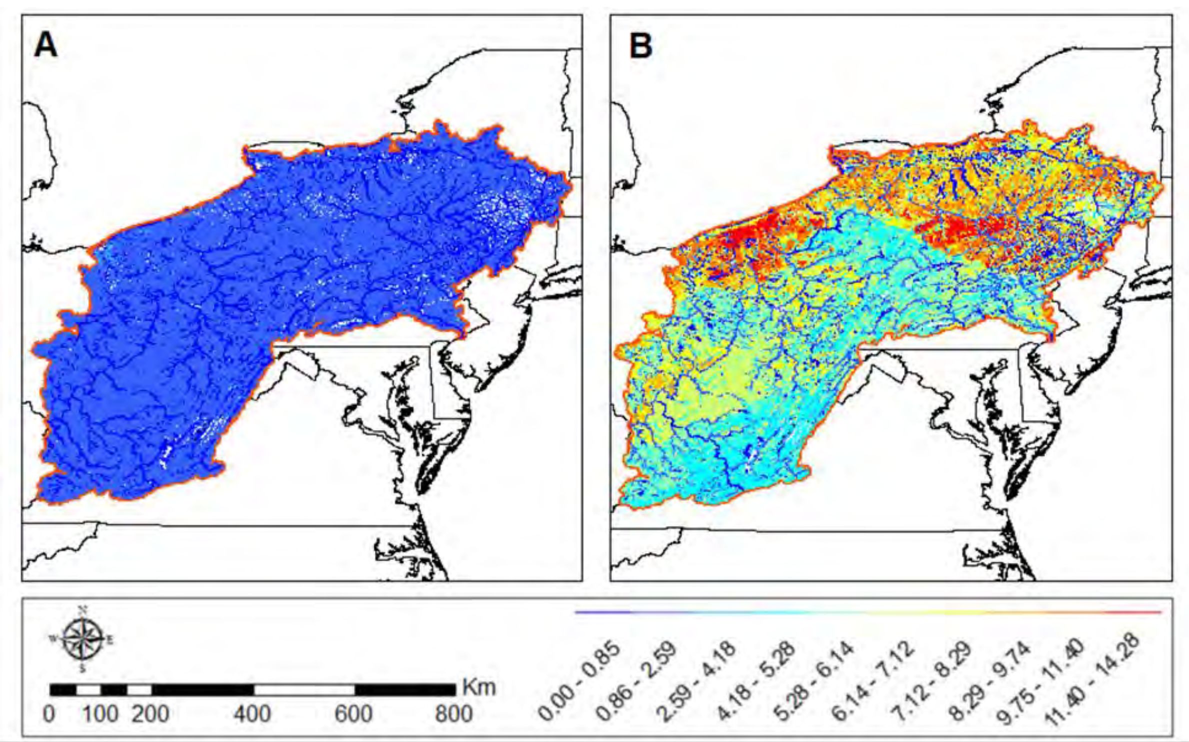

Figure ES-6. Hydrologic sensitivity indices for low flows in spring (A) and summer (B) due to the low local pumping scenario.

The results of this analysis provide insights into how hydrologic sensitivity to water withdrawals varies spatially and should help identify particularly sensitive streams for targeted management. Moreover, the mapped HSIs can be overlaid with species distribution maps and the locations of existing and projected natural gas development to further prioritize streams threatened by hydraulic fracturing activities that coincide with species of concern.

Management Implications

A thorough understanding of the effects of surface water withdrawals for hydraulic fracturing activities on riverine ecosystems is a crucial step in making prudent management decisions. Applying the ELOHA framework to stream systems within the Marcellus Shale Region revealed a number of significant findings that may be useful for defining environmental flow standards in the context of water withdrawals, as well as for providing guidance to water resource managers and future studies. Salient management and flow policy implications are as follows:

- Our pumping analysis suggests environmental flow standards and monitoring campaigns concerning water withdrawals for hydraulic fracturing should focus on low-flow hydrologic indices during the summer and fall, as these are most sensitive to alteration. However, higher low-flow requirements will only protect fish communities if depletion of low-flows is the principle hydrologic stressor acting on aquatic biota. Our flow-ecology relationships indicate that biotic integrity of fish communities is adversely affected by changes in average- and high-flow metrics, indicating that low-flow provisions alone may be inadequate to protect riverine ecosystems.
- Managers and policy makers should consider a conservative, perhaps season-specific, approach for particularly sensitive streams (e.g. low-order streams < 1000 km^2, with low annual precipitation or flashy hydrologic characteristics) or for streams lacking adequate hydrologic or biological data.
- The hydrologic risk maps presented here offer a useful initial screening tool, allowing water resource planners to identify streams or areas for targeted management, more conservative flow standards or areas that require more detailed analysis and monitoring (e.g. on-site evaluation). Species-specific risks to flow alteration from hydraulic fracturing withdrawals can also be assessed by combining the hydrologic risk maps with species distribution models and projections of future water use in the MSR.
- Streams with high observed flow alteration or those deemed a high risk to flow regime change due to water withdrawals may be good candidates for remediation, while streams with minimal alteration represent sites that may benefit from protection to prevent negative impacts to stream biota. The F-E relationships presented here could be used as decision support tools to evaluate whether an observed or predicted level of water extraction will result in unacceptable biological effects and devise an appropriate response that protects or restores the stream's hydrology and ecology.
- Overall, our study provides support for the seasonally variable flow recommendations outlined by DePhilip and Moberg (2010, 2013) as opposed to fixed minimum annual flow standards.

Ultimately, the results from this study were intended to provide a scientific foundation for the development or refinement of defensible flow standards in the MSR. Actual ecological limits to flow alteration were not quantified in this report. This is the last step in the ELOHA process, requiring that acceptable ecological conditions and environmental flow standards be defined through an adaptive process of stakeholder input, scientific analysis, monitoring and feedback. This may be best "pursued at a watershed jurisdictional scale, in accordance with state- and local-level priorities, needs, and regulatory mandates" (USACE, 2013).

Introduction

Context

Horizontal hydraulic fracturing has led to rapid expansion of natural gas drilling in the Marcellus Shale deposit in West Virginia and Pennsylvania, and is projected to expand into Ohio and New York. Horizontal drilling describes the process by which a vertical bore hole is drilled to the depth of a shale deposit, redirected laterally to a horizontal orientation and driven for thousands of meters into the shale bed. Subsequently, high volumes of water, sand and other chemical additives (e.g. surfactants, biocides, corrosion inhibitors) are pumped under high pressure into perforations in the well casing. This creates a network of small interconnected fractures which propagate large distances into the surrounding shale (Figure 1). Together, these processes greatly increase the pay zone and extraction rate of a well by enhancing both borehole-shale contact and the density of interconnected pore space. Without these unconventional technologies, extraction of natural gas from the Marcellus Shale would not be a commercially viable enterprise.

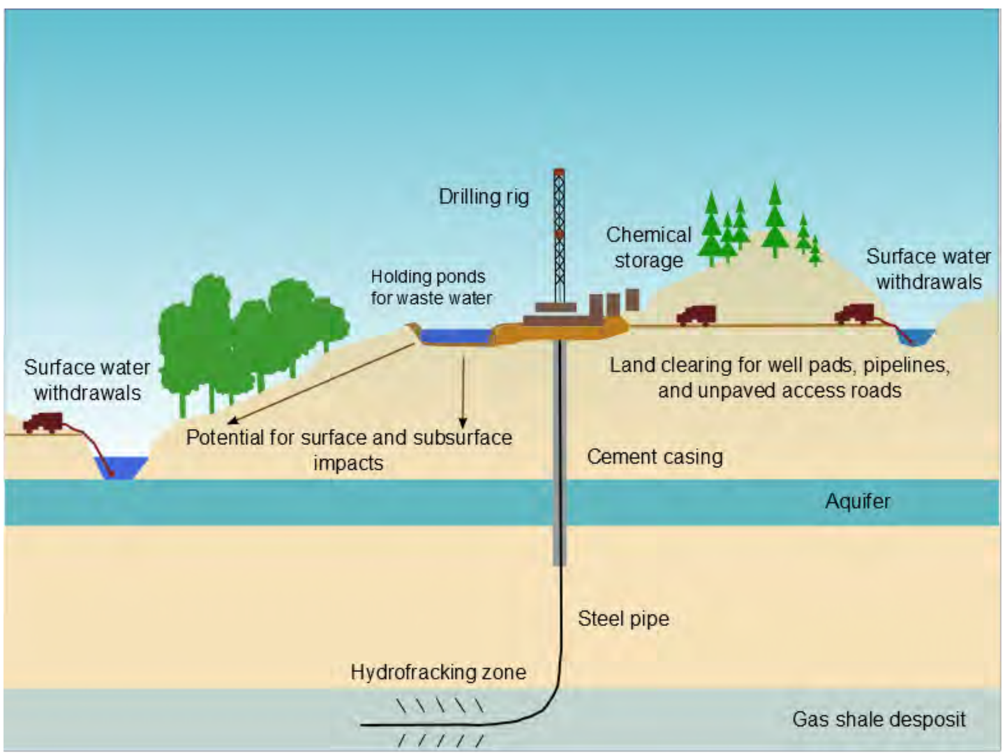

Figure 1. Conceptual diagram depicting the hydraulic fracturing process. Drilling rigs bore into shale formations and wells are lined with steel pipe. Well casings are then sealed with cement to limit groundwater contamination. Upon reaching the shale deposit, the bore-hole is directed horizontally, after which holes are blasted through the steel well casings. Water, chemicals and other additives are pumped into the well under high pressure, fracturing the shade-bed, thereby increasing the pay zone and extraction rate of wells by enhancing borehole-shale contact and the density of interconnected pore space. Associated activities include land clearing for well pads and supporting infrastructure (e.g. roads and pipelines), as well as the extraction and transportation of water from nearby freshwater bodies. Flowback water is stored in shallow holding

ponds until it can be transferred to treatment facilities or re-used in another hydraulic fracturing operation. These activities may impact nearby streams through surface and subsurface pathways. Adapted from Weltman-Fahs and Taylor (2013).

A typical hydraulic fracturing well requires between two and seven million gallons of water to fully develop and a single well pad often hosts as many as 20 wells (Rahm and Riha, 2012). Moreover, to maintain high yields, wells are frequently re-fractured several times over their life spans, which may last several decades (Entrekin et al., 2011). These large per-pad water requirements, in conjunction with burgeoning gas development across the region, suggests hydraulic fracturing activities may put substantial strain on already well-exploited regional water supplies (Rahm and Riha, 2012). Although the pool of streams and rivers that may be viable water sources is vast, only a small subsample of these will serve as practical withdrawal points for hydraulic fracturing activities due to logistical constraints primarily related to transport costs. Naturally, the natural gas industry will attempt to minimize these costs by preferentially locating withdrawal points proximal to their drilling pads. Additionally, well pad densities often vary greatly over the landscape due to the uneven spatial distribution of gas deposits, viable access points and available drilling leases. Altogether, this has the effect of concentrating water demands on an even smaller subset of surface water bodies, potentially compounding local water demands.

Research Need

The wide-spread, yet locally concentrated water consumption related to natural gas drilling, combined with existing concerns over climate change and future non-drilling water resource needs, have sparked concern among hydrologists and aquatic biologists regarding implications for freshwater ecosystems in the region. For example, the cumulative effects of rapid water extraction for multiple purposes may lead to altered flow regimes, changes in the diversity and composition of riverine ecosystems, reductions in the quality of critical habitat for freshwater biota and deleterious changes in important ecological processes (e.g. nutrient cycling). In addition to flow-related changes, hydraulic fracturing may impact aquatic biota by: i) reducing hydrologic connectivity leading to habitat fragmentation, ii) increasing sediment inputs which can adversely affect fish and macroinvertebrate communities and iii) degrading groundwater and surface water quality via contaminated flowback water (Entrekin et al., 2011; Rahm and Riha, 2012; Weltman-Fahs and Taylor, 2013). Importantly, these myriad effects do not operate in isolation, but may overlap in time and space, resulting in additive or synergistic effects.

Despite the serious ecological implications of continued gas development, little guidance currently exists for water resource managers who are tasked with balancing human and ecosystem water demands. Currently, regulations governing permitting procedures for water withdrawals related to hydraulic fracturing are developed and enforced by a confusing matrix of state and interstate agencies and river basin commissions. The lack of federal oversight, due to an exemption to the Safe Drinking Water Act written into the Energy Policy Act of 2005, has resulted in an incoherent, piecemeal regulatory framework where withdrawal limits (e.g. pass-by flows) are based on hydrologic rules-of-thumb rather than credible, ecologically meaningful science. As Rahm and Rhia (2012) note, the lack of a coherent strategy for managing hydraulic fracking activities has contributed to a largely reactive rather than proactive regulatory approach – with environmental issues being detected and addressed after they have occurred.

Task

This project aims to provide guidance for water resource managers and environmental planners seeking to establish ecologically-defined limits on water withdrawals for hydraulic fracturing in the Marcellus Shale. To achieve this goal, we applied the Ecological Limits of Hydrologic Alteration (ELOHA) approach proposed by Poff et al. (2010). ELOHA provides a conceptual framework for quantifying environmental flows that "flexibly allows scientists, water resource managers and other stakeholders to analyze and synthesize available scientific information into coherent, ecologically based and socially acceptable goals and standards for management of environmental flows"(Poff et al., 2010). ELOHA is especially well suited for assessing environmental flow needs across larger regions such as the MSR, where time and resource constraints render river-by-river assessments unfeasible.

Thus, ELOHA seeks to establish credible, evidence-based flow limits that help sustain healthy aquatic ecosystems while recognizing the need to balance human water needs. Briefly, ELOHA is a systematic, five-step process that facilitates the analyzing and synthesizing of scientific information about streamflow and the flow-related needs of riverine ecosystems. The primary steps consist of (Figure 2):

Scientific Process

I. Building a hydrologic foundation by compiling all existing observed flow records for the region of interest (ROI). Using a combination of observed flow data and predictions from either statistical- or process-based hydrologic models, establish baseline (natural) and altered hydrologic characteristics for streams and rivers throughout the ROI.
II. Classify all basins into similar hydrologic types on the basis of their natural hydrologic characteristics. This serves to reduce natural biological variability, strengthening flow-ecology relationships.
III. Calculate flow alteration by computing the difference in natural vs. altered flows.
IV. Relate flow alteration to observed changes in meaningful ecological metrics for biological communities of concern. These so called "flow-ecology" relationships are generally developed for each individual hydro-type identified in step II.

Social Process

V. Use flow-ecology relationships to manage environmental flows through an informed social process.

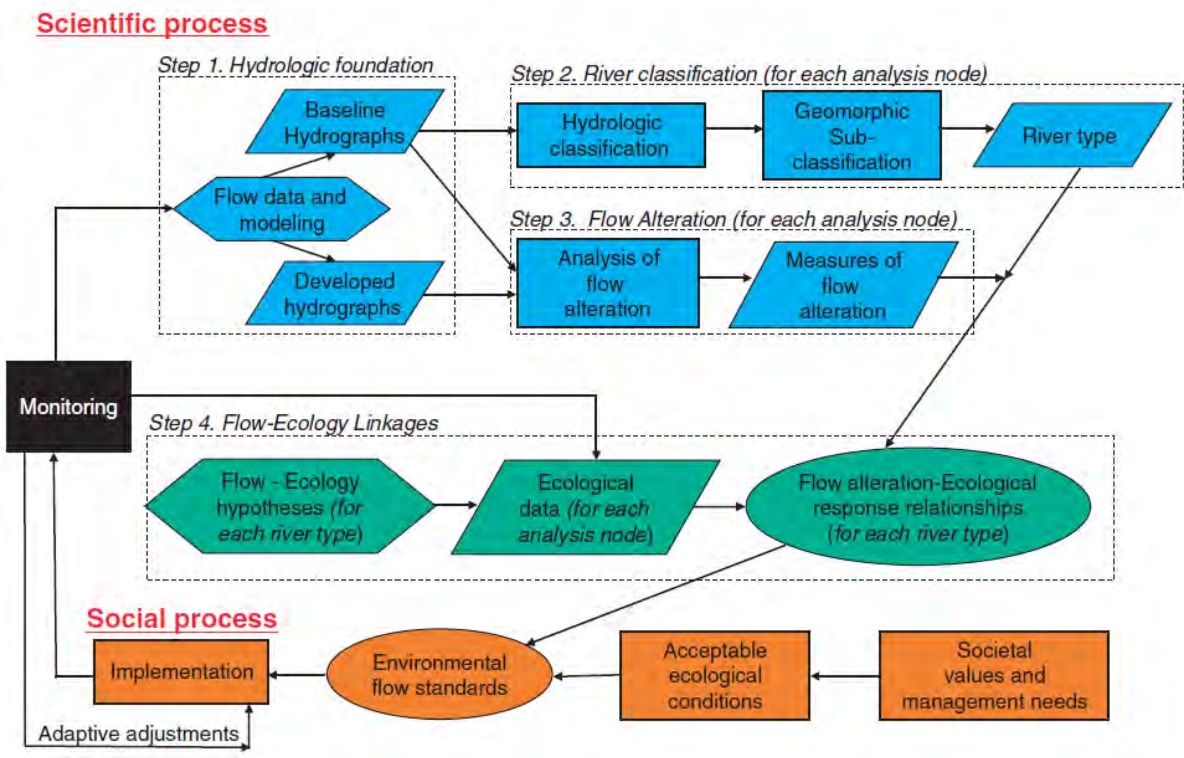

Figure 2. Conceptual diagram of the major steps in the ELOHA process (Poff et al., 2010).

This report outlines progress made towards the application of the *scientific* phase of the ELOHA framework. We comment on the implications for environmental flow standards, but stop short of specifying quantitative flow limits as this requires input and feedback from the *social* component of the ELOHA process, which is outside the scope of this project.

Objectives

To accomplish the task of applying ELOHA to the question of water resource impacts from shale gas development in the MSR, we divided the project into two phases.

Phase I

The goals of the first phase were to review existing tools and approaches, as well as gather and format available and relevant data within the Marcellus Shale Region. The specific objectives of Phase I were:

- Gather all relevant hydrologic data in the MSR and create georeferenced database of flow timeseries data.
- Identify the most appropriate biological community to use for ELOHA application in the MSR.
- Gather all relevant ecological data and format to a common, georeferenced database format.

- Determine whether a hydrologic foundation for the MSR is more appropriately established via process-based hydrologic models – or by applying statistical models.

Phase II

The second phase applies the most appropriate tools to: i) build a hydrologic foundation, ii) estimate flow alteration and iii) develop flow-ecology relationships which will provide guidance for establishing quantitative limits to water withdrawals associated with gas development.

Our specific objectives for this phase included:

- Selecting the most appropriate hydrologic indices to characterize alterations in flow regimes.
- Calculating the degree and direction of flow alteration at all basins of interest.
- Classifying all streams into similar hydrologic types according to their baseline hydrologic characteristics.
- Calculating a suite of meaningful ecological metrics to characterize fish communities in the MSR.
- Constructing flow-ecology relationships to quantify how flow alteration may be affecting aquatic ecosystems.
- Creating a series of consumptive water use scenarios to simulate the potential hydrologic effects of shale gas related water withdrawals across a range of development intensities. This analysis will also elucidate factors influencing a stream's relative sensitivity to withdrawals.
- Results of the pumping analysis will be used to compute hydrologic sensitivity indices, which will inform Marcellus-wide risk assessment.
- Interpreting our results and translate to a clear set of management implications to support water resource decision making.

Report Outline

This report is divided into two primary sections corresponding to phases I and II. Phase I outlines the application of a process-based hydrologic model to a subset of stream basins in the MSR in order to test the feasibility of this approach. We also summarize the results of our consultation with other experts in the field of hydrologic modeling, as well as present the results of a literature review. Finally, we provide a set of recommendations, as well as describe our acquisition and formatting of pertinent ecohydrologic data.

The second section of the report details major steps involved in our application of ELOHA in the MSR. This section is formatted much like a standard scientific journal article with sub-sections including: methods, study area description, results and discussion, and conclusions. Appendices providing supplemental material are also included at the end of the report. References for both report sections are compiled together and appear, along with Appendices, at the end of this report.

Phase I

One of the first steps in applying ELOHA involves establishing a hydrologic foundation of daily streamflow data for every stream reach under baseline (natural) and current (altered) conditions for a single time period. We investigated the suitability of two means of constructing a hydrologic foundation: (i) process-based hydrologic models, which simulate the dominant hydrological processes within a watershed through physically-based equations and (ii) an empirical approach which uses statistical relationships between hydrologic metrics calculated at gaged basins and physical basin characteristics (e.g. slope, elevation) to predict natural hydrologic indices across all basins of interest. In this way, a hydrologic foundation is built such that all streams in the Marcellus will have estimated flow indices under reference and non-reference conditions.

Evaluation of Process-Based Hydrologic Model Approach

Following the recommendations of the Marcellus Shale Milestone Report submitted to the AppLCC, we investigated the feasibility of applying the ABCD model to compute daily hydrographs for streams across the Marcellus Shale Region. We used a combination of approaches to corroborate our findings. For instance, we: i) contacted experts in the fields hydrologic modeling and environmental flows analysis, ii) performed preliminary hydrologic modeling with the ABCD and Soil and Water Assessment Tool (SWAT) and iii) performed a literature review.

Consultation with Modeling Experts

We contacted Austin Polebitski (U. of Wisconsin-Platteville) to obtain a newer version of the ABCD model that had been ported into the R programming environment. Dr. Polebitski informed us that their attempts to regionalize the ABCD model in the North Atlantic Landscape Conservation Cooperative (NALCC) were largely unsuccessful. Additional consultation with Scott Steinschneider (UMass - Amherst) and Ben Letcher (USGS Silvio Conte Fish Center and UMass) corroborated Dr. Polebitski's advice. Both Drs. Steinschneider and Letcher advised that trying to regionalize process-based hydrologic models across large geographic areas (i.e. the Marcellus) may prove prohibitively challenging at a daily time-step. They further advised us to consider a statistical approach to predicting hydrologic indices – emphasizing that data input requirements for a hydrologic model (e.g. existing sources of hydrologic alteration, such as dams and water withdrawals and climate data) would be extensive and that model uncertainty coupled with the challenges of model regionalization would likely result in inaccurate predictions and wide confidence bounds.

SWAT Modeling

In the interest of assuring due diligence regarding the questionable suitability of process-based hydrologic modeling across the Marcellus Region, we modeled a subset of reference and non-reference basins in the Marcellus Region with the SWAT model. All model runs were calibrated using the DEoptim function in the R programming environment. We then quantified model performance by comparing observed and simulated flow and calculated Nash-Sutcliffe efficiencies (NSEs are similar to an R^2 - the closer to one, the better the model fit). We noted a very wide range of model performance with NSEs ranging from 0.14 to 0.92 over both the reference and non-reference catchments. In general, the reference gages were better simulated by SWAT. This is not surprising as it was very difficult to obtain all the data relevant to properly simulate altered flows. For example, many of the non-reference basins contained dams and it proved difficult to obtain the operations data for each dam. This is important as without the dam release data, SWAT could not properly account for dam effects, which can have a profound effect on flow regimes. This highlights one of the key limitations of the process-based hydrologic modeling approach - namely, that it would be exceedingly challenging to acquire quantitative data concerning all the relevant sources of anthropogenic alteration across the Marcellus Region (e.g. thousands of dams and ground- and surface-water withdrawals). Despite the lack of data concerning man-made alterations to natural flow regimes, SWAT did perform reasonably well in a number of catchments. This was due to the fact that SWAT was able to compensate for the lack of appropriate anthropogenic parameterization data through the calibration process. In other words, in many cases SWAT was able to arrive at the right answer, but for the wrong reasons.

Another key finding from the exploratory SWAT modeling was that the greatest uncertainty in model estimates were observed in the smaller catchments (refer to the large range of NSE values in basins smaller than 1000 km^2 in Figure 3). This is important because it indicates that the hydrologic model will perform most poorly in the basins that would be most sensitive to hydrologic alteration resulting from surface water pumping.

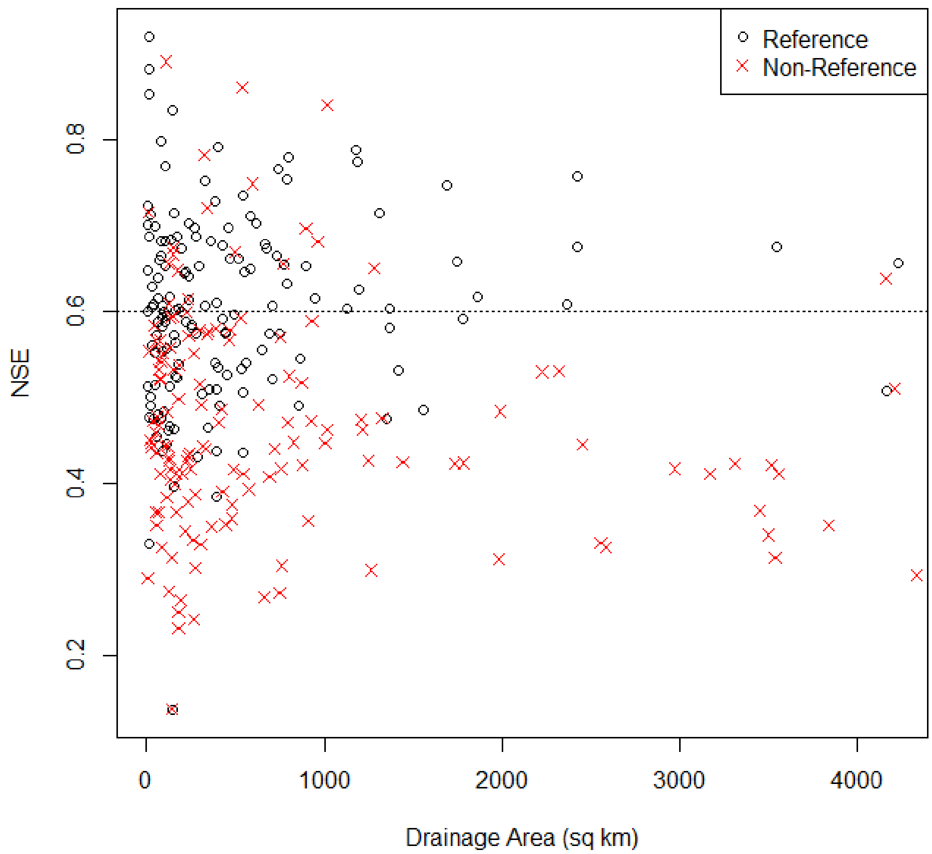

Figure 3. Calibrated SWAT model NSEs as a function of drainage area for a subset of reference and non-reference USGS gages in the Marcellus Shale Region. The dashed horizontal line indicates the lower limit of acceptable NSE values.

Unlike the ABCD model, SWAT is capable of simulating a complex system of water-related infrastructure such as inter-basin transfers, irrigation, surface and ground water withdrawal and dams. Despite, the mixed performance of the SWAT model, we determined that it would be suitable for performing case-study analyses, such as assessing the local and cumulative effects of water withdrawal for gas development using a gradient of pumping scenarios in select study catchments.

Literature Review

A review of relevant literature was performed in order to determine the current state of knowledge and guidance regarding methods of predicting baseline and altered hydrologic conditions across wide geographic areas. Below, we provide a brief summary of several particularly relevant articles. Additional supporting literature is provided in the annotated bibliography in Appendix G.

Booker and Woods (2014)

Booker and Woods (2014) compared a variety of available methods for estimating several hydrological indices and flow duration curves at ungauged catchments across New Zealand.

Specifically, they compared the following: i) a process-based spatially distributed hydrologic model (TopNet), ii) empirical regression models based on hydrologic theory, iii) empirically-based random forest models and iv) random forest corrected TopNet estimates. The purpose of this comparison was to assess which method best predicted several hydrological indices given current climatic and land cover conditions. Importantly, they found that empirically-based random forest models outperformed all other methods, including the process-based spatially distributed hydrologic model. This suggests that applying a statistical approach in the Marcellus Shale Region would prove more effective.

Buchanan et al. (2013)

The only peer-reviewed example of a process-based hydrologic model being applied across a large basin for the purposes of determining environmental flows following an ELOHA-style framework was that of Buchanan et al. (2013). In this study, the authors applied the Chesapeake Bay Program Hydrologic Simulation Program–FORTRAN (HSPF) model and the Virginia Department of Environmental Quality Online Object Oriented Meta-Model (WOOOMM) routing module to the Potomac River Basin.

They found that the combined HSPF-WOOOMM model resulted in a wide range of Nash-Sutcliffe efficiencies (0.33 to 0.82), indicating a very wide range of model performance (i.e. very poor to good). The model performed most poorly in smaller urbanized basins or on or near karst geology. We should emphasize that this study likely represents a best case scenario in terms of data availability and parameterization. For instance, the study was conducted in the Chesapeake Bay Watershed, which has been the subject of intensive study for many decades. Through the combined efforts of numerous non-profit organizations and state and federal agencies, an extensive database of information necessary for a well parameterized model has been amassed. Furthermore, the HSPF-WOOOMM model was expressively designed and calibrated for the Chesapeake Bay Watershed. Even under these relatively ideal conditions, the process-based model yielded results of questionable utility in many of the modeled catchments. This is in accordance with the result of our SWAT modeling – further suggesting that hydrologic modeling may be problematic at the scale of the Marcellus Region.

Carlisle et al. (2010)

Carlisle et al. (2010) used national- and regional-scale predictive models and models based on landscape classifications, including major river basins, ecoregions and hydrologic landscape regions (HLR) to estimate thirteen indices of the magnitude, frequency, duration, timing and rate of change of streamflow. They then compared model performance, measured with bias and precision metrics, to determine which method most accurately simulated the observed flow regime. They found that statistically-based random forest models provided substantially better estimates of hydrologic indices than the landscape stratification models. This provides further evidence that random forest models may provide the most accurate estimates of relevant hydrologic indices for the Marcellus Shale Region.

Kendy et al. (2012)

Kendy et al. (2012) offer a thorough overview of the recommended practices for conducting environmental flow analyses. They explicitly evaluated the strengths and weaknesses of the various methods of generating streamflow data necessary for constructing a hydrologic foundation, including: hydrologic simulation using process-based models, drainage area ratio methods, and regression modeling (Table 1) They point out that process-based models are not well-suited to large-scale regional applications (the intended geographic scope of ELOHA), due to their complexity. Moreover, when process models are applied at regional scales, computational challenges often limit spatial discretization to the scale of larger watersheds, which is too coarse for meaningful ELOHA results. However, a great strength of process models is the ability to simulate the hydrologic effects of future land use and climate change. Overall, Kendy et al. (2012) found that regression models were simpler to apply, faster, cheaper and more appropriate for regional scale ELOHA applications. Their primary limitations are: i) prediction inaccuracies at the extremes of the observed data (low and high flows in very small and large basins) and ii) predictions are generally limited to only certain set of flow indices as opposed to daily time series data from which hundreds of different indices could be calculated.

Table 1. Evaluation of the various means of establishing a hydrologic foundation for ELOHA applications, including examples, strengths and weaknesses of each approach. Methods are listed in order of complexity, expense and level of expertise required. Example applications are detailed in Kendy et al. (2012).

Approach	Examples	Strengths	Limitations
Drainage-area ratio method	StateMod (Colorado), CT Basin flood flows	Low cost, easy to generate.	Limited accuracy if index gages do not represent the natural range of flow regimes. Dependent on gage availability.
Regression – generated monthly statistic	Median August flow (Michigan), mean September flow (Ohio)	Low cost, easy to generate, widely accepted.	Current-condition only. Not a time series. Represents only one environmental flow component.
Regression with water accounting and flow routing	U.S. Geological Survey (USGS) AFINCH (No ELOHA case study)	High spatial resolution; linked to NHD+.	Monthly time series only. Has not been tested outside Great Lakes basin.
Duration-curve regression plus water accounting	USGS Sustainable Yield Estimator (SYE) (Massachusetts, Pennsylvania)	Relatively low cost, easy to generate. Daily time step.	Difficult to simulate flows at hydrograph and basin-size extremes. Has not been applied outside eastern US.
Duration-curve regression plus dam operations model	USGS SYE plus US Army Corps of Engineers HEC-DSS (Connecticut River basin)	Same as above, with ability to model dam releases.	Relatively time-consuming (several years) to develop; requires two federal agencies.

Hydrologic process model plus water use accounting and channel routing	WOOOMM (Watershed Online Object Oriented Meta-Model) (Potomac River basin)	Can model land-use and climate change.	Resolution typically too coarse or area too small for regional application without modification.

Recommendations for Future Work

After conducting a preliminary feasibility analysis of the ABCD and SWAT models, performing a literature review, as well as consulting with other researchers who have attempted similar regional model applications, we determined that process-based hydrological models would not be the most appropriate means of establishing a hydrologic foundation for the Marcellus. We did, however, determine that SWAT may be useful for evaluating local and cumulative effects of surface water withdrawal, similar to those associated with hydro-fracking. This analysis could be conducted in a case-study format and would help to elucidate which hydrologic indices and stream classes within the Marcellus Region are most sensitive to hydrologic alteration. Another possibility is to construct a simpler water use accounting algorithm in a programming environment such as R. Water use scenarios covering a gradient of development intensities could be used to subtract water from daily flow data from gaged streams in the MSR. The main advantage of this approach is that the algorithm would be computationally efficient allowing for simulation of water withdrawal effects at *all* gaged basins.

As an alternative to hydrologic modeling for establishing a hydrologic foundation, we propose to develop and apply statistical models which first calculate hydrologic indices (HIs) at all reference and non-reference gages, then use statistical relationships between the HIs and characteristics of their respective basins to extrapolate relevant flow indices to all ungaged basins. We suggest using the powerful and recently developed technique of random forest regression to predict HIs based on topographic, geologic, land use and climate attributes of its respective basin. Random forests are a very robust statistical modeling technique that have been shown to explain complex variations in hydrologic patterns (e.g. timing, magnitude, frequency and duration of flows; Booker and Snelder, 2012; Snelder and Booker; Booker and Woods, 2014). The approach uses machine-learning through the synthesis of many regression trees into an ensemble prediction, resulting in more accurate/reliable regressions by drawing bootstrapped samples from the original "training" data and fitting a regression tree to each sample (Booker and Snelder, 2012).

Acquisition of Relevant Hydrologic and Ecologic Data

Ecologic Data

Ample research has demonstrated that most forms of aquatic biota are responsive to changes in the natural flow regime. Thus a variety of freshwater taxonomic groups may provide meaningful endpoints for establishing ecological limits to flow alteration. We chose to focus on fish as opposed to macroinvertebrates or aquatic/riparian vegetation because: i) they have been shown to respond more predictably to anthropogenic flow alteration than macroinvertebrates or vegetation (McManamay et al., 2013; Poff and Zimmerman, 2010) , ii) data availability and spatial coverage was better in the MSR, iii) fish are a highly valued resource and therefore represent a more charismatic biological endpoint which facilitates meaningful communication with the public and iv) fish encompass a wide range of life history characteristics (e.g. life-spans and mobility), which helps reveal long-term disturbances to aquatic ecosystems over broad spatial scales (Karr, 1981; Barbour et al., 1998).

Drawing on resources from multiple state and federal agencies, we created a database describing fish presence and abundance patterns in six states (i.e. MD, VA, WV, OH, PA and NY). We first obtained Multistate Aquatic Resource Information System (MARIS) fish data for NY, PA, WV, VA and MD. We then integrated fish survey data from the United States Geological Survey (USGS) (NAQWA) program, the United States Environmental Protection Agency (USEPA) Mid-Atlantic EMAP program, and the Ohio Environmental Protection Agency (OEPA). The database contained measures including the total number and species of fish observed at a particular sampling site, the location and sampling methodology, target standard, and, in some cases, the degree of sampling effort (recorded as either time or distance). The database was then clipped to the HUC-8 Marcellus boundary. The final database contained a total of 186,518 records at 11,104 unique fish sampling sites with over 220 unique species (Appendix A).

Hydrologic Data

We first defined a project boundary using all NHD HUC-8 subwatersheds that intersected the geologic boundary of the Marcellus Shale (n=661). This resulted in a hydrologically- as opposed to geologically-defined analysis extent that we deemed more appropriate for a water resource oriented study. Next, we downloaded daily discharge data associated with all USGS gages that were: i) located within the project boundary (n=571), and ii) contained greater than 15 years of largely continuous data to ensure flow regimes could be adequately characterized. A more detailed description of this step is presented in Phase II.

Phase II

Methods

Study Area

This study focused on the Marcellus Shale deposit, a geographically expansive US shale gas reservoir estimated to contain over 13 trillion m^3 of recoverable natural gas (Rozell and Reaven, 2012). Covering an area of over 170,000 km^2, the Marcellus Shale Region (MSR) underlies much of New York, Pennsylvania and West Virginia, as well as portions of Ohio, Virginia and Maryland (Figure 4). The recent development of horizontal drilling and hydraulic fracturing has facilitated rapid expansion of natural gas drilling in the Marcellus Shale. Intense activity in West Virginia and Pennsylvania is expected to continue, with expansion into Ohio (for increased extraction of the Utica Shale formation) and New York possible. On December 17th, 2014 New York officially banned all high-volume hydraulic fracturing in shale formations, but conventional drilling practices are still allowed. This decision may reduce the future impacts of natural gas development on water resources in New York's Southern Tier, but such a ban is subject to mercurial political forces, and we therefore included the MSR in New York in this analysis.

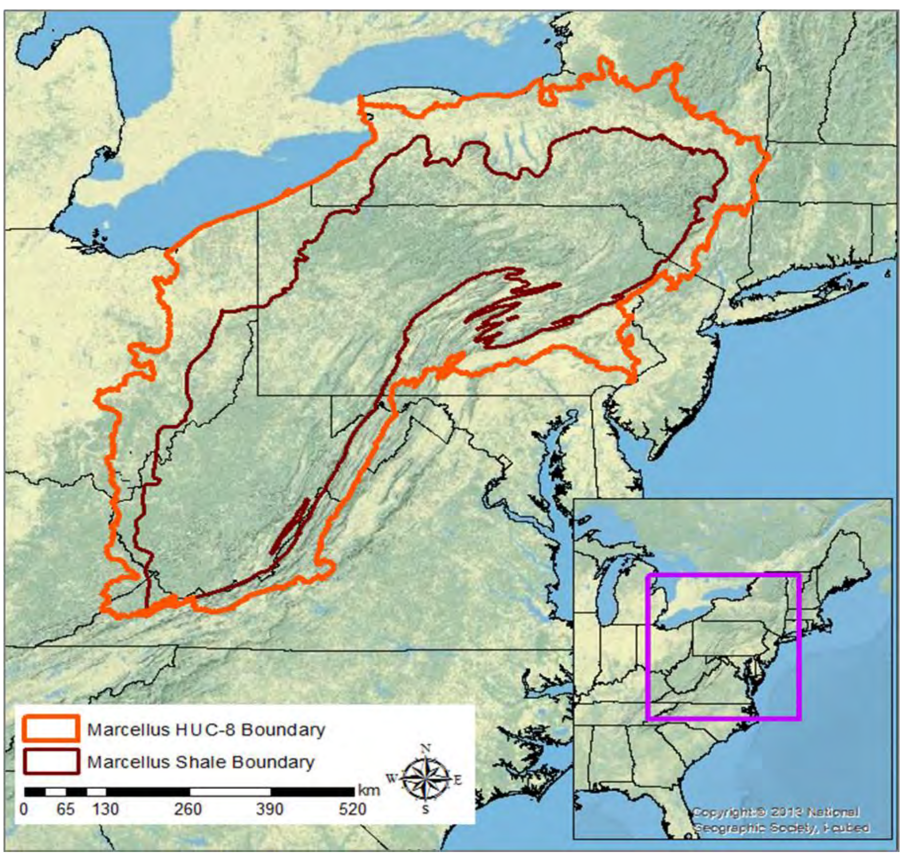

Figure 4. Overview of study area, Marcellus Shale Play (red polygon) and analysis extent (orange polygon).

The MSR encompasses substantial geologic, topographic and climatic variation, containing six physiographic provinces as well as 79 Level IV Ecoregions (USEPA, 2014). The resulting range of hydrologic conditions drives the formation of diverse stream habitats and aquatic communities. Stream classifications in the region have identified as many as 3 distinct types on the basis of natural hydrologic characteristics (McManamay et al., 2014b) and this abiotic variation supports considerable biodiversity. For example, field surveys summarized in this study suggest the MSR is home to more than 220 different fish species, including some threatened and locally endangered species (Appendix A).

The MSR's ~135,000 streams (USGS, 2014) drain three major, economically and ecologically important watersheds: the Susquehanna, Ohio and Delaware River basins. High flow frequency and magnitude in the MSR is typically greatest in the spring and lowest in the summer and early fall. Due to relatively low evapotranspiration and intermittent snow melt, winter months are often characterized by moderately-high flow events. Dry periods, characterized by low and infrequent precipitation, can occur at any time during the year, but primarily result in diminished stream discharge during late summer. Although summer low-flows are important for maintaining adequate habitat volume, temperature and dissolved oxygen, natural flow regimes across all seasons are important for ensuring healthy aquatic communities in the MSR.

Even though the region generally has abundant precipitation relative to other areas of shale gas development (e.g. North Dakota), the additional water demand placed on lotic systems may prove deleterious to the MSR's aquatic biota. Interestingly, the Energy Policy Act of 2005 rendered hydraulic fracturing activities exempt from the Safe Drinking Water Act, which limited federal regulation and oversight. This results in a complicated and largely inconsistent regulatory framework in the MSR with existing laws and regulations promulgated, monitored and enforced by a combination of state agencies and river basin commissions (e.g. Susquehanna River Basin Commission). The lack of a scientifically credible and coherent strategy for managing hydraulic fracking activities has contributed to the largely reactive rather than proactive regulatory approach – with environmental issues being detected and addressed after they have occurred (Rahm and Riha, 2012).

ELOHA Application

Build a hydrologic foundation

Our first step was to define an appropriate spatial extent within which to conduct our analyses. Given the goal of assessing water resource impacts, we delineated the analysis boundary as the intersection of the geographic Marcellus Shale Region with NHD HUC-8 catchment boundaries (USGS, 2014). The final extent included 661 NHD HUC-8 units either within or touching the Marcellus boundary (Figure 4). We then clipped NHD Version 1 streamlines and catchments, as well as all USGS streamflow gages and fish sampling sites to this boundary.

Streamflow gages were divided into reference and non-reference categories as per McManamay et al. (2014), designating a low or high degree of upstream anthropogenic disturbance (e.g., dams, diversions and native vegetation conversion). A total of 198 reference and 373 non-reference USGS gaging stations were available for analysis after removing stations with large continuous blocks of missing data and periods of record < 15 yrs (a period of record that adequately captures flow variability for hydrologic classification (Kennard et al., 2010).

After downloading mean daily flow records for each gage from the USGS National Water Information System, we used the Hydrologic Index Tool (HIT) program (Henriksen et al., 2006) to calculate 171 hydrologic indices for each discharge record. Hydrologic or "flow" indices measure various components of a hydrograph, often distinguished as the magnitude, timing, frequency, duration, and rate of change in the discharge time series (Gao et al., 2009; Olden and Poff, 2003). For each of the 571 focal gages, we merged the 171 HIT indices with 46 different basin attributes describing topographic, geologic, land use, climate and anthropogenic development variables from the GAGES II database (Table 2). In order to permit analyses at ungaged catchments (i.e., those where GAGES II data were not available), we also compiled comparable natural and anthropogenic watershed characteristics from the following sources: NHD Version 1, National Fish Habitat Partnership and data compiled by The Nature Conservancy (TNC) (Table 2). All basin attribute data represented accumulated attributes from the entire upstream contributing area, as opposed to only local characteristics (i.e. within the drainage area below the preceding stream junction).

Table 2. Basin attributes compiled from the Gages II Database, NHD Version 1, National Fish Habitat Partnership and data compiled from The Nature Conservancy (TNC). Note, all basin attribute data for all gaged basins was provided solely by the GAGES II database.

Basin Attribute Code	Definition	Data Source for Ungaged Basins
COMID	HUC ID of Basin	NHD
GRID_CODE	ArcHydro Grid Code of Basin	NHD
PROD_UNIT	HUC Production Unit	NHD
DRAIN_SQKM	Drainage Area (sq km)	TNC
LONG_CENT	Longitude of Basin Centroid	NHD
LAT_CENT	Latitude of Basin Centroid	NHD
AWCAVE	Average Available Water Content	TNC
BDAVE	Average Bulk Density	TNC
BFI_AVE	Average Baseflow Index	TNC
CLAYAVE	Average Clay Content	TNC
CONTACT	Average Contact Time	TNC
ELEV_MEAN_M_BASIN	Mean Elevation of Basin	TNC
HGA	Percentage of Hydrologic Group A	TNC
HGB	Percentage of Hydrologic Group B	TNC
HGC	Percentage of Hydrologic Group C	TNC

HGD	Percentage of Hydrologic Group D	TNC
STOR_NID_2009	Dam Storage - National Inventory of Dams 2009	TNC
NO10AVE	Average percent by weight of soil < 3" in size and passing a No. 10 sieve	TNC
NO4AVE	Average percent by weight of soil < 3" in size and passing a No. 4 sieve	TNC
NO200AVE	Average percent by weight of soil < 3" in size and passing a No. 200 sieve	TNC
OMAVE	Average Percent Organic Matter	TNC
PERMAVE	Average Permeability	TNC
ROCKDEPAVE	Average Depth to Rock	TNC
SANDAVE	Average Percent Sand	TNC
SILTAVE	Average Percent Silt	TNC
SLOPE_PCT	Average Percent Slope	TNC
STREAMS_KM_SQ_KM	Stream Density	TNC
WATERNLCD06	Percent Land Cover as Water	TNC
DEVOPENNLCD06	Percent Land Cover as Developed Open Land	TNC
DEVLOWNLCD06	Percent Land Cover as Developed - Low Density	TNC
DEVMEDNLCD06	Percent Land Cover as Developed - Med Density	TNC
DEVHINLCD06	Percent Land Cover as Developed - High Density	TNC
BARRENNLCD06	Percent Land Cover as Barren	TNC
DECIDNLCD06	Percent Land Cover as Deciduous Forest	TNC
EVERGRNLCD06	Percent Land Cover as Evergreen Forest	TNC
MIXEDFORNLCD06	Percent Land Cover as Mixed Forest	TNC
SHRUBNLCD06	Percent Land Cover as Shrub	TNC
GRASSNLCD06	Percent Land Cover as Grass	TNC
PASTURENLCD06	Percent Land Cover as Pasture	TNC
CROPSNLCD06	Percent Land Cover as Crop	TNC
WOODYWETNLCD06	Percent Land Cover as Woody Wetland	TNC
EMERGWETNLCD06	Percent Land Cover as Emergent Wetland	TNC
WTDEPAVE	Average depth to seasonally high water table	TNC
PDEN_2000_BLOCK	Population Density	NFHAP
ROADS_KM_SQ_KM	Road Density	NFHAP
NDAMS_2009	Number of Dams	NFHAP
HYDRO_DISTURB_INDX	Hydrologic Disturbance Index	NFHAP
PPTAVG_BASIN	Average Precipitation in Basin	NHD
T_AVG_BASIN	Average Temperature in Basin	NHD

Calculating the Degree of Flow Alteration

In order to compute the degree of flow alteration, we constructed 171 random forest (RF) models that related each of the flow indices to the 46 basin attributes at all reference gages. From this training set, we could then predict the natural or "expected" flow indices at all 373 non-reference gages. The difference between the observed, potentially altered, index value and the predicted natural index value yielded a measure of flow alteration as:

$$FA_{i,g} = 100 * \left(\frac{QObs\ metric_{i,g} - Expected\ metric_{i,g}}{Expected\ metric_{i,g}}\right)$$

where i indexes the 171 indices, and g indexes the 373 gages. Negative flow alteration therefore indicated a reduced value in a particular index (e.g. a smaller annual peak than predicted for that gage), whereas positive alteration represented an increase.

Briefly, a random forest is a machine learning method developed from partitioning trees (i.e., classification and regression trees, CART). A single "tree" consists of the hierarchical sequences of best-supported divisions in a response variable according to values of potentially many predictor variables. For instance, the distribution of mean annual flow for a sample of gages might be most strongly related to the mean annual basin precipitation among a pool of variables; some amount of precipitation then produces the best split among the gages according to a criterion such as minimizing the variance across subsidiary groups. Each of these "child" nodes is then further divided up to a pre-defined stopping rule (hence, "recursive partitioning"). A "forest" consists of a large number of such decision trees constructed using randomly chosen subsets of predictor variables and bootstrapped subsets of available observations. This provides an ensemble prediction that can overcome possible weaknesses in any single tree, and that can be validated via the "out-of-bag" (OOB) error rate calculated from mis-prediction of the observations withheld from the forest training set. The OOB validation procedure also helps to estimate the relative predictive power of each of the explanatory variables (i.e., "variable importance"). We chose to implement random forest models using "cForest" in the "Party" package (CITE) in the R programming environment. Conditional inference RF is suited to non-linear relationships, and correlated predictor variables of mixed types (i.e. continuous vs. categorical), and it provides a robust means to avoid the over-fitting to which standard RF is prone (by using permutation tests to define traditional statistical significance as a stopping rule). All HIT indices and explanatory variables were log(x+1) transformed.

We chose to focus our evaluation on watersheds smaller than 2,500 km^2. The training dataset of reference gages did not include an adequate sample of locations with larger drainage areas (Figure 5). Exceeding the support of the observed data undermines valid inference, and a preliminary evaluation of RF model performance across all drainage areas indicated 2,500 km^2 as a conservative threshold to ensure informative partitioning. This drainage area threshold corresponds to headwaters, creeks, small rivers and medium tributary river categories of the Northeast Aquatic Habitat Classification System (NEAHCS; Table 3). We assumed these categories were also the most vulnerable to adverse ecohydrologic impacts of surface water pumping associated with gas development. Indeed, our pumping analysis (outlined below) revealed that realistic surface water pumping is not likely to have a substantial impact on catchments larger than roughly 1,000 km^2 (see Figures 31 and 32).

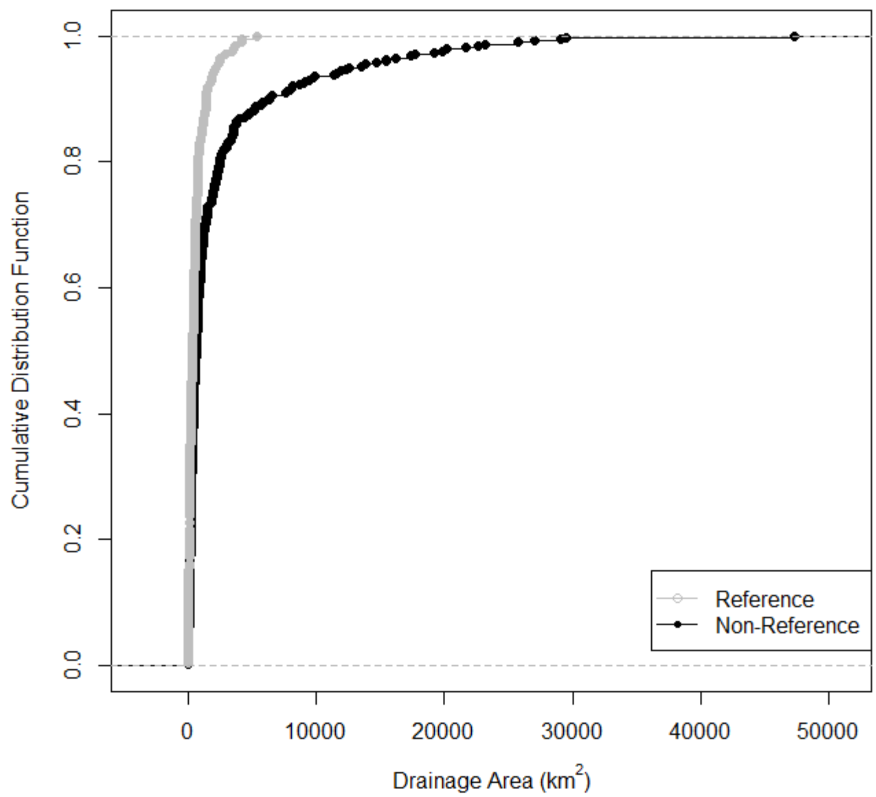

Figure 5. Cumulative drainage areas for reference and non-reference gages.

Table 3. Watershed size categories of the Northeast Aquatic Habitat Classification System (NEAHCS). Adapted from Potomac River Basin Report.

Size Category	Headwaters	Creek	Small River	Medium Tributary River	Medium Mainstem River	Large River	Great River
Drainage Area (km^2)	< 10	10 - 99	100 - 517	518 - 2,589	2,590 - 9,999	10,000 - 24,999	≥ 25,000

Selecting Flow Indices

Selection of appropriate, ecologically relevant hydrologic indices is a key step in the ELHOA process. Hydrologic indices are sometimes determined *a priori* based on established or hypothesized flow-ecology relationships, but we felt it necessary to take an exploratory, data-driven approach given the largely unstudied nature of possible gas development impacts. Desirable hydrologic indices have ecological significance (i.e., a measurable influence on organismal success), and are accurately predicted across the landscape of interest in ELOHA. Thus, we began by determining which of the 171 hydrologic indices were accurately predicted, retaining only those that achieved an OOB R^2 value ≥ 0.8 in RF models. This threshold ensured acceptable accuracy while allowing us to capture major facets of the hydrologic regime (i.e. magnitude, timing and duration and rate of change). We further reduced the set of accurately predicted indices, by focusing on those that were commonly used and easily understood, as well as those that minimized redundancy and those that were sensitive to surface water withdrawals. An additional consideration was whether the index was consistent with hypothesized flow-ecology relationships from two comprehensive environmental flow reports for two major basins within the MSR (i.e. the upper Susquehanna and Ohio River basins; DePhilip and Moberg, 2010 and 2013).

Calculating Ecological Metrics

We computed a total of 19 different ecological metrics from the MARIS database created in Phase I. Metrics covered a range of fish community and assemblage information, including: species richness, composition (total and relative abundance), tolerance to disturbance, trophic structure, and life history strategies (Table 4). We also provided flow-ecology hypotheses specific to each ecological metric based on documented and theorized ecological responses to altered flow regimes in the context of water withdrawals (i.e. reduced flows). The MARIS database contained roughly 64,000 records designated with a target standard of "Target", meaning that field crews were targeting specific fish species and ignored all others caught. All such target records were removed for non-species-specific analyses. Importantly, different MARIS sites and NHD reaches may have had different levels of sampling effort (i.e. sampled a different number of times). To control for differing degrees of sampling effort, all metrics were normalized to the number of times a particular MARIS site was sampled and the number of MARIS sites in a given NHD reach (i.e. some reaches were associated with multiple MARIS sites).

Table 4. Ecological metrics and descriptions.

Ecologic Metric	Units & Description
Species Richness Metrics	
Species Richness	# species within reach
Fish Abundance Metrics	
Total Abundance	Catch per unit effort time (CPUE-T; hrs)
Tolerance Metrics	
Percent Tolerant Species (USEPA)	Proportional abundance of non-sensitive species
Percent Intolerant Species (USEPA)	Proportional abundance of sensitive species
Indicator Guild/Species Metrics	
Abundance of Cold Headwater Species	Proportional abundance (CPUE-T; hrs)
Abundance of Nest Builders	Proportional abundance (CPUE-T; hrs)
Abundance of Riffle Obligates	Proportional abundance (CPUE-T; hrs)

Abundance of Riffle Associates	Proportional abundance (CPUE-T; hrs)
Abundance of Brook Trout	Proportional abundance (CPUE-T; hrs)
Abundance of Smallmouth Bass	Proportional abundance (CPUE-T; hrs)
Abundance of Northern Hog Sucker	Proportional abundance (CPUE-T; hrs)
Abundance of Central Stoneroller	Proportional abundance (CPUE-T; hrs)
Trophic Composition Metrics	
Percent Generalist (USEPA)	% of species with generalist feeding traits
Percent Invertivore (USEPA)	% of species with invertivorous feeding traits
Percent Herbivore (USEPA)	% of species with herbivorous feeding traits
Percent Piscivore (USEPA)	% of species with piscivorous feeding traits
Life History Metrics	
Percent Periodic Species	% of species with periodic strategy
Percent Opportunistic Species	% of species with opportunistic strategy
Percent Equilibrium Species	% of species with equilibrium strategy

Species Richness and Total Abundance

Species richness provides a measure of the overall fish diversity of a particular stream reach, but ignores the abundance of each individual species. Total abundance, on the other hand, estimates the total number of individuals observed in a study reach without regard to species composition. Generally speaking, higher species richness and abundance scores are indicative of healthier stream ecosystems, although they can be strongly influenced by stream size. Species richness and total abundance were calculated as the total number of species and total number of individuals observed per NHD reach, respectively. We used raw measures of abundance, which were standardized to the number of fish caught per unit effort (hours).

Flow-Ecology Hypotheses

- Both species richness and total abundance will decrease with increasing negative flow alteration (Bunn and Arthington, 2002; DePhilip and Moberg, 2010, 2013; Poff and Zimmerman, 2010)

Tolerance Metrics

Tolerant species are typically comprised of fish well-adapted to a range of perturbed habitat conditions and are particularly tolerant of impaired water quality. Intolerant species, on the other hand, typically exhibit strong, predictable negative responses (e.g. reduced abundance) to anthropogenic perturbation and are often the first species to disappear following disturbance (Barbour et al., 1998; DePhilip and Moberg, 2013). Reductions in flow due to water abstraction for gas development will likely favor tolerant vs. intolerant species due to a combination of temperature and water quality stress due to less water available for dilution and the maintenance of thermal refugia. Tolerance values were assigned following Barbour et al. (1998).

Flow-Ecology Hypotheses

- Tolerant species will increase with increasing flow alteration.
- Intolerant species will decrease with increasing flow alteration (Knight et al., 2014; Rader and Belish, 1999; Apse et al., 2008; Walters, 2010, Konrad et al., 2008)

Trophic Composition Metrics

Trophic composition metrics provide a measure of the "quality of the energy base and trophic dynamics of the fish assemblage" (Barbour et al., 1998). Published literature has documented shifts in trophic diversity and composition related to flow regime alteration (Horwitz, 1978; Gleason, 2007). Typically, more stable regimes are characterized by trophic and habitat specialists, whereas more hydrological variable or extreme sites possess more trophic generalist species (Hoeinghaus et al., 2007; Poff and Allan, 1995).

Fish were assigned to one of four trophic guilds based on their predominant feeding ecology as outlined by Barbour et al. (1998). Trophic guilds included: piscivores, invertivores, generalists and herbivores (Table 5). Note: the invertivore guild includes insectivores, while trophic generalists included omnivores. All trophic composition metrics were estimated as a proportional abundance.

Table 5. Description of trophic guilds.

Trophic Guild	Description
Piscivores	opportunistic top predators consuming predominantly other fish
Invertivores	mostly consume immature and adult insects, as well as array of other invertebrates, including, mollusks and crustaceans
Generalists	quite adaptable and can consume both plant and animal matter
Herbivores	primarily of plant matter such as periphyton

According to Barbour et al. (1998), the relative abundance of top-predators helps distinguish between moderate and high integrity aquatic ecosystems. Based on F-E hypotheses proposed by DePhilip and Moberg (2010, 2013; Appendix B), we posited that decreases in flow magnitudes during summer low-flow periods would result in a loss of refugia and a shift in trophic composition towards top-predator dominated systems.

Invertivores, principally insectivores, are likely the dominant trophic guild in most lentic systems in the MSR (Barbour et al., 1998) and are generally associated with high quality stream systems. Conversely, generalists are more often found in streams with degraded physical and chemical habitat (Barbour et al., 1998; Gleason, 2007). Empirical data directly relating herbivorous trophic strategies to stream habitat quality are rare. Generally however, altered habitat conditions can precipitate a shift in the energy base of lentic ecosystems towards autotrophic processes – especially in headwaters (Karr et al., 1981). This may favor recruitment of herbivorous species and a decline in invertivory and piscivory. However, some forms of flow alteration may negatively affect aquatic vegetation (e.g. augmented high flows increase scour or sedimentation), which may lead to deterioration of herbivore populations.

Overall, we anticipated that reductions in habitat quality and availability due to changes in flow regime would result in declines in invertivore prey abundance (e.g. loss of macroinvertebrates), as well as potentially negative effects to aquatic vegetation. Thus, increasing flow alteration will results in a shift from trophic specialists (i.e. invertivores and herbivores) to trophic generalist fish species as generalists are better adapted to exploit a less diverse, often more variable food base (Gleason, 2007).

Flow-Ecology Hypotheses

- Piscivores:
  - Relative abundance will increase with reduced summer flows due to loss of refugia and increased predator-prey interactions (DePhilip and Moberg, 2010, 2013).
- Invertivores & Herbivores:
  - Relative abundance of these trophic specialists will decline with increasing flow alteration due to loss of food base (Knight et al., 2013).
- Generalists:
  - Relative abundance will increase with increasing flow alteration as they are better adapted to habitat disturbance (Freeman and Marcinek, 2006; Poff and Allan, 1995).

Life History Strategies

A number of studies have concluded that freshwater fish can be grouped, according to their life history traits, into three main strategies "that represent the endpoints of a triangular continuum arising from essential trade-offs among the basic demographic parameters of survival, fecundity, and onset and duration of reproduction" (Figure 5; Olden and Kennard, 2010).

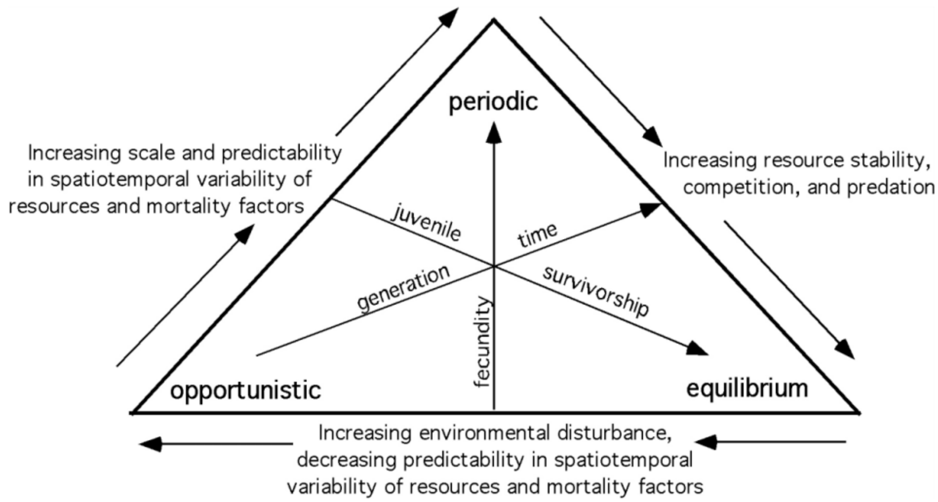

Figure 6. Triangular life history model depicting environmental gradients selecting for endpoint strategies defined by optimization of demographic parameters generation time, age-specific survivorship, or age-specific fecundity (from Winemiller, 1995).

Table 6 provides additional description of the characteristic biological and environmental habitat attributes associated with the equilibrium, opportunistic and periodic strategies outlined in Figure 6.

Table 6. Description of the different traits of the three life history strategies.

Strategy	Body Size	Maturation Age	Fecundity per Spawning Event	Juvenile Survivorship	Habitat Preference
Opportunistic	Small	Early	Low	Low	frequent and intense disturbances
Equilibrium	Small-Medium	Moderate	Low	High	low variation in habitat quality and strong biotic interactions
Periodic	Large	Late	High	Low	inhabit seasonal, periodically suitable environments with large-space spatial (patchiness) and temporal heterogeneity

It is important to note that the three life history strategies represent a continuum rather than a discreet set of mutually exclusive attributes. Consequently, the life histories of most fish species fall into intermediate positions within the life history space outlined in Figure 6 (Olden & Kennard, 2010, Mims and Olden, 2012). Recognizing this fact, we calculated the proportional abundance of each strategy using weights rather than binary assignments to each strategy. This involved calculating the Euclidian distance in the trivariate life history space for each species, normalizing to a 0-1 scale, and then estimating the strategy weight as the inverse of these values.

Subsequently, strategy weights were multiplied by the relative abundance of each species at each site and then, summed and averaged over the entire reach.

Ecological theory predicts that periodic life history strategists will be favored in streams with seasonal, yet predictable flow regimes that create periodically suitable environments (Winemiller, 2005). Such streams usually possess large spatial and temporal heterogeneity (i.e. patchiness and seasonality, respectively). Opportunistic fish species, on the other hand, are typically well-suited to stream environments characterized by frequent and intense disturbances. Equilibrium strategists are typically found in streams with stable, predictable flow regimes with low variation in habitat quality and strong biotic interactions. Accordingly, we formulated the following flow-ecology hypotheses:

Flow-Ecology Hypotheses (McManamay and Frimpong, 2015; Winemiller, 2005)

- Equilibrium
  - Decreased abundance with decreasing flow stability (i.e. increased flow variability, lower low flows and baseflows and lower predictability)
- Periodic
  - Reduced abundance with decreasing flow seasonality (i.e. higher flow variability, lower predictability)
- Opportunistic
  - Reduced abundance with increasing flow regime stability (i.e. lower variability, higher predictability, higher seasonality)

Functional Guilds & Sentinel Species

A suite of relative abundance metrics were calculated for four different functional guilds who shared common physical habitat requirements and life history strategies (Table 7). Functional guilds were derived from two comprehensive studies conducted in the Susquehanna and Ohio River Basins (DePhilip and Moberg, 2010, 2013). Fish were grouped according to whether they shared similar body size, fecundity, home range, habitat associations, feeding habits and flow-velocity tolerances. These common traits translate into similar flow requirements. For instance, all fish in the nest-building guild are predicted to be sensitive to spring high flows that may scour nests in channel margins (DePhilip and Moberg, 2010, 2013). Thus, the functional guilds are complementary to the life history ecological metrics, but perhaps more tailored to the MSR. We calculated all functional guild metrics as total vs. relative abundance due to sample size issues.

Table 7. Description of the key traits and species in each of four functional guilds. Sentinel species for each guild are indicated by bold font. Adapted from DePhilip and Moberg (2010, 2013).

Group	Key Traits	Species
Cold Headwaters	Thermal tolerance limits distribution to cool and cold habitats; sensitive to decreases in dissolved oxygen or increases in turbidity; across group, spawning occurs in all seasons	**brook trout**, burbot, mottled sculpin**,** brown trout, *Cottus* spp.
Riffle Obligates	Small bodied, flow-velocity specialists who spend most of their life in moderate-fast velocity riffle/run habitat. Small home range renders them sensitive to localized disturbance	**central stoneroller,** marginged madtom, longnose dace, blacknose dace, greenside darter, rainbow darter, tessellated darter, johnny darter, banded dater, fantail darter, bluntnose minnow, cutlip minnow
Riffle Associates	Resident species with moderate-sized home range that migrate to spawn and need access to, and connectivity between, riffle habitats. Upstream migration is cued by both temperature and rising water levels. Have a preference for clear streams.	**northern hog sucker**, white sucker, *Catostomus* spp., shorthead redhorse, golden redhorse, silver redhorse, walleye, smallmouth buffalo
Nest Builders	Similar timing of flow needs (during nest building, spawning, and egg and larval development), but a diverse group in terms of nesting strategy (includes true nests, mound construction and ledge spawners). Particularly sensitive to flow conditions during spring and summer nest building. Most require maintenance coarse substrate for nest building.	**smallmouth bass**, fallfish, creek chub, river chub, *Nocomis* spp., redbreast sunfish, spotted bass

Additionally, we also computed the abundance of four sentinel species - one per functional guild (species in bold font; Table 7) - in order to facilitate species-specific flow requirements and risk mapping. Species were chosen according to how well they were represented across the MSR.

Flow-Ecology Hypotheses (DePhilip and Moberg, 2010 and 2013)

- Cold Headwaters
  - Decreased abundance with decreasing low flows, especially during summer and fall when thermal stress may be highest.
- Riffle Obligates
  - Increased flashiness or high flow magnitudes may reduce recruitment due to egg scour or loss of access to spawning habitats.
  - Conversly, substantial reductions in spring high-flows may reduce riffle obligates as they are necessary for the maintenance of sandy substrates and serve as important spawning cues.
  - Reduced median flow in summer and winter months may limit the quality and availability of riffle habitats leading to reductions in abundance.
- Riffle Associates
  - Similar to riffle obligates.
  - Loss of low- and median-flows reduces access to and connectivity between riffle habitats, resulting in reduced abundance.
  - Changes in high flow characteristcs reduce abundance as spawning is cued by rising water levels and correct temperature ranges.
- Nest Builders
  - Especially sensitive to alterations in spring and summer flows – the primary nest building seasons.
  - Reduced abundance with diminished high flows due to lack of flushing and maintenance of coarse substrates for nest building.
  - Similarly, overall reductions in discharge (i.e. median annual flows) may increase siltation and/or dewater nests.

Stream Classification

It is customary to classify streams and rivers in ELOHA applications into different hydrologic types on the basis of natural baseline hydrologic characteristics because it is hoped that doing so will potentially improve the statistical significance of F-E relationships by reducing natural variability in the biological communities of interest (Poff et al., 2010, USACE, 2013). Accordingly, streams within the Marcellus HUC-8 boundary were classified into stream classes or hydrologic types via a hierarchical clustering analysis developed for all reference gages in the Appalachian LCC region. All HIT metrics for all reference gages were log (x+1) transformed, scaled and centered from 0-1 and used in a correlation-based principle component analysis (PCA). This effectively reduced the redundancy and dimensionality of the dataset while ensuring that the most important HIT metrics were retained for clustering. We found that over 90% of the variation was explained by the first 13 components.

We used a Bayesian mixed-model approach to cluster gauges on the basis of component scores. The approach applies multiple models and mixtures (clusters) to the data and uses Bayes Information Criteria (BIC) to determine the most likely model and number of clusters (Fraley et al., 2014). Hierarchical modeling is used to specify a prior number of clusters and then Gaussian mixture modeling is used to estimate parameters of each model. Ten different models with varying covariance structure and numbers of clusters are compared and the model-cluster combination with the highest BIC values is considered the optimal solution.

Using the clustering results, we then we then constructed a random forest model to predict stream class at all ungaged basins using the reference gages as training data. The prediction dataset for this model consisted of all ungaged NHD catchments in the HUC-8 Marcellus Region merged with basin attributes listed in Table 2. OOB error rates were used to assess model accuracy and variable importance plots of the various explanatory variables were generated.

Flow-Ecology Relationships

In order to establish flow-ecology relationships, it was necessary to devise an appropriate means of associating each MARIS site with a non-reference USGS gaging station. MARIS sites and USGS gaging stations do not necessarily overlap in space, so this is an important step. We chose to pair MARIS sites with USGS gages based on whether they shared the same National Hydrography Dataset (NHD) Version I "reachcode" (HSC, 2013). To accomplish this it was necessary to assign the NHD reachcodes to each MARIS and USGS site via a spatial join in a GIS program using a 350m join radius. This worked well, but did result in some USGS gages being paired with more than one MARIS site. Thus, it was necessary to calculate the average ecologic metric across all dates and MARIS sites per reach. Once each USGS gage had a single metric associated with it, we then began the process of using quantile regression to assess whether observed flow alteration was significantly related to ecological metrics.

Using the degree of flow alteration calculated as the difference between predicted reference and observed non-reference flows we then developed flow-ecology relationships by relating the percent alteration of particular flow indices to changes in the 19 ecologic metrics. Flow-ecology curves (F-E) were developed using quantile regression in R (quantreg package by R. Koenker, 27 February 2011, available at www.r-project.org). Quantile regression (QR) is particularly well-suited to F-E relationships as aquatic biota respond to a variety of non-flow related natural and anthropogenic factors (e.g. water temperature, contaminants, invasive species, lost connectivity, etc.). These multiple drivers produce scatter in the ecological response at any particular level of flow alteration. For example, two streams with similar flow alteration may exhibit very different ecological responses due to differing degrees of water quality impairment. The resulting scatter weakens the statistical significance of the relationship between the explanatory variable(s) and the mean of the response variable (the 50th quantile or standard OLS regression). By examining other parts of the response variable distribution, QR is able to characterize significant trends which may otherwise be missed. In the context of ELOHA, QR is often used to explore the upper bounds of F-E relationships as this provides a description of how the best possible biological status changes across varying degrees of flow alteration. For this study, we chose to use 90th quantiles to describe the upper-bound response. Significance of the 90% quantile regression was assessed via p-values derived from a xy-pair bootstrap procedure. Please refer to Cade and Noon (2003) for a more detailed exploration of QR in the context of ecological studies.

One of the key principles of the ELOHA framework is the idea that streams that share similar flow regimes (stream classes) will possess comparable ecological characteristics and thus, will respond to flow alteration in similar ways. As previously mentioned, it is hoped that the process of stream classification will reduce natural ecological variability within classes and improve F-E relationships. To assess whether F-E relationships were stream-class-specific we visually examined the wedge-shaped F-E distributions looking for clustering of points according to stream class (Knight et al., 2013). We also investigated whether clustering according to physiographic province was evident.

It is important to note many of our ecological metrics may be affected by the stream size. For instance, both species richness and catch per unit effort are generally positively correlated with watershed area (Barbour et al., 1998). This may confound flow-ecology relationships derived from raw measures of ecological response to flow alteration (particularly when stream size is not controlled in calculation of flow alteration). To control for this, we introduced drainage area as an explanatory variable in a multivariate QR for all F-E relationships.

Pumping Scenarios

We performed a preliminary scenario analysis of consumptive surface water pumping associated with hydraulic fracturing to: i) determine the relationship between sensitivity to withdrawals and stream size, ii) establish which hydrologic indices and stream classes are most sensitive to withdrawals, and iii) construct hydrologic sensitivity indices (HSIs) that could inform a Marcellus-wide risk assessment.

As previously mentioned, it is difficult to synthesize the myriad rules, regulations and management guidelines regarding limits to surface water abstraction for gas development because of the diversity of regulatory policies within the MSR. In an effort to create a realistic set of water withdrawal scenarios, we constructed a set of low and high pumping scenarios for all reference gages within the HUC-8 Marcellus Basin under a range of different development intensities. Pumping analyses were divided into two groups: (i) local pumping that reflects consumptive surface water withdrawal only within the gaged reach and (ii) cumulative pumping that reflects the collective impact of multiple pumping sites throughout the entire upstream basin. Though clearly simplified, these scenarios are likely to cover the range of pumping rates and development intensities that may occur in the field. Note: we are assuming that all water requirements for developing shale gas wells will derive from nearby streams. We are also assuming that no flowback water will be reused in the hydraulic fracturing process and that pass-by flow requirements are non-existent. These assumptions likely translate to conservative estimates as gas developers can obtain water from other surface water bodies, groundwater or municipal sources and often re-use a small fraction of flowback water (Rahm and Riha, 2012).

Pumping was implemented by subtracting the different scenario amounts from mean daily flows at each reference gaging station. Some smaller streams were pumped dry during certain parts of the year in which case the percent alteration due to pumping was capped at 100%. HSIs were then calculated as the median percent difference between the natural baseline HI value and the values under the high local and cumulative pumping scenarios across all reference gages. HSIs were computed for all HIs with an OOB pseudo-$R^2 \geq 0.8$. Additionally, HSIs were computed for median low, average and high flows (ML, MA and MH), as well as grouped by season. The relationship between stream size and sensitivity to pumping was explored by examining plots of drainage area and percent alteration in various hydrologic sensitivity indices. The sensitivity of the HIT indices was assessed by ranking the HSIs from most to least sensitive for both the high local and cumulative pumping scenarios.

Local (pumped at the gaged reach):

The "local" pumping scenarios reflect pumping from only one site within the gaged reach and were calculated as low and high. In lieu of actual recorded pumping rates we used permitted rates obtained by downloading all available surface water withdrawal data from the Susquehanna River Basin Commission (SRBC) and calculating the average and standard deviation of the permitted pumping rates. These were plotted against the mean annual flow for each pumping location to determine whether a relationship between pumping rate and flow could be established. Interestingly, no significant relationship existed, suggesting that permitted withdrawal rates are (curiously) independent of stream size (data not shown). The low local scenario was therefore calculated as the average SRBC (1.5 cfs) permitted pumping rate less 1 S.D. (0.5 cfs) and the high was the mean plus 1 S.D. (2.5 cfs). We assumed that the pumping occurred for 10 hours each day.

Cumulative (pumped throughout basin):

The cumulative scenario was calculated by assuming newly formed NY regulations regarding gas well development apply everywhere in the Marcellus Region (Best and Lowry, 2014). Specifically, NY regulations currently limit well pad density to no more than 1 pad per 1 mi^2. At each pad there can be as many as 4-9 wells – with wells using between 3-4 Mgal of water. We then developed a range of development scenarios which varied the pad density between 5-30%, number of wells per pad ranging between 4-9 and the number of gallons used per well ranging from 3-4. This equates to an overall water withdrawal ranging from 12-32 Mgal per pad. According to the Susquehanna River Basin Commission, most wells take between 2-5 days to develop (Best and Lowry, 2014, SRBC, 2015). Wells were assumed to take 5 days to develop for the low cumulative scenario and 2 days for the high cumulative scenario. We further assumed that wells were developed sequentially rather than simultaneously per pad.

The assumptions necessary to make this analysis tractable result in a number of limitations regarding interpretation of results and realism of withdrawal estimates. For instance, the rate and timing of pumping will vary through time and space, but we applied a constant daily withdrawal across the entire period of record in order to establish the long-term annual, seasonal and monthly average effects. Accordingly, this approach was not appropriate to assess pumping impacts on the timing-related His. In addition, we note that this analysis ignores *interactive* cumulative impacts resulting from pumping in conjunction with multiple non-shale gas development activities such as industrial water withdrawals, irrigation withdrawals, etc.

Risk Analysis

The sensitivity of a select group of magnitude HIs to surface water withdrawals associated with shale gas development was predicted across all streams within the MSR using RF models. First, a sensitivity index for each HI (HSI) across all reference and non-reference gages was calculated as the percent change in a hydrologic index from the natural baseline under a subset of pumping scenarios chosen to represent low (low local), medium (high local) and high (cumulative high) extraction scenarios. HSIs were predicted for monthly median low, average and high flows for February, April, August and October in order to capture the seasonal flow magnitudes for winter, spring, summer and fall, respectively. We also computed two annual flow magnitude HIs: annual runoff and median annual flow. Next, a training dataset was constructed by associating each HSI with the biophysical attributes of their respective catchments, including anthropogenic factors such as dams. The performance of the RF models was assessed via the OOB pseudo-R^2 and the predictive power of each of the independent variables was assessed via unbiased variable importance plots. HSI predictions were then mapped to polyline shapefiles of all NHD streamlines in the MSR. The results of this analysis should afford insights into how hydrologic sensitivity to water withdrawals varies spatially and should help identify particularly sensitive streams for targeted management. Moreover, the mapped HSIs can be overlaid with species distribution maps and the locations of existing and projected natural gas development to further prioritize streams threatened by hydraulic fracturing activities. For instance, it may be helpful to visualize the coincidence of sensitive streams, high gas development and the presence of a particularly important fish species or functional trait guild (threatened or endangered). Towards that end, we constructed a set of species distribution models (SDMs) for a select group of fish species using binomial RF models (Appendix F). Probabilities of occurrence of functional guilds were computed as the average of the individual species comprising that guild. This resulted in a probability of occurrence prediction at every stream in the MSR for every species and guild of interest.

Results and Discussion

Selecting Flow Indices

An OOB pseudo-R^2 threshold of ≥ 0.8 reduced the number of HIT indices from 171 to 60 (Table 8). Of those 60, 47 were predicted with an R-squared ≥ 0.9, indicating the RF regression models achieved acceptable performance. Applying the remainder of our selection criteria narrowed the list of HIs further to 28. The remaining 28 HIs captured critical components of the natural flow regime, including flow magnitude, duration, rate of change and timing. We chose to retain certain monthly flow magnitude HIs for further analysis as ecological responses to hydrologic alterations are highly seasonal (DePhilip and Moberg, 2010, 2013). Specifically, we chose February to represent winter flows, April to represent spring flows, July and August to represent summer flows and October to represent autumn flows. We should also point out that many of the HIs represent more than one flow regime component. For instance, mean low-flow for April (ML4) reflects both flow magnitude and timing as it is specific to the spring. Likewise, the annual minimum of 7-day moving average flow (DL3) represents both a flow duration and a magnitude.

Table 8. Hydrologic indices with OOB error rate ≤ 20%, retained for further analysis

Index	OOB R-squared	Index Description	Flow Component
ML2	0.95	Monthly Median Low Flow - February	Magnitude
ML4	0.94	Monthly Median Low Flow - April	Magnitude
ML7	0.9	Monthly Median Low Flow - July	Magnitude
ML8	0.9	Monthly Median Low Flow - August	Magnitude
ML10	0.9	Monthly Median Low Flow - October	Magnitude
ML20	0.9	Base Flow	Magnitude
MA2	0.94	Median of the daily mean flow values	Magnitude
MA13	0.97	Monthly Median Average Flow - February	Magnitude
MA15	0.96	Monthly Median Average Flow - April	Magnitude
MA18	0.95	Monthly Median Average Flow - July	Magnitude
MA19	0.94	Monthly Median Average Flow - August	Magnitude
MA21	0.93	Monthly Median Average Flow - October	Magnitude
MA30	0.82	Variability of monthly flow values - July	Magnitude
MA31	0.81	Variability of monthly flow values - August	Magnitude
MA41	0.9	Annual runoff	Magnitude
MH2	0.95	Monthly Median High Flow - February	Magnitude
MH4	0.96	Monthly Median High Flow - April	Magnitude
MH7	0.93	Monthly Median High Flow - July	Magnitude
MH8	0.94	Monthly Median High Flow - August	Magnitude
MH10	0.93	Monthly Median High Flow - October	Magnitude
DH4	0.97	Annual maximum of 30-day moving average flows	Duration

DH5	0.97	Annual maximum of 90-day moving average flows	Duration
DL4	0.9	Annual minimum of 30-day moving average flow	Duration
DL5	0.94	Annual minimum of 90-day moving average flow	Duration
RA6	0.82	Rise Rate - Median of log10 of positive flow changes over entire record	Rate of Change
RA7	0.8	Fall Rate - Median of log10 of negative flow changes over entire record	Rate of Change
TA1	0.83	Constancy	Timing
TA2	0.83	Predictability	Timing

Calculating the Degree of Flow Alteration

Consistent with other published studies, hydrologic alteration was both negative and positive at most non-reference stations for most indices (i.e. observed flows were both less than and greater than the expected natural flow index values predicted from RF models) (Eng et al., 2012; McManamay et al., 2013). Despite little change in overall annual discharge volumes, hydrologic alteration displayed an overall trend of decreased high flows, increased low flows, and greater flow stability (Figure 7; computed as the deviation between predicted and observed flow attributes for the non-reference gages below 2,500 km^2). The largest flood magnitude decreases were evident in February and October, and were somewhat more pronounced for the NEAHCS “small” and “medium” river classes relative to the “headwater” and “creek” classes. In contrast, the percentage increases in low flows were greatest in July, August and October, and were strongest for the smallest systems. Notwithstanding these trends, substantial variation across individual gages was apparent, with the range of alteration extending to approximately 100% change in both positive and negative directions for most flow metrics.

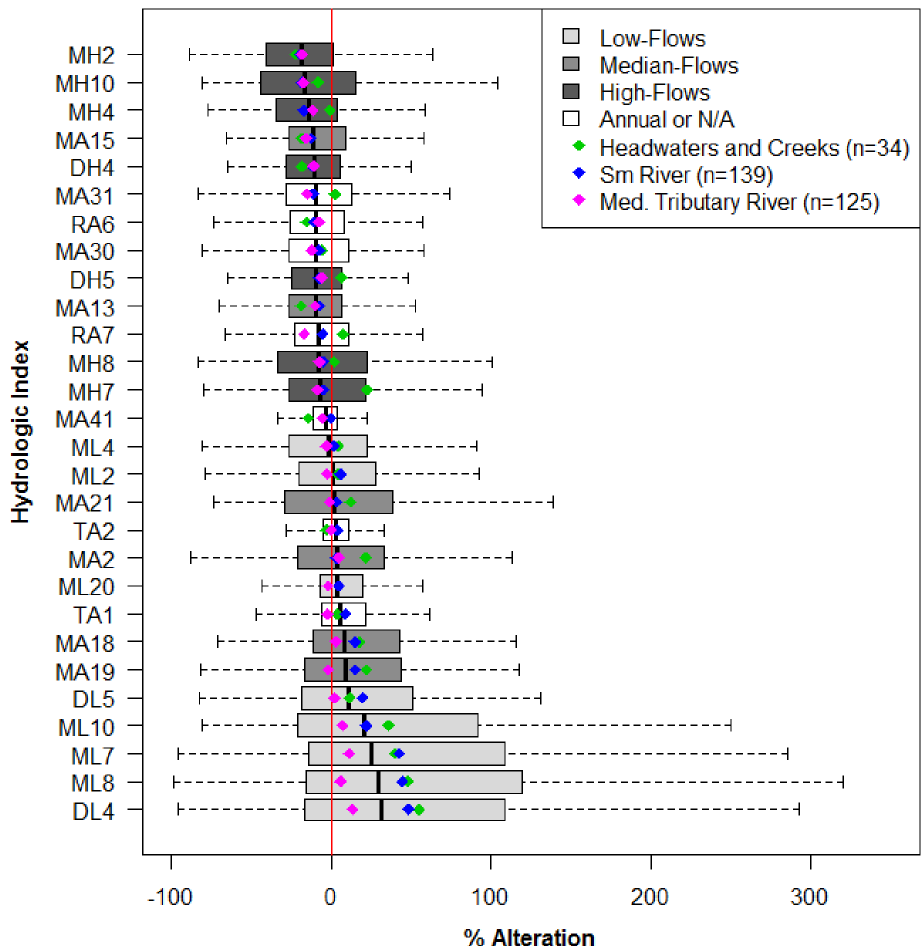

Figure 7. Observed flow alteration for non-reference gages (n=298 with drainage area < 2500 km2) across well-predicted flow metrics (pseudo R2 > 0.8). Median values for headwater reaches (green diamonds), small rivers (blue diamonds) and medium rivers (purple diamonds) are illustrated in addition to the full sample median (solid black bars). The red vertical line indicates no deviation between the value calculated from the observed flow record and the expected natural value predicted from the per-metric model fit to reference gages.

Stream Classification

The random forest classification model predicted a total of four different stream classes with an 87% OOB accuracy which we deemed acceptable (Table 9). Based on the hydrologic characteristics of each class we hypothesized the different degrees of sensitivity to flow alteration due to water withdrawals associated with hydraulic fracturing activities (Table 9). For example, given the high flow variability, propensity for intermittent flows and low minimum flows and low baseflows, perennial flashy streams were theorized to be the most sensitive to water extraction. We will test these hypotheses in the pumping scenario section.

Table 9. Stream class names, codes, narrative description and geographic setting as per McManamay et al. (2014).

Class Name	Code	Description	Geography	Hypothesized Sensitivity to Withdrawals
Stable High Baseflow	SHBF	High Baseflow Index, Low Variability, High minimum & low flows, low frequency of high flow events, low rise rates	Blue Ridge Mountains	Low
Perennial Runoff 1	PR1	Similar to SHBF but lower baseflows, semi-stable	Eastern piedmont	Low-Moderate
Perennial Runoff 2	PR2	Similar to PR1, but lower baseflows and higher runoff than PR1	Eastern Appalachians	Moderate
Perennial Flashy	PF	High variability, some intermittency, low minimum & baseflows, high frequency of high flows, high rise rate	NA	High

Variable importance analysis revealed that baseflow index, drainage area, average temperature, soil properties (e.g. percent silt or sand), mean elevation, percent of basin with poorly drained soils and latitude/longitude of the basin centroid were the some of most influential predictors of stream class (Figure 8).

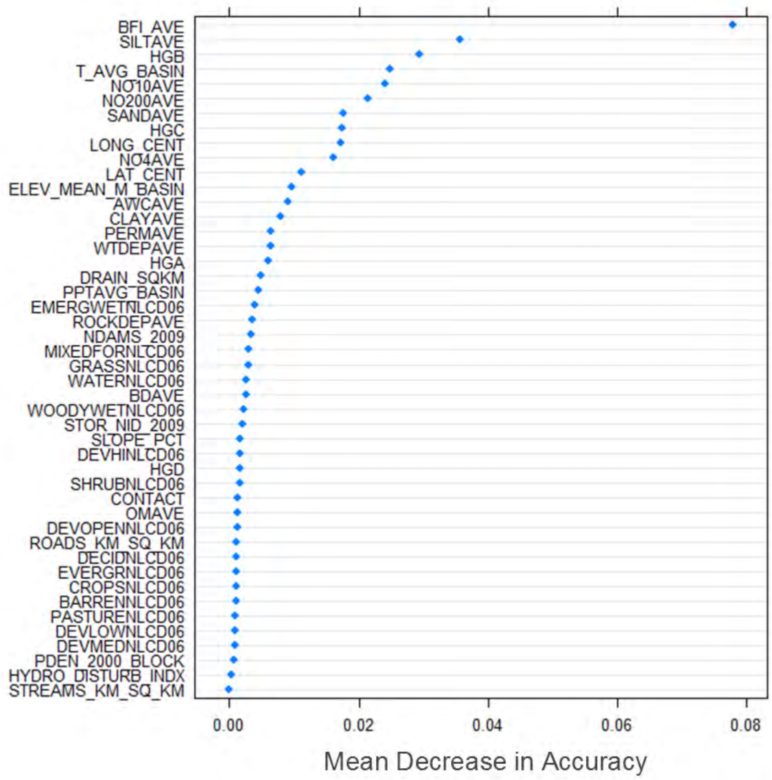

Figure 8. Unbiased variable importance plot from conditional random forest model of stream class within the MSR. Variables are ranked from top-to-bottom according to their relative predictive power.

Figure 9 depicts the results of the random forest predictions across the Marcellus Region. The different colors represent different NHD reaches classified into one of the four stream types. The majority of the basins were classified as “perennial runoff”, which is consistent with McManamay et al. (2013).

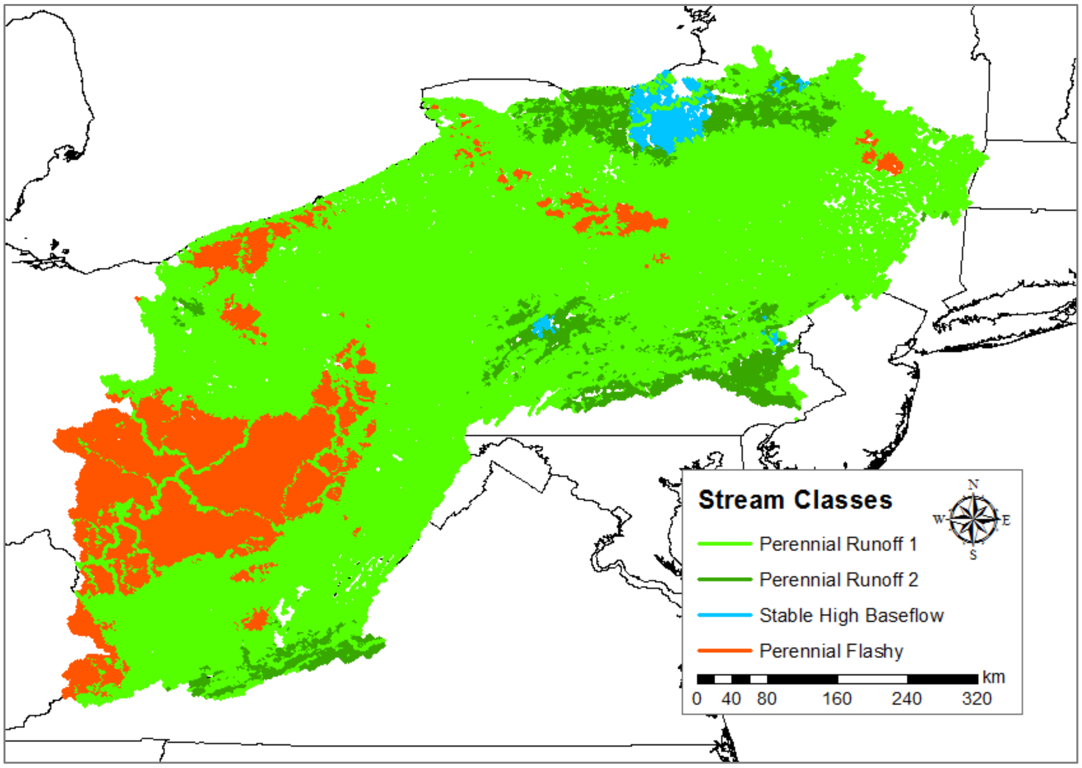

Figure 9. Stream classes predicted across all gaged and ungaged basins in the MSR by random forest models.

Flow-Ecology Relationships

Flow-ecology relationships were developed by pairing the 373 non-reference USGS gages with 11,104 unique MARIS sites on the basis of whether they shared an NHD reachcode. This resulted in a total of 83 USGS gages paired with 176 MARIS fish sampling sites. In many cases, USGS stations were matched to more than one MARIS site, in which case a reach-averaged ecological metric was calculated in order to control for differing sampling efforts.

Quantile regression revealed significant relationships between % alteration in flow and numerous ecological metrics at the 90th quantile ($\alpha = 0.05$). The majority of F-E relationships exhibited a wedge-shaped distribution with negative slopes and, in many cases, significant scatter. As previously mentioned, the high degree of variability is to be expected as aquatic biota respond to a variety of non-flow related natural and anthropogenic factors (e.g. eutrophication, toxic chemical pollution, acidity, sediment, temperature and pathogens). Flow alteration was predominantly in the negative direction for most HIs. Additionally, most statistically significant F-E relationships were associated with reduced HIs. This is consistent with Buchanan et al. (2013) and McManamay et al. (2013), who found that anthropogenic flow disturbance primarily reduced rather than increased HIs – especially magnitude-related HIs. They, along with Carlisle et al. (2010), also found that F-E relationships were much more predictable in the negative direction.

The x-axis of all F-E relationships depicts the degree of flow alteration in both negative and positive directions. Thus, zero represents no change in a given HI, indicating hydrological conditions are close to the natural baseline. Fish metrics that decline as % flow alteration becomes more negative (right-to-left from zero) indicate a decrease in fish-response with increasing negative flow alteration (i.e. a negative correlation). Similarly, fish metrics that decrease with increasing positive flow alteration (left-to-right from zero) indicate declining fish-response to inflation of a given flow index. Slopes of the QR regression lines provide a measure of the strength of the relationship between dependent and independent variables. Although we present F-E relationships for both negative and positive flow alteration, here we discuss only the negative side as this is more relevant to the effects of water withdrawals for gas development.

All points in the F-E distributions were colored according to like physiographic province – with different shapes indicating stream class membership. A key to the various physiographic provinces is provided in Figure 10.

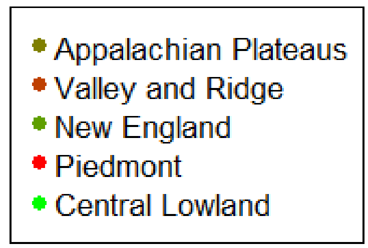

Figure 10. Key to physiographic provinces as depicted in all flow-ecology relationships.

Contrary to ELOHA theory, our visual examination of the F-E points across all ecological endpoints revealed little clustering on the basis of stream class or physiographic province. This suggests that F-E relationships derived in this study can be applied across the entire MSR. However, we should caution that many stream classes were not well represented in our paired USGS-MARIS sites. For instance, the vast majority of sites were in the "Perennial Runoff 1" stream class; with no sites characterized as "Stable High Baseflow". The lack of adequate sample size precludes definitive evaluations of class-specific F-E relationships and should be considered as a topic for future research. In particular, the lack of flow-ecological data in several stream classes provides impetus for the design of targeted ecological monitoring campaigns to better describe F-E relationships in these classes.

Species Richness

Based on F-E hypotheses from DePhilip and Moberg (2010, 2013; Appendix B) and a preponderance of published literature, we anticipated that species richness and abundance would respond negatively to reductions in magnitude, timing and duration of flows. DePhilip and Moberg (2010, 2013) also suggest that increased rate of change or flashiness should result in similar ecological declines, however evidence predicting fish response to decreases in flashiness is less consistent. In the absence of clear guidance, we make the assumption that any substantial departure from natural fall rates (negative or positive) will result in reduced species richness and abundance. It is important to note that while water withdrawals for hydraulic fracturing purposes are extremely unlikely to increase magnitude or duration HIs. In the case of rate-of-change indices, we expect withdrawals to reduce rise rates (RA6), whereas fall rates (RA7) may increase (a positive % flow alteration).

Three ecological metrics were significantly related to fluvial-fish species richness at the 90th quantile, including: average August flow (MA19), rise rate (RA6) and annual runoff (MA41; Figure 11). In the case of MA19 and MA41, species richness declines with increasing departure from natural baseline conditions (i.e. depleted flow magnitudes). Reductions in flow magnitude are generally associated with decreases in habitat availability and quality. Negative effects may include: accumulation of fine sediments due to a reduction in flushing events, reduction in depth and associated dewatering of riffle habitats, increases in stream temperatures and decreases in dissolved oxygen (D.O.), reductions in preferred spawning habitat, and increases in predator-prey interactions (DePhilip and Moberg, 2010, 2013). Numerous researchers have documented significant, predominantly deleterious, changes to native species richness, abundance and assemblage composition resulting from these direct and indirect effects of flow regime alteration (Knight et al., 2013; McManamay et al., 2013; Poff and Zimmerman, 2010; Rolls and Arthington, 2014). Of particular relevance to this study are the finding of Armstrong et al. (2010) who showed that depletion of median August discharge due to water abstraction resulted in substantial declines in both species richness and abundance.

The slope value associated with MA41 suggests that every 10% drop in annual runoff will result in a loss of approximately five fluvial fish species (Figure 11). The higher slope value of MA41 relative to MA19 suggests that fish diversity is more sensitive to changes in annual runoff than to reductions in average August flow. This is somewhat counterintuitive as August generally represents a warm, low-flow period during which many fish species are particularly vulnerable to hydrologic alterations (DePhilip and Moberg, 2010). In Massachusetts, Armstrong et al. (2010) observed a loss of one fluvial species with each 14% decline in August median flow at the 90th quantile. Here in the MSR, an equivalent drop in August flow would result in a loss of roughly 4 species.

On the other hand, increased negative alteration in rise rate (RA6) was associated with increased species richness. This implies that a more stable flow regime leads to greater species diversity, perhaps due to a reduction in scouring events and extreme changes in flow rates that may set an ecological limit on fish populations. This is consistent with other studies that detected reduced native species richness due to increasing flow variability – especially for species with narrow hydraulic niches such as shallow, fast-flowing riffles (Gehrke et al., 1995; Meador and Carlisle, 2012; Rolls and Arthington, 2014).

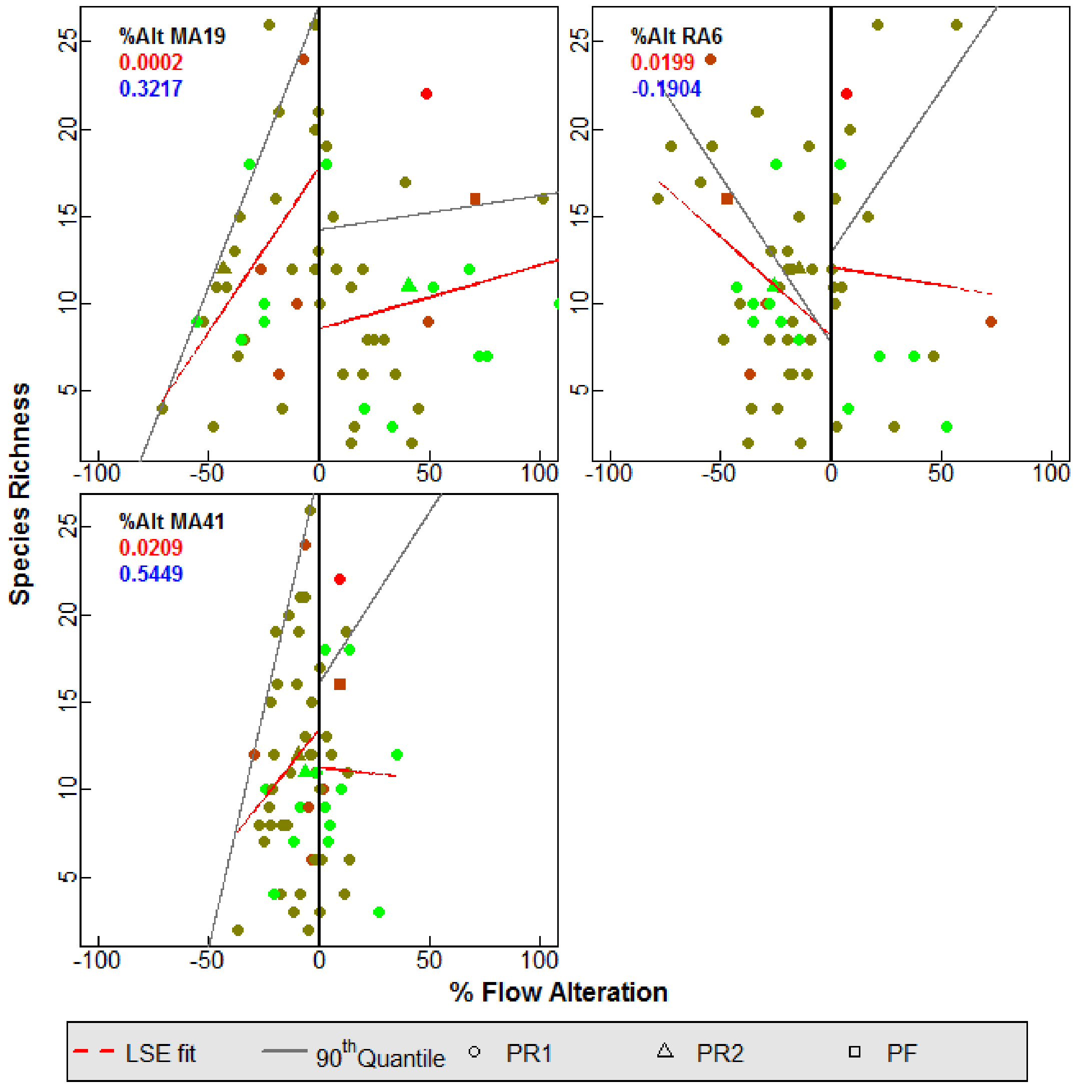

Figure 11. Plots of species richness vs. percent alteration in median August flow (MA19), rise rate (RA6) and annual runoff (MA41). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

As previously noted, water abstraction for hydraulic fracturing purposes may increase fall rates (RA7), resulting in a positive alteration. We found a significant negative correlation between species richness and increased fall rate (i.e. positive alteration; Figure 12). Among other adverse effects, accelerated fall rates may limit time for passage of fish between feeding and spawning areas and can lead to the stranding of fish in isolated pools (Knight et al., 2008).

Interestingly, although the relationship was not significant (p-value=0.09), we observed a positive relationship between negative alteration in RA7 and fish diversity, similar to RA6. The opposite responses to positive and negative alteration in RA7 indicate an overall preference for less flashy flow regimes. However, since we did not distinguish between native and non-native species, we cannot rule out the possibility that a more stable flow regime is favoring an influx of non-native species – inflating the total species richness score.

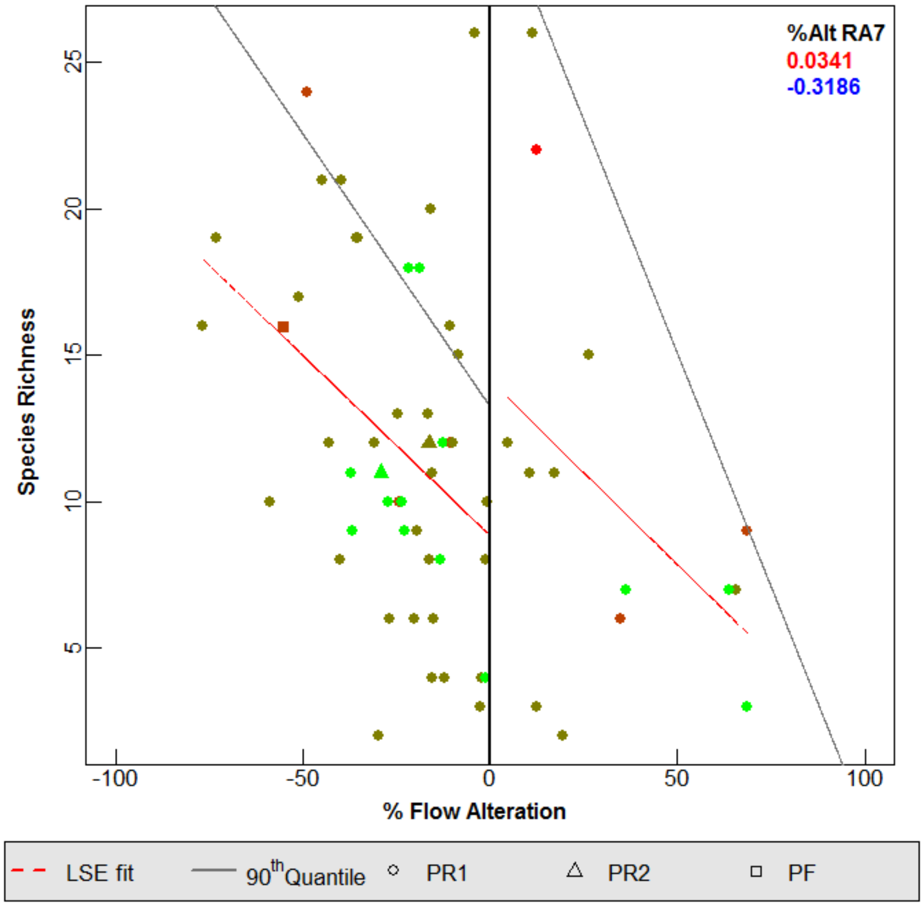

Figure 12. Plot of species richness vs. percent alteration in fall rate (RA7). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Total Abundance

Total fish abundance (CPUE) at the 90th quantile was negatively correlated with % alteration of two different flow magnitude-related HIs: April high flow (MH4) and August mean flow (MA19) (Figure 13). Similar to total species richness, this indicates that fish abundance in the MSR is also sensitive to a loss of habitat availability and quality associated with reduced flows. In particular, it seems abundance is sensitive to reductions in spring and summer discharge. The steepest 90th quantile slope was observed for mean August flow, indicating total fish abundance is more responsive to disturbance during this season. Spring flows help to maintain sandy spawning substrates, serve as important spawning cues and ensure connectivity to upstream tributary habitats (DePhilip and Moberg, 2010). Thus, reductions in spring discharge may disrupt flows during the breeding season for many species resulting in decreased recruitment due to impaired growth and survival of eggs and juvenile fish. However, we should emphasize the presence of a clear outlier, as well as the lack of parallel least squares and 90th quantile regressions lines suggests the April high flow F-E relationship may be spurious.

Summer flows (i.e. MA19) generally set strong ecological limits on fish populations. Depletion of summer discharge reduces available habitat volume, increases stream temperatures, lowers D.O., and dewaters vulnerable shallow-water habitats (DePhilip and Moberg, 2010). This explains the negative correlation between fish abundance and median August flows. These findings are corroborated by numerous other researchers who found lowered fish abundance in response to flow alteration. For instance, Freeman and Marcinek (2006) found that reduced flows negatively affected the persistent of shallow habitats, which was shown to be strongly related to juvenile fish abundance in the Tallapoose River, Alabama. Likewise, researchers in Massachusetts found that fluvial species abundance was negatively correlated with alteration of August median flow (Armstrong et al., 2011). Moreover, McManamay et al. (2013) and Poff and Zimmerman (2010) demonstrated that changes in flow magnitudes lead to predictable declines in fish abundance.

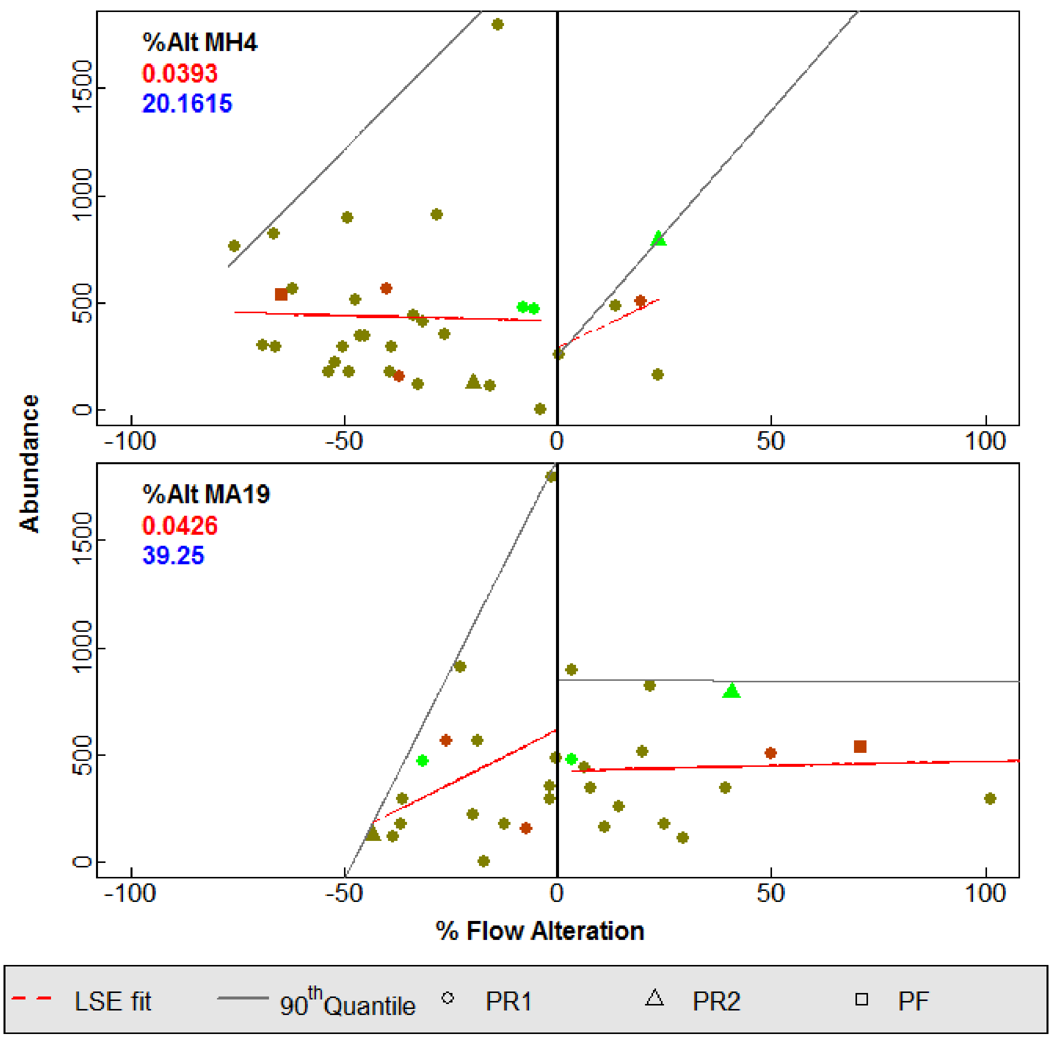

Figure 13. Plots of total abundance vs. percent alteration in April high flow (MH4) and median August flow (MA19). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Tolerance

We anticipated that the proportion of fish species tolerant to environmental degradation would increase in response to increasing flow alteration. However, we detected no significant relationships between the proportional abundance of tolerant species and % flow alteration, suggesting tolerance is relatively insensitive to changes in flow regime in the study region.

In contrast, the proportional abundance of species intolerant to anthropogenic disturbance decreased, significantly, as HIs departed from their natural baseline conditions (i.e. flow variability in July and August (MA30 and MA31, respectively), February high flow (MH2) and increased fall rates (RA7) (Figures 14 and 15). These relationships are consistent with the intuitive notion that species intolerant to habitat degradation would decline in the face of progressively altered flow regimes. Slopes of the 90th quantile regression lines were > 0.7 in most cases, indicating that every 10% change in these HIs result in the loss of roughly 7 individuals poorly adapted to habitat degradation and water quality impairment. Knight et al. (2013) observed lower rates of intolerant species loss with increasing flow alteration in the Tennessee River Basin than shown here (i.e. roughly 3 vs. 7). Our results are not, however, directly comparable to Knight et al. (2013) as they computed flow alteration as the cumulative departure from the natural baseline, regardless of whether the alteration was positive or negative. In addition, they determined flow alteration as the percent difference between observed vs. an estimated regional baseline.

Rolls et al. (2012) suggest that such flow regime changes can lead to reduced habitat and water quality through increased contaminant concentrations due to less water available for dilution. Similarly, other studies have found that flow regime changes decreased the proportion of intolerant fish and macroinvertebrate species and attribute it to the combined effects of cumulative thermal and water quality stress and habitat degradation (DePhilip and Moberg, 2010, 2013)

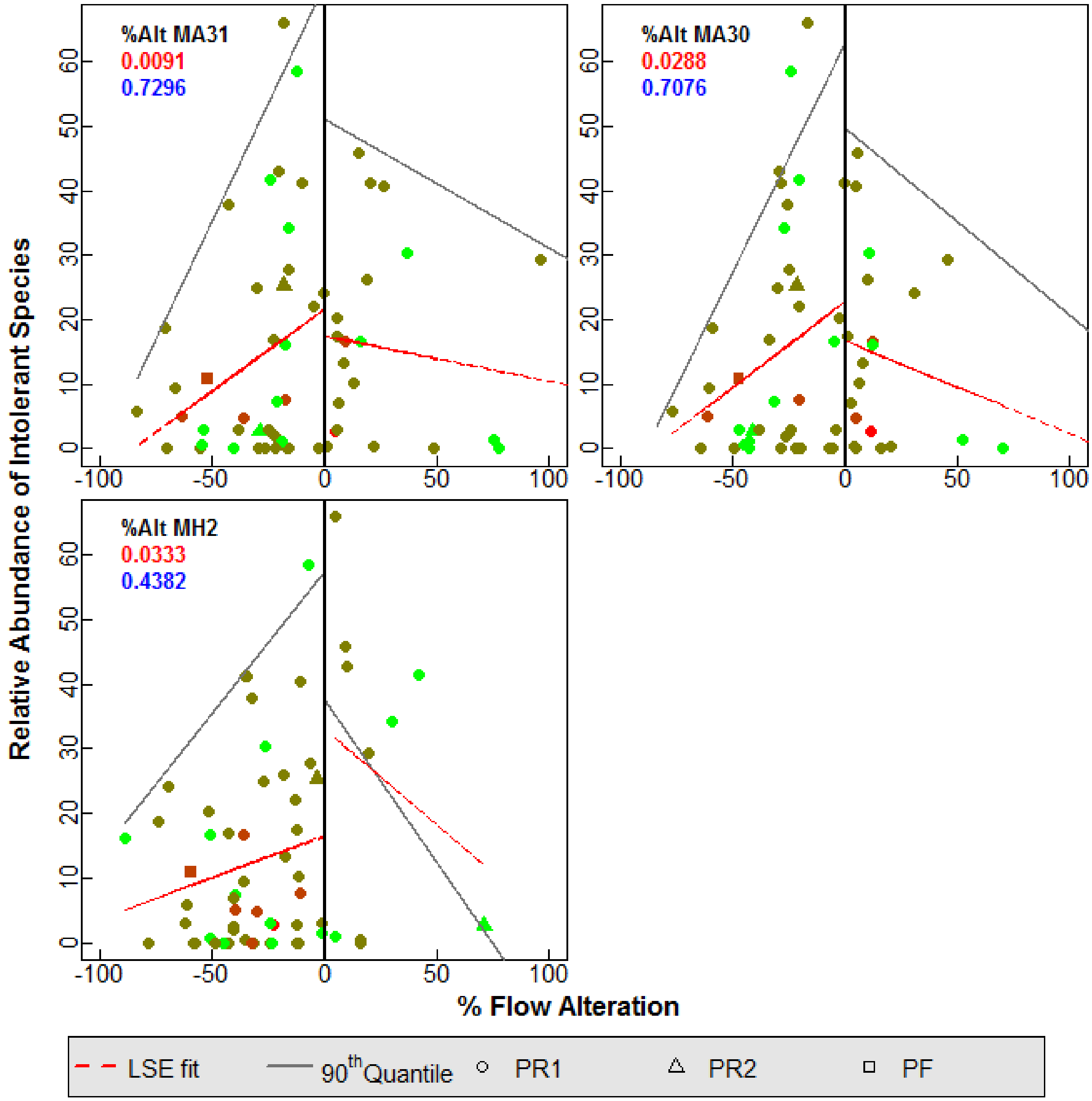

Figure 14. Plots of relative abundance of intolerant species vs. percent alteration August and July flow variability (MA31 and MA30, respectively) and February high flow (MH2). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

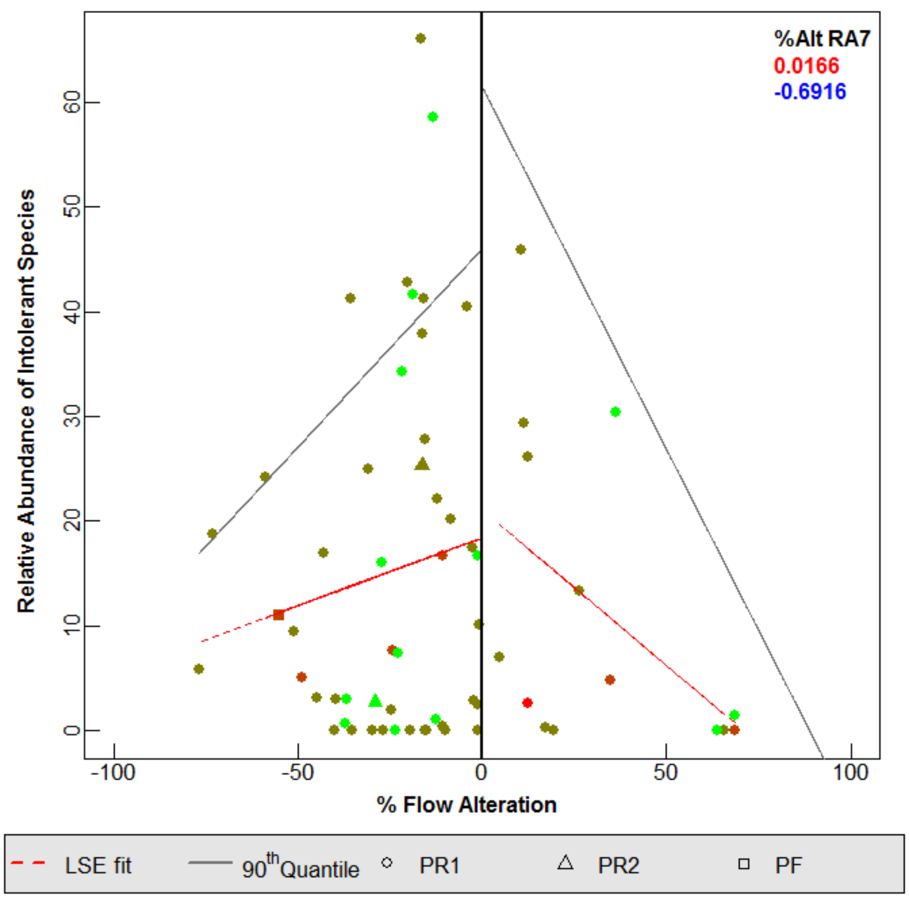

Figure 15. Plot of relative abundance of intolerant species vs. percent alteration in fall rate (RA7). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Trophic Structure

Piscivores

Based on F-E hypotheses proposed by DePhilip and Moberg (2010, 2013; Appendix B), we anticipated that decreases in flow magnitudes during summer low-flow periods would result in a loss of refugia and a shift in trophic composition towards top-predator dominated systems. The positive correlation between reductions in August flow (MA19) and the relative abundance of piscivorous species supports this hypothesis (Figure 16). It is also worth noting that relative abundance of top predators between 5 and 20% has been associated with healthy, trophically diverse fish communities. However an overabundance of piscivores (>20%) suggests stream degradation (Gleason, 2007). The slope of the 90th quantile line suggests that once August flow has been reduced by greater than 17%, piscivores exceed 20% of the community abundance – indicating impaired ecological conditions.

Lower summer flow variability (MA30 and MA31) and spring flows (MA15) were associated with lower proportional abundance of piscivorus fish. Although not directly comparable, our findings seem to contradict those of Poff and Allan (1995) and Pyron and Lauer (2004) who generally found that piscivory increased with increasing hydrologic stability. We speculate that reduced piscivory with decreasing July and August flow variability may be related to indirect adverse effects to their prey-base. For instance, lower August flow variability may limit periphyton and macroinvertebrates, which, in turn, reduces the abundance of prey-fish for piscivores (Delong et al., 2011; Osmundson et al., 2002). This is in accordance with the idea that trophic specialists may respond more negatively to flow disturbance, relative to generalists, as this may decrease the diversity of the overall food web (Gleason, 2007)

Similarly, depressed spring flows (MA15) may reduce piscivory by disrupting spawning of native fish (spawning mis-cues and altered nursery habitat), decreasing the abundance of suitably-sized prey. Indeed, Franssen et al. (2007) found that natural flow regimes favored higher densities of prey fish that were within the gape dimensions of the piscivorous Pike minnow (Ptychocheilus lucius), while "under an artificially depressed flow regime, native prey fishes have lowered spawning success and nonnative species often exceed the gape dimensions of age-1 *P*. lucius until later in the summer".

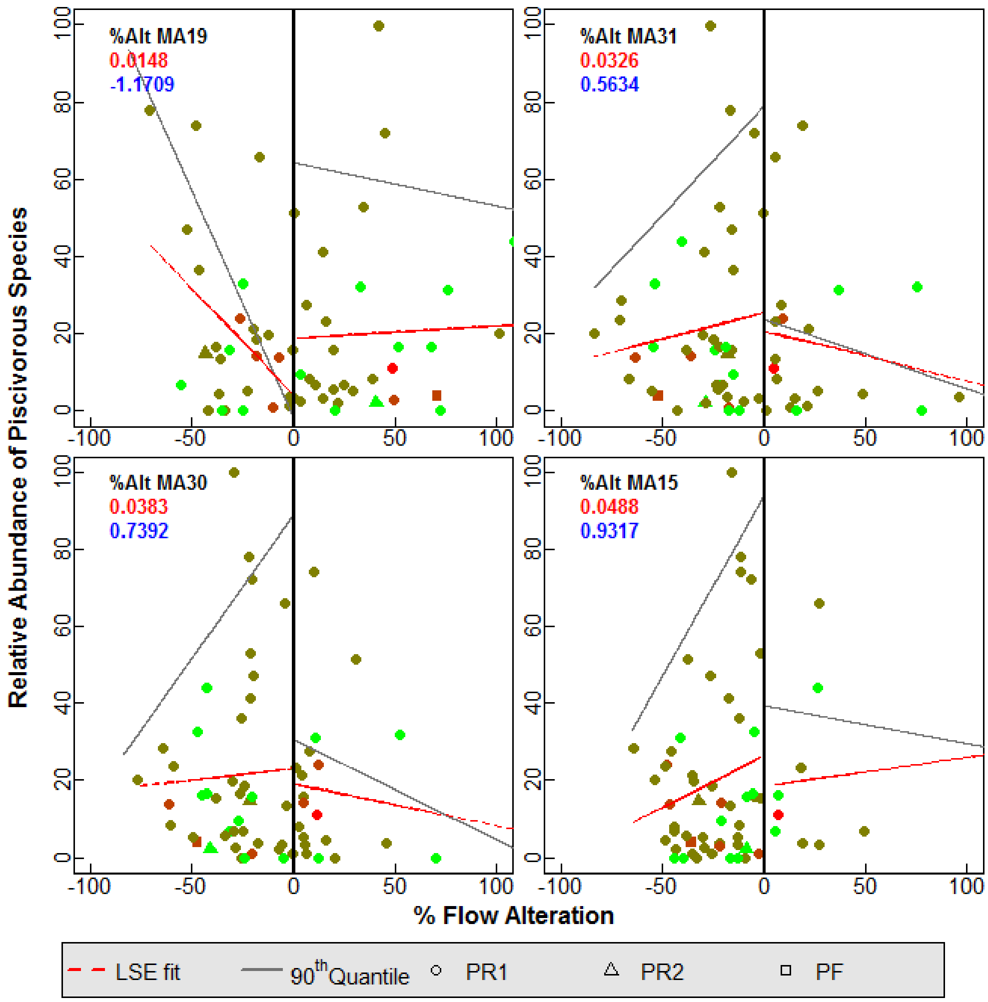

Figure 16. Plots of relative abundance of piscivorous fish vs. percent alteration in median August flow (MA19), August and July flow variability (MA31 and MA30, respectively) and median April flow (MA15). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Invertivores

Generally speaking, a high relative abundance of invertivores is indicative of a healthy fish community (Niemela and Feist, 2002). Thus, we hypothesized that increased flow alteration would result in reduced invertivore abundance. The proportional abundance of invertivorous fish was significantly related to only one hydrologic index: February high flow (MH2) (Figure 17). Compared to other seasons, there are relatively few studies evaluating fish responses to flow alteration in winter, but there is evidence that overwinter survival of insectivores may be reduced because changes in winter flows can result in decreased prey abundance. Research has demonstrated, for example, that macroinvertebrate communities are substantially impaired by reductions in winter low- and high-flows (Rader and Belish, 1999; Carlisle et al. (2012). Whether this is related to increased anchor ice formation, thermal modification, changes in migration cues or reductions in macroinvertebrate species with a high-flow preference remains to be seen.

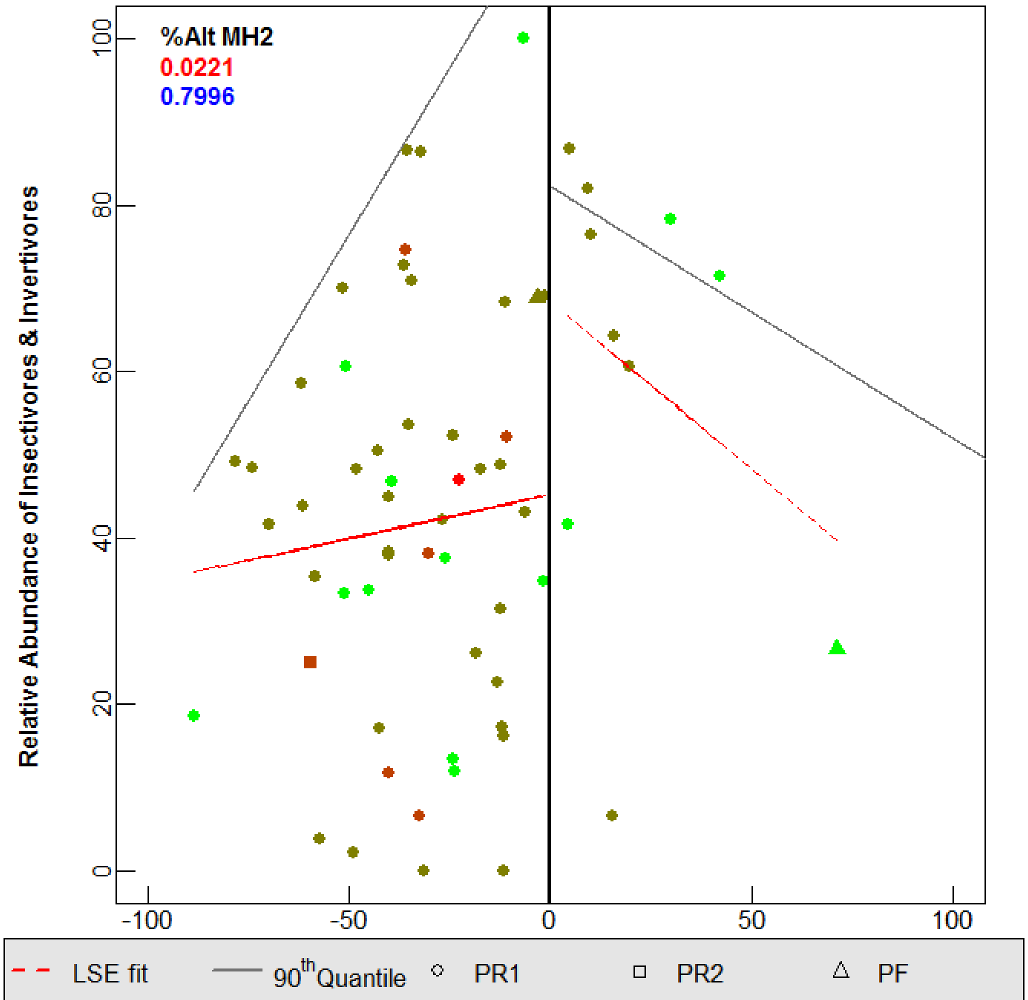

Figure 17. Plot of relative abundance of insectivores and invertivores vs. percent alteration fall rate (RA7). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Generalist

Based on previous research, we anticipated that the abundance of fish in the generalist trophic guild would be favored in increasingly altered flow environments. For example, Poff and Allan (1995) found that sites characterized by heightened flow regime disturbance were associated with trophic and habitat generalist species. Similarly, Freeman and Marcinek (2006) observed that altered hydrology related to water withdrawals in Piedmont streams were associated with marked declines in fluvial specialist species, favoring instead, trophic and habitat generalists.

In contrast to these previous studies, we detected a significant negative correlation between trophic generalist abundance and percent negative flow alteration in October flow (MA21) (Figure 18). In a study designed to evaluate the effects of water withdrawals on fish assemblages in the Susquehanna River Basin, Shank and Stauffer (2014) also expected to find increases in macrohabitat generalists. Instead, they noted greater proportions of generalists in less altered sites and suggested that flow changes resulting from withdrawals may not substantially impact macrohabitat generalists.

As previously stated, we combined omnivores and trophic generalists together for analysis. Although studies directly linking omnivorous fish guilds with flow alteration are rare, limited existing data suggests that the abundance of omnivorous fish should actually decrease in the face of increasing flow alteration. For instance, in streams located in the Ridge and Valley physiographic province of the Tennessee River Basin, Knight et al. (2013) demonstrated that omnivores decline significantly at the 85th, 80th and 30th quantiles with increasing hydrologic departure. In the Wabash River in Indiana, Pyron and Lauer (2004) found that sites with higher hydrologic variability were negatively correlated with omnivorous feeding strategies. The fact that we combined omnivores and generalists may have confounded this analysis.

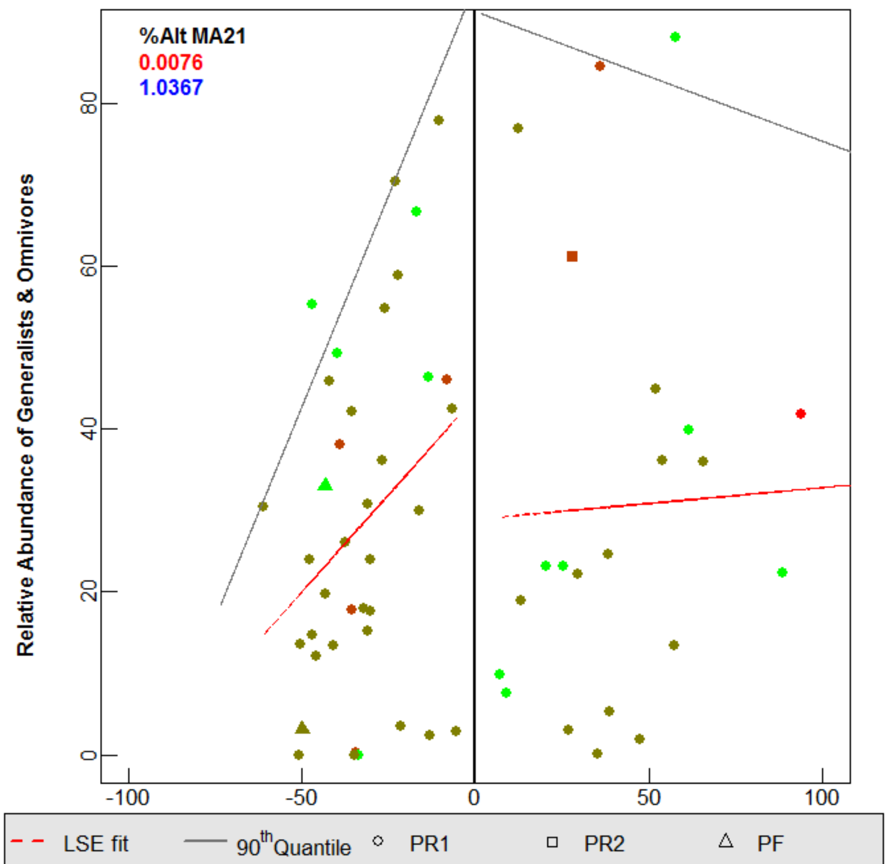

Figure 18. Plot of relative abundance of generalists and omnivores vs. percent alteration October flow (MA21). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Herbivores

Literature examining relationships between herbivorous fish and flow alteration is lacking. Thus, it was difficult to formulate F-E hypotheses specific to this group. Intuitively, one might expect that changes in flow will lead to alterations in siltation rates, substrate composition and aquatic vegetation assemblages, which may affect herbivores. Although they did not specifically evaluate fish, Chester and Norris (2006) found that sites downstream of dams with altered hydrology experienced shifts in periphyton composition. This brought about substantial declines in herbivorous macroinvertebrates utilizing periphyton as a food source. Berkman and Rabeni (1987) and Osmundson et al. (2002) both found that higher sedimentation rates negatively affected herbivorous fish – hence, any flow disturbance that alters sediment transport dynamics may cause trophic shifts away from herbivorous strategists.

Results of our quantile regression analyses indicate somewhat inconsistent relationships between herbivory and negative flow alteration (Figure 19). For instance, decreasing magnitudes of 30-day low flow (DL4) were positively correlated with herbivory, whereas annual runoff (MA41) was negatively correlated. Additionally, increased fall rates (RA7) resulted in significant reductions in the relative abundance of herbivores (Figure 20). The lack of consistent trends suggests herbivores may not provide a suitable, predictable measure of flow alteration.

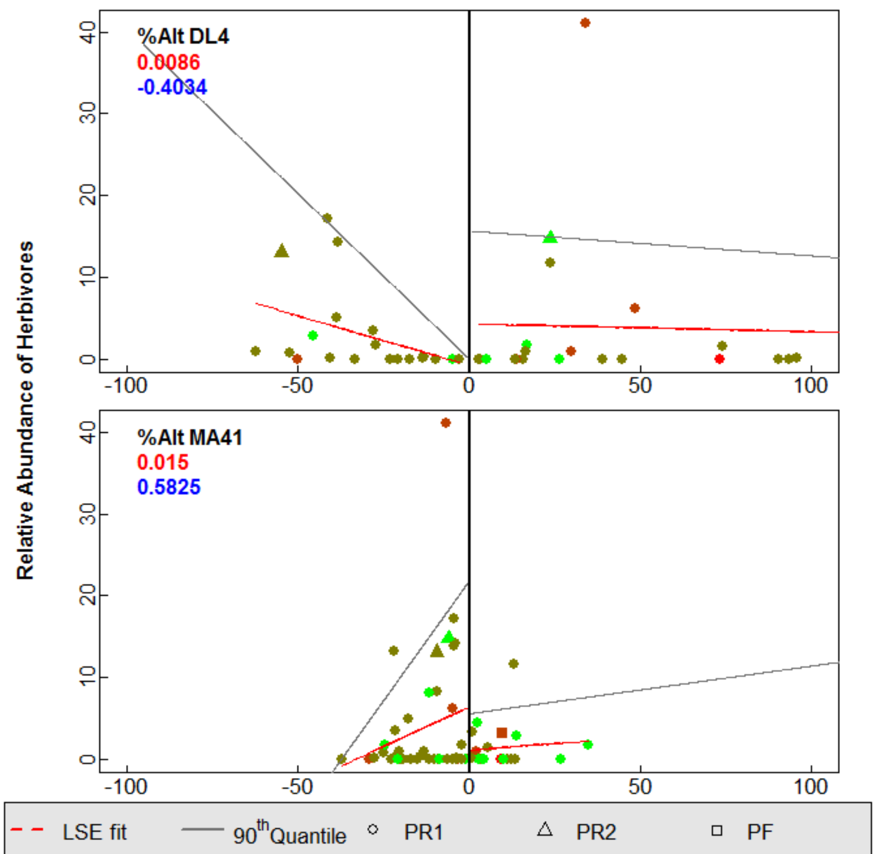

Figure 19. Plots of relative abundance of herbivores vs. percent alteration 30-day low flow (DL4) and annual runoff (MA41). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

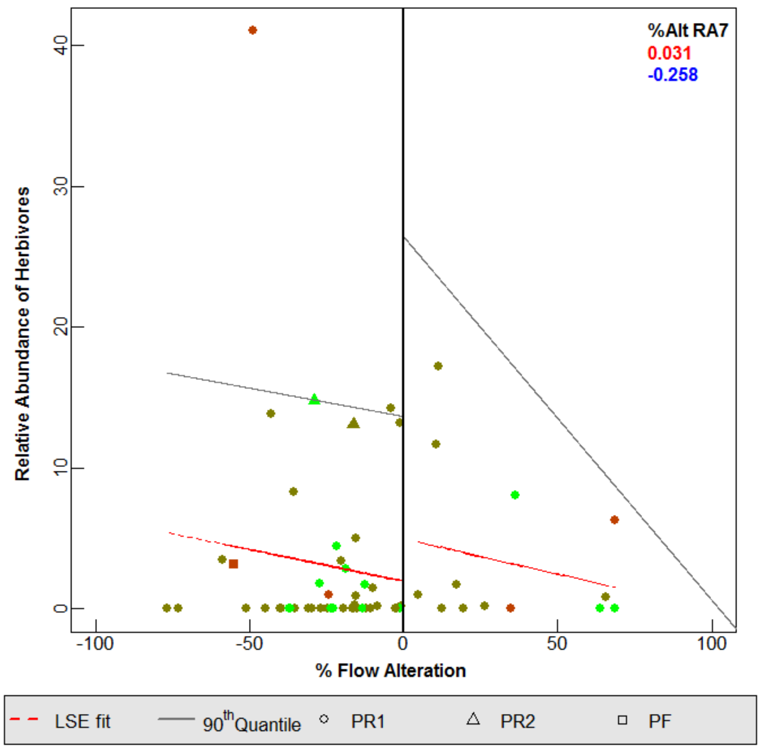

Figure 20. Plot of relative abundance of herbivores vs. percent alteration in fall rate (RA7). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Life History Strategies

Ecological theory predicts that periodic life history strategists will be favored in streams with seasonal, yet predictable flow regimes that create periodically suitable environments (Winemiller and Rose, 1992; Winemiller, 2005). Opportunistic fish species, on the other hand, are typically well-suited to stream environments characterized by frequent and intense disturbances. Equilibrium strategists are typically found in streams with stable, predictable flow regimes with low variation in habitat quality and strong biotic interactions. Accordingly, we formulated the following flow-ecology hypotheses

Periodic Strategy Weights

We anticipated that periodic life history strategists would decline in the face of reduced seasonality or predictability of flow (Winemiller and Rose 1992; Winemiller 1995). Our quantile regression results support this hypothesis in that reduced seasonal flows in the month of October (MH10) were associated with reduced abundance of periodic strategists (Figure 21). However, the negative relationship between periodic abundance and % alteration in July and August flow variability (MA30 and MA31, respectively) was somewhat unexpected. Typically, periodic strategists, with high fecundity and large age at maturity are negatively related to flow variability (McManamay et al., 2014a; Mims and Olden, 2013). Gido et al. (2013) also observed responses in fish life history strategies that were in conflict with ecological theory. They attributed discrepancies to "greater variation in key aspects of flow regimes (variability, predictability, and seasonality) across regions than within river systems across years. Even within the limited regional extent of our study, there was as much variation in flow attributes among river systems as within systems. Thus, species with different life-history strategies, once established, might not respond consistently to more subtle differences in flows across years". They further suggest that "differences in other ecological traits might override interannual variation in abundance attributed to trilateral life-history traits (fecundity, size and maturity, and parental investment). For example, because flow magnitude is tightly linked to temperature (e.g., Gido and Propst, 2012), a species' thermal preference might predict response to flow attributes such as mean spring or summer discharge".

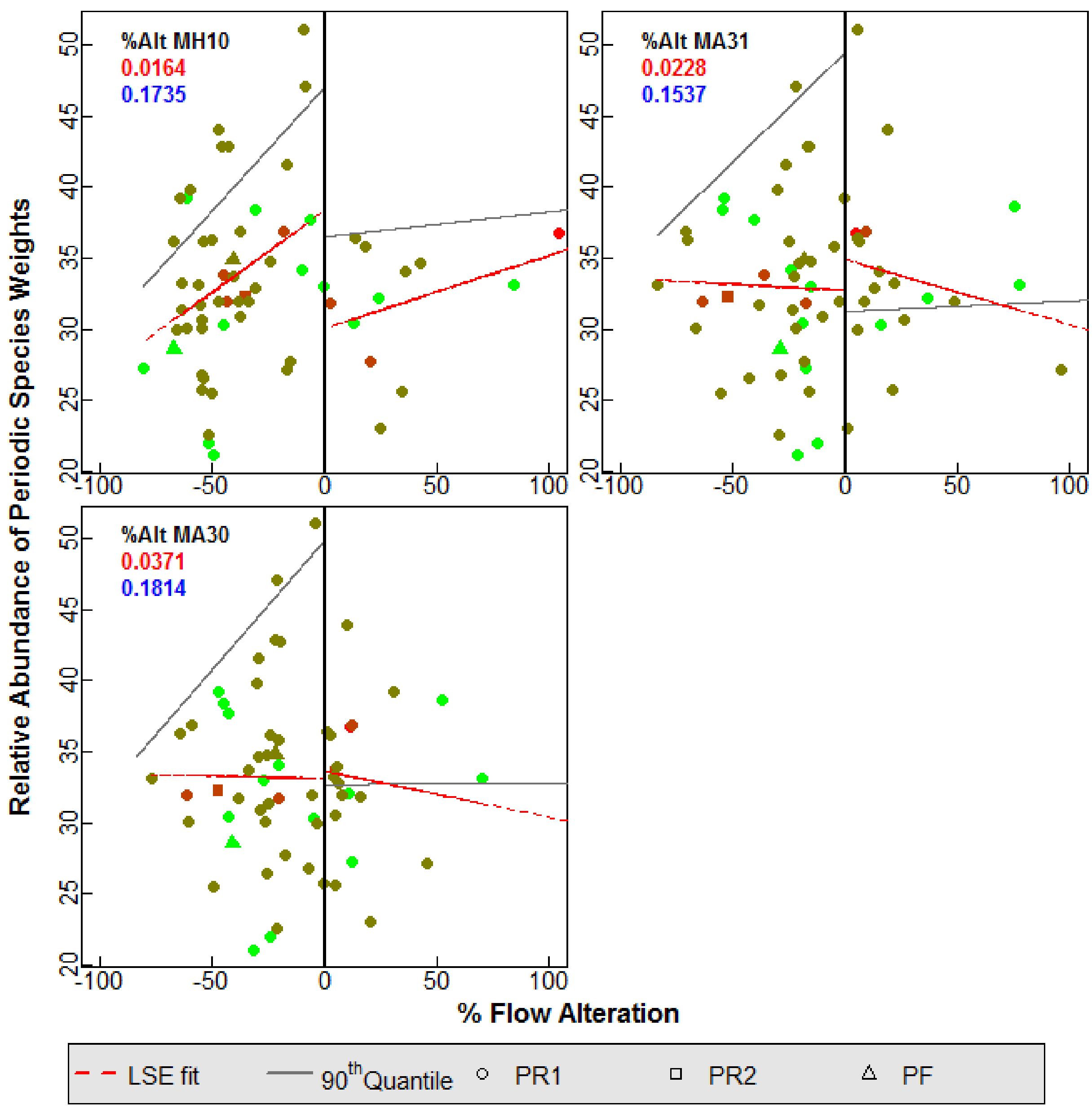

Figure 21. Plots of relative abundance of periodic strategists vs. percent alteration in October high flow (MH10), and August and July flow variability (MA31 and MA30, respectively). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Opportunistic Strategy Weights

Our findings regarding opportunistic strategists are somewhat contradictory to ecologic theory in that lower winter flows (ML2) were significantly associated with fewer opportunistic strategists (Figure 22). Discrepancies between our results and the expected ecological outcome may be explained by the fact that we did not distinguish between native and non-native species in our analysis of life history strategies. Research has demonstrated that anthropogenic flow alteration can have opposite effects on native vs. non-native opportunistic species, whereby non-natives are favored in increasingly disturbed flow regimes.

We should note, however, that we did observe a borderline significant (p-value = 0.054) positive relationship between annual runoff (MA41) and the relative abundance of opportunists. This suggests that opportunistic strategists are favored in streams with artificially low annual flows, whereas reduced winter flow has the opposite effect.

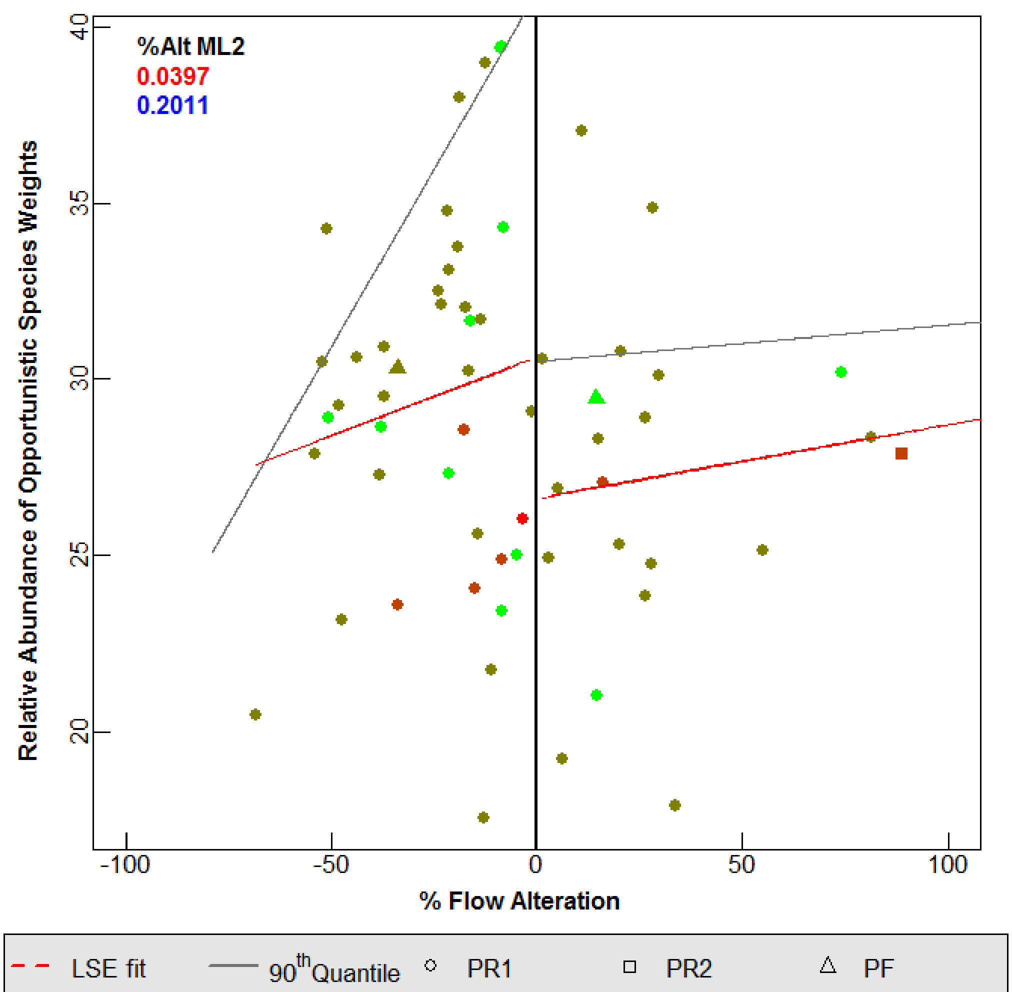

Figure 22. Plot of relative abundance of opportunistic species vs. percent alteration in February high flow (ML2). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Percent Equilibrium Weights per REACH

Quantile regression revealed no significant relationships in equilibrium strategists and flow alteration. This may be due to the fact that the stream systems within the MSR are likely dominated by equilibrium species. In contrast, opportunistic and periodic species are far fewer and, therefore, may be more likely to demonstrate stronger F-E patterns.

Functional Guild & Sentinel Species

Cold Headwaters Guild

The cold headwaters functional guild consisted of sculpins, brook trout, and brown trout. As they share similar thermal requirements, it was expected that they would respond similarly to flow alteration. In particular, we hypothesized that they would decline with increasing flow alteration as this may alter thermal regimes. QR results support this supposition in that February low flow (ML2) and base flow index (ML20) were significantly negatively correlated with abundance of cold headwater species (Figure 23). The slope of ML20 was especially high, indicating that each 10% drop in baseflow would result in a 55% decline in these species. Seasonal baseflows (winter and summer) are particularly important for coldwater fish as they maintain critical thermal refuge for these temperature-sensitive species. They also help ensure the integrity of spawning habitats and maintain healthy nest conditions throughout the winter. In addition, macroinvertebrate communities, upon which many coldwater species depend, have been shown to be negatively affected by reductions in winter and summer baseflows (Wills et al. 2006, Dewson et al. 2007). The importance of median and low flows during fall and winter for cold headwater species is also well established (DePhilip and Moberg, 2010, 2013).

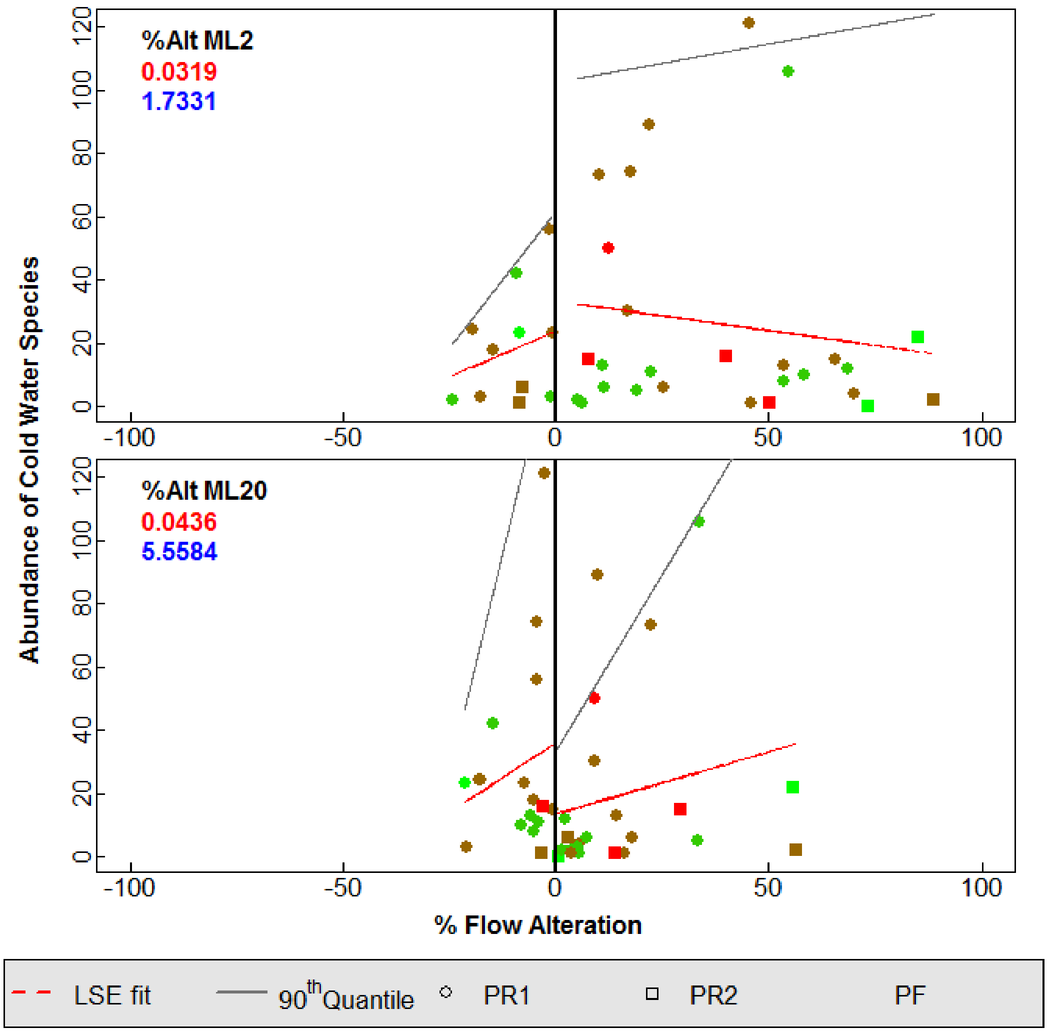

Figure 23. Plots of cold headwater species abundance vs. percent alteration in February low flow (ML2) and baseflow (ML20). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Brook Trout

We chose brook trout as a sentinel species for the cold headwater guild as they are a native, recreationally important and particularly temperature-sensitive species (Raleigh et al., 1986). We expected that brook trout would exhibit a similar; perhaps more pronounced relationship with low flow and baseflow – especially during winter and summer seasons. Interestingly, brook trout abundance was negatively correlated with summer flow variability (MA31, MA30), rise rate (RA6) and February high flows (MH2) (Figure 24). Meador and Carlisle (2012) provide some support for this finding in that they showed a loss of streamflow variability was associated with a 35% reduction in native fish species and over a 50% loss of riffle-dependent species. Also, it is possible that reduced flashiness may favor other fish species which, in turn, out-compete brook trout for food and habitat. Additionally, a loss of high flows may lead to increase sedimentation of redds which can limit recruitment (Alexander and Hansen, 1986, Argent and Flebbe, 1999).

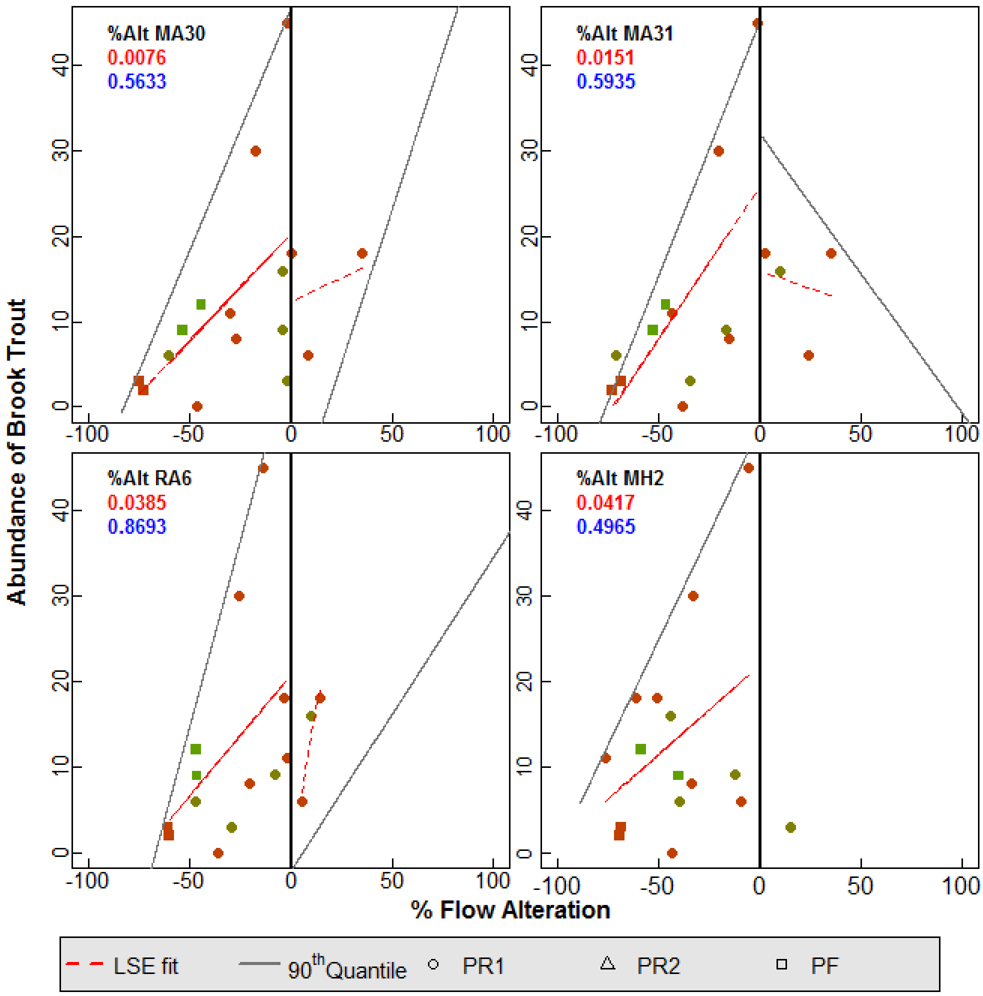

Figure 24. Plots of Brook trout abundance vs. percent alteration in July and August flow variability (MA30 and MA31, respectively), rise rate (RA6) and February high flow (MH2). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Riffle Obligate Guild

For this study, riffle obligates were comprised of margined madtoms (*Notorus insignis*), longnose dace (*Rhinichthys cataractae*), blacknose dace (*Rhinichthys atratulus*), central stoneroller (*Campostoma anomalum*), fantail darter (*Etheostoma flabellare*), rainbow darter (*Etheostoma caeruleum*), greenside darter (*Etheostoma blennioides*), tessellated darter (*Etheostoma olmstedi*), johnny darter (*Etheostoma nigrum*), banded darter (*Etheostoma zonale*), bluntnose minnow (*Pimephales notatus*), and cutlip minnow (*Exoglossum maxillingua*).

Unfortunately, this guild was not well represented in our flow-ecology dataset. This resulted in small sample sizes, reducing statistical inference. Even so, three HIs were significantly related to riffle obligate abundance. Specifically, riffle obligate abundance was negatively correlated with decreasing median August flows. The slope of the QR regression line was very high suggesting this guild was extremely sensitive to changes in August flow. However, this slope was undoubtedly influenced by potential outliers (Figure 25). Nonetheless, the strong association with August flow is in accordance with the life history traits of this guild. For instance, according to the DePhilip and Moberg (2010), reduced summer discharge (i.e. MA19) is particularly detrimental to riffle obligates who "specialize in highly oxygenated, lower riffle/plunge turbulent environments", because they are "sensitive to decreasing flow magnitude which would contract or eliminate this habitat niche". Among the various negative effects of artificially lowered August flows are impaired egg and larval development and lowered recruitment from reduced juvenile growth. Riffle obligate abundance was also reduced by declines in April and October high flows (MH4 and MH10, respectively). Substrate specialists such as riffle obligates require high flow events to maintain sandy substrates. Reductions in high flows may adversely affect habitat quality or abundance. Furthermore, riffle obligates need stable flows during spawning and egg and larval development – a significant decrease in high flows in the spring may reduce recruitment (i.e. many species are spring spawners and rely on high flows as spawning cues).

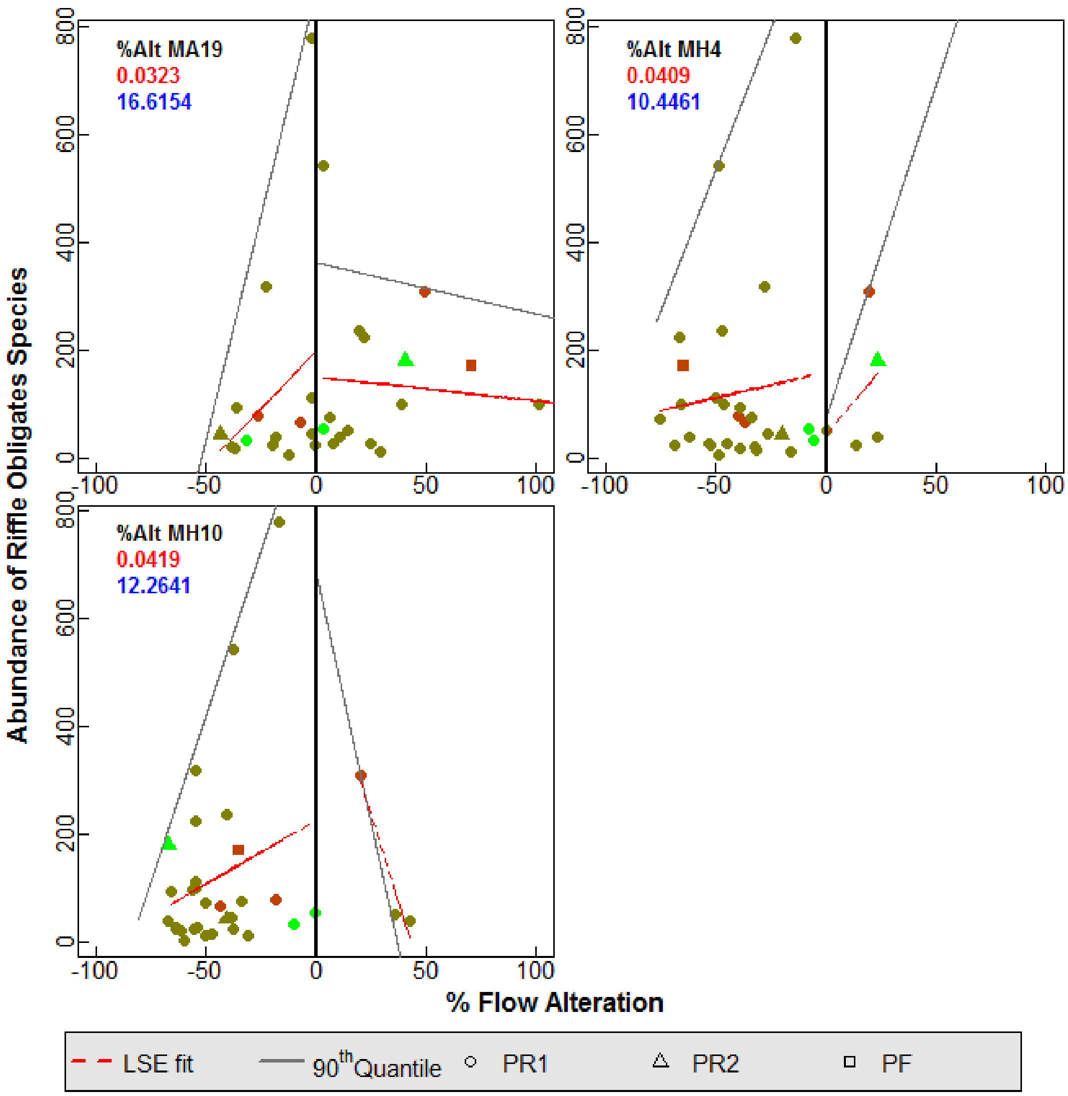

Figure 25. Plots of riffle obligate abundance vs. percent alteration in median August flow (MA19), April high flow (MH4) and October high flow (MH10). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Central Stoneroller

Being a riffle obligate species, central stonerollers prefer the shallow, fast-flowing environments associated with riffle habitats, although they will occupy deeper pools during low-flow periods (Power and Matthews, 1983). Although our sample size of central stonerollers was relatively small, we did detect significant relationships between their abundance and two HIs: duration of 90-day high flows (DH5), April high flows (MH4) (Figure 26). The small sample size suggests QR results should be interpreted with caution – however, the near-parallel association of the least squared error (LSE) fit and the 90th quantile regression lines indicates the negative correlation is consistent across numerous quantiles, boosting confidence in the QR results.

Reductions in high flows may lead to inadequate flushing of fine sediments, compromising the integrity of spawning substrates. Additionally, central stonerollers have been shown to be relatively intolerant to siltation as it negatively effects algal growth, their preferred food (DePhilip and Moberg, 2010, 2013). Alteration in MH4 likely reduces stoneroller abundance because it disrupts flows during the critical spawning period (i.e. reduces access to off-channel habitats and backwaters, thereby increasing predation risk (Gido et al., 2013).

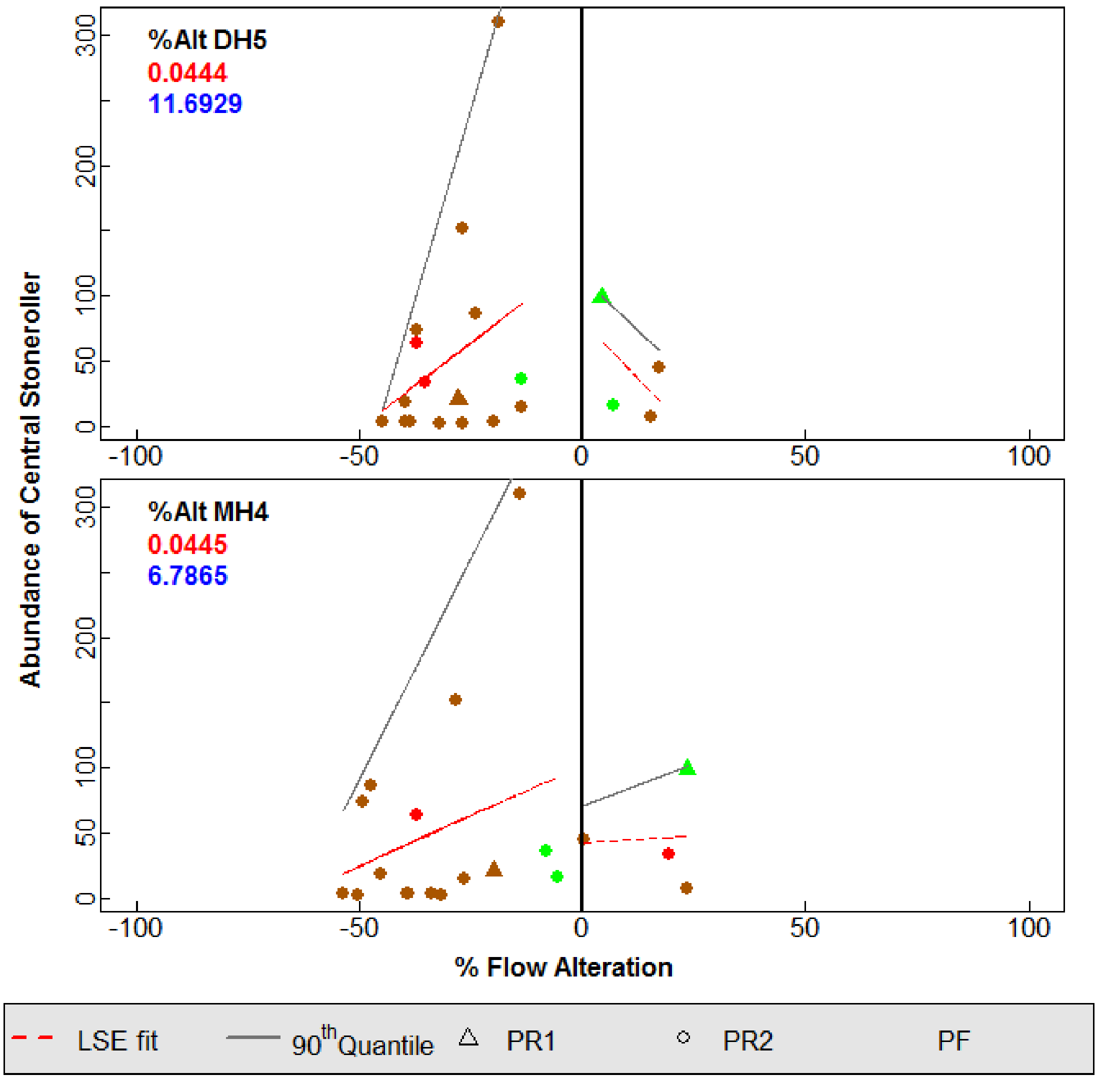

Figure 26. Plots of central stoneroller abundance vs. percent alteration in 90-day high flow (DH5) and April high flow (MH4). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Riffle Associates Guild

The riffle associate guild was comprised of *Catostomus* spp. (e.g. white sucker), *Moxostoma* spp. (e.g. shorthead redhorse, golden redhorse, silver redhorse), northern hogsucker (*Hypentelium nigricans*), walleye (*Sander vitreus*) and smallmouth buffalo (*Ictiobus bubalus*). These species are characterized by a moderate home range, typically migrate to spawn and "need access to, and connectivity between, riffle habitats" (DePhilip and Moberg, 2010). Based on life history traits and TNC flow-ecology hypotheses (Appendix B), we expected riffle associates to respond primarily to spring and summer HIs as these reflect flow requirements for migration, spawning and adult and juvenile growth. Our data did not support this theory (Figure 27). However, the significant negative trend between reductions in annual runoff (MA41) and abundance of riffle associates lends credence to the hypothesis that the overall maintenance of flow is important to maintain connectivity and quality of spawning habitat. The significance of October high-flows (MH10, suggests high flows in fall may also play an important role in the health and integrity of riffle associates by maintaining the flushing of fine sediments from spawning gravels.

The slopes of all significant relationships were quite high relative to other life history guilds indicating that riffle associates are more sensitive to flow alteration than many other fish. This is likely related to their reliance on riffle habitats which are disproportionately negatively affected by flow reductions and changes in variability.

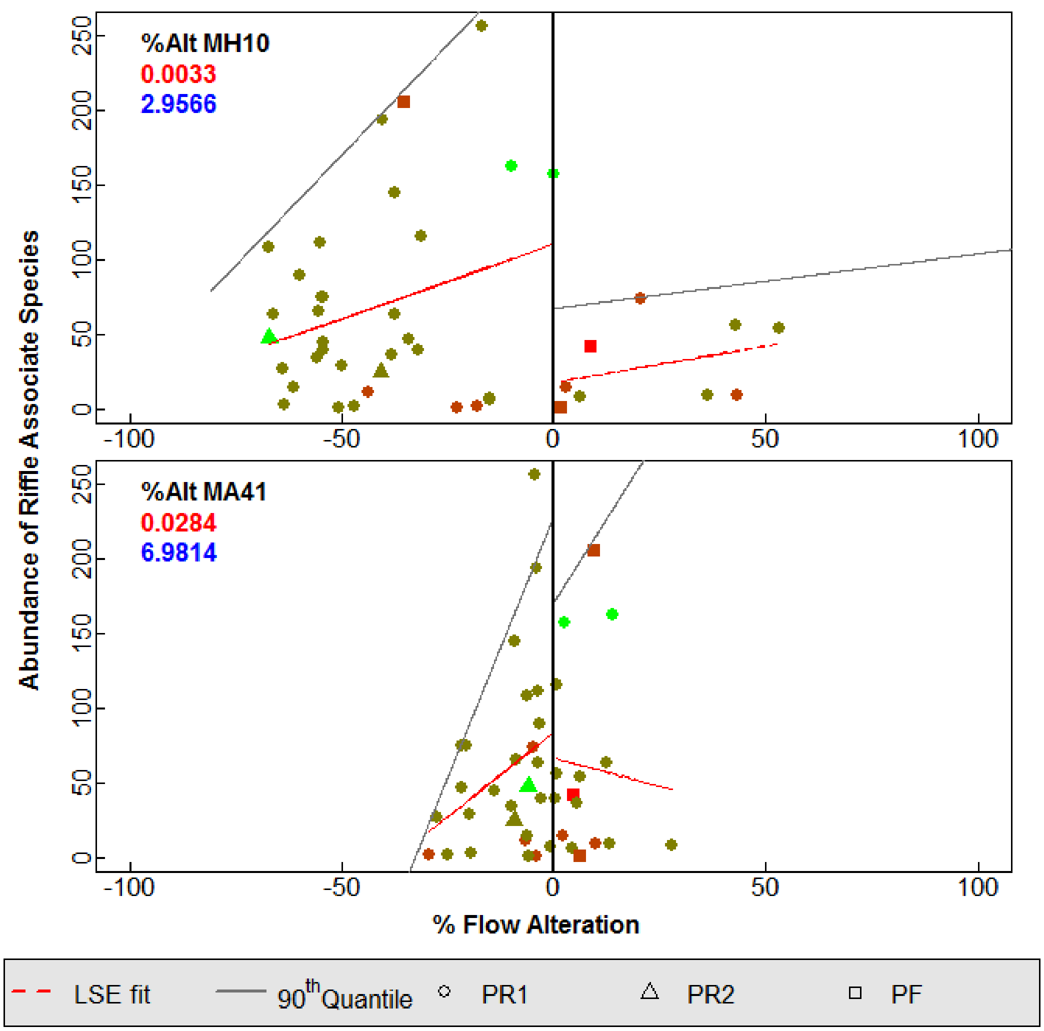

Figure 27. Plots of riffle associates abundance vs. percent alteration in October high flow (MH10) and annual runoff (MA41). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Northern Hog Sucker Abundance

Interestingly, Northern hog suckers seemed to respond differently to flow alteration than their guild as a whole. In particular, their abundance declined with decreases in August flow variability (MA31) and rise rate (RA6) (Figure 28). This may reflect the deleterious effects of enhanced interspecific competition from other species that are able to invade and out-compete hog suckers when flashier flow regimes are stabilized. Indeed, Meador and Carlisle (2012) showed that reduced streamflow variability was related to a 35% loss in native fish species, on average, and a >50% loss of species with a preference for riffle habitats.

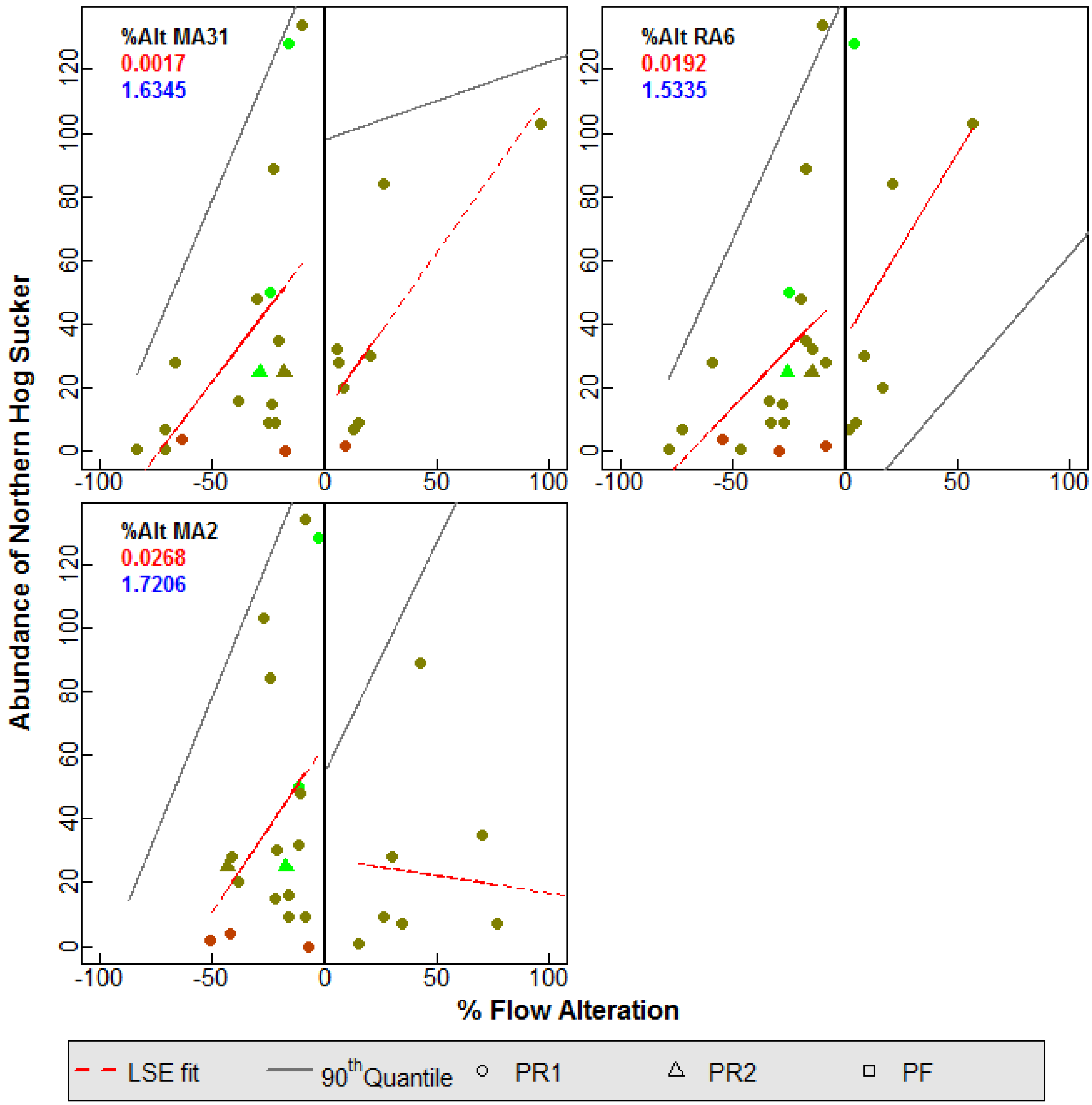

Figure 28. Plots of northern hog sucker abundance vs. percent alteration in August flow variability (MA31), rise rate (RA6) and February flow (MA2). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Nest Builders Guild

The nest building fish guild included smallmouth bass (*Micropterus dolomieu*), fallfish (*Semotilus corporalis*), *Nocomis* spp. (e.g. creek chub, river chub), redbreast sunfish (*Lepomis auritus*), rock bass (*Ambloplites rupestris*) and spotted bass (*Micropterus punctulatus*). These species are spring spawners, typically constructing nests on sand, gravel, or rocky ledges along channel margins (DePhilip and Moberg, 2010). Thus, they share similar flow requirements during nest building, spawning, and egg and larval development. For instance, if discharge is too low for nest builders, "siltation may occur or nests may be dewatered, desiccating eggs and stranding larvae" (DePhilip and Moberg, 2010). Additionally, nest degradation may have negative implications for numerous other minnow species which either co-inhabit or take over abandoned *Nocomis* nests to spawn. The summer months are also important to nest builders as juvenile growth occurs predominantly during this season.

Accordingly, TNC flow hypotheses (Appendix B) suggest nest builders are sensitive to flow alteration during the spring and summer nest building seasons as they require suitable flows to maintain coarse substrate for nest building. Moreover, nest builders may be affected by changes in the frequency and magnitude of high flow events. Our QR results support these hypotheses in that they demonstrated strong negative correlations with a number of low, median and high flow magnitudes (Figure 29). Median and high April flow (MA15 and MH4, respectively), for example, indicated highly significant trends at the 90th quantile, with relatively steep slopes. Specifically, for every 10% decrease in April high flow, the QR results suggest abundance of nest builders will decrease by 19 individuals. The lower nest builder abundance with decreasing annual flows (MA41) also points to sensitivity to reduced habitat volume and nest degradation. Additionally, sensitivity to February and October high flows (MH2 and MH10, respectively) suggests that nest builders require adequate high flows to prevent siltation and maintain coarse substrate for nest building (DePhilip and Moberg, 2010, 2013).

Many nest builders exhibit higher parental care as they will often guard their nests. We observed a declined in nest builders with reduced August flow variability (MA31), implying that they prefer less stable flow environments. This is consonant with McManamay and Frimpong (2015), who noted a positive correlation between daily flow variation and nest-guarding fish.

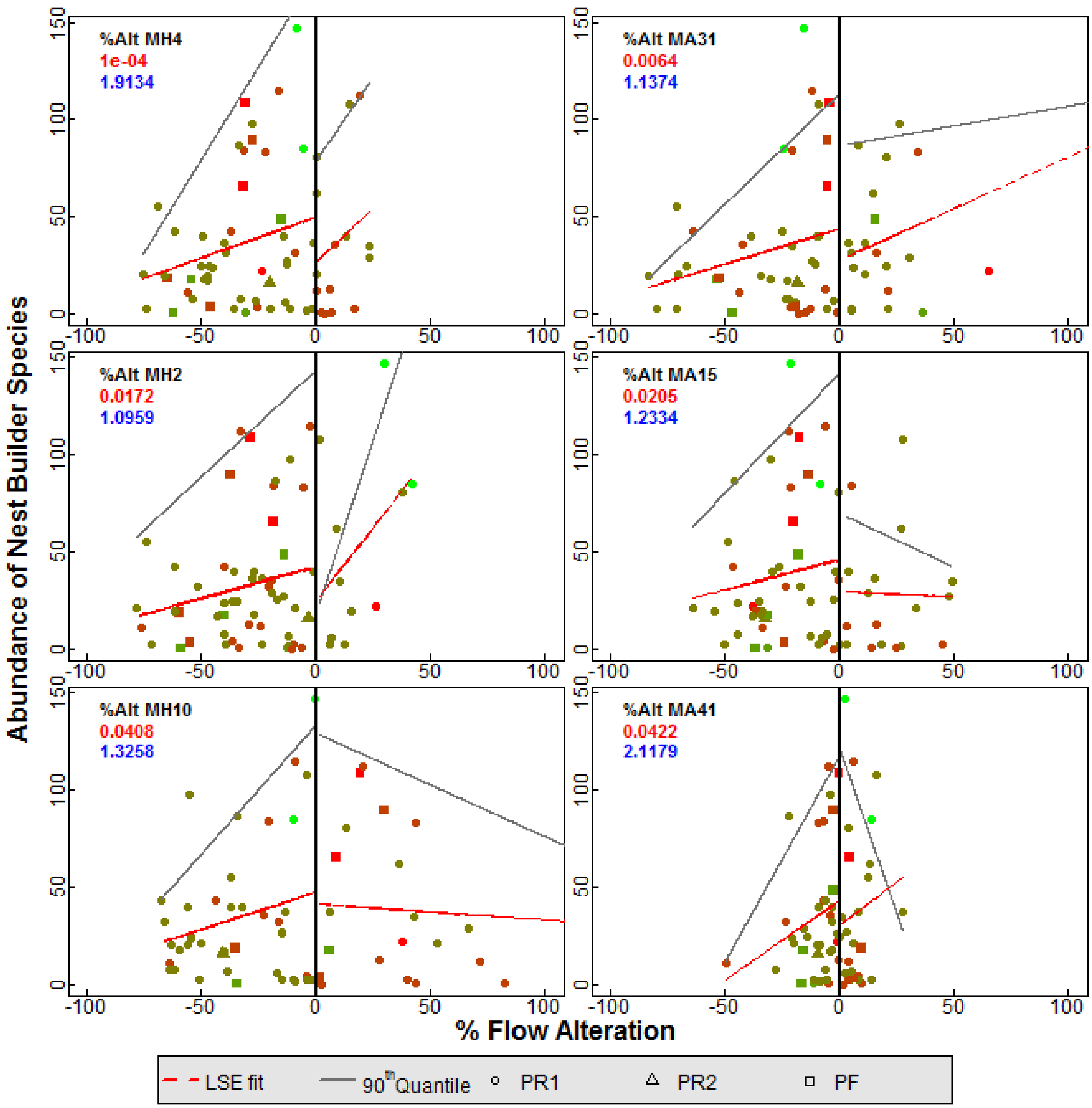

Figure 29. Plots of nest builder abundance vs. percent alteration in April median and high flows (MA4 and MH15, respectively), August flow variability (MA31), February high flow (MH2), October high flow (MH10) and annual runoff (MA41). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

Small Mouth Bass

Smallmouth bass were chosen as a suitable sentinel species for the nest building guild, due to the fact that they were well represented in the MSR and are recreationally important. Perhaps unsurprisingly, smallmouth bass possessed similar flow requirements to their fellow guild members in that they responded to decreases in October high-flow events (MH10) (Figure 30). However, they were less sensitive overall as indicated by shallower regression slopes. Smallmouth bass also showed considerable dependence on the maintenance of 30-day high flow

(DH4). The maintenance of high flow volumes likely indicates a need to maintain channel margin spawning habitat.

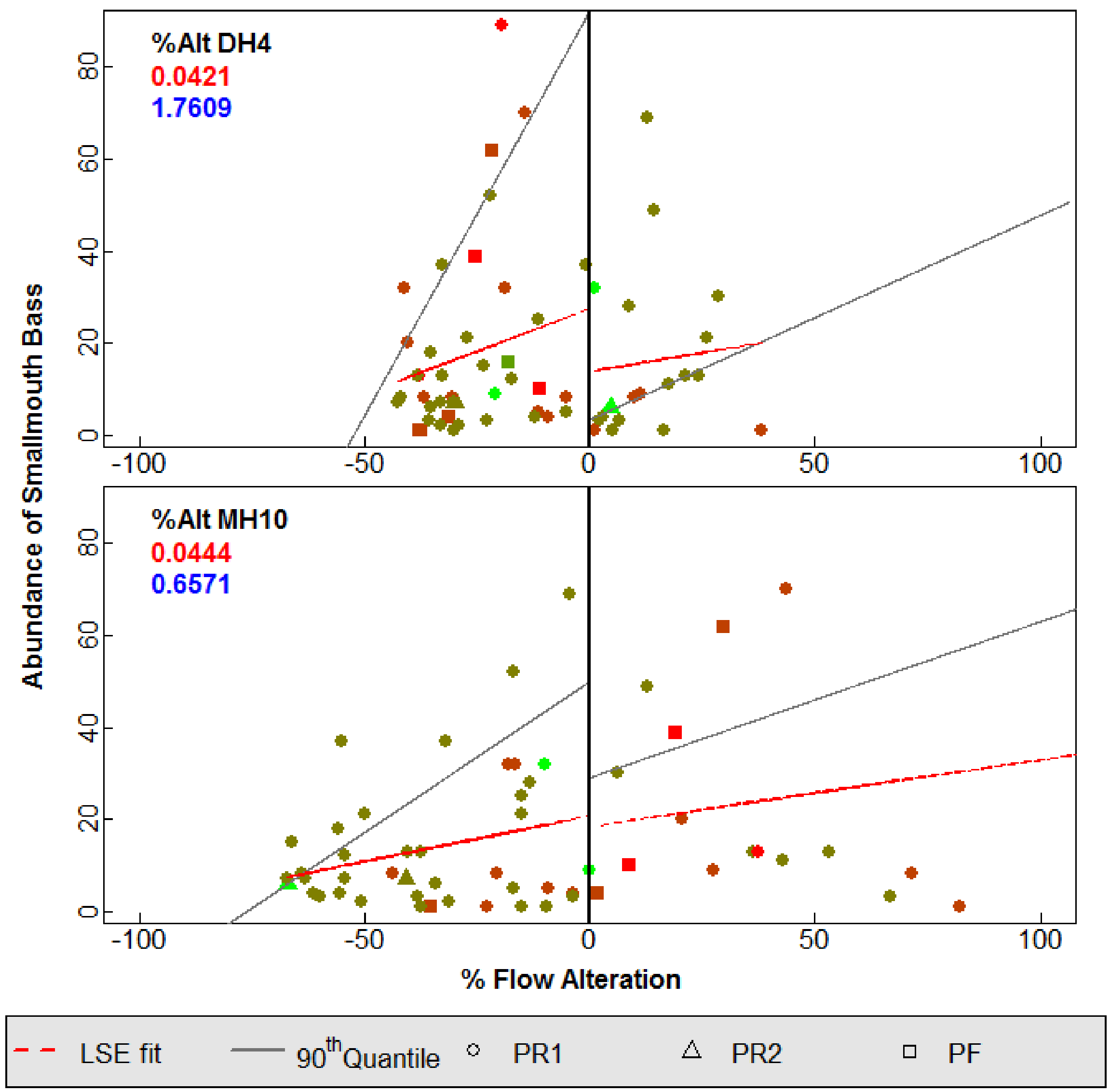

Figure 30. Plots of smallmouth bass abundance vs. percent alteration in 30-day high flow (DH4) and October high flow (MH10). P-values (red text) and slopes (blue text) associated with the 90th quantile regression line are indicated.

With the exception of percent tolerant species and two life history groups, all ecological metrics were significantly related to alteration in at least one hydrologic index. Indeed, in most cases, measures of fish assemblage health and integrity were related to multiple HIs (Table 10). F-E relationships were most commonly driven by seasonal (monthly) flow indices, particularly summer HIs. Timing-related HIs (i.e. TA1 and TA2) were not significantly related to any ecological metrics – though we did note several trends with p-values between 0.05 and 0.1 (data not shown). With few exceptions, all F-E relationships were negative suggesting that deflated

HIs consistently resulted in reduced ecological metrics. The majority of QR slopes for proportional abundance F-E curves were between 1-1.5, indicating that 10% reductions in most flow indices roughly translate to 10-15% reductions in relative abundance metrics.

Table 10. Hydrologic indices that demonstrated a significant relationship ($\alpha = 0.05$) with various ecological metrics. HIs in bold font represent positive slopes with increasingly negative HIs, while non-bold font represents negative slopes.

Ecological Metric	Seasonal HIs				Annual HIs		
	Spring	Summer	Fall	Winter	Rate of Change	Magnitude	Duration
Species Richness		MA19			**RA6,** RA7	MA41	
Abundance	MH4	MA19					
% Intolerant		MA30, MA31		MH2	RA7		
% Piscivore	MA15	**MA19,** MA30, MA31					
% Invertivore				MH2			
% Generalist			MA21				
% Herbivores					RA7	MA41	**DL4**
% Periodic		MA30, MA31	MH10				
% Opportunistic				ML2			
# Cold Headwater				ML2		ML20	
Brook Trout		MA30, MA31		MH2	RA6		
# Nest Builders	MA15, MH4	MA31	MH10	MH2		MA41	
Smallmouth Bass			MH10				DH4
# Riffle Obligates	MH4	MA19	MH10				
# Central Stoneroller	MH4						DH5
# Riffle Associates			MH10			MA41	
# Northern Hog Sucker		MA31		MA2	RA6		

We expected low flow HIs, principally during dry seasons, to dominate the F-E curves. However, significant seasonal HIs covered a range of low-; median- and high-flows during all four seasons (Table 10). This suggests fish communities in the MSR have diverse flow requirements, which is consistent with the flow-ecology hypotheses of DePhilip and Moberg (2010, 2013; Appendix B). Although not as common, several annual HIs, regarding flow magnitude, rate of change and flow duration were also significant. The unexpected lack of significant low flow F-E relationships is likely attributable to the fact that anthropogenic flow disturbance in the MSR has primarily increased low flows (Figure 7). As the variable importance plots in Appendix D suggest, the most influential anthropogenic explanatory variable in predicting hydrologic sensitivity across most HIs was the number of dams. Dam operations often result in attenuation of peak and low flows, which explains the pattern of decreased high flows vs. increased low flows in Figure 7. The overall inflation of low flow HIs greatly reduced the sample size of fish sampling sites paired with negative low flow alteration, contributing to the lack of significant low-flow F-E relationships. This is a clear limitation of this study. However, monthly medians were reasonably well correlated with low flows (data not shown); suggesting it is reasonable to conclude that F-E relationships based on median flows would translate, at least qualitatively, to low flow metrics.

The majority of F-E relationships were consistent with ecological theory and our flow-ecology hypotheses. In a few instances, such as the proportion of trophic generalists, the F-E curves were contrary to our expectations or were not well supported by the published literature. This could be reflective of flow-ecology interactions unique to the MSR or perhaps, to errors related to low sample sizes. It could also be related to uncertainty in the estimates of both flow alteration and fish metrics. Lotic ecosystems are generally complex, possessing a high degree of structural uncertainty (Williams et al., 1996). Moreover, it is often challenging to accurately quantify fish metrics such as species richness or abundance due to "partial observability". Although, measurement error of biological metrics is often <20% (Wright et al., 2000; Ostermiller and Hawkins, 2004, Van Sickle et al., 2007), this degree of uncertainty can make it difficult to accurately predict the effects of flow alteration.

Furthermore, errors in observed discharge and geospatial data used to construct RF models can lead to inaccuracies in model predictions. In this study, the majority of RF models explained > 90% of variance in HIs. This implies only flow alterations that exceed 10% would be reliably predicted with our RF models. This may further compound uncertainty in F-E curves, rendering interpretation and application of results problematic.

Overall, the QR analysis indicates that flow alteration in the MSR results in a suite of adverse ecological impacts, including: (i) reduced species richness, total abundance and relative abundance of intolerant species, (ii) changes in trophic structure and life history strategies and (iii) declines in certain functional guilds and sentinel species. The proportional abundance of nest builders appear to be a particularly sensitive metric in the MSR as it was related to 6 different HIs. Other sensitive ecological metrics (significant F-E relationships are ≥ 4) include species richness, proportion of intolerant species, proportion of piscivores, and abundance of brook trout.

The question remains: will water extraction from gas development activities result in the exceedance of flow alteration thresholds thereby causing significant ecological impairment? Our consumptive water use analysis in the following section elucidates this question.

Pumping Scenarios

The pumping analysis explores the following three research questions: i) what is the relationship between stream size and sensitivity to realistic surface water pumping associated with hydraulic fracturing, ii) which hydrologic indices and stream classes are most sensitive to surface water pumping and iii) can hydrologic sensitivity be reliably predicted across the Marcellus Shale Region? A total of 138 of 195 reference stations in the Marcellus Region were included in the pumping analysis. As previously stated, we applied a constant daily withdrawal across the entire period of record in order to establish long-term annual, seasonal and monthly-average effects of pumping. Accordingly, this analysis was only used to evaluate pumping effects on HIs pertaining to magnitude, duration and rate of change flow regime components (i.e. timing and frequency HIs will experience little change).

Relationship between Stream Size and Hydrologic Sensitivity

Figure 31 depicts the relationship between drainage area and magnitude-related HSIs constructed by taking the mean of the percent change in a hydrologic index from the natural baseline due to the cumulative high pumping scenario. Cumulative high pumping rates represent a worst-case-scenario and should therefore provide conservative estimates. HSIs were calculated over all seasons for low, median and high flow indices (HSI_ML, HSI_MA, HSI_MA, respectively). It is clear that as drainage area increases, magnitude-related HSIs decline exponentially. Moreover, low-flows have the highest sensitivity to pumping, followed by average and high flows. This is to be expected as surface water withdrawals will disproportionately affect low-flow magnitudes. Figure 31 clearly demonstrates a strong seasonal pattern in sensitivity; summer is the most sensitive, followed by fall, winter and spring. Moreover, low flow HSIs (Figure 31B) are considerably more variable than median or high flows HSIs (Figure 31C and D, respectively). Overall, this suggests that, concerning monthly magnitude HIs, low-flow HIs during the summer and fall are the most responsive to surface water withdrawal. Additionally, it is evident that under all scenarios, a threshold in drainage area can be seen at roughly 1000 km^2 – after which pumping has minimal effects.

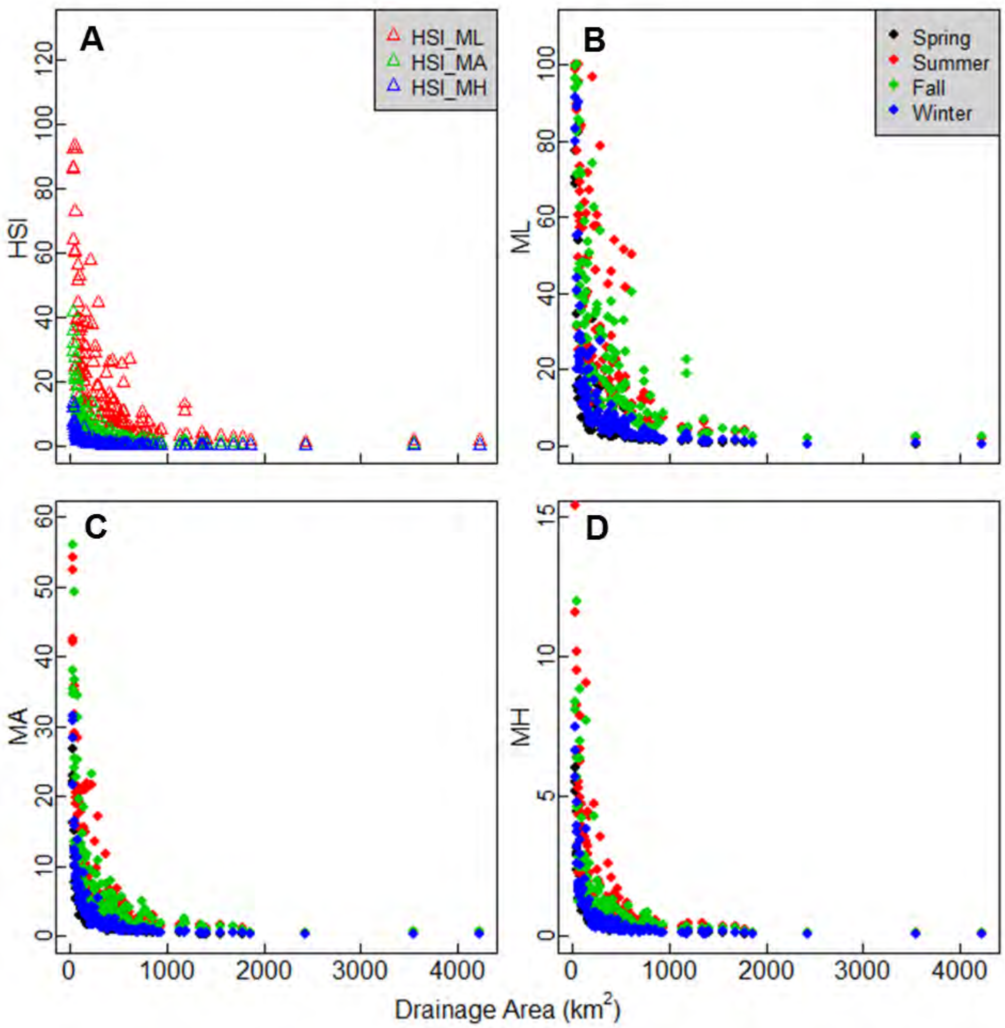

Figure 31. Relationship between drainage area and magnitude-related HSIs constructed by taking the mean of the percent change in a hydrologic index from the natural baseline due to cumulative high pumping. Plot (A) represents HSIs for all seasons and plots (B-D) represent low-, median and high-flows for all for seasons.

Figure 32 illustrates how the sensitivity of median August flow (MA19), a commonly used flow index, is affected by consumptive water extraction under a variety of pumping scenarios. The percent change in MA19 increases with increasing abstraction rates (i.e. low local to high cumulative pumping). Similar to Figure 31 above, a drainage area threshold is evident at approximately 1,000 km^2 – after which pumping has minimal effects. This finding supports our decision to limit the ELOHA application in the context of hydraulic fracturing on the basis of drainage area, as well as confirms the intuitive notion that water extraction will have a disproportionate effect on smaller streams. The strong influence catchment area exerts on sensitivity to surface water pumping also suggests that it should be feasible to accurately predict pumping sensitivity across the landscape and that catchment area should prove an important predictor.

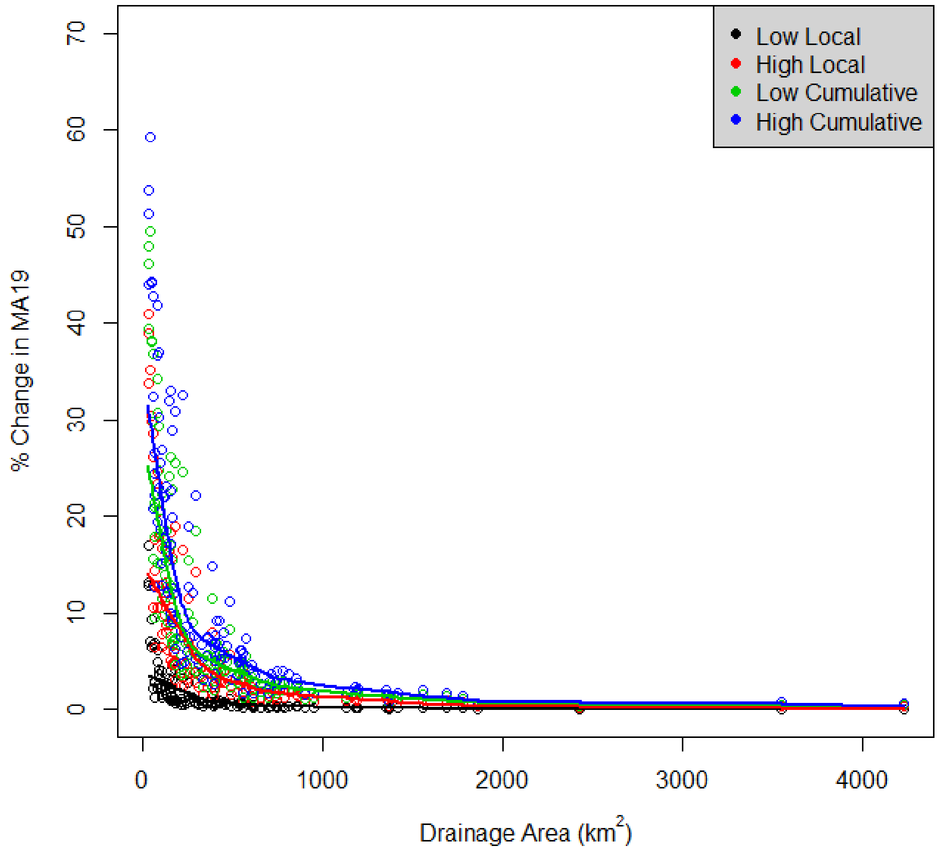

Figure 32. Percent change in mean August flow as a function of basin drainage area for the low/high local and cumulative pumping scenarios. Curvilinear lines represent locally weighted regression (LOWESS) curves fit to the data to guide the eye.

Sensitivity of hydrologic indices to surface water pumping

The pumping analysis also helped to determine which HIs are most sensitive to surface water pumping associated with shale gas development. Table 11 lists all HIs that were predicted via RF models with R-squared values ≥ 0.8, ranked according to the median percent change across all USGS reference gages as a result of the high local and high cumulative pumping scenarios. Please refer to Henriksen et al. (2006) for a description of each index. A full list of all HIT indices ranked according withdrawal sensitivity is provided in Appendix C. In general, low flow HIs were the most sensitive to pumping – especially 1-, 3- and 7-day low flow durations and seasonal low flows occurring during the summer and fall. On the other hand, high flow duration HIs, as well as high flows during the winter and spring months were least sensitive. These results confirm the idea that low flow hydrologic indices, particularly during low flow periods, are most responsive to hydraulic fracturing-related water withdrawals; and further, that these indices would be good candidates for future monitoring programs – especially those designed to detect the long-term impacts of local and cumulative surface water pumping.

Table 11. Median percent difference between natural and pumped scenarios for all HIs with an OOB error rate less than 20%.

HI	Description	Local High	Cumulative High
DL1	1-day low flow	26.06	30.82
DL2	3-day low flow	24.645	29.24
DL3	7-day low flow	21.815	25.655
ML9	Median September low flow	15.235	18.035
ML8	Median August low flow	13.365	15.85
DL4	30-day low Flow	11.99	14.22
ML7	Median July low flow	11.28	13.4
ML10	Median October low flow	9.25	10.97
ML6	Median June low flow	6.635	7.875
ML11	Median November low flow	5.43	6.45
TA1	Constancy	4.88	6.25
DL5	90-day low flow	4.725	5.615
MA19	August flow	3.72	4.42
ML12	Median December low flow	3.59	4.265
MA20	September flow	3.47	4.105
ML1	Median January low flow	3.43	4.075
TA2	Predictability	3.32	3.825
ML5	Median May low flow	3.285	3.91
ML20	Base flow	2.94	3.495
MA18	July flow	2.935	3.49
MA2	Median of daily mean flows	2.765	3.28
MA21	October flow	2.74	3.26
ML3	Median March low flow	2.21	2.625
ML4	Median April low flow	2.15	2.55
ML20	Base flow	2.13	2.245
MA17	June flow	1.845	2.19
MA22	November flow	1.52	1.8
MA1	Mean of daily flows	1.235	1.465
MA41	Annual runoff	1.225	1.405
MA16	May flow	1.035	1.23
MA23	December flow	1.035	1.235
MA12	Jan flow	0.965	1.15
MA13	February flow	0.855	1.02
MH8	August high flow	0.75	0.89
MA15	April flow	0.705	0.84
MA14	March flow	0.63	0.75
MH9	September high flow	0.63	0.745
DH5	90-day high flow	0.605	0.72
MH7	July high flow	0.56	0.665

MH10	October high flow	0.545	0.645
DH4	30-day high flow	0.41	0.48
MH6	June high flow	0.365	0.43
MH11	November high flow	0.335	0.395
MH5	May high flow	0.275	0.325
MH12	December high flow	0.25	0.3
MH2	February high flow	0.215	0.255
MH1	January high flow	0.21	0.25
DH3	7-day high flow	0.205	0.245
MH4	April high flow	0.17	0.2
MH3	March high flow	0.15	0.18
DH2	3-day high flow	0.13	0.15
DH1	1-day high flow	0.08	0.09

Sensitivity of stream classes to surface water pumping

Regarding which stream classes were more responsive to surface water abstraction, it is clear that of the three predominant classes, perennial flashy stream types (PF) were the most sensitive to withdrawals (Table 12). The least sensitive was perennial runoff 1 (PR1), followed by perennial runoff 2 (PR2). This is consistent with the characteristics of each class. For example, PR1 possesses the highest baseflows and least variability, rendering it the most resilient to surface water pumping. PR2 is characterized by more flow variability and lower baseflows, translating to an intermediate level of sensitivity. Finally, PF stream types have lower, sometimes intermittent flows, lower baseflows, and higher rise/fall rates. Naturally then, these flashier systems would be the least resilient to flow perturbation. As such, they should be considered higher priority management concerns. The stable high baseflow class was omitted from this analysis because it comprised a very small portion of the MSR.

Table 12. Mean HSIs for the high local, high cumulative and average pumping scenarios for each stream class.

Stream Class	Local High	Cumulative High	Average
Perennial Runoff 1	15.3	25.0	20.1
Perennial Runoff 2	16.9	27.8	22.5
Perennial Flashy	18.2	28.8	23.5

Risk Analysis

In general, the RF models for predicting hydrologic sensitivity to surface water withdrawals across the Marcellus achieved acceptable accuracy (i.e. most R^2 values were > 0.65; Table 13). HSIs were predicted for mean low, average and high flows, as well as for annual runoff and median annual flow. In general, average and low flow sensitivities were better predicted than high flows. Moreover, the high cumulative scenario was better predicted than either low- or high-local pumping scenarios.

Table 13. Variance explained by each RF model for mean low (ML), mean annual (MA), and mean high (MH) flows across three pumping scenarios.

Metric	Pumping Scenario		
	Low Local	High Local	High Cumulative
ML2	0.71	0.79	0.82
ML4	0.68	0.76	0.8
ML7	0.75	0.81	0.83
ML8	0.76	0.81	0.83
ML10	0.74	0.81	0.84
MA13	0.69	0.77	0.8
MA15	0.64	0.73	0.76
MA18	0.74	0.8	0.82
MA19	0.75	0.81	0.83
MA21	0.72	0.79	0.82
MH2	0.59	0.68	0.73
MH4	0.57	0.66	0.69
MH7	0.61	0.71	0.75
MH8	0.62	0.71	0.75
MH10	0.59	0.69	0.73
MA2	0.69	0.77	0.8
MA41	0.62	0.75	0.79

Regarding the relative importance of predictor variables, drainage area, average precipitation, land cover type and number of dams were consistently the most influential explanatory variables across scenarios (Appendix D). It was further noted that for lower and lower flows during dry seasons, more basin attributes (e.g. baseflow index, average available water content, mean elevation) played a larger predictive role. Additionally, mean basin precipitation and elevation were much more influential in predicting low-flow sensitivity (Figure 33A), whereas high-flow sensitivities were more influenced by land cover characteristics and mean basin slope (Figure 33B).

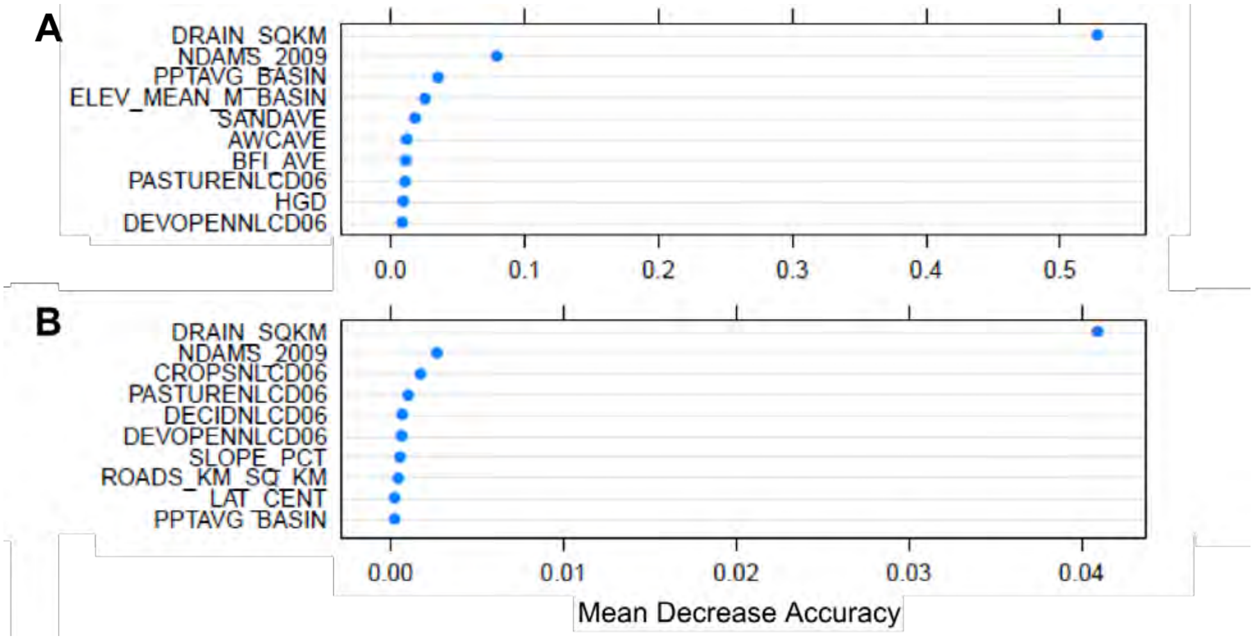

Figure 33. Top ten unbiased variable importance scores for the low-flow (A) and high-flow (B) sensitivity indices.

Predicted hydrologic sensitivities to surface water withdrawals were highest during summer for magnitude-related HIs; followed by the fall, winter and spring seasons (Figures 34-36). Low flows were altered considerably more than median or high flows (Figures 34-36). For example, the maximum predicted alteration in low flows was > 50%, whereas the maximum predicted alteration in high flows was almost an order of magnitude lower (< 6%, Figure 34-36). In general, smaller catchments with lower annual rainfall, smaller baseflow indices, more dams and higher percentages of pasture and developed land were associated with higher HSI values.

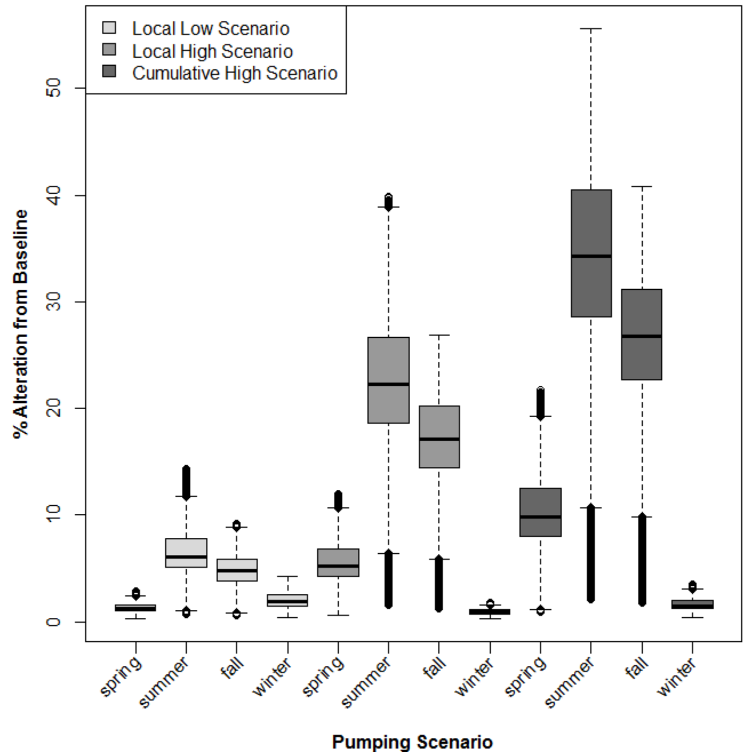

Figure 34. Boxplots of percent alteration from natural baseline in seasonal <u>low-flows</u> over three pumping scenarios.

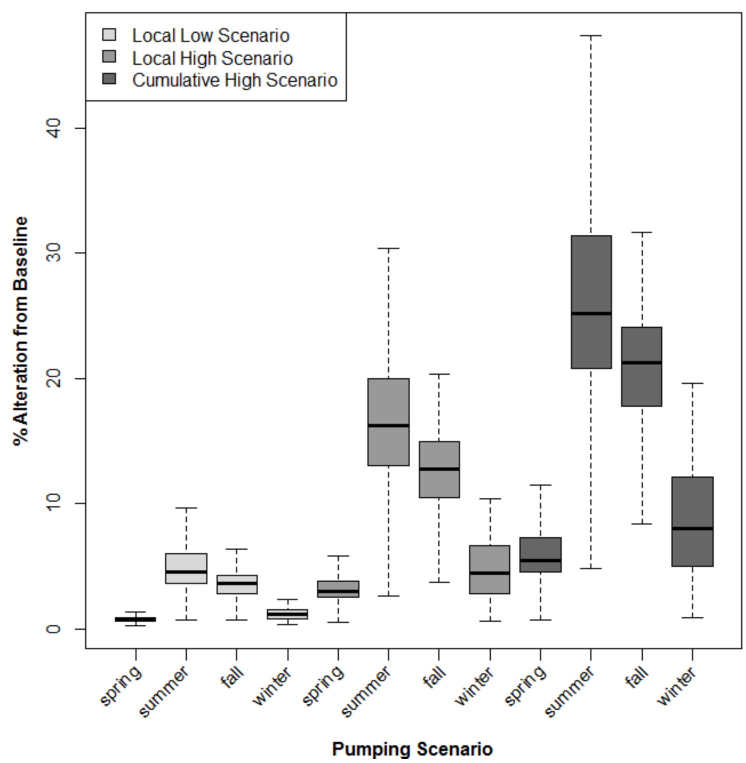

Figure 35. Boxplots of percent alteration from natural baseline in seasonal <u>median-flows</u> over three pumping scenarios.

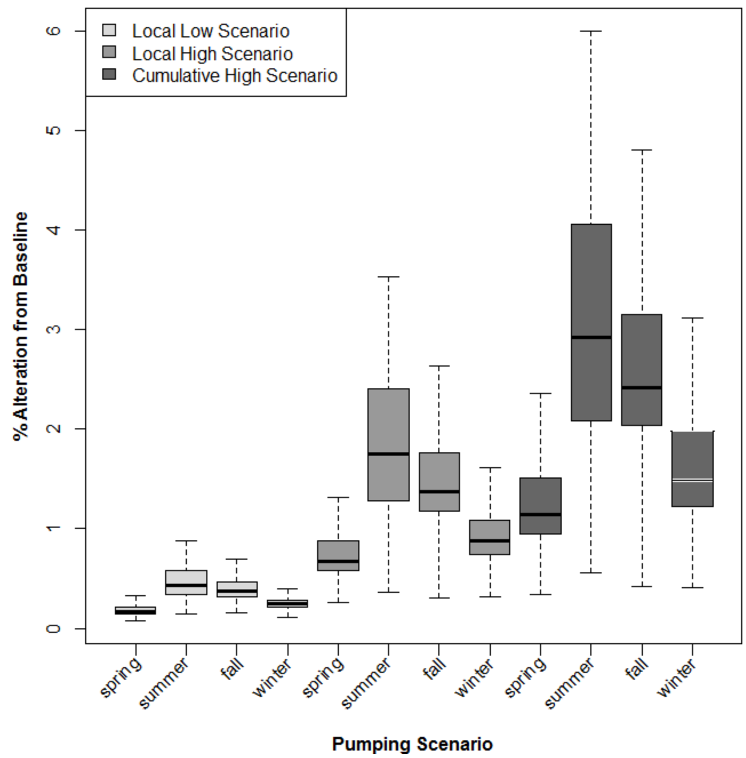

Figure 36. Boxplots of percent alteration from natural baseline in seasonal high-flows over three pumping scenarios.

Mapping the predicted HSIs across all NHD streams in the MSR revealed interesting spatiotemporal patterns (Figures 37-39 and Appendix E). For instance, the majority of streams that are sensitive to water extraction during the summer season are lower order systems located primarily in two areas within the MSR: (i) a southwestern zone (Western Allegheny Plateau and Erie Drift Plain level III ecoregions located in the Upper Ohio River, Muskingum and Southern Lake Erie basins) and (ii) in a northern band (Northern Allegheny Plateau ecoregion located in the Upper Susquehanna River Basin and tributaries of the Upper Hudson River Basin). Streams at lower risk are generally located in the central MSR (Central and North Central Appalachian ecoregions located in the West Branch of the Susquehanna and Allegheny River Basins) and along the eastern border (Ridge and Valley ecoregion located in the Potomac River Basin). Additionally, high risk stream reaches were generally characterized by the following biophysical and anthropogenic attributes: i) smaller drainage areas (headwaters and creeks), ii) fewer dams and lower overall dam storage, iii) lower average depth to seasonal water table, iv) lower elevation basins with lower average temperatures and flatter slopes, v) higher percentages of pasture and crop landuses, vi) lower percentages of evergreen and mixed forest and vii) higher percentages of poorly drained soils. Below, we provide risk maps associated with the local low pumping scenario. Risk maps for the remaining pumping scenarios are provided in Appendix E.

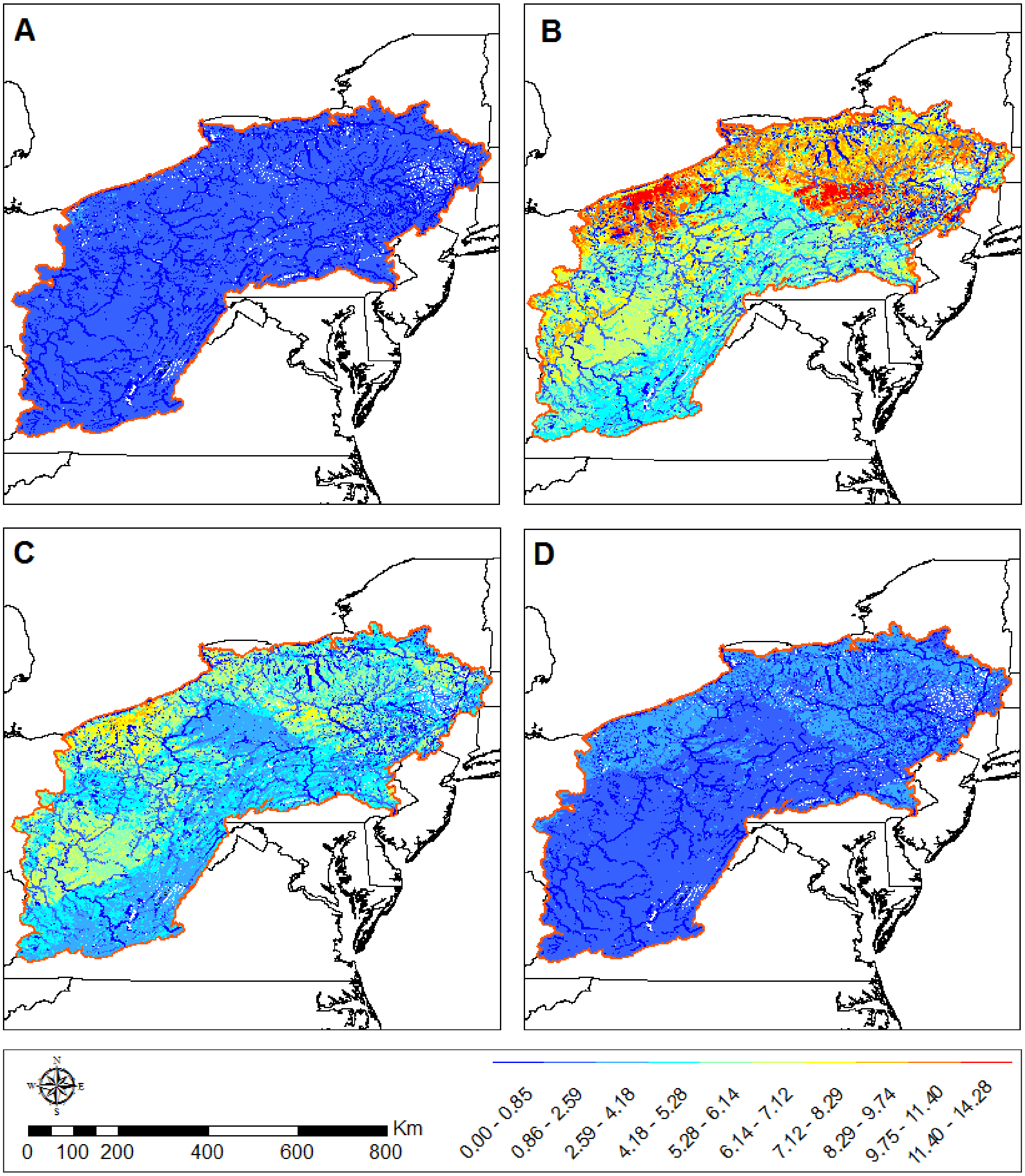

Figure 37. Maps of hydrologic risk to low-flows from the local low pumping scenario during spring (A), summer (B), fall (C) and winter (D). Note the legend scale relative to other pumping risk figures.

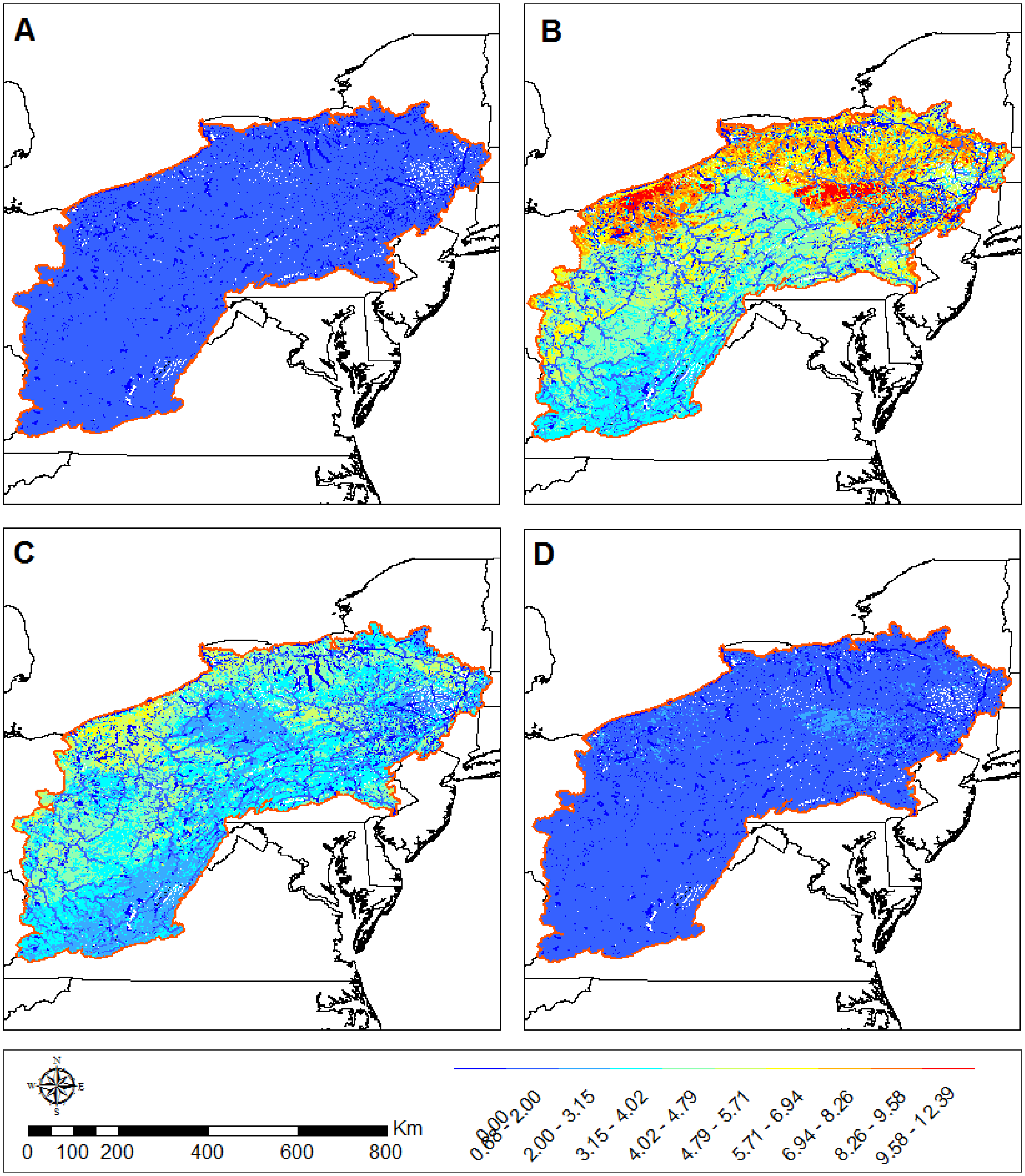

Figure 38. Maps of hydrologic risk to median-flows from the local low pumping scenario during spring (A), summer (B), fall (C) and winter (D). Note the legend scale relative to other pumping risk figures.

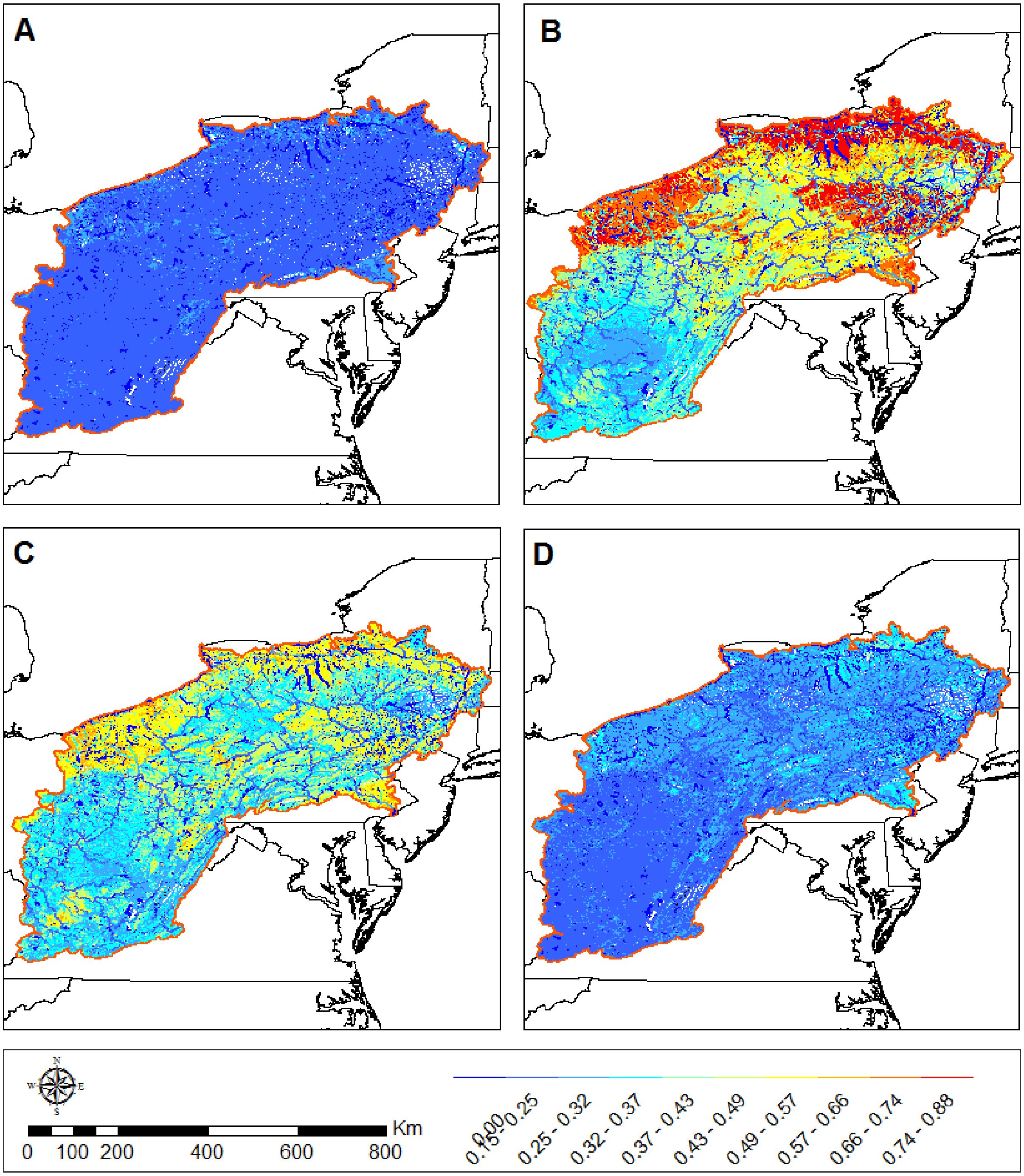

Figure 39. Maps of hydrologic risk to high-flows from the local low pumping scenario during spring (A), summer (B), fall (C) and winter (D). Note the legend scale relative to other pumping risk figures.

Combining our F-E relationships with predictions of hydrologic alteration allows us to visualize how ecological responses to hydrologic alteration vary spatially across the MSR. For example, applying the slope of the 90th QR for nest builders to projected changes in annual runoff (MA41) due to withdrawals from the local low, local high and cumulative high pumping scenarios highlights streams likely to experience larger shifts in nest builder abundance at increasing levels of extraction. According to Figure 40, some catchments in the MSR are predicted to experience substantial reductions (>20%) in nest builder abundance under the cumulative high scenario.

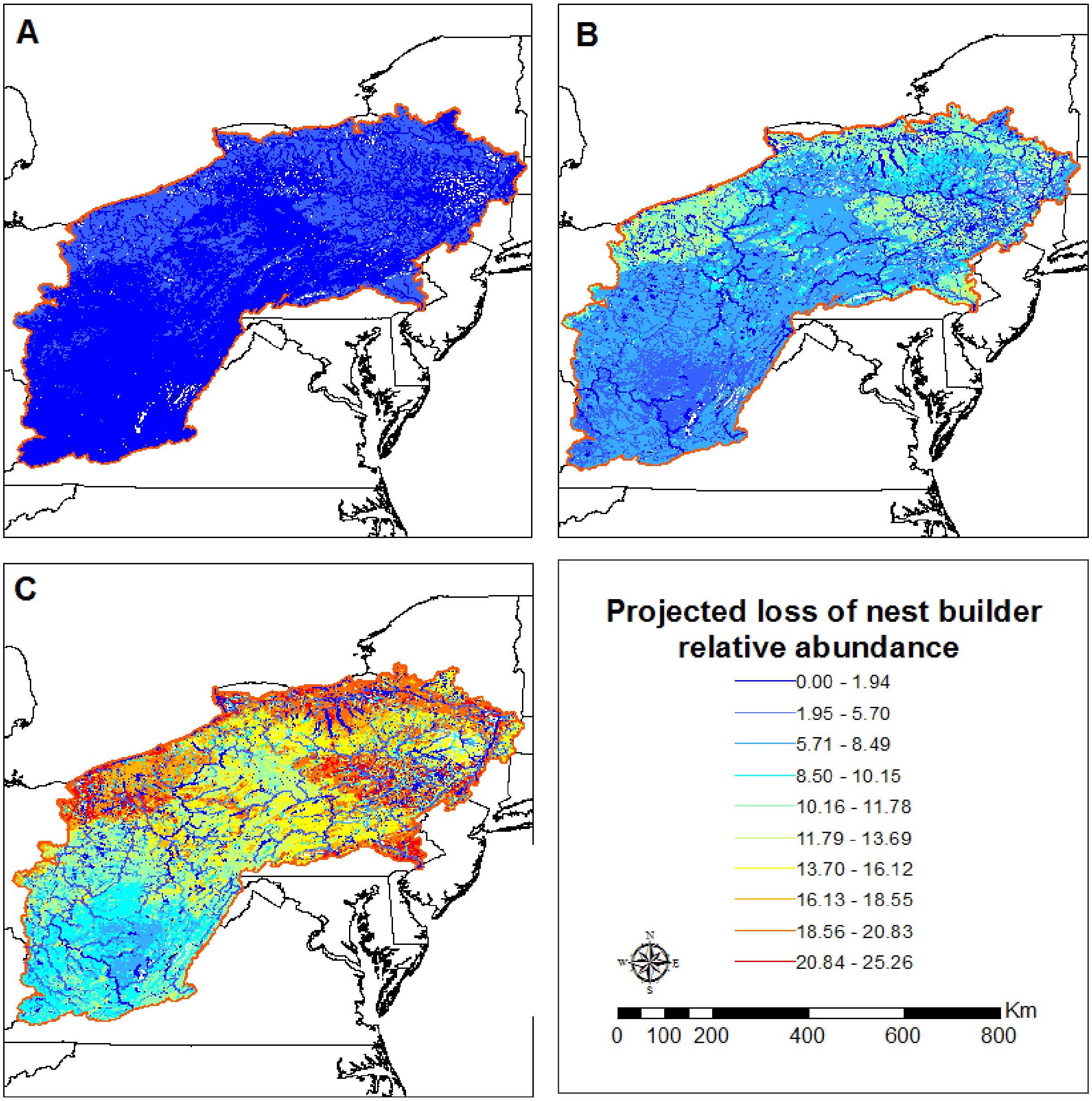

Figure 40. Maps of projected loss to the relative abundance of nest builder species (%) due to local low (A), local high (B) and cumulative high (C) pumping scenarios.

Importantly, Figure 40 represents reductions in nest builder abundance without regard to whether they existed in the stream in the first place. To obtain a more realistic estimate of the *cumulative* risk to particular fish groups due to withdrawals we combined HSI maps with empirical flow-ecology relationships and overlayed them with maps of projected gas development intensity, and probability of occurrence of particular species and functional guilds. Again, using nest builders as an example, we first calculated the probability of occurrence as the mean of occurrence probabilities of all nest builders (refer to Appendix F for a brief description of methods; Figure 41). We then overlayed it with the predicted loss of nest builder abundance maps (Figure 40, normalized from 0-1) to obtain an estimate of the relative risk of nest builder degradation resulting from hydraulic fracturing withdrawals.

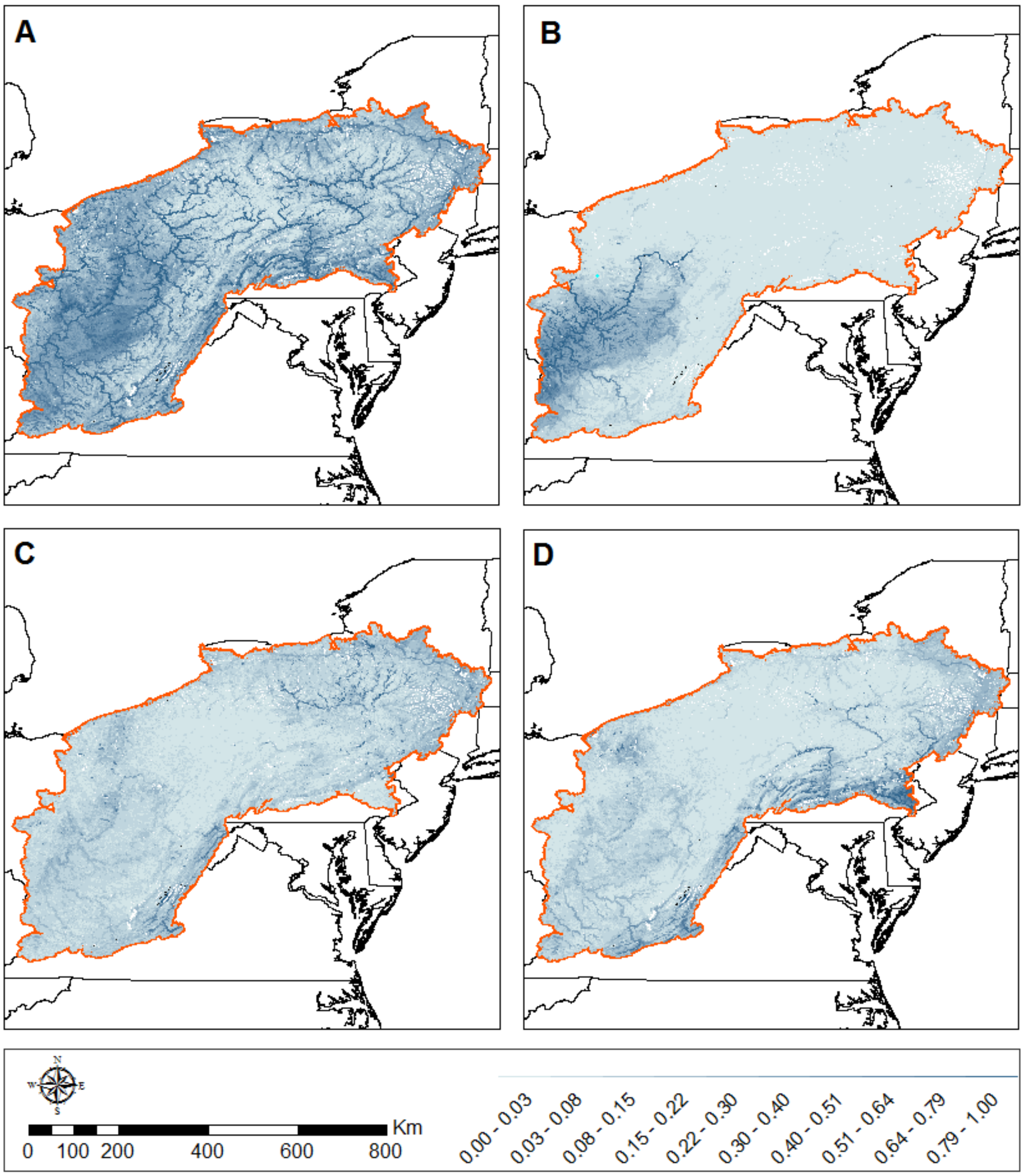

Figure 41. Results of species distribution models for smallmouth bass (A), spotted bass (B), fallfish (C) and red breasted sunfish (D). Legend displays probability of occurrence based on random forest model output. The probability of occurrence of nest builders was computed as the mean of all species comprising this guild.

In addition to hydrologic sensitivity to withdrawals, our cumulative risk assessment also incorporated a measure of the intensity of projected shale gas development in the MSR. Dunscomd et al. (2014) calculated the potential relative risk of shale gas development for every HUC-12 watershed in the AppLCC region (Figure 42). Although the analysis boundaries do not directly overlap, by overlaying their results with maps of nest builder occurrence and HSIs, we obtain a map of cumulative risk to nest builders that accounts for hydrologic sensitivity, probability of species presence and likelihood of exposure to pumping activities for most of the MSR. Accordingly, areas highlighted as high risk in Figure 43 (red lines) represent streams habitats favorable to nest builders, but also with a high relative risk of flow regime alteration from water withdrawals and high probability of shale gas development. Areas highlighted as high risk may be candidates for more judicious flow permitting, further study, and monitoring.

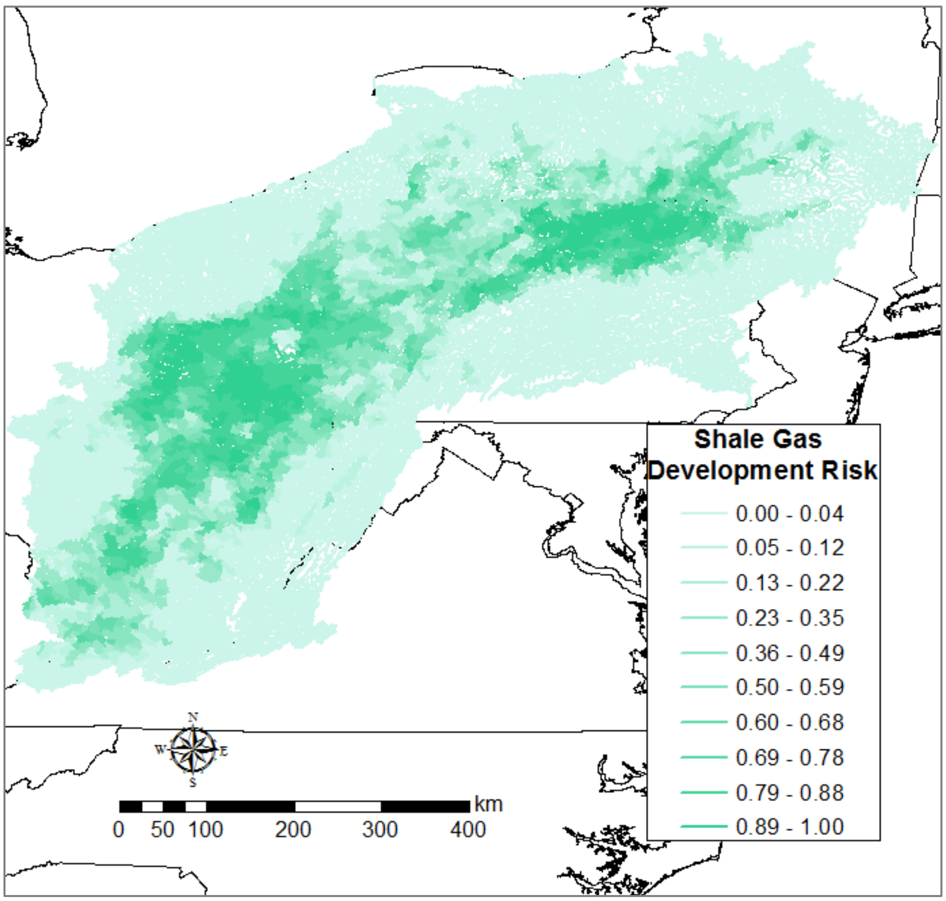

Figure 42. Map of potential relative risk to shale gas development for all HUC-12 watersheds in the MSR. Note, portions of the MSR were not included in the original analysis for the AppLCC region. Watersheds falling outside the AppLLC boundary, but within the MSR

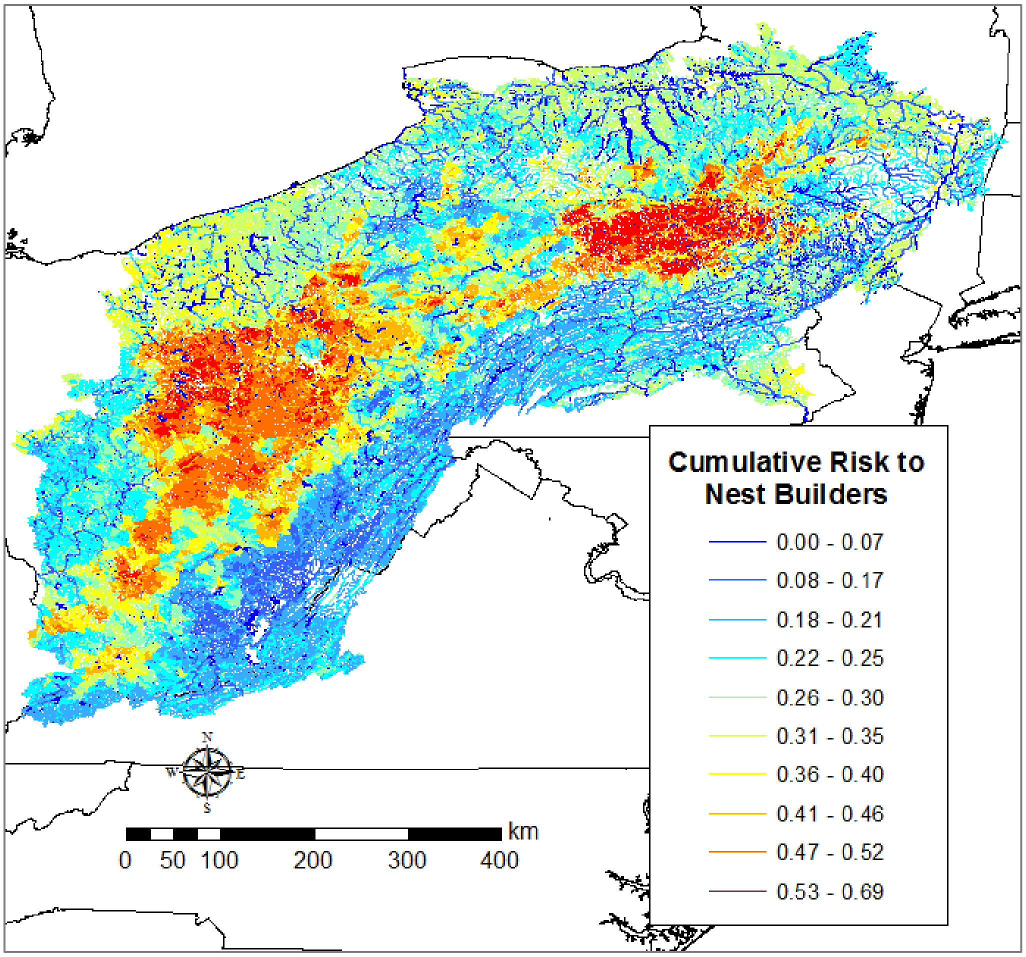

Figure 43. Map of cumulative relative risk of hydraulic fracturing activities to nest builders, reflecting: i) the likelihood of nest builder occurrence, ii) a stream's sensitivity to water withdrawals and iii) the probability that it will experience water extraction for shale gas development.

Management Implications

A key challenge facing water resource managers and conservation planners is the translation of quantitative flow-ecology relationships into actionable management strategies and tools. The ELOHA framework specifies that this step be informed by a social process (Figure 2), whereby acceptable ecological conditions and environmental flow standards are defined through an adaptive process of stakeholder input, scientific analysis, monitoring and feedback. Although this is beyond the scope of this report, we offer an example of how some of the more compelling F-E relationships may be applied to management questions within the MSR. Using fluvial species richness (Figure 11) as an example and applying the biological condition categories listed in Table 14, we can visualize how species loss associated with declining August flows interacts with boundaries of acceptable ecological status – as well as the maximum potential flow alteration resulting from four different pumping scenarios. The biological conditions categories in Table 14 and Figure 44 are hypothetical. Poff et al. (2010) suggests that: "one possible process for setting such risk levels is to use expert panels to identify 'thresholds of potential concern' (Biggs and Rogers 2003; Acreman et al. 2008), which establish where along the flow alteration gradient there is agreement among stakeholders (including scientists and managers) that further hydrologic change carries with it unacceptably high ecological risk."

Table 14. Hypothetical biological condition categories.

Biological Condition Category	Loss of Species Richness
1	< 5% (Healthy level of biodiversity)
2	5 - 15% (Reductions in sensitive species)
3	15 -35% (Moderate loss of sensitive species)
4	35 - 65% (Severe loss of sensitive species)
5	> 65% (Substantial overall reduction in biodiversity)

From Figure 44, it is evident that maintaining biological condition (BC) 1 requires roughly <5% alteration in August flow and that even under the maximum level of flow alteration for the "local low" pumping scenario predicted in the MSR by the RF models (vertical black dashed line), this threshold would not be exceeded. The lack of significant reduction in biological status under the "Local Low" pumping scenario suggests this level of extraction is not likely a management concern. Though not directly comparable, this level of flow alteration is similar to that of Shank and Stauffer (2014) who observed localized water withdrawals for hydraulic fracturing purposes rarely exceeded 6.5% of the mean daily flow in the Susquehanna River Basin. Indeed, most withdrawals were on the order of 0.04% and 0.10% of average daily flow in cold water and larger warm water streams, respectively. They also found that permitted withdrawals rates were generally substantially higher than actual withdrawals across all streams in their study, suggesting our pumping rates may be overly conservative. However, their analysis was based on only 12 streams and did not reflect the potential effects of cumulative water extraction.

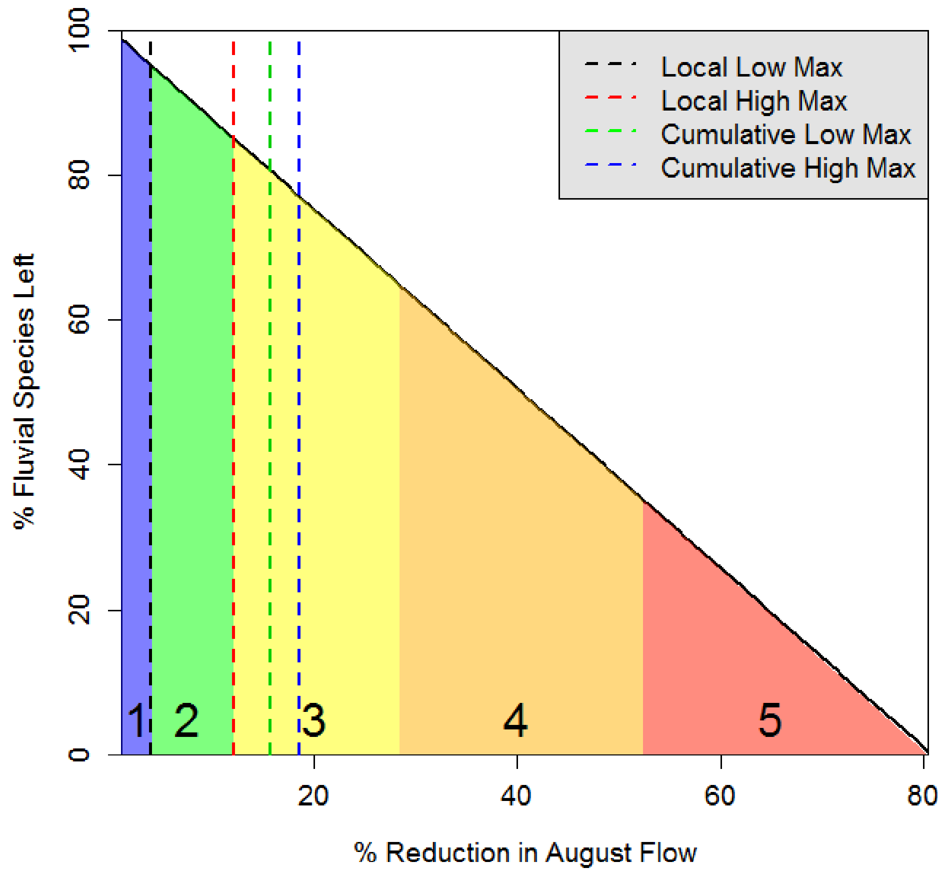

Figure 44. Conceptual relationship between percent fluvial species remaining with increasing reductions in August flow. Shaded polygons colors correspond to Table 14. Maximum levels of alteration predicted in the MSR for the local low & high and cumulative low & high pumping scenarios are indicated by dashed vertical lines.

Progressively higher extraction scenarios result in the potential for increased biological degradation. Under the worst-case-scenario (Cumulative High, dashed vertical blue line), it is possible that fluvial species richness may be reduced by approximately 25%, which equates to biological condition 3. Thus, under no extraction scenario would we see more than a "moderate" loss of species richness.

We must stress, however, that Figure 44 represents the predicted relationship under more ideal biological conditions (i.e. solid line in Figure 45). Streams that may be impaired due to other non-flow factors, such as water quality (dashed line, Figure 45) may exceed acceptable biological thresholds at lower levels of flow alteration (point B; Buchanan et al., 2013).

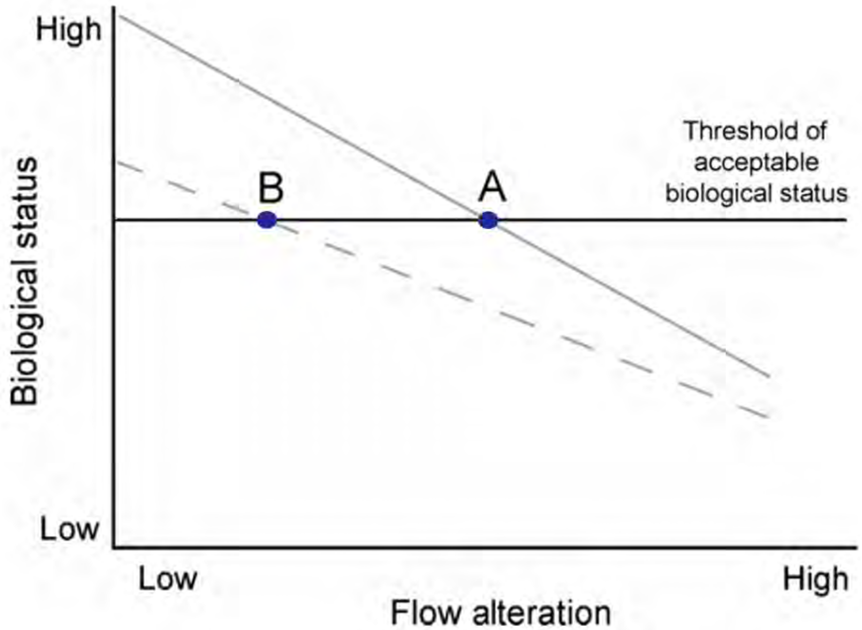

Figure 45. Conceptual relationship between stream biological condition and flow alteration in high- (solid line) and low-quality streams (dashed line). Points A and B highlight the lower levels of flow alteration required to exceed the threshold of acceptable biological status in high-quality vs. impaired stream systems. Adapted from Buchanan et al. (2013)

Streams with high observed flow alteration or those deemed a high risk to flow regime change due to water withdrawals may be good candidates for remediation, while streams with minimal alteration represent sites that would benefit from protection to prevent negative impacts to stream biota. The linear F-E relationships presented here could be used as decision support tools by managers and policy makers to decide where a particular level of water extraction falls on the biological condition continuum (i.e. worst, moderate or best-case scenario) and devise an appropriate response that protects or restores the streams hydrology and ecology.

Examining multiple quantiles within the wedge-shaped distribution of points in the species richness-August flow F-E relationship reveals several important points (Figure 46). First, all interior quantiles between 40 and 90 are significant, indicating that the explanatory variable (August flow alteration), is a principle limiting factor for species richness within this analytical space (Knight et al., 2013). Second, stream sites with lower or impaired species richness, such as indicated by the 70th quantile, would indeed exceed acceptable biological thresholds at lower levels of flow alteration. Third, the non-parallel slopes between significant interior quantiles (e.g. 40th, 50th, 70th and 90th) indicate that other non-flow related environmental factors are interacting with hydrology to influence species richness. The lower the quantile slope value, the more other non-flow related environmental factors play a role in influencing the dependent variable. Thus, in our example, species richness in the lower quantiles is influenced by August flow alteration *and,* increasingly, by an interaction with another unmeasured environmental factor(s). Consequently, interpreting quantile regression results and using them to inform policy decisions is not as straightforward as Figure 44 implies.

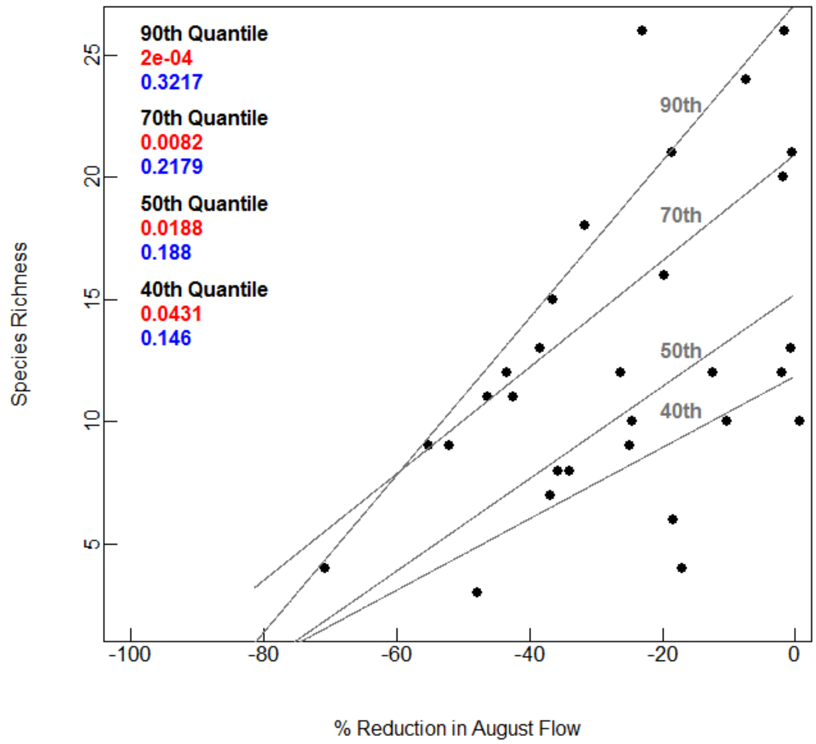

Figure 46. Species richness vs. percent alteration in median August flow (MA19). Grey lines represent 40th, 50th, 70th and 90th quantile regressions. P-values (red text) and slopes (blue text) associated with quantile regressions are indicated.

We anticipated that fish communities in the MSR would be most responsive to alterations in flow during low-flow periods such as the summer and fall. However, our F-E relationships suggested that fish were sensitive to flow regime alteration throughout the year. Interestingly, we also found that low-flow periods were more important than low-flow statistics in the F-E relationships. At the same time, our pumping analysis revealed that flow regimes were particularly sensitive to water withdrawals during the summer and fall - confirming the intuitive notion that abstractions will have a disproportionate effect during low-flow periods.

Thus, our pumping analysis suggests environmental flow standards and monitoring campaigns concerning water withdrawals for hydraulic fracturing should focus on low-flow hydrologic indices during the summer and fall as these are most sensitive to alteration. However, higher low-flow requirements will only protect fish communities if depletion of low-flows is the principle hydrologic stressor acting on aquatic biota. Our flow-ecology relationships indicate that biotic integrity of fish communities is also adversely affected by changes in average- and high-flow indices, indicating that low-flow provisions alone may be inadequate to protect riverine ecosystems in the MSR.

Altogether, these findings support multi-season flow recommendations that are protective of a range of natural flow regime components, such as those outlined by DePhilip and Moberg (2010, 2013). In the context of water withdrawals for hydraulic fracturing, it may also be prudent to ensure more conservative flow requirements for specific stream types (e.g. high risk streams), seasons and flow regime components that were shown to be more responsive to withdrawals. For example, DePhilip and Moberg (2013) suggest "higher levels of protection (i.e., more conservative limits to hydrologic alteration):

- To small streams as compared to large rivers (e.g., no change to monthly median in headwaters, < 10% change in small rivers, and < 15% change in medium tributaries and large rivers).
- In dry seasons compared to wet seasons (e.g., for medium tributaries and large rivers: no change to monthly Q90 in summer and fall and < 10% change to monthly Q90 in winter and spring).
- For low flow conditions than median or high flow conditions. (e.g., for medium tributaries and large rivers: <15% change to monthly median and < 10% change to monthly Q90)"

According to the withdrawal analysis, hydrologic disturbance from a single withdrawal point (local scenario) would not likely result in significant ecological effects except under the high local pumping scenario during summer and fall. In many cases, withdrawing at the "local high" rate of 2.5 cfs would exceed 6.8% of the mean daily flow – the maximum level of water extraction for hydraulic fracturing observed by Shank and Stauffer (2014). In addition, this level of extraction might be prohibited in many areas in the MSR due to pass-by flow regulations (i.e. a prescribed streamflow below which withdrawal must cease). For instance, for the vast majority of permits it issues, the Susquehanna River Basin Commission (SRBC) currently applies pass-by flow requirements that are a function of the lowest average flow that would be experienced during a consecutive 7-day period estimated to occur only once in 10 years (Q7-10). More specifically, the SRBC has determined that pass-by flows apply "if a proposed withdrawal, either individually or cumulatively when coupled with withdrawals for upstream users, exceeds 10 percent of the Q7-10 flow" (SRBC, 2015). Thus, the pumping analysis is likely overly conservative in many basins with pre-existing low-flow requirements. Given the lack of consistent environmental flow standards across the MSR and the dearth of empirical data on actual withdrawal rates, it is difficult to ascertain the degree to which our pumping analysis overestimates hydrologic risk. A potentially very useful exercise for water resource managers and conservation planners is to overlay regulatory boundaries with the HSI maps (Figures 37-39 and Appendix E) to highlight areas that are at high risk, but currently under-protected. These areas should be prioritized for management and monitoring.

An additional concern, which may complicate future water resource management decisions in the MSR, is global climate change. However, predicting the ecological consequences of water withdrawals in the context of climate change is not trivial given the host of other potential factors, including land-cover change, water-related infrastructure (e.g. dams), and other non-gas-development related consumptive water uses. Looking at the combined effects of projected changes in impervious cover, water withdrawals and climate change under both low and high growth and emission scenarios, Caldwell et al. (2012) showed that much of the northern and eastern portions of the MSR will experience modest increases in annual flows by 2060 (Figure 47). This was corroborated by Palmer et al. (2009) who conducted a similar study by evaluating projections from 12 different climate models. Caldwell et al. (2012) attributed enhanced annual flows mostly to increased precipitation and impervious surface, resulting in more runoff which exceeded projected increases in water withdrawals. Nevertheless, some watersheds in the southwestern MSR may experience up to a 10% reduction in average annual discharge, which may lead to adverse ecological impacts, particularly in streams with existing impairment. Moreover, the areas predicted to be more affected by climate, water use and land cover change overlap with the perennial flashy stream type, which our pumping analysis indicated would be particularly sensitive of withdrawals.

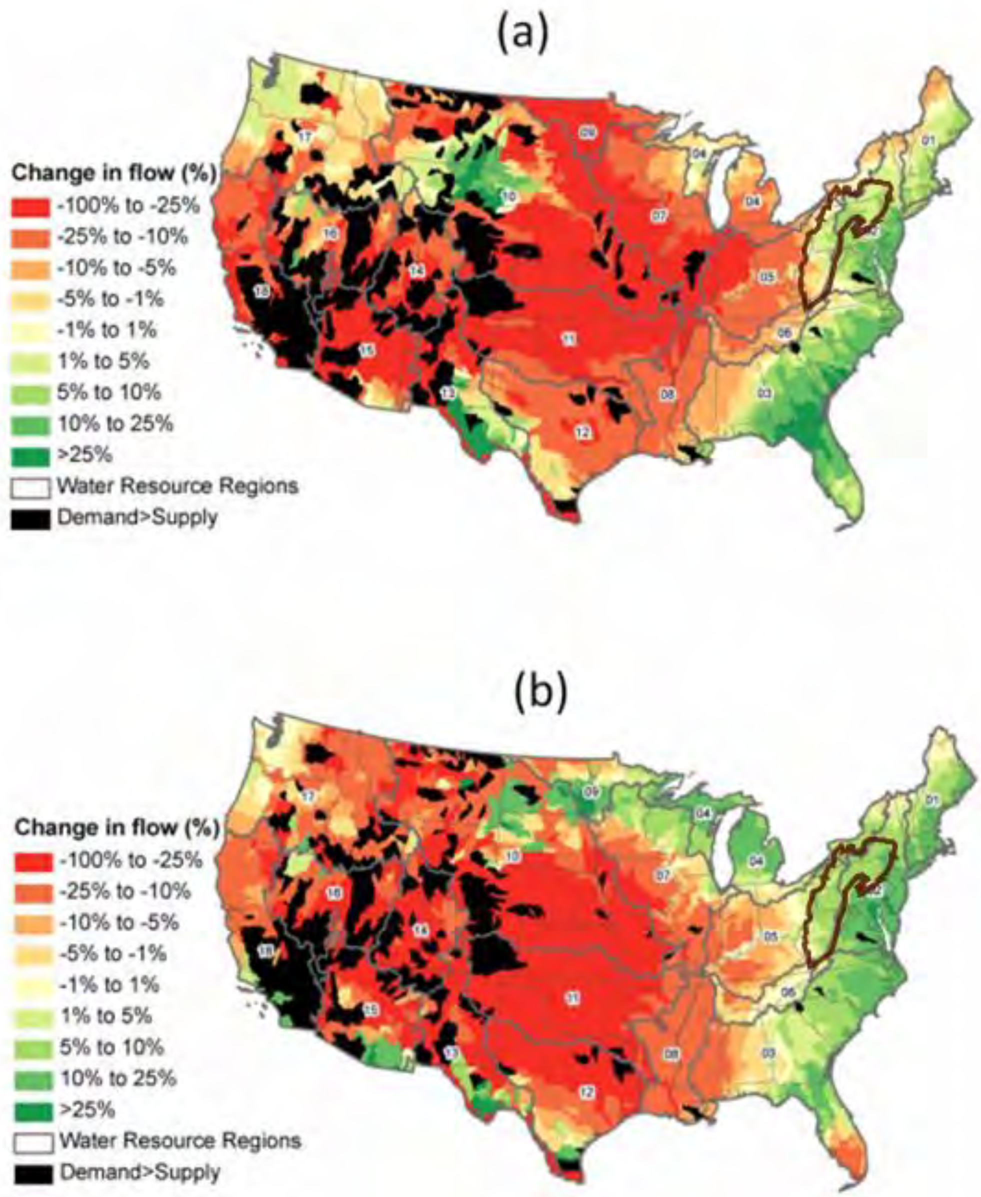

Figure 47. Projected changes in mean annual flows in 2060 given land use, population, and climate change under low (a) and high (b) growth and emission scenarios. Dark red polygons represents the approximate boundary of the Marcellus Shale Region. Black polygons represent watershed where gross demand exceeds the sum of surface water supply and groundwater withdrawals, indicating likely transfer of water from other watersheds. Adapted from Caldwell et al. (2012).

While this suggests that future climate and land use change may buffer increased withdrawals in much of the MSR, such analyses do not address the issue of increased climate variability on flow regimes. Hejazi and Moglen (2008) found that increased temperature and precipitation extremes associated with climate projections will lead to lower low-flows and higher peak flows in 6 urbanizing watersheds in the Piedmont region of Maryland. Such flow effects will likely disproportionately impact intermittent and flashy stream systems such as those identified in this report (Brooks, 2009). Given the lack of consensus and uncertainties regarding climate change there is a clear need for further research into the combined effects of climate change, water abstraction for gas development and other anthropogenic disturbances on freshwater ecosystems.

Limitations, Knowledge Gaps and Future Directions

We encountered a number of data limitations and knowledge gaps in the process of applying the ELOHA framework to the MSR. They are summarized as follows:

- Although our reference USGS gages were chosen because they were minimally altered, many have experience some level of human disturbance. Indeed, few watersheds in North America can reasonably be considered "pristine". This may obfuscate F-E relationships.
- Sampling periods of fish and hydrologic datasets did not necessary overlap directly in time. Thus, it is possible that in some cases, historical flow records, which partially overlapped fish sampling efforts in time, may not accurately reflect the degree of flow alteration experienced by the fish community at that site.
- The smaller range of drainage areas in the reference gage dataset restricted our analysis to basins <2,500 km^2, limiting the applicability of our F-E relationships to smaller streams and rivers. However, as the pumping analysis revealed, even the cumulative effect of multiple water extraction sites within the same watershed is not likely to result in appreciable flow alteration in basins larger than this threshold.
- Our analysis did not account for stream temperature classes due to a lack of temperature data in the MSR. This may also help to further refine F-E relationships.
- Our fish database contained fish sampling data from a variety of sources, which may introduce a certain degree of sampling error. For instance, different agencies may employ different sampling protocols (e.g. gear, gear deployment, sampling seasons, etc.) and data collection methods. To maintain a sufficient sample size it was necessary to make the assumption that the differences in sampling methodologies are negligible, but this is not necessarily the case.
- For ease of calculation we only determined statistical significance of F-E relationships by examining p-values associated with 90^{th} quantile regressions. To some extent, the choice of the 90^{th} quantile is arbitrary. Had we investigated other quantiles we would likely have observed other interesting F-E relationships. Future studies may want to pursue this line of inquiry.
- The pumping analysis was difficult to construct given the lack of data concerning actual observed water withdrawals by the gas industry. To date, there is only one study that explicitly evaluated empirical effects of hydraulic fracturing withdrawals in the MSR. More real-world data regarding the activities of hydraulic fracturing operators would greatly benefit future studies.
- The limited sample size of paired USGS gages and MARIS fish sampling sites prevented us from constructing F-E relationships specific to particular stream classes or

physiographic regions. An expanded flow-ecology dataset may allow for refinement of the F-E curves outlined here. Towards this end, it may be useful to develop RF models to predict natural and altered HIs across all basins in the MSR. This would greatly expand the flow-ecology dataset. However, it would come at the cost of additional uncertainty in estimates of flow alteration.

- Streams or basins within the MSR found to be at high risk and possessing good ecological data, yet with little existing flow data or flow standards in place may be good candidates for more detailed flow simulation using a process-based hydrologic model such as SWAT. For this step, it may be advisable to identify a subset of representative reference stream basins in each hydro-type identified by our stream classification effort. SWAT models could then be developed for each basin under "natural" and "altered" conditions to estimate flow alteration in all streams within the study basins. Consumptive pumping scenarios, which vary the rate and density at which surface water is withdrawn from subbasins within chosen catchments, could also be simulated.

Conclusions

Understanding the potential effects of surface water withdrawals for hydraulic fracturing activities on riverine ecosystems is a key step in making informed and prudent management decisions. Applying the ELOHA framework to stream systems within the Marcellus Shale Region revealed a number of significant findings that may be useful for defining environmental flow standards in the context of surface water withdrawals, as well as for providing guidance to future studies. For clarity, we summarize our salient findings by topic.

Constructing a hydrologic foundation

- Statistically based models performed well and provided reasonable estimates of natural flow regimes. This method is likely preferable to process-based hydrologic models over such a broad region due to excessive parameterization requirements, computational challenges, difficulties associated with the regionalization of calibration parameters and a lack of data regarding existing anthropogenic impacts (e.g. dam operations, industrial and agricultural water withdrawals) necessary to accurately simulate natural and altered flows in the MSR.

Selecting Flow Indices

- Using RF model performance (i.e. out-of-bag error) as a first cut of our HI selection protocol proved effective and practical. Using a threshold of 0.8 reduced the field of potential indices from 171 to 60. The remaining 60 HIs covered the major facets of the natural flow regime. However, other than flow constancy and predictability, flow timing-related HIs were not well predicted by the RF models, yet these could be affected by withdrawals and may be important to the fish community. The remaining HIs were further winnowed down to a more tractable set of 28 by considering both the indices sensitivity to modeled water extraction and its importance according to ecological theory.

Stream Classification

- Using a hierarchical stream classification of the Appalachian LCC and a set of geomorphic and climatic basin characteristics as training data for RF models, we predicted a total of four different stream classes in the MSR: i) Perennial Runoff 1, ii) Perennial Runoff 2, iii) Stable High Baseflow and iv) Perennial Flashy.
- Conceptually, streams in the perennial flashy category should be more sensitive to flow alteration than other classes in the MSR. These may be candidates for more targeted or conservative management.

Flow-Ecology Relationships

- Significant F-E relationships covered a range of fish assemblage and structure metrics, as well as a variety of seasonal and annual flow statistics.
- The vast majority of significant relationships were associated with negative flow alteration and resulted in declining ecological metrics.
- Some ecological metrics, such as life history traits and trophic structure displayed inconsistent or mostly insignificant linkages with changes in flow regime. This may indicate that these metrics are not the most responsive to flow alteration, or perhaps, that errors and uncertainties in our analysis, due to small sample sizes, lead to some spurious results.
- Sample size limitations also prevented a rigorous investigation of class- or region-specific F-E relationships. Even so, our F-E relationships achieved significance for numerous ecological endpoints, suggesting many of the relationships hold over the entire MSR, regardless of hydro-type or bioregion.

Pumping Scenarios

- Our consumptive water use analysis revealed that the hydrologic effects of water abstraction decay exponentially with increasing drainage area across all pumping scenarios. Importantly, we noted a threshold in the flow alteration-area relationship. Specifically, basins larger than roughly 1000 km^2 would not likely experience substantial flow alteration from local or cumulative water withdrawals associated with hydraulic fracturing activities. This provides guidance for managers in that it suggests smaller watersheds should be prioritized for hydraulic fracturing related flow standards.
- The pumping scenarios also revealed that low-flow statistics including low-flow duration and seasonal low-flows (summer and fall) were most sensitive to withdrawals, while high flows were least sensitive. The nature of our pumping analyses precluded evaluation of withdrawal effects on timing- or frequency-related HIs.

- Local or cumulative pumping rates are much less likely to substantially affect stream hydrology during high flow seasons such as the winter and spring (median predicted percent alteration in median low flows during spring or winter < 15%). Additionally, median and high monthly flow statistics are unlikely to be altered much by hydraulic fracturing withdrawals. However, pumping during low flow seasons (summer and fall), especially during low flow periods (median low flows), may result in considerable changes in flow regimes (e.g. flow alteration in median low flows during the summer season across all pumping scenarios ranged from ~8-40%).
- The lack of pass-by flow limitations and re-use of flowback water likely resulted in conservative estimates.
- Based on the findings of Shank and Stauffer (2014), the local low scenario may represent the most realistic pumping rate for single withdrawal sites.
- Perennial flashy stream types are more sensitive than other stream classes in the MSR. These would also be good candidates for targeted management. In addition, stream flow depletion associated with climate, land use and water use change is projected to be strongest over this region of the MSR.

Risk Analysis

- RF models achieved acceptable performance in predicting hydrologic sensitivity indices across the MSR (i.e. most OOB pseudo-R^2 values exceeded 0.85).
- Mapping hydrologic sensitivity indices across the MSR revealed spatial and temporal patterns in risk of flow alteration due to withdrawals. Few streams are at high risk during the spring and winter, whereas a considerable number are at risk during the summer and fall seasons. In general, high risk streams are located in the southwestern (i.e. western portions of the Ohio River Basin) and northern (i.e. headwaters of the Upper Susquehanna and Hudson River Basins) sections of the MSR.
- High risk streams are characterized by smaller drainage areas, lower average annual precipitation, greater number of dams, higher elevation, fewer dams and lower overall dam storage, lower average depth to seasonal water table, lower elevation basins with lower average temperatures and flatter slopes, higher percentages of pasture and crop landuses, lower percentages of evergreen and mixed forest and higher percentages of poorly drained soils.
- The analysis provided information for targeted management by highlighting areas at greater risk of alteration. Combining the HSI mapping with species distribution models and existing or projected hydraulic fracturing well densities also provided species-specific risk assessments. Such assessments are important for evaluating the locations where species of concern (i.e. threaten, endangered or simply of particular management concern) are at greatest risk.

Management Implications

The above findings can ultimately be distilled to a number of important management implications and guidelines.

- Environmental flow standards and monitoring campaigns concerning water withdrawals for hydraulic fracturing should focus on low flow hydrologic indices during the summer and fall as these are most sensitive to alteration. However, our flow-ecology relationships indicate that fish communities are also adversely affected by changes in average- and high-flow indices, indicating that low-flow provisions alone (e.g. Q7-10) may be inadequate to protect riverine ecosystems in the MSR. Thus, we suggest multi-season flow standards that are protective of a range of natural flow regime components, such as those outlined by DePhilip and Moberg (2010, 2013).
- In the context of water withdrawals for hydro-fracking, it may also be prudent to ensure more conservative flow requirements for specific stream types (small streams with high risk indices), seasons and flow regime components that were shown to be more responsive to withdrawals.
- Many of the existing pass-by flow requirements in the MSR are based more on hydrologic rules-of-thumb rather than empirically-based quantitative analyses. The F-E relationships outlined here may provide guidance for the refinement and justification of environmental flow regulations. Likewise, in areas with minimal flow protections in place this analysis should provide necessary baseline data for constructing defensible initial flow standards.
- Streams with high observed flow alteration or those deemed a high risk to flow regime change due to water withdrawals may be good candidates for *remediation,* while streams with minimal alteration represent sites that would benefit from *protection* to prevent negative impacts to stream biota. The linear F-E relationships presented here could be used as decision support tools by managers and policy makers to pinpoint where a particular level of water extraction falls on the biological condition continuum (i.e. worst, moderate or best-case scenario) and devise an appropriate response that protects or restores the stream's hydrology and ecology.
- Another concern is that managers and policy makers understand that these estimates have a degree of uncertainty that remains unquantified. Thus, policy decisions based on these findings should occur as part of an adaptive process, wherein flow provisions are designed and implemented as experiments with appropriate monitoring and feedback.

Water resources and rainfall in the Marcellus Shale Region are abundant. The total amount of water required for gas development is small relative to the overall regional water demand. However, this report demonstrates that, while seemingly trivial at a regional scale, surface water withdrawals at the scale of individual streams, especially headwaters, can be considerable and must be appropriately managed to ensure that human water needs are well balanced with those of riverine ecosystems.

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Appendix A – Fish Lists

Table A-1. Common names, scientific names, and number and percentage of individuals observed at fish sampling sites within the MSR.

Common Name	Scientific Name	Total Catch	% of Total
Central Stoneroller	Campostoma anomalum	497,695	13.754
Brown trout	Salmo trutta	344,820	9.529
Brook Trout	Salvelinus fontinalis	279,342	7.720
Creek Chub	Semotilus atromaculatus	248,723	6.874
Bluntnose Minnow	Pimephales notatus	195,443	5.401
Eastern Blacknose Dace	Rhinichthys atratulus	185,663	5.131
White Sucker	Catostomus commersonii	180,951	5.001
Smallmouth Bass	Micropterus dolomieu	108,089	2.987
Western Blacknose Dace	Rhinichthys obtusus	95,377	2.636
Mottled Sculpin	Cottus bairdii	78,054	2.157
Common Shiner	Luxilus cornutus	71,590	1.978
Johnny Darter	Etheostoma nigrum	70,455	1.947
Longnose Dace	Rhinichthys cataractae	68,215	1.885
Northern Hog Sucker	Hypentelium nigricans	68,109	1.882
Greenside Darter	Etheostoma blennioides	64,387	1.779
Rock Bass	Ambloplites rupestris	59,001	1.631
Striped Shiner	Luxilus chrysocephalus	57,006	1.575
Gizzard Shad	Dorosoma cepedianum	56,081	1.550
Bluegill	Lepomis macrochirus	51,053	1.411
Fantail Darter	Etheostoma flabellare	46,867	1.295
Chattahoochee Sculpin	Cottus chattahoochee	45,301	1.252
Rainbow Trout	Oncorhynchus mykiss	41,418	1.145
Spotfin Shiner	Cyprinella spiloptera	38,528	1.065
Rainbow Darter	Etheostoma caeruleum	34,520	0.954
Cutlip Minnow	Exoglossum maxillingua	32,821	0.907
Green Sunfish	Lepomis cyanellus	32,632	0.902
Emerald Shiner	Notropis atherinoides	29,087	0.804
Redbreast Sunfish	Lepomis auritus	27,875	0.770
Sand Shiner	Notropis stramineus	26,852	0.742
Silverjaw Minnow	Notropis buccatus	25,571	0.707
Pumpkinseed	Lepomis gibbosus	25,350	0.701
Rosyface Shiner	Notropis rubellus	24,486	0.677
Common carp	Cyprinus carpio	23,123	0.639
Banded Darter	Etheostoma zonale	22,726	0.628
Slimy Sculpin	Cottus cognatus	22,250	0.615
Golden Redhorse	Moxostoma erythrurum	20,225	0.559

River Chub	Nocomis micropogon	19,946	0.551
Largemouth Bass	Micropterus salmoides	16,668	0.461
Spottail Shiner	Notropis hudsonius	15,670	0.433
Yellow Bullhead	Ameiurus natalis	13,618	0.376
Tessellated Darter	Etheostoma olmstedi	13,466	0.372
Logperch	Percina caprodes	11,998	0.332
Mimic Shiner	Notropis volucellus	11,183	0.309
Redside Dace	Clinostomus elongatus	10,831	0.299
Fallfish	Semotilus corporalis	10,592	0.293
Yellow Perch	Perca flavescens	9,659	0.267
Golden Shiner	Notemigonus crysoleucas	8,860	0.245
Walleye	Sander vitreus	8,249	0.228
Margined Madtom	Noturus insignis	7,956	0.220
Black Redhorse	Moxostoma duquesnei	7,347	0.203
Longear Sunfish	Lepomis megalotis	7,147	0.198
Brown Bullhead	Ameiurus nebulosus	6,767	0.187
Southern Redbelly Dace	Phoxinus erythrogaster	6,012	0.166
Silver Shiner	Notropis photogenis	5,923	0.164
Variegate Darter	Etheostoma variatum	5,606	0.155
Spotted Bass	Micropterus punctulatus	5,367	0.148
Fathead Minnow	Pimephales promelas	5,140	0.142
Mountain Redbelly Dace	Phoxinus oreas	4,580	0.127
Freshwater Drum	Aplodinotus grunniens	4,504	0.124
Channel Catfish	Ictalurus punctatus	4,414	0.122
White Perch	Morone americana	4,318	0.119
Blue Ridge Sculpin	Cottus caeruleomentum	4,026	0.111
Torrent Sucker	Thoburnia rhothoeca	3,948	0.109
Brown Trout	Salmo trutta	3,924	0.108
Redfin Pickerel	Esox americanus	3,582	0.099
Central Mudminnow	Umbra limi	3,323	0.092
Black Crappie	Pomoxis nigromaculatus	3,192	0.088
Silver Redhorse	Moxostoma anisurum	3,067	0.085
Blackside Darter	Percina maculata	2,970	0.082
Smallmouth Redhorse	Moxostoma breviceps	2,898	0.080
Pearl Dace	Margariscus margarita	2,719	0.075
Bluehead Chub	Nocomis leptocephalus	2,477	0.068
Trout-perch	Percopsis omiscomaycus	2,327	0.064
Stonecat	Noturus flavus	2,314	0.064
Quillback	Carpiodes cyprinus	2,186	0.060
Northern Redbelly Dace	Phoxinus eos	2,017	0.056
Redfin Shiner	Lythrurus umbratilis	1,955	0.054
White Crappie	Pomoxis annularis	1,947	0.054
Chain Pickerel	Esox niger	1,893	0.052

Telescope Shiner	Notropis telescopus	1,876	0.052
Warmouth	Lepomis gulosus	1,719	0.048
Banded Killifish	Fundulus diaphanus	1,621	0.045
White Bass	Morone chrysops	1,511	0.042
Hornyhead Chub	Nocomis biguttatus	1,448	0.040
Least Brook Lamprey	Lampetra aepyptera	1,302	0.036
Shield Darter	Percina peltata	1,264	0.035
Suckermouth Minnow	Phenacobius mirabilis	1,254	0.035
Blueback Herring	Alosa aestivalis	1,198	0.033
Sauger	Sander canadensis	1,172	0.032
Brook Stickleback	Culaea inconstans	1,133	0.031
Brook Silverside	Labidesthes sicculus	1,094	0.030
Spotted Sucker	Minytrema melanops	1,091	0.030
Checkered Sculpin	Cottus sp. 7	1,067	0.029
Shorthead Redhorse	Moxostoma macrolepidotum	972	0.027
Smallmouth Buffalo	Ictiobus bubalus	839	0.023
Alewife	Alosa pseudoharengus	811	0.022
Orangespotted Sunfish	Lepomis humilis	765	0.021
Longfin Darter	Etheostoma longimanum	727	0.020
Bigeye Chub	Hybopsis amblops	705	0.019
American Shad	Alosa sapidissima	701	0.019
Northern Pike	Esox lucius	701	0.019
Flathead Catfish	Pylodictis olivaris	695	0.019
Gravel Chub	Erimystax x-punctatus	664	0.018
Walleyes and Saugers	Sander spp.	621	0.017
American Eel	Anguilla rostrata	608	0.017
Satinfin Shiner	Cyprinella analostana	583	0.016
Brindled Madtom	Noturus miurus	583	0.016
Black Bullhead	Ameiurus melas	582	0.016
Round Goby	Neogobius melanostomus	542	0.015
Snubnose Darter	Etheostoma simoterum	534	0.015
Orangethroat Darter	Etheostoma spectabile	512	0.014
Atlantic Salmon	Salmo salar	440	0.012
Burbot	Lota lota	432	0.012
Redear Sunfish	Lepomis microlophus	431	0.012
Dusky Darter	Percina sciera	428	0.012
Channel Shiner	Notropis wickliffi	425	0.012
Longhead Darter	Percina macrocephala	385	0.011
Potomac Sculpin	Cottus girardi	355	0.010
Round Whitefish	Prosopium cylindraceum	347	0.010
Banded Sunfish	Enneacanthus obesus	344	0.010
White Shiner	Luxilus albeolus	344	0.010
Muskellunge	Esox masquinongy	318	0.009

Longnose Gar	Lepisosteus osseus	280	0.008
Swallowtail Shiner	Notropis procne	264	0.007
American Brook Lamprey	Lampetra appendix	261	0.007
Streamline Chub	Erimystax dissimilis	250	0.007
Bluestone Sculpin	Cottus sp. 1	248	0.007
Creek Chubsucker	Erimyzon oblongus	239	0.007
Bigmouth Chub	Nocomis platyrhynchus	234	0.006
Pirate Perch	Aphredoderus sayanus	227	0.006
Blacktip Jumprock	Moxostoma cervinum	226	0.006
River Carpsucker	Carpiodes carpio	222	0.006
Lake Chub	Couesius plumbeus	201	0.006
Bullhead Minnow	Pimephales vigilax	200	0.006
Silver Chub	Macrhybopsis storeriana	193	0.005
Longnose Sucker	Catostomus catostomus	183	0.005
Scarlet Shiner	Lythrurus fasciolaris	165	0.005
River Redhorse	Moxostoma carinatum	158	0.004
Whitetail Shiner	Cyprinella galactura	146	0.004
Roanoke Darter	Percina roanoka	145	0.004
Highfin Carpsucker	Carpiodes velifer	140	0.004
Tonguetied Minnow	Exoglossum laurae	136	0.004
Channel Darter	Percina copelandi	135	0.004
Mountain Brook Lamprey	Ichthyomyzon greeleyi	131	0.004
Eastern Mudminnow	Umbra pygmaea	130	0.004
Banded Sculpin	Cottus carolinae	129	0.004
Crescent Shiner	Luxilus cerasinus	128	0.004
Western Mosquitofish	Gambusia affinis	127	0.004
Eastern Silvery Minnow	Hybognathus regius	124	0.003
Striped Bass	Morone saxatilis	122	0.003
Bigmouth Shiner	Notropis dorsalis	116	0.003
Bluebreast Darter	Etheostoma camurum	102	0.003
White Catfish	Ameiurus catus	101	0.003
Eastern Sand Darter	Ammocrypta pellucida	96	0.003
Bowfin	Amia calva	93	0.003
Ghost Shiner	Notropis buchanani	83	0.002
Slenderhead Darter	Percina phoxocephala	79	0.002
Steelcolor Shiner	Cyprinella whipplei	76	0.002
Blueside Shiner	Lythrurus ardens	71	0.002
Mooneye	Hiodon tergisus	65	0.002
Bull Chub	Nocomis raneyi	65	0.002
Skipjack Herring	Alosa chrysochloris	63	0.002
Black Sculpin	Cottus baileyi	61	0.002
Tadpole Madtom	Noturus gyrinus	61	0.002
Blacknose shiner	Notropis heterolepis	58	0.002

Unidentified lamprey	Petromyzontidae	56	0.002
Ohio Lamprey	Ichthyomyzon bdellium	54	0.001
Candy Darter	Etheostoma osburni	51	0.001
Fourspine Stickleback	Apeltes quadracus	50	0.001
Brassy Minnow	Hybognathus hankinsoni	50	0.001
Cutthroat Trout	Oncorhynchus clarkii	36	0.001
Bigmouth Buffalo	Ictiobus cyprinellus	34	0.001
Bridle Shiner	Notropis bifrenatus	31	0.001
Gilt Darter	Percina evides	23	0.001
Blackchin Shiner	Notropis heterodon	21	0.001
Ironcolor Shiner	Notropis chalybaeus	20	0.001
New River Shiner	Notropis scabriceps	20	0.001
Stripeback Darter	Percina notogramma	20	0.001
Sharpnose Darter	Percina oxyrhynchus	20	0.001
Black Buffalo	Ictiobus niger	19	0.001
Saffron Shiner	Notropis rubricroceus	19	0.001
Grass carp	Ctenopharyngodon idella	18	0.000
Kanawha Minnow	Phenacobius teretulus	18	0.000
Iowa Darter	Etheostoma exile	17	0.000
Blackstripe Topminnow	Fundulus notatus	17	0.000
Mountain Madtom	Noturus eleutherus	16	0.000
Comely Shiner	Notropis amoenus	15	0.000
Stripetail Darter	Etheostoma kennicotti	14	0.000
Rainbow Smelt	Osmerus mordax	13	0.000
Swamp Darter	Etheostoma fusiforme	12	0.000
Spotted Darter	Etheostoma maculatum	12	0.000
Bluespotted Sunfish	Enneacanthus gloriosus	10	0.000
Highland Shiner	Notropis micropteryx	10	0.000
Lake Chubsucker	Erimyzon sucetta	9	0.000
River Shiner	Notropis blennius	8	0.000
Slender Chub	Erimystax cahni	7	0.000
Greater Redhorse	Moxostoma valenciennesi	7	0.000
Ninespine Stickleback	Pungitius pungitius	7	0.000
Threespine Stickleback	Gasterosteus aculeatus	6	0.000
Appalachia Darter	Percina gymnocephala	6	0.000
Carpsuckers	Carpiodes sp.	5	0.000
Northern Brook Lamprey	Ichthyomyzon fossor	5	0.000
Sea Lamprey	Petromyzon marinus	5	0.000
Blue Sucker	Cycleptus elongatus	4	0.000
Highscale Shiner	Notropis hypsilepis	4	0.000
Roughhead Shiner	Notropis semperasper	4	0.000
Slender Madtom	Noturus exilis	4	0.000
Coho Salmon	Oncorhynchus kisutch	4	0.000

Chinook Salmon	Oncorhynchus tshawytscha	3	0.000
Tippecanoe Darter	Etheostoma tippecanoe	2	0.000
Eastern Mosquitofish	Gambusia holbrooki	2	0.000
River Darter	Percina shumardi	2	0.000
Jack Dempsey	Cichlasoma octofasciata	1	0.000
Threadfin Shad	Dorosoma petenense	1	0.000
Roanoke Hog Sucker	Hypentelium roanokense	1	0.000
Lampreys	Ichthyomyzon	1	0.000
Northern Madtom	Noturus stigmosus	1	0.000
Muscadine Darter	Percina smithvanizi	1	0.000
Lake Trout	Salvelinus namaycush	1	0.000
Atlantic Needlefish	Strongylura marina	1	0.000

Table A-2. Common names, scientific names, and percentage of sites sorted by the number of sites at which each species was observed.

Common Name	Scientific Name	# of Sites	% of Total
Brown trout	Salmo trutta	5,882	8.299
Brook Trout	Salvelinus fontinalis	4,455	6.286
Smallmouth Bass	Micropterus dolomieu	2,830	3.993
White Sucker	Catostomus commersonii	2,768	3.906
Creek Chub	Semotilus atromaculatus	2,471	3.487
Rock Bass	Ambloplites rupestris	2,310	3.259
Bluegill	Lepomis macrochirus	2,015	2.843
Rainbow Trout	Oncorhynchus mykiss	1,978	2.791
Eastern Blacknose Dace	Rhinichthys atratulus	1,857	2.620
Central Stoneroller	Campostoma anomalum	1,829	2.581
Pumpkinseed	Lepomis gibbosus	1,822	2.571
Bluntnose Minnow	Pimephales notatus	1,804	2.545
Largemouth Bass	Micropterus salmoides	1,706	2.407
Northern Hog Sucker	Hypentelium nigricans	1,624	2.291
Green Sunfish	Lepomis cyanellus	1,477	2.084
Fantail Darter	Etheostoma flabellare	1,425	2.011
Common Shiner	Luxilus cornutus	1,363	1.923
Johnny Darter	Etheostoma nigrum	1,334	1.882
Longnose Dace	Rhinichthys cataractae	1,265	1.785
Yellow Bullhead	Ameiurus natalis	1,056	1.490
Greenside Darter	Etheostoma blennioides	915	1.291
Western Blacknose Dace	Rhinichthys obtusus	880	1.242
Chattahoochee Sculpin	Cottus chattahoochee	869	1.226
Rainbow Darter	Etheostoma caeruleum	852	1.202
Common carp	Cyprinus carpio	813	1.147
Brown Bullhead	Ameiurus nebulosus	799	1.127

Cutlip Minnow	Exoglossum maxillingua	745	1.051
Yellow Perch	Perca flavescens	710	1.002
Striped Shiner	Luxilus chrysocephalus	704	0.993
Spotfin Shiner	Cyprinella spiloptera	624	0.880
Logperch	Percina caprodes	607	0.856
Brown Trout	Salmo trutta	585	0.825
Golden Redhorse	Moxostoma erythrurum	581	0.820
Silverjaw Minnow	Notropis buccatus	569	0.803
Tessellated Darter	Etheostoma olmstedi	527	0.744
Walleye	Sander vitreus	526	0.742
Golden Shiner	Notemigonus crysoleucas	520	0.734
Redbreast Sunfish	Lepomis auritus	517	0.729
Sand Shiner	Notropis stramineus	503	0.710
Banded Darter	Etheostoma zonale	496	0.700
Black Crappie	Pomoxis nigromaculatus	490	0.691
Fallfish	Semotilus corporalis	483	0.682
Blackside Darter	Percina maculata	472	0.666
Chain Pickerel	Esox niger	445	0.628
Redside Dace	Clinostomus elongatus	442	0.624
Gizzard Shad	Dorosoma cepedianum	442	0.624
Redfin Pickerel	Esox americanus	406	0.573
Channel Catfish	Ictalurus punctatus	379	0.535
Slimy Sculpin	Cottus cognatus	367	0.518
Fathead Minnow	Pimephales promelas	365	0.515
River Chub	Nocomis micropogon	359	0.507
Rosyface Shiner	Notropis rubellus	356	0.502
Mottled Sculpin	Cottus bairdii	352	0.497
White Crappie	Pomoxis annularis	335	0.473
Margined Madtom	Noturus insignis	299	0.422
Mimic Shiner	Notropis volucellus	290	0.409
Warmouth	Lepomis gulosus	288	0.406
Emerald Shiner	Notropis atherinoides	283	0.399
Stonecat	Noturus flavus	264	0.372
Northern Pike	Esox lucius	249	0.351
Quillback	Carpiodes cyprinus	244	0.344
Silver Redhorse	Moxostoma anisurum	242	0.341
Spotted Bass	Micropterus punctulatus	241	0.340
Longear Sunfish	Lepomis megalotis	240	0.339
Freshwater Drum	Aplodinotus grunniens	234	0.330
Central Mudminnow	Umbra limi	221	0.312
Spottail Shiner	Notropis hudsonius	214	0.302
Silver Shiner	Notropis photogenis	189	0.267
Trout-perch	Percopsis omiscomaycus	188	0.265

Southern Redbelly Dace	Phoxinus erythrogaster	183	0.258
Black Redhorse	Moxostoma duquesnei	179	0.253
Brook Silverside	Labidesthes sicculus	161	0.227
Least Brook Lamprey	Lampetra aepyptera	159	0.224
Variegate Darter	Etheostoma variatum	156	0.220
Black Bullhead	Ameiurus melas	143	0.202
Pearl Dace	Margariscus margarita	141	0.199
Muskellunge	Esox masquinongy	137	0.193
Spotted Sucker	Minytrema melanops	128	0.181
Shorthead Redhorse	Moxostoma macrolepidotum	119	0.168
Redfin Shiner	Lythrurus umbratilis	118	0.166
White Bass	Morone chrysops	116	0.164
Smallmouth Redhorse	Moxostoma breviceps	115	0.162
Flathead Catfish	Pylodictis olivaris	114	0.161
White Perch	Morone americana	112	0.158
Brook Stickleback	Culaea inconstans	104	0.147
Sauger	Sander canadensis	104	0.147
Walleyes and Saugers	Sander spp.	100	0.141
Redear Sunfish	Lepomis microlophus	83	0.117
Brindled Madtom	Noturus miurus	78	0.110
Shield Darter	Percina peltata	74	0.104
Longnose Gar	Lepisosteus osseus	72	0.102
Suckermouth Minnow	Phenacobius mirabilis	67	0.095
Banded Killifish	Fundulus diaphanus	66	0.093
Hornyhead Chub	Nocomis biguttatus	66	0.093
Orangespotted Sunfish	Lepomis humilis	65	0.092
Northern Redbelly Dace	Phoxinus eos	63	0.089
American Eel	Anguilla rostrata	60	0.085
Burbot	Lota lota	58	0.082
Bigeye Chub	Hybopsis amblops	57	0.080
Bluehead Chub	Nocomis leptocephalus	54	0.076
Alewife	Alosa pseudoharengus	53	0.075
Smallmouth Buffalo	Ictiobus bubalus	53	0.075
Dusky Darter	Percina sciera	53	0.075
Atlantic Salmon	Salmo salar	53	0.075
Blue Ridge Sculpin	Cottus caeruleomentum	51	0.072
Bowfin	Amia calva	48	0.068
Streamline Chub	Erimystax dissimilis	47	0.066
Tonguetied Minnow	Exoglossum laurae	46	0.065
Longhead Darter	Percina macrocephala	43	0.061
Highfin Carpsucker	Carpiodes velifer	39	0.055
River Redhorse	Moxostoma carinatum	37	0.052
Torrent Sucker	Thoburnia rhothoeca	37	0.052

Longnose Sucker	Catostomus catostomus	36	0.051
Gravel Chub	Erimystax x-punctatus	35	0.049
Creek Chubsucker	Erimyzon oblongus	34	0.048
Channel Darter	Percina copelandi	33	0.047
Mountain Redbelly Dace	Phoxinus oreas	33	0.047
Satinfin Shiner	Cyprinella analostana	32	0.045
Eastern Sand Darter	Ammocrypta pellucida	30	0.042
Telescope Shiner	Notropis telescopus	30	0.042
Bullhead Minnow	Pimephales vigilax	30	0.042
River Carpsucker	Carpiodes carpio	29	0.041
Checkered Sculpin	Cottus sp. 7	28	0.040
American Brook Lamprey	Lampetra appendix	27	0.038
Round Goby	Neogobius melanostomus	27	0.038
Unidentified lamprey	Petromyzontidae	27	0.038
Eastern Mudminnow	Umbra pygmaea	26	0.037
Potomac Sculpin	Cottus girardi	25	0.035
Ohio Lamprey	Ichthyomyzon bdellium	24	0.034
Orangethroat Darter	Etheostoma spectabile	21	0.030
Blueside Shiner	Lythrurus ardens	19	0.027
Bluebreast Darter	Etheostoma camurum	18	0.025
Skipjack Herring	Alosa chrysochloris	16	0.023
White Catfish	Ameiurus catus	16	0.023
Whitetail Shiner	Cyprinella galactura	16	0.023
Striped Bass	Morone saxatilis	16	0.023
Bigmouth Shiner	Notropis dorsalis	15	0.021
Channel Shiner	Notropis wickliffi	15	0.021
Slenderhead Darter	Percina phoxocephala	15	0.021
Blueback Herring	Alosa aestivalis	14	0.020
Banded Sunfish	Enneacanthus obesus	14	0.020
Longfin Darter	Etheostoma longimanum	14	0.020
Silver Chub	Macrhybopsis storeriana	13	0.018
Roanoke Darter	Percina roanoka	13	0.018
Mooneye	Hiodon tergisus	12	0.017
Eastern Silvery Minnow	Hybognathus regius	12	0.017
White Shiner	Luxilus albeolus	12	0.017
Crescent Shiner	Luxilus cerasinus	12	0.017
Steelcolor Shiner	Cyprinella whipplei	11	0.016
Scarlet Shiner	Lythrurus fasciolaris	11	0.016
Blacktip Jumprock	Moxostoma cervinum	11	0.016
Ghost Shiner	Notropis buchanani	11	0.016
Grass carp	Ctenopharyngodon idella	10	0.014
Bigmouth Buffalo	Ictiobus cyprinellus	10	0.014
Bigmouth Chub	Nocomis platyrhynchus	10	0.014

Blacknose shiner	Notropis heterolepis	9	0.013
Tadpole Madtom	Noturus gyrinus	9	0.013
American Shad	Alosa sapidissima	8	0.011
Bluestone Sculpin	Cottus sp. 1	8	0.011
Western Mosquitofish	Gambusia affinis	8	0.011
Brassy Minnow	Hybognathus hankinsoni	8	0.011
Mountain Brook Lamprey	Ichthyomyzon greeleyi	8	0.011
Bull Chub	Nocomis raneyi	7	0.010
Pirate Perch	Aphredoderus sayanus	6	0.008
Snubnose Darter	Etheostoma simoterum	6	0.008
Swallowtail Shiner	Notropis procne	6	0.008
Mountain Madtom	Noturus eleutherus	6	0.008
Rainbow Smelt	Osmerus mordax	6	0.008
Fourspine Stickleback	Apeltes quadracus	5	0.007
Banded Sculpin	Cottus carolinae	5	0.007
Candy Darter	Etheostoma osburni	5	0.007
Greater Redhorse	Moxostoma valenciennesi	5	0.007
Comely Shiner	Notropis amoenus	5	0.007
Lake Chub	Couesius plumbeus	4	0.006
Bridle Shiner	Notropis bifrenatus	4	0.006
River Shiner	Notropis blennius	4	0.006
Blackchin Shiner	Notropis heterodon	4	0.006
Cutthroat Trout	Oncorhynchus clarkii	4	0.006
Sharpnose Darter	Percina oxyrhynchus	4	0.006
Sea Lamprey	Petromyzon marinus	4	0.006
Blackstripe Topminnow	Fundulus notatus	3	0.004
Black Buffalo	Ictiobus niger	3	0.004
Roughhead Shiner	Notropis semperasper	3	0.004
Coho Salmon	Oncorhynchus kisutch	3	0.004
Stripeback Darter	Percina notogramma	3	0.004
Lake Chubsucker	Erimyzon sucetta	2	0.003
Iowa Darter	Etheostoma exile	2	0.003
Spotted Darter	Etheostoma maculatum	2	0.003
Saffron Shiner	Notropis rubricroceus	2	0.003
New River Shiner	Notropis scabriceps	2	0.003
Chinook Salmon	Oncorhynchus tshawytscha	2	0.003
River Darter	Percina shumardi	2	0.003
Round Whitefish	Prosopium cylindraceum	2	0.003
Carpsuckers	Carpiodes sp.	1	0.001
Jack Dempsey	Cichlasoma octofasciata	1	0.001
Black Sculpin	Cottus baileyi	1	0.001
Blue Sucker	Cycleptus elongatus	1	0.001
Threadfin Shad	Dorosoma petenense	1	0.001

Bluespotted Sunfish	Enneacanthus gloriosus	1	0.001
Slender Chub	Erimystax cahni	1	0.001
Swamp Darter	Etheostoma fusiforme	1	0.001
Stripetail Darter	Etheostoma kennicotti	1	0.001
Tippecanoe Darter	Etheostoma tippecanoe	1	0.001
Eastern Mosquitofish	Gambusia holbrooki	1	0.001
Threespine Stickleback	Gasterosteus aculeatus	1	0.001
Roanoke Hog Sucker	Hypentelium roanokense	1	0.001
Lampreys	Ichthyomyzon	1	0.001
Northern Brook Lamprey	Ichthyomyzon fossor	1	0.001
Ironcolor Shiner	Notropis chalybaeus	1	0.001
Highscale Shiner	Notropis hypsilepis	1	0.001
Highland Shiner	Notropis micropteryx	1	0.001
Slender Madtom	Noturus exilis	1	0.001
Northern Madtom	Noturus stigmosus	1	0.001
Gilt Darter	Percina evides	1	0.001
Appalachia Darter	Percina gymnocephala	1	0.001
Muscadine Darter	Percina smithvanizi	1	0.001
Kanawha Minnow	Phenacobius teretulus	1	0.001
Ninespine Stickleback	Pungitius pungitius	1	0.001
Lake Trout	Salvelinus namaycush	1	0.001
Atlantic Needlefish	Strongylura marina	1	0.001

Appendix B – TNC Flow-Ecology Hypotheses

Table B-1. Flow-ecology hypotheses from DePhilip and Moberg (2010).

Low Flow Component
Seasonal Flow Component
High Flow Component
Rate of Change

Working Hypotheses	Flow Component and Timing												River Types
	oct	nov	dec	jan	feb	mar	apr	may	jun	jul	aug	sep	
During winter, a decrease in streamflow and groundwater contributions may decrease depth and temperature. These conditions may encourage ice infiltration of salmonid eggs leading to reduced survival or impaired development.			x	x	x								All habitats where present
During winter, a decrease in streamflow and groundwater contributions may decrease depth and temperature. These conditions may encourage ice infiltration of salmonid eggs leading to reduced survival or impaired development.			x	x	x								Headwaters, cool/cold, Small rivers, cool-cold,
During winter, a decrease in streamflow may decrease availability and access to riffle habitats needed by riffle obligate fishes			x	x	x								All habitats where present
After spawning, during egg and larval development, a decrease in seasonal flows may dewater salmonid redds impairing development or reducing survival rates	x	x	x	x	x	x							Headwaters, cool/cold, Small rivers, cool-cold,
During spring, seasonal flows needed to maintain sediment free salmonid redds. A decrease in flow magnitude may lead to suffocation.						x	x						Headwaters, cool/cold, Small rivers, cool-cold,

During spawning and egg and larval development, riffle obligates need stable flows, if the magnitude of low flows decreases, fines may accumulate, suffocating eggs.				x	x	x	x	x	All river types
During March and April, riffle associates (redhorses) and potadromous fish (specifically walleye, sauger and Escocids), rely on temperature and increased streamflow to provide spawning cues. If low flow magnitude decreases, spawning cues and connectivity may be lost				x	x				All river types
During spawning and egg and larval development, riffle obligates need stable flows, if the magnitude of high flows increases, it may cause egg scour				x	x	x	x	x	All river types
During spawning and egg and larval development, riffle obligates need stable flows, increased flashiness may restrict access to gravel spawning habitats				x	x	x	x	x	All river types
Similarly, if high flow magnitude and duration increase, upstream spawning migration may be delayed (salmonids, burbot, migratory residents, riffle associates)	x	x	x	x	x				All river types
From March to June, a decrease in median flows may reduce fish movement to, and availability of, preferred spawning habitats. Fish spawning in riffles are especially sensitive and they vary in body-size and river types (eg darters, redhorses, paddlefish)				x	x	x	x		All river types
From March to June, great river fish and riffle associates in the navigation reaches need high flows to provide connectivity to upstream tributary habitats				x	x	x	x		Large navigational river
From April to July, the larvae larvae of migratory residents (walleye) and riffle associates (suckers) need slackwater habitats (often in stream margin), for development. An increase in the magnitude or frequency of high flow events would increase the velocity along stream margins reducing available slackwater habitat.					x	x	x	x	All Small, Tributary and Large river types
From April to July, larvae of migratory residents (walleye) and riffle associates (suckers) need slackwater habitats (often in stream margin), for development. A decrease in low flow magnitude may disconnect stream margin and backwater habitats from the main channel					x	x	x	x	All Small, Tributary and Large river types
From April to June, great river species including longnose gar and bigmouth buffalo need SAV or floodplain access for adhesive egg laying. Flooding duration must allow larvae to move back out into the channel					x	x	x		Large navigational river, Large river

During spring, an increase in the magnitude or frequency of high flows can scour nests. River chub may be particularly sensitive to this change in tributaries and large rivers and hornyhead chub in headwaters and small rivers.	x	x	x	x	x		All river types
During nest building and egg and larval development (spring) increased flashiness, may dewater nests and has been associated with decreased abundance of YOY.	x	x	x	x	x		All river types
During egg and larval development (spring), increased magnitude, frequency or duration of high flows may decrease egg and larval survival and associated year class strength.	x	x	x	x	x		All river types
From April through August, in riffles, if seasonal flows are too low then egg and larval development may be impaired by oxygen depletion, desiccation or suffocation	x	x	x	x	x		All river types
From April through August, in riffles, if high flow magnitude or frequency increase, developing eggs and larvae may be scoured and/or physically damaged	x	x	x	x	x		All river types
During summer months, a decrease in median flow may limit the quality and availability of riffle habitats for riffle obligate fishes			x	x	x	x	All river types
During the summer low flow period, a decrease in low flow magnitude can result in downstream migration of headwater fishes, compressing the species and thermal gradient, and increasing predator-prey interactions (eg brook trout and brown trout)			x	x	x	x	All headwater types
During the summer low flow period, a decrease in low flow magnitude may result in loss of refugia and a shift toward a top-predator dominated system			x	x	x	x	All river types
During summer months, riffle obligates that specialize in highly oxygenated, lower riffle/plunge turbulent environments (redside dace in headwaters, rosyface shiner in small warm streams, silver shiner in small cool-cold streams) are sensitive to decreasing flow magnitude which would contract or eliminate this habitat niche.			x	x	x	x	All river types
A decrease in the magnitude of summer low flows may restrict access for centrarchids and escocids to SAV habitats			x	x	x	x	All small river, medium tributaries and large river types

For substrate specialists an increase in high flow frequency, magnitude or duration may destabilize habitats and flush preferred substrates	x	x	x	x	x	x	x	x	x	x	x	x	All river types
For substrate specialists (specifically the eastern sand darter), high flow events maintain sandy substrates, a decrease in high flow magnitude, frequency or duration or may reduce habitat quality or abundance. Similarly, an increase in extreme high flow events may flush sands reducing abundance and quality of habitat	x	x	x	x	x	x	x	x	x	x	x	x	Small river, cool glaciated and Medium tributary, warm glaciated

Appendix C – HI Sensitivity

Please refer to Henrikson et al. (2006) for a detailed description of each hydrologic index.

Table C-1. Hydrologic indices and decriptions ranked according to their sensitivity to water withdrawals under high local and cumulative pumping scenarios.

HI Code	Description	Local High	Cumulative High
ML18	Base flow variability	31.73	38.96
DL6	1-day low flow variability	31.31	39.50
ML21	Annual low flow variability	31.29	39.49
DL7	3-day low flow variability	30.03	36.70
ML16	Median of annual low flows	27.27	33.33
ML19	Base flow	26.67	31.68
FL3	Frequency of low pulse spells	26.35	33.37
DL1	1-day low flow	26.06	30.82
DL8	7-day low flow variability	25.73	31.40
ML14	Annual low flow	25.00	29.71
ML15	Low flow index	25.00	33.33
ML22	Specific mean annual low flow	25.00	27.77
DL11	1-day low flow /median	25.00	27.27
DL2	3-day low flow	24.65	29.24
ML17	Base flow variability	22.22	25.00
DL3	7-day low flow	21.82	25.66
DL12	7-day low flow /median	18.18	23.08
MA6	Range in daily flows	15.85	18.99
ML9	Median September low flow	15.24	18.04
DL9	30-day low flow variability	13.61	16.46
DL15	Low exceedence flows (90%)	13.49	16.67
ML8	Median August low flow	13.37	15.85
DL4	30-day Low Flow	11.99	14.22
ML7	Median July low flow	11.28	13.40
MA32	Sept. flow variability	10.99	13.99
DL13	30-day low flow / median	10.00	11.11
MA31	Aug. flow variability	9.28	10.96
ML10	Median October low flow	9.25	10.97
RA7	Fall rate (log)	9.09	9.55
MA7	Range in daily flows (20-80)	7.63	9.12
MA33	Oct. flow variability	7.37	8.90
RA6	Rise rate (log)	6.67	9.09
ML6	Median June low flow	6.64	7.88
MA30	Jul. flow variability	5.73	7.20
MA8	Range in daily flows (25-75)	5.50	6.57

ML11	Median November low flow	5.43	6.45
TA1	Constancy	4.88	6.25
DL10	90-day low flow variability	4.86	5.75
DL5	90-day low flow	4.73	5.62
ML13	Variability in minimum monthly flows	4.49	5.29
FH4	High flood pulse count (7)	4.12	4.79
DL14	Low exceedence flows (75%)	4.06	4.76
MH22	High flow volume (3)	3.83	4.32
MA19	August flow	3.72	4.42
ML12	Median December low flow	3.59	4.27
MA34	Nov. flow variability	3.54	4.20
MA20	September flow	3.47	4.11
ML1	Median January low flow	3.43	4.08
TA2	Predictability	3.32	3.83
ML5	Median May low flow	3.29	3.91
MA29	Jun. flow variability	3.28	3.48
ML2	Median February low flow	2.94	3.50
MA18	July flow	2.94	3.49
MH14	Median of high flows	2.91	3.44
MH21	High flow volume index	2.84	3.37
MA11	Spread in daily flows (25-75)	2.83	3.38
MA9	Spread in daily flows (10-90)	2.81	3.32
DH11	1-day high flow / median	2.78	3.32
MA2	Median of daily mean flows	2.77	3.28
MA21	October flow	2.74	3.26
MA10	Spread in daily flows (20-80)	2.71	3.31
MH15	High flow discharge index	2.65	3.09
MH23	High flow volume	2.64	3.36
DH12	7-day high flow / median	2.61	3.12
MH27	High peak flow (75)	2.44	2.86
MA40	Monthly skewness	2.41	2.70
MH24	High peak flow (1)	2.41	2.90
DH13	30-day high flow / median	2.41	2.73
ML3	Median March low flow	2.21	2.63
FH3	High flood pulse count (3)	2.20	2.54
MH16	High flow discharge index (10)	2.18	2.51
ML4	Median April low flow	2.15	2.55
ML20	Base flow	2.13	2.25
FH7	Flood frequency (7)	2.07	2.43
MA35	December flow variability	2.03	2.20
MA43	Annual flow variability	1.92	2.13
MA17	June flow	1.85	2.19
MA37	Monthly variability (25-75)	1.81	2.18

MA38	Monthly variability (10-90)	1.79	2.15
DH18	High flow duration (3x's median)	1.78	2.06
MH25	High peak flow (3)	1.71	1.96
MA36	Monthly variability	1.67	2.03
MA28	May flow variability	1.59	1.64
MA4	Standard deviation/mean	1.56	1.85
MA22	November flow	1.52	1.80
MH17	High flow discharge index (25)	1.46	1.80
MA44	Annual flow variability (25-75)	1.41	1.61
DH19	High flow duration (7x's median)	1.38	1.75
MA27	April flow variability	1.35	1.46
MA24	January flow variability	1.33	1.53
MA42	Annual flow variability (max-min)	1.30	1.56
MA5	Skewness	1.28	1.56
MA3	Median annual variability	1.25	1.48
MA39	Monthly variability (SD)	1.25	1.47
MA1	Mean of daily flows	1.24	1.47
MA25	February flow variability	1.23	1.29
MA41	Annual runoff	1.23	1.41
MA26	March flow variability	1.13	1.22
MA16	May flow	1.04	1.23
MA23	December flow	1.04	1.24
MA12	Jan flow	0.97	1.15
MH26	High peak flow (7)	0.96	1.16
MA13	February flow	0.86	1.02
DH14	Flood duration	0.81	0.96
MH8	August high flow	0.75	0.89
MA15	April flow	0.71	0.84
MA14	March flow	0.63	0.75
MH9	September high flow	0.63	0.75
DH10	30-day high flow variability	0.61	0.71
DH5	90-day high flow	0.61	0.72
MH7	July high flow	0.56	0.67
MH10	October high flow	0.55	0.65
FH6	Flood frequency (3)	0.54	0.68
DH4	30-day high flow	0.41	0.48
DH9	30-day high flow variability	0.41	0.50
MH6	June high flow	0.37	0.43
MH11	November high flow	0.34	0.40
MH13	Monthly high flow variability	0.29	0.35
MH5	May high flow	0.28	0.33
MH12	December high flow	0.25	0.30
MH2	February high flow	0.22	0.26

MH1	January high flow	0.21	0.25
DH3	7-day high flow	0.21	0.25
DH8	7-day high flow variability	0.20	0.25
MH4	April high flow	0.17	0.20
TL2	Julian date of annual minimum variability	0.17	0.54
MH18	Annual high flow variability	0.16	0.18
MH3	March high flow	0.15	0.18
DH2	3-day high flow	0.13	0.15
DH7	3-day high flow variability	0.13	0.16
MH20	Specific mean annual maximum flow	0.08	0.09
DH1	1-day high flow	0.08	0.09
DH6	1-day high flow variability	0.08	0.10
RA1	Rise rate	0.04	0.08
RA2	Rise rate variability	0.02	0.05
RA3	Fall rate	0.02	0.06
TL1	Julian date of annual minimum	0.02	0.12
RA4	Fall rate variability	0.01	0.04
MA45	Skewness of annual flows	0.00	0.00
MH19	Skewness annual high flows	0.00	0.00
FL1	Low flood pulse count	0.00	0.00
FL2	Low flood pulse count variailbity	0.00	0.00
FH1	High flood pulse count	0.00	0.00
FH2	High flood pulse count variability	0.00	0.00
FH5	Flood frequency (1)	0.00	0.00
FH8	Flood frequency (25%)	0.00	0.00
FH9	Flood frequency (75%)	0.00	0.00
FH10	Flood frequency (1_min)	0.00	0.00
FH11	Flood frequency (Bnkfl)	0.00	0.00
DL16	Low flow pulse duration	0.00	0.00
DL17	Low flow pulse duration variability	0.00	0.00
DH15	High flow pulse duration	0.00	0.00
DH16	High flow pulse duration variability	0.00	0.00
DH17	High flow duration (median)	0.00	0.00
DH20	High flow duration (75%)	0.00	0.00
DH21	High flow duration (25%)	0.00	0.00
DH22	Flood interval	0.00	0.00
DH23	Flood duration	0.00	0.00
DH24	Flood-free days	0.00	0.00
TA3	Predictability of flooding	0.00	0.00
TL3	Seasonal predictability of low flow	0.00	0.00
TL4	Seasonal predictability of non-low flow	0.00	0.00
TH1	Julian date of annual maximum	0.00	0.00
TH2	Julian date of annual maximum variability	0.00	0.00

TH3	Seasonal predictability of nonflooding	0.00	0.00
RA5	Number of day rises	0.00	0.00
RA8	Reversals	0.00	0.03
RA9	Reversal variability	0.00	0.00

Appendix D – Variable Importance Plots

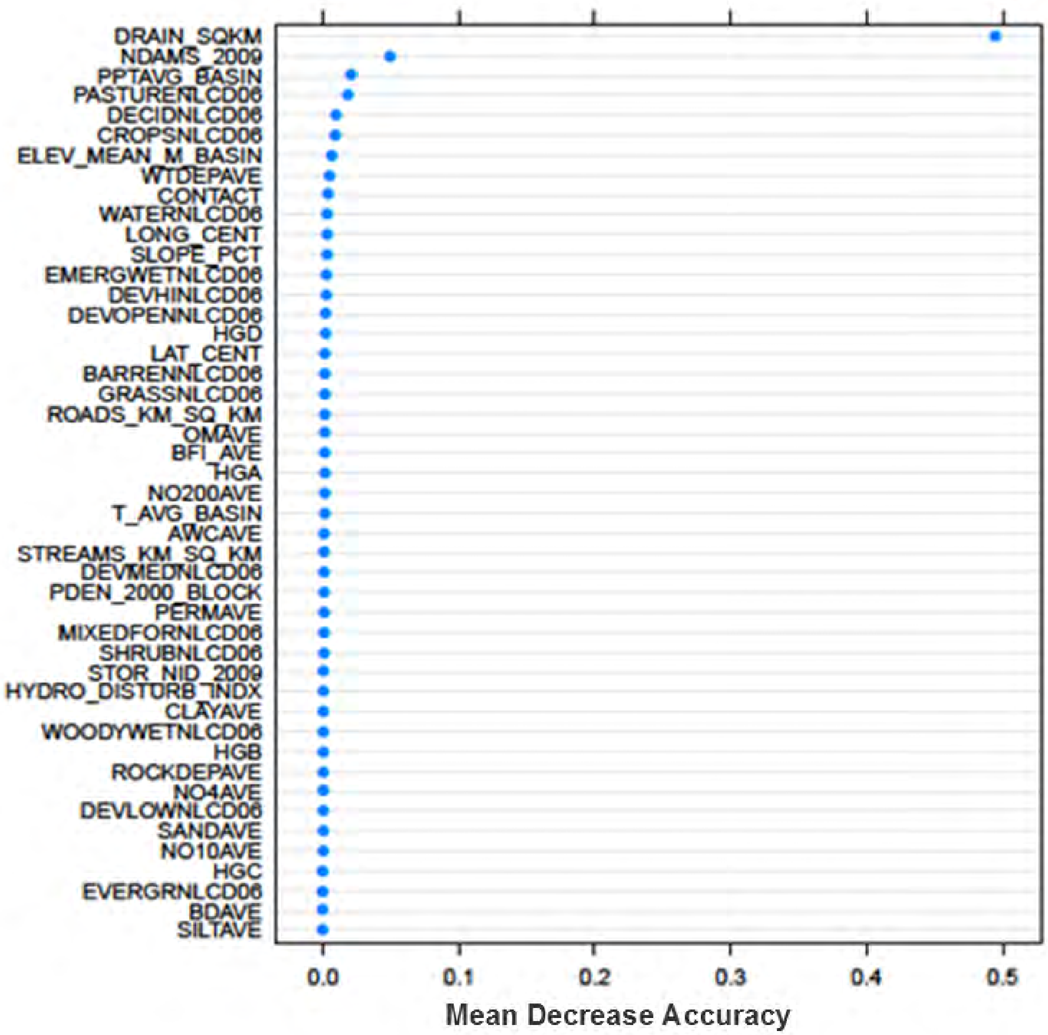

Figure D-1. Unbiased variable importance plot for the winter low-flow hydrologic sensitivity RF model.

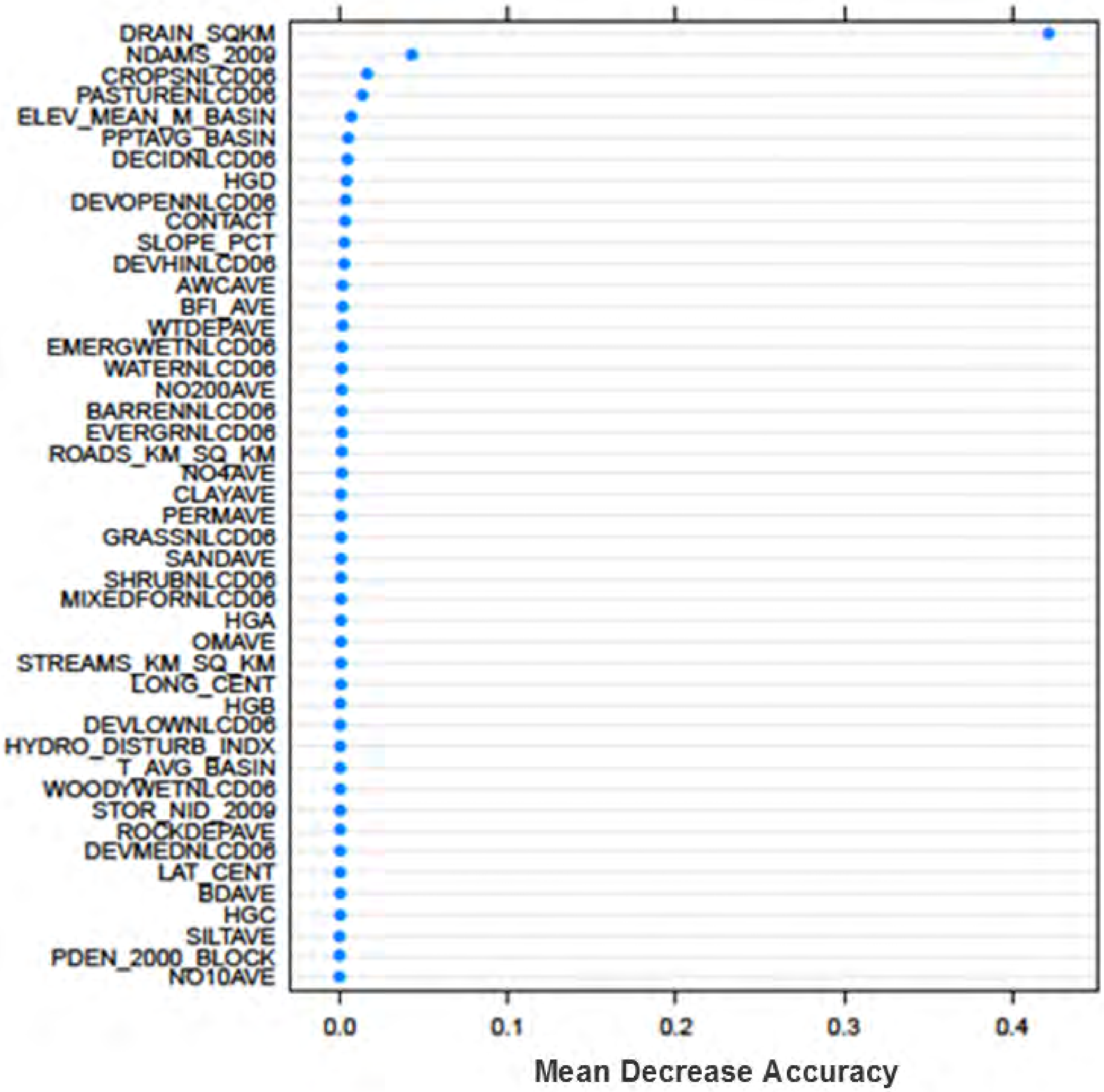

Figure D-2. Unbiased variable importance plot for the spring low-flow hydrologic sensitivity RF model

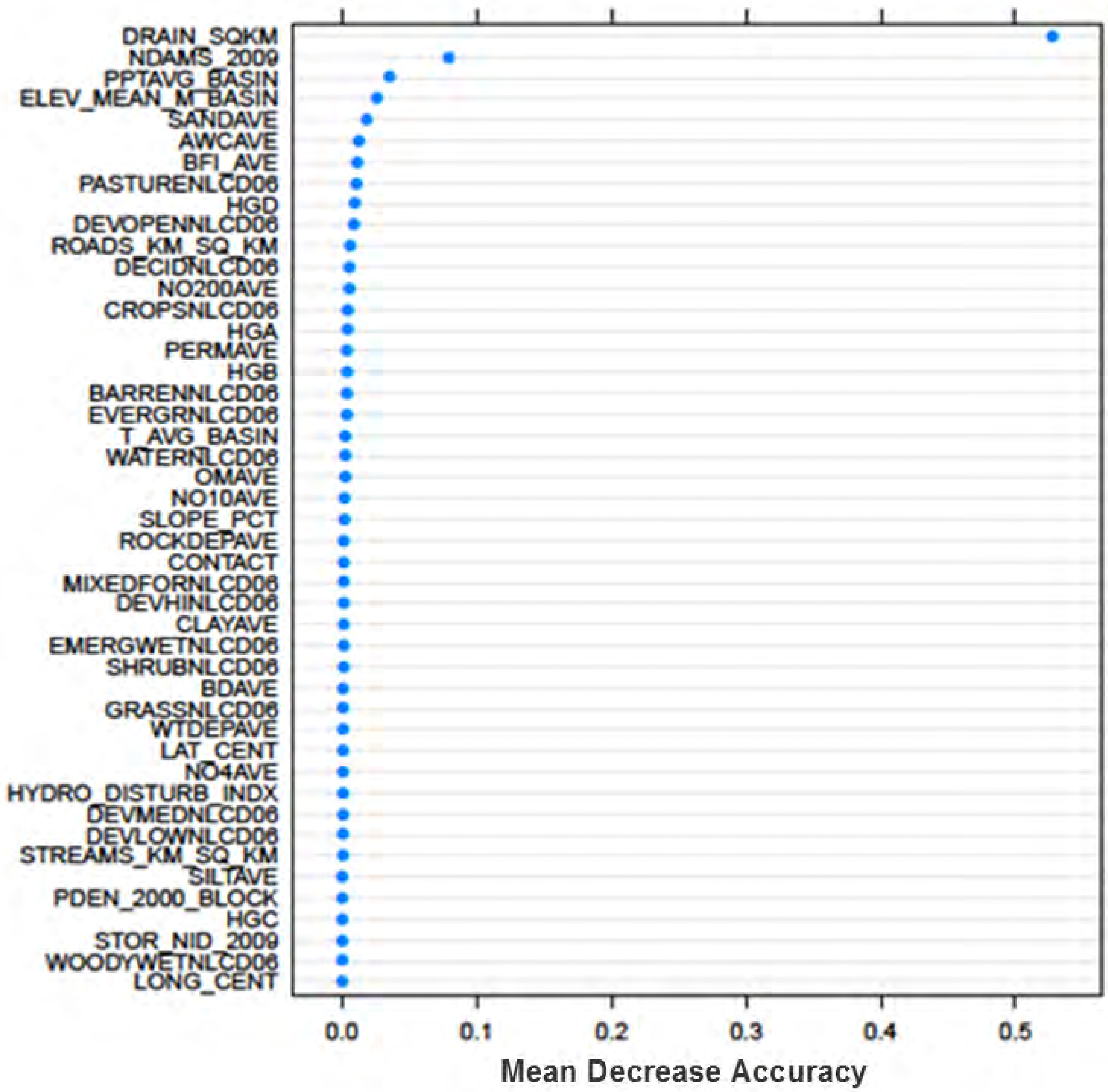

Figure D-3. Unbiased variable importance plot for the summer low-flow hydrologic sensitivity RF model

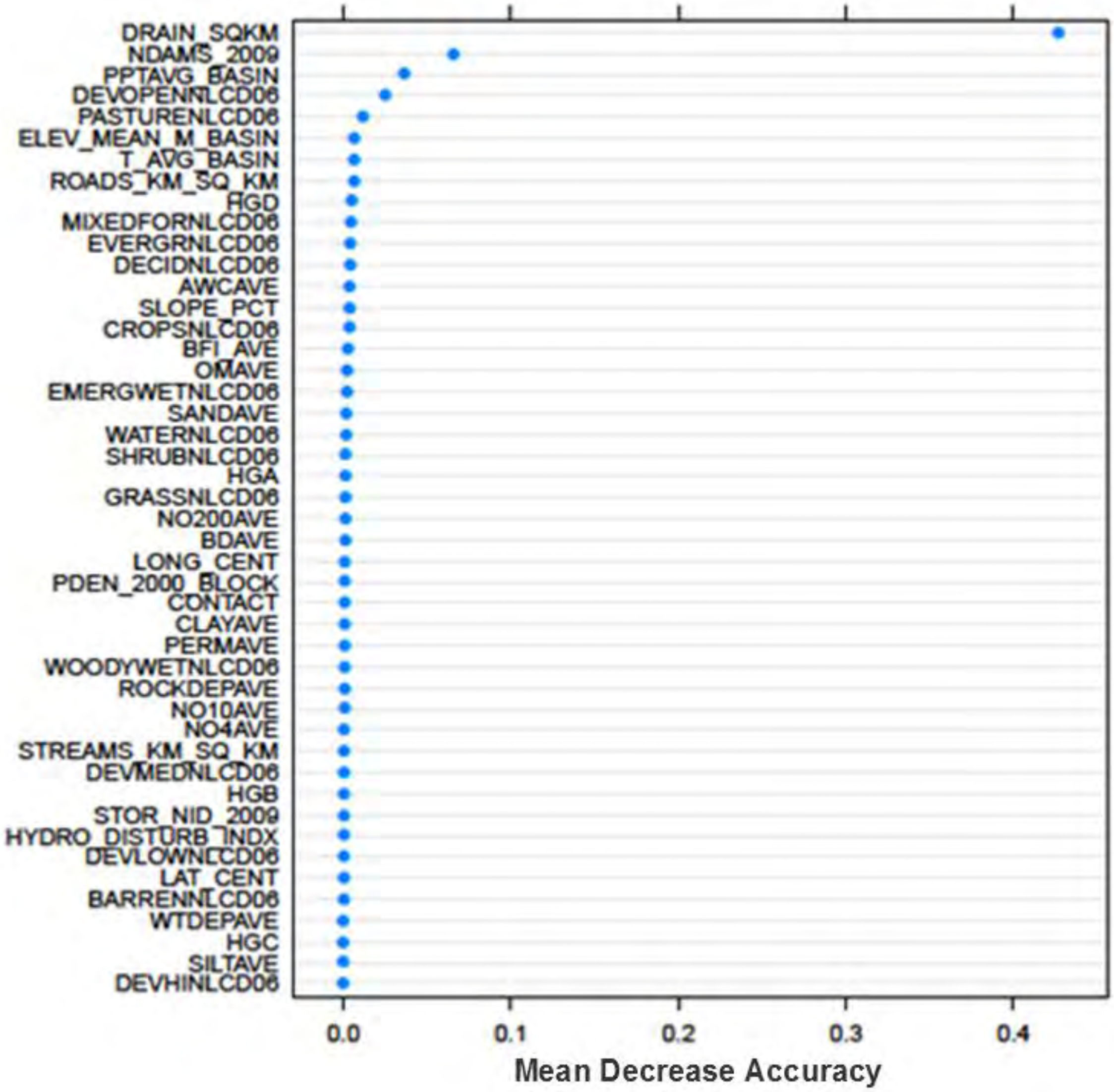

Figure D-4. Unbiased variable importance plot for the fall low-flow hydrologic sensitivity RF model.

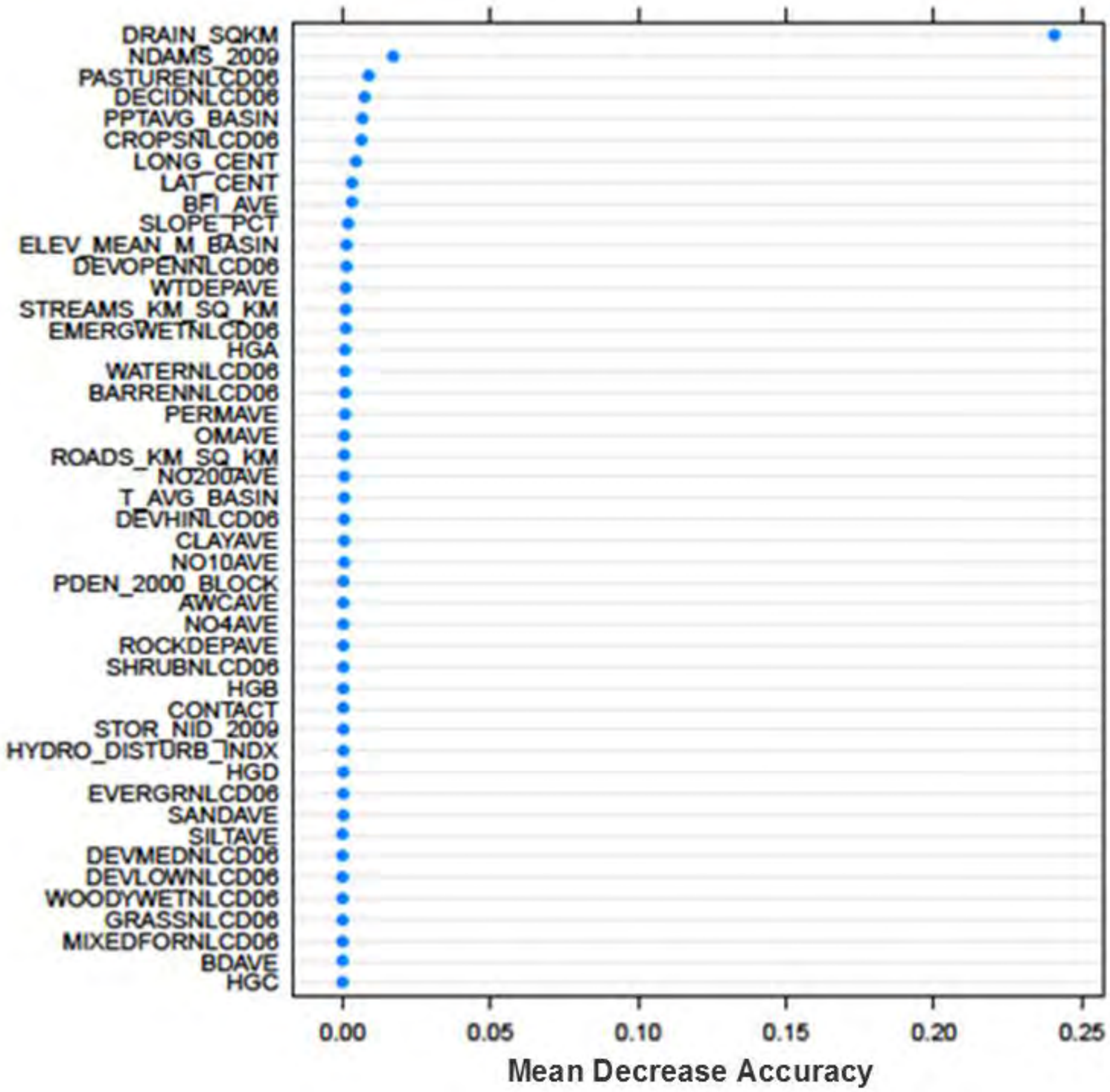

Figure D-5. Unbiased variable importance plot for the winter median-flow hydrologic sensitivity RF model.

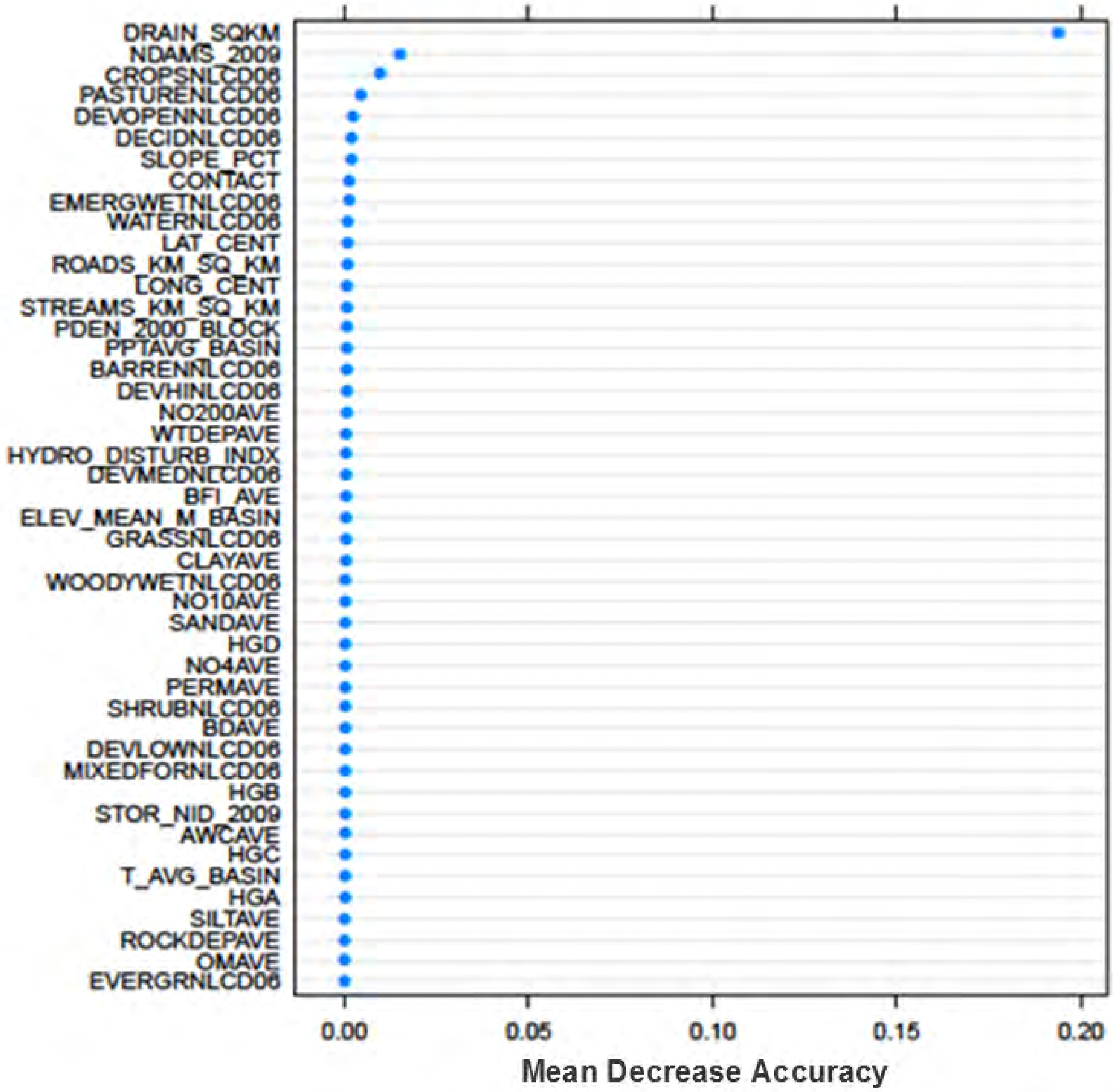

Figure D-6. Unbiased variable importance plot for the spring median-flow hydrologic sensitivity RF model.

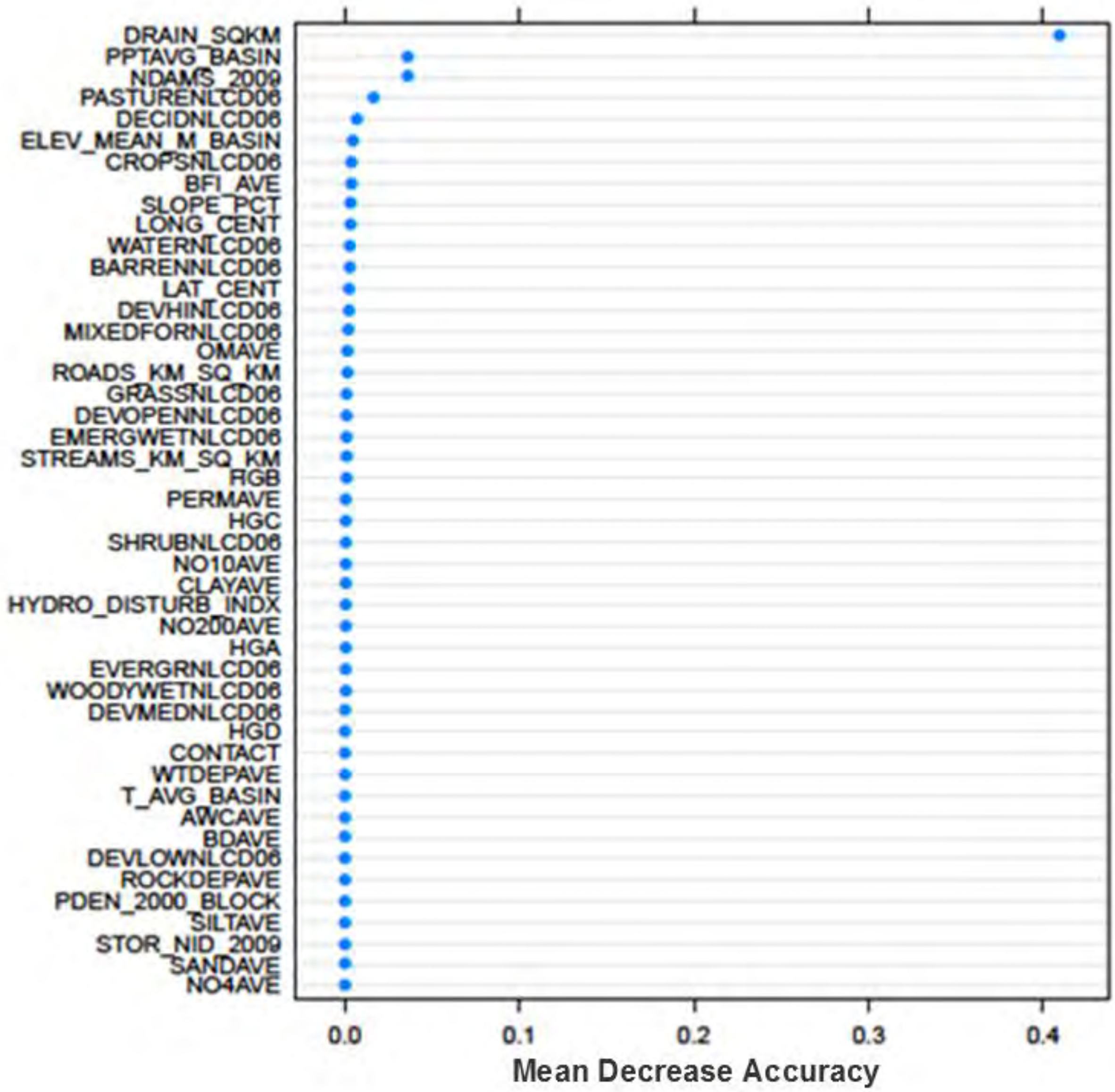

Figure D-7. Unbiased variable importance plot for the summer median-flow hydrologic sensitivity RF model.

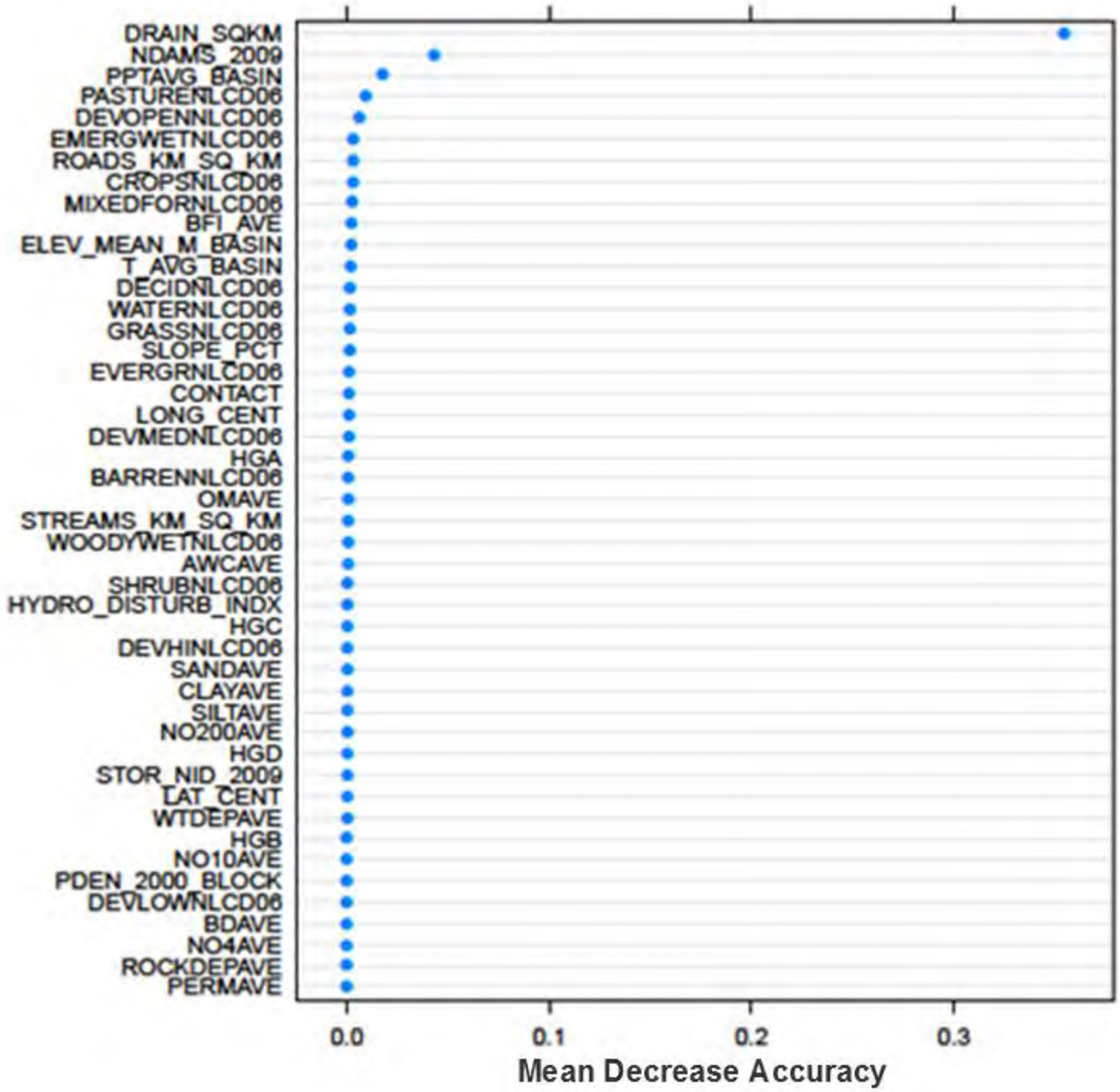

Figure D-8. Unbiased variable importance plot for the fall median-flow hydrologic sensitivity RF model.

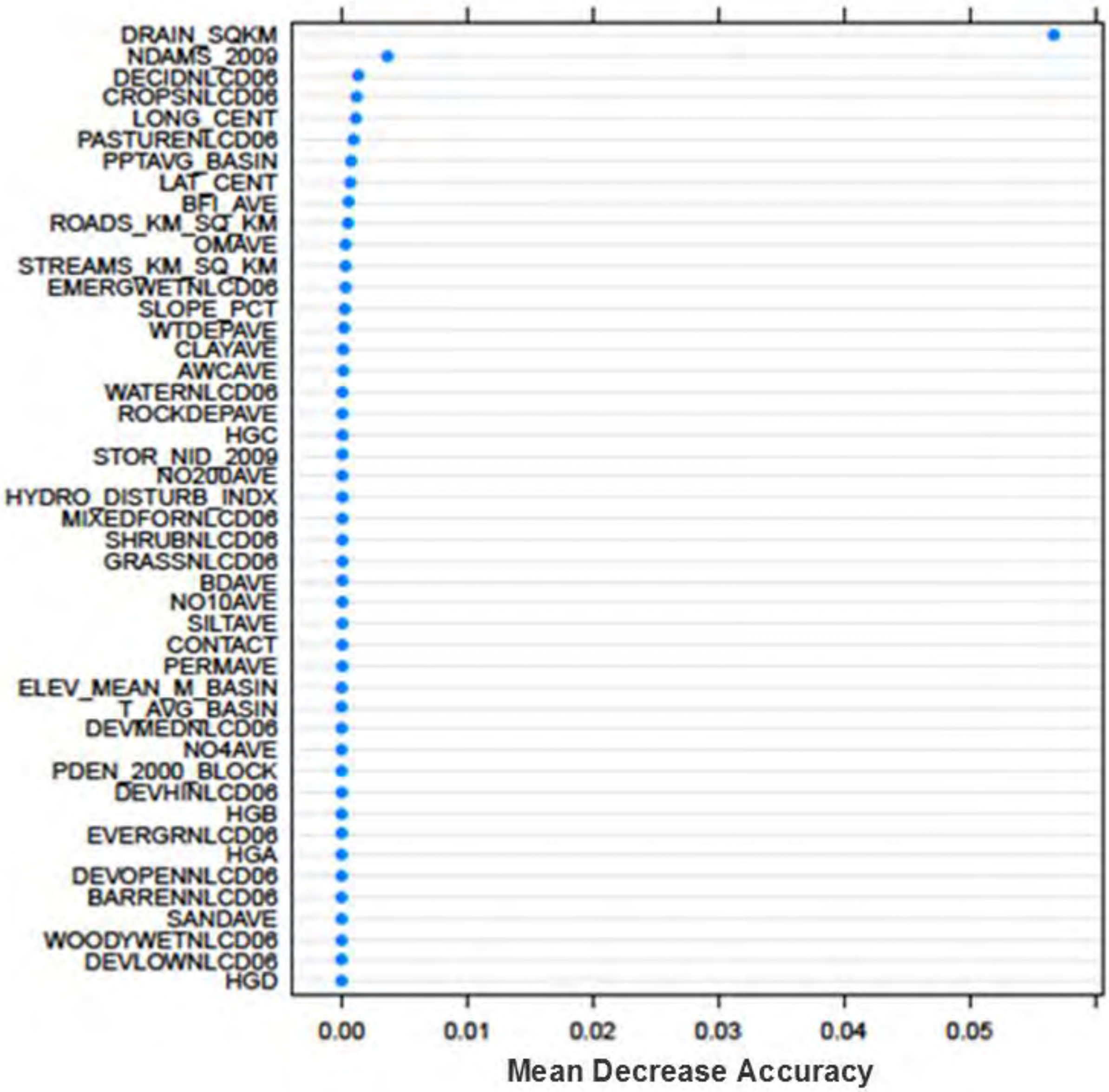

Figure D-9. Unbiased variable importance plot for the winter high-flow hydrologic sensitivity RF model.

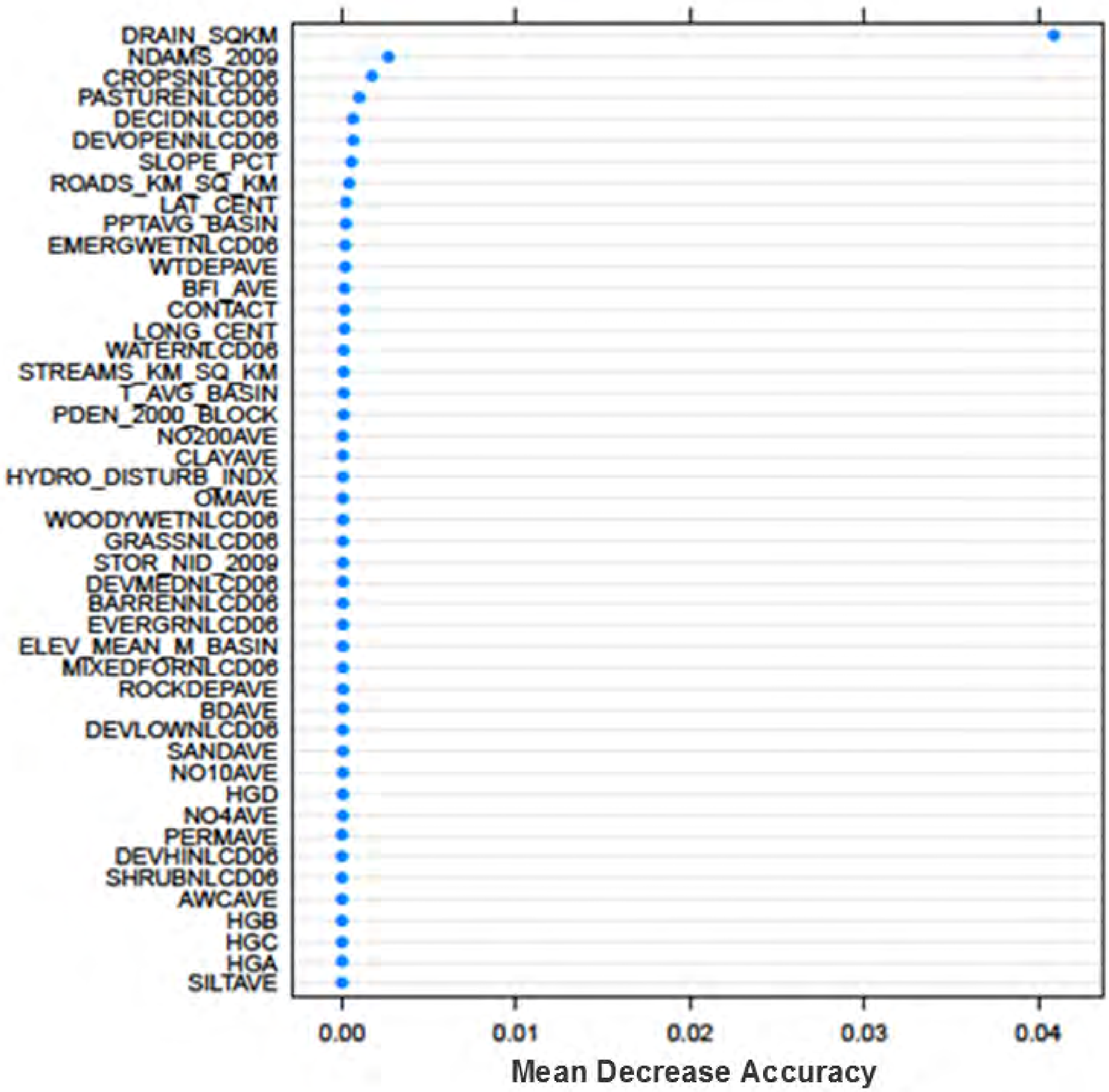

Figure D-10. Unbiased variable importance plot for the spring high-flow hydrologic sensitivity RF model.

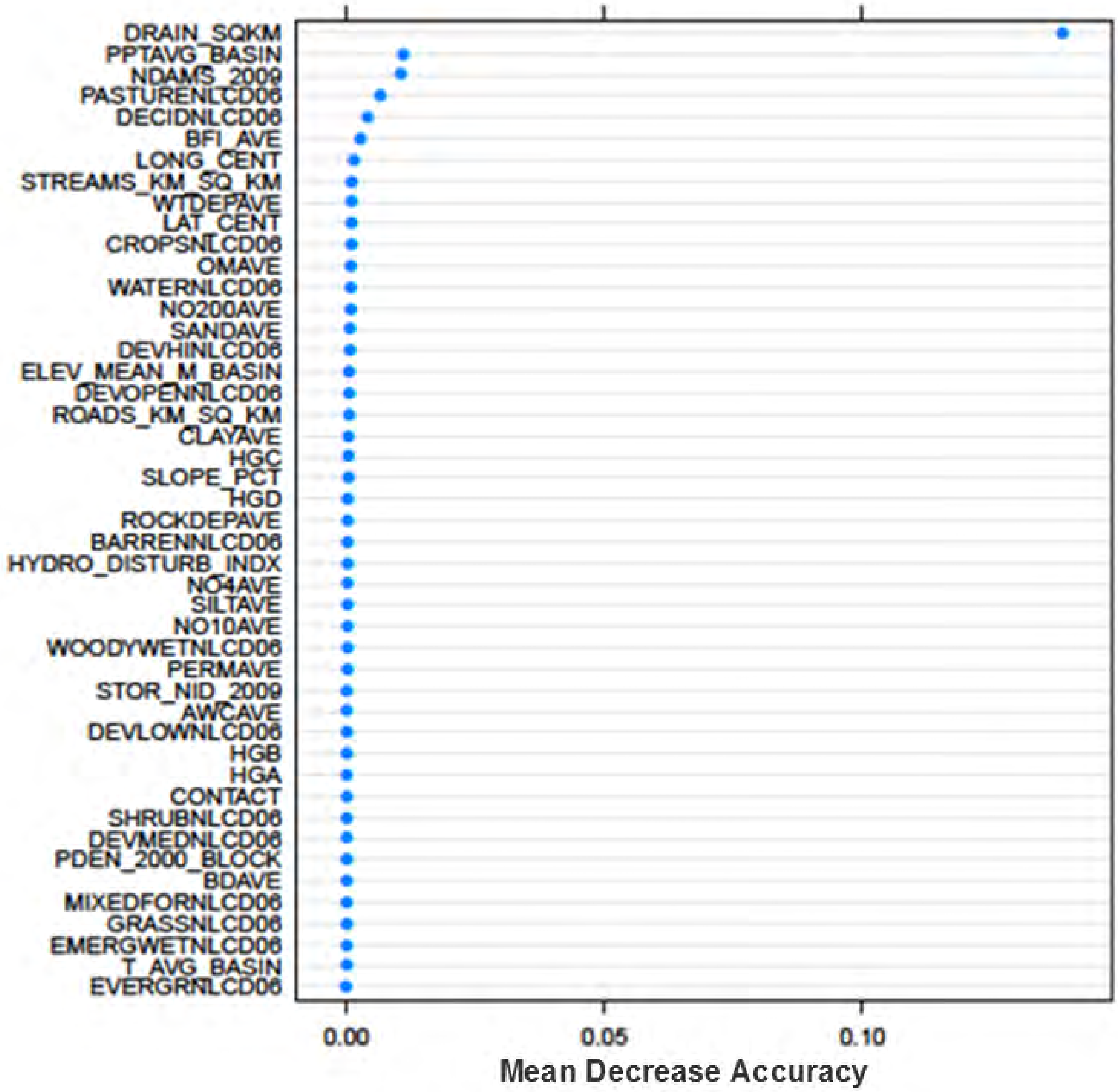

Figure D-11. Unbiased variable importance plot for the summer high-flow hydrologic sensitivity RF model.

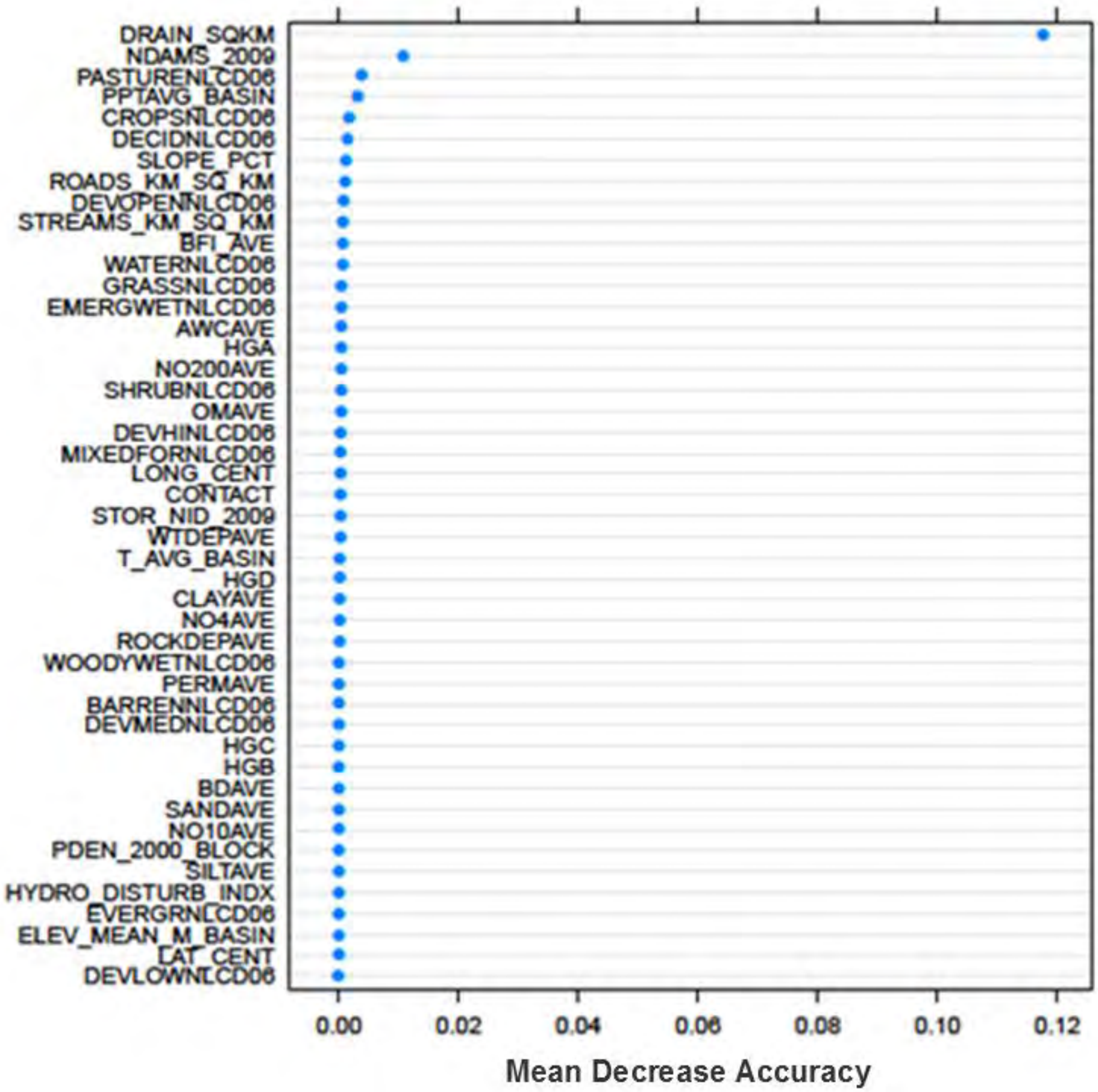

Figure D-12. Unbiased variable importance plot for the fall high-flow hydrologic sensitivity RF model.

Appendix E – Risk Maps

Local High Pumping Scenario – Low-Flows

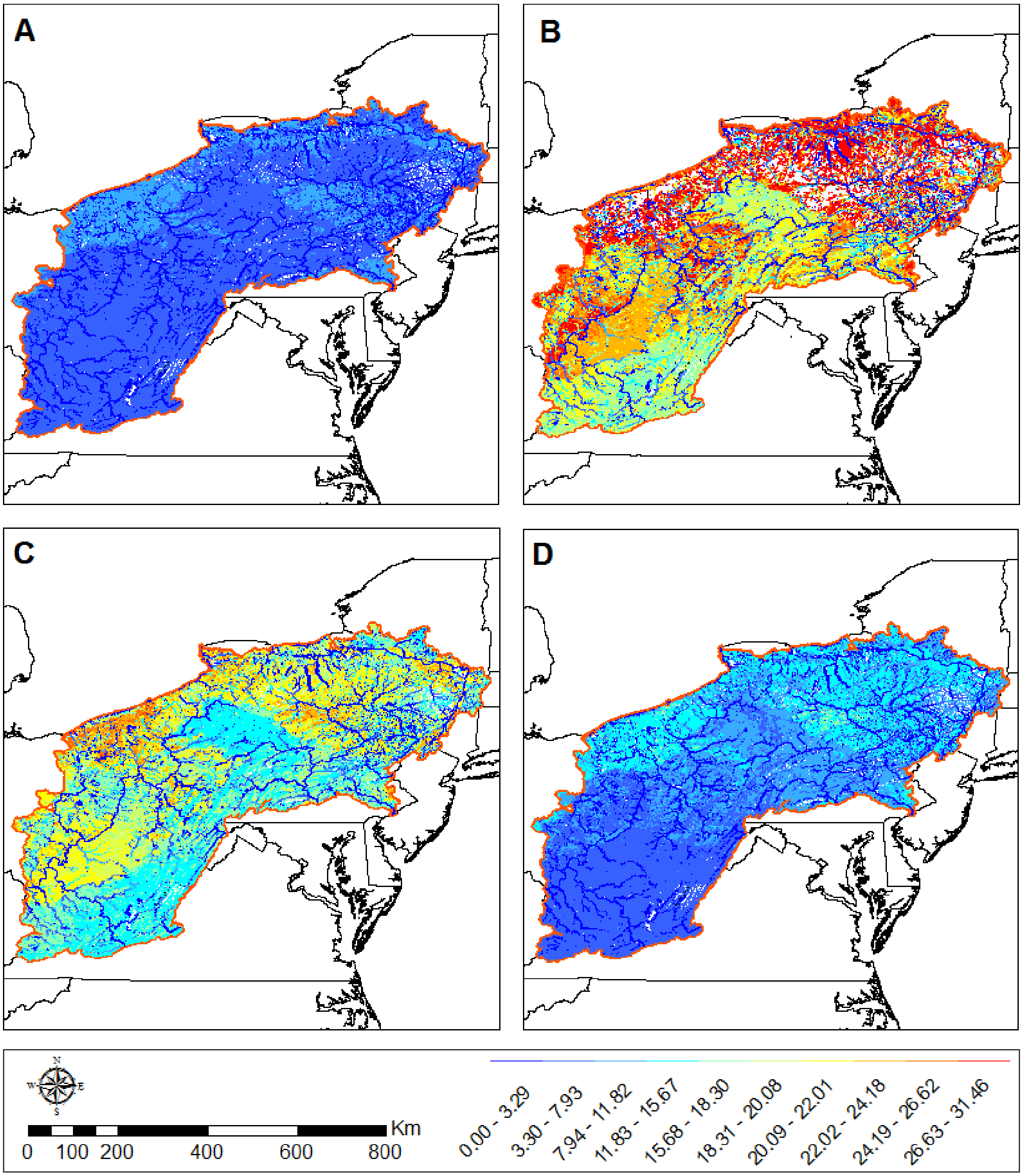

Figure E-1. Maps of hydrologic risk to low-flow from the local high pumping scenario during spring (A), summer (B), fall (C) and winter (D). Note the legend scale relative to other pumping risk figures.

Local High Pumping Scenario – Median-Flows

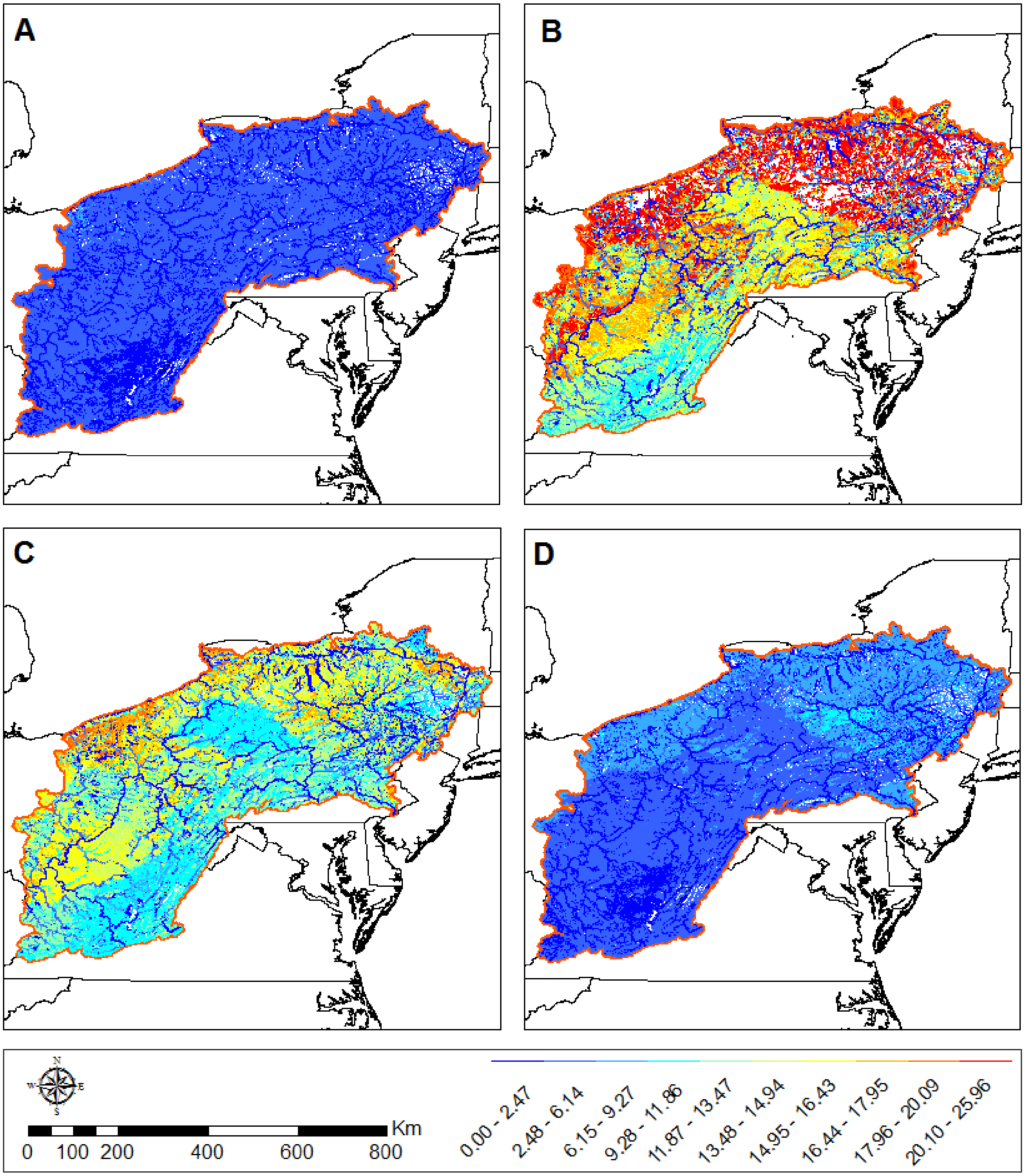

Figure E-2. Maps of hydrologic risk to median-flow from the local high pumping scenario during spring (A), summer (B), fall (C) and winter (D). Note the legend scale relative to other pumping risk figures.

Local High Pumping Scenario – High-Flows

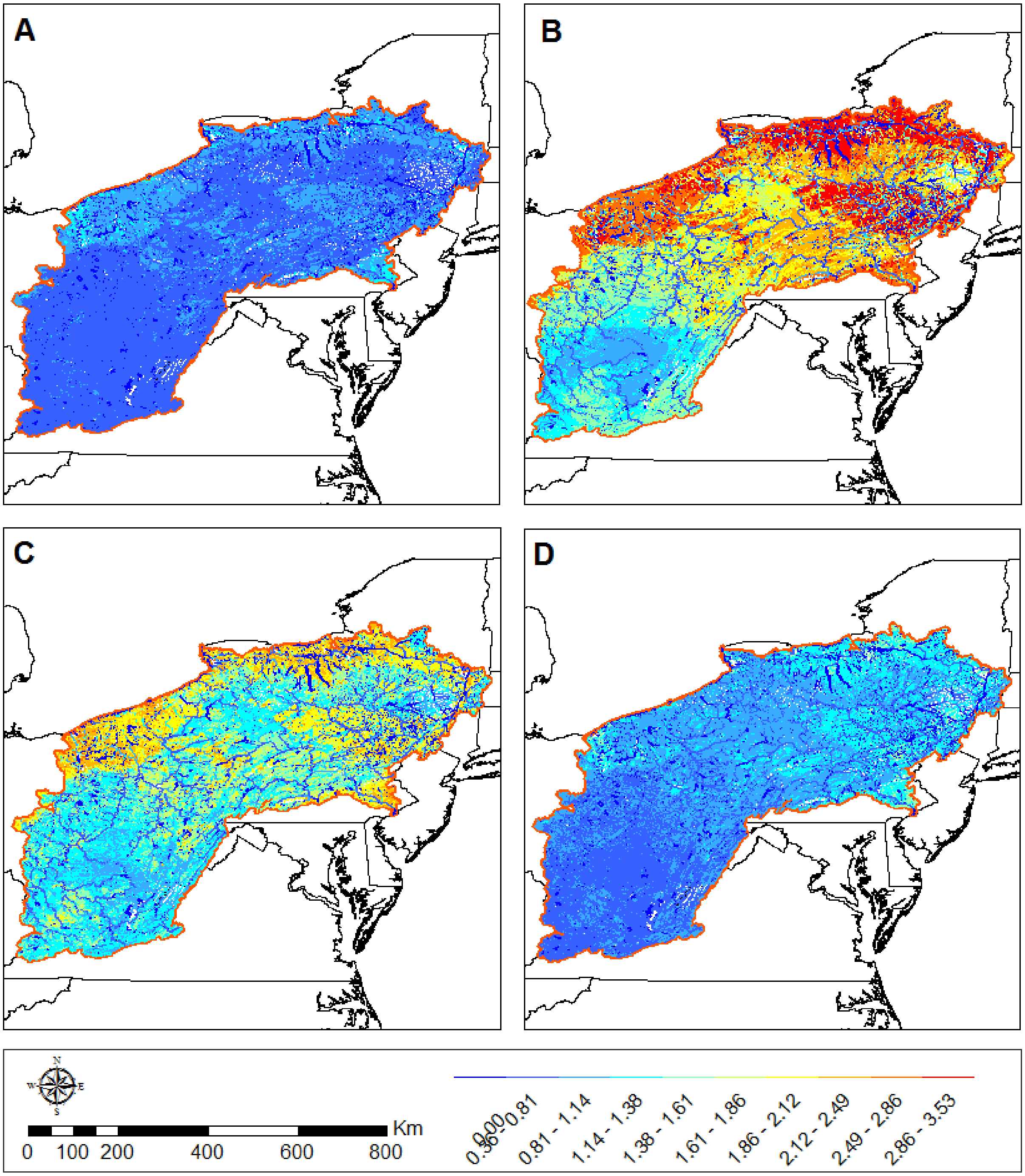

Figure E-3. Maps of hydrologic risk to high-flow from the local high pumping scenario during spring (A), summer (B), fall (C) and winter (D). Note the legend scale relative to other pumping risk figures.

Cumulative High Pumping Scenario – Low-Flows

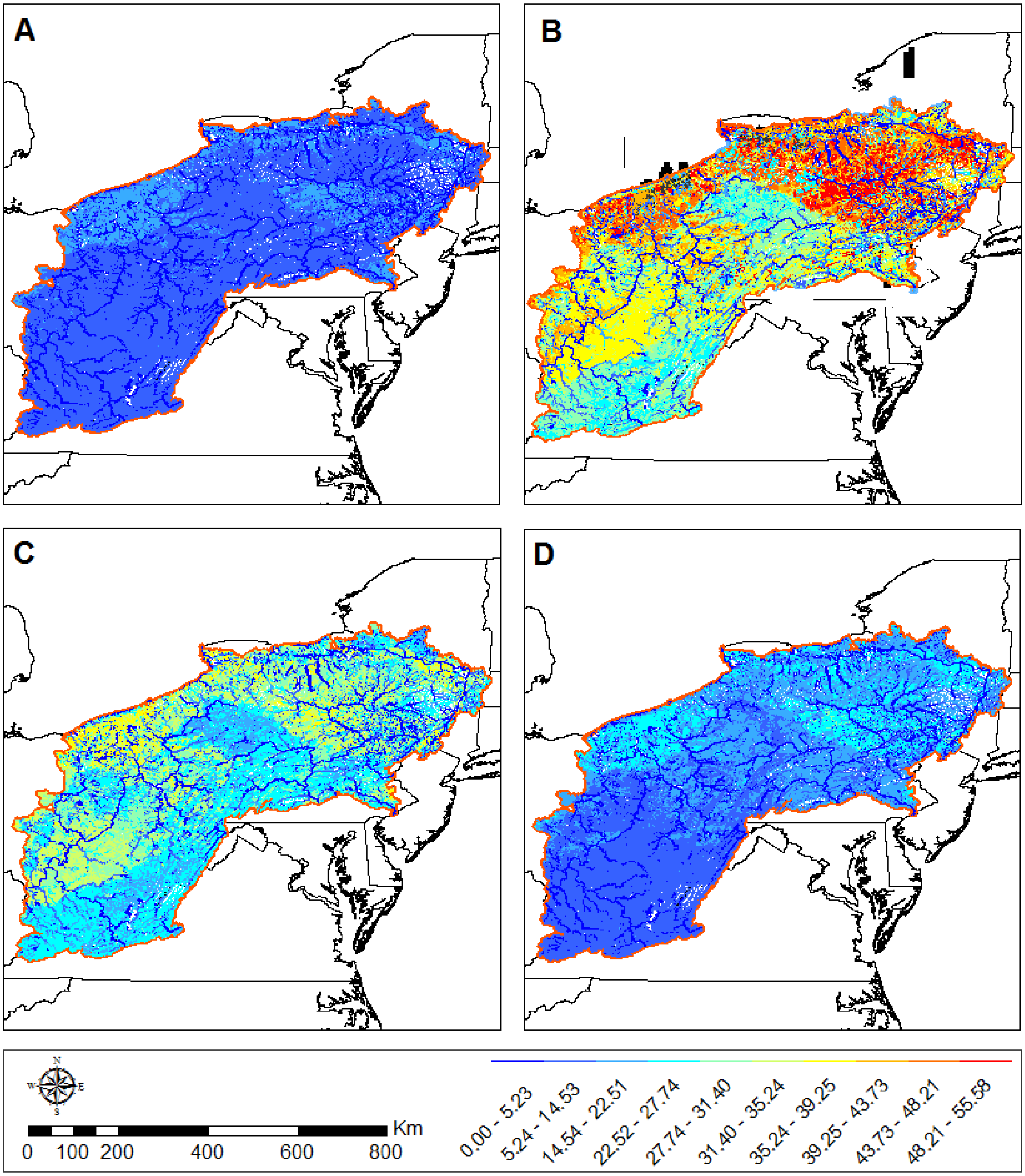

Figure E-4. Maps of hydrologic risk to low-flow from the cumulative high pumping scenario during spring (A), summer (B), fall (C) and winter (D). Note the legend scale relative to other pumping risk figures.

Cumulative High Pumping Scenario – Median-Flows

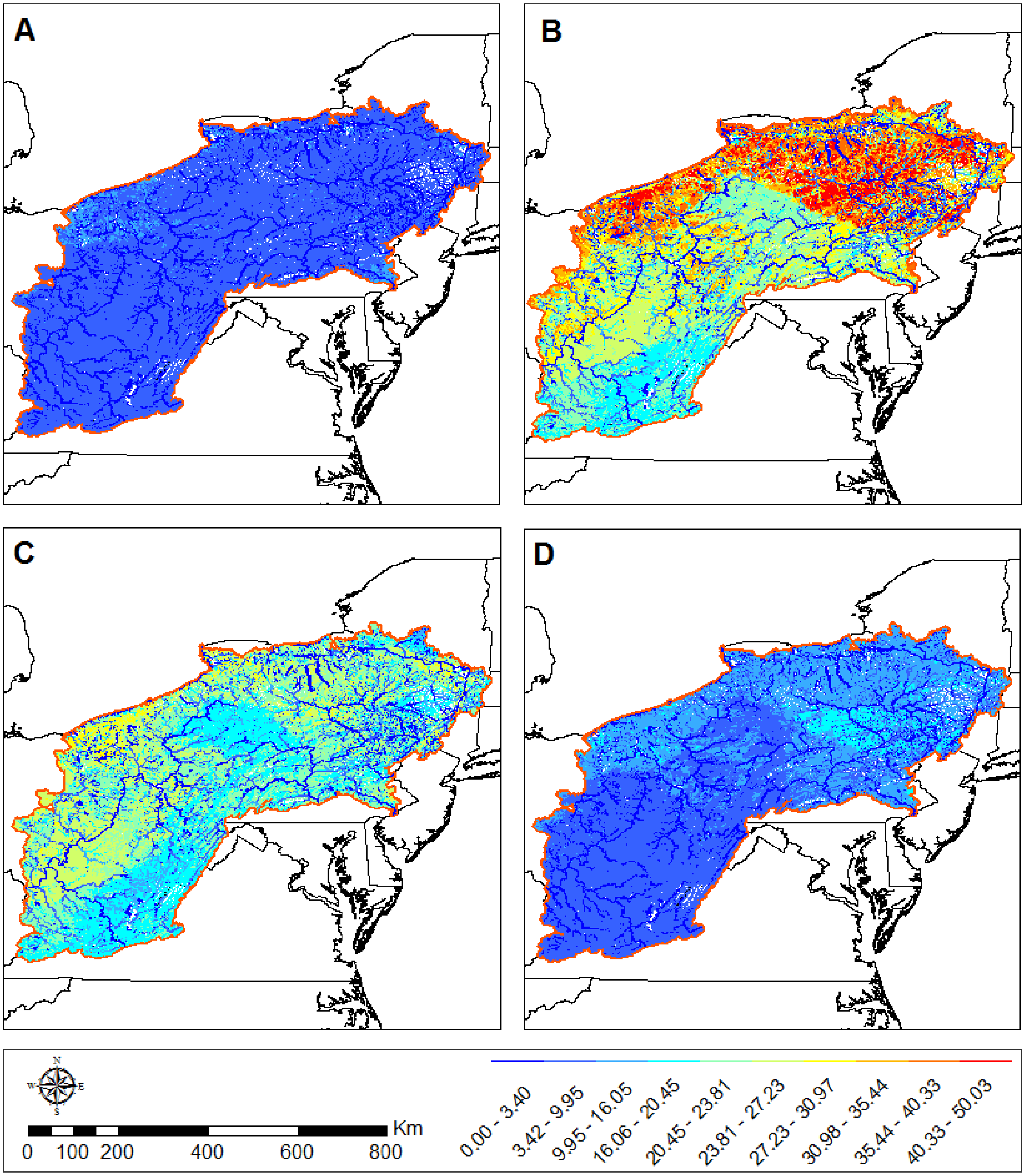

Figure E-5. Maps of hydrologic risk to median-flow from the cumulative high pumping scenario during spring (A), summer (B), fall (C) and winter (D). Note the legend scale relative to other pumping risk figures.

Cumulative High Pumping Scenario – High-Flows

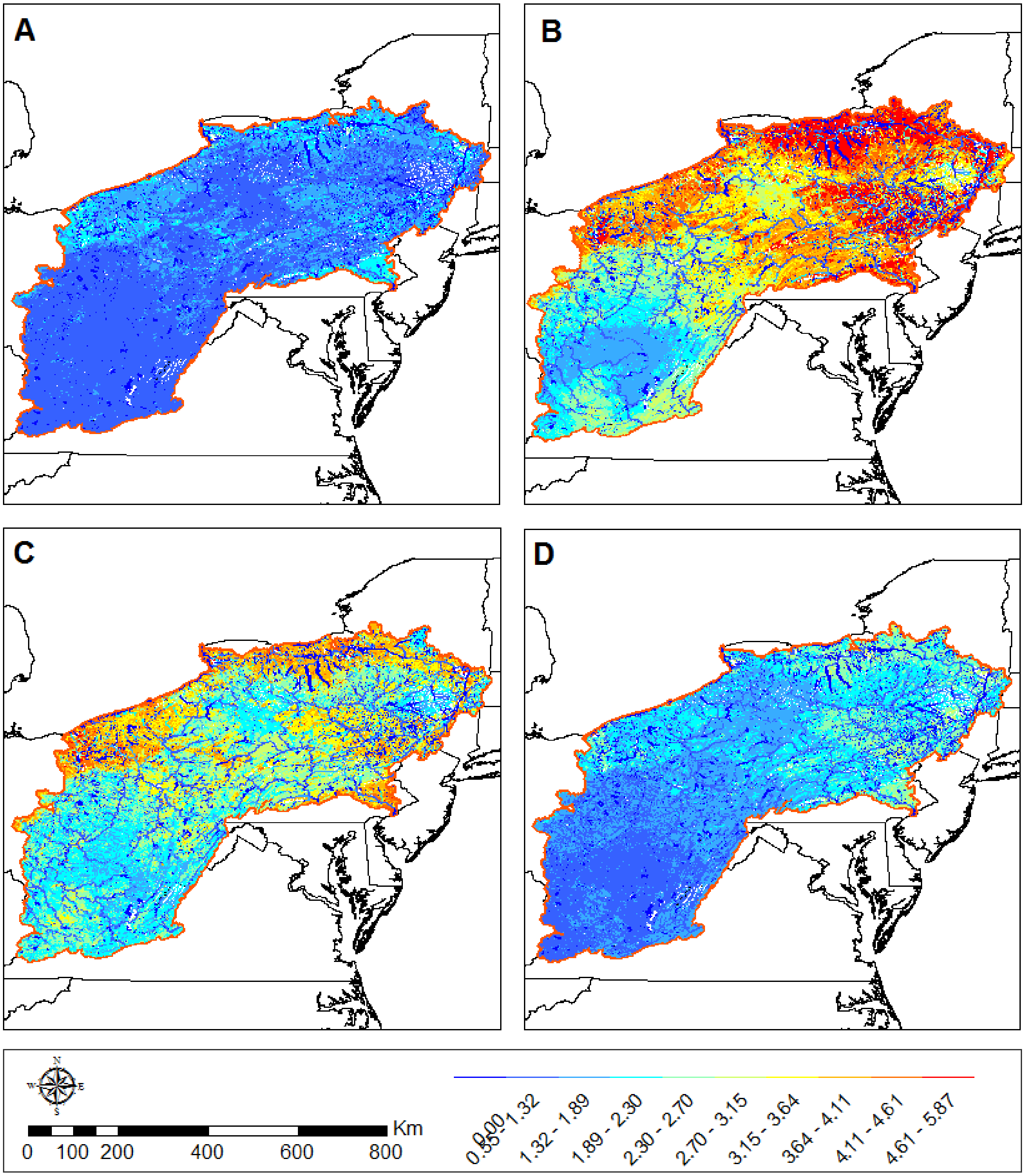

Figure E-6. Maps of hydrologic risk to high-flow from the cumulative high pumping scenario during spring (A), summer (B), fall (C) and winter (D). Note the legend scale relative to other pumping risk figures.

Appendix F – Species Distribution Models

Methods

Using the MARIS fish database, we assembled a species presence/absence matrix according to occurrences within National Hydrography Dataset (NHD V1) catchments (based on distance thresholds to stream lines). A total of 119 predictor variables were assembled for NHD catchments and represented natural characteristics, landscape disturbances, habitat fragmentation, and sampling effort. Natural characteristics included (but were not limited to) drainage area, mean annual flow, climate, gradient, soils, bedrock geology, level III Ecoregions, and hydrologic classes. Landscape disturbances included variables such as upstream dams, urbanization, agriculture, roads. Habitat fragmentation represented river length measures of fragmented habitats (bounded by up-and down-stream dams) or binary measures of whether NHD stream reaches had unobstructed flow to the ocean or Great Lakes. Because species occupancy is partially an artifact of detection probability (MacKenzie et al. 2006), we included two measures of sampling effort, one being the number of sites sampled within each reach and the second being the number of sampling occurrences within each reach.

Based on regional expertise (DePhilip and Moberg 2010, 2013), we identified seven functional trait groups of fish species, each representing hypothesized linkages between different components of the flow regime (Table 7). Hence, these functional groups are expected to respond differently to hydrologic alterations, depending on which aspects of flow regimes are disturbed. In total, we used 42 different species or guilds (i.e., combinations of species if individual species sample size was too low) to represent functional trait groups. This roughly translated to 4 to 8 species or guilds representing each functional trait group; the lamprey functional group was represented by only one guild, which consisted of all lamprey species (Ichthyomyzon or Lampetra). Random forests (Breiman 2001) were used to model presences or absences of each species or guild, which then yielded presence probabilities based on maximum values for sampling effort measures.

Probabilities of presence were then averaged to provide a measure of functional trait prevalence within each NHD reach. These were then overlain with maps of hydrologic risk, representing different aspects of flow regimes at risk from hydrologic fracturing pumping scenarios. Depending on hypothesized functional trait–flow regime linkages, catchments with higher risk were prioritized if functional trait prevalence was also high.

Results

Random forest model performance was satisfactory at predicting species or guild presence, with out-of-bag (OOB) error rates ranging from 1.3% to 23%.

Table F- 1. Out-of-bag (OOB) error rates from random forest models of species presence/absence across the Marcellus Shale region.

Species	OOB Error (%)	Species	OOB Error (%)
Shorthead Redhorse	1.31	Striped Shiner	5.02
Spotted Bass	1.49	Tessellated Darter	5.4
Silver Redhorse	1.79	Cutlip Minnow	5.59
Diadromous	1.9	Rainbow Darter	5.88
Southern Redbelly Dace	2.09	Greenside Darter	6.03
Mimic Shiner	2.71	Johnny Darter	6.85
Margined Madtom	2.79	Bluntnose Minnow	7.73
Lamprey	2.87	Creek Chub	9.13
Rosyface Shiner	3.06	Catostomus	9.28
River Chub	3.2	Fantail Darter	9.55
Mottled Sculpin	3.37	Northern Hog Sucker	9.73
Golden Redhorse	3.64	Longnose Dace	9.77
Banded Darter	3.78	Blacknose Dace	9.82
Sand Shiner	3.94	White Sucker	9.84
Nocomis	3.96	Central Stoneroller	9.92
Silverjaw Minnow	4.23	Common Shiner	9.93
Spotfin Shiner	4.43	Cottus	10.22
Redhorse	4.52	Smallmouth Bass	15.54
Fallfish	4.71	Rock Bass	15.95
Redbreast Sunfish	4.73	Brook Trout	18.07
Walleye	4.75	Brown Trout	23.15

Literature Cited

MacKenzie D.I., Nichols J.D., Royle J.A., Pollock K.H., Bailey L.L. & Hines J.E. (2006) Occupancy Estimation and Modeling: Inferring Patterns and Dynamics of Species Occurrence. Boston, MA: Academic Press, 312 pp.

Appendix G – Annotated Bibliography

Booker, D.J. & T.H. Snelder. Comparing methods for estimating flow duration curves at ungauged sites Journal of Hydrology, 434–435 (2012), pp. 78–94

Booker and Woods (2014) compared a variety of available methods for estimating several hydrological indices and flow duration curves at ungauged catchments across New Zealand. Specifically, they compared a process-based spatially distributed hydrologic model (TopNet), empirical regression models based on hydrologic theory, empirically-based random forest models and random forest corrected TopNet estimates in order to assess which method best predicted several hydrological indices given current climatic and land cover conditions. Importantly, they found that empirically-based random forest models outperformed all other methods, including the process-based spatially distributed hydrologic model. This suggests that applying a statistical approach in the Marcellus Shale Region would prove more effective.

Buchanan, C., Moltz, H. L. N., Haywood, H. C., Palmer, J. B. & Griggs, A. N. A test of The Ecological Limits of Hydrologic Alteration (ELOHA) method for determining environmental flows in the Potomac River basin, U.S.A. Freshw. Biol. 58, 2632–2647 (2013).

The only peer-reviewed example of a process-based hydrologic model being applied across a large basin for the purposes of determining environmental flows following an ELOHA-style framework was that of Buchanan et al. (2013). In this study, the authors applied the Chesapeake Bay Program Hydrologic Simulation Program–FORTRAN (HSPF) model and the Virginia Department of Environmental Quality Online Object Oriented Meta-Model (WOOOMM) routing module to the Potomac River Basin.

They found that the combined HSPF-WOOOMM model failed to properly simulate streamflow in smaller urbanized basins or on or near karst geology. In addition, Nash-Sutcliffe efficiencies, ranged from 0.33 to 0.82, indicating a very wide range of model performance (i.e. very poor to good). We should emphasize that this study likely represents a best case scenario in terms of data availability and parameterization. For instance, the study was conducted in the Chesapeake Bay Watershed, which has been the subject of intensive study for many decades. Through the combined efforts of numerous non-profit organizations and state and federal agencies, an extensive database of information necessary for a well parameterized model has been amassed. Furthermore, the HSPF-WOOOMM model was expressively designed and calibrated for the Chesapeake Bay Watershed. Even under these relatively ideal conditions, the process-based model yielded results of questionable utility in many of the modeled catchments. This is in accordance with the result of our SWAT modeling.

Carlisle, D. M., Falcone, J., Wolock, D. M., Meador, M. R. and Norris, R. H.: PREDICTING THE NATURAL FLOW REGIME : MODELS FOR ASSESSING HYDROLOGICAL ALTERATION IN STREAMS , River Research and Applications, 136, 118–136, doi:10.1002/rra, 2010.

This paper investigated the ability of statistical models developed using random forest modeling at national and regional scales to correctly predict 13 flow indices. The authors found that the random forest-based national and regional scale models performed equally well and outperformed landscape stratification models, which were based on classifications such as ecoregions and major river basins. The authors assert that such models can be applied to accurately predict natural flow regimes at ungaged catchments and that they are sensitive to long-term land use change.

Carlisle, D. M., Wolock, D. M. & Meador, M. R. Alteration of streamflow magnitudes and potential ecological consequences: a multiregional assessment. *Frontiers in Ecology and the Environment* **9,** 264–270 (2011).

This paper presents a national scale analysis of 2888 streamflow monitoring sites in the U.S. The authors detected changes in the magnitudes of mean annual, minimum and maximum streamflows. A second analysis conducted on a subset of these stream gages suggested that reduced flow magnitudes were the primary predictors of biologic integrity for fish and aquatic insect communities.

DePhilip, M. & Moberg, T. Ecosystem Flow Recommendations for the Susquehanna River Basin. The Nature Conservancy. Harrisburg, PA. 192 (2010).

This document presents a set of flow recommendations for the Susquehanna River Basin developed by The Nature Conservancy for the Susquehanna River Basin Commission (SRBC) and the U.S. Army Corps of Engineers (USACE). The flow recommendations address the full range of ecologically relevant flow conditions (i.e. low and high flows, seasonal flows, etc.) across the suite of characteristics that comprise the “natural flow regime” (i.e. timing, magnitude, frequency and duration of flows). The ultimate goal of the report is to provide key guidance for the establishment of flow limitations for water withdrawals within the Susquehanna River basin that minimize ecological impacts of consumptive water use – especially during critical low flow periods. The authors made use of existing field data, hydrologic analyses, published literature and expert opinion to develop their recommendations. Additionally, the flow recommendations were devised taking into account a wide range of aquatic and terrestrial biota, including: birds, mammals, riparian and aquatic vegetation, reptiles and amphibians, fish, mussels and aquatic macroinvertebrates. The resulting flow recommendations are summarized as follows:

High flows

For all streams and rivers

- Maintain magnitude and frequency of 20-yr (large) flood
- Maintain magnitude and frequency of 5-yr (small) flood
- Maintain magnitude and frequency of 1 to 2-yr high flow (bankfull) event
- Limit the change to the monthly Q10 to less than 10%
- Maintain the long-term frequency of high pulse events during summer and fall

Seasonal flows

For all streams and rivers

- Maintain the long-term monthly median between the 45th and 55th percentiles
- Limit change to "typical monthly range" to less than 20%

Low flows

For all streams and rivers with drainage areas greater than 50 square miles

- Limit change to "monthly low flow range" to less than 10%
- Maintain the long-term monthly Q95

For headwater streams with drainage areas less than 50 square miles

- Maintain the long-term "monthly low flow range"
- Maintain the long-term monthly Q75

DePhilip, M. & Moberg, T. Ecosystem Flow Recommendations for the Upper Ohio Ecosystem Flow Recommendations for the Upper Ohio River Basin in Western Pennsylvania. The Nature Conservancy. Harrisburg, PA. 193 (2013).

The Nature Conservancy adapted the methodology used in the Ecosystem Flow Recommendations for the Susquehanna River Basin report (DePhilip and Moberg, 2010) to the Upper Ohio River Basin. The Conservancy utilized a series of workshops attended by experts in hydrology, water quality, and aquatic and terrestrial ecology to construct a series of hypotheses regarding flow-ecology relationships and develop flow recommendations.

Entrekin, S., Evans-White, M., Johnson, B. and Hagenbuch, E.: Rapid expansion of natural gas development poses a threat to surface waters, Frontiers in Ecology and the Environment, 9, 503–511, 2011.

This study highlights ecological threats that natural gas extraction poses to aquatic biota. The authors explore a host of potential impacts, including increased sediment loads from road runoff and pipeline construction, flow regime alteration from surface water pumping activities, and water quality degradation through the introduction of hydraulic fracturing chemicals and/or flowback water. The paper concludes that our understanding of these potential effects is currently lacking and that additional study is needed in order to ensure appropriate management policies are developed.

Falcone, J., Carlise, D., Wolock, D. & Meador, M. GAGES : A stream gage database for evaluating natural and altered flow conditions in the conterminous United States. Ecology 91, 621 (2010).

This paper describes the construction of the GAGESII database, which contains several hundred watershed and site characteristics associated with 6,785 USGS stream gages. The attributes were calculated or compiled from national data sources and include environmental features (e.g., climate, geology, soils, topography) and anthropogenic influences (e.g., land use, roads, presence of dams, or canals). The USGS gages were also classified into reference and non-reference groups based on their level of anthropogenic disturbance.

Henriksen JA, Heasley J, Kennen JG, Niewsand S. 2006. Users' manual for the hydroecological integrity assessment process software (including the New Jersey Assessment Tools): U.S. Geological Survey, Biological Resources Discipline, Open File Report 2006-1093, 71.

This program facilitates the calculation of HIT indices used in this study.

Kennard, M. J., Mackay, S. J., Pusey, B. J., Olden, J. D. & Marsh, N. Quantifying uncertainty in estimation of hydrologic metrics for ecohydrological studies. *River Research and Applications* **26,** 137–156 (2010).

This study provides critical guidance regarding the effect of discharge record length and time period of record on uncertainty in the calculation of 120 commonly used hydrologic metrics. The authors conclude that: 1) hydrologic indices should be calculated based on a minimum of 15 years of discharge data and 2) discharge records should have considerable overlap (ideally >50%).

McManamay, R. A., Orth, D. J., Dolloff, C. A. & Frimpong, E. A. A regional classification of unregulated stream flows: spatial resolution and hierarchical frameworks. *River Research and Applications* **28,** 1019–1033 (2012).

Using 66 hydrologic indices, this study classified 292 streams across an eight state region of the Southeastern U.S. The authors identified a total of six stream types within the study region and also provide recommendation for a reduced set of hydrologic indices based on a classification tree analysis. Additionally, the study found that flow classification schemes are sensitive to the spatial resolution of the analysis.

McManamay, R., Orth, D. J., Dolloff, C. & Mathews, D. C. Application of the ELOHA framework to regulated rivers in the Upper Tennessee River Basin: a case study. *Environmental management* **51,** 1210–35 (2013).

This study applied the ELOHA framework to inform flow restoration recommendations in the Upper Tennessee River Basin. The authors constructed univariate flow-ecology

relationships and compared their predictive ability to that of multivariate flow-ecology models. Results suggest that the univarate models were outperformed by the multivariate models in terms of providing guidance for flow restoration in regulated rivers. The multivariate models indicated an inverse relationship between flow magnitude and riparian encroachment – and further, that alterations in substrate, stream temperature and the disturbance regime may reduce fish colonization.

McManamay, R. A., Orth, D. J., Kauffman, J., Mary, M. & Davis, M. M. A Database and Meta-Analysis of Ecological Responses to Stream Flow in the South Atlantic Region A Database and Meta-Analysis of Ecological Responses to Stream Flow in the South Atlantic Region. **12,** 1–36 (2013).

Empirical and theoretical flow- ecology relationships were compiled from numerous studies in the South Atlantic region (SAR) of the U.S. The authors found that ecological responses to natural source of flow alteration were highly variable and difficult to generalize. However, they found consistent negative relationships between ecology (fish abundance, diversity, reproduction and diversity) and anthropogenic sources of flow alteration. Some ecological responses (aquatic insects and riparian vegetation) were inconsistent and in some cases exhibited a positive response to flow alteration (algal abundance). Importantly, the authors also found that developing flow-ecology relationships at a regional scale is challenging and suggest instead that the relationships are far more meaningful when stratified into specific flow categories or by geomorphic setting.

Mobley, J. T., Culver, T. B. and Burgholzer, R. W.: Environmental Flow Components for Measuring Hydrologic Model Fit during Low Flow Events, Journal of Hydrologic Engineering, (December), 1325–1332, doi:10.1061/(ASCE)HE.1943-5584.0000575., 2012.

This study compared the performance of two different methods for estimating indicators of hydrologic alteration (IHA): 1) a simple drainage area ratio (DAR) technique and 2) lumped parameter hydrologic model developed for the Chesapeake Bay region (Chesapeake Bay Program Phase 5 Model). IHAs were calculated using both methods and results were compared with IHAs calculated from observed data. The authors found that the simple DAR method characterized low-flow IHAs better than the far more complex, difficult to parameterize and calibrate Chesapeake Bay Program Model.

NHDPlus User Guide. United States Environmental Protection Agency and United States Geological Survey. <ftp://ftp.horizon-lsystems.com/NHDPlus/documentation/NHDPLUS_UserGuide.pdf>. 2008.

The National Hydrography Dataset is a data-rich This database contains georeferenced national hydrography for the US, including stream networks, watershed boundaries, headwater nodes, cumulative drainage area characteristics, flow direction and accumulation grids and flow volume and velocity estimates for all stream segments in the network. It is particularly useful as a source of physical basin characteristics for use in predictive statistical models.

Olden, J. D. & Poff, N. L. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. *River Research and Applications* **19,** 101–121 (2003).

This paper evaluates 171 hydrologic indices to provide guidance for researchers who must choose metrics that will minimize computational effort and reduce redundancy and multicollinearity. The authors also explore the transferability of the recommended indices over different stream types in order to ensure accurate flow regime characterization across different geological and climatic environments.

Poff, N. L., Richter, B. D., Arthington, A. H., Bunn, S. E., Naiman, R. J., Kendy, E., Acreman, M., Apse, C., Bledsoe, B. P., Freeman, M. C., Henriksen, J., Jacobson, R. B., Kennen, J. G., Merritt, D. M., Keeffe, J. H., Olden, J. D., Rogers, K., Tharme, R. E. and Warner, A.: The ecological limits of hydrologic alteration (ELOHA): a new framework for developing regional environmental flow standards, Freshwater Biology, 55(1), 147–170, doi:10.1111/j.1365-2427.2009.02204.x, 2010.

This is the seminal paper which first outlined the ecological limits of hydrologic alteration (ELOHA) framework. The authors synthesize a number of pre-existing environmental flow analysis techniques into a cohesive framework for regional flow management. The method involves first building a hydrologic foundation consisting of baseline flow patterns for relevant streams in the area of interest. Secondly, streams area classified into flow regime types based on ecologically relevant flow variables. Third, the degree of hydrologic alteration is calculated as the difference in baseline vs. current flow metrics. Finally, a set of flow-ecology relationships are constructed for each of the stream types identified in the second step. The authors also emphasize the importance of acknowledging uncertainty in flow-ecology relationships and recommend applying the ELOHA method in a "consensus context where stakeholders and decision-makers explicitly evaluate acceptable risk as a balance between the perceived value of the ecological goals, the economic costs involved and the scientific uncertainties in functional relationships between ecological responses and flow alteration".

Richter, B. D., Davis, M. M., Apse, C. & Konrad, C. Short communication: Presumptive standard for environmental flow protection. *River Research and Applications* **28,** 1312–1321 (2012).

This paper points out that while significant progress has been made in the field of environmental flow protection, it is unlikely that the newly developed techniques will be successfully applied to most rivers in the U.S. and especially in more data-scarce regions around the world. The authors suggest that this will leave most rivers unprotected from flow alteration and argue for the adoption of a "presumptive standard" based on the Sustainability Boundary Approach of Richter (2009). They go one to discuss the management implications of their proposed approach.

Sanderson, J. S. *et al*. Getting to scale with environmental flow assessment: the watershed flow evaluation tool. *River Research and Applications* **28,** 1369–1377 (2012).

This study presents the development of the Watershed Flow Evaluation Tool (WFET) to "estimate flow-related ecological risk in the state of Colorado. The model was applied to two watersheds with differing data availability. The WFET successfully applied to assess ecological risk associated with flow alteration in one of the study watersheds. However, in the other watershed, active channel erosion and bed degradation prevented successful application of the tool. Despite the limited success for the tool, the authors conclude that it is appropriate for evaluating ecological risk associated with anthropogenic flow alteration.

Shrestha, R. R., Peters, D. L. and Schnorbus, M. A.: Evaluating the ability of a hydrologic model to replicate hydro-ecologically relevant indicators, Hydrological Processes, doi:10.1002/hyp.9997, 2013.

The authors used the Variable Infiltration Capacity (VIC) hydrologic model in two headwater catchments in the Fraser River, British Columbia, Canada to evaluate whether the mode was able to accurately simulate a suite of water resource indicators (WRIs) and indicators of hydrologic alteration (IHAs). The VIC model yielded mixed results – correctly simulating some WRIs and IHAs, but demonstrated statistically significant differences in modeled and observed WRIs and IHAs. The authors go on to point out specific model issues which contributed to discrepancies in modeled and observed flow statistics (e.g. model input/output data) and emphasize caution when using model-derived flow indicators.

USACE, U.S. Army Corps of Engineers, The Nature Conservancy, and Interstate Commission on the Potomac River Basin. 2013. Middle Potomac River Watershed Assessment: Potomac River Sustainable Flow and Water Resources Analysis. Final Report. 144 pp. and 10 appendices.

This report provides an excellent summary of efforts of the U.S. Army Corps of Engineers, The Nature Conservancy, and Interstate Commission on the Potomac River Basin to conduct a Sustainable Flow and Water Resources Analysis for the Potomac River Basin. It details their collaborative to determine the "relationship between streamflow alteration and ecological response in the Potomac River and its tributaries". The assessment is divided into five sub-groups, including i) a large river environmental flow needs assessment, ii) a stream and small rivers environmental flow needs

assessment, iii) a projection of future water uses, iv) a stakeholder engagement process, and v) development of a concept or scope for a strategic comprehensive plan for watershed management. The analysis was further broken up into two separate flow assessment strategies: i) large rivers were evaluated using the Ecologically Sustainable Water Management (ESWM) approach, and ii) ELOHA was used for streams and small rivers. Importantly, they found that "in the large rivers included in this study, based on currently available information, there has been no discernible adverse ecological impact on focal species due to human modification of flows". This suggests that smaller rivers and streams should be the focus of the Marcellus Shale study - and indeed, these would be the most sensitive to hydraulic fracturing related water withdrawals. "As a precautionary measure, the team did recommend that the current large river flow regime be maintained for the entire range of flows as defined by 20 flow statistics based on a 21-year period of record (1984-2005)". They did find that small stream and rivers were quite sensitive to hydrologic alteration resulting from urbanization (e.g. increases in impervious surface). Interestingly, land use change was found to be a stronger cause for hydrologic alteration than water withdrawals and impoundments. The team used macroinvertebrate metrics as their ecological endpoints for measuring degradation. They found strong relationships between increase flashiness and decrease indices of biotic integrity, but little response from changes to low flow magnitudes. They also conducted a scenario analysis which evaluated the effects of: i) three different forecasts of per capita domestic water use, ii) climate change, iii) hot and dry summer conditions, and iv)conversion of power plants to closed cycle operation. They found no regional pattern of flow alteration applied to all scenarios and, "within scenarios, impacts on flow varied for each subwatershed's unique combination of land and water uses".

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