Developing Instream Flow Criteria to Support Ecologically Sustainable Water Resource Planning and Management

Final Report to the Pennsylvania Instream Flow Technical Advisory Committee
Project funded by a Growing Greener Environmental Stewardship and Watershed Protection Grant

Submitted by The Nature Conservancy
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with contributions from the Pennsylvania Instream Flow Technical Advisory Committee

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EXECUTIVE SUMMARY AND RECOMMENDATIONS

This report, "Developing Instream Flow Criteria to Support Ecologically Sustainable Water Resource Planning and Management," was produced to support efforts to develop and implement an instream flow protection system through a collaborative process with the public agencies that make up the Pennsylvania Instream Flow Technical Advisory Committee (TAC). This final report builds on a draft report, released March 16, 2007, and substantive engagement with the TAC over a period of almost two years. The project was completed by The Nature Conservancy (TNC) with funding from the PA Department of Environmental Protection's Growing Greener Grant Program.

The scope of work committed TNC to assess "available data, tools and approaches that can be used to meet the overall goal of statewide instream flow protection criteria." The scope also included a "pilot watershed study effort using existing data to develop general stressor-response relationships between flow alteration and ecological health." In accordance with the project scope, this report includes an evaluation of the primary options for statewide instream flow (i.e. environmental flow) criteria development, estimates of related costs, and evaluation of pilot studies classifying Pennsylvania rivers statewide and linking flow alteration to biological response in the Susquehanna watershed.

Based on the "Pennsylvania Statewide Instream Flow Studies Issues Paper" (Young 2006), the Ecological Limits of Hydrologic Alteration (ELOHA) framework (Poff et al. in review), input from the TAC, and our own assessment, development of an instream flow protection system should be guided by the components that are discussed in the following sections:

Section 1	Introduction
Section 2	Building a Hydrologic Foundation
Section 3	Developing and Applying an Appropriate River Classification
Section 4	Selecting Hydrologic Statistics and Assessing Hydrologic Alteration
Section 5	Developing Flow Alteration—Ecological Response Relationships
Section 6	Defining and Using Environmental Flow Criteria in Decision-Making

Each of these sections begins by providing background information, definitions, and goals related to each of the components of an overall instream flow protection framework. The body of the text in each section evaluates options for accomplishing the goals based on information gathered from the published literature, grey literature, and personal contacts with professionals in this field. Each section concludes with a set of recommendations. Although each of the instream flow protection system components described in Sections 2-6 are useful parts of an overall framework, they should not be seen as a linear approach without room for modification or variable levels of investment. Pennsylvania's state and basin resource agencies may find it desirable to invest heavily in certain elements of the framework while taking a less sophisticated approach to others. Similarly, although early development of a hydrologic foundation is critical for the completion of some elements of the framework, it is not necessary to develop all components outlined in Sections 2-6 in sequence. Rather, much could be accomplished by working on components simultaneously and exploring a multi-stage approach to fulfilling the recommendations in this report.

Following Sections 1-6, the report includes a set of key case studies, describing:

1. The Virginia water management decision support model
2. Results of the pilot hydrologic stream classification in Pennsylvania
3. Work in the United Kingdom to set environmental flow standards
4. The Michigan water withdrawal assessment process
5. Results of the Susquehanna pilot study linking flow alteration to ecological response
6. The Connecticut draft regulatory framework for streamflow protection

The report also includes a Project Options Table in Appendix 1, which clearly sets out a range of potential approaches for building a Pennsylvania Water Management Decision Support System and state-specific flow-ecology relationships that can inform instream flow criteria. For each option, costs, strengths, weaknesses and other relevant information are included.

Summary and Recommendations

The Commonwealth of Pennsylvania is fortunate to have a wealth of rivers, significant aquatic biodiversity, popular game fish resources, and an engaged public interested in river protection and restoration. Pennsylvania also has a set of state and federal public agencies interested in environmental flow protection as demonstrated through their involvement in the Pennsylvania Instream Flow Technical Advisory Committee (TAC). The TAC is already very knowledgeable about the state-of-the-art tools and common issues associated with comprehensive environmental flow protection. This common level of understanding and shared goals creates an important opportunity to move forward. This report provides the information to make informed decisions about which methods and approaches are most promising for Pennsylvania.

TAC members have noted that there is a range of existing state and basin commission authorities that can be more effectively employed to implement environmental flow protection in Pennsylvania. A strong, comprehensive and statewide approach to environmental flow protection would allow agencies to realize these opportunities and provide a framework for comprehensive water management decision-making that meets environmental and human water needs.

<u>Recommendation: Pennsylvania should define environmental flow goals and criteria through a process that includes a broad range of stakeholders, supported by scientists.</u>

Public agencies, informed by stakeholders, need to work together to define ecological goals and the level of protection for environmental flows that are acceptable to meet those goals. It is inevitable that there will be scientific uncertainties and conflicting visions of the importance of ecological goals relative to human uses. Given this, a structured, inclusive approach should be employed to take flow alteration-ecological response relationships (whether quantitative or semi-quantitative) and other available scientific information, and use this information to define ecologically and socially acceptable thresholds of hydrologic alteration. These instream flow criteria should be linked to a tiered set of goals which, in turn, can be applied to all rivers or river segments within Pennsylvania.

This work can and should build upon the Pennsylvania Instream Flow Model (PA IFM) and its current implementation in many of the state's coldwater trout streams. The process can also build on other existing tools, including the PA Water Analysis Screening Tool (WAST). The PA IFM demonstrates the value of regionalizing information on flow alteration and ecological response as it is based on generalized changes to trout habitat associated with hydrologic alteration.

Recommendation: The Pennsylvania Instream Flow Technical Advisory Committee (TAC) should move from an advisory committee to a program development committee.

The TAC is well suited to design and lead a process to develop environmental flow criteria. The broad expertise and experience of its members provides the knowledge and insight to identify the most promising opportunities and develop solutions that are appropriate to Pennsylvania's future. The TAC can build on the state's long history of working on environmental flow criteria and previous investments in water management tools. The committee should develop a charter outlining the goals of the effort and the responsibilities of the committee leadership and members. This charter would describe the goal and common commitment to develop a comprehensive, statewide approach to environmental flow protection, including the development of tools to build a hydrologic foundation for Pennsylvania. Given a set of shared goals, the committee would develop specific short-term and long-term tasks towards comprehensive environmental flow protection and work to maximize available resources to accomplish these tasks in a timely manner.

Using the ELOHA Framework

The Ecological Limits of Hydrologic Alteration (ELOHA) framework (Poff et al. in review) provides guidance on a set of specific building blocks to develop comprehensive instream flow policies. ELOHA provides a timely and scientifically credible means for broadly assessing environmental flow needs given that in-depth studies cannot be performed for all rivers. By linking changes in river flows to changes in ecological conditions, water managers and stakeholders can develop regional environmental flow criteria and apply them to many rivers at a time, without requiring site-specific hydrologic or biological studies on each river.

The method involves five steps:

1. Build a **hydrologic foundation**, a state or regional database of daily or monthly streamflow hydrographs representing both baseline (undeveloped) and developed conditions for a recent time period long enough to represent past climate variability.
2. **Classify river segments** based primarily on similarity of flow regimes with consideration of other aspects of river variability.
3. Determine **the amount of hydrologic alteration,** using flow variables linked to ecological responses and useful as water management targets.
4. **Develop flow-ecology response relationships** by associating degrees of hydrologic alteration with associated changes in ecological condition.

5. Use these flow-ecology relationships to **develop ecological and human use goals and identify specific environmental flow criteria** which define the limits of acceptable and unacceptable change to hydrologic and ecological conditions.

This report provides detailed discussion of the possible approaches to complete these steps and recommends approaches that appear to be most appropriate and applicable in Pennsylvania.

Building a Hydrologic Foundation

Building a "hydrologic foundation" is a critical element of ensuring the statewide availability of water for human use and ecological protection. As described in the ELOHA framework (Poff et al. in review), this is a hydrologic modeling effort that consists of developing "daily flow time series representing simulated baseline and developed conditions throughout the region."

<u>We recommend that a statewide GIS-based computer application be developed that allows for the estimation of baseline (i.e., "minimally impacted") and current/future daily flow time series at any point location, both gaged and ungaged, throughout the state.</u> We refer to this application as Pennsylvania Water Management Decision Support System (PA DSS).

This PA DSS would build upon the PA Water Analysis Screening Tool (WAST). Development of a daily flow time series will permit the calculation of various flow statistics of ecological significance at a range of time steps (from daily to inter-annual), which include magnitude, duration, frequency, timing, and rate-of-change. This capability is important for both water management and for the development of flow-ecology response relationships that can serve as a basis for instream flow criteria.

The PA DSS would allow for simulation of baseline and developed daily streamflow over discrete time periods (e.g., 1960-2000). The recommended approach for determining a daily flow time series is a regression-based approach that estimates baseline flow duration statistics to construct flow duration curves at any point location of interest. Daily time series data would result from linking these estimated flow duration curves to flow duration curves defined at index (i.e., least impacted) gages in the state. These index gages have been identified by USGS and refined during the course of this project, as described in the report. This approach is similar to the one that MA USGS has developed under contract with Massachusetts Department of Conservation and Recreation as part of their Sustainable Yield Estimator Project.

In addition to simulating baseline daily streamflow, the PA DSS would develop daily streamflow estimates that account for both individual and cumulative current diversions, return flows, and reservoir operations upstream of a point of interest. Although there are challenges involved in simulating reservoir operations, we believe that the Sustainable Yield Estimator approach holds the most promise as a basis for statewide water management decision-support.

The PA DSS could initially use the WAST pour points as analysis locations, but it may be desirable to expand to a more "point and click" application. Initial input to this DSS would be from WAST's comprehensive water use database, which currently accounts for withdrawals (registered and estimated) as well as discharges (USGS "Methodology for the Development of Water Analysis Screening Tool" undated draft). The PA DSS should be constructed to allow for continuous improvement in estimates of water management impacts and to incorporate any new available data. If at all possible, monthly estimates of water use and discharge should be incorporated into a PA DSS, to more fully account for seasonal variation in water use. In 2008, monthly water use reporting will begin, and these data can be incorporated into a PA DSS.

A PA DSS with the capability to simulate current conditions would also allow for the evaluation of future scenarios on a site-specific basis or statewide. Future water use or discharge proposals could be incorporated directly in the application and the resulting alteration in hydrology subsequently analyzed. This functionality would likely be critical for sustainable, integrated management of statewide water resources through state and basin commission authorities.

Developing and Applying an Appropriate River Classification

As described in the ELOHA framework (Poff et al. in review), river classification serves three purposes. First, it can be used to develop flow alteration-ecological response relationships with data from a relatively small set of rivers that can be applied to a broader set of rivers within the same type. Second, once flow-ecology relationships are developed for stream classes, it can be used to assign any river location to a type for which the natural range of variation in key flow statistics and ecological responses are known. Third, classification can be useful for directing future monitoring efforts to improve the quality of flow-ecology relationships or to detect management impacts. Classification is completed using primarily hydrologic regime characteristics based on the premise that natural flow attributes, as well as hydrologic alterations, shape their associated ecological communities in a similar way within a river class.

The goal for developing a river classification is to group rivers and stream segments that have similar hydrologic behavior into distinct classes. There are two main approaches to achieving this goal: (1) based on hydrologic statistics; or (2) using landscape variables that influence hydrology. The ecohydrological regionalization of Australia reviewed in the report uses a combination of these two approaches and also uses a novel statistical approach to group streams into hydrologic classes.

As part of this report, we completed a pilot application of the Hydroecological Integrity Assessment Process (HIP) hydrologic classification in PA, reviewed existing classifications applied to Pennsylvania, including the Pennsylvania Aquatic Community Classification (PACC) and the Northeast Aquatic Habitat Classification System (NEAHCS), and reviewed other applicable approaches. Based on this, we believe there is merit in further pursuing a hydrologic classification that can be extrapolated to river reaches throughout Pennsylvania and that considerable work toward this goal has already been completed. <u>We recommend that a scientific committee that includes members of the Pennsylvania Instream Flow Technical Advisory Committee (TAC) review the work to date and finalize a classification.</u> As part of this process, we specifically recommend that they:

1. Review the statistical approaches in the pilot application of Hydroecological Integrity Assessment Process (HIP) in Pennsylvania and, if necessary, suggest any alternative clustering methods.

2. Review the Bayesian statistical methods used in the ecohydrological regionalization of Australia and decide whether or not these methods should be applied in Pennsylvania.

3. Consider using a similar approach to the New Zealand River Environment Classification (REC) to develop a flow classification using a subset of variables in the existing physical classifications in PA (e.g., PACC, NEAHCS) and additional variables that may not be included (e.g., climatic variables).

4. Evaluate correlations between selected physical habitat variables in the PACC and/or the NEAHCS and hydrologic statistics calculated for the index gages.

5. If a classification based on hydrologic statistics (e.g., HIP) is chosen, agree on an approach to assign stream classes to ungaged stream segments.

6. Even though hydrology may be the primary factor in determining river types, factors other than hydrology can and should be used to further subdivide stream types.

The result of this classification effort would provide a strong basis for the development of flow-ecology relationships. It would also be useful for assigning river sites to a class in which the general variation in flow conditions is known.

Selecting Hydrologic Statistics and Assessing Alteration

Assessing hydrologic alteration using a small set of ecologically-relevant flow statistics is important for assessing the impacts of current or future water management, the development of flow alteration-ecological response relationships, and the development of instream flow criteria. As discussed above, we recommended using estimated daily time series for baseline (minimally impacted) and current/future (developed) conditions. Once a daily time series has been developed, there are two readily available programs to analyze this type of data: the Indicators of Hydrologic Alteration (IHA) and the Hydrologic Assessment Tool (HAT). The difference between the two analysis packages is not substantial, but there are trade-offs associated with each that are described in the body of the text. Although other programs (e.g., Microsoft Excel) could be coded to provide similar statistics, the advantage of using the statistics in these existing programs is that they are currently being used around the country and have been, or can easily be, screened for ecological applicability.

<u>Our recommendation is to develop a Pennsylvania-specific Water Management Decision Support System that can either link directly to either program (IHA or HAT) or can be programmed to calculate the array of statistics that these programs have the capability to provide.</u> The most economically efficient approach would be to create a PA Water Management DSS with IHA and HAT compatibility, as is being developed as part of the Massachusetts Sustainable Yield Estimator.

<u>To develop a set of statistics for use in water management and developing instream flow criteria, we recommend focusing on a set of screening criteria that should:</u>

1. Be sensitive, and have explainable behavior related to flow changes that are likely to occur due to human uses;
2. Have limited redundancy;
3. Represent natural variability in hydrologic regime, including magnitude, timing, duration, frequency, and rate-of-change if possible;
4. Have conceptual and empirical linkages to ecological response;
5. Be repeatable, therefore not involve subjective user settings; and
6. Facilitate communication among state policy makers, hydrologists, and ecologists as well as to water users.

Arriving at a set of statistics that meets these goals involves an iterative process beginning with the results of the hydrologic classification. The pilot classification for Pennsylvania using the HIP approach includes a set of "primary and secondary hydrological indices" that resulted from a principal components analysis. These statistics generally explain the dominant pattern of hydrologic variation for each of the 11 identified components of the flow regime and are differentiated by each river class. These statistics have limited redundancy and represent intra- and inter-annual variability in flow regimes, but may not satisfy some of the other goals listed above (e.g., conceptual linkages to ecological response, easy to use by water managers). Any statistics resulting from the final Pennsylvania hydrologic classification should be modified and supplemented based on an expert input process.

Developing Flow Alteration-Ecological Response Relationships

A central feature of the ELOHA (Poff et al. in review) and Arthington et al. (2006) frameworks, as well as our recommendations, are flow alteration-ecological response relationships applicable to Pennsylvania river types. These relationships can be either conceptual or empirical, and can be linked to a range of ecological information from habitat characteristics to community responses. Best available information can be used to link flow alteration to ecological response through conceptual relationships, risk thresholds, or empirical models for rivers and streams in Pennsylvania. Quantitative or semi-quantitative flow-ecology relationships, in contrast to the typical hydrological "rule of thumb" approaches, provide decision-makers explicit information on the risks of excessive hydrological alteration to natural resources of concern. Final instream flow criteria should be set through a stakeholder process (described in Section 6) that uses these quantitative or semi-quantitative flow ecology relationships.

<u>We recommend a statewide approach to instream flow protection through development of ecologically-based environmental flow (also "instream flow" or "streamflow") criteria.</u> Environmental flow criteria define an acceptable degree (limit) of flow alteration for a set of ecologically-relevant flow statistics to meet state ecological goals.

<u>The following steps summarize our recommended approach for developing flow-ecology relationships that can be used to define environmental flow criteria:</u>

1. Draft a set of flow alteration - ecological response hypotheses for major river and stream types in Pennsylvania.

This set of hypotheses should:

- *Address a variety of relevant flow components* (e.g., extreme low flow, low flow, high pulses, and large floods) and characteristics (magnitude, timing, duration, frequency, and rate-of-change)**;**
- *Address a variety of relevant taxonomic groups* (e.g., fish, macroinvertebrates, riparian vegetation) and habitat types (e.g., floodplain, riffle area);
- *Describe the direction of the anticipated ecological response* (e.g., fish diversity will *decrease* as low flows *decrease*); and
- *Describe the functional form of the response* (e.g., linear, threshold, curvilinear).

Hypotheses should be developed for all major stream types in an existing or revised classification of Pennsylvania rivers and streams. Habitat-flow relationships have been developed for a number of rivers across the state, and a subset of those relationships have been generalized across the coldwater trout streams type as part of the PA/MD Instream Flow Studies (Denslinger et al. 1998). These habitat-flow relationships typically focus on low flow parameters and on a small suite of species. Nonetheless, they act as an important basis for hypotheses about how flow alteration is likely to impact key aspects of ecological integrity in one system type. The conceptual relationships in Poff et al. (in review) can also serve as a starting point for additional hypotheses. Greater detail on this step is provided in the Project Options Table (Appendix 1), approach B1.

2. Convene experts and conduct literature review to determine potential risk thresholds.

We recommend that expert input and literature review be used to define thresholds of hydrologic alteration beyond which ecological impacts would be anticipated. This process should be a collaborative effort between resource agencies, academic scientists, and other stakeholders. Depending on the existing literature and research supporting these thresholds, these thresholds could be used to establish initial flow criteria. *Case Study 3: Environmental Flow Standards to Meet the EU Water Framework Directive* provides one model for this process.

In addition, these thresholds would guide the research and monitoring to develop quantitative relationships between flow alteration and ecological response. These investigations support an adaptive management process that defines risk thresholds and streamflow criteria based on best available information, with refinement and validation over time.

3. Begin validating flow alteration-ecological response hypotheses using existing data.

As hypotheses are developed, experts should also begin identifying site-specific or regional hydrological and ecological data that can be used to test these hypotheses. Hypothesis testing should build on experience and lessons learned in the pilot study to assess the impacts of water withdrawals on macroinvertebrate assemblages in the Susquehanna River Basin. Results are described in detail in *Case Study 5: Development of flow alteration-ecological response curves for Pennsylvania streams.*

The approach taken by the Michigan Groundwater Conservation Advisory Committee to define "flow-fish functional response curve" is also particularly promising for Pennsylvania. Existing fish databases, including those compiled and formatted by Pennsylvania Natural Heritage Program as part of the Pennsylvania Aquatic Community Classification, should be further reviewed to determine if they could be used to develop these types of relationships. A similar approach could be taken using other taxa, including macroinvertebrates, if sufficient data are available.

4. Define additional research and data collection needed to link hydrologic alteration to ecological condition.

Existing biological databases and hydrological information in Pennsylvania are likely sufficient for initial examinations of flow-ecology relationships, but will ultimately need to be supplemented. This is due primarily to the spatial and temporal disconnect between biological sampling sites and sites with adequate hydrologic information, but is also due to the different methodologies used by different resource agencies. Pennsylvania should also begin to supplement existing databases with additional sampling designed to detect impacts on indicators of ecological integrity from flow alteration as suggested by any conceptual models and initial flow-ecology relationships. This monitoring program would also be part of an adaptive management cycle that allows for validation and refinement of flow-ecology relationships, risk thresholds, and instream flow criteria. A monitoring program established specifically to detect ecological effects from flow alteration would take into account lessons learned from the Susquehanna River Basin pilot project (described in *Case Study 5: Development of flow alteration-ecological response curves for Pennsylvania streams*). Specifically, this program should select ecological monitoring sites that (1) spatially and temporally match sites with hydrologic data, (2) are distributed across a range of hydrologic alteration, and (3) are distributed across stream classes (if stream classes are defined) or distributed across streams and rivers with different sizes and physical and chemical characteristics. In addition, the monitoring program should develop or identify ecological response metrics that are hypothesized to respond to hydrologic alteration and address multiple taxonomic groups.

Any effort to develop environmental flow criteria would benefit significantly from a near-term investment in capacity to estimate baseline and current hydrologic conditions statewide. These recommendations reflect a sequence of approaches listed in the Project Options Table (Appendix 1).

Defining and Using Environmental Flow Criteria in Decision-Making

A comprehensive water management flow program to protect ecological structure and functions of streams and other freshwater resources is, like many other environmental protection programs, based on a few key elements. <u>We recommend Pennsylvania begin the process of identifying desired ecological and human use goals, identifying specific criteria of ecological condition that can be used to define acceptable and unacceptable limits to changes to these conditions, and defining measures of hydrologic change associated with these criteria and operating rules by which these criteria and measures can be achieved by water managers.</u> Such goals, criteria, measures and operating rules can be used to inform a broad range of regulatory and planning programs within Pennsylvania.

For the protection of hydrologic conditions, this entails defining appropriate levels of protection for specific water bodies based on existing and desired designated and beneficial uses. It also entails providing a framework for assessing both existing conditions as well as assessing new uses and projects to determine their ability to meet these goals or standards. Central to hydrologic protection are *environmental flow criteria*, the flow conditions or degrees of hydrologic alteration that must be consistently met by water users in any particular river segment. Environmental flow criteria (i.e. instream flow criteria) should be based on a scientific understanding of how changes in hydrology are likely to impact natural resource conditions. These criteria, much like water quality criteria, include a definition of what is acceptable and what is unacceptable in terms of changes to hydrologic conditions.

Scientists can play a role in defining environmental flow criteria, but since this is a policy action with significant consequences, relevant public agencies informed by stakeholders will need to make the final decision. The Michigan Groundwater Conservation Advisory Council provides a good example of a stakeholder process, informed by science, that resulted in quantitative "adverse resource impact" thresholds applicable to rivers across the state.

As with any environmental program, establishing standards is necessary for achieving desired outcomes. But strong criteria alone are not sufficient. Meeting Pennsylvania's water needs while protecting ecological systems and processes affects how we think about water infrastructure – that is, where we get our water, how we use it, where we return it. Meeting the requirements of public health, providing water to support a strong economy and ensuring a sustainable environment requires integrated approaches that look at storm water management, wastewater returns, road and bridge construction, and stream restoration and rehabilitation.

Clear environmental flow goals and criteria that lead to water use guidelines provide the foundation from which the many entities and individuals can guide their actions to achieve these predefined goals. Experience with other environmental programs, from water quality to air quality, demonstrates how standards and criteria are the foundation from which a broad array of actions can be guided and directed over a multi-decadal time scale.

SECTION 1: INTRODUCTION

Resource agencies in the Commonwealth of Pennsylvania, including the Department of Environmental Protection, the Department of Conservation and Natural Resources, and the Fish and Boat Commission, have expressed a clear desire for an approach to evaluating the impacts of flow alteration on aquatic resources that is applicable statewide. This need is grounded in the goal to have long-term ecologically sustainable management of water in the rivers of Pennsylvania, a goal which is shared by The Nature Conservancy. The development and application of "instream flow" or "environmental flow" criteria is central to this goal. These criteria can act as the basis for water management decisions by defining the acceptable levels of flow alteration to meet aquatic resource goals. To that end, The Nature Conservancy, working under a Growing Greener Grant from the state, has reviewed options and is making final recommendations for the development and implementation of these instream flow criteria.

The development of environmental flow criteria that can be applied across broad spatial scales is the central challenge of implementing an instream flow protection program. There are a wide variety of methodologies available for determining site-specific environmental flow needs, each with their strengths and weaknesses. In contrast, very few methodologies exist, or have been applied, across the regional spatial scale (i.e., of a state or large river basin). This report focuses on an integrated approach for the development of scientifically sound, implementable environmental flow criteria statewide. As an introduction, this section includes a brief review of what, according to the scientific literature, constitutes scientifically defensible instream flow criteria at the regional scale.

Recent publications on environmental flow protection (Postel and Richter 2003, Annear et al. 2004, NRC 2005, Acreman 2006, Arthington et al. 2006, Acreman 2007) all agree on a set of key principles. These principles have been influenced by what has been called the natural flow regime paradigm. This paradigm essentially states that "the structure and function of a riverine ecosystem and many adaptations of its biota are dictated by patterns of temporal variation in river flows" (Arthington et al. 2006). In practice, this leads to a set of principles that include:

1. The goal of environmental flow standards should be to protect entire ecosystems instead of single species (NRC 2005).
2. Environmental flow standards should provide inter- and intra-annual variability in a manner that maintains the form and function to the greatest extent possible (Annear et al. 2004). This includes protecting natural magnitude, frequency, timing, rate-of-change, and duration of different hydrologic conditions, particularly high and low conditions, to the greatest extent possible.
3. Environmental flow standards should be based upon site-based information about the species, communities, and ecosystems that naturally occur or that could be expected to naturally occur in the watershed.
4. Adaptive management should be used so that changes in the ecological system can be observed and the management approaches adjusted as necessary to achieve the goal of protecting and restoring ecological integrity.
5. A margin of safety should be included in hydrologic regime management programs.

There is a major gulf between these generally-accepted principles and the practice of statewide instream flow protection, which typically only protects minimum flows. When considering how and whether to limit alteration of surface and ground waters, the "natural flow regime" paradigm provides only general guidance on a methodological approach to environmental flow standards. The Instream Flow Council (2004) points out that historically, "the word 'standard' came to mean 'minimum'- a line in the sand above which all water could be apportioned and below which the water was reserved for aquatic life." Minimum flow protection continues to dominate the practice since there is no accepted method of defining ecosystem-based standards. It is well understood that different flow levels provide for ecosystem and species needs at different times of year. However, it is difficult to quantify those flow levels in a manner that is meaningful for the development of ecosystem-based environmental flow criteria. Our recommendations build on relevant experience of minimum flow protection programs but expand on this to develop a system that protects other aspects of flow that are critical to supporting the aquatic resources of Pennsylvania. Thus, we focus, at a statewide level, on the question of "How much flow alteration can occur without negatively impacting our aquatic resource goals?"

Fortunately, a framework for developing scientifically-credible regional environmental flow criteria has been recently developed by an international group of river scientists. The recommendations of this group, which includes one of the authors of this report and one member of the Pennsylvania Instream Flow Technical Advisory Committee (TAC), are included in "The Ecological Limits of Hydrologic Alteration (ELOHA): A New Framework for Developing Regional Environmental Flow Standards" (Poff et al. in review). This framework consists of five major parts, each of which is represented in this report and mirrored by the report structure (Figure 1.1):

1. Hydrologic Foundation
2. River Classification
3. Computing Flow Alteration
4. Developing Flow-Ecology Relationships
5. Developing and Implementing Environmental Flow Standards (Social Process)

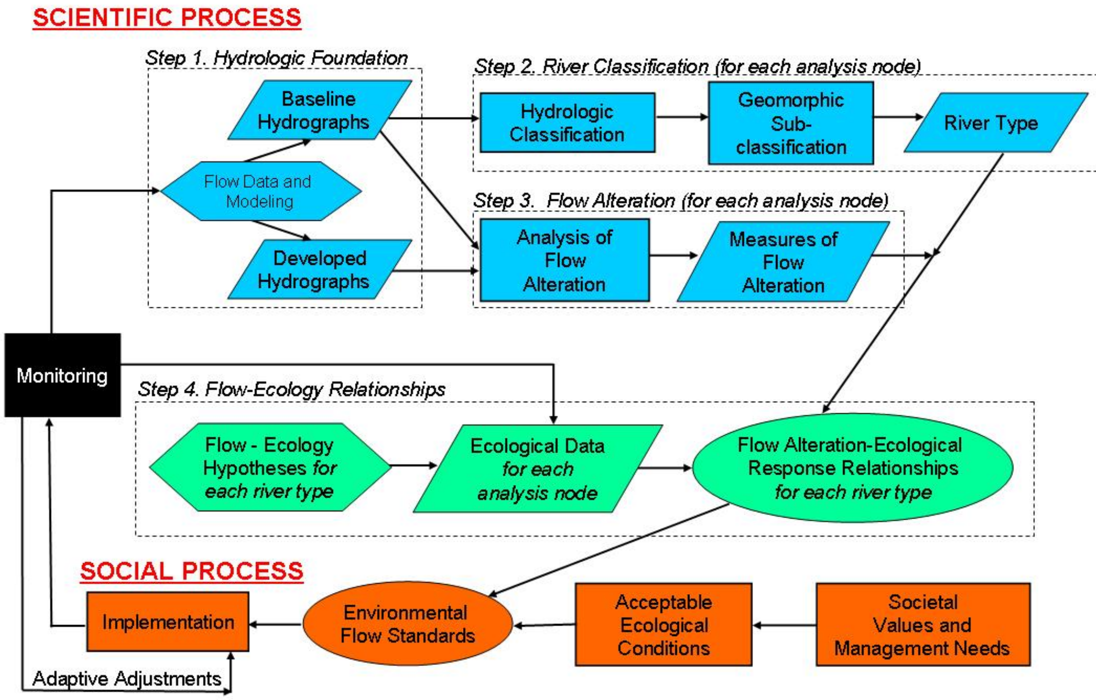

Figure 1.1 Flow chart illustrating ELOHA framework for defining and applying regional environmental flow criteria

The "Hydrologic Foundation" is the output of a hydrologic modeling effort that allows for the estimation of baseline and developed hydrologic conditions at locations throughout a region. This hydrologic foundation can be created at a range of time and spatial scales, but using daily flows simulated at the state scale is clearly achievable in the United States. "Baseline flow" conditions are estimates of flows that are minimally impaired (or "least impacted") by dam and reservoir operations, water withdrawals and diversions. "Developed" or "current conditions" represent the effects of current water development in the state and can be modified to account for future scenarios. Developing this capability is critical to: (1) estimate the degree to which current water use is altering natural streamflows; (2) permit the development of flow-ecology relationships by providing flow alteration estimates at sites with existing biological data; and (3) allow site-specific implementation of instream flow criteria and understanding of water availability. Modeling approaches for estimating baseline and current/future hydrologic conditions are covered in detail in Section 2. They vary significantly in cost and intensity, from statistical approaches to hydrologic process models.

"River Classification," as presented in the ELOHA framework, is designed to extrapolate understanding of hydrologic conditions at particular sites to other similar river systems and to support regionalized development and application of criteria. Hydrologic regime is recommended as the primary basis for river classification since flow regime is considered a key driver of river systems and of significant interest to water managers. Classifying rivers based on similarity of hydrologic regime can be approached using a range of methods reviewed in this report. To be amenable to management, the goal of classification should be to define a relatively small number of hydrologic river types, trading detail for interpretability. In addition, sub-classification or stratification of river types based on factors such as geomorphology or habitat conditions can be useful to management and the development of flow-ecology relationships. "Developing and Applying Appropriate River Classification" is described in Section 3.

Section 4 explains how "Selecting Hydrologic Statistics and Assessing Hydrologic Alteration" is the key step in determining which hydrologic variables should be used to assess alteration and how this should be accomplished at locations of interest. Given development of a hydrologic foundation, the difference between baseline and developed (current/future) hydrologic conditions can be quantified at a range of analysis nodes, which serves at least two purposes. First, this type of assessment permits cumulative impact assessment at any location of interest (e.g., withdrawal point) as part of a comprehensive water management program. Second, understanding hydrologic alteration across a broad landscape can be combined with biological information to assist in the development of flow alteration-ecological response relationships. To complete both of these functions effectively requires use of a relatively narrow set of hydrologic statistics. This report describes and recommends approaches to defining a small set of non-redundant, ecologically-relevant hydrologic statistics that can be used for management purposes.

"Developing Flow Alteration—Ecological Response Relationships" is a core concept in ELOHA and in our recommendations for expanding the scope of instream flow protection statewide. These relationships between flow alteration and ecological condition are grounded in the biological condition gradient approach (Davies and Jackson 2006) in which increasing degrees of anthropogenic stress lead to decreasing ecological condition. As described in Section 5, this work can begin with the development of hypotheses, or conceptual relationships, guided by expert opinion and knowledge of existing literature. Although non-quantitative, these conceptual relationships can prove useful to managers in instream flow criteria development and serve as a basis for empirical testing. More empirical approaches, using ecological data (both existing and future monitoring data) and hydrological data (both measured and estimated conditions) to develop linkages between flow alteration and ecological impact, provide the strongest scientific basis for decision-making. Well constructed empirical approaches, in contrast to the typical hydrological "rule of thumb" approaches, provide decision-makers explicit information on the risk of excessive hydrological alteration to natural resources of concern. Creation of a hydrologic foundation in combination with a spatial ecological database and river classification should permit these flow-ecology response relationships to be developed statewide. The report provides examples of existing regional flow-ecology response relationships, guidelines for developing new conceptual and empirical relationships, and recommendations for use of these relationships in instream flow criteria.

Policy and social considerations are the focus of the final step in the ELOHA framework, as the scientific process gives way to developing and implementing environmental flow standards. This social process begins with setting of ecological goals and associated levels of acceptable risk to ecological health in relation to flow alteration. Scientific understanding of flow alteration-ecological response relationships informs this process, but can not determine standards or criteria, given the range of economic and social factors at play and the range of anthropogenic factors that impact ecological integrity. In Section 6 of the report, "Defining and Using Environmental Flow Criteria in Decision-Making," concepts and frameworks useful to water management and policy are introduced and explained. These include goal-setting, criteria development, measuring consistency, and translating criteria into operating rules and withdrawal limits.

SECTION 2: BUILDING A HYDROLOGIC FOUNDATION

2.1 Background

A hydrologic foundation includes a database of streamflow hydrographs (daily or monthly) that represent both baseline and current flow conditions (Poff et al. in review). This hydrologic database should also be able to be modified to account for new withdrawals and other future activities affecting water availability, thereby estimating future flow conditions. Understanding "baseline" or "reference" flow conditions and their natural range of variability is fundamental to management of instream flow conditions. Baseline flow conditions provide a basis for understanding both water availability and ecologically important natural flow variability (Poff et al. 1997). A variety of techniques for characterizing available flow data or estimating hydrologic characteristics are discussed in Gordon et al. (1992) and Richter et al. (1997), as well as in many standard hydrology texts.

Within Pennsylvania, as with any area across the United States, there is a limited amount of hydrologic information from unimpaired gaging stations. In Pennsylvania, USGS has identified a set of 195 "index gages" or continuous record station gages that can be considered least impaired due to limited upstream regulation, diversion, and mining impacts (Stuckey 2006). Despite the importance of this information, these stream gages provide directly relevant information for only a tiny portion of over 83,000 miles of streams and rivers in the state. Therefore, to understand minimally impaired conditions in rivers statewide, it is necessary to be able to estimate unregulated flows at any point location of interest.

Estimates of current flow conditions reflecting existing water use and expressed as flows over a historical time period (e.g., 1960-2000) can be used to understand existing water availability challenges. These estimated current flow conditions can be compared with baseline flow conditions to assess flow alteration in rivers and serve as a basis for developing flow alteration-ecological response curves. The finer the temporal and spatial scale of this estimate of flow conditions, the more useful it is likely to be for management. Estimates of future flow conditions can be developed in order to evaluate permit applications, renewals, and reservoir operation proposals as well as large scale changes in water use intensity.

Current flow conditions are well known or easily extrapolated in river reaches at or downstream of stream gaging stations. However, for most of the state, estimating current flow conditions at ungaged locations is necessary to account for the effects of current water uses, incorporate the impacts of potential future actions, and serve as a basis for comprehensive statewide instream flow protection. Approaches and software tools exist that can be used to develop these current or future flow estimates, each with their own limitations. The options for approaching this goal are severely constrained by quality and spatial/temporal resolution of available data on water withdrawals, return flows, and reservoir management. This section will describe the challenges as well as promising options for developing current and future flow conditions.

2.2 Definition of Baseline, Current, and Future Flow Conditions

Our working definition of baseline flow conditions is an estimate of flows that are minimally impaired (or "least impacted") by dam and reservoir operations, water withdrawals and diversions. This is a "least impaired" approach, because even the best gage data may be somewhat altered by land use and other anthropogenic factors. One major complication of estimating unregulated conditions is the effect of land use on hydrology. Few gages considered unimpaired by the USGS can be considered to have "natural" land use (i.e., pre-Columbian, typically heavily forested). All gages have various levels of upstream impervious surface and loss of forest cover due to ongoing forestry practices, agricultural uses and human settlement. We are not, however, recommending that pre-Columbian land use flow conditions be defined for any streams, nor are we suggesting that streams with current withdrawals represent baseline flow conditions.

We define current hydrologic conditions as those that incorporate existing impacts from water diversions, discharges, reservoir operations and other upstream water uses that could affect flow patterns. Similarly, future hydrologic conditions incorporate the anticipated impacts of these water uses, whether at a site-specific or larger-scale.

Baseline, current and future hydrologic condition estimates can be useful at monthly, daily, or even subdaily time scales. Daily time series estimates over a consistent modeling period of at least 20 years are relatively cost effective and provide significant information (TNC 2005). Estimated daily flow time series information can also be used to estimate the degree of alteration from baseline conditions in terms of selected flow statistics using readily available free software packages.

Goals for simulating baseline flow conditions and estimating current or future flow conditions:
1. Applicable at any location in Pennsylvania through a GIS application
2. Generates data that represents baseline, current, and future hydrologic conditions over at least 20 years for each period
3. Can be used to calculate a wide range of ecologically-relevant flow statistics for baseline, current and future periods for any location within Pennsylvania
4. Developed at the finest time scale possible within cost limitations
5. Low calibration and estimation error
6. Can incorporate new and improved water use and reservoir operations information
7. Enables the user to partition flow alterations to multiple categories including diversions, discharges, reservoir operations, and land use

In this section, we review several approaches to simulating baseline flow conditions and estimating current or future flow conditions. We also make recommendations on implementation of a hydrologic foundation in Pennsylvania and review how existing data and models in the Commonwealth could contribute.

2.3 Potential Approaches Considered for Simulating Baseline Conditions

2.3.1 Statistical Regression Equation Techniques to Develop Baseline Statistics

This statistical approach to estimating baseline flow statistics is represented in Pennsylvania by the recent USGS report "Low-Flow, Base-Flow, and Mean-Flow Regression Equations for Pennsylvania Streams" (Stuckey 2006). This report uses data from 293 unregulated gages across the state, 195 of which are continuous-record "index gages," and information on a wide range of GIS-derived watershed characteristics to develop statistics that estimate flows at ungaged locations using generalized least-squares regression. A range of flow statistics can be estimated, from as low as the 7 day, 10 year low flow to the mean flow at an ungaged site in which a set of basin characteristics are available (e.g., drainage area, mean elevation, mean precipitation, land use, and bedrock geology). Specifically, regression equations were developed for five "low flow regions" in the state to predict the following low flow statistics: Q7,10; Q7,2; Q30,10; Q30,2; and the Q90,10 (i.e., 90 day, 10 year low flow). The USGS report also defines statewide equations for the 10-, 25-, and 50-year baseflows, harmonic mean, and mean annual flow.

An analysis of prediction error was completed for the estimates which resulted in standard error results that varied from 12 to 66 percent. Error rates generally were higher for predictions of extreme low flow statistics (i.e., 7Q10) and relatively low for predictions of statistics closer to central tendency (i.e., mean annual flow, Q90,10). Error rates were modest for estimates of infrequent events such as the 10 to 50 year baseflow (21-23 percent). A small portion of the results of this PA USGS work has been incorporated by USGS into the Water Analysis Screening Tool as part of the State Water Plan process. This tool currently allows user-friendly estimation of unregulated 7Q10 at over 10,000 "pour points" around the state.

The regression equations used to define flow statistics at ungaged locations in the state have been incorporated into the USGS StreamStats program for Pennsylvania. The first phase of StreamStats was recently completed in Pennsylvania and is designed to make "the process of computing streamflow statistics for ungaged sites much faster, more accurate, and more consistent than previously used manual methods" (Ries et al. 2004). In terms of functionality, StreamStats provides managers with the ability to define flow and basin characteristics at ungaged sites through the state of Pennsylvania. This application of StreamStats is limited by the regression equations developed to date by USGS. In practice, this means that only flow statistics developed as part of Stuckey (2006) (listed above) can be defined. It is worthwhile to note that the only exceedance flow value PA StreamStats can simulate is the 50% exceedance probability flow, which makes it impossible to estimate flow duration curves at ungaged sites. PA StreamStats can be found at http://water.usgs.gov/osw/streamstats/pennsylvania.html

Predictive regression equations have been developed for statistics other than these represented in Stuckey (2006), although only for portions of Pennsylvania. A recent USGS report, "Selected Streamflow Statistics and Regression Equations for Predicting Statistics at Stream Locations in Monroe County, Pennsylvania" by Thompson and Hoffman (2006) estimates a wide range of statistics. Statistics included in this report that were not included in Stuckey (2006) are: mean monthly flows (12), exceedance flow statistics (Q1-Q99), and the mean annual baseflow. Standard errors of prediction in this study were relatively low, although as in Stuckey (2006), extreme low flow statistics tended to have higher errors associated with them. The report also identified that predictions were best for ungaged locations in which drainage area is between 2 and 350 square miles. In addition, Stuckey and Reed (2000), developed regional regression equations statewide for flood flows including the 10, 25, 50, 100, and 500 year flows. These equations were designed to be used only on basins with drainage areas below 2,000 square miles and above 1.5 square miles. This work has been revised by USGS PA, with new equations and revised regions as a result. The associated publication should be released in July 2008.

As described in Section 3, The Nature Conservancy evaluated the set of 195 PA USGS index gages considered least impaired due to limited upstream regulation, diversion, and mining impacts (Stuckey 2006). The Nature Conservancy screened and reduced this list of gages further using additional criteria related to land cover (<15% urban in catchment) and period of record (>15 years). The revised list of 136 continuous-record, least impaired gages may be the most appropriate set of index gages for use in developing regression equations that estimate unimpaired streamflow at ungaged sites due to their relatively long records and limited urban development.

Other regression-based approaches of note include the Region-of-Influence Method. This method is similar to the work done by Stuckey (2006), but can be used to define subregions within a state based on "predictor variable space" instead of geographic space (Eng et al. 2005). In such cases, each ungaged site ends up with a unique "region" defined based on similarity of its basin characteristics rather than proximity. This approach is being used in a USGS project titled "Regional Determination of Hydrologic Requirements of Aquatic Ecosystems of Watersheds in Tennessee." This project builds upon an existing Region-of-Influence work done in the state to estimate flood frequency and magnitude, low flow statistics, and duration estimation. Ecologically relevant hydrologic statistics, defined through a process of correlation between ecological metrics and hydrologic statistics, will be estimated using the enhanced model.

A recently published regression approach (Sanborn and Bledsoe 2006) from the Western U.S. is promising as it was able to estimate 84 ecologically relevant streamflow metrics for ungaged sites over a large heterogeneous region. The approach stratified streamflow regimes of gaged locations, classified the regimes of ungaged streams based on watershed characteristics using discriminant analysis, and then used multiple regression analysis to predict ecologically relevant streamflow metrics for these ungaged streams. Statistics accurately predicted for ungaged streams were not solely magnitude, but represented timing, duration, frequency, and rate of change. The multiple regression models did not perform well for predictions of streamflow variability statistics. Many of the statistics are currently represented in the USGS Hydroecological Integrity Assessment Process (HIP) and the TNC Indicators of Hydrologic Alteration program so they may be quite useful for impact assessment using these tools.

Finally, in the United Kingdom, a regression-based approach has been implemented throughout the country through a simulation program called "Low Flows 2000" (Young et al. 2003). This program allows simulation of a range of natural flow regime statistics through regression techniques including annual and monthly exceedance flow statistics. It also has some capability to estimate altered flow statistics when combined with information on withdrawals or impoundment effects, thereby providing a basic "hydrologic foundation" for the country. *Case Study 3: Environmental Flow Standards to Meet the EU Water Framework Directive* provides context for use of this program.

Strengths and Weaknesses of Using Statistical Regression Equation Techniques to Develop Baseline Statistics

Regression-based approaches are valuable and well-tested for defining baseline conditions. They can provide a wide range of flow statistics that often have low standard errors of prediction. Managers are becoming more comfortable using these statistics for ungaged locations, as evidenced by the use of the predicted 7Q10 as part of the Water Analysis Screening Tool. It is also a relatively inexpensive way to develop baseline hydrologic statistics statewide.

However, the statistics that can be estimated currently (i.e., based on Stuckey 2006) do not represent the range of hydrologic statistics that characterize natural variability nor do they encompass those statistics that have been found to be ecologically relevant in site-specific studies (Poff et al. 1997). Relatively simple extreme low flow, base flow and mean flow statistics limit how current or future flow changes relative to unregulated flows can be described in ways that are ecologically meaningful. Such a regression model can only generally predict magnitude statistics over the period of record. However, as referenced above, recent work by Sanborn and Bledsoe (2006) may provide an improved way to use a regression approach to define more ecologically significant flow metrics.

Enhancing StreamStats with additional ecologically-relevant flow statistics that cover the range from low to high flows, including exceedance flow values, is an option for developing a statewide hydrologic baseline. Development of these additional regression equations and combining them with water use information in a PA Water Management Decision Support System (PA DSS) to develop current condition estimates is described and costed as an option in the Project Options Table (Appendix 1). This may be the least time- and cost-intensive approach to developing a hydrologic foundation. However, key limitations identified include the inability to develop estimated daily flow time series hydrographs which constrains both analysis of hydrologic alteration and development of flow-ecology relationships. Statistics that can not be or are typically not calculated through regression approaches include frequency and timing statistics. This weakness is significant given the goals for a water management approach that are laid out in this document.

There are other weaknesses to these approaches. First, they often directly incorporate land use (% forest, % urban) into the regression equations, leading to estimates of "unregulated flows" rather than "unimpaired flows." Estimates of unimpaired flows are, of course, very difficult to test for strength of prediction and require more assumptions about "natural" land use. The regression approach also can not fully account for the effects of karst terrain or underground mines that may lead to movement of water across basins.

Other limitations are associated with errors of estimation. Standard errors of prediction can become significant when estimating hydrologic statistics that are infrequent or extreme. Notably, 7Q10 is one of these statistics, and is currently being used as the sole baseline for the Water Analysis Screening Tool. Decision-makers in Pennsylvania should be clear about the level of error they are willing to accept in developing a baseline flow condition. In addition, there are often relatively high errors associated with estimating flow conditions at ungaged sites with very large or very small drainage areas. Again, since baseline flow conditions will likely need to be simulated at a range of basin sizes it is necessary to clearly analyze errors when baseline flow conditions in headwaters or major rivers are being estimated. For larger rivers, it may be appropriate to use other, more sophisticated, techniques to estimate baseline flows.

The region-of-influence method has not been used to date in Pennsylvania, though it has been examined. A USGS study estimated flood-frequency characteristics using both generalized least-square regression and the region-of-influence approach (Koltun 2003). The findings were that for all recurrence intervals, the generalized least-square regression approach (e.g., Stuckey 2006) was superior in performance. These issues make the region-of-influence approach less attractive, although it still has potential.

2.3.2 Statistical Techniques to Develop Daily Flow Time Series Data

This set of statistical approaches, which essentially builds upon regression approaches described above, is designed to produce daily flow time series of unregulated flows without use of the more expensive hydrologic process modeling. In general, these approaches are designed around transferring a time series of daily flows from an existing unregulated gage site to an ungaged location. The flow duration curve is used as the means of transfer, since it provides a summary of flow conditions over a given time period at a site in common terms (exceedance frequencies).

The original **QPPQ-transform method** developed by Fennessey (1994) assumes an underlying model for daily streamflow and uses regional-regression equations to relate measurable basin characteristics to parameters used to characterize the assumed underlying process. The regression equations result in a flow duration curve (FDC) at the ungaged site. The FDC is then transformed to a time series of streamflow through an index gage, which transfers the timing of the daily flows at the index gage to the ungaged site by equating the exceedance probabilities at an index gage and the ungaged site. The original method was used by Waldron and Archfield (2006) to estimate monthly streamflows to drinking-water reservoirs in Massachusetts.

This transformation of a flow-duration curve to a time series of streamflow is illustrated in Figure 2.1, which is from Fennessey (1994) and Waldron and Archfield (2006).

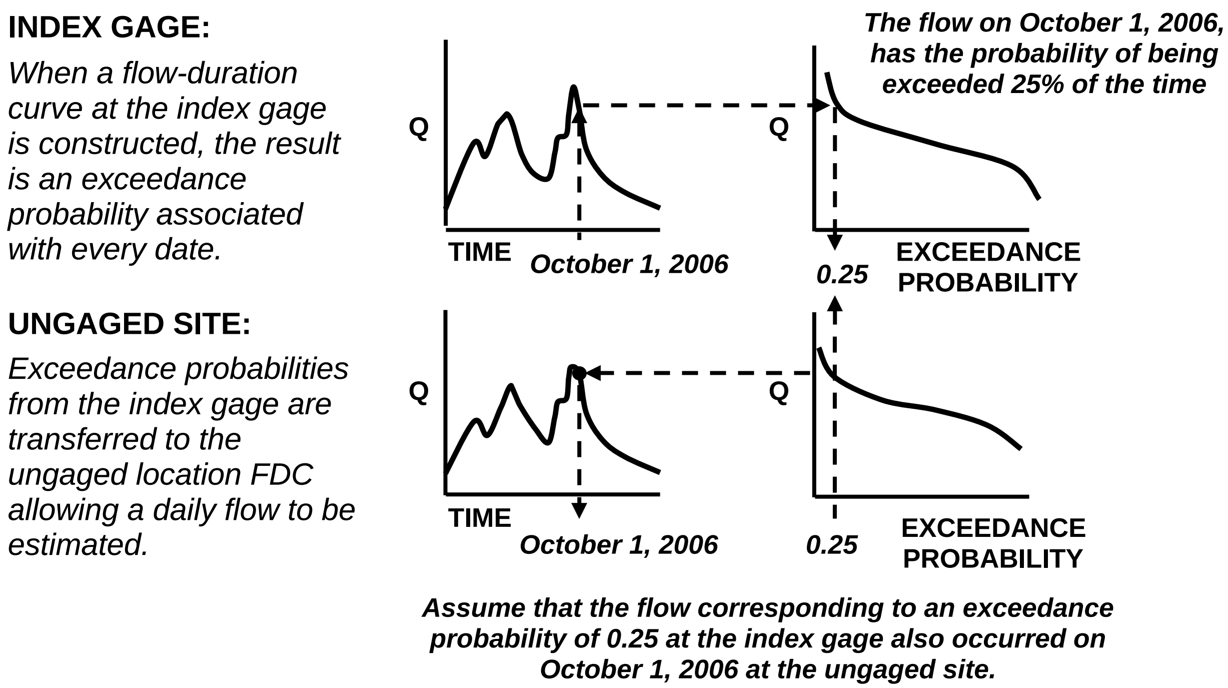

Figure 2.1 Hypothetical example of the modified QPPQ methodology.

The regression equations used in the original QPPQ-transform method were specifically developed for the Northeast, using gage stations from throughout the region (including Pennsylvania). Therefore, the equations could be applied in Pennsylvania currently without modification. The equations require that 6 basin characteristics be estimated at the ungaged site, including basin area, average annual precipitation, average annual snowfall, channel slope, elevation, and soil retention number. This approach was examined by USGS in Massachusetts (USGS MA) using basin characteristics derived from available GIS data (Waldron and Archfield 2006). The study found that simulated flows closely matched gaged flows at HCDN (Hydro-Climatic Data Network, USGS) sites in Massachusetts with long term records. The exception was extreme high and low flows, which had substantially larger errors associated with them. The study also noted that the QPPQ flow duration curve equation is constrained by the basin characteristics used to develop the equation. For example, small watershed sizes (below 1.5 square miles) are outside of the range used to develop the initial regression equations. Another limitation of the method is that soil characteristics, as included in the QPPQ flow duration curve equation, are currently difficult to derive from GIS layers.

Additional work has been completed by the USGS MA, which compared the assumption of an underlying process to characterize streamflow with other methods to estimate the flow-duration curve at ungaged sites (Archfield et al. 2007). In comparing the original QPPQ-transform method, Archfield et al. (2008) determined that regional regression for individual flows on the FDC and interpolation between those flows to obtain a continuous FDC provided better agreement between observed and estimated streamflows than the original QPPQ-transform, which assumed an underlying model for daily streamflow. Regional regression equations have been developed for flows at 19 exceedance probabilities along the FDC.

The USGS MA has also developed a method to select the most appropriate index gage to use in the transformation of the FDC to a time series of flows; however, the selection of the index gage is still an active area of research in the USGS MA study (S. Archfield, USGS MA, personal communication). The current selection criteria picks the index gage that minimizes the sum of the distance and relative percent differences between the basin characteristics (Archfield and Vogel 2008). Linking ungaged sites to index gages could also be accomplished through similarity of weighted basin characteristics in an approach similar to the region-of-influence method currently implemented in the SYE tool. This linkage could be made more efficient through a statewide classification that might approximate the hydrologic landscape regions found in Wolock (2003). The gains in accuracy of estimation associated with adding more complexity to the linkage of ungaged locations to index gages has not been fully explored and would likely vary from state to state. Another factor to consider is the period of record of the index gage used to develop the estimate of the ungaged flow record. These options could be pursued in Pennsylvania if this type of approach for estimating a daily time series was selected.

The limitations of the modified QPPQ-transform method are similar to the original QPPQ method in that the applicability of the equations is limited to ungaged basins whose characteristics are within the ranges under which the equations were developed. Errors in estimated flows are larger for high-flow values (flows corresponding to exceedance probabilities of 0.01 or less) than for flows at other exceedance probabilities; however, the modified method offers substantial improvement in the estimation of low flows over the original method (Archfield et al. 2007). Full documentation and publication of these methods is expected by winter 2009.

Other statistical approaches for developing daily baseline flow time series that have been published include Smakhtin (1999) and Sharma et al. (1997). Sharma et al. (1997) uses a non-parametric approach to generate synthetic daily flow time series. This work was tested in Utah and has apparently not been applied or tested in the East so it will not be dealt with further here. Smakhtin (1999) describes a non-linear spatial interpolation technique that, like the QPPQ transform, uses flow duration curves as a basis for developing estimates of a daily time series. The primary difference between this approach and the QPPQ method is that index gage "sources" for simulating ungaged sites are weighted based on degree of similarity between the source and the ungaged site. This approach also focuses on developing monthly flow duration curves across the period of record for exceedance probability values ranging from .01 to 99.99 percentile. The ungaged site estimates are done through a weighted average of all estimated flows from the source sites. This approach also has not been applied in the Eastern U.S. based on the literature reviewed. Although it may have merit, its lack of testing makes it less desirable as an approach for estimating unregulated daily flows.

Strengths and Weaknesses of Statistical Approaches to Daily Flow Time Series Simulation

The primary strength of using this set of approaches is the ability to develop a daily flow time series without the expense of a hydrologic process modeling approach. First, this allows for the calculation of an extremely wide range of ecologically-relevant flow statistics using software tools such as the USGS Hydroecological Assessment Tool (HAT) and TNC Indicators of Hydrologic Alteration program (IHA) described in Section 4. Simulation of a flow time series, whether it is monthly or daily, would allow for statistics of frequency, duration, timing, and rate of change (if daily) to be developed. Recent scientific literature has emphasized the importance of these attributes of natural flow variability to the ecological health of river and stream systems (e.g., Poff et al. 1997).

Second, when index gage information is available, it permits estimation of hydrologic conditions during particular periods of interest. These periods may be periods of extreme drought (e.g., mid 1960s) or periods in which biological data is available to facilitate the development of response relationships between flow alteration and biological condition. Although water use data will rarely be available at a daily time step, estimating how monthly or even annual withdrawal values impact daily flows is likely to be important to decision-making.

Of the two primary approaches reviewed for developing flow duration curves at ungaged sites, regression of duration statistics has the advantage of being readily available as soon as regression equation estimates have been developed for full range of exceedance flow statistics. In Pennsylvania, only a subset of these statistics can be estimated currently. Yet the additional cost of developing regression equations for these exceedance flow statistics would be moderate relative to the other approaches listed below.

One weakness of these methodologies is that they are generally less familiar to hydrologists and have not been tested as rigorously for sources of error. Although initial testing by USGS Massachusetts has shown favorable results, pursuing an approach that develops flow time series information will need to be tested for error to better understand the limitations of the estimate for basins in Pennsylvania. As with all regression techniques, estimates of extremely high or low daily flow are expected to have relatively high errors associated with them. The relative level of error and the range of likely acceptability are currently being examined by USGS in Massachusetts (S. Archfield, USGS MA, personal communication). Finally, all statistical approaches share a key weakness that they are limited in their ability to account for impacts of land use, climate change, or represent areas with complicated hydrology.

2.3.3 Surface and Groundwater Hydrologic Process Modeling Approaches

A final major category of approaches is the use of detailed rainfall-runoff or watershed/groundwater process modeling approaches to synthesize baseline flow conditions. There are a number of examples of this type of work in Pennsylvania and in the region, but none at the scale of the entire state.

The **Precipitation Run-off Modeling System (PRMS)** was developed to evaluate the impacts of various combinations of precipitation, climate, and land use on streamflow, sediment yields, and general basin hydrology (Leavesley 1983). It therefore can develop daily time series streamflow estimates for various land use scenarios, including historic natural cover or future development. PRMS has been used by PA USGS in the Highlands region in conjunction with the US Forest Service (USGS Scope of Work 2006). PRMS can be used to estimate baseline hydrology at daily or monthly time steps. Its application in the PA Highlands has been focused on monthly water budgeting, which is further discussed below in the section on estimating current hydrologic conditions.

Hydrological Simulation Program Fortran (HSPF) modeling has been used in a number of locations in PA, including in the Christina River basin (http://pa.water.usgs.gov/malvern/chesco_christina.html), in Fayette County (the Poplar Run Basin), and in Purdy & Ariel Creek in Northeast Pennsylvania. This modeling approach requires more detailed inputs that PRMS and is often coupled with MODFLOW to account for groundwater impacts. It requires significant site-based information in order to develop accurate estimates of watershed hydrology. It has been used effectively as part of instream flow studies in a number of areas, including in the Ipswich River in Massachusetts in a study that looked at impacts from a diverse set of water withdrawals in relation to streamflow goals (Zarriello and Ries 2000). A statewide application of HSPF that incorporates water use data is described below with an example from Virginia.

In New Jersey, a NJ USGS team led by Jonathan Kennen used **TOPMODEL**, combined with TR-55 (to include impervious surface impacts), to simulate unregulated flow conditions for locations across the state from 1948-2000 (Kennen et al. 2008). This Statewide Watershed Runoff Model (SWRM) daily flow simulation was a unique statewide application designed to support water management decision-making in New Jersey. TOPMODEL, like PRMS, is a relatively simple model that uses hourly/daily precipitation and daily temperature data as well as topography and coarse soils information. The overall approach, SWRM, was applied to 856 sites across New Jersey in which aquatic invertebrate community data was available. The vast majority of these sites are ungaged. As further described below, SWRM accounted for water withdrawals and discharges through annual average values that were directly subtracted from or added to the daily stream hydrograph. Groundwater withdrawals were included in this model and incorporated into the estimates of current hydrologic condition (Kennen et al. 2008).

Each of these surface water modeling approaches has been successfully coupled with groundwater models, particularly **MODFLOW**. MODFLOW is a three-dimensional groundwater flow model that can be used to examine the impacts of well withdrawals, changes in recharge, changes in evaporation, and impacts to associated stream systems (Harbaugh et al. 2000). In the context of this report, MODFLOW would be particularly useful for defining the current or future streamflow regime due to impacts from water withdrawals or land use, assuming resources were available to apply it in particular areas.

Strengths and Weaknesses of Surface and Groundwater Modeling Approaches

These approaches allow for simulation of daily flow regimes, and even subdaily flow regimes, over a multi-year period with varying land uses. Thus, they can be well suited for supporting decisions associated with instream flow and water allocation. Their major weaknesses are the extensive data input parameters and expertise required to run, and the time and costs of developing the models. Due to these challenges, it may be difficult to apply these models across a broad enough area to allow for site-specific decision-making throughout Pennsylvania. However, the statewide HSPF model in Virginia and the TOPMODEL applications in New Jersey demonstrate that these approaches are feasible given adequate resources and time. Key strengths and weaknesses of these approaches are summarized in the Project Options Table (Appendix 1).

The statewide application of HSPF in Virginia required significant resources, a portion of which was covered by the federal government. As part of the Project Options Table, Appendix 1, it was estimated that statewide development of HSPF for development of baseline and current conditions would likely cost over a million dollars. We believe this is unlikely to meet the resource constraints of Pennsylvania agencies. Setting aside its cost, the HSPF model has some clear strengths as described in *Case Study 1: A Model for Water Supply Planning in Virginia* and documented in the Project Options Table. In addition, although it may be unlikely in the near term that the state of Pennsylvania will develop HSPF for the rest of the state due to resource constraints, use of the HSPF model for the Susquehanna River Basin (developed by the Chesapeake Bay Program) for defining baseline and current conditions might be possible at the spatial scale of the existing model (approximately 50-square-mile and larger basins). As far as limitations, HSPF modeling is not only data intensive for model development and calibration, but difficult and computing power intensive to run, and requires dedicated and experienced staff to manage. It also can not handle a large number of control points in any watershed. For all of these reasons, an HSPF approach may not be appropriate for desktop applications with Pennsylvania agencies.

The New Jersey Statewide Watershed Runoff Model was developed at a relatively moderate cost of approximately $200,000. In our evaluation in the Project Options Table, we include an estimate of $300,000-$400,000 to develop baseline hydrologic conditions statewide though TOPMODEL rainfall-runoff modeling. This cost also includes developing current flow time series information and incorporating both results in a Decision Support System. As a rainfall-runoff approach, TOPMODEL has the advantage of not only estimating historic baseline conditions, but incorporating estimates of unregulated flows under various climate change scenarios through modeled patterns of annual rainfall. As a weakness, the quality and spatial coverage of precipitation data is uneven across Pennsylvania and may lead to high error in baseline hydrologic estimates on ungaged streams in parts of the Commonwealth.

PRMS modeling is generally intermediate in cost between HSPF modeling and a statewide runoff model based in TOPMODEL. This model would also be capable of estimating current hydrologic conditions if combined with WAST or an updated statewide spatial water use database. Estimated costs, strengths and weaknesses of this approach are further detailed in the Project Options Table (Appendix 1).

2.4 Potential Approaches Considered for Defining Current or Future Conditions

As reviewed in the Project Options Table (Appendix 1), there are a number of options to develop current or future hydrologic conditions and all are contingent on the approach used for estimation of baseline hydrologic conditions. All of these approaches are dependent upon the quality of data in a spatial database of known and estimated water usage, withdrawals and discharges, statewide. Fortunately, Pennsylvania has a first iteration of such a database included in the Water Analysis Screening Tool, and the current quality and potential of this database will be addressed below. In addition to having a spatial database of water use to simulate current conditions and the capability to estimate future conditions, a decision support system interface will be required to make it usable by state resource agency staff and not just model developers. To that end, this review will build on the discussion of baseline simulation approaches above and include consideration of the potential development of a Pennsylvania Water Management Decision Support System (PA DSS).

2.4.1 Statistical Approaches Coupled with a Water Use Database

The first approach included in the Project Options Table (Appendix 1), "Enhanced StreamStats with Water Use Information," would take the static baseline flow statistics that result from the regression equation approach of PA StreamStats and modify them at pour points using withdrawal and discharge information relevant at each pour point. Current conditions could be developed for annual magnitude statistics only, through an arithmetic approach dependent on the temporal resolution of water use and discharge data. Future conditions, similarly, could only be developed for magnitude statistics. The overall approach would be analogous to the current Water Analysis Screening Tool, but with a set of hydrologic statistics that would expand beyond 7Q10. Unlike hydrologic processes modeling, land use impacts would not be able to be incorporated. It would be extremely difficult to incorporate impacts of reservoir operations. Other strengths and weaknesses are described in the Project Options Table (Appendix 1).

"Flow Duration Curve Regression and Decision Support Application," as reviewed in the Project Options Table (Appendix 1), is based on the promising statistical approach to daily flow simulation described above. The sole current application is work by Massachusetts USGS, in cooperation with the Massachusetts Department of Environmental Protection, to develop the Sustainable-Yield Estimator (SYE) tool. The Massachusetts SYE is an interactive decision-support application which can estimate the unimpacted and impacted continuous, daily hydrograph from October 1, 1960 to September 30, 2004 at user-selected ungaged sites in Massachusetts. The application includes all relevant, geo-referenced information on rates of permitted withdrawals, reported withdrawals, estimate withdrawals, and return flows in Massachusetts. The Massachusetts SYE was designed as a desktop application that employs ArcGIS, a geographic information system (GIS) (Figure 2.2), and Microsoft Excel and Microsoft Access, commonly-used and widely-available spreadsheet and database programs, respectively. Baseline and current condition daily streamflow results can be easily exported to the USGS Hydroecological Assessment Tool (HAT) and TNC Indicators of Hydrologic Alteration program (IHA) to analyze flow alteration at locations of interest. The results can also be used in the development of flow alteration-ecological response curves that can serve the basis for instream flow criteria.

In addition, based on user-defined constraints such as existing water-use in the basin and instream-flow regimes necessary for sustainability of aquatic habitat, the SYE computes the sustainable yield of the basin, defined as the difference between the estimated hydrograph and the user-specified instream-flow regime. Users can quickly and easily compute the "sustainable yield" of the basin for a variety of water-management scenarios and instream-flow regimes. The calculation of sustainable yield is based not only on user-specified instream-flow targets (time-varying or constant flow targets) but also for a user-specified time period (drought year, wet year, or average year) (Figure 2.3).

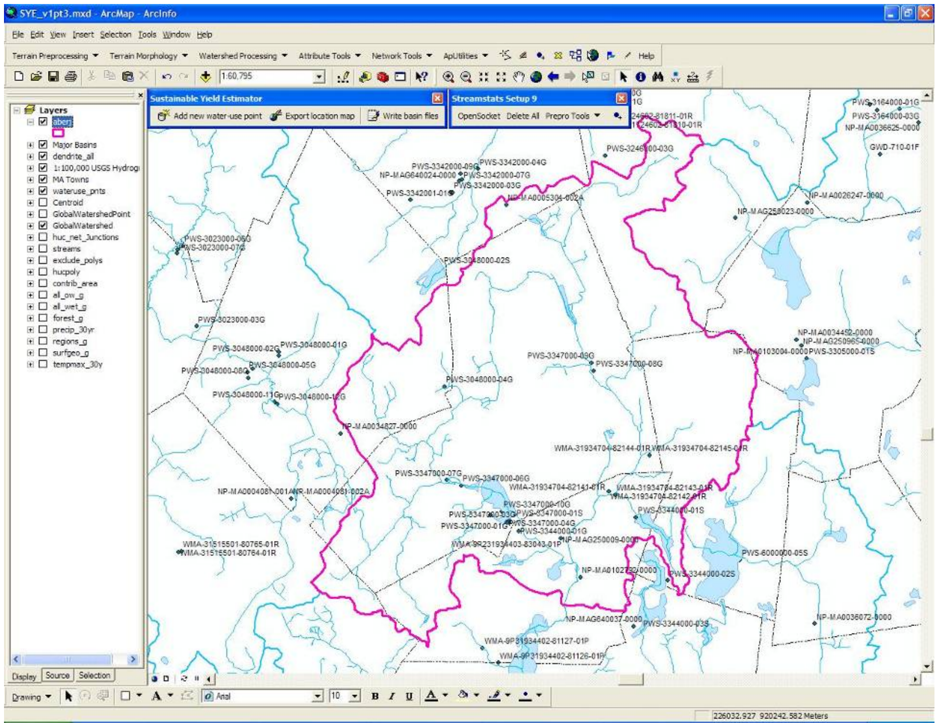

Figure 2.2 ArcGIS user interface and related toolbars for the Massachusetts Sustainable-Yield Estimator. Delineated watershed shown above is located approximately 15 kilometers north of Boston, Massachusetts. The user clicks on the stream location to delineate an on-the-fly watershed, compute basin characteristics, and query the water-use database for points within the watershed.

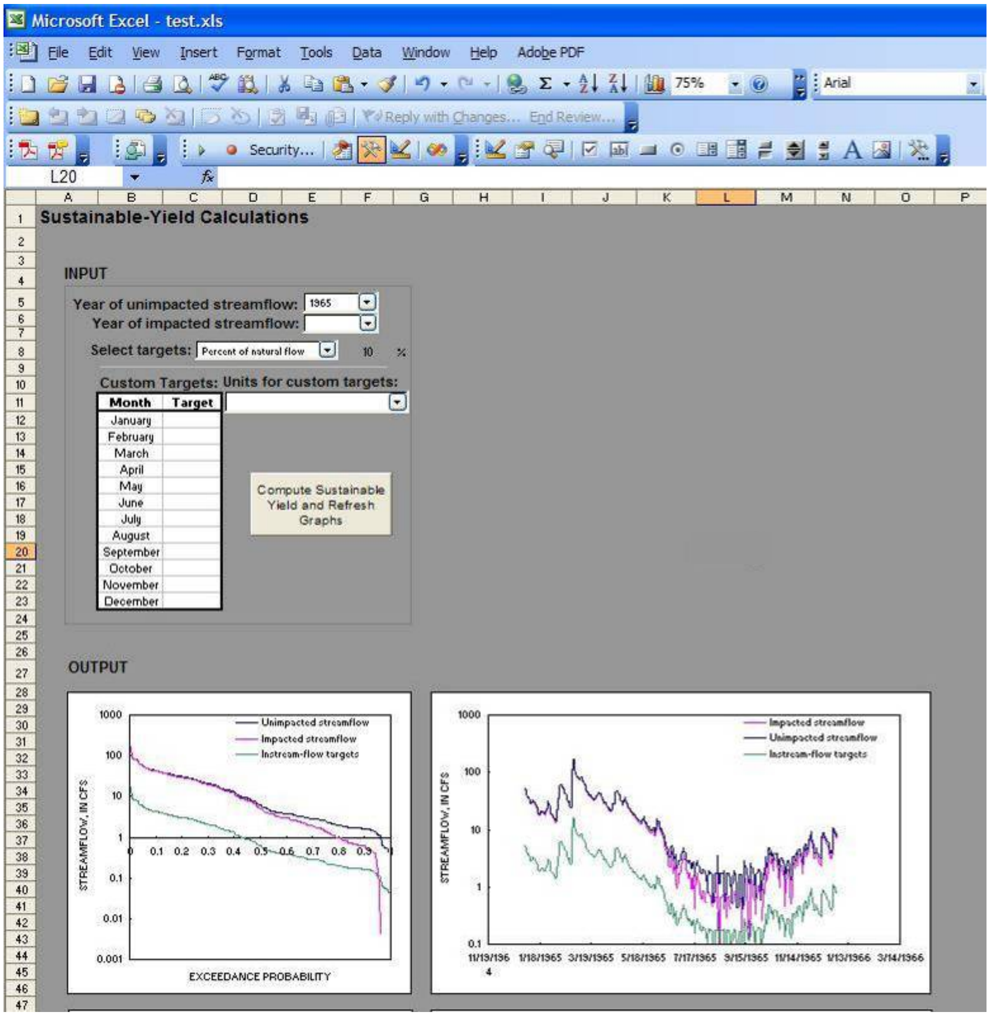

Figure 2.3 Microsoft Excel user interface for the calculation of sustainable yield in the Massachusetts Sustainable Yield Estimator (SYE) tool. Users can quickly and easily compare unimpacted (shown in black on the graphs) and impacted streamflows (shown in pink on the graphs) for any combination of years for which data is available. Estimated streamflows can be compared to default or custom instream-flow targets (shown in green on the graphs).

To define actual withdrawals, return flows, and streamflow depletion, the project incorporates five years of annual, site-specific withdrawal rates up to the most recent available year. These water use data are taken from the Massachusetts DEP database of annual withdrawals maintained for all public-water supplies and other permitted withdrawals. Other withdrawals and return flows "will be estimated by using GIS land-use coverages and the standard methods of the USGS National Water-Use Information Program." Paper records of monthly flow data (Annual Statistical Reports) from a representative subset of about 50 cities and towns for the 2001-2005 period were analyzed to determine ratios of monthly to mean-annual withdrawals for each month of the year. Average values of these monthly ratios, the variability of the ratio, and their spatial trends across the state were evaluated. Monthly return-flows were obtained from the EPA-NPDES database (Weiskel and Garabedian 2005).

The Sustainable Yield Estimator (SYE) application is designed primarily for the use of Massachusetts DEP professional staff, and will be served locally from a CD-ROM. Users will be able to input either (1) constant rates of withdrawal and return flow for the basin, or (2) a time-varying set of human inflows and outflows (reflective of the typical annual water-use cycle for the part of the State in question). If the time-varying option is chosen, the STRMDPL program (Zarriello and Ries 2000) will be used to estimate the time-varying pattern of streamflow depletion caused by groundwater withdrawals in the basin (Weiskel and Garabedian 2005).

This approach provides some significant advantages because it simulates baseline, unregulated flows and current condition flows over a modeling period of 1960-2004. The SYE's flexibility allows for estimation of these flows at any location in the state (within the bounds of the regression equations) and will allow current condition to be examined in a number of ways. In Massachusetts, like Pennsylvania, data on major withdrawals is only available for a limited time period and water use is not consistently reported on a monthly time scale. The SYE application will allow decision-makers to examine the current condition several ways, including developing a 45-year current condition record based on: (1) a single year withdrawal and return flow volume; (2) multiple years of withdrawals and return flows. Each of these options can have a seasonally varying time series based on the estimation approaches described above. This approach is also relatively moderate in cost, with a total project budget of $450,000. A similar approach has been proposed for the fractured bedrock portions of Maryland (K. Ries, MD USGS, personal communication).

Weaknesses of this SYE-type approach to developing current or future conditions are, as with the StreamStats approach, the inability to effectively account for land use and climate change impacts. In addition, accounting for reservoir operations can only be done through use of simplifying rules. More accurate output requires an integrated reservoir operations and water management model (e.g., OASIS, WEAP). Other strengths and weaknesses are covered in the Project Options Table.

2.4.2 Hydrologic Process Modeling and Water Budgeting

Three well established hydrologic process models that can produce time series information on baseline and current conditions statewide over a modeling period of record, TOPMODEL, PRMS, and HSPF, are included in the Project Options Table (Appendix 1). All three are discussed in detail above with regards to their capability for modeling baseline hydrologic conditions and are constrained by the quality, temporal resolution, and spatial extent of the applicable water use database. In areas of significant groundwater extraction, all three can be coupled with a groundwater model (e.g., MODFLOW) or an analytical approach (see below) to increase accuracy. It should be noted, however, that MODFLOW can be very costly to apply at broad spatial scales.

TOPMODEL, generally the least costly of the three models, has been demonstrated for this type of use through the New Jersey Statewide Watershed Runoff Model (SWRM) (see Kennen et al. 2008). NJ SWRM provides an example of a Decision Support System which incorporated information on water use into a TOPMODEL streamflow simulation application. The NJ

SWRM accounts for water withdrawals and discharges by subtracting or adding average annual water use to the daily stream hydrograph. Groundwater withdrawals were handled by adding to a term in the model called the "saturation deficit" which is essentially "the depth of water table multiplied by the readily drained soil porosity" (Kennen et al. 2008). A simple channel routing approach was applied through the GIS platform of the model and lake storage was accounted for.

The accuracy of the simulation may have been improved by regionalizing the state based on streamflow regime types. Accuracy in estimating altered conditions definitely would have been improved if daily withdrawal and discharge data were available statewide. Below is a figure of the inputs to and outputs from the SWRM (Figure 2.4). Although this project explicitly linked ecological community indices with hydrologic indices, it did not attempt to define hydrologic alteration in the sense of simulating a baseline and comparing it to a current condition hydrograph.

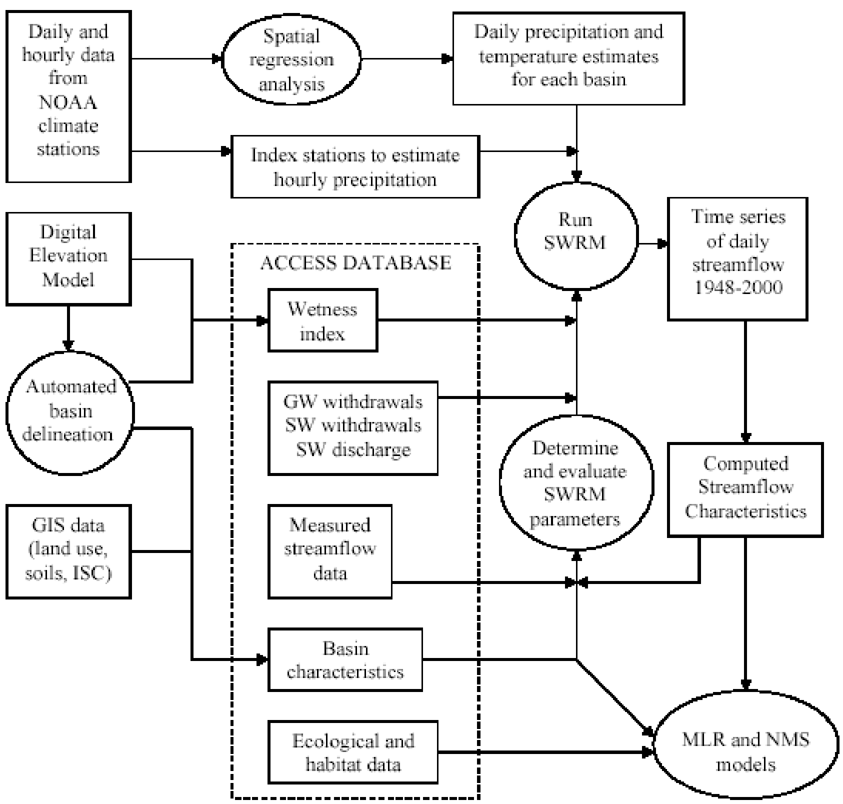

Figure 2.4 New Jersey Statewide Runoff Model Schematic (from Kennen et al. 2008)

The **Precipitation Run-off Modeling System (PRMS)**, introduced in the baseline discussion above, has been used by PA USGS in the Highlands region in conjunction with the US Forest Service (USGS Scope of Work 2006). This project includes a "detailed analysis of existing groundwater and surface-water data for the purpose of preparing a hydrologic system water budget of the Highlands region" (Figure 2.5). This project covers a substantial area of the state, including 95 Hydrologic Unit Codes 14 (HUC14s) that are wholly or partially within the Highlands region of Pennsylvania. As described in the Scope of Work, the water budget accounts for average annual withdrawals and transfer of water from many existing reservoirs.

The watershed model has been used to project potential changes in annual runoff and baseflow characteristics due to changes in land use. This work did not include examination of changes in the magnitude and spatial distribution of water withdrawals and discharges, but this could be added into the project at a later date (Jennifer Cox, Regional Planning Association, personal communication). The model's relatively simple water balance approach can incorporate water withdrawal impacts from surface water, but would need to be linked to a groundwater model for groundwater withdrawal impacts. One of the major sources of error for PRMS is the precipitation input data, the quality of which varies depending upon which portion of the state being simulated.

This recently completed application of PRMS to the Pennsylvania Highlands, an area of approximately 2,100 square miles, cost approximately $80,000. Model output was a general water balance (Precipation_in, Baseflow_out, Annual Runoff_out, Evapotranspiration_out) for each of the 95 14-digit HUC watersheds. Calibration was done for total annual streamflow. Beyond visual confirmation that the simulated daily hydrograph seemed reasonable, no attempt was made to calibrate to daily, peak, or low flows which would have resulted in greater time and cost. Thus, due to the temporal resolution of the study and level of modeling effort, the Pennsylvania Highlands Study can not assess how many flow statistics of interest, like those representing high flow magnitude and timing, duration, frequency, and rate-of-change will change with land use. The final results will be published by US Forest Service (E. Koerkle, USGS PA, personal communication).

Figure 2.5 Map of the Pennsylvania Highlands Study Region (USGS Scope of Work, 2006)

Hydrological Simulation Program Fortran (HSPF): In a rare case of large scale application of HSPF, Virginia has developed a statewide HSPF model integrated with water use information into a DSS for water supply planning. This model was developed at significant cost with contributions from the U.S. EPA Chesapeake Bay Program, USGS VA, and the Virginia

Department of Conservation and Recreation. This model, as currently being used by the Virginia Department of Environmental Quality, can be used to develop baseline and current hydrologic conditions throughout the state of Virginia over the period 1985-2005 at an hourly time step. It is also available for the Susquehanna Basin portion of Pennsylvania. The capabilities of this sophisticated water supply planning model are described in *Case Study 1: A Model for Water Supply Planning in Virginia*.

Using primarily the **Soil and Water Assessment Tool (SWAT),** the U.S. EPA ORD Science and Technology Network for Sustainability supported an "ecological flow" project on Pocono Creek, in Monroe County, PA that is evaluating the effects of land use change on both groundwater and stream flow, and the effect of those flow changes on the aquatic ecology (in particular trout), of Pocono Creek. Pocono Creek is a high quality wild trout stream whose watershed is threatened by a high rate of development with its accompanying land use change. Collaborating in the EPA and the Delaware River Basin Commission led study, were the USGS-Fort Collins, Colorado Science Center, the USGS PA Water Science Center, the PA Fish and Boat Commission, the PA Department of Environmental Protection, the Monroe County Conservation District, the Monroe County Planning Commission and other local stakeholders.

A groundwater flow model and a hydrologic model of the Pocono Creek watershed were developed by the USGS PA office and the EPA Office of Research and Development respectively. The groundwater flow model (**MODFLOW**) used the areal recharge values from the EPA hydrologic model and was calibrated to base flow measurements at 27 sites in the watershed. The groundwater model was used to examine impacts of groundwater withdrawals and land use changes on ground water baseflow.

The hydrologic model of the watershed utilized the Soil and Water Assessment Tool (SWAT) modeling framework to estimate the effect of land use changes on Pocono Creek stream flow. SWAT is a distributed, process-based watershed model with a significant number of empirical relationships and is one of the most suitable models for assessing the impact of land use change on stream flow. The model links precipitation data to the physical characteristics of the watershed to simulate daily stream flow. For modeling purposes, the watershed was divided into 37 sub-basins, ranging in size from 0.1 square miles to about 5 square miles. The model was calibrated to daily stream flow data for the period July 1, 2002 to May 31, 2004 and validated for the period June 1, 2004 to April 30, 2005.

The models were used to estimate the effect of land use changes (increased imperviousness) in Pocono Creek watershed on groundwater recharge rates and the stream flow of Pocono Creek. A future "build out" scenario and a forested watershed (predevelopment) scenario were modeled and compared to the existing condition scenario. A 20-year Monte Carlo statistical simulation based on historical precipitation records was performed with the watershed model to evaluate the various land use scenarios.

For the build out scenario, watershed model results indicate that the watershed-averaged groundwater recharge is predicted to decline by 31%, causing the average daily baseflow to be reduced by 31%. The seven day, ten year low flow (7Q10), is expected to decline by 11%, and the monthly median daily flow is expected to be reduced by 10% on the average. The monthly peak of simulated daily flows and annual maximum daily flow on the average are predicted to increase by 21% and 19%, respectively. In addition, subwatersheds were ranked based on their relative impact on watershed response to anticipated land developments (Hantush and Kalin, 2006).

In addition to estimating the effects of land use changes on stream flow, an important project goal is to relate those stream flow changes to their effect on the aquatic ecosystem. This is being done through the Hydreoecological Integrity Assessment Process (HIP) approach described in the following section. While this work is still ongoing, preliminary results for the build out scenario indicate an ecologically significant degree of flow alteration using the Range of Variability Approach (RVA) as documented in Richter et al (1997). In addition, work to relate HIP flow metrics to trout population data in Pennsylvania is also ongoing (Charles App, U.S. EPA, personal communication.)

In addition, several detailed water budgets have been developed for watersheds within Pennsylvania. These water budgets provide valuable information that can be fed into models describing current or future flow conditions. These will not be discussed in depth, but any approach for developing current or future condition at the regional or state scale would benefit from water budgeting information, especially if the water budgeting is done at a monthly or daily time scale.

Sloto and Buxton (2005) defined water budgets for three small watersheds in Eastern PA. These budgets incorporated information on water imported, consumptive use, and surface and groundwater withdrawals exported from each watershed. In addition, the water budget demonstrated how water moved within the watershed. GIS was used to make withdrawal and discharge sites spatially explicit. The result was annual water budgets, in which streamflow was quantified in inches along with all other factors. In each of the watersheds, withdrawals and return flows were defined for at least 10 years, but no estimate of change from unregulated (baseline) conditions was made for key flow statistics (e.g., annual or period-of-record summer low flow). Detailed water budget information is also available through the Northern Lancaster County Groundwater Study (Edwards and Pody 2005).

Generally, estimates of current or future hydrologic conditions can be made with annual water use and return flow data, but are made much more realistic with assumptions about seasonal use patterns. This type of estimation is included in the Texas (U.S.) Water Availability Model, which examines unregulated water availability, current allocation, and potential future allocation scenarios on a monthly time step and allows impacts on streamflows to be assessed at any location of interest within Texas (NRC 2005).

2.4.3 Linking Groundwater Withdrawals and Reservoir Impacts to Streamflows

Quantifying the effect of groundwater withdrawals and reservoirs on streamflows is a hurdle for accurately defining current and future flow conditions. Although many groundwater models that can be linked to surface water models exist (e.g., MODFLOW, BRANCH), most, if not all, are too expensive and data intensive for statewide application. Incorporating 2-D or 3-D groundwater models is recommended where these models already exist (e.g., Pocono Creek, French Creek) or in watersheds where groundwater movement is particularly complex (e.g., karst watersheds).

In the absence of existing models, the effects of well pumping can be estimated using analytic techniques. One example is a computer program developed in Massachusetts by USGS called STRMDEPL. This model is being used statewide in Massachusetts as part of the MA USGS Sustainable Yield Estimator project. The program includes two analytical methods, one that calculates unimpeded flow at the stream-aquifer boundary and another that calculates resistance to flow caused by semipervious streambed and streambank material (Zarriello et al. 2001). With this information, the effects of streamflow depletion over time can be calculated based on daily or monthly pumping data. This method, like others, assumes a simplified, homogenous aquifer and therefore has estimation error associated with it. Figure 2.6 illustrates the depletion associated with distance of withdrawals in Rhode Island (Zarriello and Bent 2004).

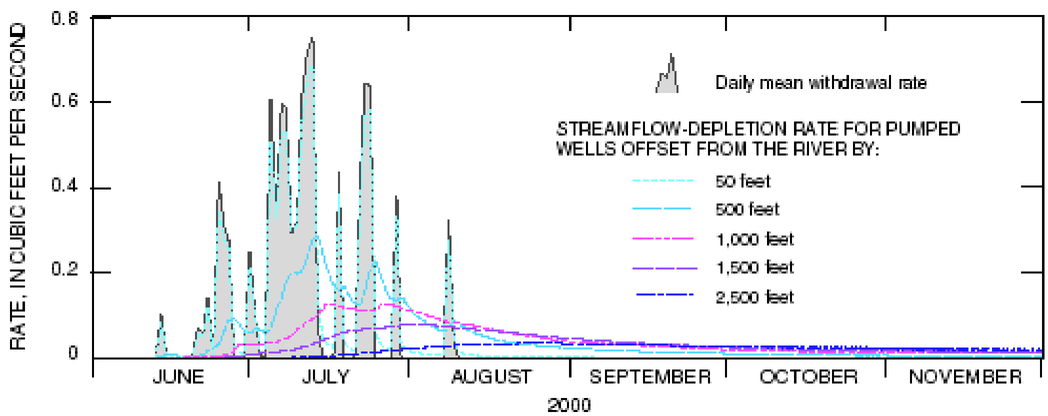

Figure 2.6 Observed daily mean irrigation withdrawals (2000) and corresponding streamflow depletions computed with the STRMDEPL program for hypothetical wells 50 to 2,500 feet from the river, Usquepaug–Queen River Basin, Rhode Island.

There are other analytical solutions available, but all involve assumptions about simplified aquifer properties. According to Zarriello et al. (2001), the most widely applied equation is documented in Jenkins (1968):

$$Q_s = Q_w \, erfc(U)$$

where

$$U = \sqrt{\frac{d^2 S}{4Tt}}$$

and

Q_s is the rate of streamflow depletion (cubic length per time);

Q_w is the pumping rate of the well (cubic length per time);

d is the perpendicular distance from the well to the stream (length);

S is the storativity (or specific yield) of the aquifer (dimensionless);

T is the transmissivity of the aquifer (square length per time); and

t is time.

In Michigan, as part of the Water Withdrawal Assessment Process (described in Case Study 4), USGS is leading the development of a withdrawal model that assesses the impacts of a potential withdrawal on streamflow. The model assumes that surface water withdrawals reduce streamflow in the closest stream segment by the volume withdrawn. For groundwater withdrawals, the model accounts for the fact that (1) the effect of a withdrawal may be distributed among multiple nearby stream segments; (2) it may take weeks, years, or decades for a groundwater withdrawal to affect streamflow; (3) intermittent groundwater withdrawals may have substantially less impact on streams than continuous withdrawals; and (4) the impact of a groundwater withdrawal will depend on the characteristics of the aquifer. This model is based on statewide estimates of aquifer characteristics that were generated for Michigan (http://gwmap.rsgis.msu.edu). For groundwater withdrawals, the model uses an analytical solution for streamflow depletion by a nearby well (Hunt 1999, 2003). For intermittent pumping, USGS MI uses an approach by Jenkins (1968; H. Reeves, USGS MI, personal communication). The withdrawal model accounts for the following factors: the type of withdrawal (surface water or groundwater); the amount and continuity of the withdrawal; the depth of the well; the distance of the well from the stream; and the aquifer properties. The result of the withdrawal model is that the reduction in streamflow can be calculated for every stream segment in Michigan, given the specific characteristics of a proposed withdrawal (Michigan Groundwater Conservation Advisory Council 2007).

To date, Pennsylvania has assumed a 1:1 impact in volume and time associated with groundwater withdrawals in keeping with state regulations. This is also reflected in the current version of the Water Analysis Screening Tool. This assumption was examined by a MODFLOW study in the French Creek Basin in Chester County, PA (Sloto 2004). The USGS found that typically over 90% of the groundwater extraction was derived from reductions in baseflow in the associated stream. This USGS study also found that "(1) if the contributing area of a well is in a basin, pumping will affect stream base flow and water levels in that basin whether the well is inside or outside that basin; (2) wells in different areas of a basin away from a divide produce a similar reduction in base flow; (3) a well within a basin will derive more water from diverted base flow and less water from storage than a well on or near a basin divide; and (4) the reduction in base flow at the mouth of the stream is the same for a well in the headwaters and a well downstream near the confluence."

Generally, given the complications associated with precisely defining the impacts of groundwater withdrawals on streamflow in terms of timing and volume, simple analytical approaches should be adequate for estimates of current or future condition.

Reservoir and lake storage also complicates the process of defining current or future condition. Reservoirs and lakes tend to delay runoff and dampen hydrologic variability. The current version of the Water Analysis Screening Tool "flags" pour points that have upstream reservoirs, but it does not estimate the effects of these reservoirs. Similarly, the Massachusetts Sustainable Yield Estimator does not explicitly deal with reservoirs. A more advanced approach in Pennsylvania that incorporates the storage effects of reservoirs could be modeled after the NJ SWRM (Kennen et al. 2007). In New Jersey, "an exponential decay function was applied to the fraction of runoff equivalent to the lake-affected fraction of the watershed." This would clearly be useful if a TOPMODEL or other rainfall-runoff model is used to develop baseline and current conditions, but could also be useful for other applications.

Operational water models have typically been used to incorporate the routing of water through a watershed with many reservoirs and complicated operating rules. Such models include OASIS, Riverware, RES-SIM, and WEAP21. Each of these more sophisticated water management models tends to be expensive to develop, but can incorporate the complications associated with reservoir operation rules. The OASIS model, for example, has been developed in both the Susquehanna and Delaware Basins, for the SRBC and DRBC respectively. The OASIS model, developed by Hydrologics, Inc. (see http://www.hydrologics.net/oasis.html), can provide detailed results of the consequences of alternative dam operation scenarios, and in that way can be useful for assessing the impact of meeting instream flow requirements. In the case of OASIS applications in the Delaware and Susquehanna, the program tends to be more suited as a planning tool over large spatial scales because of its parameterization and calibration needs. Given existing development of OASIS in the Delaware and Susquehanna, however, it is appropriate for Pennsylvania to explore how the existing OASIS models can be integrated into a larger water management decision-support structure and whether these basin models can be improved in spatial scale or detail.

In some simpler cases, information will be known about reservoir releases through water use reporting, and this information can be incorporated into a current condition estimate of seasonal baseflows as part of a decision-support system. However, seasonal timing and magnitude of spills may be more difficult to incorporate in even these simpler cases. Reservoirs can be treated as having limited effect on flows at or downstream of a location of interest if the spills were relatively constant during a particular season, as they can be in reservoirs that are small relative to their basin size with relatively small associated withdrawals. However, except in cases where there is gage information below a reservoir or release information kept by the operator, it will be difficult to accurately determine the impact of reservoirs on downstream nodes in any statewide approach to determining current flow conditions.

2.4.4 Incorporating Land Use Impacts

Land use change has a well demonstrated impact on hydrology, as exemplified by the Pocono Creek Watershed Study, so making a provision for assessing land use impacts when estimating current or future condition is a logical step. However, land use can be difficult to regulate in the statewide water management context and assessing its impacts accurately without watershed modeling is extremely challenging. Even with regression equations to define baseline flow conditions that eliminate land use parameters, most index gages in the state will have at least minimal hydrologic impacts from land use.

To deal with this issue, we recommend further evaluation through the PA Instream Flow Technical Advisory Committee of how general land use impacts could be incorporated into a Pennsylvania Water Management Decision Support System. Furthermore, in areas in which land use issues are of major concern, current or future condition estimates should be developed using estimated impacts from both water use and land use. This should be done through available watershed modeling approaches (e.g., PRMS, SWAT, HSPF, TOPMODEL) when models exist or resources are available.

2.5 Current Spatial Water Use Data in Pennsylvania: Opportunities and Limitations

Pennsylvania DEP, in conjunction with the Delaware River Basin Commission (DRBC) and the Susquehanna River Basin Commission (SRBC), are fortunate to have a water use reporting program in place. Act 220, the Water Resources Planning Act, requires the Department of Environmental Protection (DEP) to administer a water withdrawal and use registration and reporting system. The DEP also maintains information on discharges to waterbodies in the state and monitors monthly values of discharges across the state since approximately 2000. Reporting is required for all hydropower facilities and persons who withdraw or use more than 10,000 gallons per day in a 30-day period. Reports are annual and are kept by DEP in the Water Use Data System (WUDS). The coverage in WUDS is not complete, but does include annual use quantities for reporting users across the state (with some monthly data included). Public water suppliers have been reporting monthly data or annual data since approximately 2003 under the mandatory reporting system. Non-public water supply users are reporting on a voluntary basis, so this database is not complete. Because annual reporting has been required statewide since 2003, it is possible to examine trends in water use for public water supply users who have reported their 2004 or 2005 water supply. Some public water supply records extend into the 1990s, but these records are incomplete and difficult to access (D. Jostenski, PA DEP, personal communication).

In addition, the DRBC and SRBC have review and approval authority for groundwater and surface-water withdrawals and consumptive uses. Withdrawals from surface or groundwater that exceed 100,000 gallons per day (gpd) require approvals. New or increased consumptive uses that exceed 20,000 gpd in the Susquehanna River basin also require approvals. In the Delaware River Basin, any increase in withdrawal for an approved project requires approval. The reporting associated with these withdrawals in the SRBC and DRBC dockets will be integrated into the PA WUDS over time. However, currently the DRBC and SRBC have some reporting information not in the WUDS which is being identified through use of the Water Analysis Screening Tool.

There are some significant gaps and limitations associated with the existing water use database that could be used to estimate current flow conditions. First, since not all users have registered, the reporting data are incomplete. For example, approximately half of the golf courses in the state have yet to register and report (D. Jostenski, PA DEP, personal communication). These non-reporting uses can be estimated based on known use types. Second, agricultural withdrawals are not included in the database. This information has to be indirectly derived from estimates based on the relative water use requirements per acre of cultivation for different agricultural practices. Third, smaller water users (e.g., self-supply homeowners) are exempt from reporting. Use data from these users are probably not worth collecting, but the cumulative impacts of multiple domestic users could be estimated based on the number of self-supply households in a watershed and the average consumption of each household. This type of estimate could be generated for areas experiencing or likely to experience water stress. Finally, the time scale of reported data varies from daily to annual values. This may be corrected in the 2008 reporting year when users that withdraw over 10,000 gpd will have to report monthly data. The future database may also include information on projected water use for registered locations.

The water withdrawal and discharge information in WAST is a step toward being able to simulate current flow conditions. Currently, it has the capability to examine withdrawal and discharge information at over 10,000 “pour points” statewide. This data can be edited to take into account more accurate information or new withdrawal or discharge information. This water use information can be examined relative to a single flow statistic, the 7-day 10 year low flow statistic (7Q10). As described in the previous section, unimpaired (baseline) 7Q10 is estimated for ungaged locations in WAST based on the regression equation documented in Stuckey (2006). The screening tool can examine the percentage of 7Q10 that is allocated at each pour point. This information is designed to help in the identification of areas that could be considered as Critical Water Planning Areas under Act 220 or areas that may need more detailed planning. Thus, now that WAST is operational, the PA DEP has developed a single statistic (7Q10) baseline condition and current condition (expressed as a percentage deviation from 7Q10) for all pour points in the state.

The information in the WAST provides the foundation for a more holistic examination of current flow conditions at locations across the state and is a good starting point for assessment of current flow alteration and development and implementation of instream flow standards. Limitations in how the water use data have been collected and formatted impact the quality of current (or future) hydrologic condition estimates that can be developed. Many of these limitations have been documented by the PA USGS in their draft “Methodology for the Development of the Water Analysis Screening Tool: State Water Plan Update of 2008.” Some key limitations include:

1. WAST generally applies only to watersheds between 15 and 2,000 square miles;
2. Hydrologic alterations associated with reservoir operations are not included;
3. All upstream withdrawals (and discharges) are aggregated for each pour point, so potential local impact of particularly large withdrawals (or discharges) may be missed;
4. Surface water and ground water divides are not differentiated in the model;
5. Some water use data has been verified, but some areas of the state have not; and
6. Water use data (withdrawals or discharges) is not at a monthly time step, but rather annualized based on actual days used.

At present, the WAST includes only one hydrologic statistic, 7Q10, that can be linked to ecological integrity metrics at biomonitoring sites. 7Q10 can provide information about the magnitude of baseline and current drought conditions and has been linked to ecological integrity metrics in Georgia (Freeman and Marcinek 2006). There are many other useful hydrologic indicators of low flow conditions that have been associated with habitat or ecological integrity metrics. 7Q10 also has the major limitation of having relatively high estimation error associated with it, as described in Stuckey (2006). Hydrologic statistics such as August median flow, currently being used a key hydrologic statistic in Michigan, may be more easily linked to ecological integrity measures and have less associated estimation error. In addition, as will be discussed in Section 5, an effective instream flow standard should incorporate limits on alteration of multiple aspects of flow regime variability including aspects of high flows and low flows, as well as consideration of annual and inter-annual variability.

Any PA Decision Support System application that defines current or future conditions should allow for estimation of a variety of ecologically-relevant flow statistics. This could be accomplished by enhancing the existing screening tool or by developing a new application. To determine the effects of current water use on statistics related to flow duration, timing, frequency or rate of change would require a simulated time series. Several approaches to simulating a time series are discussed in detail in the Section 2.3. If a baseline time series is available, withdrawals from the most recent water use reporting year can be subtracted (and discharges added) from this baseline series in order to create a current time series that could be used to calculate monthly magnitude or any statistics related to flow duration, timing, frequency, or rate of change. The description of the Massachusetts Sustainable Yield Estimator Application Project, included above, provides an excellent example of an approach to defining a time-varying current condition flow time series which could be used as a model for a Pennsylvania application.

An alternative approach, which could be accomplished by enhancing the existing screening tool, would use a range of baseline regression-based statistics. At present, such baseline estimates are only available for the flow statistics listed in the Stuckey (2006) report, but, regression equations could be developed to predict values of many other flow statistics (M. Stuckey, PA USGS, personal communication). To estimate current condition for these statistics, one could estimate magnitude statistics by defining cumulative withdrawals as a percentage of the statistic of interest; this is the approach that the screening tool currently uses for 7Q10. Baseline values of monthly flow magnitudes, if predicted, could be adjusted for existing uses in locations where monthly water use data are available or monthly withdrawals can be estimated from annual reporting data.

2.6 Recommendations for Developing the Hydrologic Foundation

We recommend that a statewide GIS-based computer application be developed that allows for the estimation of baseline (i.e., “minimally impacted”) and current/future time series at any point location, both gaged and ungaged, throughout the state. This application would allow for simulation of unimpacted and impacted daily streamflow over discrete time periods (e.g., 1960-2000). These flow time series could be used to calculate various flow statistics of ecological significance at a range of time steps (from daily to inter-annual). Hydrologic statistics would not be limited to magnitude, but could also include duration, frequency, timing, and rate-of-change. Calculating the differences between values of these flow statistics under baseline and current conditions will yield an assessment of hydrologic alteration.

The recommended approach for determining a daily flow time series is a statistical, regression-based approach that estimates baseline flow duration statistics to construct flow duration curves at any point location of interest. This would be coupled with a flow duration curve transform approach that links estimated flow duration curves to flow duration curves defined at index (i.e., least impacted) gages in the state. This approach, which can be called “Modified QPPQ” (Waldron and Archfield 2006), is similar to the one that MA USGS is developing under contract with the Massachusetts Department of Conservation and Recreation. A cooperative project between USGS PA, the Susquehanna River Basin Commission, and The Nature Conservancy has been developed in accordance with this recommendation and submitted for Growing Greener funding. The Pennsylvania USGS estimates that developing additional statewide regression equations for flow duration curves at ungaged locations and developing a tool to simulate baseline daily flow time series estimates throughout the state would cost approximately $235,000.

Alternative statewide approaches with significant merit include (1) using regression approaches to predict flow statistics without simulating a flow time series, and (2) developing a statewide rainfall-runoff model to produce a daily time series that could be analyzed directly using a hydrologic statistic analysis program (e.g., TOPMODEL for New Jersey: Kennen et al. 2008). In addition, for areas of the state in which water resources are particularly stressed (e.g., Southeastern PA Ground Water Protected Area) or hydrologic processes are extremely complicated (e.g., karst areas) it may be most appropriate to invest in regional hydrologic process models that can incorporate groundwater dynamics and provide a higher degree of predictive accuracy.

The decision to recommend a regression-based approach in conjunction with a flow duration curve transformation was made in part by examining the relative costs of the potential approaches. The Project Options Table (Appendix 1) illustrates the tradeoffs in costs, including estimated costs for current condition development, which generally track the amount of work and data required for parameterization.

Steps to implement this recommendation**:**

1. Agree upon a set of index gages that can be used to define baseline flow conditions.

To finalize the identification of index gages used to develop baseline flow conditions, the criteria applied by USGS and TNC should be reviewed and revised if necessary. Once agreed upon, these criteria can be applied to select a final list of Pennsylvania index gages. To expand upon the list of index gages from Pennsylvania, the agreed upon criteria should also be applied to gages in adjacent states that are within the same physiographic regions as Pennsylvania. Gages for screening may exist in Maryland, New York, New Jersey, Ohio, and/or West Virginia. This use of adjacent state gages has been applied by USGS in New England (D. Armstrong, MA USGS, personal communication).

2. Link all locations in the state to appropriate index gage(s) to develop baseline flow conditions.

Our recommendation is to use the index gages to identify baseline flow conditions for any point location within the state of Pennsylvania (including non-index gages and all ungaged locations). To accomplish this, all points within the state need to be associated with an appropriate index gage or set of gages. Pour points and associated basins used in WAST may be an appropriate spatial unit for setting baseline flow conditions.

There are at least two approaches to accomplishing linkage to an index gage or set of gages:

Option 1. Link each pour point to an index gage based on similarity in hydrologic or catchment characteristics. *In this option, one index gage is used to define baseline flow conditions for one or more pour points.*

a) **The index gage that has the most similar value of a selected hydrologic statistic, drainage area-normalized**. For example, the predicted 7Q10 for each pour point could be compared with the 7Q10 for each index gage and the gage that has the closest value would be used to define baseline flow statistics for that pour point. Although predicted flow duration curves do not currently exist statewide, these could be developed and used to model flow duration curves for all pour points. Curves could be associated with the appropriate index gage by minimizing the area between the two curves or by minimizing the difference between selected points along the two curves (e.g., difference between Q10, Q50, Q90).

b) **The index gage that is most similar in terms of catchment characteristics**. Another way to link pour points to an index gage would be to associate the two gages using one or more catchment characteristics, as catchment characteristics are known to be significant predictors of flow statistics and provide good candidate variables. Candidate variables could include drainage area, basin slope, elevation, precipitation, soil thickness and porosity, glacial and bedrock geology (See examples from New Jersey (Kennen et al. 2007) Pennsylvania (Stuckey 2006), and Michigan, Illinois, and Wisconsin (Seelbach et al. 2002, Allan et al. 2000)). Many of these variables have already been calculated for stream reaches in the National Hydrography Dataset (USGS 2006) through complementary projects and are available for use in this analysis.

With both choices, the index gage may not be geographically closest to the pour point, rather it is the most similar or 'closest' relative to some selected hydrologic or catchment characteristic. Also with both choices, an additional stratifying variable could be introduced. For example, the pour point could be linked to the nearest index gage within the same ecoregion or major basin. Or the pour point could be linked to the gage most similar in terms of some hydrologic statistic within a specified distance.

The advantage of linking to a single index gage is that a daily time flow series can be developed for each pour point that serves as a baseline for future analysis using a hydrologic statistical analysis program such as the Indicators of Hydrologic Alteration (IHA) program or the statistical programs that are part of the USGS Hydroecological Integrity Assessment Process (HIP). Such analysis would require development of a current/future condition daily time series, but given this step, the assessment of alteration is relatively straightforward (see Section 3). The disadvantage, however, is that you have to completely rely upon a single index gage for development of your time series, which subjects the pour point time series to any errors or limitations associated with the index gage time series.

Option 2: Link each pour point to a set of index gages using some kind of classification procedure. In this option, similar index gages would be grouped using some form of classification process and *groups of gages would be used to define the baseline flow conditions for a set of pour points.* There are at least two approaches that could be used to group gages:

a) **Group gages within an existing geospatial unit.** Possible classification units include physiographic provinces, ecoregions (Level III or IV), Hydrologic Landscape Regions (Wolock 2003), Ecological Drainage Units (Higgins et al. 2005). Testing which classification scheme works best could be accomplished by comparing whether the values of selected flow metrics from reference gages within a 'class' (defined through classification) are more similar to each other than they are to values from gages in another class. Chalfant and McGarrell (personal communication) used a similar approach to classify streams into groups for establishing water quality reference conditions.

b) **Group gages with similar values of selected hydrologic statistics.** This is the approach used in the first step of the Hydroecological Integrity Assessment Process (HIP). This step results in a hydrologic classification of streams for a geographic area based on long-term gage records for relatively unmodified streams and 171 ecologically relevant indices. (Henriksen et al. 2006)

With either approach, all index gages within a given class would be grouped and used to define baseline flow conditions for other pour points that fall within the class. In addition to these two approaches, other methods for grouping gages into classes, including using one or more catchment characteristics (e.g., drainage area, geology/lithology, precipitation) could be explored (see Sanborn and Bledsoe 2006).

The advantage of such an approach is that it does not rely on any individual index gage and is therefore not subject to its associated error and limitations. The primary disadvantage of this approach is that it is more difficult to create a single daily time series from multiple index gages. Furthermore, since this approach relies on classification, it may group gages that have normalized flow variables that have a wide range. This would complicate analysis of the degree of alteration associated with human uses at a site, since the baseline may incorporate a wide frequency distribution for each statistic of interest.

3. Simulate a daily flow time series using regional regression and a flow duration curve (FDC) – transformation method.

Our recommendation for setting baseline flow conditions incorporates a regression approach to simulate a daily flow time series of unregulated or “minimally impaired” flows. Index gages are identified and linked to other locations in the state in the two steps outlined above. In this third step, the daily flow series from the index gage are transferred to other locations. The flow duration curve is used as the means of transfer, since it provides a summary of flow conditions over a given time period at a site in common terms (exceedance frequencies).

Daily streamflow is first estimated by constructing a flow-duration curve (FDC) at the ungaged basin. This FDC is estimated by a set of equations that relate basin characteristics to properties of the FDC. A time series of daily streamflow is then created from the estimated FDC by use of the QPPQ-transform method (originally introduced by Fennessey 1994), which transfers the timing of the daily flows at an index gage to the ungaged site by equating the exceedance probabilities at the index gage and ungaged site. This approach is currently being applied by the MA USGS, in cooperation with the Massachusetts Department of Environmental Protection, towards the development of an interactive, GIS-based decision-support tool for water management. Daily streamflow in Massachusetts estimated from this method shows good agreement with observed daily flows and the agreement between observed and estimated streamflow is comparable to the agreement obtained from a calibrated rainfall-runoff model (Archfield and Vogel 2008).

This allows for the calculation of an extremely wide-range of ecologically-relevant flow statistics using software tools such as the USGS Hydroecological Assessment Tool (HAT) and TNC Indicators of Hydrologic Alteration program (IHA) (see Section 4, Selecting Hydrologic Statistics and Assessing Hydrologic Alteration). Simulation of a flow time series, whether it is monthly or daily, would allow for statistics of frequency, duration, timing, and, in the case of daily, rates of change to be developed. Recent scientific literature has emphasized the importance of these attributes of natural flow variability to the ecological health of river and stream systems (e.g., Poff et al. 1997).

We recommend that a Pennsylvania Water Management Decision Support System (PA DSS) be developed as the central tool for statewide water management. The PA DSS would allow for simulation of unimpacted daily streamflow over discrete time periods (e.g., 1960-2000). The recommended approach for determining a daily flow time series is a regression-based approach that estimates baseline flow duration statistics to construct flow duration curves at any point location of interest. This approach, which is called "Modified QPPQ" (Waldron and Archfield 2006) is similar to the one that MA USGS has developed under contract with Massachusetts Department of Conservation and Recreation as part of their Sustainable Yield Estimator Project.

Work as part of the Sustainable Yield Estimator Project has shown that the accuracy of results of this statistical approach to estimating baseline conditions is comparable to results of HSPF and presumably other process-based models. It is also significantly more cost-effective, and can be run on a basic desktop computer with little training beyond familiarity with Excel. On the other hand, hydrologic process models like HSPF, as demonstrated by the Virginia HSPF case study, provide considerable flexibility and power. This includes the ability to simulate changing land use conditions and climate conditions. So although we recommend an approach akin to the Sustainable Yield Estimator, clearly a functional PA DSS could be created using a watershed modeling approach if the additional funds, and staffing to effectively utilize it, were available.

In addition to simulating unimpacted daily streamflow, the PA DSS would develop daily streamflow estimates that account for both individual and cumulative current diversions, return flows, and reservoir operations upstream of a point of interest. This application should be modeled after the Sustainable Yield Estimator application designed for similar purposes in Massachusetts. In Pennsylvania, this application could initially use the WAST pour points as analysis locations, but it may be desirable to expand to a more "point and click" application. Initial input to this DSS would be from WAST's comprehensive water use database, which currently accounts for diversions (registered and estimated) as well as discharges (USGS "Methodology for the Development of Water Analysis Screening Tool" undated draft). The current water use database in the state is not complete and reflects a variety of time steps (from daily to annual). However, as work on the screening tool has shown, major water uses and discharges otherwise unaccounted for in the PA Water Use Data System (WUDS) can be estimated and used to develop cumulative assessments of water use at locations throughout the state. In addition, impacts of reservoir operations have not been incorporated into WAST and may be difficult to incorporate statewide, but could be incorporated for specific sites. The PA DSS should be constructed to allow for continuous improvement in estimates of water management impacts and to incorporate any new available data. If possible, monthly estimates of water use and discharge should be incorporated into a PA DSS to better account for seasonal variation in water use. In 2008, monthly water use reporting will begin, and these data can be incorporated into a PA DSS.

A PA DSS with the capability to simulate current conditions would also allow for the evaluation of future scenarios on a site-specific basis or statewide. Future water use or discharge proposals could be incorporated directly in the application and the resulting alteration in hydrology subsequently analyzed. This functionality would likely be critical for sustainable, integrated management of statewide water resources through state and basin commission authorities.

The approach taken to estimate baseline flow conditions will determine the spatial and temporal resolution necessary for current/future condition estimation. We recommend an initial approach that develops a daily time series of at least twenty years at each pour point in the state. Using a relatively long period of record helps to account for interannual variability in some of the flow statistics that may be calculated using estimated daily flows (TNC 2005). Estimations of the impact of current or future water use from groundwater withdrawals or reservoirs should be initially based on relatively simple approaches (e.g., assumed 1:1 groundwater:surface water impacts for wells within a certain distance of a waterbody), but should be refined based on available data and impact estimation approaches (e.g., analytical solutions for groundwater withdrawal impacts on streamflow). In areas where more detailed estimates of current condition are possible (e.g., PA Highlands), such information should be incorporated to enhance assessment of hydrologic alteration when appropriate for the development of flow-ecology relationships or instream flow decision-making. Finally, in areas in which major land use issues exist, hydrology is particularly complicated, or other site-specific factors are significant, we recommend consideration of more detailed watershed-specific hydrologic process modeling (e.g., HSPF, PRMS) or operational water management modeling (e.g., OASIS) to ensure these challenges can be met.

SECTION 3: DEVELOPING AND APPLYING AN APPROPRIATE RIVER CLASSIFICATION

3.1 Background

By classifying rivers according to ecologically-meaningful streamflow characteristics (e.g., Poff and Ward 1989, Harris et al. 2000, Henriksen et al. 2006), groups of similar rivers can be identified, such that within a river type there is a *range* of hydrologic and ecological variation that can be considered the natural variability for that type (Poff et al. in review). Classification using primarily hydrologic characteristics is based on the premise that natural flow attributes, as well as hydrologic alterations, shape their associated ecological communities in a similar way within each river type. As described in the ELOHA framework (Poff et al. in review), river classification can serve two purposes. First, it can be used to develop flow alteration-ecological response relationships with data from a relatively small set of rivers that can be applied to a broader set of rivers within the same type. Second, classification can be useful for directing future monitoring efforts to improve the quality of flow-ecology relationships or to detect management impacts.

For each river type within a classification, the following items should be developed and applied:

1. A range of hydrologic variation that represents the baseline hydrologic condition against which hydrologic alterations and corresponding ecological responses can be assessed;
2. A corresponding set of hydrologic statistics that is used to assess hydrologic alteration for all rivers assigned to the same river type;
3. A set of flow alteration–ecological response relationships that can be applied to rivers within the same type; and
4. A monitoring plan, stratified by river type, designed to improve the flow alteration–ecological response relationships for all types.

There are various approaches to river classification depending on the application. In this section, we review classifications that are potentially relevant to development and application of instream flow criteria, including (1) river classifications based on hydrologic statistics and (2) river classifications based on other habitat variables or biological attributes. We also emphasize that the classifications discussed in this section are applied *to characterize hydrological or ecological types* (e.g., the “coldwater streams” type within the Michigan stream classification, or the “Atlantic basin coldwater [fish] community” within the Pennsylvania Aquatic Community Classification), rather than to define *classes of protection* (e.g., special value streams, typical streams, working rivers) that have corresponding protection goals. Assigning classes of protection may be an important aspect of a water management program; examples of protection classes are discussed in Section 6.

Goals for an appropriate river classification**:**

1. A river classification should result in a relatively small number of river types that capture the major dimensions of streamflow variability within a region. Classes can be further subdivided, if need, to reflect additional ecological conditions.
2. The classification should be applicable to all rivers within a region.
3. Ideally, if a classification based on hydrologic metrics is selected, these metrics should be amenable to management, so that environmental flow standards can be established using these same hydrologic metrics.
4. Flow metrics used to develop a hydrologic classification should collectively describe the full range of natural hydrologic variability, including the magnitude, frequency, duration, timing, and rate of change of flow events (Poff et al. 1997, Richter et al. 1996, Olden and Poff 2003, Kennen et al. 2007, Mathews and Richter 2007).
5. Flow metrics must be "ecologically relevant," i.e., they are known to have—or can reliably be extrapolated from ecological principles to have—some demonstrated or measurable ecological influence (Arthington et al. 2006, Monk et al. 2007) and hence will be important in assessing ecological responses to hydrologic alteration.

3.2 Potential Classification Approaches Considered

3.2.1 River Classifications Based on Hydrologic Statistics

Poff (1996) developed a hydrologic classification of river types in the United States based on analysis of hydrologic statistics. Examples of river types in this classification include stable groundwater-fed rivers; seasonally predictable snowmelt rivers; intermittent, rain-fed prairie and desert rivers; and highly dynamic, unpredictable rain-fed perennial rivers. The 10-12 river types were subsequently reduced to six general types by Olden and Poff (2003). Olden and Poff (2003) also presented a reduced set of hydrological indices that could explain the dominant proportion of statistical variation and minimize redundancy while representing the critical aspects of the flow regime. These indices are the foundation of the Hydroecological Integrity Assessment Process (HIP), which includes a hydrological classification approach that has been applied in several states, including New Jersey and Missouri. Kennen et al. (2007) defined four river types in New Jersey; a similar analysis resulted in three primary and five secondary stream types in Missouri (Jim Henriksen, USGS, personal communication).

The **Hydroecological Integrity Assessment Process (HIP)** was developed by the USGS Fort Collins Science Center based on a large body of research linking hydrological variability and aquatic ecosystem integrity. HIP *is a process* that includes a range of software tools for conducting hydrologic classification of streams, addressing instream flow needs, and assessing past and proposed hydrologic alterations on streamflow. The HIP approach addresses several steps in the ELOHA framework, including river classification and calculating hydrologic alteration using a set of stream class-specific hydrologic statistics. We also discuss HIP in *Section 4: Selecting Hydrologic Statistics and Assessing Hydrologic Alteration.*

As described in the HIP Manual (Henriksen et al. 2006), "The HIP is intended for use by any Federal or State agency, institution, private firm, or nongovernmental entity that has a responsibility or interest in the management and (or) regulation of streams with an objective to address ecological integrity at the reach or watershed scale. In addition, HIP can assist researchers by identifying critical, stream class specific, hydrologic indices (HIs) that adequately characterize the five major components of the flow regime (magnitude, frequency, duration, timing, and rate of change) by using 10 nonredundant indices (see Olden and Poff 2003)."

Applying the HIP involves four **steps**:

1. Perform a hydrologic classification of streams in a geographic area using long-term gage records for relatively unmodified streams and 171 ecologically relevant indices.
2. Identify statistically significant, nonredundant, hydroecologically relevant indices associated with the five major components of the flow regime for each stream class.
3. Develop an area-specific Stream Classification Tool (SCT) for placing streams not used in the classification analysis into one of the identified stream classes.
4. Develop an area-specific Hydrologic Assessment Tool (HAT).

Four computer software tools have been developed.

1. The Hydrologic Index Tool (HIT) calculates 171 biologically relevant hydrologic indices using daily and peak flow records. The indices are then used for a regional (state) stream classification analysis. The program is designed to import U.S. Geological Survey mean daily and peak flow discharges from the National Water Information System databases.
2. The National Hydrologic Assessment Tool (NatHAT) is based on a hydrologic classification of streams which used 420 gaging stations across the contiguous United States. The National Hydrologic Assessment Tool has six stream classes identified. The program is used to establish a hydrologic baseline (reference time period), to establish environmental flow standards, and to evaluate past and proposed hydrologic modification. This is accomplished by using flow statistics, trend analysis, and 10 primary stream class-specific indices that address the five major components of flow.
3. A state-specific Stream Classification Tool classifies any stream within the state. For example, four classes of streams have been identified in New Jersey.
4. A state-specific Hydrologic Assessment Tool (HAT) is used to establish a hydrologic baseline (reference time period), establish environmental flow standards, and evaluate past and proposed hydrologic modifications. This is accomplished by using flow statistics, trend analysis, and 10 primary stream class-specific indices that address the five major components of flow. Two states have a Hydrologic Assessment Tool – New Jersey and Missouri. Development of HIP and the associated software is scheduled for Texas in 2007.

We contracted with USGS to complete a pilot classification of Pennsylvania rivers using HIP. Results of this study are described in *Case Study 2: Stream Classification in Pennsylvania using Hydroecological Integrity Assessment Process (HIP).* Appendix 2 is the final report that USGS-Fort Collins prepared for The Nature Conservancy and provides additional details on the pilot classification.

Ecohydrological regionalization of Australia: Researchers in Australia recently completed and are in the process of publishing a continental-scale classification of Australia's riverine flow regimes using two approaches (Pusey et al. in review). The first approach involved analysis of empirically-derived stream flow data from 830 gages throughout Australia. One-hundred twenty hydrologic metrics related to flow magnitude, frequency, duration, timing and rate of change were calculated for each gage using 15 years of continuous discharge data. A Bayesian clustering technique was used to group gages according to similarity in flow regime. This probabilistic approach recognizes and accounts for the uncertainty attributable to flow data, flow class definition, and flow class assigned; this approach is notably different from other deterministic approaches (i.e., that assign a stream or gage to one class without accounting for the uncertainty associated with defining classes, assigning classes to a particular stream or gage, and other factors). The analysis resulted in the most likely classification having 12 flow regime types. The classes differed in the degree of flow permanence, seasonal discharge patterns, variations in flood magnitude and frequency, and other aspects of flow predictability and variability.

The second approach grouped Australian rivers and streams according to their similarities in climatic and landscape factors. This analysis produced a 30-group classification based on 48 attributes describing catchment climate, water balance, topography, substrate, and vegetation cover for 1.2 million stream reaches. Classes are primarily distinguished by attributes related to catchment climate and water balance and secondarily by catchment morphology, substrate, and vegetation. Subsequent analysis used combinations of these environmental variables to explain and predict flow regime class membership. These analyses revealed that geographic, climatic, and some topographic factors were generally strong discriminators of flow regime classes, supporting the view that spatial variation in hydrology is determined by interactions between climate, geology, topography, and vegetation at multiple spatial and temporal scales.

3.2.2 River Classifications Based on Other Landscape (Catchment) Variables that Influence Hydrology

Using landscape variables, often derived from a GIS, to describe and classify rivers is an alternative to classification based on analysis of hydrologic statistics. Many examples of these classification approaches exist, and they often include variables used to describe physical and biological characteristics of rivers at a variety of spatial scales (Higgins et al. 2005, Sowa et al. 2007). Below we briefly describe several classification approaches that either could serve as a model for Pennsylvania or have been completed for streams or watersheds within Pennsylvania and could be adapted for or incorporated into a hydrologic classification.

Hydrologic landscape regions: Wolock (2003) grouped watersheds in the United States into hydrologic landscape regions (HLRs) according to their similarity in landscape and climate characteristics. The selected landscape and climate characteristics represent factors assumed to affect hydrologic processes. Hydrologic landscape regions in the United States were delineated by using GIS tools and statistical methods including principal components and cluster analyses. The GIS and statistical analyses were applied to landsurface form, geologic texture (permeability of the soil and bedrock), and climate variables that describe the physical and climatic setting of 43,931 small (roughly 200 square kilometer) watersheds in the United States. The analyses then grouped the watersheds into 20 noncontiguous HLRs on the basis of similarities in land-surface form, geologic texture, and climate characteristics. Figure 3.1 shows the HLRs within Pennsylvania.

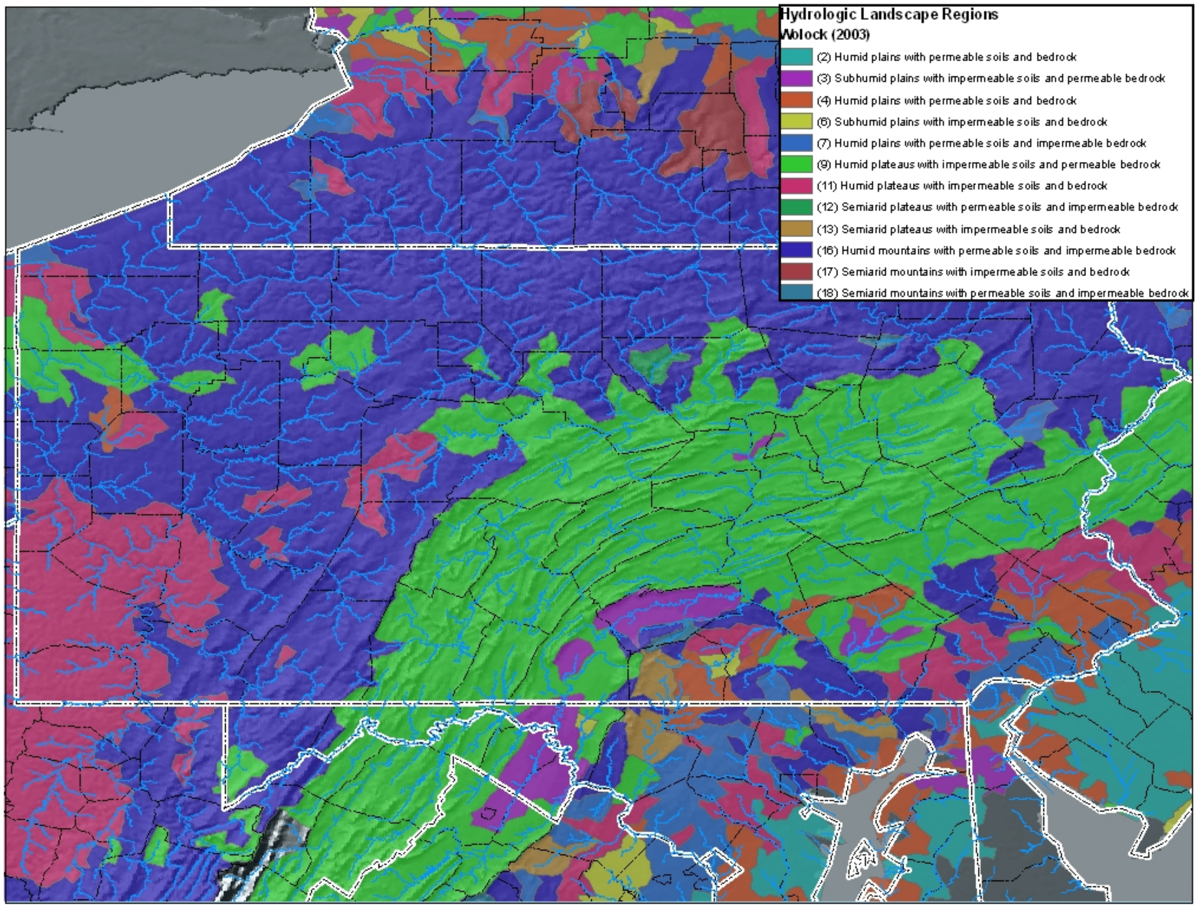

Figure 3.1 Hydrologic landscape regions (Wolock 2003) within Pennsylvania.

Northeastern Aquatic Habitat Classification System (NEAHCS): With the creation and implementation of State Wildlife Action Plans (SWAP) by state fisheries and wildlife agencies, the need for consistent, current digital habitat maps has grown dramatically. In response to this need, Northeast Regional Conservation Needs (RCN) Grant Program of the Northeast Association of State Fish and Wildlife Agencies (NEAFWA) provided financial support to develop consistent terrestrial and aquatic habitat GIS datasets (see http://rcngrants.org/habitat_classification.shtml). Each of the 13 Northeastern states (ME, NH, VT, MA, RI, CT, NY, PA, NJ, DE, MD, VA, and WV) and the District of Columbia are participating and contributing in kind. This project will compile and standardize terrestrial and aquatic habitat classification systems and provide map products that will form the foundation of state and regional conservation in the Northeastern United States.

Unlike terrestrial communities, no national standard aquatic habitat or community classification for the U.S. currently exists. The comprehensive biological sample data necessary to develop a classification and map of aquatic ecosystems from biological data alone is also lacking across large regions, including the 13 Northeast states. The Northeast Aquatic Habitat Classification System will therefore use the aquatic biophysical classification approach developed by The Nature Conservancy (Higgins et al. 2005) and recommended by the National Fish Habitat

Science and Data Committee (2006). This classification approach can be implemented at regional scales and emphasizes the environmental gradients of climate, elevation, landform, and geology which are known to shape aquatic ecosystems at several spatial scales and influence the physical aquatic habitat diversity (Higgins et al. 2005). At each spatial scale of the classification, the variables selected for classification are those physical entities that are most general, invariant, and causal for the given scale (Frissell et al. 1986). The classification product will provide an estimate of the expected natural aquatic habitat type, but is not intended to account for variation in the occurrence of aquatic habitats due to human alteration.

The NEAHCS will nest finer levels of classification within a hierarchy. The focus of the NEAFWA aquatic classification workgroup is on development of macrohabitats, which will be nested within a hierarchy of previously defined units or units under development, including Freshwater Ecoregions, Ecological Drainage Units, and Aquatic Ecological Systems (Higgins et al. 2005). Macrohabitats are the finest mapped unit of NEAHCS classification and define individual stream reach or lake types. Macrohabitats are based on abiotic variables known to structure aquatic communities at this reach or lake scale that can be modeled in a GIS. These variables include potential factors such as stream or lake size, gradient, water acidity, flashiness/stability of flow, water temperature, and local connectivity. Macrohabitats are designed to be relatively homogeneous with respect to energy and nutrient dynamics, habitat structure, and position within the drainage network (Higgins et al. 2005). The macrohabitats will use the NHD-Plus medium resolution hydrography for modeling and mapping.

To assist with macrohabitat development, the Northeastern Aquatic Habitat Classification Workgroup has been formed with more than 30 state, federal, university, and NGO representatives from the Northeast states. This workgroup has been meeting monthly since September 2007 to review potential macrohabitat classification variables and agree on methods for modeling each variable in GIS, developing ecologically relevant thresholds for each variable, and combining the variables into unique macrohabitat types.

The workgroup agreed upon four primary stream macrohabitat classification variables: Size, Temperature, Gradient, and Geologic Buffering Capacity (Table 3.1). When these four variables are combined, they yield 399 unique combinations in the region. These combinations can be reduced to a smaller number for reporting purposes. The workgroup is currently considering several methods to simplify this classification and reduce the number of types. Figure 3.2 shows a simplified version of the macrohabitat classification (including 124 types) for Pennsylvania.

Table 3.1: Primary Classification Variables and Thresholds for Stream Segments in Northeastern Aquatic Habitat Classification System

	Size	Drainage area (mi^2)
Headwaters	1a	0-3.861
Creeks	1b	>= 3.861 and <38.61
Small Rivers	2	>=38.61 and <200
Medium Tributary Rivers	3a	>=200 and <1000
Medium Mainstem Rivers	3b	>=1000 and <3861
Large Rivers	4	>=3861 and <9653
Great Rivers	5	>=9653

Temperature (expected natural)		
Cold	33	Complex rules based on CART analysis
Transitional Cool	31	
Transitional Warm	13	
Warm	11	
Gradient		
Very Low	1	<0.02 %
Low	2	>=0.02 and <0.1
Moderate-Low	3	>=0.1 and <0.5
Moderate-High	4	>=0.5 and <2
High	5	>=2 and <5
Very High	6	>5
Norton Geology Class		
Low Buffered; Acidic	1	100-200
Moderately Buffered; Neutral	2	>=200 and <300
Highly Buffered; Calc-Neutral	3	>= 300

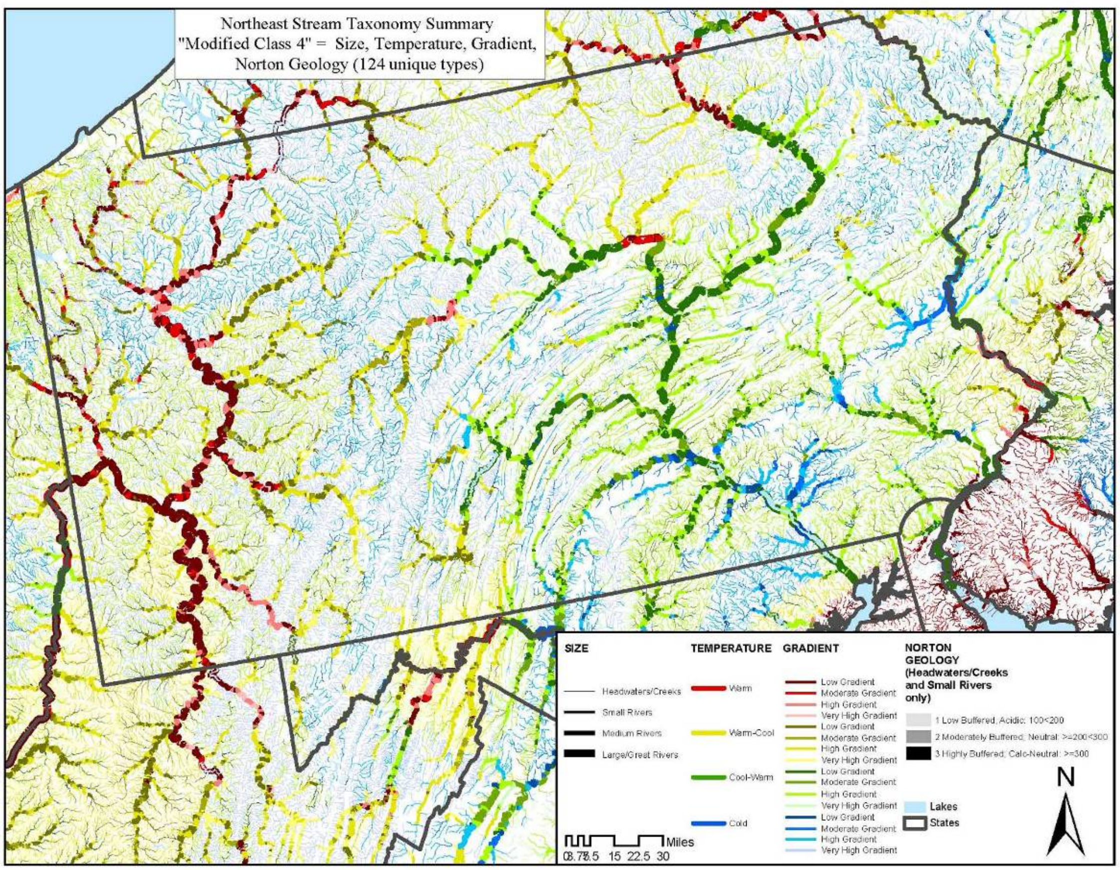

Figure 3.2 Northeastern Aquatic Habitat Classification System for Pennsylvania.

In addition to the variables used in the classification, this project will add a number of additional local and cumulative attributes to each reach in the NHD-Plus. These attributes provide additional habitat descriptors for the stream reaches but are not used in the regional classification. Some of these attributes are currently assigned to stream reaches in the NHD-Plus and other attributes were calculated and added by The Nature Conservancy (Appendix 4).

River Environment Classification (New Zealand): The River Environment Classification (REC) (Snelder and Biggs 2002) is an *a priori* classification based on a hierarchical model of "controlling factors," which are assumed to be the dominant causes of variation in physical and biological characteristics of rivers at a variety of spatial scales. The first and second levels of the REC, the "Climate" and "Source-of-Flow" levels respectively, are based on climate and topography and are expected to discriminate rivers according to differences in their flow regimes. Classes are assigned to individual "sections" of the river network based on categorical description of the climate and topography of each section's unique watershed. Snelder et al. (2005) tested the REC's ability to explain variation in hydrological character of rivers using continuous flow measurements from 335 sites throughout New Zealand. These sites were assigned to REC classes and 13 flow variables were calculated using continuous flow data. Principal components analysis was used to show that the REC classes have distinctive flow regime characteristics.

Michigan Stream Classification: In the state of Michigan, the Groundwater Conservation Advisory Council developed and adopted a stream classification that groups all Michigan streams into 11 stream types based on catchment area and July mean temperature (Michigan Groundwater Conservation Advisory Council 2007). For each segment, catchment area was calculated using GIS and July mean water temperature was predicted using a regression model and geostatistical kriging of water temperature data from 830 stream sites across Michigan (Zorn et al. in prep). Examples of stream types are cold streams, warm streams, and warm small rivers. Stream segments were assigned to one of four temperature classes (Table 3.2) and one of three watershed size classes (Table 3.3). The stream temperature classes used were proposed by members of the Great Lakes Regional Aquatic GAP science team.

Table 3.2 Temperature classes used to classify Michigan streams

Temperature Class	Description
Cold	July mean water temperature ≤ 63.5° F (17.5° C). The fish community is nearly all coldwater fishes; small changes in temperature do not affect species composition.
Cold-transitional	July mean water temperature > 63.5° F (17.5° C) and ≤ 67° F (19.5° C). The fish community is mostly coldwater fishes, but some warmwater fishes are present; small changes in temperature cause significant changes in species composition.
Cool (warm-transitional)	July mean water temperature > 67° F (19.5° C) and ≤ 70° F (21.0° C). The fish community is mostly warmwater fishes, but some coldwater fishes are present; small changes in temperature cause significant changes in species composition.
Warm	July mean water temperature > 70° F (21.0° C). The fish community is nearly all warmwater fishes; small changes in temperature do not affect species composition.

Table 3.3 Watershed size classes used to classify Michigan streams

Watershed size class	Description
Stream	Segment catchment area ≤ 80 mi^2 (207 km^2).
Small river	Segment catchment area > 80 mi^2 (207 km^2) and ≤ 300 m^2 (777 km^2).
Large river	Segment catchment area >300 mi^2 (777 km^2).

Approximately 9000 stream valley segments were assigned to one of the 11 temperature-size classes (no "cold large river" types exist). Each stream type has associated with it a characteristic relationship between fish abundance and streamflow. These relationships are presented as a set of response curves, referred to as the "Flow-fish functional response curves," which show how fish population abundances change as flow is incrementally reduced (Michigan Groundwater Conservation Advisory Council 2007). The fish models used to develop these response curves are described in detail in Zorn et al. (in prep) and discussed further in Section 5.

3.2.3 Other Classifications of Pennsylvania Rivers and Streams

Pennsylvania Aquatic Community Classification: Pennsylvania is fortunate to have the Pennsylvania Aquatic Community Classification (Walsh et al. 2007a, Walsh et al. 2007b) for flowing waters that includes both biological and physical stream habitat classifications. Seven biological classifications were developed for mussels, macroinvertebrates and fish in the major river basins in Pennsylvania using existing data acquired from state and regional monitoring projects (Table 3.4). Data from 27 data sets with compatible collection methods were used to develop the classifications.

Table 3.4 Biological Communities in the Pennsylvania Aquatic Community Classification

Community Type	Community Name
Mussels (13 communities)	
Delaware Basin	Eastern Elliptio
	Alewife Floater
	Other
Ohio – Great Lakes Basin	Pink Heelsplitter
	Fluted Shell
	Fatmucket
	Spike
Susquehanna-Potomac Basin	Lanceolate Elliptio
	Squawfoot
	Yellow Lampmussel
	Elktoe
	Eastern Elliptio
	Eastern Floater
Macroinvertebrates (12 genus-level communities; 8 family-level communities)	
Genus-level	High Quality Small Stream
	High Quality Headwater Stream
	High Quality Large Stream
	Sluggish Headwater Stream
	Common Large Stream

Family-level	Limestone / Agricultural Stream Small Urban Stream Large Stream Generalist Forested Headwater Stream Common Small Stream Ohio River Mixed Land Use Stream Low Gradient Valley Stream High Quality Mid-Sized Stream Common Headwater Stream Limestone / Agricultural Stream High Quality Small Stream Common Large Stream High Quality headwater Stream AMD Stream
Fish (11 communities)	
Atlantic Basin	Warmwater Community 1 Warmwater Community 2 Coolwater Community 1 Coolwater Community 2 Coldwater Community River and Impoundment Lower Delaware River Community
Ohio-Great Lakes Basin	Warmwater Community Coldwater Community Coolwater Community Large River Community

Known locations of all community types are mapped to stream reaches and available in a GIS. Regression models were also built to predict the distribution of these communities in unsampled stream reaches. Predicted distributions are also mapped to stream reaches and available in a GIS. Finally, all values for predictor variables used in the regression equations are available for all stream reaches.

Physical stream types were classified based on geology, stream gradient and watershed size (Table 3.5). Types were created by concatenating the codes for each variable class. These three physical habitat variables, especially watershed size and geology, could be correlated with and possibly used as predictors of streamflow characteristics.

Table 3.5 Variables and classes used to determine physical stream type (modified from Table 8-2 in Walsh et al. 2007b).

Physical Variables and Classes		Description (with emphasis on flow-related characteristics)
Geology		
1	Sandstone	Most common type in study area; comprised of sand-sized particles; moderate/variable stream flashiness

2	Shale	A fine-grained sedimentary rock, the second-most common geology type in study area; generally flashy streams
3	Calcareous	Limestone and dolomite rock types; flow is more stable in these streams because of porosity and fracturing
4	Crystalline Silicic	Igneous or metamorphic rock containing silica ions
5	Crystalline Mafic	Igneous or metamorphic rock containing calcium, sodium, iron, and magnesium ions
6	Unconsolidated materials	Sands and gravels (mainly along coastal zones and larger rivers); geological characteristics derived from surrounding rock types in the area
Stream Gradient		
1	Low Gradient	Stream slope is 0.0-0.5%
2	Medium Gradient	Stream slope is 0.51-2.0%
3	High Gradient	Stream slope is >2.0%
Watershed Size		
1	Headwater stream	0-2 mi^2
2	Small stream	2-10 mi^2
3	Mid-reach stream	10-100 mi^2
4	Large streams and rivers	> 100mi^2

Stream classes used in Instream Flow Studies: Pennsylvania and Maryland: A classification of reproducing trout streams with drainage areas less than 100 square miles was completed as part of the 1998 Instream Flow Studies Report developed for Pennsylvania and Maryland. This classification was generally based on physiographic provinces and sections, with some study regions defined within physiographic provinces based on geology or glacial history. A sub-classification was completed for study streams based on stream length, which was used as a surrogate for stream slope. This classification approach, designed to group streams with similar key physical features related to fish habitat, was seen as effective as trout habitat impacts from withdrawals were found to be similar within stream regions.

Stream classes used by DEP for macroinvertebrate sampling: Pennsylvania DEP uses a stream classification for biological monitoring associated with their Integrated Water Quality Monitoring and Assessment Reporting requirements under the Clean Water Act Sections 305(b)/303(d). For wadeable streams, DEP currently recognizes three stream classes: (1) true limestone spring streams; (2) freestone pool-glide (or low-gradient) type streams; and (3) freestone riffle-run (or higher-gradient) type streams. Each stream class has (an) associated sampling protocol(s) and each protocol uses a specific set of metrics to yield a final macroinvertebrate index of biotic integrity (IBI) score (Brian Chalfant, PA DEP, personal communication).

3.3 Recommendations for Developing and Applying an Appropriate River Classification

The goal for developing and applying a river classification that is applicable in a water management context is to group rivers and streams that have similar hydrologic behavior and apply the classification to all stream segments in Pennsylvania. There are two main approaches to achieving this goal. River classifications may be developed (1) based on hydrologic statistics (e.g., HIP); or (2) based on other landscape variables that influence hydrology (e.g., HLRs, REC, Michigan classification). The ecohydrological regionalization of Australia uses a combination of these two approaches and also uses a unique statistical approach to group streams into hydrologic classes.

As an example of the first approach, we completed a pilot application of a statewide hydrologic classification using the Hydroecological Integrity Assessment Process (HIP) and developed a regression equation that could be used to assign ungaged segments to stream classes. The PACC physical habitat classification and the NEAHCS currently under development include many attributes that could be used as a starting point to develop a classification using the second approach.

We believe there is merit in further pursuing a hydrologic classification that can be extrapolated to river reaches throughout Pennsylvania and that considerable work toward this goal has already been completed. We recommend that a scientific committee that includes members of the Instream Flow Advisory Committee review the work to date and finalize a classification. As part of this process, we specifically recommend that they:

1. Review the statistical approaches in the pilot application of HIP and, if necessary, suggest any alternative clustering methods.

2. Review the Bayesian statistical methods used in the ecohydrological regionalization of Australia and decide whether or not these methods should be applied in Pennsylvania.

3. Consider using a similar approach to the River Environment Classification to develop a flow classification using a subset of variables in the PACC and NEAHCS and additional variables that may not be included (e.g., climatic variables).

4. Evaluate correlations between selected physical habitat variables in the PACC and/or the NEAHCS and hydrologic statistics calculated for the index gages.

5. If a classification based on hydrologic statistics (e.g., HIP) is chosen, agree on an approach to assign stream classes to ungaged stream segments. This spatial extrapolation is a critical process to transform any gage-based classification to stream reaches across the state.

6. Even though hydrology may be the primary factor in determining river types, factors other than hydrology can and should be used to further subdivide stream types. New Zealand provides an excellent example of a multilevel classification, and the ELOHA framework also emphasizes this point.

The result of this classification effort would provide a strong basis for the development of flow-ecology relationships. As recommended in the ELOHA framework (Poff et al. in review), the final river classification should include a relatively small number of types to be practical for management while capturing major differences in hydrologic variability. The five types documented in this report represent a good example of a reasonable number of river classes.

Based on our survey of efforts to develop hydrologic classifications for other states and countries, input from the Instream Flow Advisory Committee, and our understanding of the existing data available, we estimate the cost to develop a statewide classification would be between $25,000 and $100,000. The cost range associated with each classification approach is listed in the Project Options Table (Appendix 1).

SECTION 4: SELECTING HYDROLOGIC STATISTICS AND ASSESSING HYDROLOGIC ALTERATION

4.1 Background

The general premise of the ELOHA framework, and of instream flow management in general, is that increasing degrees of hydrologic alteration will generally lead to increasing levels of ecological change. Describing the degree of hydrologic alteration at gaged and ungaged locations requires use of individual flow statistics or variables. This section will review two "off the shelf" programs to calculate hydrologic alteration: the Indicators of Hydrologic Alteration (IHA) and the Hydrologic Assessment Tool (HAT), which is part of the Hydrecological Integrity Assessment Process (HIP). It will also briefly discuss other approaches to assessing hydrologic alteration.

The choice of flow statistics used to assess alteration will have substantive consequences for water management. As described in Section 3, a wide range of flow statistics was used in the pilot Pennsylvania hydrologic classification using HIP. As part of that pilot application, a set of statistically significant, non-redundant variables associated with the five major flow components was identified for each stream class. The approach used to select these statistics is described below and the resulting statistics for Pennsylvania are listed in *Case Study 2: Stream Classification in Pennsylvania using Hydroecological Integrity Assessment Process (HIP)*.

Identifying a set of key flow statistics is also important for the development of flow-ecology relationships and for the implementation of flow criteria. A range of flow alteration metrics should be used in the process of defining flow-ecology relationships. Some potential metrics have been defined in the literature in the region (e.g., Kennen and Ayers 2002, Kennen et al. 2007) and others are being used in practice (e.g., Denslinger et al. 1998). Given the wide range of hydrologic statistics that could be used for assessing alteration and developing environmental criteria, it is important to keep some practical **goals** in mind. A hydrologic alteration assessment approach should:

1. Be relatively easy to use by water managers;
2. Include statistics that represent intra- and inter-annual variability in flow regimes;
3. Facilitate use of modeled hydrologic data, including data on future use scenarios; and
4. Be tested to ensure accuracy of calculations.

In addition, hydrologic statistics used to assess alteration, to develop flow-ecology relationships, and to implement instream flow criteria should:

1. Be sensitive, and have explainable behavior related to flow changes that are likely to occur due to human uses;
2. Have limited redundancy;
3. Represent natural variability in hydrologic regime, including magnitude, timing, duration, frequency, and rate-of-change if possible;
4. Have conceptual and empirical linkages to ecological response;

5. Be repeatable, and therefore not involve subjective user settings; and
6. Facilitate communication among policy makers, hydrologists, ecologists, and water users.

4.2 Potential Approaches for Selecting Hydrologic Statistics and Assessing Hydrologic Alteration

There are two readily available hydrologic assessment tools: the Indicators of Hydrologic Alteration (IHA) and the Hydrologic Assessment Tool (HAT). Both of these desktop tools have been used in instream flow applications around the country, and IHA has been used extensively around the world. They have been evaluated for purposes of water management and instream flow management in Texas by Hersh and Maidment (2006). Another approach that has been developed more recently examines changes in flow duration curves (FDCs) and associated metrics. This more graphical approach is also reviewed below.

4.2.1 Indicators of Hydrologic Alteration

As described in Mathews and Richter (2007), "The Nature Conservancy developed a software program, called the Indicators of Hydrologic Alteration (IHA), to support hydrologic evaluations (Richter et al. 1996, Richter et al. 1997). This program, available at no cost from The Nature Conservancy, has been used by scientists and water managers in river basins throughout the US and worldwide (TNC 2004). The Nature Conservancy recently made substantive enhancements to the IHA to include new capabilities to support environmental flow assessments. The IHA was originally developed to enable rapid processing of daily hydrologic records to characterize natural flow conditions and facilitate evaluations of human-induced changes to flow regimes. The program was designed to calculate the values of 33 hydrologic parameters that characterize the intra- and inter-annual variability in water conditions, including the magnitude, frequency, duration, timing and rate of change of flows or water levels (Richter et al. 1996). Values are computed for each of the 33 parameters for each year of record, enabling users to assess the inter-annual variability and changes in each hydrologic parameter for selected time periods or for the entire period of record. Two primary criteria were used in selecting the original suite of 33 hydrologic parameters: their ecological relevance, and particularly their use in published ecological studies; and their ability to reflect human-induced changes in flow regimes across a broad range of influences including dam operations, water diversions, ground water pumping, and landscape (catchment) modification.

Values are computed for each of the 33 parameters for each year of record, enabling users to assess the inter-annual variability and changes in each hydrologic parameter for selected time periods or for the entire period of record. Users can conduct an impact analysis using data from before and after an impact such as dam construction, or perform a trend assessment of more gradual changes in hydrologic conditions, such as those attributable to conversion of a landscape from forest to agricultural use. While less dramatic than the alterations evident in the hydrograph after construction of a dam, trend assessments can identify parameters that have changed over time. An IHA analysis can also be applied to output from hydrologic models that are used in testing future water management scenarios, or in comparing model-simulated naturalized flows with current conditions. The IHA software facilitates analysis of variability and change in hydrologic parameter values over time by producing tabular summaries and graphical output."

In recent years, river scientists in countries around the world have been forging new methodologies for establishing environmental flows. Holistic approaches such as Benchmarking and the Holistic Method in Australia (Arthington et al. 1992, Brizga et al. 2002), and the Building Block Methodology and DRIFT in South Africa (King and Louw 1998) address the complex nature of riverine ecosystems by evaluating the flows necessary to maintain the geomorphic, hydraulic, biotic, and water chemistry aspects of a healthy river and floodplain ecosystem. These holistic methodologies each rely upon statistical characterizations of key aspects of flow regimes. The hydrological characterizations differ to some degree, but five major components of flow have repeatedly been considered as being ecologically important: extreme low flows, low flows, high flow pulses, small floods, and large floods. These five components of flow regimes have been incorporated into the new version of the IHA software as "environmental flow components" (Table 4.1).

Table 4.1 List of environmental flow components (EFCs) that can be used in developing environmental flow recommendations, as calculated by the IHA software (TNC 2005).

Environmental Flow Component	Definition	IHA Statistics
Extreme Low Flows	10^{th} percentile of all low flows	Mean or median values for: • Magnitude • Frequency • Duration • Timing *(subtotal 4 parameters)*
Low Flow	Low flow (base flow) in each month	Mean or median values for: • Monthly low flows *(subtotal 12 parameters)*
High Flow Pulses	Flows greater than low flows but less than bankfull	Mean or median values for: • Magnitude • Frequency • Duration • Timing • Rate of rise and fall *(subtotal 6 parameters)*
Small Floods	Flows equal to or greater than bankfull flows but less than the 10-year flood	Mean or median values for: • Magnitude • Frequency • Duration • Timing • Rate of rise and fall *(subtotal 6 parameters)*

Large Floods	Flows equal to or greater than the 10-year flood	Mean or median values for: • Magnitude • Frequency • Duration • Timing • Rate of rise and fall *(subtotal 6 parameters)*

4.2.2 Hydroecological Integrity Assessment Process

As described in Section 3, the Hydroecological Integrity Assessment Process (HIP) involves four steps: (1) a hydrologic classification of relatively unmodified streams in a geographic area using long-term gage records and 171 ecologically relevant indices; (2) the identification of statistically significant, nonredundant, hydroecologically relevant indices associated with the five major flow components for each stream class; (3) the development of a stream classification tool; and (4) the development of an area-specific hydrologic assessment tool. *Case Study 2: Stream Classification in Pennsylvania using Hydroecological Integrity Assessment Process (HIP)* describes Steps 1 and 2 of HIP applied in Pennsylvania. Steps 3 and 4 could be pursued in Pennsylvania based on a holistic examination of needs and opportunities.

For assessing alteration, two of the software tools that are part of the HIP process are relevant. The Hydrologic Index Tool (HIT) calculates a wide range (171 currently) of hydrologic statistics and this list may be expanded to include the Environmental Flow Components (EFCs). This range of statistics, derived from the work of Olden and Poff (2003), provides an extensive basis to assess hydrologic conditions. The other software tool (NatHAT), is an analysis tool that provides the ability to assess alteration in statistics of interest. The state-specific versions of HAT are much the same as the NatHAT, but they facilitate analysis of a subset of the 171 statistics that have been derived from statewide stream classification. NatHAT/HAT has similar analysis capabilities as IHA for comparing baseline hydrologic conditions to current or future conditions. HAT currently better facilitates use of modeled data than IHA, and both can be used to define degree of hydrologic alteration in a number of ways.

The pilot work in Pennsylvania has been an opportunity to test the use of TNC's Environmental Flow Component statistics as an additional set of ecologically-relevant hydrologic statistics beyond those found in the Hydrologic Index Tool (HIT). Although adding in these 34 Environmental Flow Component statistics did not significantly change the outcome of the pilot classification in Pennsylvania, their inclusion may provide a model for future analysis to combine some of the strengths of the two approaches (IHA and HIP).

4.2.3 Texas Review of Hydrologic Alteration Software

Under contract with the Texas Water Development Board, Hersh and Maidment (2006) completed a review of both IHA and HAT for purposes of decision support for the Texas Instream Flow Program and for evaluating flow regime impacts in priority Texas sub-basins. Texas has a four-level flow characterization (subsistence flow, base flow, high flow pulses, and overbank flows) for purposes of instream flow management that is similar to the Environmental Flow Components of the IHA described above. They also sought to identify the inter-annual variability of flow component statistics through examining them in wet, dry and normal years. The study concluded that the IHA and the HAT programs characterize hydrographs similarly well and noted that there may be greater flexibility associated with characterizing hydrographs using an independent software program such as Excel or SAS.

Given the goals of the Texas program listed above, Hersh and Maidment concluded that the HAT program was a better choice for application in Texas due to its use of non-dimensional indices (often normalized by median streamflow), its ability to calculate the 7Q2 (a regulatory threshold for water quality standards in Texas), and for its capability to facilitate "regionalization." This regionalization step is developed through the state specific classification tool and is based on multivariate statistical analysis of all the statistics that derive from the HIT software tool. (This classification/regionalization step could be done with IHA statistics as well as HIT results, but to date such an effort has not been documented.) The study concluded that a Texas-customized version of HAT, as part of a Hydroecological Integrity Assessment Process, would be "suitable and preferable to IHA for application in the Texas Instream Flow Program." Hersh and Maidment (2006) also concluded that "regardless of the tool selected, the consideration of the variability of the natural flow regime and the development of instream flow prescriptions accordingly represents a leap forward in the science and policy of instream flow and in the ability to protect and/or restore a 'sound ecological environment'."

4.2.4 Flow Duration Curve Approach

A Flow Duration Curve (FDC) approach is another relevant approach to assessing hydrologic alteration. This approach is best represented by Vogel et al. (2007), referred to as the "Ecodeficit" (Figure 4.1), and Acreman (2005). The FDC also has been applied by water managers in the United States including the Suwanee River Water Management District (Florida). FDC approaches are good graphical approaches to assessing alteration of magnitude, and they can be used for seasonal analysis to help determine seasonal frequency of particular flow conditions. They are, however, inadequate for defining the timing, duration, and frequency of most events (especially infrequent events) and for evaluating rate-of-change alteration. Inter-annual variability is often lost as well, because flow duration curves are often created by summarizing multiple years of flow data.

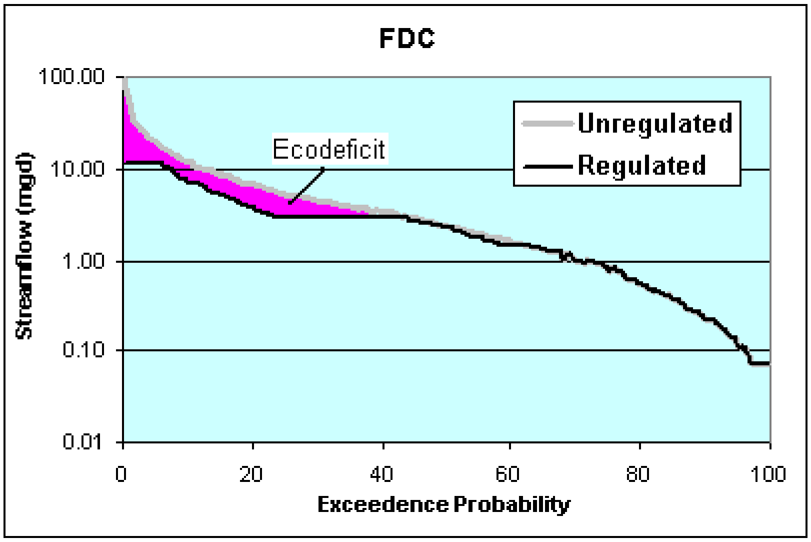

Figure 4.1 The Ecodeficit Flow Duration Curve Estimate (from Vogel et al. 2007)

4.3 Selecting Hydrologic Statistics for Assessment of Alteration and Criteria Development

Whether an existing software program (e.g., IHA, HIT/HAT) or another approach (FDC approach or newly-programmed) is used, a subset of hydrologic statistics should be chosen for (1) assessing alteration; (2) evaluating flow alteration for the development of flow-ecology relationships; and (3) implementation of flow standards. Around the world, a range of approaches have been used to narrow hydrologic statistics to a set that is useful for management. These typical flow variable-selection approaches can be generally grouped into four categories that need not be mutually exclusive:

1. statistical (e.g., as part of HIP process);
2. multi-metric (e.g., DHRAM);
3. expert-input driven (e.g., UK case study); and
4. flow-ecology response driven (e.g., TN project, PA IFM).

The ELOHA framework suggests that the subset of hydrologic statistics to be used by managers result from river classification (i.e., a statistical approach) plus any "additional hydrologic variables of management interest" (i.e., expert-input driven). Additional narrowing of key flow variables can be done adaptively as empirical flow alteration-ecological response relationships are developed for specific flow parameters as covered in Section 5. But, given that this process of developing empirical relationships takes time, this section will focus on defining an initial set of key flow statistics for use in water management.

A key factor to consider in selection of a hydrologic parameter is how accurately it can be simulated or estimated. As discussed in Section 2, error rates differ for various flow parameters, with parameters representing extreme or infrequent events often having the highest error. As pointed out in Kennen et al. (2007), some IHA statistics, such as the frequency of rises or falls in the hydrograph, are very sensitive to errors in daily flow estimation. Such statistics should not be selected to assess alteration if the baseline and current daily flow hydrographs are estimated.

4.3.1 Statistical Approaches

The Hydroecological Integrity Assessment Process (HIP) approach takes a statistical analysis-driven approach to defining key statistics for assessing alteration. As described in Henriksen et al. (2006) and in Section 3, HIP uses Un-Weighted Pair Group Method Analysis and Principal Components Analyses to classify rivers in a geographic area by hydrologic regime type. This hydrologic classification is followed by the identification of statistically significant, nonredundant, hydroecologically relevant indices associated with the five major flow components (a total of 11 subcomponents) of the flow regime for each stream class. These 5 major flow components are magnitude, timing, duration, frequency and rate of change. The 11 subcomponents are: low, average, and high magnitude; low and high frequency; low and high duration; low, average, and high timing; and average rate of change. In HIP, this suite of hydrologic indices is seen to adequately characterize the flow regime for each stream class (Olden and Poff 2003). *Case Study 2: Stream Classification in Pennsylvania using Hydroecological Integrity Assessment Process (HIP)* provides an example of methods and results of this statistical approach to defining key flow statistics.

To date, the statistics that have come out of similar classification processes in New Jersey and Missouri using the HIP process have only been used to assess hydrologic alteration. They have not yet been applied for the development of flow-ecology relationships or for the development and implementation of flow criteria. One of the major strengths of the HIP approach is that it avoids the substantial redundancy examined in Olden and Poff (2003). The potential weaknesses of the subset of statistics that result from this process are that (1) they may be non-intuitive to managers and difficult to manage for (e.g., coefficient of variation statistics); (2) they may be difficult to communicate to water users; and (3) it may be difficult to associate their alteration with ecological integrity metrics due to lack of clear mechanistic linkages. In New Jersey, for example, the New Jersey Department of Environmental Protection has generally found that monthly flow statistics are more explainable to water managers and may be used in New Jersey for standard setting (J. Hoffman, NJ DEP, personal communication). These potential limitations need to be tested, however, since application of HIP is recent and flexible enough to allow for adaptation based on experience in the field.

4.3.2 Multi-metric Statistical Approaches

Regardless of the statistics used to evaluate change from a baseline condition, there will be a trade-off between ability to represent natural variability in flow regimes and utility for development of instream flow criteria. Defining and implementing criteria for even 11 statistical variables, as above, may likely be unwieldy for managers. To that end, some approaches have been developed to try to condense a range of statistical variables into overall indices of hydrologic impact. One such approach is the Dundee Hydrological Regime Alteration Method (DHRAM), which uses results of the IHA analysis of a flow time series to classify the degree of alteration of a flow regime (Black et al. 2005) through a multi-metric approach. The 64 IHA statistics (not including EFCs), grouped into 10 categories, are used to arrive at summary "impact points" associated with the hydrologic alteration at the point of interest (a location in which impacted and un-impacted conditions can be estimated). These impact points are used to define an overall severity of hydrologic alteration in a manner that "classifies" the degree of hydrologic alteration into 5 levels, from unimpacted to severely impacted.

Although the summary "impact points" in DHRAM suffer from the redundancy inherent in using all the IHA statistics as well as the lack of relative weighting given to different types of flow alteration, this approach can be used to define an overall composite index of hydrologic alteration relevant to a particular resource of concern. A modification of this DHRAM multi-metric approach has been developed in Rhode Island for non-regulatory hydrologic alteration impact assessment (A. Richardson, RI DEM, personal communication).

4.3.3 Expert-input Driven Approaches

In attempts to define a set of hydrologic statistics that are easy to understand, linked to ecological response, and applicable to water management, a number of scientists and water managers have turned to expert-driven approaches to define key hydrologic statistics. *Case Study 3: Environmental Flow Standards to Meet the EU Water Framework Directive* is an excellent example of this approach. The key hydrologic variables that came out of that process included mean January flow, mean April flow, mean July flow, mean October flow, Q95, Q5, and a baseflow index.

In another example of an expert-driven set of key flow statistics, the Massachusetts Water Resources Commission (WRC) developed a methodology in 2001 for designating "stressed basins" in Massachusetts, and published a map of stressed basins. This method ranked gaged basins in Massachusetts according to three low-flow statistics. The 2001 method did not indicate whether the low flows were caused by natural or human factors. In addition, the method considered only potential flow-related stresses; potential water-quality-related stresses were not considered, and biological responses were not included. Working for the WRC, the Massachusetts Department of Conservation and Recreation, Office of Water Resources, in consultation with a task force and the USGS, is developing a revised methodology for basin stress classification. The project focuses on quantifying and mapping indicators of flow-alteration and water-quality at appropriate scales across the State. A subsequent phase is intended to use the results of a concurrent USGS-DCR aquatic ecology study relating fish-community characteristics to potential flow-alteration and water quality indicators, and incorporate fish-community response into the basin stress methodology. In this way, this project will be working to develop empirical flow alteration-ecological response functions.

Key to the Massachusetts project is the development and mapping of flow-alteration indicators to represent the degree of present flow alteration from water withdrawals and return flows. The analysis relies upon data and simulations of the MA Sustainable Yield Estimator (SYE) application and is being carried out at two scales: Hydrologic Unit Code 12 basins (HUC-12~40 mi^2) and a finer basin scale derived from the National Hydrography Dataset (NHD) basins (~5 mi^2). Flow alteration indicators include mean annual flow, median January flow, median April flow, median August flow, median October flow, median annual 7-day minimum flow, and, median annual low-pulse duration.

These seven hydrologic indicators were decided upon through an expert-input process lead by DCR and USGS. USGS will prepare a Scientific Investigation Report summarizing the methodologies applied and the resulting maps of each of the parameters evaluated. The report is expected to be completed in the spring of 2009 (L. Hutchins, MA DCR, personal communication).

The IHA program and associated publications do not provide guidance or a methodology to narrow the range of statistics examined for assessment of alteration. Recent work from TNC emphasizes use of the Environmental Flow Components (Mathews and Richter 2007). These EFCs have recently been incorporated as a model for instream flow management in a National Research Council report that provides recommendations to the state of Texas (NRC 2005). This report recommends that four hydrologic flow components be protected: overbank flows, high pulses, baseflows, and subsistence flows. These flows can be approximated by either program, but are more easily calculated using the IHA program.

There have been other attempts to define key flow statistics for purposes of management that rely on expert-input to narrow hydrologic variables, but also use other techniques. The technical committee of non-profit, private, and agency scientists convened as part of the Connecticut Department of Environmental Protection's instream flow standard development process arrived at a set of statistics proposed for use in evaluating hydrologic alteration in Connecticut rivers. As described in *Case Study 6: Connecticut Draft Streamflow Protection Regulations – Framework and Standard Development*, the process began with a set of variables defined by a nationwide hydrologic classification approach (Olden and Poff 2003) and was modified by expert input. This subset of variables was not linked to particular stream types as were those defined through the classification process that is part of HIP.

Finally, as described in Section 5, the process of developing conceptual flow-ecology response relationships can be useful to defining a subset of flow statistics for water management. This is a structured way to lead a group of experts towards a subset of flow statistics that, when altered, are likely to have ecological responses associated with them.

4.3.4 Flow-ecology Response Driven Approaches

This last category of approaches to defining a subset of flow statistics for water management represents a more analytical approach to selecting key hydrologic statistics based on the results of significant technical work. An example of this approach is work by USGS in Tennessee in which hydrologic characteristics that are correlated to fish community health and structure are identified. The investigation revealed functional connections between insectivorous fish and three hydrologic metrics: constancy, moderate floods, and streamflow recession rates. This work is described further in Section 5.

4.4 Recommendations for Selecting Hydrologic Statistics and Assessing Hydrologic Alteration

Assessing hydrologic alteration using a small set of ecologically-relevant flow statistics is important for assessing the impacts of current of future water management, the development of flow alteration-ecological response relationships, and the development of instream flow criteria.

This section reviewed the primary approaches for assessing hydrologic alteration and defining a set of hydrologic statistics useful to water management. The manner in which hydrologic alteration is assessed depends upon the methodologies used for defining baseline and alternative (current or future) conditions. As discussed in Section 2, our recommended approach is to develop estimated daily time series for baseline and current/future condition. Once a daily time series has been developed, there are two readily available programs to analyze this type of data: the Indicators of Hydrologic Alteration (IHA) and the Hydrologic Assessment Tool (HAT). The difference between the two analysis packages is not substantial, but there are trade-offs associated with using one or the other that described in the body of the text. Whether or not a daily time series is developed, there are also other approaches for assessing alteration that can be readily programmed into SAS, Excel, or directly into a water management decision support application.

Although other programs (e.g., Excel) could be coded to provide similar statistics, the advantage of using the statistics in these existing programs is that they are currently being used around the country and have been, or can easily be, screened for ecological applicability. Our recommendation is to develop a Pennsylvania-specific Water Management Decision Support System that can either link directly to either program (IHA or HAT) or can be programmed to calculate the array of statistics that these programs, combined, can provide. The most economically efficient approach would be to create a PA Water Management DSS with IHA and HAT compatibility. The Massachusetts Sustainable Yield Estimator, for example, has such capability.

The recommended approach for arriving at a limited set of statistics that meets the goals set out at the beginning of this section involves an iterative process beginning with the results of the hydrologic classification. The pilot classification for Pennsylvania in this report includes a set of "primary and secondary hydrological indices" that resulted from a principal components analysis. These statistics generally explain the dominant pattern of hydrologic variation for each of the 11 identified components of the flow regime and are differentiated by each river class. These statistics have limited redundancy and represent intra- and inter-annual variability in flow regimes, but may not satisfy some of the other goals listed above (e.g., conceptual linkages to ecological response, easy to use by water managers).

Any statistics resulting from the final Pennsylvania hydrologic classification should be modified and supplemented based on an expert input process. This process should begin with the definition of conceptual flow alteration-ecological response relationships as discussed in Section 5. Statistics that meet the goals set out at the beginning of this section, including linkage to ecological response are recommended for use in water management. These flow metrics would be used to assess hydrologic alteration associated with current or future water use. They should also be central to the development of flow alteration-ecological response relationships and the development of instream flow criteria.

SECTION 5: DEVELOPING FLOW ALTERATION – ECOLOGICAL RESPONSE RELATIONSHIPS

5.1 Background

A variety of methods are available for determining site-specific environmental flow requirements, each with their strengths and weaknesses. In contrast, few methods exist or have been applied at the spatial scale of a state or large river basin. Furthermore, environmental flow criteria in the U.S. are still generally focused on minimum flows for a river or stream. This emphasis can have unintended consequences if, for example, tributary reservoirs are necessary to "meet minimum flow requirements" downstream. Nonetheless, minimum flow protection continues to dominate the practice for state policy since there is no accepted method to define ecologically-based flow criteria at the state or regional scale.

The recent paper by Arthington et al. (2006) provides a model for the development of "regional environmental flow standards." The authors, an international group of river ecologists, assert that despite advances in site-specific approaches to determining environmental flow requirements, most decisions about water quantity issues are made based on simple hydrological "rules of thumb." This is a direct result of the inability of current methods to deliver relevant information on environmental flow criteria within the cost and time constraints that face water managers. So while site-specific studies will continue to be important for the most challenging water management issues, the authors emphasize the need for a holistic approach that can be readily applied across broad spatial scales.

Building upon the work in the Arthington et al. (2006) paper, Poff et al. (in review) have recently provided the Ecological Limits of Hydrologic Alteration (ELOHA) framework. Both papers emphasize the importance of developing relationships between flow alteration and ecological responses that can be applied to multiple rivers within a state or basin. This section reviews options for and makes recommendations on promising approaches to serve as a basis for environmental flow criteria statewide, featuring the development of flow alteration-ecological response relationships.

The recommendations in this report are generally consistent with the work of the authors of Arthington et al. (2006) and of those crafting the ELOHA approach. We begin this section with a review of the general types of river-specific approaches for determining environmental flow requirements that have been applied globally. Experience implementing these approaches worldwide led to the development of the ELOHA framework. Then, we review statewide approaches that can serve as a basis for developing environmental flow criteria that have recently been completed or are underway. Finally, we conclude with recommendations on developing semi-quantitative and quantitative flow-ecology relationships to serve as a basis for setting statewide environmental flow criteria. Note that the content of Section 5 and Section 6 are tightly linked, with Section 6, "Defining and Using Environmental Flow Criteria in Decision-

Making" focused on the final setting and implementation of criteria, while this section provides the foundation for that work.

5.2 River-Specific Methods for Determining Environmental Flow Criteria

The vast majority of applications of environmental flow methodologies around the world are river-specific, and it is worthwhile reviewing them because each type has elements that are applicable in a statewide context. Tharme (2003) reviewed river-specific environmental flow methodologies from around the world, ranging from "relatively simplistic, reconnaissance-level approaches for the early phases of countrywide, water resource planning initiatives, to resource intensive methodologies for highly utilized, individual catchments or sites." Descriptions of environmental flow methodologies (EFMs) used to define environmental flow requirements (EFRs) are excerpted from Tharme (2003). Each description is followed by notes and example applications relevant to development of statewide instream flow criteria in Pennsylvania. The four general categories are (1) hydrological; (2) hydraulic rating; (3) habitat simulation; and (4) holistic methodologies.

"The simplest, typically desktop EFMs, **hydrological methodologies**, rely primarily on the use of hydrological data, usually in the form of naturalized, historical monthly or daily flow records, for making environmental flow recommendations. They are often referred to as fixed-percentage or look-up table methodologies, where a set proportion of flow, often termed the minimum flow, represents the EFR intended to maintain the freshwater fishery, other highlighted ecological features, or river health at some acceptable level, usually on an annual, seasonal or monthly basis."

Many of these methodologies have been applied at the regional or statewide scale and they are relevant to the development of initial criteria that can be validated through use of empirical flow-ecology relationships. Examples: New England Aquatic Base Flow Method, Tennant Method.

"**Hydraulic rating methodologies** are approaches that use changes in simple hydraulic variables, such as wetted perimeter or maximum depth, usually measured across single, limited river cross-sections (e.g., riffles), as a surrogate for habitat factors known or assumed to be limiting to target biota. The implicit assumption is that ensuring some threshold value of the selected hydraulic parameter at altered flows will maintain the biota and/or ecosystem integrity. Environmental flows are calculated by plotting the variable of concern against discharge. Commonly, a breakpoint, interpreted as a threshold below which habitat quality becomes significantly degraded, is identified on the response curve, or the minimum EFR is set as the discharge producing a fixed percentage reduction in habitat."

These methodologies tend to be particularly difficult to generalize to larger spatial scales. However, they may be useful to help develop initial, generalized linkages between high flow hydrologic characteristics and metrics of riparian health on a river-by-river basis to inform statewide hypotheses. Examples: Wetted Perimeter Method, R2CROSS.

"**Habitat simulation methodologies** are also referred to as microhabitat or habitat modeling methodologies. These techniques attempt to assess environmental flow requirements on the basis of detailed analyses of the quantity and suitability of instream physical habitat available to target species or assemblages under different discharges (or flow regimes), on the basis of integrated hydrological, hydraulic and biological response data. Typically, the flow-related changes in physical microhabitat are modeled in various hydraulic programs, using data on one or more hydraulic variables, most commonly depth, velocity, substratum composition, cover and, more recently, complex hydraulic indices (e.g., benthic shear stress), collected at multiple cross-sections within the river study reach. The simulated available habitat conditions are linked with information on the range of preferred to unsuitable microhabitat conditions for target species, lifestages, assemblages and/or activities, often depicted using seasonally defined habitat suitability index curves. The outputs, usually in the form of habitat–discharge curves for the biota, or extended as habitat time and exceedance series, are used to predict optimum flows as EFRs."

The Pennsylvania Instream Flow Method, currently in use in headwater trout streams, is one of the few examples of a habitat simulation methodology modified for use at a regional scale. Beyond expansion of this approach to other portions of Pennsylvania (e.g., the Piedmont), this is unlikely to be a primary source of regional flow-ecology relationships due to the resource intensive and site-specific nature of the approach. Examples: PHABSIM, MesoHABSIM.

"**Holistic methodologies** emerged from a common conceptual origin (Arthington et al. 1992) to form a distinct group of EFMs focused from the outset towards addressing the EFRs of the entire riverine ecosystem. They rapidly took precedence over habitat simulation EFMs in South Africa and Australia, countries that lack the high profile freshwater fisheries characteristic of North America and where the emphasis is on ensuring the protection of entire rivers and their often poorly known biota.

In a holistic methodology, important and/or critical flow events are identified in terms of select criteria defining flow variability, for some or all major components or attributes of the riverine ecosystem. This is done either through a bottom-up or, more common recently, a top-down or combination process that requires considerable multidisciplinary expertise and input. The basis of most approaches is the systematic construction of a modified flow regime from scratch (i.e., bottom-up), on a month-by-month (or more frequent) and element-by-element basis, where each element represents a well defined feature of the flow regime intended to achieve particular ecological, geomorphological, water quality, social or other objectives in the modified system. In contrast, in top-down, generally scenario-based approaches, environmental flows are defined in terms of acceptable degrees of departure from the natural (or other reference) flow regime, rendering them less susceptible to any omission of critical flow characteristics or processes than their bottom-up counterparts. The most advanced holistic methodologies routinely utilize several of the tools for hydrological, hydraulic and physical habitat analysis featured in the three types of EFM previously discussed, within a modular framework, for establishing the EFRs of the riverine ecosystem. They also tend to be reliant on quantitative flow-ecology models as input."

The approaches recommended in Arthington et al. (2006) and as part of the ELOHA framework, grew out the experience developed through the application of these holistic methodologies. For example, the Habitat Analysis Method has been used in Queensland to develop environmental flow criteria at the large basin scale. This approach identifies key habitats (e.g., within channel, floodplain) and develops a set of flow statistics to maintain those habitats and the processes that support them. This approach to deriving flow-ecology relationships depends heavily on expert opinion in the form of a Technical Advisory Panel (Arthington et al. 1998). This type of approach, adapted to regional application, is potentially appropriate for Pennsylvania. Examples: DRIFT, Building-block Method.

There are, of course, hybrid approaches that include aspects of multiple types described above. Annear et al. (2004) and Dyson et al. (2003) provide additional details on each of these methodologies.

5.3 Statewide Methods for Setting Environmental Flow Criteria

5.3.1 Hydrological methodologies

As mentioned above, among the most readily applied methodologies at the statewide scale are those based in hydrological "standard-setting" approaches. Prominent examples used or developed in the Eastern U.S. include the New England Aquatic Baseflow methodology (USFWS 1981), which establishes summer instream flow requirements based on the unregulated median August flow and has provisions for other hydrologically determined seasonal flows. Other hydrological standard-setting methods include the Maryland Method, which is based on the seasonal 85-percent exceedance value, and the Tennant Method, which is based on percentages of mean annual flow and has been applied in some Eastern states. The Susquehanna River Basin Commission uses a hydrological approach in its 2002 "Guidance for using and determining passby flows and conservation releases…," in dictating a passby flow of 20% of average daily flow for Pennsylvania streams that are neither Exceptional Value nor High Quality Waters. The Range of Variability Approach (RVA; Richter et al. 1997) is another hydrological methodology that has been used in site-specific applications but has not been applied in a statewide context. Building on a state-specific implementation of the Hydroecological Integrity Assessment Process (HIP), New Jersey DEP may use a modification of the RVA approach as part of their State Water Planning process. This may include using the 25^{th} to 75^{th} percentile range of variability as a basis for defining an acceptable range of alteration for key flow statistics derived from comparing developed flow conditions to a baseline hydrologic daily time series. (J. Hoffman, NJ DEP, written communication).

The RVA approach is similar to the Dundee Hydrological Regime Alteration Method (DHRAM) which uses results from the Indicators of Hydrologic Alteration (IHA) analysis of a flow time series to classify the degree of alteration of a flow regime (Black et al. 2005). This method was developed to support the European Union's Water Framework Directive which seeks to bring water bodies in Europe up to a basic quality standard. In DHRAM, hydrologic "impact points" based on deviation scores in IHA parameters, are summed up at any site, and the risk of damage to instream ecology is defined based a simple look-up table of "DHRAM classes" that ranges from unimpacted condition to severely impacted condition.

Finally, Arthington et al. (2006) suggest that a hydrological approach could be used to develop first approximation environmental flow criteria. The authors suggest that frequency distributions of key index gage flow statistics could be developed for each river type once classification has occurred. These frequency distributions (Figure 5.1(c)) can be examined for variables of interest and "benchmarked" for risk of ecological damage. For example, the authors recommend a 95th percentile of the reference frequency distribution for each variable (e.g., 7Q10) as a basic benchmark (or criteria) when ecological validation can not be achieved. Thus, if a measured or estimated 7Q10 value fell outside the 95th percentile of the distribution of 7Q10 values within that class, it would violate the benchmark. This benchmark could be refined by expert scientific workshop input, but is meant primarily as an initial basis for criteria until ecologically-validated criteria can be developed.

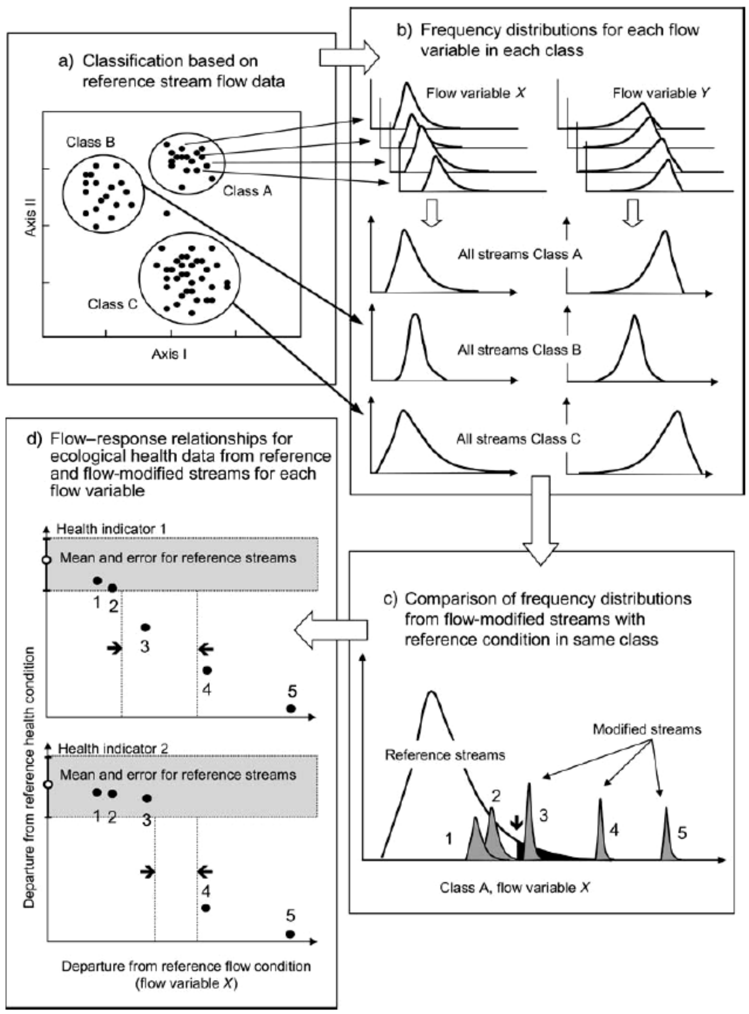

Figure 5.1 Approach to Regional Environmental Flow Standards (from Arthington et al. 2006)

<u>5.3.2 Developing Criteria Based on Flow-Ecology Relationships</u>

The National Research Council report (2005) that reviewed the state instream flow program in Texas evaluated the trade-offs associated with different approaches to linking flow alteration to biological response (Table 5.1). The authors suggested that approaches that link flow alteration to assemblage structure or coarse ecological indicators are more appropriate for large spatial scale applications than more resource-intensive approaches based on habitat modeling (e.g., Instream Flow Study: Pennsylvania and Maryland (Denslinger et al. 1998)). However, Table 5.1 points out the drawbacks associated with these approaches, including higher uncertainty and a lack of specificity of mechanisms leading to the ecological response.

Table 5.1 Approaches to Develop Flow-Ecology Linkages (from NRC 2005)

Holistic <——> Specific Higher Uncertainty <——> Lower Uncertainty						
	Ecosystem Indicator <——> Metrics with Direct Response to Flow					
Approach	Coarse Ecological Indicators	Assemblage Structure	Habitat Guilds	Species HSC	Population Models	Individual-based Models
Key Characteristics	Integrates many components/ processes; correlational	Integrates many components; correlational	Integrates many components; correlational	Individual or species; correlational	Dynamic simulation of aggregate response variable	Dynamic simulation of ecological mechanisms
Strengths (Benefits)	Rapid, cheap, repeatable <——> Predictive with high resolution					
Weaknesses (Limitations)	Ecological responses & Mechanisms not specified <——> System site specific expensive, time consuming					
Settings where appropriate	Any <——> Those for which much information is available					
Appropriate spatial scales	Intermediate to large	Intermediate to large	Intermediate	Small	Large	Small to large
Outputs	A single target value & correlations with flow	A set of values & correlations with flow	A set of values & correlations with flow	A series of values & correlations with flow	A series of simulated values at variable flows	A series of simulated values at variable flows
Examples/Applications	Multiple applications water quality, watersheds, etc.	Channel-floodplain connectivity	Leonard & Orth 1988	Many ISF programs	Long history but few formal applications to ISF	Jager et al., 1997, 2001, Railsback et al., 1999, 2002, others

The primary recommendation in the PA Statewide Instream Flow Studies Paper (Young 2006), Arthington et al. (2006), and the ELOHA approach is to use data from reference and flow-modified rivers to define flow alteration-ecological response relationships for key flow statistics. These relationships serve as a general predictive model that describes a probable biological response based on the best available data and expertise. Young (2006) and Davies and Jackson (2006) provide examples of conceptual models of relationships between flow alteration and ecological responses (Figure 5.2, Figure 5.3). In both figures, the critical attributes are (1) some scale of biological impact; (2) some scale of hydrologic alteration (or stressor); and (3) thresholds (represented by vertical lines or numbered zones) that delineate different levels of impact, transforming a continuous relationship into a more discrete scale.

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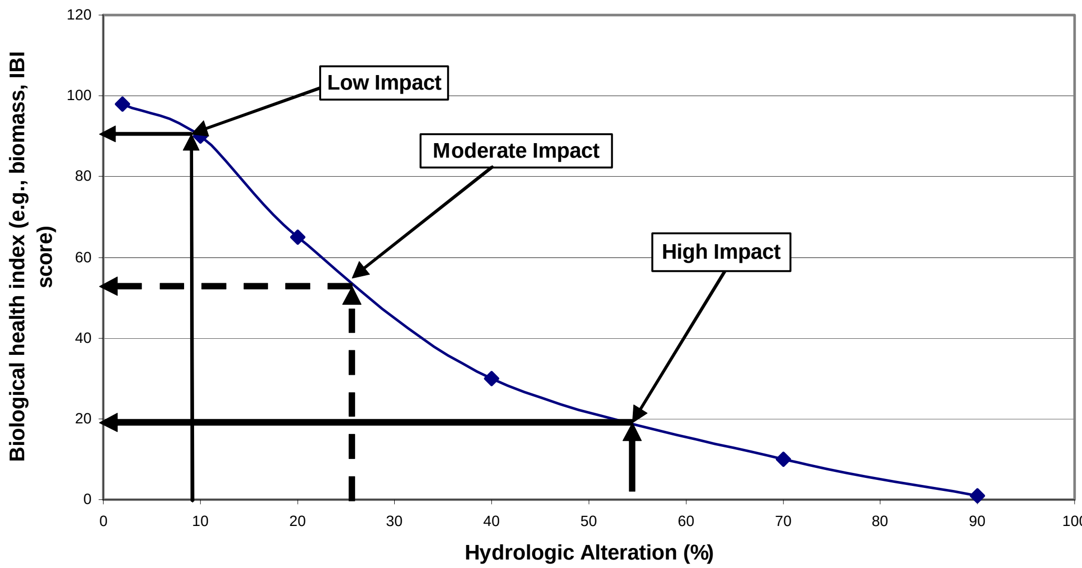


Figure 5.2. Conceptual model of the relationship of hydrologic alteration to biological IBI scores (from Young 2006)

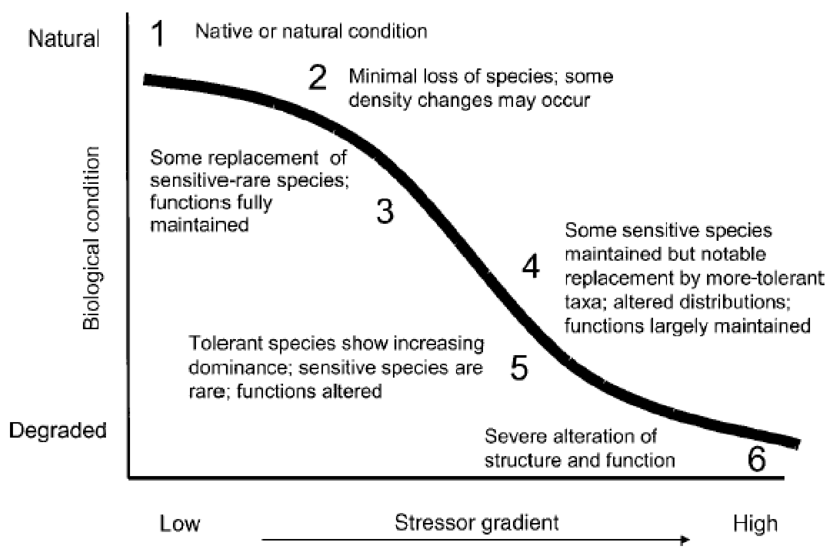


Figure 5.3 The Biological Condition Gradient (from Davies and Jackson 2006)

These flow-ecology response curves act as a set of hypotheses about linkages between flow alteration and ecological response. These hypothesized relationships are defined for each river type, using the best data available at the time. These relationships are tested and validated over time using monitoring data, allowing for modification of initial relationships or the addition of new ones. These relationships will likely have varying functional forms, such as linear, threshold, or curvilinear (Figure 5.4).

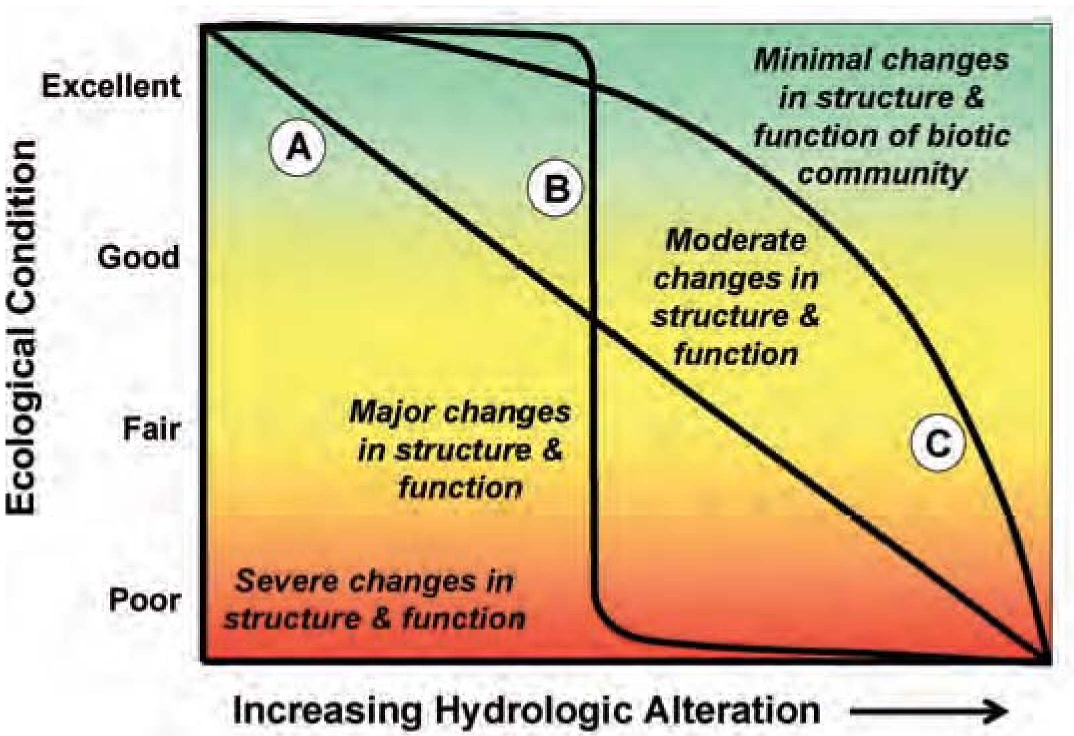


Figure 5.4 Conceptual flow alteration-ecological response relationships. Possible forms include: linear (A), threshold (B), and curvilinear (C). The form of the relationship depends on the specific ecological and flow statistics analyzed.

Conceptual models are hypotheses that link flow alteration to anticipated ecological impacts. These models are useful for guiding empirical investigation of flow-ecology relationships. The scientific literature provides a basis for many relationships representing the range of flow components (e.g., extreme low flow, low flow, high pulses, and large floods) and characteristics (magnitude, timing, duration, frequency, and rate-of-change) (see Poff et al. 1997, Extence et al. 1999, Bunn and Arthington 2002). In a recent USGS publication, Bencala et al. (2006) suggest that synthesizing "the existing science of riverine ecosystem processes to produce broadly applicable conceptual models" should be one of the USGS's highest priorities. This recognition of the importance of developing applicable conceptual models should lead to more investment on this topic by USGS. Examples of some of these conceptual relationships have also been compiled for use in developing flow-ecology relationships in the ELOHA paper (Poff et al. in review). Table 5.2 lists selected examples of possible hypotheses. Flow categories are based on "environmental flow components" from Mathews and Richter (2007).

Table 5.2 Examples of hypotheses to describe expected ecological responses to flow alteration formulated by the authors of the ELOHA paper during a 2006 workshop.

Extreme low flow
Hypothesis: Depletion of extreme low flows in perennial streams and subsequent drying will lead to rapid loss of diversity and biomass in invertebrates and fish due to declines in wetted riffle habitat, lowered residual pool area/depth when riffles stop flowing, loss of connectivity between viable habitat patches, and poor water quality.

Low Flow
Hypothesis: Augmentation of low flows will cause a decline in richness and abundance of non-fluvatile species with preferences for slow-flowing, shallow-water habitats, whereas fluvatile or obligate rheophilic species would shift in distribution or decline in richness and abundance if low flows were depleted.

Small floods / high flow pulses
Hypothesis: Lessened frequency of substrate-disturbing flow events leads to reduced benthic invertebrate species richness as fine sediments accumulate, blocking substratum interstitial spaces. Hypothesis: A decrease in inter-annual variation in flood frequency (i.e., stabilized flows) will lead to a decline in overall fish species richness and riparian vegetation species richness, as habitat diversity is reduced. Hypothesis: Changes in small flood frequency will lead to changes in channel geometry (dependent upon boundary materials).

Large floods
Hypothesis: Lessened frequency or extent of floodplain inundation will lead to reduced invertebrate and fish production or biomass due to loss of flooded habitat and food resources supporting growth and recruitment.

Ideally, any conceptual models or hypotheses will eventually be tested using existing hydrologic, biological and habitat data or new data collected specifically for this purpose. Such testing using regionally-relevant data can empirically establish relationships between flow alteration and ecological responses and strengthen the defensibility of these relationships if challenged. Examples of some of these **empirical flow-ecology relationships** are included in Section 5.3.3 and an initial attempt to complete this work in Pennsylvania is included in *Case Study 5: Development of Flow Alteration-Ecological Response Curves for Pennsylvania Streams*. It should be noted, despite these examples, that there are still relatively few examples of these empirical flow-ecology relationships useful for setting instream flow criteria documented from around the world. This is likely due to the recent emergence of this approach and the ELOHA framework, but it also may be due to resources required and the scientific uncertainties associated with these relationships given the confounding factors in the environment.

Given scientific uncertainties, and the fact that funding or time may not always be available for the development of empirical flow alteration-ecological response relationships, structured expert input can be used to develop **semi-quantitative flow-ecology** relationships or **risk thresholds** that can also serve as a basis for instream flow criteria. Expert scientific input is used to take flow-ecology relationships from other regions, along with other relevant scientific information, to either create a set of relationships or risk thresholds that can apply locally.

Case Study 6: Connecticut Draft Streamflow Protection Regulations – Framework and Standard Development, provides one example of such an approach. USGS researchers in Georgia developed flow-ecology relationships between a permitted water withdrawal index (as % of the

7Q10 flow statistic) and fish richness in the Georgia piedmont (Freeman and Marcinek 2006). A Connecticut technical advisory committee, made up of local scientists, decided that this Georgia relationship could be used in Connecticut, and interpreted the results in a manner that was useful to policymakers seeking instream flow criteria applicable annually in a range of flow conditions. This can be considered the creation of a semi-quantitative flow-ecology relationship.

An example of the development of risk thresholds is included in *Case Study 3: Environmental Flow Standards to Meet the EU Water Framework Directive*. In this case an expert scientific workgroup from the United Kingdom used existing literature and local knowledge to arrive at a set of thresholds of risk (low, medium, and high) for meeting "good ecological status" and linked these to degrees of hydrologic alteration as measured by six key hydrologic statistics. This is a type of flow-ecology relationship in which increasing degrees of hydrologic alteration are linked directly to ecological condition.

As is apparent from these examples, expert input applied through a workshop setting can be critical to the process of developing flow-ecology relationships for instream flow criteria. Expert input is key to making decisions about interpretation of data that leads to the development of flow-ecology relationships that are in a form more useful for decision-makers. Not surprisingly, a number of holistic river-specific environmental flows methodologies rely heavily on "expert teams," including the Building Block Methodology in South Africa (Tharme and King 1998) and a similar approach used by The Nature Conservancy on rivers in the United States (Richter et al. 2006). As discussed in the IUCN Report on environmental flows (Dyson 2003) use of an expert team benefits from flexibility which can lead to consensus building amongst experts who can come to a strong recommendation based on the data and model results.

A Pennsylvania example of the use of expert input as a basis for defining instream flow criteria is work that went into the PA/MD Instream Flow Studies (Denslinger et al. 1998). During that study, the documentation indicates that a number of expert driven decisions were required that involved the judgment of the local biologists. Examples include: (1) the decision to use the median monthly habitat as a measure of the habitat available with the natural flow regime; (2) use of the no net loss of habitat criterion for determining the level of flow that would protect the median monthly habitat; and (3) definition of inflection points in the wetted perimeter analysis. These expert decisions permitted instream habitat goals to be set, limiting the uncertainty about data interpretation that was moved into the policy (criteria-setting) process.

There are some potential weaknesses of an expert driven process approach. Acreman (2005) reports that use of expert opinion has met with some resistance from stakeholders in the United Kingdom due to lack of transparency. Another potential disadvantage is that the recommendations are less replicable. However, such issues have been mitigated through efforts at broad participation in the overall program development effort and careful documentation of the process.

5.3.3 Examples of Regional Flow-Ecology Relationships Completed or Underway in Other States

Generalized flow-ecology relationships are being developed or used for environmental flow protection in several states and countries. Relevant examples include:

USGS researchers in **Georgia** developed defined flow-ecology relationships between permitted water withdrawals (as a percentage of the 7Q10 flow statistic) and fish richness in the Georgia piedmont (Freeman and Marcinek 2006). This work is informing the policy process in Georgia, but also is being used as a foundation for a draft instream flow protection standard in the state of Connecticut.

In **Massachusetts**, the Target Fish Community (TFC) method (Annear et al. 2004, Bain and Meixler 2000) is being used to develop reference fish communities that reflect conditions where streamflow alteration and other stresses are relatively minor. The TFC method allows a quantitative description, in terms of the fish community, of a natural or near natural condition. This community acts as the "expected" community to which sampled or "observed" fish communities can be compared. This comparison of Target Fish Community to sampled community is being done through a cooperative project between USGS and the MA Department of Fish Wildlife for a range of watersheds in Eastern Massachusetts. These watersheds have estimates of unregulated baseline and current condition flows throughout the project areas, which have been developed through HSPF modeling. Initial results of this work are promising, with changes in fish communities (e.g., loss of fluvial specialist fish) linked to changes in flow metrics (e.g., August median flow) in Eastern Massachusetts in a manner that fits a flow alteration-ecological response curve. These changes in fish communities are also linked to watershed factors correlated with water use (e.g., increase in impervious surface), but it seems likely that water use sets an upper bound on the fish community integrity of Eastern Massachusetts. The result of this work is designed to inform development of flow standards in Massachusetts through the Department of Conservation and Recreation (D. Armstrong, USGS MA, personal communication). The likely regional transferability of this approach is one of its major advantages for the development of environmental flow standards.

In **King County, Washington**, the "Normative Flows Project" was designed around the objective of linking flow alteration to biological condition in the county's streams and rivers to inform county policy decisions. The project used hydrologic data, primarily from an HSPF model, and existing macroinvertebrate data from around King County. The project was able to make linkages between index of biological integrity scores for macroinvertebrates and flow metrics that indicated a change in disturbance regime (e.g., frequency of pulse events, duration above mean 2 year flow). However, this project did not fully achieve its goals due to factors including: (1) a lack of biological data; (2) use of redundant hydrologic measures; (3) errors in flow simulation; and (4) poor spatial and temporal linkages between biological and hydrologic data sites (Cassin et al. 2005). The lessons from the King County Normative Flows Project should inform any effort to develop flow-ecology response relationships in Pennsylvania.

In **New Jersey**, Kennen et al. (2008), used a statewide watershed runoff model (SWRM), as described in Section 2, to examine the major environmental and hydrologic factors contributing to changes in the aquatic invertebrate assemblage structure. Hydrologic metrics that related to invertebrate assemblage structure included the ratio of Q25 to Q75, low pulse duration, and high pulse frequency. This first phase study has yet to be applied to developing instream flow standards in New Jersey, but this work could provide a basis for conceptual models of flow ecology relationships that can inform decisions about appropriate hydrologic metrics for criteria development.

In the **Tennessee River Valley (**includes portions of Tennessee, Virginia, North Carolina, Georgia, Alabama, and Kentucky), USGS is leading an initiative to develop functional connections between fish communities and hydrology that can be applied to water management. The first phase of this project was to identify hydrologic characteristics that are correlated to fish community health and structure. Approximately 100 hydrologic metrics were calculated for each gage, including standard hydrologic descriptors such as low-flow values, duration values, and mean annual streamflow, and other ecologically relevant metrics describing the magnitude, duration, frequency, timing, and rate of change of streamflow. This investigation revealed functional connections between insectivorous fish and three hydrologic metrics: constancy, moderate floods, and streamflow recession rates. The second and third phases of this study include the development and testing of a tool to predict hydrologic metrics at ungaged locations. Using this tool, managers would then have a method to estimate fish community health at stream locations without ecological data using hydrologic metrics proven to be relevant to the fish community. Additionally, managers would have hydrologic metrics that could be monitored to estimate potential change to fish community health with changes to the river (R. Knight, USGS TN, written communication).

In **Michigan**, a legislatively-appointed Groundwater Conservation Advisory Committee oversaw the development of a “flow-fish functional response curve” for each of the eleven stream types in Michigan. These curves illustrated the response of a fish population metric to increasing water withdrawals. The curves were constructed in a manner that allowed a stakeholder group to make decisions about the level of ecological risk associated with hydrologic alteration of low flows. This process is described in the *Case Study 4: Michigan Water Withdrawal Assessment Process*.

5.3.4 Examples of Regional Flow-Ecology Relationships Completed or Underway in Pennsylvania

Habitat impact curves in the Pennsylvania and Maryland Instream Flow Studies: As part of the Instream Flow Studies: Pennsylvania and Maryland (Denslinger et al. 1998), regional habitat impact curves were developed for wild brook trout, wild brown trout, and wild brook and brown trout combined. These flow-ecology response curves represent a regionalization of flow-trout habitat relationships developed within the stream classes of the Unglaciated Plateau and Ridge and Valley Freestone regions as well as two classes within the Ridge and Valley Limestone region. These relationships between withdrawal, passby flow, and habitat impacts for the three trout targets facilitated the development of regional passby flow criteria. This approach provides a strong model of using site-specific data and stream classification to develop regional flow-ecology response relationships. However, significant data collection time and expense was required for this approach, since site-specific data on flow-habitat relations was collected (typically on 30 study segments per region) before regionalization occurred. Given our review of existing methods in this section and expected costs, we believe that the approach used in these Instream Flow Studies should not be newly developed in regions outside its area of existing application. Instead, as recommended as part of the ELOHA framework, for these river types it may be more appropriate to pursue empirical flow alteration-ecological integrity response relationships.

Pilot application to develop flow alteration–ecological response curves in Pennsylvania: During the term of this project, we completed a pilot application to "use existing data to define the relationship between biological indicators and altered flow conditions for major river types across the state." We worked with the Technical Advisory Committee to select the Susquehanna River basin in PA as the watershed for this pilot study and to select hydrological and ecological data sources that could be used in this application. The objective was to develop a generalized predictive model that would estimate the degree of biological impact that can be expected from a given degree of hydrologic impact for a given stream type.

To accomplish these goals, we developed a water withdrawal index using data from the Pennsylvania Water Analysis Screening Tool (WAST). More specifically, we used data on cumulative water withdrawals at "pour points" across a portion of the Susquehanna River basin to develop an index of cumulative water use relative to a low flow parameter (i.e., 7Q10) in a manner similar to the Freeman and Marcinek (2006) and Weiskel et al. (2007). By linking a water withdrawal index at these pour points to locations with available biological data, we developed statistically-based, quantitative estimates of ecological response. Results of this pilot application are described in detail in *Case Study 5: Development of flow alteration-ecological response curves for Pennsylvania streams.* We emphasize that this is an exploratory investigation of the feasibility of linking hydrologic alterations with ecological responses using existing data within Pennsylvania.

5.4 Recommendations for Developing Flow Alteration—Ecological Response Relationships

The recommended statewide approach to instream flow protection is through development of ecologically-based instream flow (also "environmental flow" or "streamflow") criteria. Environmental flow criteria will define an acceptable degree (limit) of flow alteration for a set of ecologically-relevant flow statistics to meet state ecological goals. Best available information can be used to link flow alteration to ecological response through conceptual relationships, risk thresholds, or empirical models for rivers and streams in Pennsylvania. Quantitative or semi-quantitative flow-ecology relationships, in contrast to the typical hydrological "rule of thumb" approaches, provide decision-makers explicit information on the risk of excessive hydrological alteration to natural resources of concern. Final instream flow criteria should be set through a stakeholder process (described in Section 6) that uses these quantitative or semi-quantitative flow ecology relationships.

We recommend that a combination of literature review, expert knowledge, conceptual models, and existing data be used to develop semi-quantitative flow-ecology relationships and ecological risk thresholds that can inform instream flow criteria (i.e., acceptable level of deviation from baseline values) for Pennsylvania. We recognize that despite growing literature and practice on the topic, there is no set of easily transferable flow-ecology relationships that can currently be used across Pennsylvania. Therefore, the flow-ecology relationships that result from this process of using existing data and literature will likely be semi-quantitative, with expert knowledge used to fill in missing information, deal with uncertainty in the data, and interpret the form of functional relationships between flow alteration and ecological response. This approach uses elements of options B1 and B2 in the Project Options Table (Appendix 1) combined with key aspects of the scientific process as described in *Case Study 3: Environmental Flow Standards to Meet the EU Water Framework Directive*. The results of this semi-quantitative approach may act as an initial basis for criteria in Pennsylvania, or given financial limitations and utility of results, it could act as a basis for criteria over a longer time period.

Our long-term recommendation is the development of empirical (quantitative) relationships linking flow alteration to ecological response across stream types in Pennsylvania. To do this, we recommend that Pennsylvania develop stressor-response relationships between flow alteration and ecological response using newly collected data and/or existing data not yet explored as outlined in the Project Options Table (Appendix 1) as option B3. Quantitative relationships, as they emerge, can be used to validate, refine, and strengthen the basis for streamflow criteria. Effort towards the development of these empirical relationships can begin in parallel with a semi-quantitative approach, or in sequence. The development of these empirical relationships should be pursued by collecting ecological integrity indicators across a gradient of flow impairment. Selection of both flow and ecological integrity indicators should be based on *a priori* conceptual models reflecting hypothesized flow-ecology linkages. However, we recognize that due to financial limitations or concerns about likely limitations of state data collection, Pennsylvania may choose not to go further than the semi-quantitative flow-ecology relationship output as a basis for instream flow criteria.

To implement these recommendations, we outline the following **steps**:

1. Draft a set of flow alteration - ecological response hypotheses for major river and stream types in Pennsylvania.

This set of hypotheses should:

- **Address a variety of relevant flow components flow components** (e.g., extreme low flow, low flow, high pulses, and large floods) and characteristics (magnitude, timing, duration, frequency, and rate-of-change);
- **Address a variety of relevant taxonomic groups** (e.g., fish, macroinvertebrates, riparian vegetation) and habitat types (e.g., floodplain, riffle area);
- **Describe the direction of the anticipated ecological response** (e.g., fish diversity will *decrease* as low flows *decrease*); and
- **Describe the functional form of the response** (e.g., linear, threshold, curvilinear).

Hypotheses should be developed for all major stream types in an existing or revised classification of Pennsylvania rivers and streams. Habitat-flow relationships have been developed for a number of rivers across the state, and a subset of them have been generalized across the coldwater trout streams type as part of the PA/MD Instream Flow Studies (Denslinger et al. 1998). These habitat-flow relationships typically focus on low flow parameters and on a small suite of species. Nonetheless, they act as an important basis for hypotheses about how flow alteration is likely to impact key aspects of ecological integrity in one system type. The conceptual relationships in Poff et al. (in review) can also serve as a starting point for additional hypotheses. Additional detail on this step is provided in the Project Options Table (Appendix 1), option B1.

2. Convene experts and conduct literature review to determine potential risk thresholds.

We recommend that expert input and literature review be used to define thresholds of hydrologic alteration beyond which ecological impacts would be anticipated. This process should be a collaborative effort between resource agencies, academic scientists, and other stakeholders. Depending on the existing literature and research supporting these thresholds, these thresholds could be used to establish initial flow criteria. *Case Study 3: Environmental Flow Standards to Meet the EU Water Framework Directive* provides one model for this process, as does the approach for developing initial criteria described in Arthington et al. (2006).

In addition, these thresholds would guide research and monitoring to develop quantitative relationships between flow alteration and ecological response. These investigations support an adaptive management process that defines risk thresholds and streamflow criteria based on best available information, with refinement and validation over time.

3. Begin validating flow alteration-ecological response hypotheses using existing data.

As hypotheses are developed, experts should also begin identifying site-specific or regional hydrological and ecological data that can be used to test these hypotheses. Hypothesis testing should build on experience and lessons learned in the pilot study to assess the impacts of water withdrawals on macroinvertebrate assemblages in the Susquehanna River Basin. Results are described in detail in *Case Study 5: Development of flow alteration-ecological response curves for Pennsylvania streams.*

The approach taken by the Michigan Groundwater Conservation Advisory Committee to define "flow-fish functional response curves" is also particularly promising for Pennsylvania. Existing fish databases, including those compiled and formatted by Pennsylvania Natural Heritage Program as part of the Pennsylvania Aquatic Community Classification, should be further reviewed to determine if they could be used to develop these kinds of relationships. A similar approach could be taken using other taxa, including macroinvertebrates, if sufficient data are available.

4. Define additional research and data collection needed to link hydrologic alteration to ecological condition.

Existing biological databases and hydrological information in Pennsylvania are likely sufficient for initial examinations of flow-ecology relationships, but will ultimately need to be supplemented. This is due primarily to the spatial and temporal disconnect between biological sampling sites and sites with adequate hydrologic information but also due to the different methodologies used by different resource agencies. Pennsylvania should also begin to supplement existing databases with additional sampling designed to detect impacts on indicators of ecological integrity from flow alteration as suggested by any conceptual models and initial flow-ecology relationships. This monitoring program would also be part of an adaptive management cycle that allows for validation and refinement of flow-ecology relationships, risk thresholds, and instream flow criteria. A monitoring program established specifically to detect ecological effects from flow alteration would take into account lessons learned from the Susquehanna River Basin pilot project (described in *Case Study 5: Development of flow alteration-ecological response curves for Pennsylvania streams*). Specifically, this program should select ecological monitoring sites that (1) spatially and temporally match sites with hydrologic data; (2) are distributed across a range of hydrologic alteration; and (3) are distributed across stream classes (if stream classes are defined) or distributed across streams and rivers with different sizes and physical and chemical characteristics. In addition, the monitoring program should develop or identify ecological response metrics that are hypothesized to respond to hydrologic alteration and address multiple taxonomic groups.

Any effort to develop environmental flow criteria would benefit significantly from a near-term investment in capacity to estimate baseline and current hydrologic conditions statewide. Recommendations for developing these estimates were covered in Sections 2 and 3, but it is worth emphasizing that validated flow-ecology relationships will be very difficult to create without this investment.

These recommendations reflect a sequence of approaches listed in the Project Options Table (Appendix 1).

SECTION 6: DEFINING AND USING ENVIRONMENTAL FLOW CRITERIA IN DECISION-MAKING

6.1 Background

The need for water resource managers to make decisions to ensure reliable supplies of water for human uses and to ensure ecologically sustainable rivers and streams underpins the need to develop comprehensive programs to protect environmental flows. Key to this process is to define environmental flow (instream flow) criteria that are linked to natural resource conditions and are used to delineate between acceptable and unacceptable water management. Environmental flow criteria, if designed appropriately, can inform a wide range of water management programs, including regulatory review programs, environmental review of new projects, local and regional water resource planning, investments in new infrastructure and other water planning guidance.

Many approaches and management frameworks in existing environmental protection programs, including water quality standards, wastewater permitting programs, and facility planning and environmental review programs can provide a basis on which to build a statewide approach to protecting environmental flows and meeting human water needs. One of the major benefits of having clearly defined allowable environmental flow goals and the associated criteria is the ability to provide predictable and transparent decision-making and to provide criteria for use in designing new projects.

Accordingly, well constructed environmental flow criteria meet the following **goals**:

1. Can be applied at most locations and river types across the state;
2. Protect critical aspects of natural flow variability in order to maintain ecological integrity;
3. Have a strong scientific foundation and peer review to ensure credibility in legal proceedings;
4. Can be adapted using new information;
5. Can be easily incorporated in permit conditions, planning decisions, or other water management efforts;
6. Can be interpreted in a way that allows for a level of protection that fits resource goals; and
7. Can be enhanced by site-specific studies.

6.2 Hydrologic Measures, Environmental Flow Criteria, and Limits and Operations

This report has analyzed a broad range of methods for analyzing the hydrologic conditions of rivers and streams, organizing river systems into classes of similar system types; developing measures of hydrologic alteration, understanding how flow alterations affect the ecology of stream systems, and building these flow-ecology relationships that can be used to develop protection criteria and risk thresholds.

In this section we discuss the socially-driven process of defining environmental flow criteria and describe how the output of the methods covered in this report can lead to useful management structures for environmental protection and water use management. A framework for this discussion is to consider four interrelated but distinct parts of such management systems. These include (1) defining protection goals; (2) defining criteria by which these goals will be achieved; (3) defining hydrologic measures which measure consistency with and deviations from these criteria; and (4) translating these criteria and measures into operating rules and withdrawal limits that can be implemented by operators of water withdrawal and management structures and facilities.

We define these terms as follows:

Environmental Flow Goals: These are the environmental and human use goals for rivers and streams. These are often tiered into classes and are usually expressed as narrative goal statements which can be achieved by meeting defined specific, often numeric, criteria.

Environmental Flow Criteria: These are flow conditions or degrees of hydrologic alteration that must be met on a consistent basis. Environmental flow criteria are linked directly to desired environmental outcomes and delineate between acceptable and unacceptable changes to hydrologic conditions. Environmental flow criteria are based on an understanding of how changes in hydrology are likely to impact ecological conditions, as represented by ecological indicators (e.g., index of biotic integrity, habitat conditions, etc.)

Hydrologic Measures: These are statistical measures used to describe the hydrology of a river and stream system and are used to describe alterations to natural or unimpacted flows of a river and stream.

Limits and Operating Rules: These are quantities of water that are allowed to be withdrawn or released from a control or water withdrawal structure. These are the "operating rules" and often are the "permit conditions" for a system in order to meet the environmental flow criteria.

6.3 Environmental Flow Protection Goals

Key to developing a system for protection is the need to answer the questions: "what level of protection?" or "protection of which ecological attributes?" Therefore, establishing protection goals or standards for water bodies is a key element of implementing environmental flow criteria. Such goal setting is typical of many environmental protection programs.

One useful frame of reference for goal setting is the state water quality standards required under the federal Clean Water Act and Pennsylvania's Clean Streams Law. These standards associate specific levels of protection to specific designated uses and to special classes of streams. The standards recognize that a "one-size-fits-all" goal for all streams is not appropriate for balancing the various uses. However, all classes must meet basic environmental goals (e.g., fishable, swimmable, or similar basic protections) and therefore all receive a fairly high level of protection. Certainly a single standard for the entire state or region is possible. However, the difficulty with a single standard is that it is likely to be either too lax and therefore allow high degree of alterations to all waterbodies, or too stringent and result in many existing conditions being unable to meet the standard.

It is important to note that in discussing protection goals we use the terms “class” and “classification” differently than in the hydrologic classification process as described in Section 3. Here we discuss *classes of protection* rather than classification of stream types.

A tiered approach to goal setting is also applicable to water quantity issues. Three hypothetical classes of protection provide a useful framework for thinking about protection classes: (1) special value streams; (2) typical streams; and (3) “working” streams. At one end of the spectrum of this hypothetical example are streams where maintaining high, near-reference ecological conditions is the primary goal. At the other end of the spectrum are streams that will be heavily used and altered while still maintaining good environmental quality. In the middle are what are euphemistically called “typical streams” which are neither of exceptional value nor are highly altered – their protection goal is to allow an “acceptable” degree of alteration while still maintaining most ecological values. Table 6.1 summarizes how these three hypothetical protection goal classes relate to generic degrees of alteration. The allowable flow alteration in this table, if made numeric, would constitute environmental flow criteria.

Table 6.1 Conceptual Stream Protection Classes

Stream class	**Brief description**	**Stream Goal**	**Allowable flow alterations**
“Special Value Streams”	Streams of exceptional or other high value where near natural conditions are desired	Natural structure and function of biotic community maintained.	de minimus alterations to natural flows and water levels.
“Typical Streams”	Streams that are neither of exceptional value nor heavily used	Minimal to moderate changes to structure and function of biotic community.	Some alteration of natural flows and water levels.
“Working Stream”	Streams that are heavily used for water withdrawal, wastewater discharge or other uses or alterations	Significant alteration of natural flows and water levels. Major changes in structure & moderate changes in function of biotic community.	Significant alteration of natural flows and water levels.

The development of environmental flow goals on a statewide or regional basis requires the development of a process to assign an appropriate protection goal to every stream or stream reach – that is to assign it to a particular class of protection. Continuing with the conceptual framework presented in Table 6.1, it requires every stream to be designated as either a special value, typical or working stream. An example of this approach is described in *Case Study 1: Connecticut Draft Streamflow Protection Regulations – Framework and Standard Development.*

For Pennsylvania, a set of hydrologic protection classes is likely to relate to both existing stream classifications and designated uses.

Since these classes of flow standards focus on goals for hydrology and water quantity, there are a range of factors that can be considered when developing protection classes, including:

- Existing stream classes and designations;
- Location within or adjacent to conservation lands;
- Level of existing development/impervious surface within watershed;
- Number and size of water withdrawals, diversions and impoundments;
- Number and size of return flows from wastewater and industrial;
- Existing 'development areas' or other planned changes;
- Presence of unique, rare or threatened species and communities; and
- Degree and type of recreational use.

Another aspect of establishing protection goals is the need to consider both existing conditions and the "desired future state" of the stream. Three basic situations can be anticipated: (1) streams where current condition and future condition are the same (e.g., currently unaltered and goal is to keep in that condition); (2) streams that are currently significantly altered where an improved protection class is desired (either not meeting any class of protection goals or meets one protection class but an high protection class is desired); and (3) streams where a protection class less protective than could currently be achieved is appropriate based on known or anticipated future water uses or alterations. Designing protection goals should include developing a process that allows for each of these situations to be anticipated.

Ensuring adequate public comment and participation as part of this process is an important element of setting and change protection goals. Structured public input provides an opportunity for both groups and individuals that seek to raise the protection status of stream or those who seek to change a goal to accommodate a new use to inform the process. Standards for the types of information that will be used to substantiate changes can be clearly articulated so changes are only made where sufficient information and public input has been received. Not "setting in stone" the protection goals allows for a process to ensure the best available information is able to be brought to bear on a regular basis to keep the benchmarks or goals up-to-date.

6.4 Environmental Flow Criteria

Environmental flow criteria are the flow conditions or degrees of hydrologic alteration that must be consistently met by water users in any particular river segment. Environmental flow criteria should be based on an understanding of how changes in hydrology are likely to impact natural resource conditions. These criteria, much like water quality criteria, include a definition of what is acceptable and what is unacceptable in terms of changes to hydrologic conditions. Section 5 of this report explored river-specific approaches to defining environmental flow requirements and then went on to review statewide approaches that could be used to set environmental criteria (i.e., hydrological and flow ecology-response approaches).

Hydrological methodologies, which are typically simple approaches based on maintaining an arbitrary degree of natural flow variability, have their value in ease of applicability and low cost. However, hydrological methodologies are considered by many scientists to lack scientific credibility, because there is no explicit connection between the degree of hydrologic alteration allowed and the associated natural resource consequences. Concern about the widespread development and application of hydrological methodologies is one of the factors that prompted a group of scientists to develop the Ecological Limits of Hydrologic Alteration (ELOHA) framework. Statewide hydrological methodologies also leave little room for a social process of evaluating risks to ecological resources of concern from hydrologic alteration and balancing those risks against the legitimate needs for water use by society as a whole. In other words, the degree to which you alter a particular flow parameter can only be examined relative to human use consequences since no explicit link between hydrologic alteration and ecological consequences has been made.

In contrast, flow alteration-ecological response relationships allow for examination of the approximate risk to ecological condition (expressed through a chosen indicator) associated with a change in hydrology. As mentioned in Section 5, flow alteration can be linked to a variety of ecological indicators using this approach, including habitat (e.g., PA/MD Instream Flow Study), population metrics (e.g., fish abundance), functional attributes (e.g., change in species guilds), and community metrics (e.g., macroinvertebrate species richness). Poff et al. (in review) include a table with considerations for selecting ecological indicators useful for developing flow alteration –ecological response relationships. No matter what indicator is chosen, the approach is designed to result in conceptual, semi-quantitative, or empirical flow-ecology relationships that can be used to set instream flow criteria for river types in a state or basin.

In *Case Study 4: Michigan Water Withdrawal Assessment Process*, the flow-ecology relationships that linked thriving and characteristic fish communities to low flow alteration provide an excellent example of the type of output sought for defining criteria (Figure 6.1). These smooth “flow-fish functional response curves” are based on empirical data and have eleven different variations, each associated with a different Michigan stream type. In this way, Michigan was able to generalize about the likely risks to ecological condition from increasing degrees of low flow hydrologic alteration. If these empirical relationships were instead semi-quantitative, they might be in the form of a curve with less scientific certainty or set risk thresholds which would specify different levels of ecological concern associated with degrees of hydrologic alteration (expressed through one or more hydrologic statistics).

Whatever the form of the scientific information about the consequences of flow alteration that is developed for rivers throughout a state or major basin (likely using river classification as a basis for generalization), it is clear that the process of setting environmental flow criteria will take place in a social context. This “social context” or “policy process” can take many forms, as shown by the water management examples and case studies reviewed in this report. But the unifying factor is that natural resource decision-making is only supported, not driven, by science. Stakeholders, and their associated values and biases, play the key role in decisions not just about the allocation of water, but about the range of costs and benefits that water has both instream and out of channel.

Environmental flow criteria are about defining thresholds associated with acceptable and unacceptable impacts. The question of how much alteration or degradation is acceptable is partly a scientific question, but mostly a societal decision. Again, *Case Study 4: Michigan Water Withdrawal Assessment Process,* provides a good model of how environmental flow criteria can be set in a transparent, science-informed process. The legislatively-appointed Michigan Groundwater Conservation Advisory Council (the Council) included stakeholders from all major water users in the state as well as environmental concerns (MGCAC 2007). This group used the flow-ecology response relationships developed by resource agency and university scientists to create four zones of ecological risk that could be directly linked to hydrologic alteration (Figure 6.1). The case study and cited documentation provide full details, but important to this discussion is that the Council made decisions that were informed by science but reflected social balancing of impacts. Specifically, there were the social decisions to limit ecological impacts to 10% of the initial population metric, to avoid the steeper portion of the curves where ecological risk was assumed to be higher, and to create a scheme in which a definition of unacceptable impacts was coupled with three zones of increasing user responsibility. The construction of these vertical lines of risk constitute environmental flow criteria that, by being based in a stakeholder decision-making process, reflected the concerns and desires of a consensus group of water users and environmental interests.

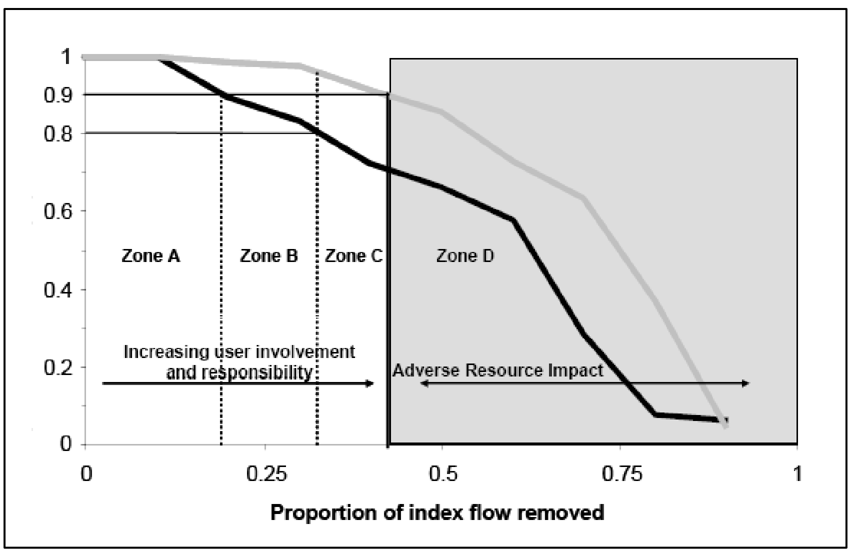


Figure 6.1 Environmental flow criteria from Michigan, defined based on percent changes from characteristic and thriving fish communities.

This example should not be seen as an isolated case. Stakeholder processes are often effectively used in natural resource decision-making, although they range in their group size, complexity and transparency. For purposes of setting environmental flow criteria, it appears that the earlier the participation of stakeholders in the process, the better. This includes keeping stakeholders informed of the progress of scientific investigations associated with criteria development, as was the case in Michigan. Similarly, in the work represented in *Case Study 6: Connecticut Draft Streamflow Protection Regulations*, key stakeholders appointed by the Connecticut Department of Environmental Protection had an opportunity to react to recommendations of the scientific and technical committee as they were developed, which amounted to a continuing stakeholder review of risk thresholds for hydrologic alteration.

The paper describing the ELOHA framework (Poff et al. in review) explains that the scientist's role at this stage of environmental flow criteria determination is "to support that decision-making process by accurately and usefully communicating the importance of ecosystem goods and services provided by streams, rivers, and wetlands and the ecological and societal consequences that will result from different levels of flow modification represented in the flow-ecology relationships." In other words, scientists play a critical role in assisting and informing decision-making that leads to "redistribution of the costs and benefits of water use within the management area." Since the setting of environmental flow criteria will inevitably be both complicated and controversial, it is appropriate to include scientists in a supporting role to provide necessary interpretation. It is clear that it is not the role of scientists to make decisions about how much risk is appropriate, but rather to provide as much specific information to decision-makers on the ecological risks of hydrologic alteration as possible.

As the Michigan and Connecticut case studies demonstrate, decisions about instream flow criteria can be made by combining the best available scientific information with the values as expressed by society. These values are usually embodied in the statutes and laws that establish programs which define the overall standard (e.g., no significant adverse impact),and which then must be informed by processes as described above. Whether completed through a less transparent internal agency process or a more transparent public stakeholder process, it should be recognized that the final decisions about environmental flow criteria are of enough consequence that they will always be affected by the social process of balancing perceived benefits and costs. Successful environmental flow criteria will likely be those that have a solid scientific foundation and a well structured policy-making process that has transparency and balancing at its core.

6.5 Hydrologic Measures

Hydrologic measures are used in two ways. The first is to define a set of measures that can be used to define how intact or altered a current stream hydrology is from a natural or minimally impacted condition. While programs such as IHA and HAT offer literally hundreds of possible statistics, for management purposes key indicator statistics are needed to consistently measure meaningful hydrologic changes. These should be linked to the flow criteria developed. For example, if environmental flow criteria are related to the extent and duration of floodplain inundation, one or more statistics related to flood flows will be important to measure for these systems. In general, these statistics:

- Be relatively simple;
- Be non-redundant;
- Be easily understood by water managers and the public;
- Account for the five major flow characteristic (magnitude, duration, frequency, rate of change, timing) and seasonal and inter-annual variability; and
- Be linked (at least conceptually) to ecological response.

Depending on the type of river and the type of withdrawal, particular flow components should be included, such as:

- Extreme low flows (drought flows, zero flow days);
- Seasonal baseflows;
- Small pulses (for water quality or migratory species);
- Channel maintenance flows; and
- Floodplain maintenance flows.

Determining hydrologic measures is often a combination of statistical analysis, such as principal component analysis (PCA) and expert input on what useful measures are given the type of stream systems, known ecological attributes, and known flow-ecology relationships.

6.6 Limits and Operating Rules

A key step to meeting environmental flow goals is to be able to transform the criteria and measures into terms that guide operations of structures and facilities by water users. This requires developing operating rules related to releases, bypass flows, and limits. Most water users think in terms of "how much water can I withdraw and when" or "how much water do I need to release from the reservoir and when." Such use limits are more intuitive and understandable for both water users, resource managers and the public.

By building protection goals that are explicit about the ecological conditions that are being protected and the allowable withdrawals that still meet these goals, the intent and desired actions are likely to be better understood by key water managers, water users, and stakeholders. In general, operating rules and limits should:

- Be seasonally appropriate;
- Account for extreme conditions (droughts and floods);
- Be linked to environmental flow criteria;
- Ensure essential uses (public health, fire, etc.) are met;
- Account for cumulative withdrawals and alterations; and
- Include a "share the pain" approach during periods of water shortages whereby both releases and demands are reduced to conserve both supply and to avoid zero release conditions.

Operating rules and limits may include features such as:

- Linkage to current conditions: e.g., increases in allowable withdrawals in wetter seasons;
- Inclusion of "Hands Off Flows": flows below which withdrawals are not permitted (must come from storage); and
- Protection of assimilation capacity (often 7Q10 flows).

For reservoir releases, operating rules and limits may:

- Be based on reservoir size (e.g., storage ratio: storage volume relative to annual runoff);
- Need to be linked to storage level and/or antecedent inflow; and

- Have impacts modeled over a long period of record, including severe drought conditions.

Operating rules may benefit from policies that:

- Include drought management plans that reduce demands based on instream flow or other hydrologic conditions;
- Minimize consumptive loss;
- Maximize return flows (wastewater, stormwater, etc.);
- Include trading programs among water users to maximize efficiency (water and economic) to meet criteria and goals to encourage investment in conservation, etc.; and
- Provide support for new water supply infrastructure that allows for more flexibility in operations and ability to meet environmental flow goals.

6.7 Anticipating Change and New Information

Since our ability to understand current conditions is always improving and since our ability to predict future conditions and needs is tenuous at best, it is important that resource managers be able to update protection goals, criteria, measures and operating rules on a regular basis. Any water management program should incorporate best available information as well as continued refinement and improvement. Such approaches are typical of many environmental programs, such as the state water quality standards which are, by statute, required to be reviewed on a triennial basis. The review should include the opportunity for public comment on both proposed changes and existing classifications and designated uses. This process allows for the continued use of the best existing information to set protection goals and to accommodate changing needs and demands.

6.8 Use of Water Allocation and Hydrologic Models to Inform Decision-making

To understand how a set of operating rules will affect the full range of natural variability, an important aspect of developing these rules and assessing compliance with the rules will be modeling how withdrawals and other uses or impacts affect instream flow over an extended period of time. In some ways the approach to understanding long-term changes under variable conditions and understanding cumulative impacts will be more similar to air quality modeling than the steady state modeling associated with water quality programs.

These approaches use long periods of record to examine how the criteria, in this case flow criteria, are met over varying conditions. The period or records are typically 20-40 years, but by extending streamflow records a synthetic period of hundreds or even 1000 years is possible. Such analysis is important for understanding how the criteria are met both during extreme conditions, such as droughts, and over less extreme conditions. By modeling existing and proposed water uses, the expected alterations can be compared to the baseline hydrologic condition and the resulting changes to the hydrograph can be understood over an extended period of time. This modeling allows decision-makers to determine in advance whether the existing or proposed alterations are within the allowable limits as defined by stream protection class. It also provides a level of confidence in future outcomes – if the system is operated as modeled, there is an understanding of the conditions that will result during various types of climate conditions, from floods to droughts. So despite our inability to do accurate long-range weather precipitation forecasting to gauge water availability, modeling can provide approximations of hydrologic conditions under a wide range of circumstances.

6.9 Recommendations for Using Environmental Flow Criteria in Decision-making

Comprehensive water management to protect ecological structure and functions of streams and other freshwater resources is, like many other environmental protection programs, based on a few key elements. These include identification of the desired ecological and human use goals, identifying specific criteria of ecological conditions that can be used to define acceptable and unacceptable limits to changes to these conditions, and defining measures of hydrologic change associated with these criteria and operating rules by which these criteria and measures can be achieved by water managers. Such goals, criteria, measures and operating rules can be used to inform a broad range of regulatory and planning programs within Pennsylvania.

For the protection of hydrologic conditions, this entails defining appropriate levels of protection for specific water bodies based on existing and desired designated and beneficial uses. It also entails providing a framework for assessing both existing conditions as well as assessing new uses and projects to determine their ability to meet these goals or standards.

As with any environmental program, establishing standards is necessary for achieving desired outcomes. But setting standards alone is not sufficient. Meeting Pennsylvania's water needs while protecting ecological systems and processes affects how we think about water infrastructure – that is, where we get our water, how we use it, where we return it. Meeting the requirements of public health, providing water to support a strong economy and ensuring a sustainable environment requires integrated approaches that look at storm water management, wastewater returns, road and bridge construction, and stream restoration and rehabilitation.

Clear environmental flow goals and criteria that lead to water use guidelines provide the foundation from which the many entities and individuals can guide their actions to achieve these predefined goals. Experience with other environmental programs, from water quality to air quality, demonstrates how standards and criteria are the foundation from which a broad array of actions can be guided and directed over a multi-decadal time scale.

Recommendation: Pennsylvania should undertake a process to define environmental flow goals and criteria that includes a broad range of stakeholders supported by scientists. The ELOHA framework provides a useful and structured approach to the key building blocks of such a program.

Public agencies, informed by stakeholders, need to work together to define ecological goals and the level of protection for environmental flows that are acceptable to meet those goals. It is inevitable that there will be scientific uncertainties and conflicting visions of the importance of ecological goals relative to human uses. Given this, a structured, inclusive approach should be employed to take flow alteration—ecological response relationships (whether empirical or semi-quantitative) and other available scientific information, and use this information to define ecologically and socially acceptable thresholds of hydrologic alteration. These instream flow criteria should be linked to a tiered set of goals which, in turn, can be applied to all rivers or river segments within Pennsylvania.

Scientists can play a role in such as process, but as with any policy decision, relevant public agencies informed by stakeholders will need to make the final decision. The Michigan Groundwater Conservation Advisory Council provides a good example of a stakeholder process, informed by science, that resulted in quantitative "adverse resource impact" thresholds applicable to rivers across the state.

This work can and should build upon the Pennsylvania Instream Flow Model (PA IFM) and its current implementation on many of the state's coldwater trout streams. The process can also build on other existing tools, including the WAST information. The PA IFM demonstrates the value of regionalizing information on flow alteration and ecological response as it is based on generalized changes to trout habitat associated with hydrologic alteration.

Recommendation: The Pennsylvania Instream Flow Technical Advisory Committee should move from an advisory committee to a program development committee. The Pennsylvania Instream Flow Technical Advisory Committee is well suited to design and lead a process to develop environmental flow criteria. The broad expertise and experience of its members provides the knowledge and insight to identify the most promising opportunities and develop solutions that are appropriate to Pennsylvania's future and build on the state's long history of working on environmental flow criteria and previous investments in water management tools. The committee should develop a charter outlining the goals of the effort and the responsibilities of the committee leadership and members. This charter would describe the goal and common commitment to develop a comprehensive, statewide approach to environmental flow protection, including the development of tools to build a hydrologic foundation for Pennsylvania. Given a set of shared goals, the committee would develop specific short-term and long-term tasks towards comprehensive environmental flow protection and work to maximize available resources to accomplish these tasks in a timely manner.

6.10 Conclusion

The Commonwealth of Pennsylvania is fortunate to have a wealth of rivers, significant aquatic biodiversity, popular game fish resources, and an engaged public interested in river protection and restoration. Pennsylvania also has a set of state and federal public agencies interested in environmental flow protection as demonstrated through their involvement in the Pennsylvania Instream Flow Technical Advisory Committee (TAC). The TAC is already very knowledgeable about the state-of-the-art tools and common issues associated comprehensive environmental flow protection. This common level of understanding and shared goals creates an important opportunity to move forward. This report provides the information to make informed decisions about which methods and approaches are most promising for Pennsylvania.

It has been noted by many TAC members that there are a range of existing state and basin commission authorities that can be more effectively employed to implement environmental flow protection in Pennsylvania. A strong, comprehensive and statewide approach to environmental flow protection would allow agencies to realize these opportunities and provide a framework for comprehensive water management decision-making that meets environmental and human water needs.

CASE STUDIES

CASE STUDY 1: A MODEL FOR WATER SUPPLY PLANNING IN VIRGINIA

Virginia Water Supply Planning regulations require the development of water supply plans throughout the Commonwealth, in order to assure adequate water supply for future off-stream uses, and the maintenance of instream resources including fish and wildlife. The Virginia Department of Environmental Quality (VA DEQ), Office of Water Supply and Planning administers these regulations. It is common, when the VA DEQ is trying to understand the impacts of a proposed withdrawal, to attempt to model the proposed withdrawal against the monitored flows in the stream in question during the drought of record. However, unless VA DEQ can know the extent of withdrawals upstream of the area of interest during the drought of record, they cannot accurately understand what impact additional upstream withdrawals may have during future droughts. Therefore, in order to gain the most accurate budget of available water for both instream and off-stream uses a hydrologic model is required. The ideal model needs to be capable of representing future withdrawals, discharges, and storage rules (i.e., a hydrologic foundation), as well as producing input for linked ecological models and recreational condition reviews (e.g., flow velocity and depth). The ideal platform/structure would build upon previously developed infrastructure, and also possess the flexibility to support new analysis tools and techniques as they arise. The underlying goal for such as model is to create a collaborative environment that enables rapid review and deployment of scientific tools, and facilitates the use of those tools by state, regional, and local water supply planners.

Model Selection and Construction

For many years, a basic Chesapeake Bay Program model using HSPF produced a contiguous hydrologic model over the portions of the state of Virginia that drain to the Chesapeake Bay. However, the resolution was coarse, and a significant portion of Virginia lies outside of the Chesapeake Bay watershed boundaries and was not covered. In recent years interest in a comprehensive statewide TMDL model rose in the state of Maryland's Department of the Environment (MDE), leading to a collaboration with the EPA's Chesapeake Bay Program to produce a new version of the Chesapeake Bay Watershed Model. This new model possesses a greatly enhanced model resolution, and representation of surface and instream processes. The Virginia Department of Conservation and Recreation (VA DCR) also saw the benefit of having a statewide hydrologic model and entered into this collaborative process with the Chesapeake Bay Program and the Virginia Water Science Center of the United States Geological Survey (USGS). VADCR provided $557,000 (including $100,000 from EPA out of their bay restoration grant) and the Virginia USGS put forth $349,000 in order to achieve the goal of extending the model to the non-Bay watershed portion of the state of Virginia. This multi-state, multi-agency collaborative effort produced what is known as the Chesapeake Bay Programs Phase 5 (CBPP5) hydrology model. In 2007, the Office of Water Supply Planning at the Virginia Department of Environmental Quality (VADEQ) began to investigate the use of this model as a major component in its water supply planning modeling toolbox.

Model Description

The CBPP5 model is a detailed hydrologic model that runs on an hourly time step (to 15 minutes in some areas), and has been calibrated and validated over a 21 year time period. Without modification or recalibration the model can be run from 1984 to 2005, using flows from over 140 continuous flow gages, and modeling over 600 stream reaches in the state of Virginia alone. With additional effort, the model can add additional nodes of interest, likely simulating basins sizes down to 10 square miles in area. The HSPF model is highly modular and programmable, and these features have been exploited in the CBPP5 implementation to represent reservoir operation rules, time varying water withdrawals, discharges, and land use evolution. This modular capability permitted the inclusion of the VA DEQ's Virginia Water Use Database (VWUDS), which has compiled reported water withdrawals since 1982, and point source discharge data. The USGS programmed the stage-discharge relationships of modeled impoundments to correspond to the actual operational rules in order to match hydrologic flows for a successful model calibration. This also has the dual benefit of matching the hydrologic alteration of the system to producing a realistic set of current flow conditions for water supply planning and instream flow analysis, as well as validating the upland portions of the model (rainfall, runoff, interflow, and groundwater). While the programming of the reservoir rules in the CBPP5 model has proven sufficient for modeling the variation in daily discharge over the period of record caused by dam operation, the HSPF model is not built to replicate the full suite of reservoir operation rules, inter-basin transfers, and outlet types. In order to facilitate a more robust reservoir operation simulation, the VADEQ has constructed a separate flow routing model to interface with the surface runoff portion of the CBPP5 model. This flow routing model is integrated with the decision support system, and possesses robust and flexible withdrawal rules to be used in conjunction with the rainfall, runoff, and potentially a separate groundwater model. The integration of this separate flow routing model further highlights the utility of the HSPF model, as its capabilities to export all manner of simulated quantities, and to likewise receive inputs from external sources makes it an ideal candidate for an integrated water supply planning and instream flow assessment model.

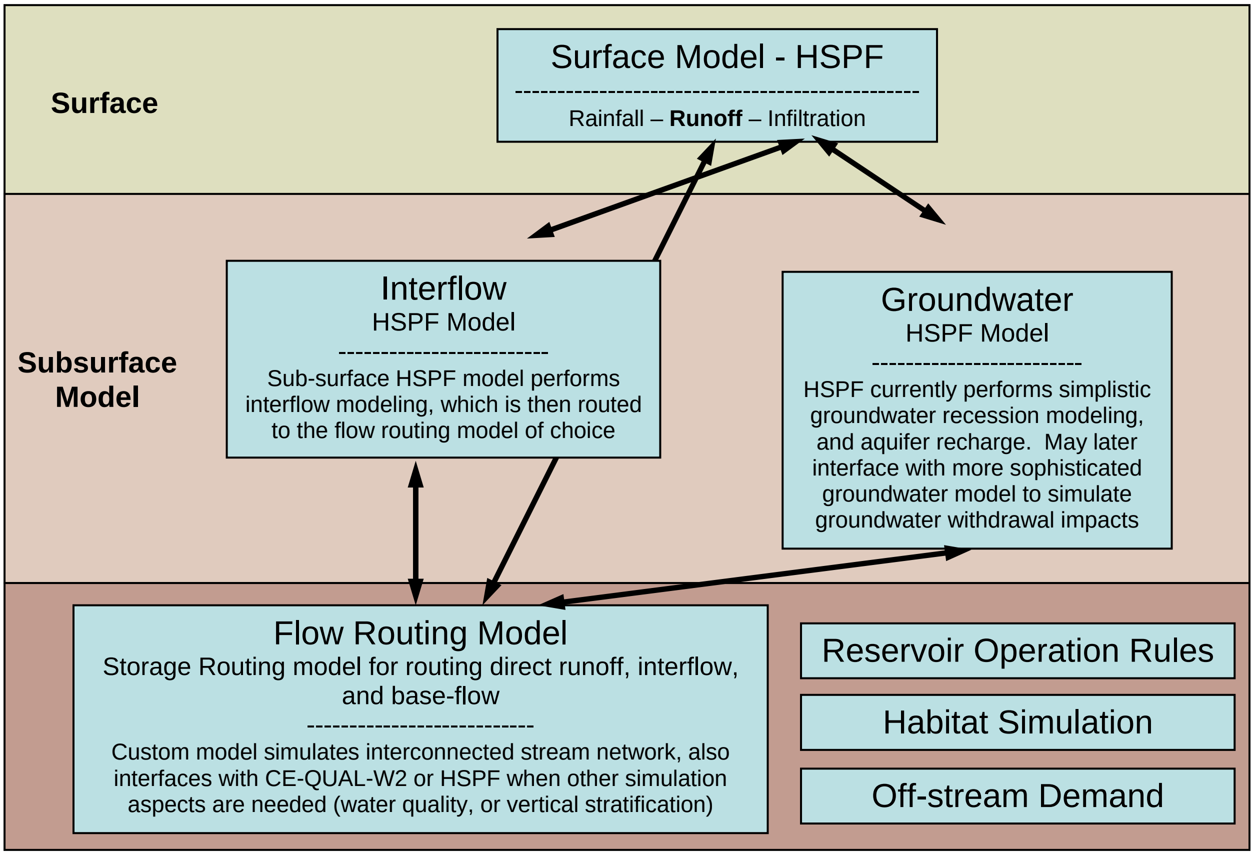


Figure 1. Interaction of existing and future models as part of the Virginia Water Supply Planning decision support system

Model Accuracy, Data Accuracy and Automated Calibration
The basic model has a reasonable level of accuracy for total water budget: 82.9% of the 141 simulated streams were within 15% of total annual measured flows for the 16 years calibration/validation period, and 92.2% were within 20%. Of the largest 50% of basins, the results show 90.1% of modeled flows within 15%, and 97.2% within 20% of the observed. Reasons for hydrologic model error can be the result of simplifications in the modeled processes, errors in input data, and errors in the observed values of flow gages when estimating flow rates outside of the gages observed record. Sources of error in the CPP5 model may be the result of errors in the water use records (VWUDS), which suffered from gaps early in the record, precipitation variations, and failure of the assumption that groundwater boundaries conform to surface water boundaries (both of which become increasingly important in smaller basins).

The VWUDS data is reported to DEQ by withdrawal permit holders and non-permitted entities that report on a voluntary basis. While this data is reported to, and by definition not estimated by DEQ, there may still be considerable estimation in the process of reporting by any individual entity. Additionally, data reporting in the early period of the VWUDS program (which began in 1982) was much less complete than in recent years. In general, as the size of the individual water using entity increases, the accuracy of the withdrawal information increases. Therefore, in areas with a substantial portion of water use is accounted for by large permitted withdrawals, the

VWUDS information has been shown to produce a reliable estimate of the usage portion of the water cycle. Groundwater withdrawals are not explicitly modeled in the current watershed model, therefore, the influence of groundwater withdrawals are not explicitly accounted for, however, the modular framework may be adapted to include the use of more robust groundwater models as they becomes available.

Precipitation data is obtained from the NOAA CO-OP system of rainfall gages. This data is then processed into a gridded form in order to mimic regional precipitation patterns more accurately, but is still essentially, a coarse precipitation input map. Recent advances in precipitation observation by the NOAA have resulted in a radar-based 4 km by 4 km square grid precipitation product. This data will be integrated into future calibrations of this model by VADEQ in order to increase the accuracy of the model in smaller basins.

Perhaps one of the most useful aspects of the CBPP5 model is its automated calibration mechanism. This enables unsupervised recalibration of the model when input data improves (such as with the addition of high resolution precipitation data), when new calibration data comes on line (e.g., new stream gages), or when the model is to be used for a different objective. The calibration system is goal-based, in that the user specifies the "goal criteria" for the model. This will allow water supply modelers to recalibrate the model with total flow as a primary criteria, in order to minimize errors in the water budget, or to emphasize flow durations, or base flow recession curves, to provide the greatest level of accuracy for use in instream flow evaluations.

Using the Model to Perform Cumulative Impact Analysis

In practice, this model will form the basis of a modeling system that can be tailored for use in Virginia for cumulative water withdrawal and instream flow analyses. In many areas, where the withdrawal record is detailed and continuous, the model can be used without modification, to provide current flow conditions (with withdrawals, discharges, and impoundments included in the simulation). Through the use of the upland components of the model (for rainfall, runoff, interflow and groundwater) in conjunction with a flow routing model, it can also be used to represent an "unaltered condition", or "minimally altered condition" by removing simulated impoundments, withdrawals, and discharges, as well as modifying land uses to simulate pre-development surface hydrology. This unaltered state will be validated in areas with long historical flow monitoring data. Once an un-altered state has been validated, the model will be used to simulate scenarios for future water withdrawals, reservoir operation rules, pipelines, side-stream storage versus on-stream storage, and regionally integrated reservoir management schemas.

Lessons Learned, Modeling Requirements, and Coping With Model Error

Some of the most difficult issues for VA DEQ so far are estimating consumptive use, probing the limits of the hydrologic model resolution, and understanding the role that model error plays in informed decision making. Additionally, the model itself requires a fairly robust computer to run the hydrologic simulation in a timely fashion.

In the existing CBP model hydrology, it was assumed that the consumptive use water budget could be described entirely by the balance of reported withdrawals and discharges. That is, the total reported water withdrawn, minus the total reported water discharged in a given watershed equaled the consumptive use. While this is a reasonable approach for a historical water quality model, it is not a workable assumption in a future-casting water supply model. The issue of return flows from projected withdrawals will require a good deal of attention.

The model resolution in HSPF is inherently flexible in that it simulates a "representative acre", and then multiplies the outflow of that acre by the area of the watershed in question. This allows, at least in theory, the simulation of virtually any size sub-watershed, using the representative acre in the region containing that watershed. In practice, however, the validity of a representative acre decreases as the modeled area gets smaller, and consequently errors increase.

Any model will have errors and uncertainties. Error handling will be the largest challenge in the use of these models. Generally, errors in model simulation lead to distrust amongst the consumers of these models outputs. However, proper understanding of errors can allow models to remain useful within their valid ranges. It is our intention to develop a means of quantifying the margins of safety in our interpretation of model results based on the model errors themselves.

The original modeling system could perform a hydrologic calibration in approximately 24 hours on a multi-processor system featuring 8 x 2.0 GHz CPUs. Hydrologic simulations themselves can be performed in under 3 hours, however, the additional overhead required by reservoir management rules have been shown to increase processing by a factor of 5-10 in preliminary runs. For a state wide model, this results in a scenario run time of around 1 day, however, most runs will be done on smaller basins. For a small basin, around 1,000,000 acres, a 10 year scenario completes in under 20 minutes, including HSPF run and reservoir rule simulation.

CASE STUDY 2: STREAM CLASSIFICATION IN PENNSYLVANIA USING HYDROECOLOGICAL INTEGRITY ASSESSMENT PROCESS (HIP)

We contracted with USGS-Fort Collins to develop and apply a stream classification using HIP. The objectives of this pilot study were to a) develop a hydrologic classification of Pennsylvania streams and b) develop a method to transfer this hydrologic classification statewide (i.e., to ungaged sites and other, impacted, gages not used to develop the classification).

We worked with USGS to identify a set of index gages to use to compute the hydrological indices and develop the classification. In Pennsylvania, USGS had identified a set of 195 "index gages" or continuous record station gages that can be considered least impaired due to limited upstream regulation, diversion, and mining impacts (Stuckey 2006). We reduced this list further using additional criteria related to land cover (<15% urban in catchment) and period of record (>15 years). The revised list of 136 gages was used to calculate the 171 hydrological indices used by HIP and the 34 Environmental Flow Components (EFC) indices. Appendix 2 is a report written by USGS Fort Collins that includes a detailed explanation of the procedures used to develop three alternative classifications and a table showing the class(es) assigned to each of the index gages. Methods and results are briefly summarized below.

Based on the results of a principal components analysis, 151 of the original 205 (171 HIP + 34 EFC flow indices) with the strongest component loadings were used in three different clustering procedures which are further described in Appendix 2:

1. *Simultaneous classification:* All 151 indices were used simultaneously in the k-means clustering procedure. This approach resulted in five classes.
2. *2-stage classification:* 71 of the 151 indices that were strongly correlated with drainage area were used as a first stage classification. This resulted in three classes: high flow streams with large drainage area (n=4), moderate flow streams with moderate drainage area (n=32), and low flow streams with small drainage area (n=100). At the second stage, the remaining 80 of 151 indices that were uncorrelated with drainage area were used to cluster streams within the moderate and low flow stream classes. The moderate flow stream class was further subdivided into two classes and the low flow class was further subdivided into three classes, resulting in six classes total.
3. *Classification without indices correlated with drainage area:* The 71 indices that were highly correlated with drainage area were eliminated and only the 80 indices that were relatively uncorrelated with drainage area were included in the clustering procedure. Several different numbers of groups were considered before deciding on five stream classes.

USGS produced the second and third clustering procedures in an attempt to address concerns that the simultaneous classification was strongly influenced by indices that were highly correlated with drainage area and that differences in drainage area may be accounting for most of the variation between classes.

Of the three classifications, the first classification is the most straightforward and easily interpretable. This procedure resulted in five classes based on a eleven of the original 151 flow indices (Table 1). Flow classes are distinguished by differences in flow magnitude, flow variation, and flood frequency (Figure 1). Figure 2 is a map of the flow classes assigned to the index gages used to develop the classification.

Table 1. Flow indices used to define five hydrologic types within Pennsylvania using simultaneous classification procedure.

Flow Component	Flow Indices (code)
Flow magnitude	Mean of daily mean flow (MA1) Median of daily mean flow (MA2) Mean of March daily flow (MA14) Mean daily high flow in June (MH6) Mean daily low flow in December (ML12)
Flow variation	Mean of the annual CV of daily flows (MA3) Skewness of daily flows, mean/median (MA5) Q10 / Q90 of daily flow (MA6) Q20 / Q80 of daily flow (MA7) Ratio of difference in Q90 and Q10 to log Q50 (MA10)
Flood frequency	Avg number of flow events >3X median daily flow (FH6)

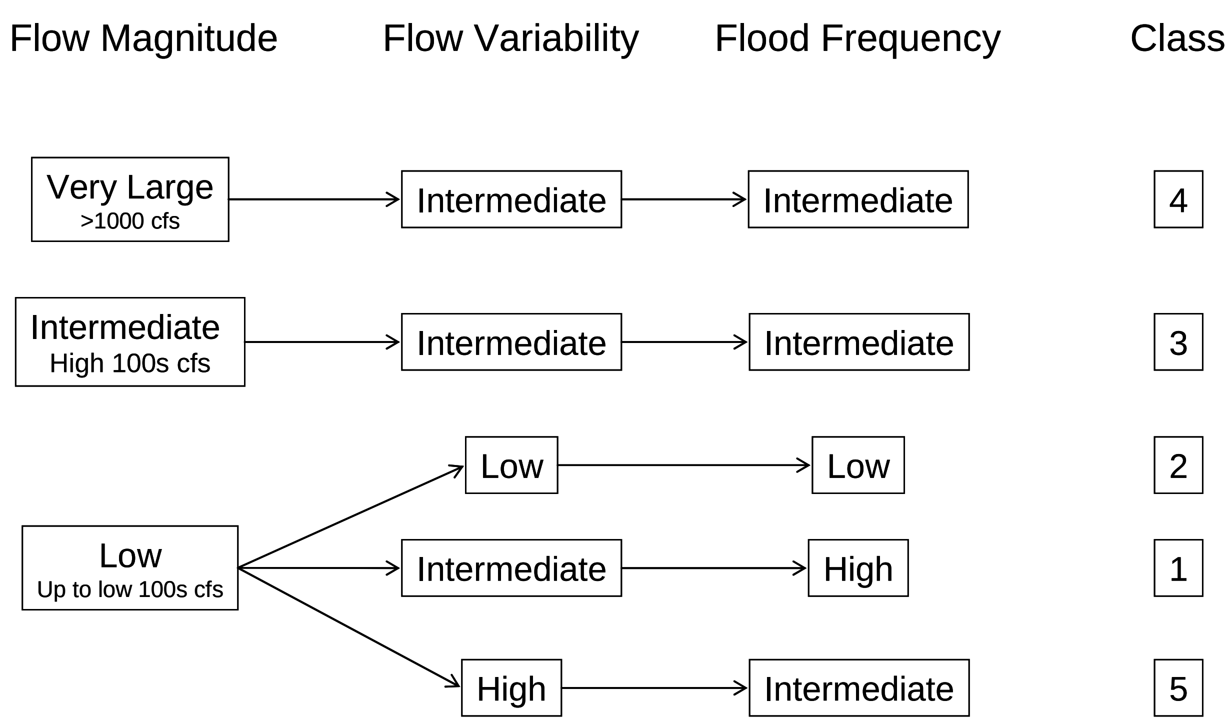


Figure 1. Hydrologic characteristics of the five flow classes resulting from simultaneous classification.

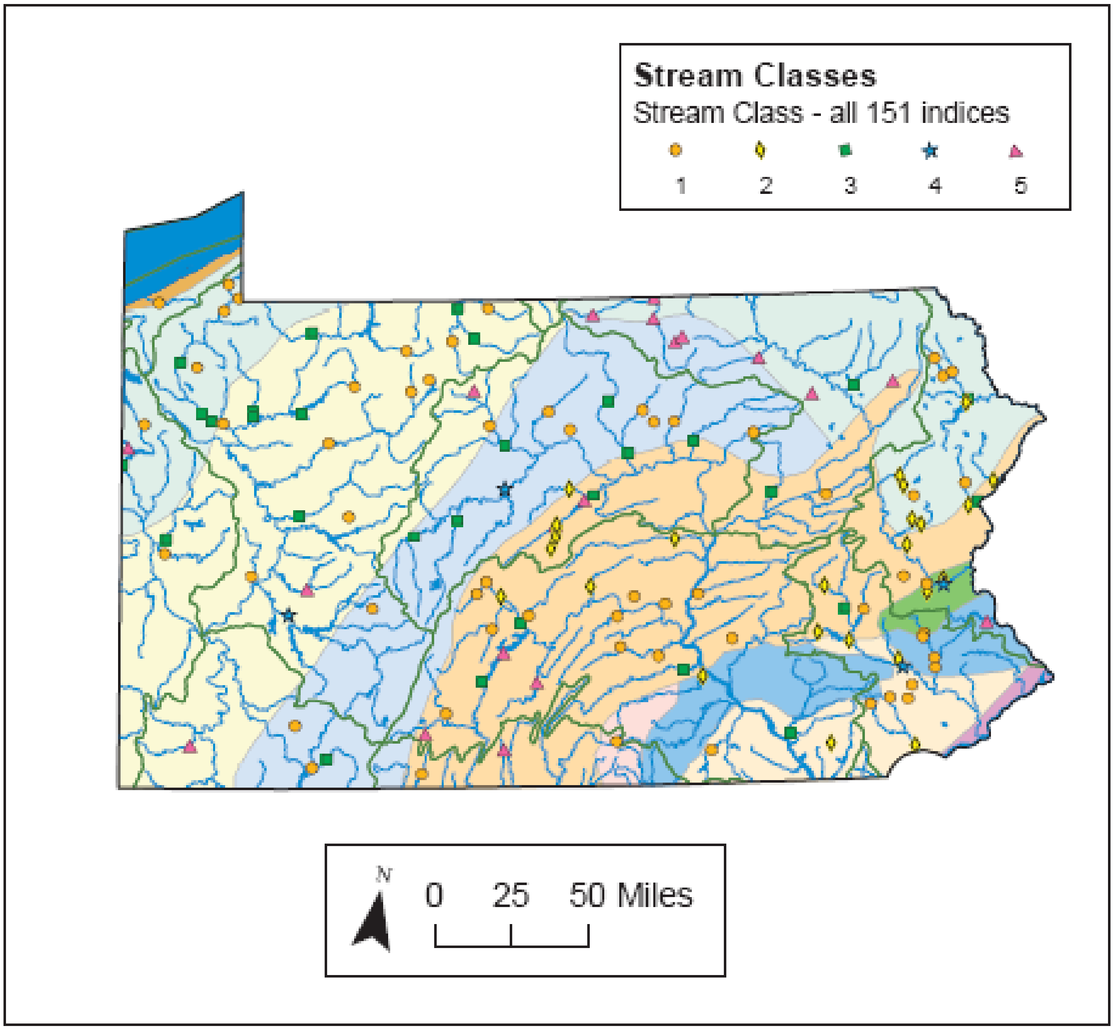


Figure 2. Geographic locations of the 5 stream classes in Pennsylvania resulting from simultaneous classification approach. Colored bands represent physiographic sections.

These five classes can be described in reference to the national classifications by Poff (1996) and Olden and Poff (2003). Class 2 streams appear to be stable groundwater as indicated by their relatively low overall flow volumes, low variability of daily flows, and low flood frequency. Class 4 streams are large volume, perennial runoff streams; class 3 streams are moderate volume perennial runoff streams. Class 1 streams are low volume, perennial runoff streams. These streams have less variable daily flows and greater flood frequency than class 5 streams, which are low volume, perennial flashy-runoff streams with high variability in daily flows.

Using discriminant function analysis to assign stream classes to ungaged streams

The ultimate goal for creating stream classes for the state of Pennsylvania is to assign stream classes to streams that were not included in the original classification procedure. However, these sites are mostly ungaged, and even sites with stream gages do not have a long period of unaltered flow data that would be needed to assign a stream class based on hydrology alone. Therefore, to accomplish this task, we first need to develop a method of predicting the stream classes of the index gages used in the classification procedure, using data that would be available at non-gaged locations. We can later use this information to predict the stream classifications of non-gaged sites.

We used a discriminant function analysis (PROC DISCRIM; SAS Institute Inc. 1999) to predict stream classes at index gage sites included in the stream classification procedure (described above) using GIS-based variables provided by USGS in Pennsylvania (Table 2). A correlation procedure (PROC CORR; SAS Institute Inc. 1999) was conducted to determine highly correlated pairs of variables in this set, and we eliminated one variable from each highly correlated pair (correlation coefficient > 0.75). Land use variables were also eliminated from the analysis because stream classes should be assigned according to unaltered flow conditions rather then hydrology that may have been altered by changes in watershed land use. Table 2 indicates the final set of variables used in the discriminant function analysis. We determined the correct classification rate of assigning index gages to stream classes based on our discriminant function model by using a cross-validation procedure. In this procedure, each observation was systematically removed from the analysis and all other observations were used to predict its stream class. The resulting correct classification rates were the proportion of observations that were correctly classified using this method. In addition, we used a stepwise discriminant analysis (PROC STEPDISC; SAS Institute Inc. 1999) to determine a subset of variables that provided significant (α=0.10) contributions to the discriminatory power of the model (i.e., were useful in discriminating among stream classes).

Overall, the variables we used allowed us to predict stream classes of index gages with a 78% correct classification rate. The correct classification rate varied by stream class, with the model illustrating the best predictive ability for stream class 4 (100%) and the worst for stream class 5 (63%; Table 3). Examination of the number of gages that were correctly and incorrectly classified within each stream class suggested that the model had the highest error rates when attempting to distinguish between stream classes 1 and 5; 20% of gages in stream class 1 were incorrectly classified as stream class 5 and 37% of gages in stream class 5 were incorrectly classified as class 1 (Table 3). The overlap between these two stream classes was not surprising, considering that stream classes 1 and 5 also exhibited considerable overlap in the stream classification analysis performed by USGS (described above). Because these stream classes were similar in terms of hydrologic data and only distinguished by relatively small differences in flow variability, it makes sense that it was difficult to distinguish among the two classes using landscape-level GIS variables. Drainage area was the variable that explained the most variation among all stream classes, followed by percent carbonate, precipitation, channel slope, groundwater head, percent wetlands, depth to bedrock, permeability, and longitude (Table 4).

We can assume that we would be able to predict stream classes of gaged and non-gaged sites throughout Pennsylvania with a correct classification rate approaching 78%, taking into consideration that the correct classification rate determined by our analysis may be optimistic when applying the discriminant function model to a new set of sites. In addition, the discriminant function model should only be applied to sites comparable to index gages in terms of the range of the variables used in the analysis (e.g., within the range of drainage area, % carbonate, precipitation, etc. exhibited by the index gages). We can apply the discriminant function model we developed in this analysis to predict the stream classes of other sites for which we have data for the GIS variables important in the model. We may be able to improve the overall predictive ability of the model if more quantitative predictive variables were available, particularly geology or geomorphology, although we are satisfied with the correct classification rate of the current model and would not expect additional variables to produce correct classification rates approaching 100%. We could increase the correct classification rate by combining stream classes 1 and 5; however, this would decrease the amount of information on differences in hydrologic character provided by the stream classifications.

Table 2. Watershed variables (many calculated using GIS) provided by USGS for inclusion in a discriminant function analysis to predict stream classes assigned to index gages without using stream gage data. Variables removed from final analysis were found to be highly correlated (correlation coefficient > 0.75) with other variables. '*' denotes land use variables that were removed from the analysis because stream classes are intended to represent least altered conditions.

Variable	Final analysis	Highly correlated variable(s)
Drainage area	Included	All predicted flow statistics, longest drainage path
Stream density	Included	None
Groundwater head	Included	None
Mean elevation	Included	None
Shape factor	Included	None
Basin slope	Included	None
Channel slope	Included	None
Longest drainage path	Removed	Drainage area, all predicted flow statistics
Precipitation	Included	None
% dominant rock type	Included	None
% carbonate	Included	Sinkhole density
% glaciated area	Included	Drainage run-off curve
Sinkhole density	Removed	% carbonate
% lakes	Included	None
Depth to bedrock	Included	None
Drainage run-off curve	Removed	% glaciated area
Infiltration index	Included	None
Available water content	Included	None
Permeability	Included	None
% urban	Removed*	None
% forest	Removed*	None
% residential	Removed*	None
% mining	Removed*	None
% commercial, industrial	Removed*	None
% wetlands	Included	None

Predicted 7 day, 10 year low flow	Removed	Drainage area, all predicted flow statistics
Predicted 7 day, 2 year low flow	Removed	Drainage area, all predicted flow statistics
Predicted 30 day, 10 year low flow	Removed	Drainage area, all predicted flow statistics
Predicted 30 day, 2 year low flow	Removed	Drainage area, all predicted flow statistics
Predicted 90 day, 10 year low flow	Removed	Drainage area, all predicted flow statistics
Predicted 10 year base flow	Removed	Drainage area, all predicted flow statistics
Predicted 25 year base flow	Removed	Drainage area, all predicted flow statistics
Predicted 50 year base flow	Removed	Drainage area, all predicted flow statistics
Predicted harmonic mean flow	Removed	Drainage area, all predicted flow statistics
Predicted mean annual flow	Removed	Drainage area, all predicted flow statistics

Table 3. The number of index gages that were predicted to occur in each stream class, and correct classification rate of index gages, by actual stream class.

Actual stream class	Predicted stream class					Correct classification rate
	1	2	3	4	5	
1	44	3	0	0	12	75%
2	6	18	1	0	0	72%
3	2	3	23	1	0	79%
4	0	0	0	4	0	100%
5	7	0	0	0	12	63%

Table 4. Variables that provided significant (α=0.10) contributions to the discriminatory power of the model (i.e., were useful in discriminating among stream classes), squared partial correlation (partial R^2, which indicates the relative rank of each variable in its ability to discriminate among stream classes), and significance value of each variable in the discriminant analysis.

Variable	Partial R^2	F-value	p-value
Drainage area	0.80	133.08	<0.01
% Carbonate	0.29	13.08	<0.01
Precipitation	0.34	16.46	<0.01
Channel slope	0.14	5.20	<0.01
Groundwater head	0.14	5.00	<0.01
% wetlands	0.11	4.05	<0.01
Depth to bedrock	0.09	3.19	0.02
Permeability	0.08	2.59	0.04
Longitude	0.07	2.38	0.06

Choosing non-redundant hydrologic indices

After the classification produced the 5 initial stream classes, USGS-Fort Collins performed a Principle Components Analysis (PCA) on each of the 5 classes of stream gauges and all classes combined (6 analyses) to identify indices that best explained variation in the 11 sub-components of the flow regime for the 171 HIP indices and the 34 TNC environmental flow components (EFCs) within stream classes. USGS retained up to the first 5 principal components that explained the majority of the variance in the indices and examined scree plots to determine whether a reduced number of components could be considered. Indices with the largest absolute loadings on the first 4 principal components for indices in each of the 11 sub-components were selected to explain the dominant pattern of hydrologic variation. Table 5 includes the primary and secondary hydrologic indices selected. This provided a reduced set of indices that were related to major components of variation in hydrology within a stream class and that were relatively uncorrelated with each other. The indices selected from the principal components for stream class 4 are not very reliable as the sample size for this class was only $n = 4$.

Table 5. Primary and secondary flow components with highest absolute loadings on first several principal components explaining majority of variance within 5 stream groups and within all streams ($n = 136$) in Pennsylvania. Principal components (PC) analysis was made with the correlation matrix. USGS used the first 4 PC for Group 1 (75% of variance); first 4 PC for Group 2 (80% of variance); first 4 PC for Group 3 (75% of variance); first 3 PC for Group 4 (100% of variance); first 4 PC for Group 5 (75% of variance); and first 4 PC for all streams (77% of variance). All indices are defined in Appendix 3.

Flow Component	Class 1 $n = 59$	Class 2 $n = 25$	Class 3 $n = 29$	Class 4 $n = 4$	Class 5 $n = 19$	All streams
MA	1, 6	14, 33	7, 19	27, 32	1, 4	2, 4
ML	1, 17	3, 19	22, 10	7, 21	3, 17	1, 13
MH	4, 16	4, 24	3, 14	15, 9	4, 27	7, 16
FL	3, 1	2, 1	1, 3	3, 1	3, 1	3, 1
FH	3, 6	3, 5	3, 7	2, 1	11, 4	7, 3
DL	5, 15	5, 1	14, 5	15, 6	5, 15	5, 14
DH	3, 12	2, 14	19, 5	12, 20	3, 12	5, 13
TA	2, 1	3, 1	1, 2	1, 2	1, 2	2, 3
TL	2, 1	2, 1	1, 2	3, 1	3, 1	1, 2
TH	2, 1	2	2, 1	2, 1	2, 1	3, 2
RA	1, 7	1, 6	3, 6	8, 5	3, 7	1, 7
EFC_ML	6,13	7, 13	7, 11	6, 3	8, 13	10, 6
EFC_MH	17, 29	17, 29	17, 23	23, 29	17, 23	17, 23
EFC_DL	14	14	14	14	14	14
EFC_DH	18, 24	24, 30	18, 24	18, 24	24, 25	24, 18
EFC_FL	16	16	16	16	16	16
EFC_FH	20, 26	26, 20	26, 20		20, 26	20
EFC_TL	15	15	15	15	15	15
EFC_TH	31, 19	31, 19	19, 31	31, 19	31, 19	31, 19
EFC_RA	21, 33	21, 33	28, 21	33, 27	21, 28	21, 28

To illustrate the pilot PA results of defining key hydrologic variables through the HIP statistical approach, we can look at table results for PA Class 2, which generally includes streams with low flow magnitude, low flow variability, and low flood frequency. Key flow statistics that explain the majority of the variance for this stream group include “ML3,” representing low magnitude, which is the mean of “minimum March flows” across the period of record. This also appears for stream Class 5 as a key statistic. “FL2,” representing the low frequency sub-component, is the “variability in low flow pulse count” and is only a key statistic for Class 2. In this way, a set of key flow variables can be described and used corresponding to each river class, or across all river classes analyzed in Pennsylvania.

USGS noted that the environmental flow components (EFCs) never loaded as strongly on any of the principal components as the hydrological indices from HIP. The EFCs also were not as strongly associated with differences among the 5 stream classes as the other hydrological indices. However, this does not imply that there were no statistical differences in EFC among the stream classes. Many of the EFCs were strongly correlated and similar in definition to the hydrological indices.

CASE STUDY 3: ENVIRONMENTAL FLOW STANDARDS TO MEET THE EU WATER FRAMEWORK DIRECTIVE

The European Union (EU) Water Framework Directive, passed in 2000, was designed to protect and restore the structure of aquatic ecosystems by setting ecological objectives. By design, the sustainable use of aquatic resources across the European Union will be judged based upon achieving these ecological goals. The "river basin management planning system" is introduced by the Water Framework Directive as an implementation structure for integrated management. The planning system is a decision-making framework where economic and social concerns can be brought into the process of defining environmental objectives and the approach to achieve those objectives. (http://www.wfduk.org/, accessed May 12, 2008)

The Water Framework Directive requires EU member states to achieve "Good Ecological Status" (GES) in all surface and ground waters that are not determined to be "heavily modified". Good Ecological Status is considered a slight deviation of reference ecological conditions and is supposed to be achieved based on environmental standards and conditions in river basins throughout Europe (Acreman et al. 2006). Heavily modified water bodies are instead required to achieve "Good Ecological Potential" (GEP), a lower standard. In the United Kingdom, the process of setting environmental standards and conditions under the Water Framework Directive is well under way. A UK Technical Advisory Group (UKTAG) has worked with conservation agencies and academics across the country to begin setting environmental standards (i.e., required thresholds) of physiochemical and hydromorphological conditions necessary to meet different levels of ecological status.

This case study will focus on environmental standards that have been developed that relate to hydrologic alteration of surface water bodies through abstraction (direct withdrawal) and impoundment management. These standards are the result of a review of existing science and standards by leading experts from the UK in the fields of aquatic ecology, hydrology, and geomorphology, and were recently released for public review (Acreman et al. 2006; Acreman 2007). The 2006 document, "Development of Environmental Standards (Water Resources)" has four key features with regard to river management: 1) an eight class river typology based on physical watershed characteristics; 2) an approach to assigning rivers in the UK into one of the eight classes; 3) detailed expert-based standards for allowable abstraction percentage based on knowledge of macrophyte, macroinvertebrate, and fish needs; and 4) recommended standards for abstraction in relation to annual flow statistics based on river type. All four features can be instructive for application of similar methods in Pennsylvania, and link to Sections 3-6 of this report.

The UK river typology is a modification of a macrophyte community-based classification documented in Holmes et al. (1998) which uses drainage area, gradient, altitude, river substrate, and nutrient-status to delineate between the eight types. This typology was seen as adequate by macrophyte and macroinvertebrate experts, but was considered imperfect by fish experts who offered an alternative typology. In a manner analogous to the pilot study in Pennsylvania, a detailed statistical approach was used to attempt to assign 733 watershed locations into each of the eight pre-determined classes based on physical characteristics of each of the locations available in GIS. This approach was more successful for certain types than for others, reinforcing the result described in *Case Study 2: Stream Classification in Pennsylvania using Hydroecological Integrity Assessment Process (HIP)*, and the overall conclusion of the authors was that their eight river class typology was appropriate for purposes of environmental flow standard-setting. In a final workshop, the tested typology was modified by the fish river typology resulting in 10 river types, one of which is salmonid spawning and nursery areas, which could not be defined by watershed characteristics alone.

The environmental standards were defined based on an expert consensus workshop approach using the precautionary principle to deal with considerable uncertainty in the best available scientific knowledge. Specific recommendations were made for macrophytes, macroinvertebrates, and fish which are best summarized in the document. Generally, however, the expert workgroup set permissible abstraction levels at approximately 20% of natural flows. The workgroup also generally agreed upon a Q95 (i.e., 5th percentile) flow as being "hands-off", meaning that at that flow withdrawal would either stop or be significantly reduced. Finally, the workgroup recognized that impoundments would not be able to meet the above standards, and that active flow management would be required below impoundments. These recommendations included ensuring floods would occur at some reduced level of frequency and that a seasonal "compensation flow" (i.e., passby flow release) would be required.

In the development of final abstraction standards for all river classes, some key principles were adopted, including: 1) standards should vary by river type, 2) the most stringent standard based on taxonomic group (e.g., macrophytes) should be the one applied, 3) standards should vary seasonally, 4) a risk-based approach should be used that can be modified by monitoring results, and 5) the only constant volume withdrawal applicable at all flows is the equivalent of 25% of annual, unregulated Q99. The documented quantitative standards are best represented by the table below, which includes four groups of river types, two applicable seasons, and four tiers of withdrawal standards based on annual flow characteristics. The % allowable abstraction values below are intended to be cumulative and applicable to any point on a river of that type.

Table 23 Recommended standards for UK river types for achieving GES given as % allowable abstraction of natural flow (thresholds are for annual flow statistics)

Type	Season	flow > Qn_{60}	Flow > Qn_{70}	flow > Qn_{95}	flow < Qn_{95}
A1	Apr – Oct	30	25	20	15
	Nov – Mar	35	30	25	20
A2 (ds), B1, B2, C1, D1	Apr – Oct	25	20	15	10
	Nov – Mar	30	25	20	15
A2 (hw), C2, D2	Apr – Oct	20	15	10	7.5
	Nov – Mar	25	20	15	10
Salmonid spawning & nursery areas (not Chalk rivers)	Jun – Sep	25	20	15	10
	Oct – May	20	15	flow > Q_{80} 10	flow < Q_{80} 7.5

The second relevant document from the United Kingdom, "Guidance on Environmental Flow Releases from Impoundments to Implement the Water Framework Directive" (Acreman 2007), is divided into three major parts. The first and most inclusive section defines a 14 step process for setting flow releases for impoundments so that a water body can meet Good Ecological Status (GES) or Good Ecological Potential (GEP). The second part provides an approach to assessing hydrologic alteration and determining whether this degree of alteration is acceptable. The final part of the document provides a three level, risk-based approach to defining an acceptable environmental flow release regime. The document also features useful "principles of environmental flow release regimes from impoundments".

The 14 step process is somewhat specific to implementation under the Water Framework Directive, but nonetheless provides a useful model for implementation of reservoir release regimes given infrastructure limitations (e.g., inadequate valves), scientific uncertainty, and site specific economic and environmental factors. It also has some clear steps relating to defining ecosystem status in relation to goals that is applicable elsewhere. The second section of the document is analogous to this report's section "Selecting Hydrologic Statistics and Assessing Hydrologic Alteration" and, similarly, it uses the Indicators of Hydrologic Alteration (IHA) statistics as a starting point for discussion. The UK guidance suggests modeling statistics of impounded and unimpounded flow regimes and using a subset of flow statistics to determine whether these flow regimes are different in a biologically significant way. Work in the UK is assisted by the availability of a flow simulation program, "Low Flows 2000" (Young et al. 2003), which allows simulation of natural flow regime statistics as well as impounded flow statistics, thereby providing a basic "hydrologic foundation" for the country.

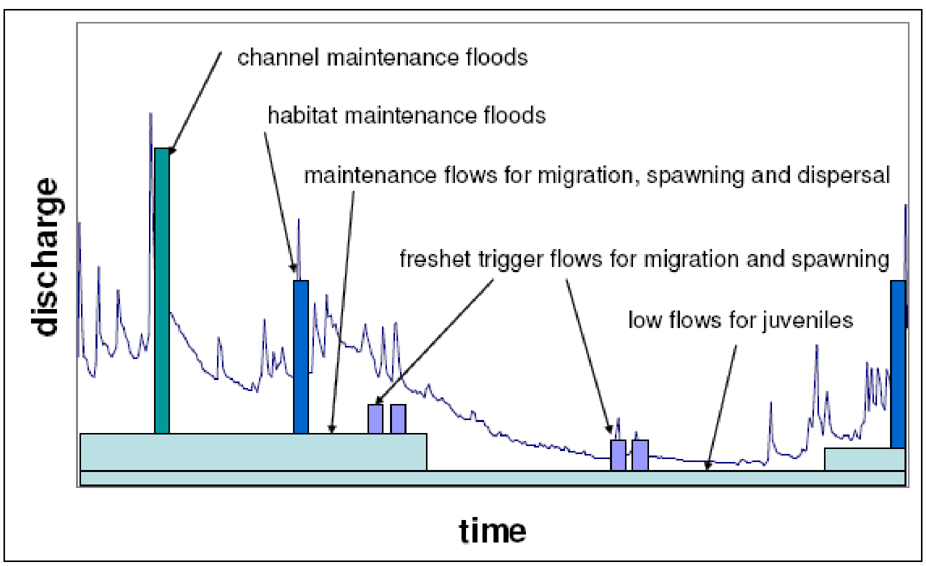
To define thresholds of flow alteration that would indicate a river would likely fail Good Ecological Status, the UK workgroup calculated statistics of flow alteration for nine UK sites.

This analysis helped to shape the recommendations in the table below, which are for a subset of Low Flows 2000 statistics that were also considered by the workgroup to be: 1) ecologically-relevant and 2) analogous to IHA statistics.

Table 4 Thresholds of hydrological alteration to meet GES

Low Flows 2000 statistics	**Low risk of failing GES if alteration less than 40% in all statistics**	**Medium risk of failing GES if alteration greater than 40 but less than 80% in any statistic**	**High risk of failing GES if alteration greater than 80% in any statistic**
mean January flow (m^3s^{-1}) mean April flow (m^3s^{-1}) mean July flow (m^3s^{-1}) mean October flow (m^3s^{-1}) Q_{95} (m^3s^{-1}) Q_5 (m^3s^{-1}) BFI			

The final section of this Guidance on Environmental Flow Releases from Impoundments goes into detail on developing future environmental flow release regimes. This provides a good review of the literature and proposes three levels of assessment that can be used to develop flow release recommendations through the "Building Block Methodology" as described by King et al. (2000). A conceptualization of this Building Block Methodology is included below from Acreman (2007).



Taken together, these two guidance documents from the United Kingdom provide a valuable example of how to: 1) use a hydrologic foundation; 2) develop a river classification and use it to structure flow standards; 3) assess hydrologic alteration; 4) develop risk-based standards for abstraction; and 5) define a process for developing environmental release regimes. These steps were all completed using best available scientific information and applied across a broad spatial scale. It is also useful to note that empirical flow-ecology relationships were not developed in this case and there was no separate social process for standard-setting. In this way, work in the United Kingdom may provide a good example of how standard-setting can proceed through an expert consensus process. Its social legitimacy and ease of implementation has yet to be tested to date.

CASE STUDY 4: MICHIGAN WATER WITHDRAWAL ASSESSMENT PROCESS

The State of Michigan recently engaged a collaborative advisory council to design a new, science-based Water Withdrawal Assessment Process and make recommendations regarding its implementation as policy. Initial development was under the legislatively-appointed Michigan Groundwater Conservation Advisory Council (Council) and is currently under Michigan Department of Environmental Quality (MDEQ). The Michigan legislature charged the Council to design and make recommendations regarding a process to guide state assessment of proposed large-quantity (>100,000 gallons per day) water withdrawals using a scientific (hydrologic and ecologic) basis for decision making as per recent state legislative mandate (PA 2006 34) and to comply with the Michigan's responsibilities under the Great Lakes Charter Annex (Council of Great Lakes Governors 2001) and the Great Lakes Water Resources Compact. Much of the material in this Case Study is taken from the final report to the Michigan Legislature (Michigan Groundwater Conservation Advisory Council 2007) and forthcoming technical reports.

The objective was to provide a suite of linked hydrologic and ecologic tools to be used in objective assessment of the potential for a proposed large-quantity withdrawal to adversely impact water-dependent natural resources. Work was done within a collaborative framework that was overseen by the Council, with members representing broad interests in societal water use; incorporated a national science review panel; and with technical work done by multiple agencies and universities (P. Seelbach, Michigan Department of Natural Resources, personal communication).

Michigan legislation states that "*A person shall not make a new or increased large quantity withdrawal from the waters of the state that causes an adverse resource impact.*" The following definitions were established in the legislation:

> "Adverse resource impact" means "Decreasing the flow of a stream by part of the index flow such that the stream's ability to support characteristic fish populations is functionally impaired".
>
> "Index flow" means the 50% exceedance flow for the lowest flow month of the flow regime (typically August or September), for the applicable stream reach, as determined over the period of record or extrapolated from analyses of the United States Geological Survey stream flow gauges in Michigan.

The Water Withdrawal Assessment Process includes a series of models used to determine whether or not a withdrawal is likely to functionally impair the ability of a stream to support Characteristic Fish Populations. Figure 1 shows the relationship between the three models:

- The **Streamflow Model** describes how much flow is in Michigan streams (i.e., predicted Index Flow [August or September Q50]) based on statewide regression models of gaged flows and GIS catchment attributes.

- The **Withdrawal Model** describes how much a withdrawal will reduce streamflow in nearby streams. A 1:1 relationship is assumed for surface water withdrawals. For any specified groundwater withdrawals, the impact of the proposed withdrawal on all nearby river segment discharges can be estimated using a generalized groundwater model that considered withdrawal distance and depth, and geologic texture.
- The **Fish Community Model** describes how reduced streamflow will affect Characteristic Fish Populations in affected streams. The fish community model is based on a form of habitat suitability indices that relate population abundance of individual fish species to key landscape-scale habitat measures (baseflow yield, July mean temperature, and catchment area) at many rivers throughout the state. These regional-based indices, when combined with measured or modeled landscape-scale habitat data for individual reaches, provided an empirical basis for setting standards (see Annear et al. 2004) for acceptable water withdrawals throughout Michigan. Individual fish species population models are used to define "characteristic" and "thriving" species for each stream segment, and the Fish Community Model predicts how these species groups would be affected by a water withdrawal. Zorn et al. (in prep) describes the how characteristic and thriving species groups were defined for each stream.

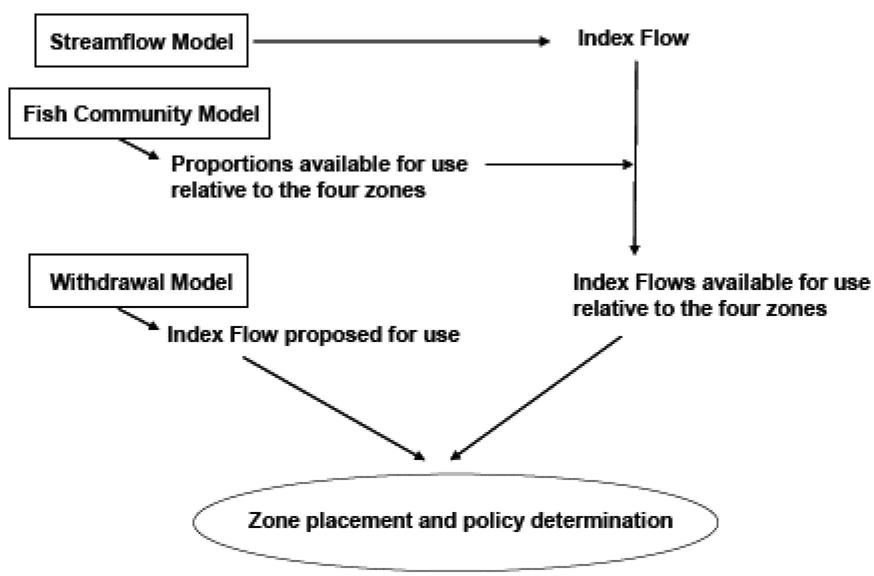


Figure 1. Linked models used to assess impact in the Michigan Water Withdrawal Process (from Michigan Groundwater Conservation Advisory Council 2007)

The Council used data from fish surveys at 1720 sites in Michigan to develop relationships between the proportion of index flow removed and the response of the fish population metric (Zorn et al. in prep). These empirical relationships were used to create smooth "flow-fish functional response curves" (Figure 2). These response curves, similar to Davies and Jackson (2006), included several levels of ecological degradation including (1) initial condition; (2) some density changes; (3) some replacement of sensitive species; (4) notable replacement of sensitive species; (5) tolerant species dominant and ecological functions altered; and (6) severe alteration of ecological structure and function.

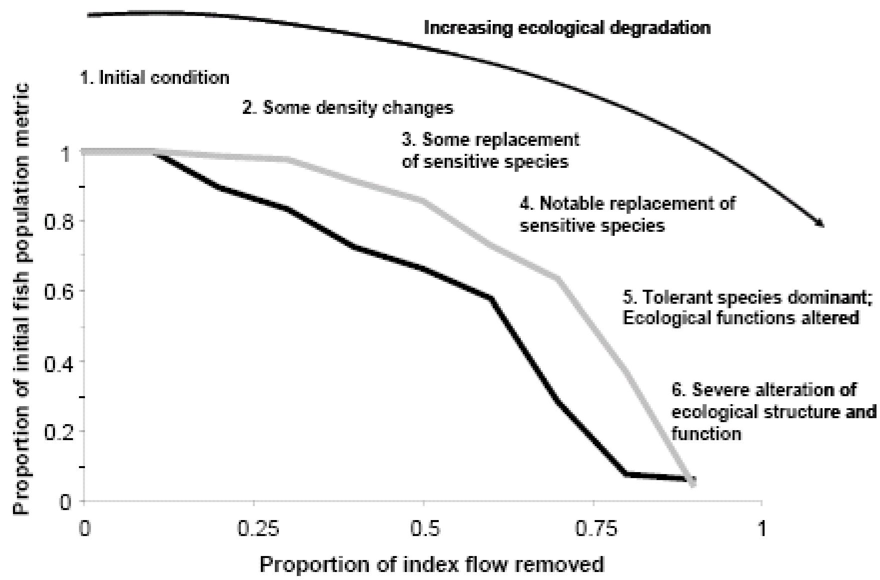


Figure 2. Steps in ecological degradation as index flow is increasingly removed (from Michigan Groundwater Conservation Advisory Council 2007).

The Council then used a stakeholder process to interpret the curves in terms of ecological risk and chose to use two horizontal lines as a tool; one at 90% of the initial fish population metric and one at 80% (Figure 3). The points where these horizontal lines intersected fish response curves A and B were used to draw several vertical lines to the bottom axis, indicating proportional flow removals associated with each threshold risk point. This approach resulted in risk threshold points that satisfied several objectives: (1) to keep ecological impacts to a minimum, primarily 10% or less of the initial fish population metrics; (2) to correspond with the levels of ecological degradation equating to “Some density changes” and “Some replacement of sensitive species,” while staying clear of “Notable replacement of sensitive species”; and (3) to stay on the upper portion of the curves, away from the inflection point that leads to steeper slopes and riskier decisions.

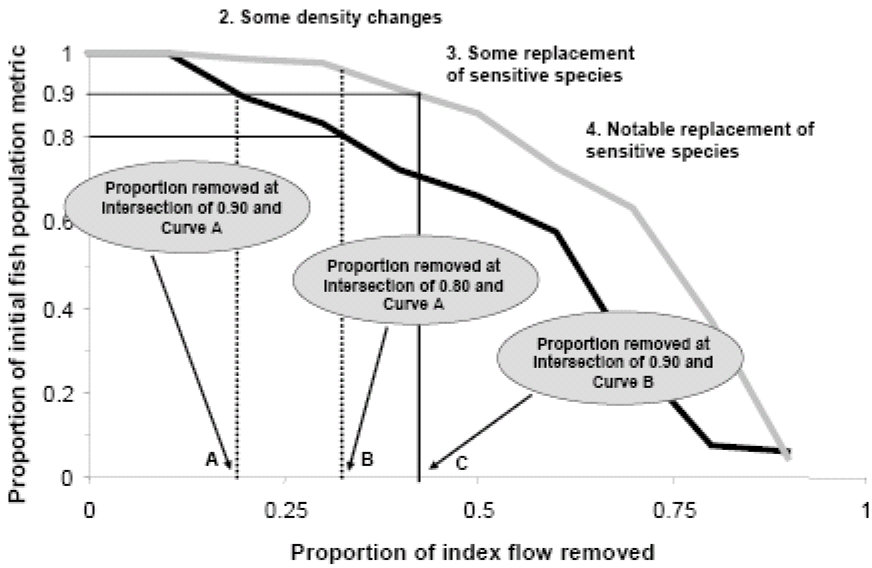


Figure 3. Analysis of the functional response curves yielded three 'proportions of index flow removed' (A, B, and C) that correspond to ecological change (from Michigan Groundwater Conservation Advisory Council 2007).

The three vertical lines in Figure 3 create four zones that represent increasing levels of ecological risk (Figure 4). The linked models within the Screening Tool are used to place a proposed withdrawal into one of the four zones. The water user may prefer to undertake a more detailed site specific analysis to determine the potential risk of the proposal. This is allowed within the Water Withdrawal Assessment Process. Whether the Screening Tool or a Site Specific Analysis is used, the Council's intent is that each zone has a corresponding action or actions to be taken by the user in order to minimize ecological risk and avoid an Adverse Resource Impact. The specific actions for some zones are still being determined; Figure 5 is an illustration of potential actions associated with each zone.

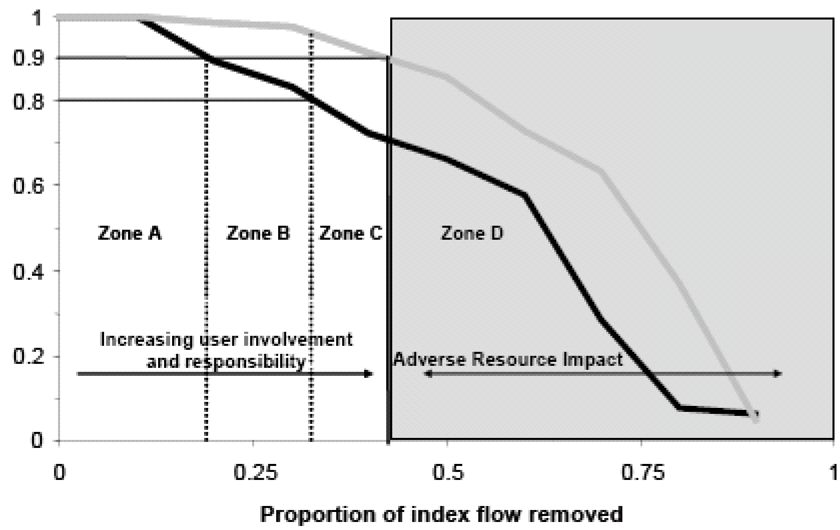


Figure 4. The four policy zones demarcated by increasing levels of index flow removal (from Michigan Groundwater Conservation Advisory Council 2007).

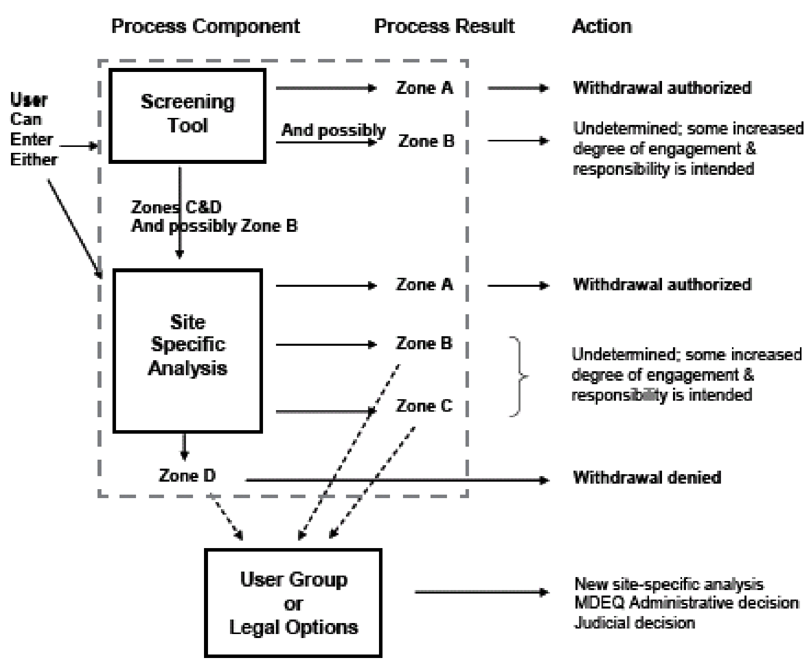


Figure 5. Diagram of the proposed Water Withdrawal Assessment Process. The dashed box contains the realm of the models to assess impact (from Michigan Groundwater Conservation Advisory Council 2007).

This approach was applied to 11 stream types classified based on drainage area and July mean temperature, as described in Section 3. The Council recognized that in some stream types, fish abundances are highly sensitive to fairly small flow reductions and in other stream types, fish abundances show fairly gradual changes in response to much larger flow reductions. The "flow-fish functional response curve" created a consistent set of rules that include both an acceptable level streamflow reduction and corresponding of fish population change, and a degree of caution (i.e., some safety factor) that recognizes the uncertainties inherent in the process. Once the Council agreed to this approach, the same rules were applied to the response curves for each stream type to determine how much flow can be removed without causing an unacceptable change in fish populations, (i.e., Adverse Resource Impact). The result was a set of 11 flow-fish functional response curves, one for each stream type (Figure 6).

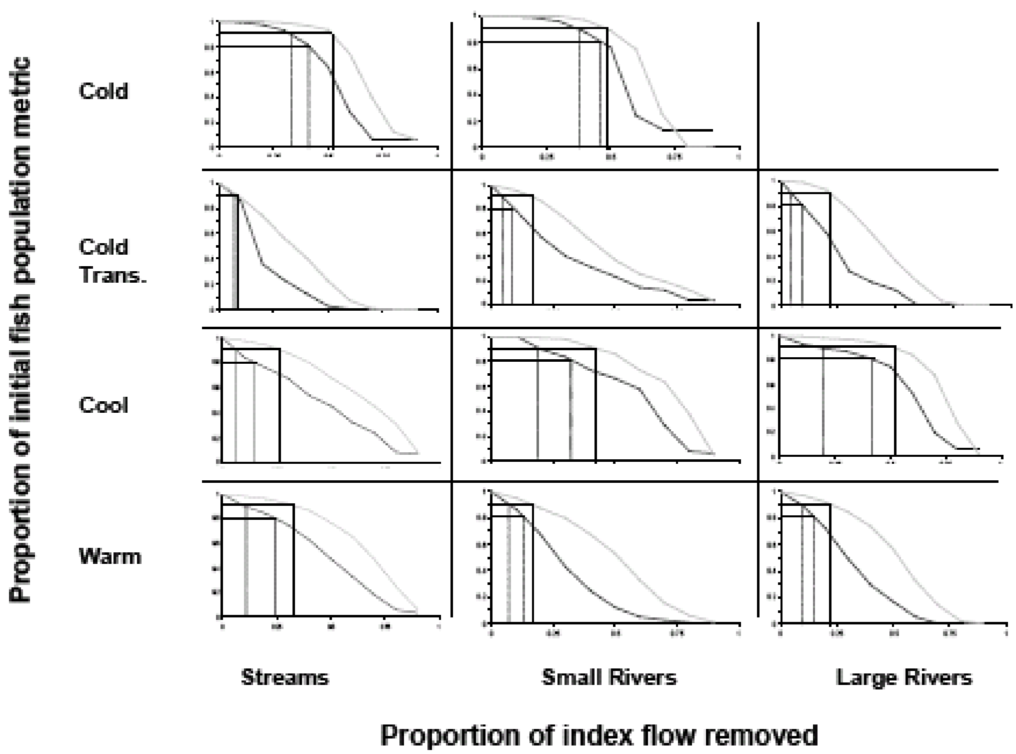


Figure 6. Flow-fish functional response curves and line creating zones of ecological risk for each of the eleven stream types (from Michigan Groundwater Conservation Advisory Council 2007).

CASE STUDY 5: DEVELOPMENT OF FLOW ALTERATION-ECOLOGICAL RESPONSE CURVES FOR PENNSYLVANIA STREAMS

Background

In the April 2006 Scope of Work for “Developing Hydrologic Criteria to Support Ecologically Sustainable Water Resource Planning and Management: Phase 1,” The Nature Conservancy proposed to use a “pilot watershed” to “attempt to use existing data to define the relationship between biological indicators and altered flow conditions for major river types across the state”. The Scope of Work further stated that “the pilot study will use biological and hydrologic data that is already available from data sources,” and “once an adequate biological database is developed, key metrics for assessment of the impacts of hydrologic alteration will need to be selected”. During the term of the project, TNC worked with the Advisory Committee to select the Susquehanna River basin in PA as the watershed for this pilot study.

As stated in the Scope of Work, “general relationships between the statistics of hydrologic impact and the selected biological data will be developed in the pilot watershed. The objective will be to develop a generalized predictive model that would estimate the degree of biological impact that can be expected from a given degree of hydrologic impact for a given stream type.” To accomplish these goals, we developed a water withdrawal index using data from the Pennsylvania Water Availability Screening Tool (WAST). More specifically, we used WAST data on cumulative water withdrawals at "pour points" across a portion of the Susquehanna basin to develop an index of cumulative water use relative to a low flow parameter (i.e., 7Q10, the 7-day low flow that is predicted to occur every 10 years on average) in a manner similar to Freeman and Marcinek (2006) and Weiskel et al. (2007).

By linking a water withdrawal index at these pour points to locations with available biological data, the project team developed statistically-based, quantitative estimates of ecological response to water withdrawals. We used changes in standard aquatic insect metrics calculated by SRBC (taxonomic richness, modified Hilsenhoff Biotic Index, percent Ephemeroptera, percent dominant taxon, number of Ephemeroptera, Plecoptera, and Trichoptera taxa, percent Chironomidae, and Shannon-Wiener Diversity Index) and aquatic insect functional traits as measures of ecological response because: (1) macroinvertebrate data are available at many sites throughout the Susquehanna River basin, and provide the best opportunity for matching sites with biological data with estimates of the water withdrawal index, (2) we were interested in examining the potential to use standard macroinvertebrate indices that are widely calculated as indicators of biological response to water quality as indicators of biological response to water quantity, and (3) analyses of functional traits allow us to examine functional changes in aquatic insect assemblages regardless of species (taxonomic) composition.

We want to emphasize that this is an exploratory investigation of the feasibility of linking hydrologic alterations with ecological responses using existing data within Pennsylvania. We developed these approaches to help quantify the risk of negative ecological response as a consequence of increasing degree of flow alteration. This information is a first step towards providing a biological foundation for statewide instream flow standards that can be effectively used by decision-makers.

Hydrologic metrics

We explored two methods of calculating hydrologic alteration in the Susquehanna River basin: (1) alteration in daily flows as assessed by the OASIS (Operational Analysis and Simulation of Integrated Systems) hydrologic model that was developed for the Susquehanna River basin, and (2) total water withdrawals (permitted and estimated, both absolute and adjusted for return flows) as a proportion of the estimated 7Q10 associated with each pour point in WAST. We evaluated each method according to spatial and temporal resolution of the data and the ability to match macroinvertebrate sampling sites with point estimates of hydrologic alteration. Ultimately, we chose to use the WAST data in our final analysis; however, we also describe the data available from the OASIS model below.

OASIS model

The OASIS model was used to calculate daily baseline and regulated flows from 1930 to 2002. Baseline flows accounted for historical hydrologic conditions (precipitation patterns) without considering any consumptive water use or reservoir operations. Regulated flows also accounted for historical hydrologic conditions and included consumptive water use and reservoir operations at 2002 levels. Regulated flows only accounted for net water balance (water withdrawals adjusted for discharges), so the final data are likely to mask sites where water withdrawals are mitigated by discharges within the same subbasin.

We used The Nature Conservancy's Indicators of Hydrologic Alteration (IHA) software to calculate metrics of hydrologic alteration at all OASIS nodes based on daily estimates of baseline and regulated flows over the entire period of record. IHA calculates 67 statistics of hydrologic alteration that describe alterations in several ecologically important aspects of the hydrologic regime: magnitude of high, low, and average flows, duration of high and low flows, frequency of high and low flows, timing of high and low flows, and rate of change during flow events. Rather than examining alteration for all 67 flow statistics calculated by IHA, we focused on a subset of 10 flow statistics that represented ecologically important aspects of the flow regime (Table 1).

Table 1. Ecologically important aspects of the hydrologic regime (flow components) and associated statistics of hydrologic alteration (a subset of all the flow statistics calculated by the Indicators of Hydrologic Alteration software). These 10 flow statistics were calculated using data from the OASIS model and were examined to characterize changes in stream flow due to consumptive use in the Susquehanna River basin.

Flow component	IHA statistic
Average magnitude	Mean annual flow
Low magnitude	3-day minimum
High magnitude	3-day maximum
Low duration	Low pulse duration

High duration	High pulse duration
Low frequency	Low pulse count
High frequency	High pulse count
Low timing	Date of minimum
High timing	Date of maximum
Rate of change	Reversals

Water Availability Screening Tool (WAST)

We calculated two different water withdrawal indices using the hydrologic data associated with each pour point in the WAST database. We obtained a copy of the WAST database from Pennsylvania DEP in November 2007. In addition to several basin attributes for each pour point (including, for example: area, percent forest and urban land cover, precipitation), the database includes several fields of hydrologic data that were used to calculate water withdrawal indices. These fields include:

TotalSW – Total Surface Water (in gallons per day). This is a modeled estimate of the 7Q10 at each pour point based on the regression models developed by USGS and described by Stuckey (2006).

TotalWaterUse – Estimate of the total water withdrawn (in gallons per day) upstream of each pour point based on registered water withdrawals. Withdrawals were adjusted to address the fact that facilities may withdrawal at different daily rates during different months (For example, if a facility reported an average of 8 mgd discharge for 8 months and 6 mgd for the other 4 months of the year, the value used would be 7.3 mgd).

Discharges – Estimate of total water discharged upstream of each pour point (in gallons per day).

We used these three values to calculate two indices, described below, for all pour points selected for the analysis.

Withdrawal index = TotalWaterUse/TotalSurfaceWater. This is a simple ratio between the water withdrawn and the 7Q10 to show the proportion of 7Q10 that is withdrawn for use. This is the withdrawal index used by Freeman and Marcinek (2006) to develop a relationship between water use and fish assemblages.

Adjusted Withdrawal index = (TotalWaterUse - Discharges)/TotalSurfaceWater. This index adjusts the *withdrawal index* by incorporating (adding back) known discharges. This index may account most fully for the overall degree of hydrologic alteration experienced in the aquatic ecosystem, although discharged water is often of a different quality than the water that was originally withdrawn.

Ecological metrics

We used macroinvertebrate data collected by SRBC at 665 sites throughout the Susquehanna River basin as the basis for our ecological response variables. We chose to use SRBC's macroinvertebrate data because: (1) the dataset had high spatial extent and resolution throughout the Susquehanna River basin, providing the best chance of matching with sites of hydrologic data of any ecological datasets available, (2) all samples were collected with the same standardized and documented methodologies, and (3) macroinvertebrates were identified to the genus level and assigned quantitative abundances, providing sufficient taxonomic resolution to examine changes in functional traits of macroinvertebrate assemblages among sites. Methods used to sample and process macroinvertebrates are documented by Susquehanna River Basin Commission (2007), and are based on a modified version of the U.S. Environmental Protection Agency's Rapid Bioassessment Protocol III (Barbour et al. 1999).

We investigated databases of macroinvertebrate samples available in Pennsylvania from various other sources and assessed the geographic coverage of sampling sites, sampling dates, taxonomic resolution, and sampling methods. We decided that we could not combine samples from different sources in this analysis because of differences in sampling methods, including sampling intensity, habitats sampled (e.g., riffles only versus multiple habitat types), taxonomic resolution of organism identifications, and use of qualitative versus quantitative estimates of taxa abundance. As mentioned above, we chose to use data collected by the Susquehanna River Basin Commission (SRBC) because of the relatively high density and broad geographic scope of samples within the Susquehanna River basin, recent dates of sampling, quantitative abundances of taxa, and identification of organisms to genus.

We used standard macroinvertebrate metrics commonly calculated for water quality monitoring as well as aquatic insect functional traits as our ecological metrics for this analysis. We used seven macroinvertebrate metrics calculated by SRBC for all sites, including taxonomic richness, modified Hilsenhoff Biotic Index (HBI), percent Ephemeroptera, percent dominant taxon, number of Ephemeroptera, Plecoptera, and Trichoptera taxa (EPT), percent Chironomidae, and Shannon-Wiener Diversity Index (Table 2). These metrics were developed to examine biological response to water quality degradation rather than changes in water quantity; however, all metrics generally measure macroinvertebrate response to disturbances. We were interested in examining how well they could be used to characterize responses to other disturbances, such as hydrologic alteration.

Table 2. Summary of metrics used by the Susquehanna River Basin Commission to evaluate the overall biological integrity of stream and river benthic macroinvertebrate assemblages, and predicted changes in each metric in response to disturbance. Table adapted from Steffy et al. (2007).

Metric	Description	Prediction
Taxonomic Richness	The total number of taxa present in the sample at each site	Number decreases with increasing stress
Shannon-Wiener Diversity Index	A measure of biological community complexity based on the number of equally or nearly equally	Index value decreases with

	abundant taxa in the community	increasing stress
Modified Hilsenhoff Biotic Index	A measure of the organic pollution tolerance of a benthic macroinvertebrate community	Index value increases with increasing stress
EPT Index	The total number of Ephemeroptera (mayfly), Plecoptera (stonefly), and Trichoptera (caddisfly) taxa present in the sample at each site	Number decreases with increasing stress
Percent Ephemeroptera	The percentage of Ephemeroptera in the sample at each site	Ratio decreases with increasing stress
Percent Dominant Taxa	Percentage of the taxon with the largest number of individuals out of the total number of macroinvertebrates in the sample	Percentage increases with increasing stress
Percent Chironomidae	The percentage of Chironomidae in the sample at each site	Ratio increases with increasing stress

In addition to the above macroinvertebrate metrics, we used aquatic insect functional traits as ecological metrics for this analysis because of the ability to mechanistically link functional trait responses to hydrologic alteration through hypotheses developed from published literature. For example, we know from previous studies that increases in disturbance (such as increased periods of drought) result in an increase in the abundance of taxa that have a small size at maturity and are abundant in the drift; these taxa can escape poor conditions due to high mobility, are good at colonizing new areas, and invest less energy in growth relative to reproduction. We can explicitly test these hypotheses using statistical models that relate hydrologic alteration to the relative abundance of individuals in a particular trait-state. Use of aquatic insect traits also allows us to combine taxa into groups expected to respond consistently over environmental gradients. In addition, we can examine functional composition at a site rather than taxonomic composition, which is more site-specific and often responds more variably to factors unrelated to hydrologic alteration (such as presence of endemic species in areas that are geographically isolated).

We converted abundance data by genus to abundance by functional trait for all SRBC samples using a published functional trait matrix (Poff et al. 2006). For each aquatic insect genus per sample, we recorded total abundance for the correct trait-state in each functional trait category. For example, Baetis (Ephemeroptera: Baetidae) has a small size at maturity. There are three trait-state categories for the functional trait labeled “size at maturity” (small, medium, and large). In the data matrix, we recorded total Baetis abundance for the sample in the small column, and zeros in the columns for medium and large. We then summed the total abundance for each trait state across all genera in each sample, with the final product of one row of data per sample, with total aquatic insect abundance per trait-state in each column. We completed these calculations for all aquatic insect taxa in all samples, and for all 20 traits (59 trait-states) described by Poff et al. (2006). Poff et al. (2006) provides a complete list of traits and trait-states organized into 4 categories (life history, mobility, morphology, ecology) for all aquatic insect genera in North America.

Linking locations of hydrologic metrics to macroinvertebrate sampling sites

OASIS model

Daily flow data computed by the OASIS model were available for 74 locations throughout the Susquehanna River watershed. We used a "near function" in ArcGIS 9.2 to match these OASIS nodes with the closest of 665 macroinvertebrate sampling points located throughout the basin. This method resulted in a total of 19 macroinvertebrate sampling points that could be matched with OASIS nodes because the sampling point was within 2 km of an OASIS node on the same stream and there were no major tributary inflows or reservoirs between them; we felt this number was insufficient as a sample size to adequately determine the statistical relationships between flow alteration and our ecological response variables. Thus, we did not use data from the OASIS model in our final analysis.

Water Analysis Screening Tool (WAST)

The Water Analysis Screening Tool provided hydrologic data for 5220 pour points throughout the Pennsylvania portion of the Susquehanna River watershed. We used a "Near function" in ArcGIS 9.2 to match the closest pour point to each of the 665 macroinvertebrate sampling points. Ultimately, we used sites in our analysis where pour points and macroinvertebrate sampling sites overlapped or could be reasonably linked within distance constraints. After this automated process, each pair of points was checked manually to confirm that they were matched appropriately (i.e., on the same stream or river and without any major tributary inflows or reservoirs between them). We set a maximum tolerance for distance separating pour points-sample points of 2 km, but given the density of pour points, we found that most pairs were much closer together. We eliminated a site from the analysis if it was determined to have poor water quality as indicated by 303(d) listing (all sites on the 303(d) list were eliminated unless hydrologic alteration was listed as the primary reason for listing) or very low pH values (<5). Our final sample size, after matching sites for hydrologic and ecological data and removing sites with poor water quality, was 298 sites.

Hypotheses of ecological response to water withdrawals

Our analysis of the relationships between water withdrawals and macroinvertebrate metrics commonly used for water quality monitoring (described above) was meant to explore use of these metrics to assess relationships with changes in water quantity. Thus, we considered this part of the analysis to be exploratory and not necessarily based on hypotheses developed from the ecological literature. However, we did have hypotheses about the nature of the relationship between increasing water withdrawals and each macroinvertebrate metric, based on the relationship between each metric and disturbance in general (Table 2). For example, taxonomic richness is expected to decrease with increasing disturbance; thus, we also expected taxonomic richness to decrease with increasing water withdrawals. We did conduct a literature survey to construct hypotheses of the relationships between water withdrawals and functional traits of invertebrate assemblages. Hypotheses of responses of aquatic insects to water withdrawals, diversions, and/or extended periods of low and extreme low flows that we explicitly examined in our analyses are listed below:

- Decreased diversity (number of taxa) of grazers and shredders (McKay and King 2006). This is equivalent to an increase in generalist feeders (collector-gatherers).
- Increase in the abundance of individuals with small body size at maturity (Richards et al. 1997, Rader and Belish 1999)
- Decrease in the abundance and number of taxa that are rare in the drift (species rare in the drift could not replenish lost populations when more favorable flow conditions were present; this is equivalent to an increase in abundance of invertebrates common in the drift - Rader and Belish 1999)
- Increase in abundance and number of taxa that are multivoltine (Richards et al. 1997)
- Increase in abundance and number of taxa that are obligate depositional (Richards et al. 1997)
- Increase in abundance and number of taxa with high thermal tolerance (eurythermal); decrease in abundance and number of taxa that are cold stenothermal (Lake 2003)

A full list of hypotheses developed from this literature survey is included as Case study supplement B.

Statistical analyses

We constructed regression models to examine the relationships between water withdrawals and the seven macroinvertebrate metrics (described in Table 2) and the six functional trait metrics indicated by the above hypotheses that were developed from the literature review. For example, one of our hypotheses stated that increased water withdrawals will result in an increase in abundance of individuals with small body size at maturity. To examine this hypothesis, we constructed a regression model with the dependent (response) variable equal to the proportion of individuals with a small body size at maturity and predictor variables that included water withdrawals as a percentage of 7Q10.

Two regression models were constructed for each of the 13 ecological response variables; the first examined the relationship between the response variable and Withdrawal Index and the second examined the relationship between the response variable and Adjusted Withdrawal Index. Of the 13 ecological response variables, nine were expressed as a proportion. For these nine variables, we used logistic regression models using the events/trials syntax (PROC LOGISTIC; SAS Institute Inc. 1999). For each sample, the response variable was expressed as the number of individuals that met the criteria of the variable (events) divided by the total number of individuals in the sample (trials). For example, for percent Ephemeroptera, the response variable was expressed as the number of Ephemeroptera (events) divided by the total number of macroinvertebrates in the sample (trials), rather than as a proportion. R^2 values for logistic regression models were derived by modeling the relationship between observed data and values predicted by the model. For the remaining four ecological response variables that were not expressed as proportions, we used linear regression to examine the relationship between each response and the predictor variables (PROC GLM; SAS Institute Inc. 1999).

In addition to our main predictor variables of interest (Withdrawal Index and Adjusted Withdrawal Index), we included covariates in all models. These covariates included pH and

Dissolved Oxygen (indicators of water quality) and drainage basin area (natural log transformed), and we also examined interactions between predictor variables for possible inclusion in the model. Before adding covariates we determined that none of the predictor variables were significantly correlated with each other. We also examined different functional forms of the hydrologic predictor variable (Withdrawal Index or Adjusted Withdrawal Index, depending on the model) to explore possible non-linear relationships (e.g., we examined the significance of a quadratic term and a natural log transformation of the predictor variable in the model). Strength of the models were assessed based on significance of the hydrologic predictor variables (Withdrawal Index or Adjusted Withdrawal Index) in explaining variation in the ecological response variable and the amount of variation in the raw data that was explained by the model (R^2). Final models for each ecological response variable are listed in Tables 3 and 4.

In addition to examining the amount of variation in the raw data explained by each full model (R^2), we were interested in examining the amount of variation explained by the hydrologic predictor variable (Withdrawal Index or Adjusted Withdrawal Index, depending on the model) relative to other covariates included in the model. To accomplish this, we calculated partial R^2 values for all predictor variables in each model. We calculated partial R^2 by comparing the R^2 for the full model (containing all predictor variables) with the R^2 for a model that did not include the predictor variable of interest (but did contain all other predictor variables). This method gives an indication of additional variance explained by adding the variable of interest, but may be influenced by correlations among all predictor variables in the model.

For all of our models, our primary interest was to examine response of ecological variables as hydrologic alteration increased, measured by Withdrawal Index or Adjusted Withdrawal Index. For both of these hydrologic variables, most of the values at the sites we used fell somewhere between 0 and 1 (indicating that withdrawals were between 0 and 100% of estimated 7Q10); however, 15% of sites had a Withdrawal Index > 1. The 15% of sites with Withdrawal Index > 1 spanned a large range, with the maximum withdrawal at 653% of 7Q10 (Table 5). This pattern of the majority of our sites in a small range (0 to 1) and a relatively small number of sites across a large range (1 to 6.53) made it difficult to use linear regression modeling; the sites with large values for Withdrawal Index had disproportionate influence on the shape of the final curve. To account for this we tried two different approaches: (1) we truncated all values of Withdrawal Index greater than 1 at 1, and (2) we eliminated all sites with Withdrawal Index greater than 1 from our analysis. The first method preserves all our data points but could potentially result in inferences that Withdrawal Index of 1 has a greater effect on ecological variables than the raw data support (if Withdrawal Index values greater than 1 have an increasingly greater effect on the ecological response variable). The second method accurately models the relationship between Withdrawal Index and ecological response variables for the interval between 0 and 1, but ignores important information about higher values for Withdrawal Index and reduces our sample size. We chose to examine both methods and compare results to determine the method that made the most sense for these data.

Results

Withdrawal Index

We examined the relationships between each of the 13 ecological response variables and Withdrawal Index to determine if there was a significant relationship between each response variable and water withdrawals (not accounting for discharges), and if the nature of these relationships matched our predictions (outlined above). We found significant relationships between Withdrawal Index and 12 out of the 13 ecological response variables (Table 3). The only response variable that was not significantly correlated with Withdrawal Index was the proportion of aquatic insects that were obligate depositional. In addition, the nature of the relationship between each ecological response variable and Withdrawal Index matched our hypotheses, with the exception of the proportion of aquatic insects that were obligate depositional (a p-value of 0.10 indicated no relationship; Table 3). The proportion of aquatic insects common in the drift responded to increases in Withdrawal Index as we hypothesized for small basins, but showed the opposite trend for large basins.

Although Withdrawal Index was a significant predictor of changes in all but one of our ecological response variables, the proportion of variation explained by each of our models was relatively low (indicated by R^2 values; Table 3). The models that explained the greatest proportion of variation in the response variables included the models for HBI, EPT, proportion of aquatic insects with small size at maturity, proportion of aquatic insects bi- or multi-voltine, and proportion of aquatic insects that were collector-gatherers (generalist feeders). Of this subset of response variables, Withdrawal Index was the predictor variable that explained the greatest proportion of variation in the response variable for two of the models, HBI and EPT (as indicated by partial R^2 values; Table 3). The relationships between HBI and Withdrawal Index and EPT and Withdrawal Index both depended on drainage area (Figure 1, 2). We present curves for four drainage areas within the range used in our analyses for each of these response variables (Table 5; Figure 1, 2).

As we described above, we used two different datasets for Withdrawal Index when analyzing the effects of Withdrawal Index on the ecological response variables: (1) all sites, with values for Withdrawal Index that were greater than 1 truncated at 1, and (2) a subset of sites, eliminating from the analysis sites that had Withdrawal Index greater than 1. All data presented here and in Table 3 are from dataset (1): all sites. Analyses using dataset (2) yielded similar results. Using a subset of sites with Withdrawal Index between 0 and 1 resulted in significant relationships for 11 of the 13 ecological response variables (compared with 12 of the 13 for the truncated Withdrawal Index). All of the resulting curves illustrated the same general trends (direction of the relationship), although the slopes of the curves tended to be different because different data were used to generate the curves. Because results were generally similar whether we used all sites with a truncated Withdrawal Index or a subset of sites with Withdrawal Index between 0 and 1, we chose to only present results that used the entire dataset of sites.

Adjusted Withdrawal Index

Similar to our analyses of Withdrawal Index, we examined the relationships between 13 ecological response variables and Adjusted Withdrawal Index to determine the relationship between each response variable and water withdrawals adjusted for discharges (i.e., net water use). We found significant relationships between Adjusted Withdrawal Index and ecological response for 8 of the 13 variables (Table 4). Ecological variables not significantly associated with changes in Adjusted Withdrawal Index included taxonomic richness, Hilsenhoff Biotic Index, percent dominant taxon, number of EPT taxa, and Shannon-Wiener diversity index. Interestingly, two of these variables (HBI and EPT) had the strongest associations with Withdrawal Index (not adjusted for discharges; Table 3). As with Withdrawal Index, the nature of the relationship between Adjusted Withdrawal Index and each ecological response variable generally followed our hypotheses, with the exception of the five variables listed above (that showed no relationship with Adjusted Withdrawal Index) and the proportion of aquatic insects abundant in the drift (that decreased with increasing values for Adjusted Withdrawal Index, contrary to our prediction).

The proportion of variation in the ecological response variables that was explained by the models used in our analysis was fairly low, even relative to the R^2 values for the Withdrawal Index models (Table 4). In addition, partial R^2 values indicated that water quality parameters (particularly pH) were generally more important predictors of variation in the ecological response variables than net water use. Also in contrast to Withdrawal Index, relationships between most of the ecological response variables and Adjusted Withdrawal Index did not depend on drainage area (i.e., the interaction between Adjusted Withdrawal Index and the natural log of drainage area was not significant for most models; Table 4).

Table 3. Response of ecological variables to Withdrawal Index. Ecological response variables are defined in the text. Predictor variables include Withdrawal Index (WI), water quality variables (pH and Dissolved Oxygen [DO]), and the natural log of drainage area (lnArea). The interaction between WI and lnArea (WI*lnArea) was included as a predictor variable if the slope of the relationship was significantly different from zero (α=0.05, indicated by a p-value<0.05). Model R^2 represents the proportion of total variation in the data that is explained by the full model (all variables). Partial R^2 represents the additional proportion of variation explained by adding Withdrawal Index to a model containing all other variables. Values in bold (under p-value) indicates that WI made a significant contribution to the model (i.e., the slope of the relationship between WI and the ecological response variable was significantly different from zero).

Response variable	Predictor variables in model	Model R^2	Significance and strength of WI term		Variable with highest partial R^2 (value for partial R^2 in parentheses)
			p-value	Partial R^2	
Taxa richness	WI, pH, DO, lnArea	0.10	**<0.01**	0.01	lnArea (0.04)
Hilsenhoff Biotic Index	WI, pH, DO, lnArea, WI*lnArea	0.23	**<0.01**	0.07	WI (0.07)
% Ephemeroptera	WI, pH, DO, lnArea, WI*lnArea	0.05	**<0.01**	0.03	WI (0.03)
% Dominant taxon	WI, pH, DO, lnArea	0.03	**0.04**	0.01	WI (0.01)
# Ephemeroptea, Plecoptera, and Trichoptera taxa	WI, pH, DO, lnArea, WI*lnArea	0.13	**<0.01**	0.05	WI (0.05)
% Chironomidae	WI, pH, DO, lnArea, WI*lnArea	0.08	**<0.01**	0.05	WI 0.05)
Shannon-Wiener diversity index	WI, pH, DO, lnArea	0.10	**<0.01**	0.01	DO (0.04)
% abundant in the drift	WI, pH, DO, lnArea, WI*lnArea	0.05	**<0.01**	0.02	pH (0.03)
% small size at maturity	WI, pH, DO, lnArea, WI*lnArea	0.17	**<0.01**	0.05	pH (0.06)
% bi- or multi-voltine	WI, pH, DO, lnArea, WI*lnArea	0.13	**<0.01**	0.03	pH, lnArea (0.06)
% collector-gatherers	WI, pH, DO, lnArea, WI*lnArea	0.24	**<0.01**	0.03	pH (0.19)
% cold stenothermal or cool eurythermal	WI, pH, DO, lnArea	0.09	**<0.01**	<0.01	pH (0.05)
% obligate in depositional habitats	WI, pH, DO, lnArea	0.07	0.10	<0.01	pH (0.04)

Table 4. Response of ecological variables to Adjusted Withdrawal Index. Ecological response variables are defined in the text. Predictor variables include Adjusted Withdrawal Index (adjWI), water quality variables (pH and Dissolved Oxygen [DO]), and the natural log of drainage area (lnArea). The interaction between adjWI and lnArea (adjWI*lnArea) was included as a predictor variable if the slope of the relationship was significantly different from zero (α=0.05, indicated by a p-value<0.05). Model R^2 represents the proportion of total variation in the data that is explained by the full model (all variables). Partial R^2 represents the additional proportion of variation explained by adding Adjusted Withdrawal Index to a model containing all other variables. Values in bold (under p-value) indicates that adjWI made a significant contribution to the model (i.e., the slope of the relationship between adjWI and the ecological response variable was significantly different from zero).

Response variable	Predictor variables in model	Model R^2	Significance and strength of adjWI term		Variable with highest partial R^2
			p-value	Partial R^2	
Taxa richness	adjWI, pH, DO, lnArea	0.09	0.94	0.01	lnArea (0.06)
Hilsenhoff Biotic Index	adjWI, pH, DO, lnArea	0.16	0.25	<0.01	pH (0.14)
% Ephemeroptera	adjWI, pH, DO, lnArea	0.03	**<0.01**	0.01	WI, pH (0.01)
% Dominant taxon	adjWI, pH, DO, lnArea	0.02	0.17	0.01	WI, pH, DO (0.01)
# Ephemeroptea, Plecoptera, and Trichoptera taxa	adjWI, pH, DO, lnArea	0.08	0.67	<0.01	pH (0.04)
% Chironomidae	adjWI, pH, DO, lnArea	0.04	**<0.01**	0.01	DO (0.02)
Shannon-Wiener diversity index	adjWI, pH, DO, lnArea	0.09	0.85	<0.01	DO (0.04)
% abundant in the drift	adjWI, pH, DO, lnArea, adjWI*lnArea	0.03	**<0.01**	<0.01	pH (0.02)
% small size at maturity	adjWI, pH, DO, lnArea	0.12	**<0.01**	<0.01	lnArea (0.07)
% bi- or multi-voltine	adjWI, pH, DO, lnArea, adjWI*lnArea	0.10	**<0.01**	<0.01	pH (0.06)
% collector-gatherers	adjWI, pH, DO, lnArea	0.20	**<0.01**	<0.01	pH (0.17)
% cold stenothermal or cool eurythermal	adjWI, pH, DO, lnArea	0.08	**<0.01**	<0.01	pH, lnArea (0.04)
% obligate in depositional habitats	adjWI, pH, DO, lnArea	0.08	**0.03**	0.01	pH (0.05)

Table 5. Range of values (mean, minimum, and maximum) for Withdrawal Index and Adjusted Withdrawal Index by drainage area of sites.

Drainage area range (mi^2)	# of sites	Withdrawal Index			Adjusted Withdrawal Index		
		Mean	Min	Max	Mean	Min	Max
< 25	87	0.46	0	6.53	0.46	-4.68	6.35
25-100	138	0.53	0	5.12	0.36	-2.97	5.90
100-1000	69	0.46	0.01	5.39	0.22	-2.07	3.11
> 1000	4	2.18	0	6.43	0.03	0	0.08

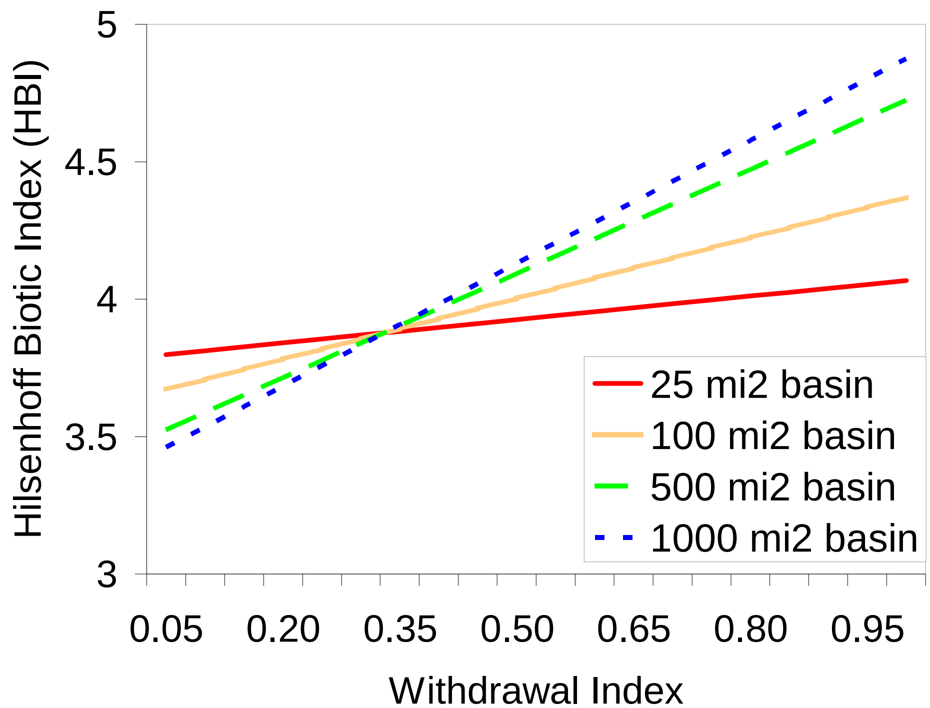


Figure 1. Relationship between modified Hilsenhoff Biotic Index (HBI) and Withdrawal Index. This relationship depended on drainage area; thus, we present curves for predicted values of the relationship for four drainage area sizes within the range of sites used in the analysis. Other covariates included in models (pH and dissolved oxygen) were held constant at mean values (pH=7 and dissolved oxygen=7).

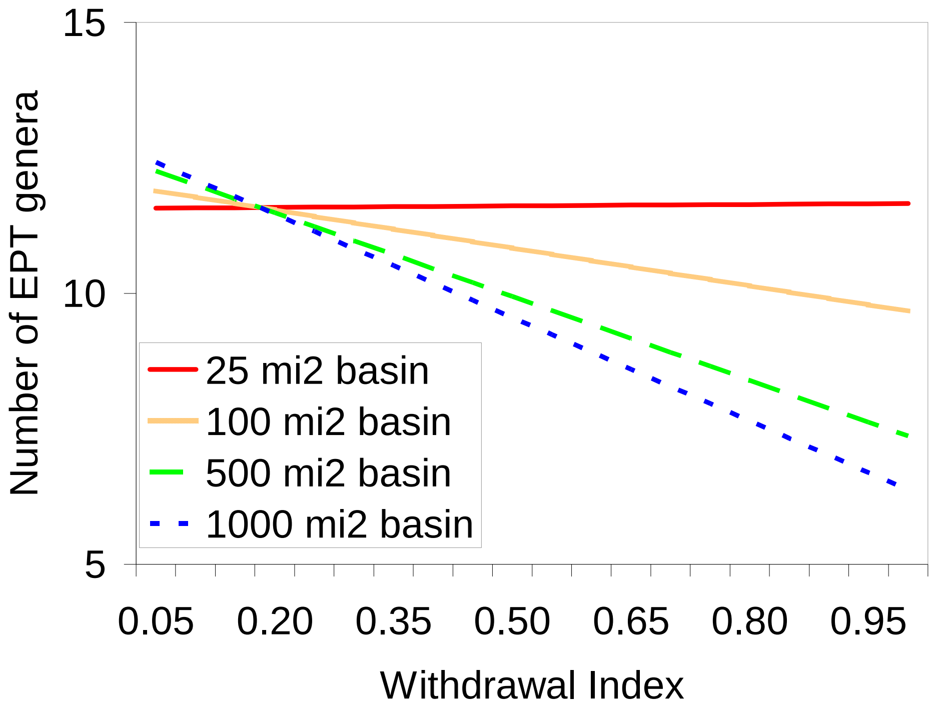


Figure 2. Relationship between the number of Ephemeroptera (mayflies), Plecoptera (stoneflies), and Trichoptera (caddisflies) genera (EPT) and Withdrawal Index. This relationship depended on drainage area; thus, we present curves for predicted values of the relationship for four drainage area sizes within the range of sites used in the analysis. Other covariates included in models (pH and dissolved oxygen) were held constant at mean values (pH=7 and dissolved oxygen=7).

Discussion

The most striking results of our analyses were the consistently significant relationships between ecological response variables and water withdrawals, and that the nature of these relationships matched our hypotheses. However, R^2 values for the models were low, indicating low predictive ability. We found significant relationships between Withdrawal Index and 12 out of the 13 ecological response variables that we examined. The only response variable that was not significantly correlated with Withdrawal Index was the proportion of individuals that were obligate depositional. These taxa are most often found in pool habitats (depositional areas), and all macroinvertebrate samples used in our analysis were collected from riffle habitats (erosional areas). Thus, the lack of a relationship between obligate depositional taxa and Withdrawal Index may have been due to sampling methodology.

Although Withdrawal Index (water withdrawals without considering discharges) had consistently significant relationships with ecological response variables and was the most important predictor for several of these variables, we did not observe similar results for Adjusted Withdrawal Index (water withdrawals adjusted for discharges, an estimate of net water use). Adjusted Withdrawal Index was significantly associated with several (8 out of 13) of the ecological response variables; however, the predictive ability of these models (measured by R^2) tended to be lower than those constructed for Withdrawal Index. In addition, partial R^2 values indicated that water quality variables (pH and dissolved oxygen) tended to be more important predictors of ecological response in these models. This suggests that the water released as discharges is not ecologically equivalent to water left in the stream, likely because discharged water often has degraded water quality and/or increased temperature. Macroinvertebrates are likely responding both to alterations in net water quantity as well as changes in pH, dissolved oxygen, and other water quality parameters.

One consistent and interesting trend in the curves generated for Withdrawal Index (see Case study supplement A) is that the relationships between each ecological response variable and water withdrawals tended to be stronger with increasing basin area. There are a few possible reasons why the curves illustrated this trend. First, small streams generally have less predictable flow regimes and greater variation in flows (measured by % change) compared with larger rivers. It is possible that hydrologic alteration has a greater effect on biota adapted to a larger river environment where the flow regime is predictable relative to small streams with unpredictable flow regimes. Second, although the same macroinvertebrate sampling methods were used at all sites, it is likely that entire riffles were sampled in small streams due to the ease of wading across the channel. In larger rivers, it is possible that samples were concentrated at stream margins rather than the middle of the channel, sites that would be more susceptible to alterations in hydrology. Third, most of the larger river sites had larger values for Withdrawal Index and also likely had poor water quality compared with smaller streams; thus, multiple stressors at larger river sites may have resulted in poorer ecological condition. Finally, water withdrawals in small streams are often from groundwater pumping or a mixture of groundwater and surface water withdrawals, whereas water withdrawals in large rivers are almost exclusively from surface water. Thus, macroinvertebrates may respond more directly to surface water withdrawals, which represent a greater proportion of total flow at larger river sites.

In general, all of the models relating predictor variables (Withdrawal Index or Adjusted Withdrawal Index, pH, dissolved oxygen, and basin area) to ecological response variables had fairly low R^2 values, indicating that the models explained a low proportion of the variation in the data (between 3% and 24%). We would expect low R^2 values for most ecological studies that use pre-existing field data to examine correlations between ecological response and a small group of predictor variables. Two primary reasons are that (1) collection of field data and location of sampling sites were not designed for the purpose of examining effects of water withdrawals on macroinvertebrate metrics, and (2) there are many other factors that influence macroinvertebrate assemblages across the Susquehanna River basin that were not accounted for in our models, including watershed land use, point and non-point source pollution, water temperature, local geology and stream geomorphology, sediment dynamics, reservoir operations, and riparian conditions, among others. Although the R^2 values of the models in our analysis suggest that the models have low predictive ability, the fact that water withdrawals were consistently and significantly correlated with changes in the ecological response variables show that the models are useful for assessing risk of increasing water use. It may be useful to view the relationships between water withdrawals and ecological response in the context of water quality regulations; water quality is regulated even though these parameters may not always be exact predictors of ecological response. However, the risk of poor ecological condition under declining water quality conditions is well established.

Contrary to our expectations, partial R^2 values showed that Withdrawal Index was the most important predictor for the more standard macroinvertebrate metrics typically used in water quality monitoring, but not for the aquatic insect functional traits that had stronger theoretical relationships with changes in water quantity based on hypotheses developed from the literature (Table 3). One reason may be that although standard macroinvertebrate metric are typically used in water quality monitoring, these metrics generally measure response of macroinvertebrate assemblages to disturbance. In addition, we chose to examine aquatic insect functional traits that were hypothesized to change with disturbance at a site, and were not expected to solely respond to changes in water quantity. Thus, we would expect these metrics to also change in response to disturbances such as alterations in water quality conditions. When deciding which curves and ecological response variables to focus on for assessing risk of increasing water withdrawals, it may make sense to choose metrics that are most associated with Withdrawal Index (WI has highest partial R^2 in the model) and have the highest overall predictive ability (highest overall R^2). In our analysis, the metrics that met these two criteria included the modified Hilsenhoff Biotic Index (HBI; figure 1) and the number of EPT taxa (figure 2). However, it is still important to establish theoretical links between the ecological metric and water withdrawals. This relationship is clearer for the number of EPT taxa (a measure of sensitive taxa that is hypothesized to decrease with increasing disturbance) than HBI (a metric developed specifically to respond to organic pollution at a site, and is hypothesized to increase with increasing disturbance). However, a metric developed to indicate ecological response to organic pollution, such as HBI, can also be generally thought of as a metric of taxonomic tolerance to disturbance, and we would expect tolerant taxa to increase as water withdrawals increase.

There are three methods that may be used in future analyses to potentially increase the predictive capacity of the models we developed, or to construct different models with more predictive power: (1) develop a new dataset of hydrologic alteration that addresses some of the limitations inherent with using the WAST data, (2) develop stream classifications and construct separate models for each stream class, and (3) include additional covariates and interaction terms in the models. Data from WAST are excellent for calculating water withdrawal indices consistently for many sites basin- or state-wide. However, this dataset also has several limitations. First, hydrologic alterations (and potential ecological responses) associated with reservoir operations may be missed and/or may confound relationships between biological indicators and altered flow conditions. Second, because all upstream withdrawals (and discharges) are aggregated for each pour point, potential local impact of particularly large withdrawals (or discharges) may be missed. Third, water withdrawals are based on permitted withdrawals (or discharges) as well as estimated water use, and may not accurately reflect actual use at a pour point. Finally, WAST does not give seasonal estimates of withdrawals, and larger withdrawals during typical low flow periods would likely have stronger ecological effects than withdrawals at other times of the year.

As discussed earlier in this report, aquatic species found in different types of streams and rivers may respond differently to hydrologic alteration. We did not group data by stream class for this analysis, but this would be a useful next step that may provide new insights into responses of aquatic macroinvertebrates to water withdrawals. In addition, analysis by stream class may better illustrate the reasons why we observed stronger relationships between water withdrawals and macroinvertebrate metrics in larger basins. Finally, as mentioned above, aquatic macroinvertebrates respond to multiple physical, chemical, and biotic variables. Inclusion of other variables in our models may increase their predictive abilities. However, our analysis has shown that water withdrawals are an important and consistent predictor of aquatic macroinvertebrate metrics and functional traits, and this information should be useful in determining instream flow needs.

Conclusions

We conclude that the consistent, significant, and predictable relationships between our response variables and Withdrawal Index provide strong evidence that there is increasing risk of ecological alteration with increasing water withdrawals. However, low R^2 values indicate that the predictive ability of these models is poor. Thus, the models developed in this analysis do not provide the ability to predict specific ecological condition for a given value of Withdrawal Index, but do provide a general assessment of ecological risk with increasing water withdrawals. Our analysis also provides a starting point for future development of models with greater predictive value, which may be achieved by (1) developing a stronger hydrologic foundation that incorporates multiple flow components, seasonal changes in water use, and multiple drivers of hydrologic alteration, (2) implementation of additional ecological monitoring that is specifically designed to address hydrologic alteration by focusing on sampling sites that match sites with hydrologic data, are distributed across a range of hydrologic alteration, and are distributed across stream classes or streams and rivers with different sizes and physical and chemical characteristics, and (3) address multiple taxonomic groups.

Case study supplement A: Flow alteration – ecological response curves developed for Withdrawal Index

Figure 1. Taxonomic richness

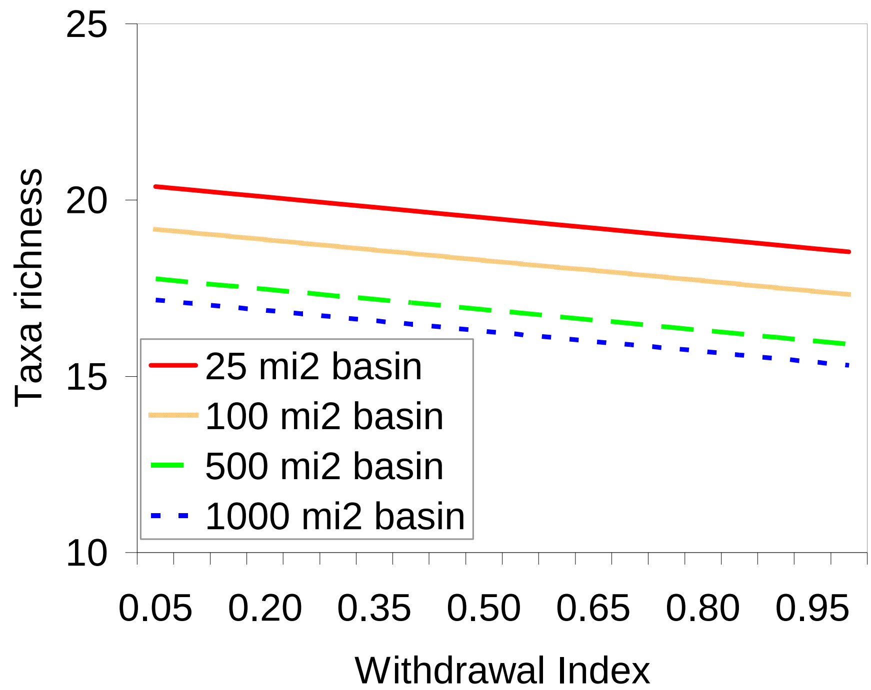


Figure 2. Modified Hilsenhoff Biotic Index (HBI)

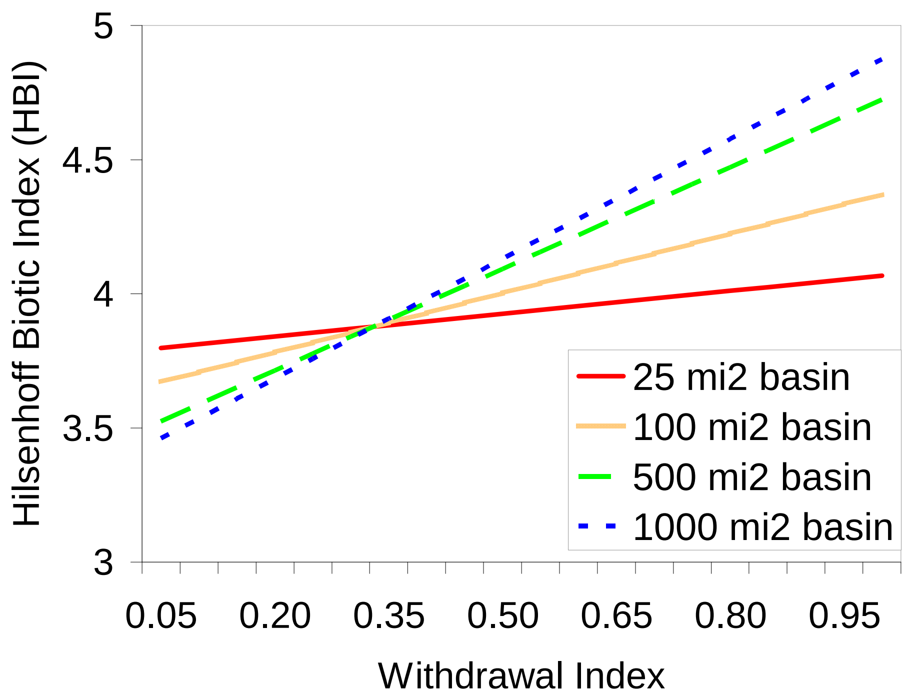


Figure 3. Percent Ephemeroptera

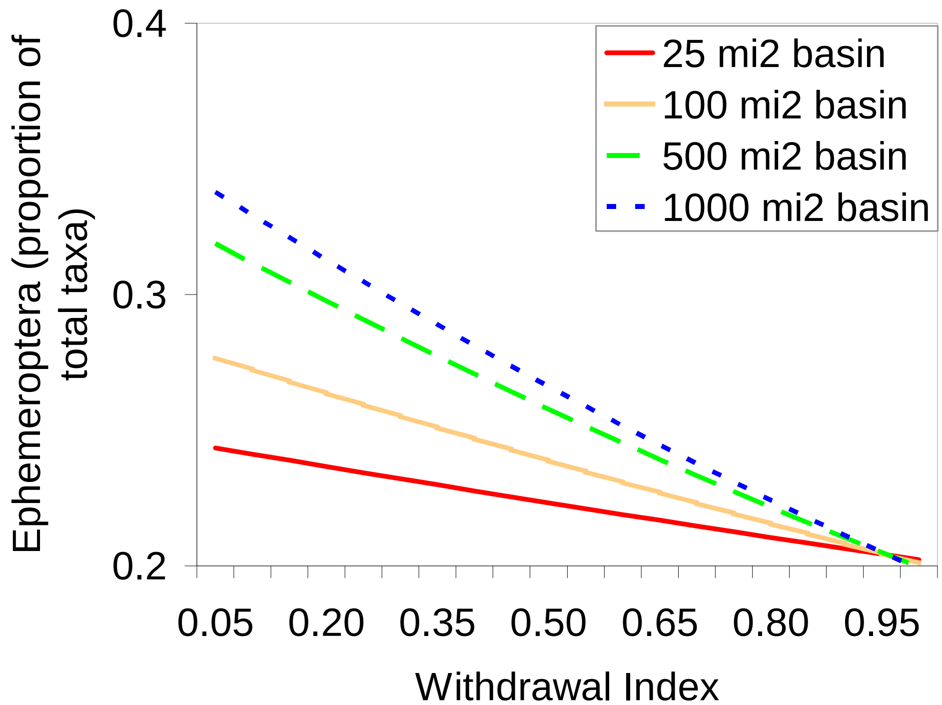


Figure 4. Percent Dominant Taxon

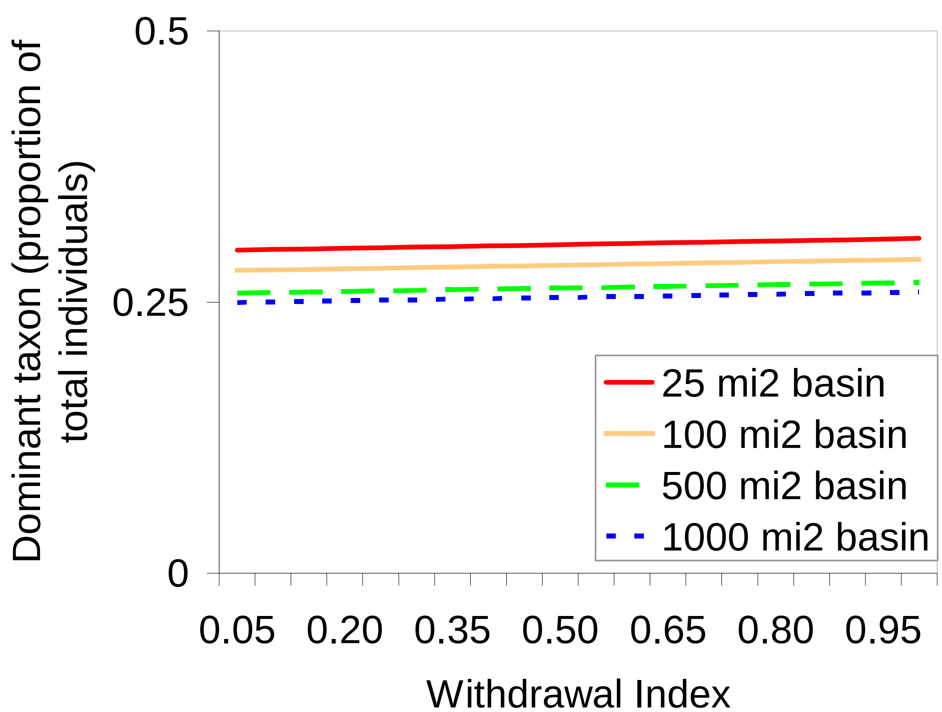


Figure 5. Number of Ephemeroptera, Plecoptera, and Trichoptera taxa

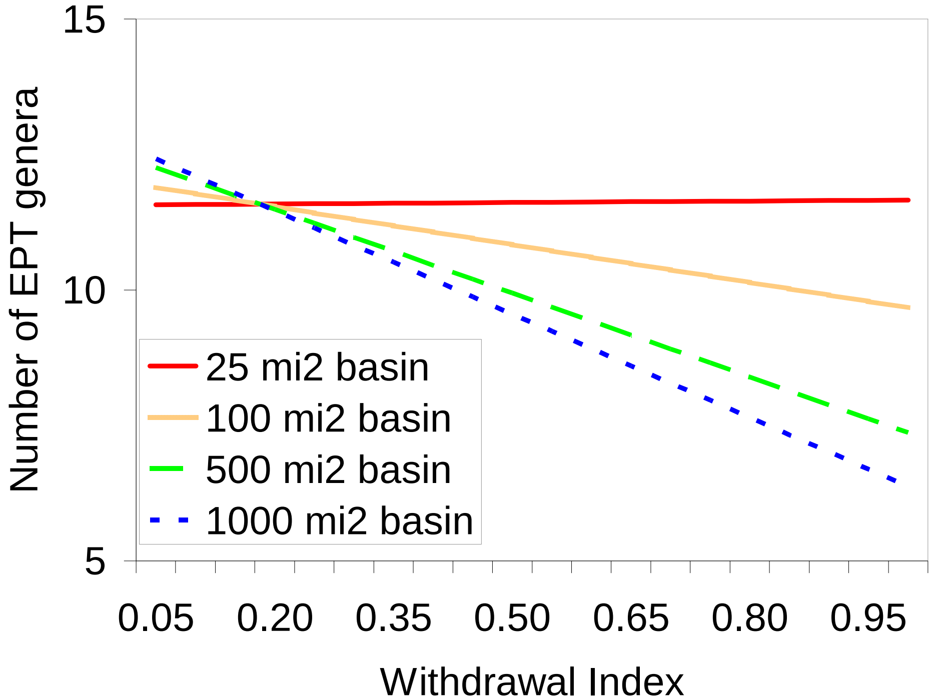


Figure 6. Percent Chironomidae

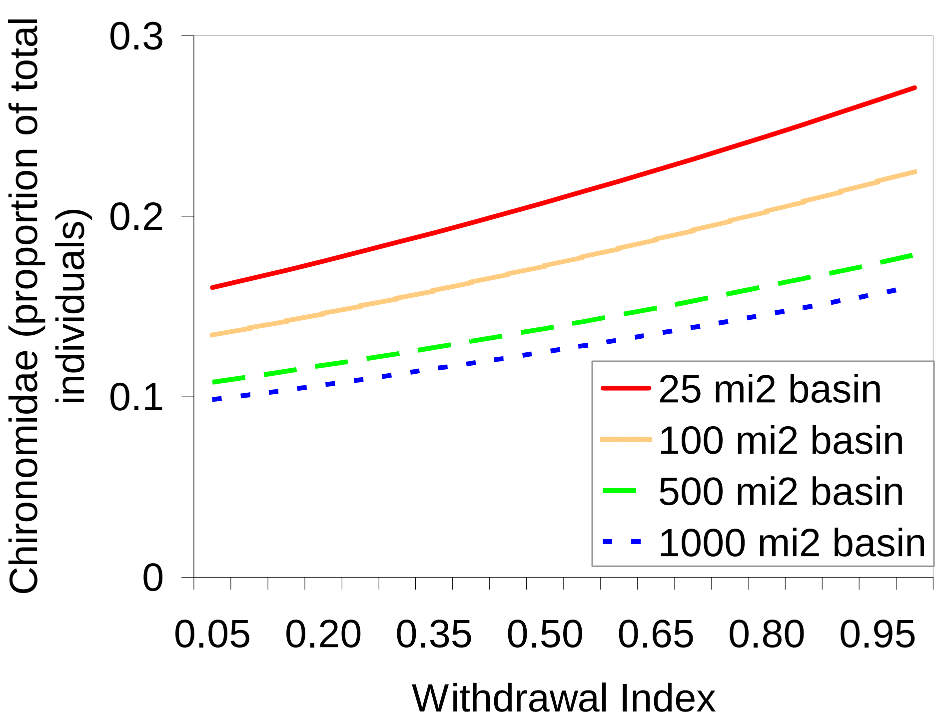


Figure 7. Shannon Wiener Diversity Index

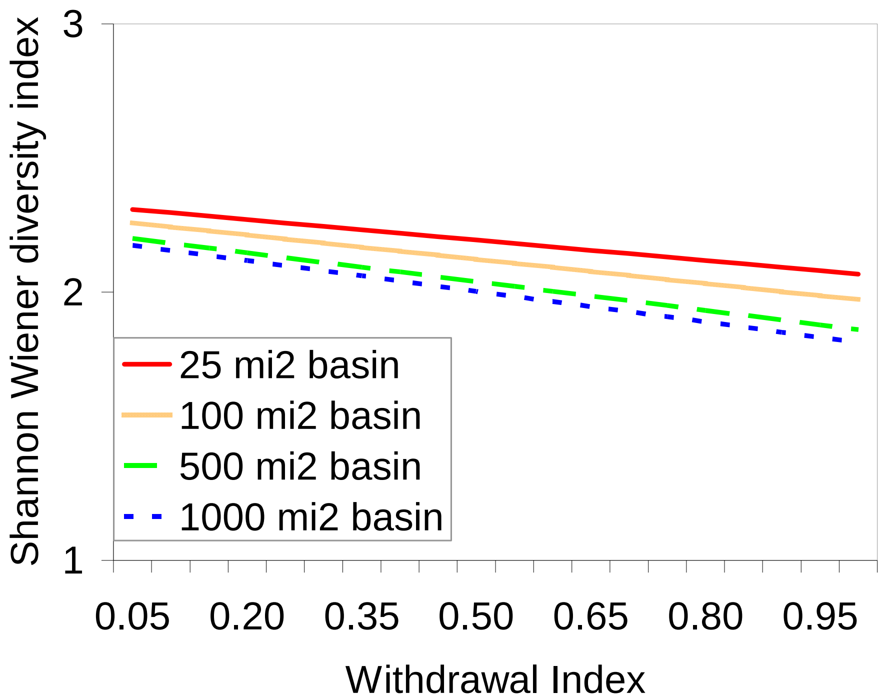


Figure 8. Proportion of aquatic insects abundant in the drift

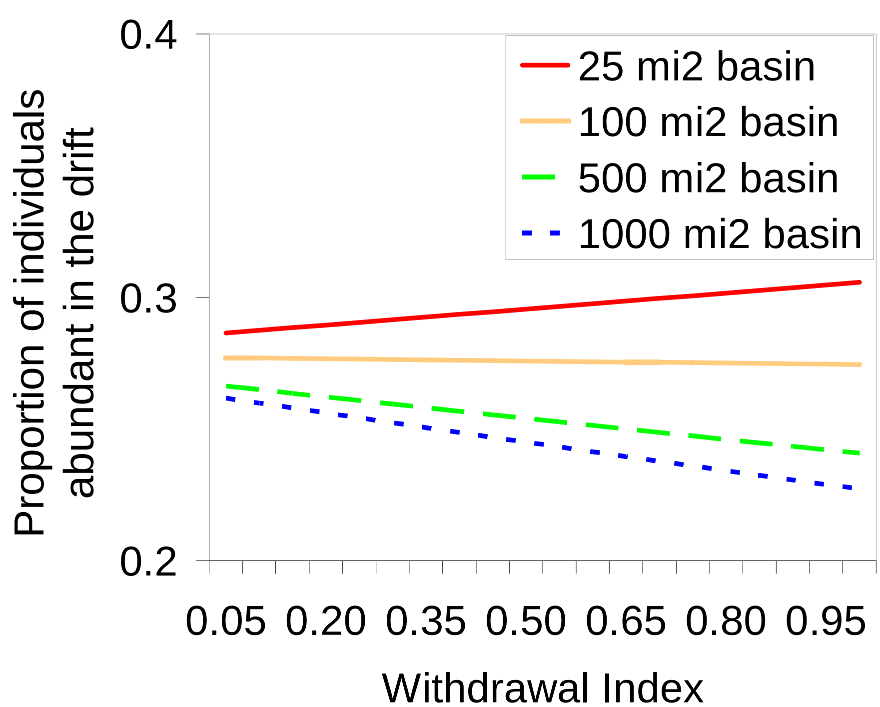


Figure 9. Proportion of aquatic insects with small size at maturity

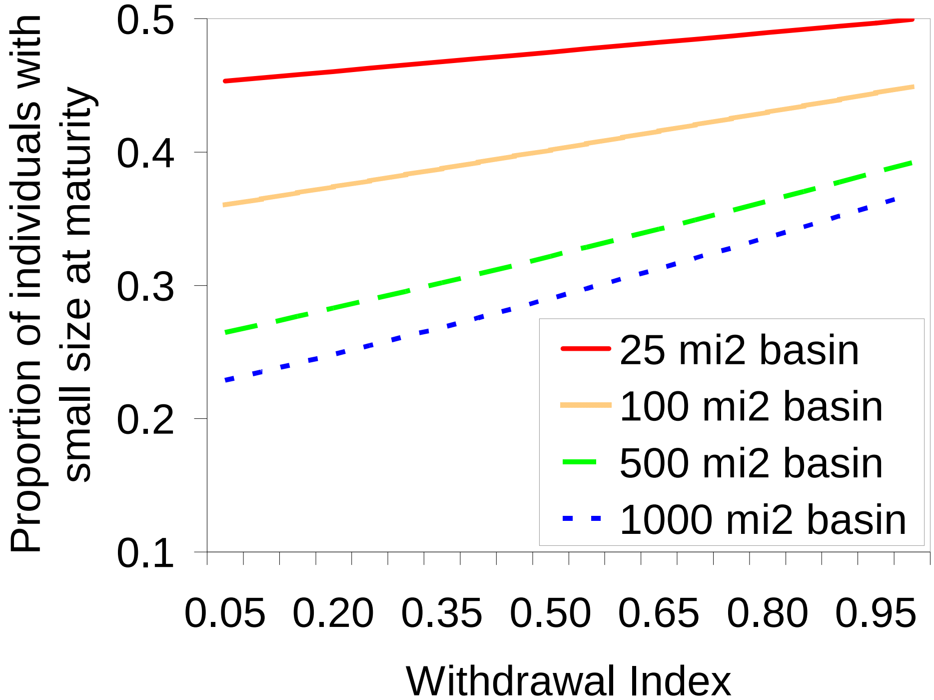


Figure 10. Proportion of aquatic insects with multiple generations per year (bi- or multi-voltine)

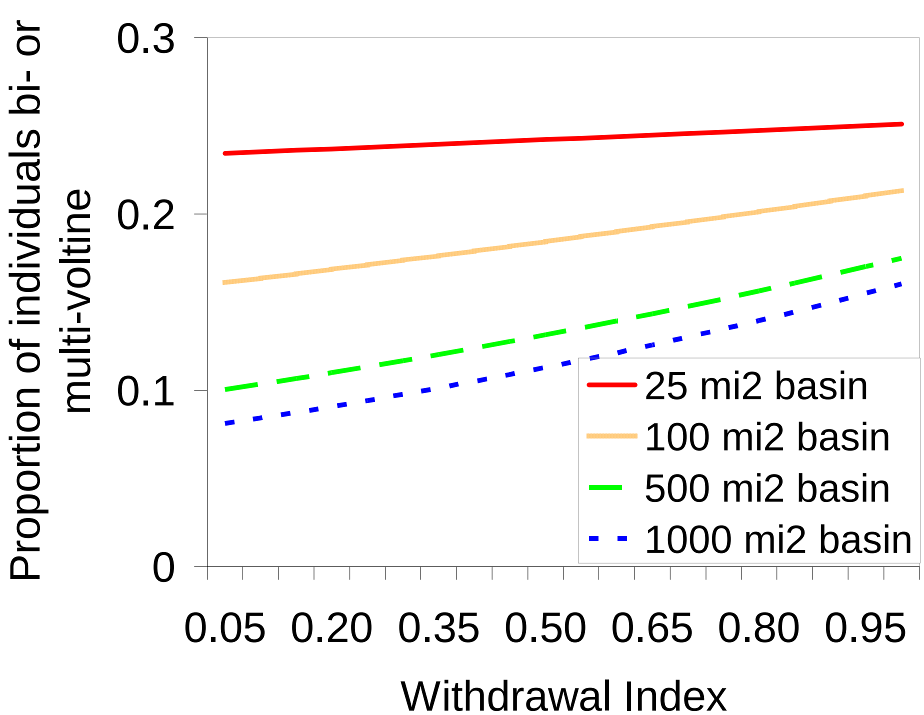


Figure 11. Proportion of aquatic insects that are generalist feeders (collector-gatherers)

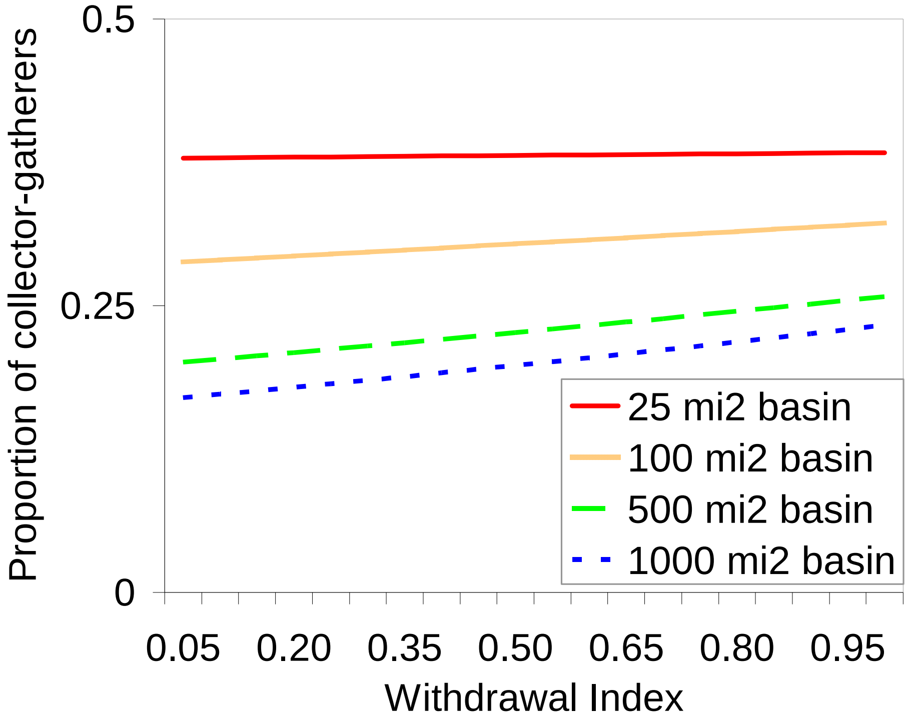


Figure 12. Proportion of aquatic insects with preference for cold water habitats (cold-stenothermal or cool-eurythermal)

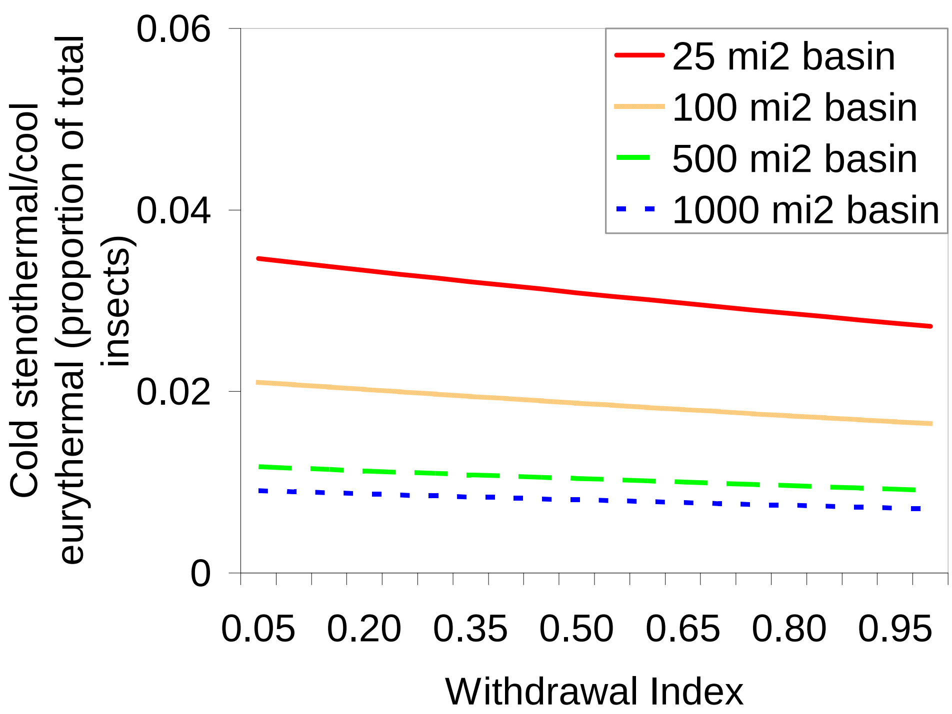
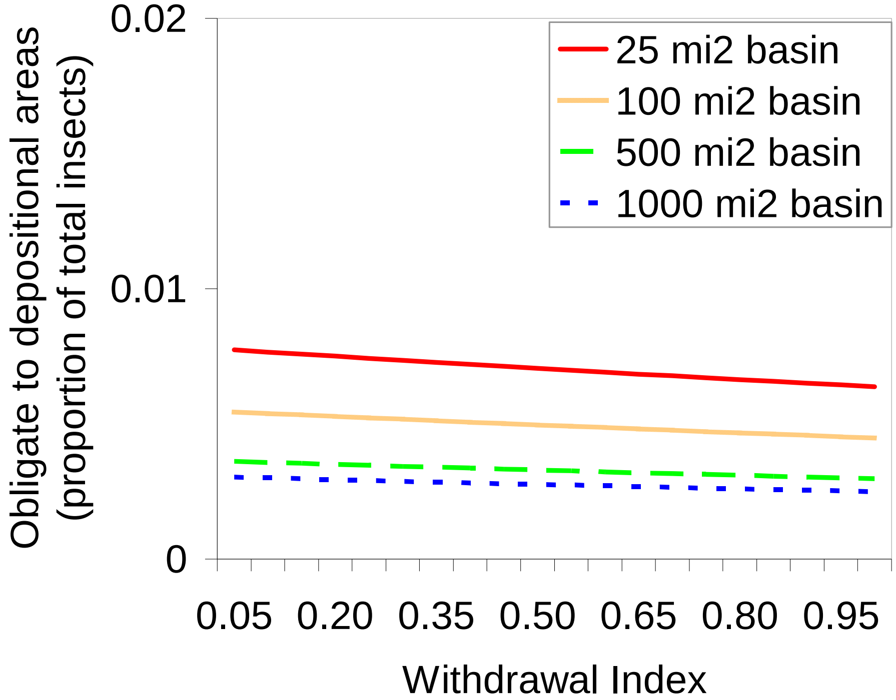


Figure 13. Proportion of aquatic insects obligate to depositional areas



Case study supplement B: Aquatic insect trait hypotheses: linking water withdrawals and diversions with trait responses

1. Direct responses of water withdrawals, diversions, and/or extended periods of low and extreme low flows (either a direct response or no intermediate processes were identified)
    1.1. Decrease in total number of invertebrate taxa below withdrawal/diversion sites (Rader and Belish 1999, McKay and King 2006)
    1.2. Decreased diversity (number of taxa) of grazers and shredders (McKay and King 2006). This is equivalent to a decrease in specialist feeders.
    1.3. Increase in the abundance of individuals with small body size at maturity (Rader and Belish 1999)
    1.4. Decrease in the abundance and number of taxa that are rare in the drift (species rare in the drift could not replenish lost populations when more favorable flow conditions were present - Rader and Belish 1999)
2. Decrease in wetted area/habitat - indirect response (includes increase in the % area of shallow habitats - Richards et al. 1997)
    2.1. Increase in abundance and number of taxa that are multivoltine (Richards et al. 1997)
    2.2. Increase in abundance and number of taxa with small body size at maturity (Richards et al. 1997)
    2.3. Decrease in abundance and number of taxa that are scrapers or shredders (Richards et al. 1997)
    2.4. Increase in abundance and number of taxa of swimmers and climbers (increased mobility and ability to seek out refugia; Richards et al. 1997)
    2.5. Increase in abundance and number of taxa that are obligate depositional (Richards et al. 1997)
    2.6. Increase in abundance and number of taxa that are free-ranging (not sessile or case-building; Richards et al. 1997)
3. Decrease in water velocity - indirect response (Lake 2003)
    3.1. Decrease in abundance and number of taxa that are rheophilic
4. Increase in water temperature - indirect response (Lake 2003)
    4.1. Increase in abundance and number of taxa with high thermal tolerance (eurythermal); decrease in abundance and number of taxa that are stenothermal
5. Decrease in dissolved oxygen availability – indirect response (McKay and King 2006)
6. Increase in the percent of fines (a by-product of decreased water velocity)
    6.1. Increase in the abundance and number of taxa that are burrowers (Richards et al. 1997)
    6.2. Decrease in the abundance and number of taxa that are obligate erosional (Richards et al. 1997)
7. Increase in ephemeral conditions
    7.1. Increase in the abundance of individuals with small body size at maturity (Rader and Belish 1999)
    7.2. Increase in the abundance of individuals that are multivoltine
    7.3. Increase in diversity and abundance of highly mobile taxa (Boulton 2003).
    7.4. Decrease in the abundance and number of taxa that are rare in the drift (species rare in the drift could not replenish lost populations when more favorable flow conditions were present - Rader and Belish 1999)
    7.5. Increase in desiccation-adapted taxa (Boulton 2003)

CASE STUDY 6: CONNECTICUT DRAFT STREAMFLOW PROTECTION REGULATIONS – FRAMEWORK AND STANDARD DEVELOPMENT

In the fall of 2005, the Connecticut State Legislature passed legislation requiring that the Connecticut Department of Environmental Protection (CT DEP) adopt flow protection regulations that apply to all rivers in the state by the end of 2006. Key provisions of this legislation direct CT DEP's regulations to "preserve and protect the natural aquatic life" while being "based, to the maximum extent practicable, on natural variation of flows and water levels". These regulations are to be based on "the best available science". Balancing language was included to make sure the regulations provide for the needs and requirements of an inclusive list of human uses. (Connecticut Public Act 05-142, 2005)

To accomplish this work, stakeholder and scientific committees were set up in early 2006, including a technical advisory committee and a policy advisory committee, each of which have been meeting for over two years (the deadline has been extended until at least summer 2008). The draft regulation described in this case study is primarily a result of the efforts of these committees and the CT DEP staff members that oversee the committee process.

Key provisions of this draft regulation as of May 2008 include:

1) Use of a tiered goal system for river segments in the state with associated narrative standards
2) Reliance upon a "hydrologic foundation" which can simulate unimpacted baseline and current condition daily flows
3) Use of six "bioperiods": groups of months or individual months that have common ecological relevance
4) Use of a direct withdrawal standard expressed as a percentage of annual Q99 (the 99th percent exceedence flow)
5) Default reservoir release standards based on estimated natural hydrology
6) Allowance for the use of a site-specific study habitat or biological study to develop site-specific reservoir release regime
7) Use of a subset of hydrologic statistics to estimate the degree of hydrologic alteration associated with a reservoir-specific release regime that can be compared with streamflow compliance criteria

Narrative standards are described in the draft regulation for river or streams that are "Classes" 1-4. Class 1 rivers have the highest level of protection while Class 4 rivers are substantially altered and may require restoration. The status goal for a particular river segment will be determined based on a range of local factors (e.g., degree of land use intensity, existing withdrawal levels, presence of trout resources) and will be reviewed as part of a public process. Key language in the narrative standards centers around the degree to which the river of interest: 1) can support an aquatic biological community similar to that typically present in free-flowing river; and 2) is similar in hydrologic characteristics to a free-flowing river.

Simulation of baseline, unimpacted conditions is seen as critical to implementation of the regulation. The State of Connecticut can currently simulate some natural flow statistics statewide, but is seeking the capacity to simulate baseline and current/future condition daily flow time series information using the same methodology as the Massachusetts Safe Yield Estimator (see report text for review). Effective estimation of a range of natural flow statistics is particularly important as it serves as a basis for the environmental standards set up in the draft rule.

The environmental flow standards set up in the draft regulation are a result of a consensus workgroup process in the technical advisory committee made up of scientists and water managers. Given the lack of flow-ecology relationship data from Connecticut that could be extrapolated statewide, the technical committee gathered the best available information from the literature—much of which is reviewed as part of this Growing Greener Grant report. The result is a set of standards that reflect best professional judgment of those on the technical committee along with a degree of balancing as required by the statute.

The draft abstraction (direct withdrawal, no storage) standard proposed by Connecticut requires estimation of the annual Q99 at any location of interest in the state (e.g., an existing or future withdrawal location). For the summer "rearing and growth" bioperiod, July through October, the cumulative withdrawal through direct river intake or groundwater pumping can be up to 5% of Q99 for Class 1 rivers, 25% of Q99 for Class 2, and 50% of Q99 for Class 3. Allowances for other bioperiods are scaled using a factor that reflects the difference between the relevant bioperiod Q99 and the summer "rearing and growth" bioperiod Q99. The basis for these standards is primarily the study of withdrawal index in relation to fish communities in the Georgia Piedmont (Freeman and Marcinek 2006). This work identified clear negative impacts to the richness of fluvial specialist fish species when approximately 50% of 7Q10 was permitted for withdrawal. The Connecticut draft standards use a comparable low flow statistics and are centered about the Class 3 summer standard (50% of Q99). The standards also rely on work from the United Kingdom (see case study) and an iterative analysis of the degree of hydrologic alteration that results from these allowances akin to Acreman (2007).

Figure 1. Sample direct withdrawal standard implementation table. The values in the Class 1, 2, and 3 columns are the decimal value of Q99 flow that can be allowably withdrawn under the presumptive standard. For example, half (0.5) of Q99 can be cumulatively withdrawn in the summer rearing and growth bioperiod on a Class 3 river.

Any River

Month	BioPeriod	HydroPeriod	cfsm	Scale factor	99th Percentile Flow Multiplier Class 1	Class 2	Class 3
Dec Jan Feb	Overwinter	High-Medium Flow	1.54	9	0.05	2.14	4.28
Mar Apr	Habitat Forming	High Flow	2.57	14	0.05	3.57	7.14
May	Clupeid Spawning	High-Medium Flow	1.54	9	0.05	2.14	4.28
Jun	Resident Spawing	Low-Medium Flow	0.70	4	0.05	0.97	1.94
Jul Aug Sep Oct	Rearing & Growth	Low Flow	**0.18**	1	0.05	0.25	0.50
Nov	Salmonid Spawning	Low-Medium Flow	0.70	4	0.05	0.97	1.94

The use of estimated Q99 and bioperiod scaling factors as a basis for the standard allows for differentiation between rivers and streams with high baseflow (and therefore more capability to absorb withdrawals) and those with low baseflow. In this way, the abstraction standard takes into account river low flow hydrologic differences in a manner somewhat analogous to river hydrologic classification. On the other hand, it adds complexity and makes the standard more difficult to explain to users and other stakeholders.

Another feature of the draft regulation is a Class 3 “presumptive” reservoir release rule which can be used in lieu of a site-specific reservoir release regime (see below). This presumptive release rule will likely require varying releases based on:

1) bioperiod
2) antecendent inflow (i.e., higher release requirement when the previous two weeks’ flows have been high, lower when they have been low)
3) reservoir storage level

As with the direct withdrawal standard, the actual release values will be based on estimated natural flow statistics at the dam location, allowing for local hydrological conditions to be taken into account. In addition, during long periods of low flow, a small one day pulse will likely be required. Finally, for reservoirs that are large relative to their watershed size, a spring high flow release may also be required. Significant difficulties relate to defining a presumptive rule that can be implemented on all sizes of reservoirs without significantly affecting “safe yield” (i.e., the amount of water that can be reliable delivered without failure across the period of record). The basis for this presumptive standard is a balancing between impacts to safe yield and impacts to natural variability in flow regime as represented by flow statistics. This balance is being worked out through an iterative process of evaluating how a “typical” water supply reservoir may behave under a range of environmentally protective release rules.

To account for cases in which a reservoir operator can not, or chooses not, to use the presumptive release rule, the draft regulation allows for use of a site-specific biological or habitat study to develop a proposed reservoir release regime. A set of flow statistics and associated compliance criteria in the draft regulation are designed to be used to assess this alternative release approach. To define "key flow statistics" under the regulation the technical committee in Connecticut began with the hydrologic statistics that were found non-redundant, and representative of all flow components for "perennial flashy or runoff" rivers representative of Connecticut hydrology in the publication "Redundancy and the Choice of Hydrologic Indices for Characterizing Streamflow Regimes" (Olden and Poff 2003). This set of twenty-seven statistics were then modified based on expert opinion of the technical committee to ensure they were comprehensible to managers and the regulated public, to ensure their behavior was explainable, and to link them to the bioperiods. The draft set of key statistics includes:

1) annual runoff
2) base flow index
3) monthly average streamflows for January, March, May, June, August, and November (representing each of the bioperiods)
4) low flow pulse count
5) variability in low pulse duration
6) date of annual minimum streamflow
7) high flood pulse count
8) high streamflow duration and magnitude
9) variability in streamflow reversals

These statistics would be used to examine estimated baseline flows in comparison with the downstream consequences of the proposed release rule (i.e., releases and spills). For Class 1 river systems, only a 5% change in all flow statistics in comparison to baseline would be allowed for compliance. An up to 40% change in all flow statistics would be allowed for Class 2 and an up to 80% change would be allowed for Class 3 compliance. As written in the draft regulation, further departure in even one of the statistics would violate the assessment criteria. This approach is heavily reliant on development of a hydrologic foundation by the CT DEP or the user. The primary basis for the hydrologic assessment criteria is work from the United Kingdom as documented in Acreman (2007).

Due to the initial tight legislative deadline and limited state data availability, the Connecticut regulatory effort is basing environmental flow standards on flow-ecology relationships and environmental flow alteration criteria that are from outside of the state and even outside of the country. In this way, it is combining a more traditional hydrological standard-setting approach with new information on flow-ecology relationships that have been peer reviewed and published in the scientific literature.

Although local expert, technical workgroup consensus has played a significant role in selecting and interpreting what has been considered the best available science, the lack of local empirical relationships may weaken the regulations if subject to a future challenge. For Pennsylvania, the framework of the Connecticut draft streamflow regulation should be instructive as an example of how a statewide regulation can be structured that takes into account local hydrologic conditions, the differing hydrologic impacts associated with active storage and direct withdrawal, the need for balancing, and ease of implementation.

LITERATURE CITED

Acreman, M.C. 2005. Linking science and decision-making: features and experience from environmental river flow setting. Environmental Modeling and Software **20**:99-109.

Acreman, M.C., M.J. Dunbar, J. Hannaford, O.M. Bragg, A.R. Black, J.S. Rowan, and J King. 2006. Development of environmental standards (water resources): final report. Project WFD48. Scotland & Northern Ireland Forum for Environmental Research (SNIFFER), Edinburgh, UK.

Acreman, M.C. 2007. Guidance on environmental flow releases from impoundments to implement the water framework directive: final report. Project WFD82. Scotland & Northern Ireland Forum for Environmental Research (SNIFFER), Edinburgh, UK.

Allan, J. D., M. J. Wiley, P. Webb, and N. L. Poff. 2000. Flow alteration in Great Lakes Basin Rivers: extent and effects. Proposal WR553. Great Lakes Protection Fund, Chicago, IL

Annear, T., I. Chisolm, H. Beecher, A. Locke, P. Aarrestad, C. Coomer, C. Estes, J. Hunt, R. Jacobson, G. Jöbsis, J. Kauffman, J. Marshall, K. Mayes, G. Smith, and R. Wentworth. 2004. Instream flows for riverine resource stewardship (revised edition). Instream Flow Council, Cheyenne, WY.

Archfield, S. A., R. M. Vogel, and S. B. Brandt. 2007. Estimation of flow-duration curves at ungaged sites in southern New England. Proceedings of the American Society of Civil Engineers World Water and Environmental Resources Congress, Tampa, FL.

Archfield, S. A. and R. M. Vogel. 2008. A decision-support system to assess surface-water resources in Massachusetts. Proceedings of the American Society of Civil Engineers World Water and Environmental Resources Congress, Honolulu, HI.

Arthington, A.H., S. E. Bunn, B. J. Pusey, D. R. Blühdorn, J. M. King, J. A. Day, R. E. Tharme, and J. H. O'Keeffe. 1992. Development of an holistic approach for assessing environmental flow requirements of riverine ecosystems. Pages 69-76 in: Proceedings of an international seminar and workshop on water allocation for the environment. John J. Pigram and Bruce P. Hooper (Editors). The Centre for Water Policy Research, Armidale, Australia.

Arthington, A.H., S. O. Brizga, and M. J. Kennard. 1998. Comparative evaluation of environmental flow assessment techniques: best practice framework. LWRRDC Occasional Paper 25/98. Accessed March 19, 2007 at http://www.lwa.gov.au/downloads/publications_pdf/pr980305.pdf

Arthington, A., S. E. Bunn, N. L. Poff, and R. J. Naiman. 2006. The challenge of providing environmental flow rules to sustain river ecosystems. Ecological Applications **16:**1311-1318.

Bain, M. and M. Meixler. 2000. Defining a target fish community for planning and evaluating enhancement on the Quinebaug River in Massachusetts and Connecticut. New York Cooperative Fish and Wildlife Research Unit, Cornell University, Ithaca, NY.

Barbour, M. T., J. Gerritsen, B. D. Snyder, and J. B. Stribling. 1999. Rapid Bioassessment Protocols for use in streams and wadeable rivers: periphyton, benthic macroinvertebrates and fish, Second Edition. EPA 841-B-99-002. U.S. Environmental Protection Agency, Office of Water, Washington, D.C.

Bencala, K. E., D. B. Hamilton, and J. H. Petersen. 2006. Science for maintaining riverine ecosystems: actions for the USGS identified in the workshop “Analysis of flow and habitat for aquatic communities”. Open File Report 2006-1256. U. S. Geological Survey, Washington, D.C.

Black, A. R., J. S. Rowan, R. W. Duck, O. M. Bragg, and B. E. Clelland, 2005. DHRAM: a method for classifying river flow regime alterations for the EU Water Framework Directive. Aquatic Conservation: Marine & Freshwater Ecosystems **15**:427-446.

Boulton, A. J. 2003. Parallels and contrasts in the effects of drought on stream macroinvertebrate assemblages. Freshwater Biology **48**:1173-1185.

Brizga, S.O., A. H. Arthington, B. J. Pusey, M. J. Kennard, S. J. Mackay, G. L. Werren, N. M. Craigie, and S. J. Choy. 2002. Benchmarking, a ‘top-down’ methodology for assessing environmental flows in Australian rivers. Proceedings of the International Conference on Environmental Flows for Rivers. Southern Waters Consulting, Cape Town, South Africa.

Bunn, S. E. and A. H. Arthington. 2002. Basic principles and ecological consequences of altered flow regimes for aquatic biodiversity. Environmental Management **30**:492–507.

Cassin, J., R. Fuerstenberg, L. Tear, K. Whiting, D. St. John, B. Murray, and B. Burkey. 2005. Development of hydrological and biological indicators of flow alteration in Puget Sound lowland streams. King County Water and Land Resources Division, Seattle, WA.

Davies, S.P, and S. K. Jackson. 2006. The biological condition gradient: a descriptive model for interpreting change in aquatic ecosystems. Ecological Applications **16**:1251–1266.

Denslinger, T. L., W. A. Gast, J. J. Hauenstein, D. W. Heicher, J. Henriksen, D. R. Jackson, G. J. Lazorchick, J. E. McSparran, T. W. Stoe, and L. M. Young. 1998. Instream flow studies: Pennsylvania and Maryland. Susquehanna River Basin Commission, Harrisburg, PA.

Dyson, M., G. Bergkamp, and J. Scanlon. 2003. Flow: The Essentials of Environmental Flow. International Union for Conservation of Nature and Natural Resources (IUCN), Gland, Switzerland.

Edwards, R. E. and R. D. Pody. 2005. Northern Lancaster County groundwater study: A resource evaluation of the Manheim-Lititz and Ephrata area groundwater basins. Susquehanna

River Basin Commission, Harrisburg, PA. Accessed March 19, 2007 at http://www.srbc.net/waterbudgetstudy.htm.

Eng, K., G. D. Tasker, and P. C. D. Milly. 2005. An analysis of region-of-influence methods for flood regionalization in the Gulf-Atlantic Rolling Plains. Journal of the American Water Resources Association **41**:135-143.

Extence, C. A., D. M. Balbi, and R. B. Chadd. 1999. River flow indexing using British benthic macroinvertebrates: a framework for setting hydro-ecological objectives. Regulated Rivers: Research & Management **15**:543-574.

Fennessey, N. M. 1994. A hydro-climatological model of daily streamflow for the northeast United States. Ph.D. dissertation, Tufts University, Medford, MA.

Freeman, M. C., and P. A. Marcinek. 2006. Fish assemblage responses to water withdrawals and water supply reservoirs in Piedmont streams. Environmental Management **38**:435-450.

Frissell, C. A., W. J. Liss, C. E. Warren, and M. D. Hurley. 1986. A hierarchical framework for stream habitat classification: viewing streams in a watershed context. Environmental Management **10**:199-214

Gordon, N. D., T. A. McMahon, and B. L. Finlayson. 1992. Stream Hydrology: An Introduction for Ecologists. John Wiley and Sons, New York.

Hantush, M. M. and L. Kalin. 2006. Impact of Urbanization on the Hydrology of Pocono Creek Watershed: A Model Study. Office of Research and Development, U.S. Environmental Protection Agency, Cincinnati, Ohio.

Harbaugh, A. W., E. R. Banta, M. C. Hill, and M. G. McDonald. 2000. MODFLOW-2000, the U.S. Geological Survey modular ground-water model—user guide to modularization concepts and the ground-water flow process. Open-File Report 00-92. U.S. Geological Survey, Washington, D.C.

Harris, N. M., A. M. Gurnell, D. M. Hannah, G. E. Petts. 2000. Classification of river regimes: a context for hydroecology. Hydrological Processes **14**:2831-2848

Henriksen, J. A., J. Heasley, J. G. Kennen, and S. Nieswand. 2006. Users' manual for the Hydroecological Integrity Assessment Process software (including the New Jersey Assessment Tools). Open-File Report 2006-1093. Fort Collins Science Center, U.S. Geological Survey, Fort Collins, CO.

Hersh, E. S. and D. R. Maidment. 2006. Assessment of Hydrologic Alteration Software: Final Report. Center for Research in Water Resources, University of Texas, Austin, TX.

Higgins, J. V., M. T. Bryer, M. L. Khoury, and T. W. Fitzhugh. 2005. A freshwater classification approach for biodiversity conservation planning. Conservation Biology **19**:1-14.

Holmes, N. T. H., J. P. Boon, and T. A. Rowell. 1998. A revised classification system for British rivers based on their aquatic plant communities. Aquatic Conservation: Marine and Freshwater Ecosystems **8**:555-578

Hunt, B. 1999. Unsteady stream depletion from ground water pumping. Ground Water **37**:98-102.

Jenkins, C. T. 1968. Computation of rate and volume of stream depletion by wells. Techniques of Water-Resources Investigations. U. S. Geological Survey, Washington, D.C.

Kennen, J. G. and M. A. Ayers. 2002. Relation of environmental characteristics to the composition of aquatic assemblages along a gradient of urban land use in New Jersey 1996-98. Water Resources Investigations Report 02-4069. U. S. Geological Survey, West Trenton, NJ.

Kennen, J. G., J. A. Henriksen, and S. P. Nieswand. 2007. Development of the hydroecological integrity assessment process for determining environmental flows for New Jersey streams. Scientific Investigations Report 2007-5206. U. S. Geological Survey, West Trenton, NJ.

Kennen, J. G., L. J. Kauffman, M. A. Ayers, D. M. Wolock, and S. J. Colarullo. 2008. Use of an integrated flow model to estimate ecologically relevant hydrologic characteristics at stream biomonitoring sites. Ecological Modeling **211**:57-76.

King, J. M. and D. Louw. 1998. Instream flow assessments for regulated rivers in South Africa: using the building block methodology. Aquatic Ecosystem Health and Management **1**:109-124.

King, J. M., R. E. Tharme, and M. S. de Villiers (editors). 2000. Environmental flow assessments or rivers: manual for the Building Block Methodology. Water Research Commission Report TT 131/00, Pretoria, South Africa.

Koltun, G.F. 2003. Techniques for Estimating Flood-Peak Discharges of Rural, Unregulated Streams in Ohio (second edition). Water Resources Investigation Report 03-4164. U. S. Geological Survey, Columbus, OH.

Lake, P. S. 2003. Ecological effects of perturbation by drought in flowing waters. Freshwater Biology **48**:1161-1172.

Leavesley, G.H., R. W. Lichty, B. M. Troutman, and L. G. Saindon. 1983. Precipitation-Runoff Modeling System: User's Manual. Water-Resources Investigations Report 83-4238, U.S. Geological Survey, Washington, D.C.

Mathews, R. and B. D. Richter. 2007. Application of the Indicators of Hydrologic Alteration Software in Environmental Flow Setting. Journal of the American Water Resources Association **43**:1400-1413.

McKay, S. F., and A. J. King. 2006. Potential ecological effects of water extraction in small, unregulated streams. River Research and Applications **22**:1023-1037.

Michigan Groundwater Conservation Advisory Council. 2007. Final report to the Michigan Legislature in response to Public Act 34. Lansing, MI. Accessed June 16, 2008 at http://www.michigan.gov/documents/deq/Groundwater_report_206809_7.pdf

Monk, W. A., P. J. Wood, D. M., Hannah, D. A. Wilson. 2007. Selection of river flow indices for the assessment of hydroecological change. River Research and Applications **23**:113-122.

National Fish Habitat Science and Data Committee (D. Beard and G. Whelan, Co-chairs). 2006. A Framework for Assessing the Nation's Fish Habitat. Accessed June 16, 2008 at http://fishhabitat.org/images/documents/science/National_Fish_Habitat_Initiative-Science_Report.pdf

National Research Council. 2005. The Science of Instream Flows: A Review of the Texas Instream Flow Program. The National Academies Press, Washington D.C.

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

Poff, N. L. 1996. A hydrogeography of unregulated streams in the United States and an examination of scale-dependence in some hydrologic descriptors. Freshwater Biology **36**:71-91

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Prestegaard, B. D. Richter, R. E. Sparks, and J. C. Stromberg. 1997. The natural flow regime: a paradigm for river conservation and restoration. BioScience **47**:769–784.

Poff, N. L., J. D. Olden, N. K. M. Vieira, D. S. Finn, M. P. Simmons, and B. C. Kondratieff. 2006. Functional trait niches of North American lotic insects: traits-based ecological applications in light of phylogenetic relationships. Journal of the North American Benthological Society **25**:730-755.

Poff, N. L. and J. V. Ward. 1989. Implications of streamflow variability and predictability for lotic community structure: a regional analysis of streamflow patterns. Canadian Journal of Aquatic Sciences **46**:1805-1818.

Poff, N. L., B. Richter, A. H. Arthington, S. E. Bunn, R. J. Naiman, E. Kendy, M. Acreman, C. Apse, B. P. Bledsoe, M. Freeman, J. Henriksen, R. B. Jacobson, J. Kennen, D. M. Merritt, J. O'Keeffe, J. Olden, K. Rogers, R. E. Tharme, and A. Warner. In review. The Ecological Limits of Hydrologic Alteration: A framework for developing regional environmental flow standards. Freshwater Biology.

Postel, S. and B. Richter. 2003. Rivers for Life: Managing Water for People and Nature. Island Press, Washington, D.C.

Pusey, B. J., M. J. Kennard, J. L. Stein, J. D. Olden, S. Mackay, M. F. Hutchinson, and F. Sheldon. In review. Ecohydrological regionalization of Australia: a tool for management and science. Land & Water Australia.

Rader, R. B. and T. A. Belish. 1999. Influence of mild to severe flow alterations on invertebrates in three mountain streams. Regulated Rivers: Research & Management **15**:353-363.

Richards, C., R. J. Haro, L. B. Johnson, and G. E. Host. 1997. Catchment and reach-scale properties as indicators of macroinvertebrate species traits. Freshwater Biology **37**:219-230.

Richter, B. D., J. V. Baumgartner, J. Powell and D. P. Braun. 1996. A method for assessing hydrologic alteration within ecosystems. Conservation Biology **10**:1163-1174.

Richter, B. D., J. V. Baumgartner, R. Wigington and D. P. Braun. 1997. How much water does a river need? Freshwater Biology **37**:231-249.

Richter, B. D., A. T. Warner, J. L. Meyer, and K. Lutz. 2006. A collaborative and adaptive process for developing environmental flow recommendations. River Research and Applications **22**:297-318.

Ries, K.G., III, P. A. Steeves, J. D. Coles, A. H. Rea, and D. W. Stewart. 2004. StreamStats: a U.S. Geological Survey web application for stream information. Accessed February 3, 2007 at http://water.usgs.gov/osw/streamstats/.

Sanborn, S. C. and B. P. Bledsoe. 2006. Predicting streamflow regime metrics for ungauged streams in Colorado, Washington, and Oregon. Journal of Hydrology **325**:241-261.

SAS Institute Inc. 1999. SAS OnlineDoc®, version 8.0. SAS Institute Inc., Cary, NC.

Seelbach, P. W., R. J. Stevenson, K. E. Wehrly, E. Baker, J. Lyons, D. Austen, A. Holtrop, M. DePhilip, C. Dolan, D. Day, L. Wang, B. Pijanowski, T. G. Zorn, J. D. Allan, and M. J. Wiley. 2002. Ecological classification of rivers for environmental assessment: demonstration, validation, and application to regional risk assessment across Illinois, Michigan, and Wisconsin. Grant Number R830596. U.S. Environmental Protection Agency, Washington, D.C.

Sharma, A. D., G. Tarboton, and U. Lall. 1997. Streamflow simulation: A nonparametric approach. Water Resources Research **33**:291-308.

Sloto, R. A. 2004. Geohydrology of the French Creek Basin and simulated effects of drought and ground-water withdrawals, Chester County, Pennsylvania. Water-Resources Investigations Report 03-4263. U.S. Geological Survey, Washington, D.C.

Sloto, R. A. and D. E. Buxton. 2005. Water budgets for selected watersheds in the Delaware River Basin, Eastern Pennsylvania and Western New Jersey. Scientific Investigations Report 2005-5113. U.S. Geological Survey, Washington, D.C.

Smakhtin, V. Y. 1999. Generation of natural daily flow time-series in regulated rivers using a non-linear spatial interpolation technique. Regulated Rivers: Research & Management: **15**:311-323.

Snelder, T. H. and B. J. F. Biggs. 2002. Multiscale river environment classification for water resources management. Journal of the American Water Resources Association **38**:1225–1239.

Snelder, T. H., B. J. F. Biggs, and R. A. Woods. 2005. Improved eco-hydrological classification of rivers. River Research and Applications **21**:609-628.

Sowa, S. P., G. Annis, M. E. Morey, and D. D. Diamond. 2007. A gap analysis and comprehensive conservation strategy for riverine ecosystems of Missouri. Ecological Monographs **77**:301-334.

Steffy, L. Y., D. L. Sitlinger, and T. E. Shenk. 2007. Assessment of interstate streams in the Susquehanna River Basin. Monitoring Report No. 20. Watershed Assessment and Protection Division, Susquehanna River Basin Commission, Harrisburg, PA.

Stuckey, M. H. and L. A. Reed. 2000. Techniques for estimating magnitude and frequency of peak flows for Pennsylvania streams. Water-Resources Investigations Report 00-4189. U.S. Geological Survey, Washington, D.C.

Stuckey, M. H. 2006. Low-flow, base-flow, and mean-flow regression equations for Pennsylvania Streams. Scientific Investigations Report 2006-5130. U.S. Geological Survey, Washington, D.C.

Susquehanna River Basin Commission. 2007. Chemung subbasin survey: a water quality and biological assessment, June - August 2006. Publication 251. Susquehanna River Basin Commission, Harrisburg, PA.

Tharme, R. E. and J. M. King. 1998. Development of the Building Block Methodology for instream flow assessments, and supporting research on the effects of different magnitude flows on riverine ecosystems. Water Research Commission Report No. 576/1/98.

Tharme, R. E. 2003. A global perspective on environmental flow assessment: emerging trends in the development and application of environmental flow methodologies for rivers. River Research and Applications **19**:397-441.

The Nature Conservancy. 2004. Flow Restoration Database. Accessed on March 19, 2007 at http://www.nature.org/initiatives/freshwater/conservationtools/art17004.html.

The Nature Conservancy. 2005. User's Manual for the Indicators of Hydrologic Alteration (IHA) Software. The Nature Conservancy, Charlottesville, VA.

Thompson, R. E. and S. A. Hoffman. Selected streamflow statistics and regression equations for predicting statistics at stream locations in Monroe County, Pennsylvania. Scientific Investigations Report 2006-5244. U.S. Geological Survey, Washington, D.C.

Thoms, M. C. and F. Sheldon. 2002. An ecosystem approach for determining environmental water allocations in Australian dryland river systems: the role of geomorphology. Geomorphology **47:**153–168.

U.S. Fish and Wildlife Service. 1981. Interim regional policy for New England streamflow recommendations. U.S. Fish and Wildlife Service, Newton Corner, MA.

U.S. Geological Survey and Pennsylvania Department of Environmental Protection. 1999. Estimated Water Withdrawals and Use in Pennsylvania, 1995: fact sheet. U.S. Geological Survey, New Cumberland, PA. Accessed June 16, 2008 at http://pa.water.usgs.gov/reports/fs174-99.pdf.

U.S. Geological Survey. 2006. Scope of work for regional determination of hydrologic requirements of aquatic ecosystems of watersheds in Tennessee. Tennessee Water Science Center, U.S. Geological Survey, Nashville, TN.

U.S. Geological Survey and U.S. Environmental Protection Agency. 2006. NHDPlus user guide. U.S. Environmental Protection Agency, Washington, D.C. National Hydrography Dataset, digital data set and documentation accessed June 16, 2008 at: http://www.horizon-systems.com/nhdplus/.

U.S. Geological Survey. 2006. Pennsylvania Highlands Regional Study Scope of Work. Pennsylvania Water Science Center, U.S. Geological Survey, New Cumberland, PA.

U.S. Geological Survey. Undated Draft. Methodology for the Development of the Water Analysis Screening Tool; State Water Plan Update of 2008. Pennsylvania Water Science Center, U.S. Geological Survey, New Cumberland, PA.

Vogel, R. M., J. Sieber, S. A. Archfield, M. P. Smith, C. D. Apse, and A. Huber-Lee. 2007. Relations among storage, yield, and instream flow. Water Resources Research **43**. see: http://engineering.tufts.edu/cee/people/vogel/publications/relationsAmongStorage.pdf

Waldron, M. C. and S. A. Archfield. 2006. Factors affecting firm yield and the estimation of firm yield for selected streamflow-dominated drinking-water-supply reservoirs in Massachusetts. Scientific-Investigations Report 2006-5044. U.S. Geological Survey, Washington, D.C.

Walsh, M.C., J. Deeds, and B. Nightengale. 2007a. Classifying Lotic Systems for Conservation: Methods and Results of the Pennsylvania Aquatic Community Classification.

Pennsylvania Natural Heritage Program, Western Pennsylvania Conservancy, Middletown, PA and Pittsburgh, PA.

Walsh, M.C., J. Deeds, and B. Nightengale. 2007b. User's Manual and Data Guide to the Pennsylvania Aquatic community Classification. Pennsylvania Natural Heritage Program, Western Pennsylvania Conservancy, Middletown, PA and Pittsburgh, PA.

Weiskel, P. K. and S. P. Garabedian. 2005. Development of a Safe-Yield Estimator Application for Massachusetts River Basins: scope of work. U.S. Geological Survey, Northborough, MA.

Weiskel, P. K., R. M. Vogel, P. A. Steeves, P. J. Zarriello, L. A. DeSimone, and K. G. Reis, III. 2007. Water use regimes: characterizing direct human interaction with hydrologic systems. Water Resources Research **43**:1-11.

Wolock, D. M. 2003. Hydrologic landscape regions of the United States. Open-File Report 03–145. U.S. Geological Survey, Washington, D.C. Digital data set, accessed June 16, 2008 at URL http://water.usgs.gov/lookup/getspatial?hlrus.

Young, A. R., R. Grew, and M. G. R. Holmes. 2003. Low Flows 2000: A national water resources assessment and decision support. Water Science and Technology **48**:119-126.

Young, L. 2006. Pennsylvania Statewide Instream Flow Studies Issues Paper. Division of Fisheries Management, Pennsylvania Fish and Boat Commission, Harrisburg, PA.

Zarriello, P. J., and K. G. Ries, III. 2000. A precipitation-runoff model for the analysis of the effects of water withdrawals on streamflow, Ipswich River Basin, Massachusetts. Water-Resources Investigations Report 00-4029. U.S. Geological Survey, Washington, D.C.

Zarriello, P. J., P. M. Barlow, and P. B. Duda. 2001. Simulating the effects of ground-water withdrawals on streamflow in a precipitation-runoff model. In: Bridging the Gap: Meeting the World's Water and Environmental Resources Challenges. Proceedings of the World Water and Environmental Resources Congress, Orlando, Florida.

Zarriello, P. J., and G. C. Bent. 2004. A precipitation-runoff model for the analysis of the effects of water withdrawals and land-use change on streamflow in the Usquepaug–Queen River Basin, Rhode Island. Scientific Investigations Report 2004-5139. U.S. Geological Survey, Washington, D.C.

Zorn, T.G., P. W. Seelbach, E. S. Rutherford, T. C. Wills, S. T. Cheng, and M. J. Wiley. In preparation. A regional-scale habitat suitability model to assess the effects of flow reduction on fish assemblages in Michigan streams.

APPENDIX 1: PROJECT OPTIONS TABLE

	Developing PA Water Management Decision Support System (Baseline & Current Hydrologic Conditions, Classification & Assessing Alteration)				
	Note that the approaches below are not mutually exclusive, and it is likely that Pennsylvania may want take multiple approaches to deal with developing baseline & current hydrologic condition. Although a single approach could be used statewide, at the watershed scale the approach might vary based on: 1) availability of an existing watershed model; 2) primary impacts on the resource (e.g. dam regulation, groundwater withdrawal); 3) hydrologic characteristics of primary concern (e.g. low flows, flood flows).				
	APPROACHES				
	A.1. "Enhanced StreamStats with Water Use Information"	A.2. "FDC Regression with Transform plus Water Budget Application"	A.3. "TOPMODEL and DSS"	A.4. "PRMS and DSS"	A.5. "HSPF and DSS"
Description	**"Enhanced StreamStats with Water Use Information"**: Building upon PA StreamStats (currently under development), this approach will allow the estimation of a wide range of ecologically-relevant static regression flow statistics (low flow to high flows) and will be linked to database of cumulative water use (ground water and surface water) for comparison at pour points. Examination of flow alteration will be done in a manner somewhat similar to the current PA Water Availability Screening Tool Application. A classification of rivers for water management will be completed using hydrologic, physical and/or ecological data.	**"FDC Regression with Transform and DSS"**: This approach will develop regression equations to estimate flow exceedence stats (e.g. Q1-Q99) and will develop a daily time series at ungaged sites using a Flow Duration Curve transform approach. This Flow Duration Curve transform will require an an approach to link index gages to ungaged sites. Current flow conditions estimated as part of Decision Support System (DSS) that takes information from a surface & ground water use database and uses a monthly time scale to estimate impacts on daily flow conditions based on relatively simple equations (e.g. for groundwater withdrawal). A classification of rivers for water management will be completed using hydrologic, physical and/or ecological data. A hydrologic alteration assessment tool (IHA, HIT/HAT, or newly recreated) will be linked to the Water Budget Application. *The baseline hydrologic conditions portion of this approach is similar to the recently submitted TNC-USGS-DRBC Growing Greener Grant.*	**"TOPMODEL and DSS"**: Baseline flow conditions simulated through rainfall-runoff modeling through TOPMODEL. Current flow conditions developed within the watershed model application by developing a spatial water use database that can be used to estimate current flow time series based on relatively simple equations. Baseline and current condition hydrologic data will be evaluated through a Decision Support System (DSS) Complete classification using hydrologic, physical and/or ecological data. Link hydrologic alteration assessment tool (IHA, HIT/HAT, or newly recreated) to model. *Could be done statewide or for selected basins in the state based on resource and technical needs.*	**"PRMS and DSS"**: Baseline flow conditions simulated through rainfall-runoff modeling through PRMS. Current flow conditions developed within the watershed model application by developing a spatial water use database that can be used to estimate current flow time series based on relatively simple equations. Baseline and current condition hydrologic data will be evaluated through a Decision Support System (DSS) Complete classification using hydrologic, physical and/or ecological data. Link hydrologic alteration assessment tool (IHA, HIT/HAT, or newly recreated) to model. *Could be done statewide or for selected basins in the state based on resource and technical needs.*	**"HSPF and DSS":** Use HSPF for development of detailed baseline and current condition simulations. Will permit more advanced assessment of impacts of land use and reservoir operation, etc. Baseline and current condition hydrologic data will be evaluated through a Decision Support System (DSS) Complete classification using hydrologic, physical and/or ecological data. Link hydrologic alteration assessment tool (IHA, HIT/HAT, or newly recreated) to model. *Could be done statewide or for selected basins in the state based on resource and technical needs.*
Cost Ranges for Statewide Application	Baseline & Current Flow Stats in DSS Application: $200,000-250,000; Classification: $25,000-100,000.	Baseline Condition Estimator = $150,000-$250,000; Current Condition & DSS $100,000-$150,000; Classification= $25,000-$100,000.	Baseline & Current Conditions (including DSS) = $300,000-400,000; Classification= $25,000-$100,000.	Baseline & Current Conditions (including DSS) = $350,000-450,000; Classification= $25,000-$100,000.	Baseline & Current Conditions (including DSS)= $500,000 to multiple million $'s; Classification= $25,000-$100,000.
Key Strengths	Least time & cost intensive approach. Methods for developing regression equations have already been applied to estimate low flow statistics in Pennsylvania.	Flexible- provides the ability to develop estimates of daily flow time series (baseline and current). Similar to application under development by USGS MA (Sustainable Yield Estimator)- likely to build on their experience. Straightforward and moderate cost, although requires more steps than A1.	Rainfall - runoff approach, has capacity for hydrologic forecasting based on anticipated changes in rainfall patterns based on climate change. Can also examine land use impacts (current or projected). Jonathan Kennan with experience developing NJ TOPMODEL.	Rainfall - runoff approach, has capacity for hydrologic forecasting based on anticipated changes in rainfall patterns based on climate change. Can also examine land use impacts (current or projected). Will be developed for 30 HUC11s in the near future by PA USGS through PA Highlands Project.	Highly comprehensive and will likely give the best representation of a synthetic hydrograph. HSPF has been developed for Susquehanna Basin, with nodes representing watershed of 50 square miles and larger (would require significant additional model development & cost to make it applicable at smaller scale). HSPF is familiar to many users. Can evaluate many types of changes within the watershed that could affect flows (e.g., land cover, temperature and precipitation, reservoir operation, water withdrawals). Can also be used for a range of other sophisticated applications (e.g., sediment runoff modeling).
Key Limitations	Less flexible than methods that simulate daily time series. Will not permit development of estimated daily flow hydrographs which will constain the analysis that can be completed (i.e., can not use IHA or HAT). Regression model will predict primarily magnitude statistics over period of record. Hydrographic information is only as good as the gaging data the regressions are based on.	Approach is still under development by MA USGS. Ability to accurately model flow duration curves is critical. It is also particularly dependent on the ability to identify representative "least impaired' sites for adequate hydrologic record. Requires confidence in ability to link ungaged sites to an appropriate reference site.	Extended time for data compilation to develop and parameterize a statewide model. Estimates of baseline condition dependent on the quality and spatial coverage of precipation data.	Extended time for data compilation to develop and parameterize a statewide model. Estimates of baseline condition dependent on the quality and spatial coverage of precipation data. Can not handle a large of number of "control points" within any single watershed model. One you define the control points within the model, it is not possible to add additional nodes. (However, could use an approach to extrapolate in between modeling nodes through creating a "runoff grid" in GIS).	Extremely parameter intensive -- extensive data needed for model development and calibration. Requires more expertise than other approaches to both develop and run- the difficulty of running the model may limit routine statewide application. Based on its complexity, may be easier to use at smaller scales. Expensive and cost varies greatly depending on extent of area modeled, available data, and need to collect new data. Can not handle a large of number of "control points" within any single watershed model. One you define the control points within the model, it is not possible to add additional nodes. (However, could use an approach to extrapolate in between modeling nodes through creating a "runoff grid" in GIS).

	Developing PA Water Management Decision Support System (Baseline & Current Hydrologic Conditions, Classification & Assessing Alteration)				
	Note that the approaches below are not mutually exclusive, and it is likely that Pennsylvania may want take multiple approaches to deal with developing baseline & current hydrologic condition. Although a single approach could be used statewide, at the watershed scale the approach might vary based on: 1) availability of an existing watershed model; 2) primary impacts on the resource (e.g. dam regulation, groundwater withdrawal); 3) hydrologic characteristics of primary concern (e.g. low flows, flood flows).				
	APPROACHES				
	A.1. "Enhanced StreamStats with Water Use Information"	**A.2. "FDC Regression with Transform plus Water Budget Application"**	**A.3. "TOPMODEL and DSS"**	**A.4. "PRMS and DSS"**	**A.5. "HSPF and DSS"**
Time of Development	1.5-2 years	2-3 years	2-3 years. Expect that modeling and project time will increase by at least 1/2 year if person responsible for effort has only minimal initial modeling experience.	3-4 years. Time can be decreased or increased dependent upon the experience of the model developer(s).	4-5 years Time can be decreased or increased dependent upon the experience of the model developer(s) Could be developed more efficiently through use of regional calibration (USGS WSP 2459).
Use in Different Regions within Pennsylvania	Easily applied statewide	Easily applied statewide	Yes	May be most appropriate for modeling specific watersheds with unique conditions or heavy water use pressures, and where detailed data exist to support model development.	May be most appropriate for modeling specific watersheds with unique conditions or heavy water use pressures, and where detailed data exist to support model development.
Applicability Across State Lines	Yes -- but only if basin boundary information is complete for interstate areas / regions. Current condition estimates dependent on quality of water use data in adjacent states.	Yes -- but only if basin boundary information is complete for interstate areas / regions. Current condition estimates dependent on quality of water use data in adjacent states.	Yes -- but only if basin boundary information is complete for interstate areas / regions. Current condition estimates dependent on quality of water use data in adjacent states.	Yes -- but only if basin boundary information is complete for interstate areas / regions. Current condition estimates dependent on quality of water use data in adjacent states.	Yes --e.g., has already been developed for the Susquehanna outside the boundaries of PA. Need basin boundary information that is complete for interstate areas/regions. Current condition estimates dependent on quality of water use data in adjacent states.
Potential Funding Sources	Growing Greener $$, EPA, Foundations (Great Lakes Protection Fund, Heinz Foundation, etc), USGS Coop	Growing Greener $$, EPA, Foundations (Great Lakes Protection Fund, Heinz Foundation, etc), USGS Coop	Growing Greener $$, EPA, Foundations (Great Lakes Protection Fund, Heinz Foundation, etc), USGS Coop	Growing Greener $$, EPA, Foundations (Great Lakes Protection Fund, Heinz Foundation, etc), USGS Coop	Growing Greener $$, EPA, Foundations (Great Lakes Protection Fund, Heinz Foundation, etc), USGS Coop
Accuracy of baseline flow estimates in small headwaters (<10 square miles)	Fair - limited by the availability of calibration gages in small watersheds	Fair - limited by the availability of calibration gages in small watersheds	Good - as long as there are resonable continuous record gages downstream in the basin, simulated hydrographs can be scaled appropriately	Good if the model is developed at smaller watershed size (e.g. HUC14s), but not using a statewide model.	Good if the model is developed at smaller watershed size (e.g. HUC14s), but not using a statewide model.
Accuracy of "current condition" in heavily dam regulated rivers	Fair - using simple representation of release rules: More accurate output requires use of reservoir operations model (e.g. OASIS)	Fair - using simple representation of release rules: More accurate output requires use of reservoir operations model (e.g. OASIS)	Fair - using simple representation of release rules: More accurate output requires use of reservoir operations model (e.g. OASIS)	Good if coupled with a reservoir operations model through Modular Modeling System (at additional cost)	Good if coupled with a reservoir operations model through Modular Modeling System (at additional cost). OASIS model already developed in Susquehanna Basin to estimate current or future hydrologic condition for major rivers.
Accuracy of "current condition" for rivers significantly impacted by groundwater withdrawals	Fair - using simple equations describing impact: More accurate output requires use of groundwater model (e.g. MODFLOW)	Fair - using simple equations describing impact: More accurate output requires use of groundwater model (e.g. MODFLOW)	Good - GW withdrawals can be added incrementally as long as state maintains usable pumping records in MGD, however, increased accuracy could be gained by coupling with a groundwater model (e.g., MODFLOW). Modflow can be very costly for broad geographic areas.	Good if coupled with MODFLOW (at additional cost) through GSFLOW. Fair if not.	Good if coupled with MODFLOW (at additional cost). Fair if not.

	Developing the Ecological Basis for Statewide Instream Flow Criteria		
	Note that none of the options below are mutually exclusive-- in fact all three could be implemented as part of a comprehensive approach		
	APPROACHES		
	B.1. Hypothesis development through expert consultation	**B.1. Hypothesis development through expert consultation**	**B.3. Flow-ecology relationships based on new data**
Description	**Hypothesis development through expert consultation:** Use expert workshop series to develop conceptual models of flow-ecology relationships and a basis for hydrologic standards. Existing instream flow studies, expert knowledge, and flow-ecology relationships from other states/regions would be used to develop these conceptual flow-ecology relationships. These results should provide meaningful guidance to decision-makers (potentially in the form of hydrologic-based standards). PA-specific quantitative flow-ecology relationships would not be developed under this task.	**Flow-ecology relationships based on existing data:** Use existing PA and adjacent state biological and hydrologic data to develop flow-ecology relationships for major river types across PA. The intensity of this approach could vary widely, based on the degree of desired investment in examining existing data and testing conceptual models (hypotheses).This approach could include at least three different analyses: (a) Pre-Post comparisons - for sites with adequate hydrological and biological data, test hypotheses about biological responses to hydrologic changes. (b) Statewide landscape analyses - relate species and / or biological community data to sites with varying degrees of flow alteration (after Freeman et al, using existing data & estimates of hydrologic alteration). (c) Fish suitability curves - relate fish (or other taxa) presence data to flow statistics at sites statewide (after approach taken in Michigan Water Withdrawal Assessment Tool).	**Flow-ecology relationships based on new data:** Develop and implement new assessment program to document impacts of flow alteration across a gradient of river types and hydrologic alteration. This assessment approach could: (a) be statewide to complement existing biomonitoring programs or (b) focus on a particular watershed or group of similar watersheds of interest. Either effort would likely build upon B.1 and/or B.2.
Cost Ranges	$15-30,000 per workshop for up to three workshops to bring scientists in to provide input and to summarize results in reports. $20-50,000 for workshop preparation, including: compilation of existing instream flow/habitat studies, organizing and hosting workshop(s), & summarizing hypotheses and conceptual models based on input.	(a) $15-25K for several sites, pending data availability; (b) $100-300K for statewide (or basinwide) analysis; (c) $100-200K to plot fish versus flow alteration, determine optima, create curves.	For: (a) intial investment required for monitoring design approach ($75-100K) and $200,000 and up per year to implement. For (b): $150,000-250,000 based on size of watershed(s) and parameters measured
Major Strengths	Pooled expertise & experience. Rapid ouput useful to develop both initial standards and as hypotheses for quantitative analysis. Does not require completion of hydrologic baseline and current conditions statewide. Expert knowledge should inform any flow-ecology analyses and hypothesis testing. This step should be part of any approach.	Broader applicability, with the potential to define quantitative relationships between flow alteration and ecological conditions. Use of statewide database provides large enough sample size to provide statistical strength. Does not require data collection. Methods from other states / watersheds (e.g., Michigan, Georgia) could be applied in Pennsylvania. Biological data have already been assembled through Pennsylvania Aquatic Community Classification Project.	For (a): Better long-term design for answering important management questions -- i.e., trends etc. May be an appropriate additional goal of statewide biomonitoring programs. Sampling can be designed to cover all environment types and methods can be customized for assessing specific impacts. Sampling program could be designed to fill in gaps within existing data. For (b): Much higher likelihood of defining strong, quantiative flow ecology relationships that when using data that was collected for other purposes (i.e. B.2).
Major Limitations	Limited quantitative conclusions possible. Questions about repeatability. Dependent on subject experts available for participation. Requires ability to find experts that have statewide knowledge.	Comparability of biological samples due to variability in sampling procedure & community types. Confounding factors that impact biological integrity (e.g., water quality). Ensuring long-term effective database management. Relies on flow and biological sampling data collected for other purposes. Data available may not cover all environment types.	For (a): Difficult and time-consuming to coordinate and implement at a statewide basis. Could be done on a rotating basin basis to limit the logistical difficulty. Will take time to accumulate enough data to be meaningful for this type of assessment. For (b): High cost and effort to implement and may provide results limited in geographic applicability
Time of Development	1+ years	1-3 years	For (a) 1-2 years to develop and begin implementation, 3+ years to accumulate data; For (b): 2-3 years
Regional Applicability within PA	Yes	Yes	Yes
Applicability Across State Lines	Yes	Yes	Yes
Potential Funding Sources?	Growing Greener, Foundations (Heinz, GLPF, etc.)	Growing Greener $$, EPA, Foundations (Great Lakes Protection Fund, Heinz Foundation, etc), USGS Coop	Growing Greener $$, EPA, Foundations (Great Lakes Protection Fund, Heinz Foundation, etc), USGS Coop, State 305b, 303d funds, DCNR funds (for B.3.b)

APPENDIX 2: FINAL REPORT - USGS STREAM CLASSIFICATION FOR PENNSYLVANIA (SUBMITTED BY USGS FORT COLLINS TO THE NATURE CONSERVANCY 13 MAY 2008).

Of the 143 gauges with multiple year records, 136 with periods of record ≥15 years for reference conditions (minimally altered) were used to compute the 171 hydrological indices used by HIP and the 34 environmental flow components (EFC). The 136 stream gauges were classified into multiple stream classes by first reducing the number of flow indices and then applying a clustering algorithm. Three (DL19, DL20, and EFC32) of the 205 indices (171 HIP + 34 EFC) were immediately deleted from consideration because they had mostly missing values or too little variation to evaluate statistically. So we performed a principal components analysis (PCA) on the reduced (202) set of indices. Loadings of the indices on the first five principal components (based on correlation matrices) determined which of the 202 indices were retained for the clustering procedure. The selected 151 indices with strongest component loadings (absolute values ≥ 0.60) were standardized to mean = 0 and variance = 1 prior to being used in *k*-means clustering procedures (using Euclidean distances).

Three different classification approaches were considered: (1) classification based on using all 151 indices simultaneously; (2) a 2-stage classification based on first using only 71 of the 151 indices that were strongly correlated ($|r| \geq 0.80$) with drainage area (Table 1) and then using the 80 of 151 indices that were uncorrelated with drainage area to classify within classes formed in the first stage; and (3) only using the 80 hydrological indices that were not highly correlated with drainage area and total daily flow (Table 1). Approach (3) was similar to classification procedures used for HIP applications in New Jersey (Kennen et al. 2007) and Missouri (J. Kennen, personal communication). In those applications, indices that were strongly correlated with drainage area were first standardized by drainage area, i.e., index/drainage area, and this effectively eliminated most of their variation among stream gauges such that these new standardized indices never loaded strong enough on the first several principal components to be included in the clustering procedure used for stream classification. A more transparent approach is to simply eliminate these indices from being included in the clustering procedure used to classify the streams as our approach (3) does. Although the original classification schemes followed by Poff (1996), and subsequently referenced by Olden and Poff (2003), excluded hydrological indices strongly related to drainage area and daily mean discharge, we elected to use classification approach (1) that included indices both strongly related and unrelated to drainage area. This provided a classification of streams that recognized differences in both the absolute magnitude of daily flows and variation in flow dynamics.

Classification Results

Our classification approach used all 151 indices simultaneously in the *k*-means clustering procedure. We examined clustering results initially for 4, 5, 6, 7, and 8 classes before settling on an analysis with 5 classes of stream gauges. The 5 classes of streams included one with median daily flows in the thousands of cfs (class 4, $n = 4$), one with median daily flows in the high hundreds of cfs (class 3, $n = 29$), one with median daily flows in the low hundreds of cfs (class 2, $n = 25$), and two (class 1, $n = 59$ and class 5, $n = 19$) with median daily flows less than a hundred cfs. A discriminant function analysis (DFA) with backwards elimination (based on an F-ratio *P*-value = 0.15) of the 151 variables was used to derive a parsimonious multivariate model that best separated the means of the 5 stream classes based on a reduced set of the flow indices that then were used to graphically depict differences in stream classes. A model with just 11 of the indices (MA1, MA2, MA3, MA5, MA6, MA7, MA10, MA14, MH6, ML12, and FH6) provided a parsimonious separation of the 5 classes and had a 93% jack-knifed classification accuracy. But this set might be combined even further for interpreting the stream classes. Indices for the magnitude of daily flows (MA1, MA2, MA14, MH6, and ML12) had similar patterns of variation among the stream classes (Figure 1) because of their shared correlation with drainage area. Class 4 had much greater flow magnitudes than class 3, which was greater than class 2, which was slightly greater than classes 1 and 5. Indices related to variation in the magnitude of the daily flow (MA3, MA5, MA6, MA7, and MA10) also had similar patterns of variation among the stream classes (Figure 2). Class 5 had the greatest and class 2 the least variation in daily flows, whereas class 1, 3, and 4 were intermediate in variability indices. Flood frequency (FH6) provided a third dimension on which to interpret the stream classes (Figure 3). Class 1 had the greatest and class 2 the least frequency of flood events, whereas classes 3, 4, and 5 were intermediate in flood frequency.

Based on these summary statistics for the 5 stream classes and with reference to national classifications of Poff (1996) and Olden and Poff (2003), a simple description of the 5 stream classes is as follows. Class 2 streams appear to be stable groundwater as indicated by their relatively low overall flow volumes, low variability of daily flows, and low flood frequency. Class 4 streams are large volume (low stream order) perennial runoff streams, class 3 streams are moderate volume perennial runoff streams, and class 1 streams are low volume (high stream order) perennial runoff streams with less variable daily flows but greater flood frequency than class 5 streams, which are low volume perennial flashy-runoff streams with high variability in daily flows. The stream classification by gauges is provided in the Excel file PA_TNC_classes.xls (included as Table 3). The stream gauges are plotted in geographic map space by latitude and longitude in Figure 4.

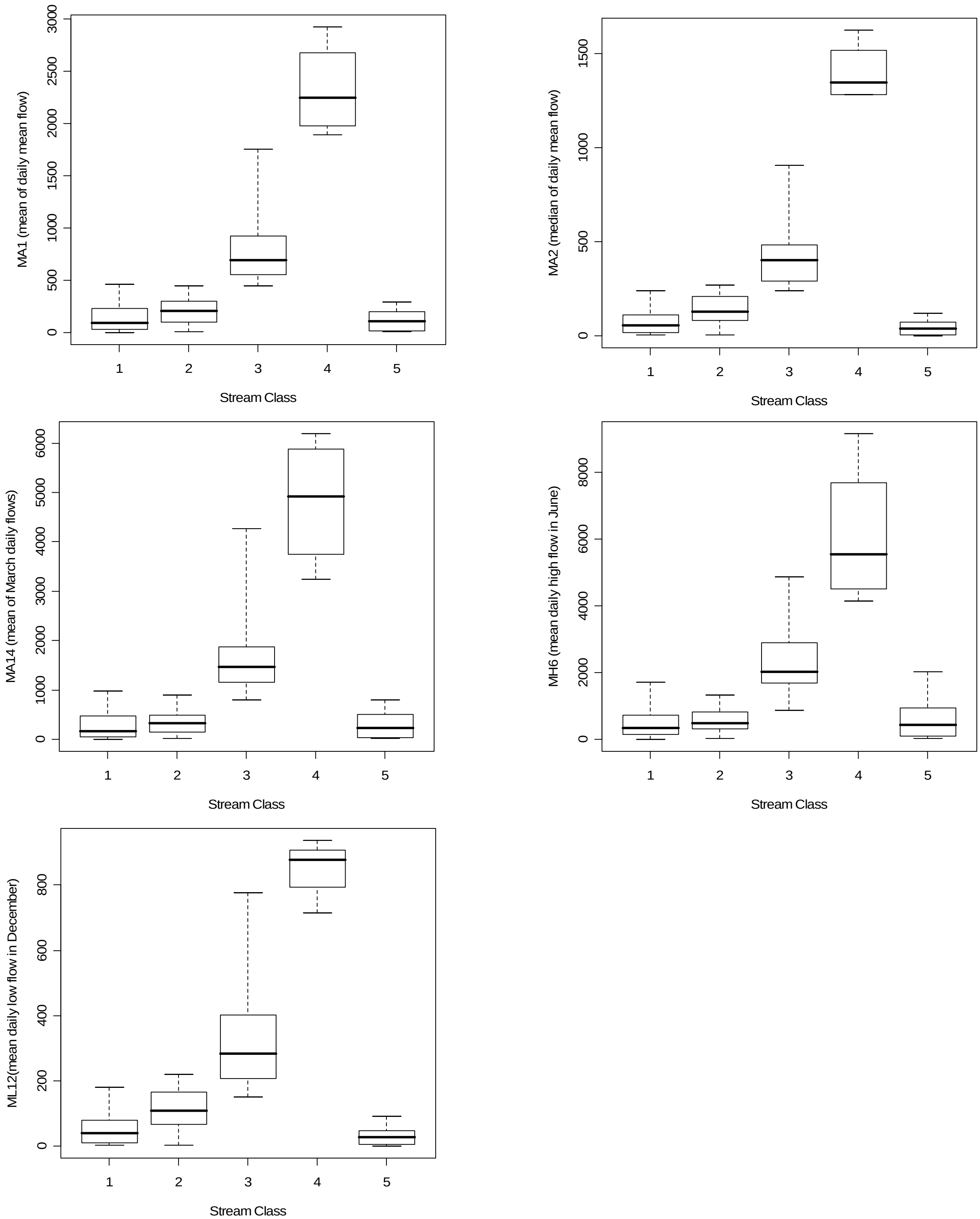


Figure 1. Boxplots of magnitude of daily flow indices (cfs) for 5 stream classes in Pennsylvania using all 151 indices simultaneously in the classification. Thick center line is median, boxes include 25th to 75th percentiles, and the whiskers extend to the minimum and maximum.

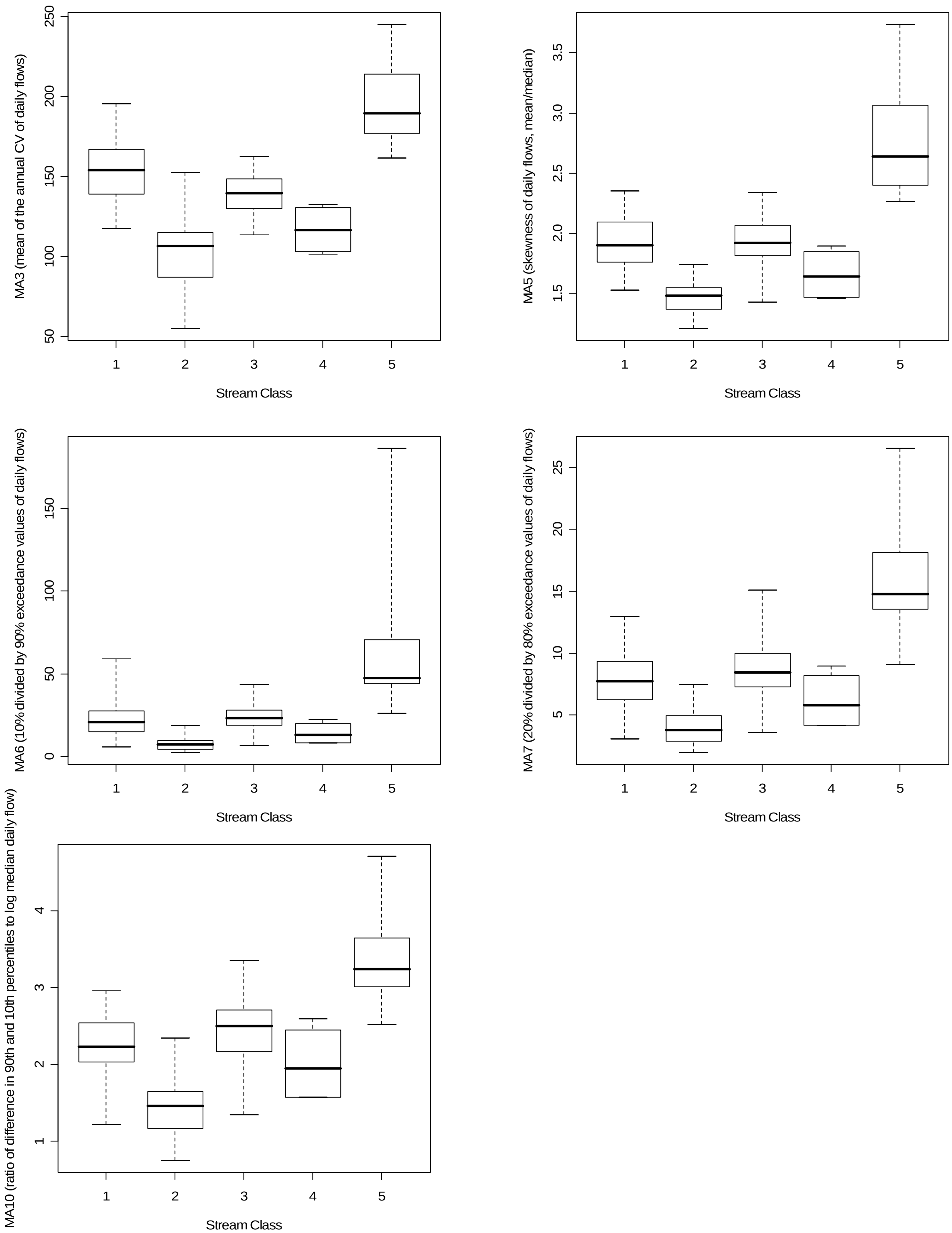


Figure 2. Boxplots of variation in magnitude of daily flow indices for 5 stream classes in Pennsylvania using all 151 indices simultaneously in the classification. Thick center line is median, boxes include 25th to 75th percentiles, and the whiskers extend to the minimum and maximum.

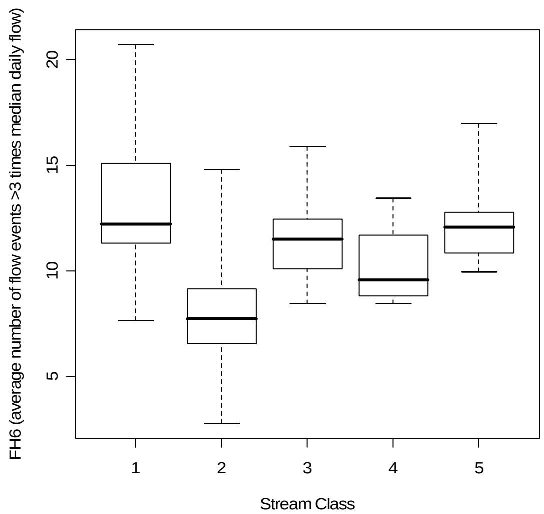


Figure 3. Boxplot of average annual flood frequency for 5 stream classes in Pennsylvania using all 151 indices simultaneously in the classification. Thick center line is median, boxes include 25th to 75th percentiles, and the whiskers extend to the minimum and maximum.

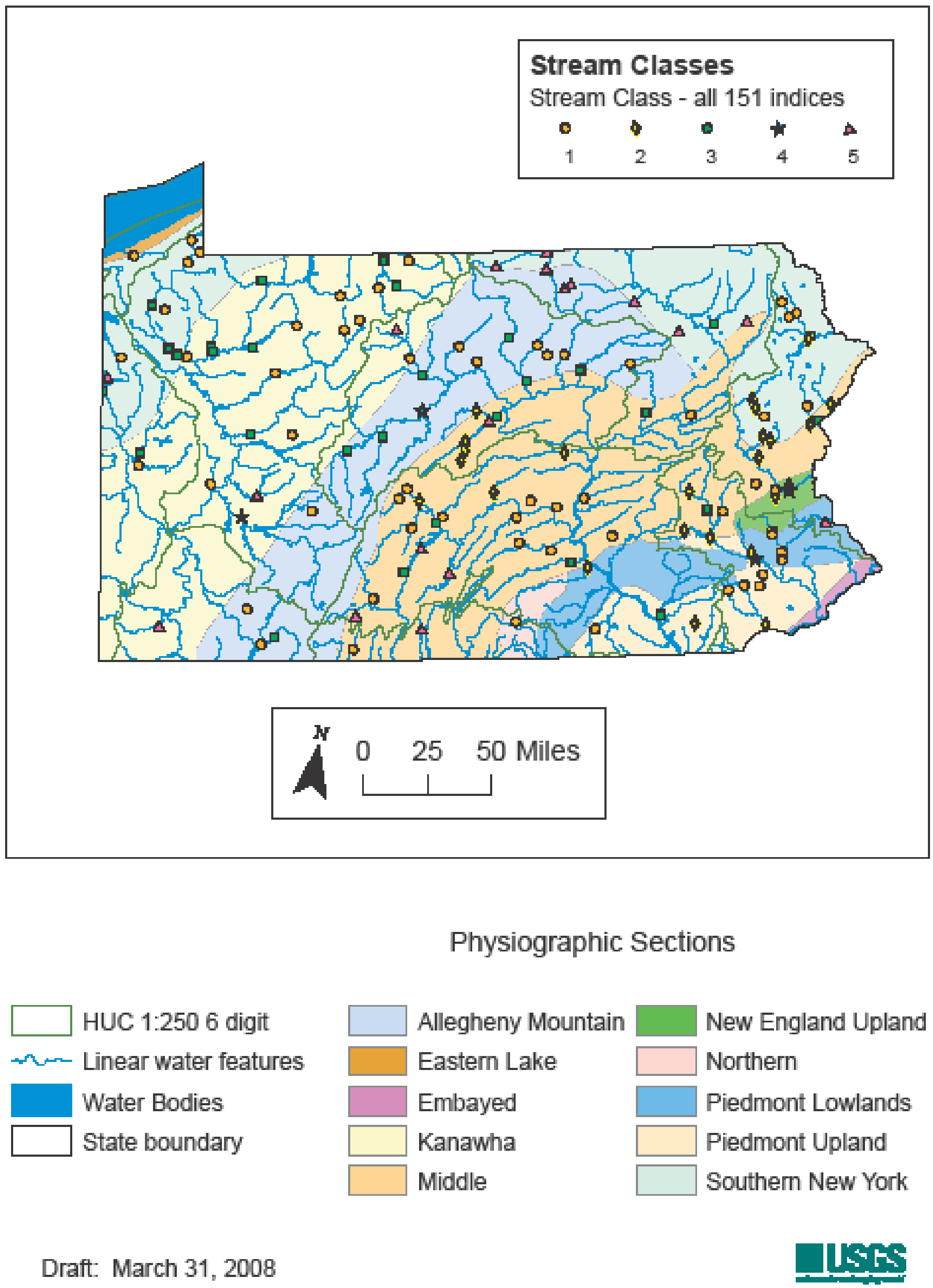


Figure 4. Geographic locations of the 5 stream classes in Pennsylvania. Class 1 streams are low volume (high stream order), perennial runoff streams with greater flood frequency; class 2 streams are stable groundwater streams; class 3 streams are moderate volume, perennial runoff streams; class 4 streams are large volume (low stream order), perennial runoff streams; and class 5 streams are low volume, perennial flashy-runoff streams with high variability in daily flows.

Table 1. Correlation between drainage area (DA) and 151 hydrological indices for 136 stream gauges in Pennsylvania. Bolded indices were highly correlated with drainage area and the others were not.

Pearson Correlation Matrix

	DA
MA1	**0.990010**
MA2	**0.972985**
MA3	-0.200566
MA4	0.011741
MA5	-0.048866
MA6	-0.077077
MA7	-0.004930
MA8	0.016736
MA9	0.035407
MA10	0.078877
MA11	0.087960
MA12	**0.964356**
MA13	**0.980399**
MA14	**0.984923**
MA15	**0.979346**
MA16	**0.980010**
MA17	**0.969897**
MA18	**0.943508**
MA19	**0.929482**
MA20	**0.928469**
MA21	**0.956811**
MA22	**0.957673**
MA23	**0.972682**
MA24	-0.163318
MA25	-0.115918
MA26	-0.098418
MA27	-0.165088
MA28	-0.105672
MA29	-0.128428
MA31	-0.255532
MA33	-0.209630
MA34	-0.277900
MA35	-0.113170
MA37	0.047393
MA38	0.027806
MA39	0.003464
MA43	-0.233646
MA44	-0.198585
ML1	**0.972430**
ML2	**0.973378**
ML3	**0.980684**
ML4	**0.967000**
ML5	**0.952920**
ML6	**0.903213**
ML7	**0.870499**
ML8	**0.853827**
ML9	**0.854139**
ML10	**0.886251**

ML11	**0.946520**
ML12	**0.956463**
ML13	0.028054
ML14	0.006464
ML15	-0.030859
ML16	-0.000944
ML17	-0.028603
ML19	-0.030773
ML20	0.034352
ML22	-0.027750
MH1	**0.953805**
MH2	**0.966888**
MH3	**0.965837**
MH4	**0.968951**
MH5	**0.958691**
MH6	**0.947161**
MH7	**0.902123**
MH8	**0.903271**
MH9	**0.884019**
MH10	**0.933010**
MH11	**0.940855**
MH12	**0.950396**
MH14	-0.210356
MH15	-0.132611
MH16	0.038145
MH17	0.102353
MH21	0.048101
MH22	-0.029375
MH23	-0.132014
MH24	-0.173793
MH25	-0.202378
MH27	-0.138926
FL1	-0.240020
FL3	-0.150287
FH1	-0.221808
FH2	-0.023913
FH3	0.120744
FH4	-0.033607
FH5	-0.217215
FH6	-0.131760
FH7	-0.135742
FH8	-0.221808
FH9	-0.290964
DL1	**0.810156**
DL2	**0.809555**
DL3	**0.817337**
DL4	**0.857918**
DL5	**0.935417**
DL9	-0.101910
DL10	-0.068557
DL11	-0.005453
DL12	-0.003312
DL13	-0.029885
DL14	-0.029011
DL15	0.002427
DH1	**0.970064**
DH2	**0.980800**

DH3	**0.982546**
DH4	**0.985291**
DH5	**0.988106**
DH9	-0.227181
DH10	-0.205604
DH11	-0.230557
DH12	-0.095216
DH13	-0.025822
DH14	-0.043778
DH15	0.198178
DH16	-0.134595
DH17	0.118641
DH18	0.210231
DH19	0.170799
DH20	0.199575
DH21	0.140351
TA1	0.521502
TA2	0.560663
RA1	**0.950488**
RA3	**0.953430**
RA4	-0.374880
RA7	-0.063896
EFC1	**0.915825**
EFC2	**0.957588**
EFC3	**0.963776**
EFC4	**0.968074**
EFC5	**0.973186**
EFC6	**0.981275**
EFC7	**0.976624**
EFC8	**0.967371**
EFC9	**0.931687**
EFC10	**0.912576**
EFC11	**0.874421**
EFC12	**0.887646**
EFC13	**0.826200**
EFC17	**0.980120**
EFC20	-0.203870
EFC21	**0.971017**
EFC22	**-0.943151**
EFC23	**0.968954**
EFC24	0.215308
EFC27	**0.825729**
EFC28	**-0.875276**
EFC29	**0.919398**
EFC33	0.717672
EFC34	**-0.805209**

Primary and Secondary Hydrological Indices

Following the cluster analysis for the simultaneous classification, PCA was performed on each of the 5 classes of stream gauges and all classes combined (6 analyses) to identify indices that best explained variation in the 11 sub-components of the flow regime (low, average, and high magnitude; low and high frequency; low and high duration; low, average, and high timing; and average rate of change) for the HIP indices and the environmental flow components (EFC) corresponding to low (EFC1-13) and high (EFC17, 23, and 29) magnitude, low (EFC14) and high (EFC18, 24, 25, and 30) duration, low (EFC16) and high (EFC20, 26, and 32) frequency, low (EFC15) and high (EFC19 and 31) timing, and rate of change (EFC21, 22, 27, 28, 33, and 34) within stream classes. Again, PCA was performed only on the subset of indices without any missing values for a given class of streams. We retained up to the first 5 principal components that explained the majority of the variance in the indices and examined scree plots to determine whether a reduced number of components could be considered. Indices selected to explain the dominant pattern of hydrologic variation were those with the largest absolute loadings on the first 4 principal components for indices in each of the 11 sub-components of the flow regime and the environmental flow components following Olden and Poff (2003). This provided a reduced set of indices that were related to major components of variation in hydrology within a stream class and that were relatively uncorrelated with each other. Table 2 includes the primary and secondary hydrological indices selected by this process. Indices ranking third and fourth on the principal components are also available. The indices selected from the principal components for stream class 4 are not very reliable as the sample size for this class was only $n = 4$.

It should be noted that the environmental flow components (EFC) never loaded as strongly on any of the principal components as the hydrological indices from HIP. The EFC also were not as strongly associated with differences among the 5 stream classes as the other hydrological indices. However, this does not imply that there were no statistical differences in EFC among the stream classes. Many of the EFC were strongly correlated and similar in definition to the hydrological indices.

Table 2. Primary and secondary flow components with highest absolute loadings on first several principal components explaining majority of variance within 5 stream groups and within all streams (n = 136) in Pennsylvania. Principal components (PC) analysis was made with the correlation matrix. We used the first 4 PC for Group 1 (75% of variance); first 4 PC for Group 2 (80% of variance); first 4 PC for Group 3 (75% of variance); first 3 PC for Group 4 (100% of variance); first 4 PC for Group 5 (75% of variance); and first 4 PC for all streams (77% of variance). When the number of variables per component is less than the number of PC used that is due to a variable occurring as highest absolute loading on multiple PC.

Flow Component	Group 1 n = 59	Group 2 n = 25	Group 3 n = 29	Group 4 n = 4	Group 5 n = 19	All streams
MA	1, 6	14, 33	7, 19	27, 32	1, 4	2, 4
ML	1, 17	3, 19	22, 10	7, 21	3, 17	1, 13
MH	4, 16	4, 24	3, 14	15, 9	4, 27	7, 16
FL	3, 1	2, 1	1, 3	3, 1	3, 1	3, 1
FH	3, 6	3, 5	3, 7	2, 1	11, 4	7, 3
DL	5, 15	5, 1	14, 5	15, 6	5, 15	5, 14
DH	3, 12	2, 14	19, 5	12, 20	3, 12	5, 13
TA	2, 1	3, 1	1, 2	1, 2	1, 2	2, 3
TL	2, 1	2, 1	1, 2	3, 1	3, 1	1, 2
TH	2, 1	2	2, 1	2, 1	2, 1	3, 2
RA	1, 7	1, 6	3, 6	8, 5	3, 7	1, 7
EFC_ML	6,13	7, 13	7, 11	6, 3	8, 13	10, 6
EFC_MH	17, 29	17, 29	17, 23	23, 29	17, 23	17, 23
EFC_DL	14	14	14	14	14	14
EFC_DH	18, 24	24, 30	18, 24	18, 24	24, 25	24, 18
EFC_FL	16	16	16	16	16	16
EFC_FH	20, 26	26, 20	26, 20	.	20, 26	20
EFC_TL	15	15	15	15	15	15
EFC_TH	31, 19	31, 19	19, 31	31, 19	31, 19	31, 19
EFC_RA	21, 33	21, 33	28, 21	33, 27	21, 28	21, 28

Table 3. Stream classification by gauge station using three classification approaches (originally provided to TNC in the Excel file PA_TNC_classes.xls)

Station Name	Station ID	Drainage Area (DA)	Start Water Year	End Water Year	No. of Water Years	Stream Class 1 (all 151 indices)	Stream Class 2 (2-stage using 151 indices)	Stream Class 3 (using subset of 151 indices not correlated with DA)
West Branch Lackawaxen River near Aldenville, Pa.	1428750	40.6	1988	2006	18	1	1	2
West Branch Lackawaxen River at Prompton, Pa.	1429000	59.7	1946	1960	14	1	1	2
Dyberry Creek near Honesdale, Pa.	1429500	64.6	1945	1959	14	1	1	2
Lackawaxen River at Hawley, Pa.	1431500	290.0	1910	1959	49	3	4	2
Wallenpaupack Creek at Wilsonville, Pa.	1432000	228.0	1911	1925	14	2	2	1
Bush Kill at Shoemakers, Pa.	1439500	117.0	1910	2001	91	2	2	1
Brodhead Creek near Analomink, Pa.	1440400	65.9	1959	2006	47	1	2	1
McMichael Creek near Stroudsburg, Pa.	1441000	65.3	1913	1938	25	2	2	1
Brodhead Creek at Minisink Hills, Pa.	1442500	259.0	1952	2006	54	3	5	1
Lehigh River at Stoddartsville, Pa.	1447500	91.7	1945	2006	61	2	2	1
Tobyhanna Creek near Blakeslee, Pa.	1447720	118.0	1963	1985	22	2	2	1
Dilldown Creek near Long Pond, PA	1448500	10.9	1949	1996	47	1	2	1
Pohopoco Creek at Kresgeville, Pa.	1449360	49.9	1968	2006	38	2	2	1
Wild Creek at Hatchery, Pa.	1449500	16.8	1942	1958	16	2	2	1
Aquashicola Creek at Palmerton, Pa.	1450500	76.7	1941	2006	65	2	2	1
Little Lehigh Creek near Allentown, Pa.	1451500	80.8	1947	2006	59	2	2	4
Jordan Creek near Schnecksville, Pa.	1451800	53.0	1967	2006	39	1	1	2
Jordan Creek at Allentown, PA	1452000	53.0	1945	2006	61	1	1	2
Monocacy Creek at Bethlehem, Pa.	1452500	44.5	1950	2006	56	2	2	4
Lehigh River at Bethlehem, Pa.	1453000	1279.0	1904	1927	23	4	6	1
Tohickon Creek near Pipersville, Pa.	1459500	97.4	1937	1973	36	5	3	3
Schuylkill River at Landingville, Pa.	1468500	133.0	1949	2006	57	2	2	1
Schuylkill River at Berne, Pa.	1470500	355.0	1949	2006	57	3	5	1
Maiden Creek at Virginville, Pa.	1470756	159.0	1974	1995	21	1	1	5
Tulpehocken Creek near Bernville, Pa.	1470779	66.5	1976	2006	30	2	2	4
Tulpehocken Creek near Reading, Pa.	1471000	211.0	1952	1979	27	2	2	1
Manatawny Creek near Pottstown, Pa.	1471980	85.5	1976	2006	30	2	2	5
Schuylkill River at Pottstown, Pa.	1472000	1147.0	1929	1979	50	4	6	1
French Creek near Phoenixville, Pa.	1472157	59.1	1970	2006	36	1	2	5
Pickering Creek near Chester Springs, Pa.	1472174	6.0	1968	1983	15	1	2	5
Perkiomen Creek at East Greenville, Pa.	1472198	38.0	1983	2006	23	1	2	5

Station Name	Station ID	Drainage Area (DA)	Start Water Year	End Water Year	No. of Water Years	Stream Class 1 (all 151 indices)	Stream Class 2 (2-stage using 151 indices)	Stream Class 3 (using subset of 151 indices not correlated with DA)
West Branch Perkiomen Creek at Hillegass, Pa.	1472199	23.0	1983	2006	23	1	2	5
Perkiomen Creek near Frederick, Pa.	1472500	152.0	1886	1913	27	1	1	5
Perkiomen Creek at Graterford, Pa.	1473000	279.0	1916	1956	40	1	4	5
West Branch Brandywine Creek near Honey Brook, Pa.	1480300	18.7	1962	2006	44	1	2	5
Marsh Creek near Glenmoore, Pa.	1480675	8.6	1968	2006	38	1	1	5
Brandywine Creek at Chadds Ford, Pa.	1481000	287.0	1913	1973	60	2	5	1
Corey Creek near Mainesburg, Pa.	1516500	12.2	1956	2006	50	5	3	3
Elk Run near Mainesburg, PA	1517000	12.2	1955	1978	23	5	3	3
Crooked Creek at Tioga, Pa.	1518500	122.0	1955	1974	19	5	3	3
Cowanesque River at Westfield, Pa.	1518862	90.6	1985	2006	21	5	3	3
Cowanesque River near Lawrenceville, Pa.	1520000	298.0	1953	1979	26	5	3	3
Towanda Creek near Monroeton, Pa.	1532000	215.0	1915	2006	91	5	3	3
North Branch Mehoopany Creek near Lovelton, Pa.	1533500	35.2	1942	1958	16	5	3	3
SB Tunkhannock Creek near Montdale, Pa.	1533950	12.6	1962	1978	16	5	1	3
Tunkhannock Creek near Tunkhannock, Pa.	1534000	383.0	1915	2006	91	3	4	2
Wapwallopen Creek near Wapwallopen, Pa.	1538000	43.8	1921	2006	85	1	1	1
Fishing Creek near Bloomsburg, Pa.	1539000	274.0	1940	2006	66	3	4	2
West Branch Susquehanna River at Bower, Pa.	1541000	315.0	1915	2006	91	3	4	2
Clearfield Creek at Dimeling, Pa.	1541500	371.0	1915	1960	45	3	4	2
WB Susquehanna River at Karthaus, Pa.	1542500	1462.0	1941	1964	23	4	6	2
Waldy Run near Emporium, Pa.	1542810	5.2	1966	2006	40	5	3	3
Driftwood Br Sinnemahoning Cr at Sterling Run, Pa.	1543000	272.0	1915	2006	91	1	1	2
Sinnemahoning Creek at Sinnemahoning, Pa.	1543500	685.0	1940	2006	66	3	4	2
Kettle Creek at Cross Fork, Pa.	1544500	136.0	1942	2006	64	1	1	2
Young Womans Creek near Renovo, Pa.	1545600	46.2	1966	2006	40	1	1	2
North Bald Eagle Creek at Milesburg, Pa.	1546000	119.0	1912	1934	22	5	3	3
Spring Creek at Houserville, Pa.	1546400	58.5	1986	2006	20	2	2	4
Spring Creek near Axemann, Pa.	1546500	87.2	1942	2006	64	2	2	4
Spring Creek at Milesburg, PA	1547100	142.0	1967	2006	39	2	2	4
Bald Eagle Creek bl Spring Creek at Milesburg, Pa.	1547200	265.0	1957	2006	49	2	5	4
Marsh Creek at Blanchard, Pa.	1547700	44.1	1957	2006	49	5	3	3

Station Name	Station ID	Drainage Area (DA)	Start Water Year	End Water Year	No. of Water Years	Stream Class 1 (all 151 indices)	Stream Class 2 (2-stage using 151 indices)	Stream Class 3 (using subset of 151 indices not correlated with DA)
Beech Creek at Monument, Pa.	1547950	152.0	1970	2006	36	2	2	1
Bald Eagle Creek near Beech Creek Station, Pa.	1548005	562.0	1912	1970	58	3	5	1
Pine Creek at Cedar Run, Pa.	1548500	604.0	1920	2006	86	3	4	2
Blockhouse Creek near English Center, Pa.	1549500	37.7	1942	2006	64	1	1	2
Pine Creek bl L Pine Creek near Waterville, Pa.	1549700	944.0	1959	2006	47	3	4	2
Larrys Creek at Cogan House, Pa.	1549780	6.8	1962	1978	16	1	1	2
Lycoming Creek near Trout Run, Pa.	1550000	173.0	1915	2006	91	1	1	2
Loyalsock Creek at Loyalsockville, Pa.	1552000	435.0	1927	2006	79	3	4	2
Muncy Creek near Sonestown, Pa.	1552500	23.8	1942	2006	64	1	1	2
Penns Creek at Penns Creek, Pa.	1555000	301.0	1931	2006	75	2	2	1
East Mahantango Creek near Dalmatia, Pa.	1555500	162.0	1931	2006	75	1	1	2
Frankstown Br Juniata River at Williamsburg, Pa.	1556000	291.0	1918	2006	88	1	2	1
Little Juniata River at Tipton, Pa.	1556500	93.7	1947	1962	15	1	1	2
Bald Eagle Creek at Tyrone, Pa.	1557500	44.1	1954	2006	52	1	1	2
Little Juniata River at Spruce Creek, Pa.	1558000	220.0	1940	2006	66	2	2	1
Juniata River at Huntingdon, Pa.	1559000	816.0	1943	2006	63	3	5	1
Standing Stone Creek near Huntingdon, Pa.	1559500	128.0	1931	1958	27	1	1	2
Sulphur Springs Creek near Manns Choice, Pa.	1559700	5.3	1963	1978	15	5	3	3
Dunning Creek at Belden, Pa.	1560000	172.0	1941	2006	65	1	1	2
Raystown Branch Juniata River at Saxton, Pa.	1562000	756.0	1913	2006	93	3	4	2
Great Trough Creek near Marklesburg, Pa.	1562500	84.6	1931	1957	26	5	3	3
Aughwick Creek near Three Springs, Pa.	1564500	205.0	1940	2006	66	5	3	3
Kishacoquillas Creek at Reedsville, Pa.	1565000	164.0	1941	1985	44	2	2	1
Little Lost Creek at Oakland Mills, Pa.	1565700	6.5	1965	1981	16	1	1	5
Tuscarora Creek near Port Royal, Pa.	1566000	214.0	1913	2006	93	1	1	2
Cocolamus Creek near Millerstown, Pa.	1566500	57.2	1932	1958	26	1	1	2
Bixler Run near Loysville, Pa.	1567500	15.0	1955	2006	51	1	2	5
Sherman Creek at Shermans Dale, Pa.	1568000	207.0	1931	2006	75	1	1	2
Conodoguinet Creek near Hogestown, Pa.	1570000	470.0	1913	1969	56	3	5	1
Yellow Breeches Creek near Camp Hill, Pa.	1571500	216.0	1911	2006	95	2	2	4
Manada Creek at Manada Gap, Pa.	1573500	13.5	1939	1958	19	1	1	2
Codorus Creek at Spring Grove, Pa.	1574500	75.5	1931	1964	33	1	1	5
Conestoga River at Conestoga, Pa.	1576754	470.0	1986	2006	20	3	5	1
Bowery Run near Quarryville, Pa.	1578400	6.0	1964	1981	17	2	2	5

Station Name	Station ID	Drainage Area (DA)	Start Water Year	End Water Year	No. of Water Years	Stream Class 1 (all 151 indices)	Stream Class 2 (2-stage using 151 indices)	Stream Class 3 (using subset of 151 indices not correlated with DA)
Evitts Creek near Centerville, Pa.	1603500	30.2	1934	1982	48	1	1	2
Tonoloway Creek near Needmore, PA	1613050	30.2	1966	2006	40	5	3	3
Conococheague Creek near Fayetteville, Pa.	1614090	5.1	1962	1981	19	1	2	1
Allegheny River at Port Allegany, Pa.	3007800	248.0	1976	2006	30	3	1	2
Potato Creek at Smethport, Pa.	3009680	160.0	1976	1995	19	1	1	2
Allegheny River at Eldred, Pa.	3010500	550.0	1941	2006	65	3	4	2
Oswayo Creek at Shinglehouse, Pa.	3010655	98.7	1976	2006	30	1	1	2
Kinzua Creek near Guffey, Pa.	3011800	38.8	1967	2006	39	1	1	1
Brokenstraw Creek at Youngsville, Pa.	3015500	321.0	1911	2006	95	3	4	2
Tionesta Creek at Lynch, Pa.	3017500	233.0	1939	1979	40	1	1	2
Tionesta Creek at Nebraska, Pa.	3019000	469.0	1911	1940	29	3	4	2
Oil Creek at Rouseville, Pa.	3020500	300.0	1934	2006	72	3	4	2
Oil Creek near Rouseville, PA	3021000	300.0	1910	1932	22	3	4	2
French Creek near Wattsburg, Pa.	3021350	92.0	1976	2006	30	1	1	5
West Branch French Creek near Lowville, Pa.	3021410	52.3	1976	1993	17	1	1	5
French Creek at Carters Corners, Pa.	3021500	208.0	1911	1971	60	1	1	2
French Creek at Saegerstown, Pa.	3022500	629.0	1923	1939	16	3	4	2
Woodcock Creek at Blooming Valley, Pa.	3022540	31.1	1976	1995	19	1	1	5
French Creek at Carlton, Pa.	3023500	998.0	1910	1925	15	3	4	2
French Creek at Utica, Pa.	3024000	1028.0	1934	1970	36	3	4	2
Sugar Creek at Sugarcreek, Pa.	3025000	166.0	1934	1979	45	1	1	2
Sevenmile Run near Rasselas, Pa.	3026500	7.8	1953	2006	53	1	1	2
West Branch Clarion River at Wilcox, Pa.	3028000	63.0	1955	2006	51	1	1	2
Toms Run at Cooksburg, Pa.	3029400	12.6	1961	1978	17	1	1	2
Big Run nr Sprankle Mills, Pa.	3031950	7.4	1965	1981	16	1	1	5
Redbank Creek at St. Charles, Pa.	3032500	528.0	1920	2006	86	3	4	2
Crooked Creek at Idaho, Pa.	3038000	191.0	1939	1967	28	5	3	3
Little Yellow Creek near Strongstown, Pa.	3042200	7.4	1962	1988	26	1	1	2
Kiskiminetas River at Avonmore, Pa.	3047500	1723.0	1909	1937	28	4	6	2
Buffalo Creek near Freeport, Pa.	3049000	137.0	1942	2006	64	1	1	2
South Fork Tenmile Creek at Jefferson, PA	3073000	133.0	1932	1995	63	5	3	3
Casselman River at Markleton, Pa.	3079000	382.0	1922	2006	84	3	4	2
Laurel Hill Creek at Ursina, Pa.	3080000	121.0	1920	2006	86	1	1	2
Poplar Run near Normalville, Pa.	3082200	9.3	1963	1978	15	1	1	2

Station Name	Station ID	Drainage Area (DA)	Start Water Year	End Water Year	No. of Water Years	Stream Class 1 (all 151 indices)	Stream Class 2 (2-stage using 151 indices)	Stream Class 3 (using subset of 151 indices not correlated with DA)
Little Shenango River at Greenville, Pa.	3102500	104.0	1915	2006	91	1	1	2
Pymatuning Creek near Orangeville, Pa.	3103000	169.0	1915	1963	48	5	3	3
Shenango River at Sharon, Pa.	3104000	608.0	1911	1938	27	3	4	2
Connoquenessing Creek near Zelienople, Pa.	3106000	356.0	1921	2006	85	1	4	2
Slippery Rock Creek at Wurtemburg, Pa.	3106500	398.0	1913	1969	56	3	4	2
Brandy Run near Girard, Pa.	4213075	4.5	1988	2006	18	1	1	5
Middle Creek near Hawley, Pa.	1431000	78.4	1947	1960	13			
Fishing Creek at Bloomsburg, Pa.	1540000	355.0	1915	1928	13			
Jackson Run near North Warren, Pa.	3015280	12.8	1964	1978	14			
Georges Creek at Smithfield, Pa.	3072590	16.3	1965	1978	13			
Youghiogheny River at Connellsville, Pa.	3082500	1326.0	1910	1925	15			
Shenango River near Jamestown, Pa.	3102000	181.0	1921	1934	13			
Raccoon Creek at Moffatts Mill, Pa.	3108000	178.0	1943	1956	13			

Appendix Literature Cited

Kennen, J.G., J.A. Henriksen, and S.P. Nieswand. 2007. Development of the hydroecological integrity assessment process for determining environmental flows for New Jersey streams. Scientific Investigations Report 2007-5206. : U.S. Geological Survey. 55 p.

Olden, J. D., and N. L. Poff. 2003. Redundancy and the choice of hydrologic indices for characterizing streamflow regimes. River Research and Applications 19: 101-121.

Poff, N. L. 1996. A hydrogeography of unregulated streams in the United States and an examination of scale-dependence in some hydrological descriptors. Freshwater Biology 36: 71

APPENDIX 3: DEFINITIONS OF HYDROLOGICAL INDICES USED IN THE HYDROLOGICAL INTEGRITY ASSESSMENT PROGRAM (KENNEN ET AL. 2007).

Code	Definition
MA1	Mean of the daily mean flow values for the entire flow record (cubic feet per second - temporal).
MA2	Median of the daily mean flow values for the entire flow record (cubic feet per second - temporal).
MA3	Mean (or median - Use Preference option) of the coefficients of variation (standard deviation/mean) for each year. Compute the coefficient of variation for each year of daily flows. Compute the mean of the annual coefficients of variation (percent - temporal).
MA4	Standard deviation of the percentiles of the logs of the entire flow record divided by the mean of percentiles of the logs. Compute the log10 of the daily flows for the entire record. Compute the 5th, 10th, 15th, 20th, 25th, 30th, 35th, 40th, 45th, 50th, 55th, 60th, 65th, 70th, 75th, 80th, 85th, 90th, and 95th percentiles for the logs of the entire flow record. Percentiles are computed by interpolating between the ordered (ascending) logs of the flow values. Compute the standard deviation and mean for the percentile values. Divide the standard deviation by the mean (percent - spatial).
MA5	The skewness of the entire flow record is computed as the mean for the entire flow record (MA1) divided by the median (MA2) for the entire flow record (dimensionless - spatial).
MA6	Range in daily flows is the ratio of the 10 percent to 90 percent exceedence values for the entire flow record. Compute the 5 percent to 95 percent exceedence values for the entire flow record. Exceedence is computed by interpolating between the ordered (descending) flow values. Divide the 10 percent exceedence value by the 90 percent value (dimensionless – spatial).
MA7	Range in daily flows is computed like MA6 except using the 20 percent and 80 percent exceedence values. Divide the 20 percent exceedence value by the 80 percent value (dimensionless - spatial).
MA8	Range in daily flows is computed like MA6 except using the 25 percent and 75 percent exceedence values. Divide the 25 percent exceedence value by the 75 percent value (dimensionless – spatial).
MA9	Spread in daily flows is the ratio of the difference between the 90th and 10th percentile of the logs of the flow data to the log of the median of the entire flow record. Compute the log10 of the daily flows for the entire record. Compute the 5th, 10th, 15th, 20th, 25th, 30th, 35th, 40th, 45th, 50th, 55th, 60th, 65th, 70th, 75th, 80th, 85th, 90th, and 95th percentiles for the logs of the entire flow record. Percentiles are computed by interpolating between the ordered (ascending) logs of the flow values. Compute MA9 as (90th – 10th) /log10(MA2) (dimensionless – spatial).
MA10	Spread in daily flows is computed like MA9 except using the 20th and 80th percentiles (dimensionless – spatial).
MA11	Spread in daily flows is computed like MA9 except using the 25th and 75th percentiles (dimensionless – spatial).
MA12–MA23	Means (or medians - Use Preference option) of monthly flow values. Compute the means for each month over the entire flow record. For example, MA12 is the mean of all January flow values over the entire record (cubic feet per second – temporal).
MA24 – MA35	Variability (coefficient of variation) of monthly flow values. Compute the standard deviation for each month in each year over the entire flow record. Divide the standard deviation by the mean for

Code	Definition
	each month. Average (or median - Use Preference option) these values for each month across all years (percent – temporal).
MA36	Variability across monthly flows. Compute the minimum, maximum, and mean flows for each month in the entire flow record. MA36 is the maximum monthly flow minus the minimum monthly flow divided by the median monthly flow (dimensionless – spatial).
MA37	Variability across monthly flows. Compute the first (25th percentile) and the third (75^{th} percentile) quartiles (every month in the flow record). MA37 is the third quartile minus the first quartile divided by the median of the monthly means (dimensionless – spatial).
MA38	Variability across monthly flows. Compute the 10th and 90th percentiles for the monthly means (every month in the flow record). MA38 is the 90th percentile minus the 10th percentile divided by the median of the monthly means (dimensionless – spatial).
MA39	Variability across monthly flows. Compute the standard deviation for the monthly means. MA39 is the standard deviation times 100 divided by the mean of the monthly means (percent – spatial).
MA40	Skewness in the monthly flows. MA40 is the mean of the monthly flow means minus the median of the monthly means divided by the median of the monthly means (dimensionless – spatial).
MA41	Annual runoff. Compute the annual mean daily flows. MA41 is the mean of the annual means divided by the drainage area (cubic feet per second/square mile – temporal).
MA42	Variability across annual flows. MA42 is the maximum annual flow minus the minimum annual flow divided by the median annual flow (dimensionless – spatial).
MA43	Variability across annual flows. Compute the first (25th percentile) and third (75^{th} percentile) quartiles and the 10th and 90th percentiles for the annual means (every year in the flow record). MA43 is the third quartile minus the first quartile divided by the median of the annual means (dimensionless –spatial).
MA44	Variability across annual flows. Compute the first (25th percentile) and third (75^{th} percentile) quartiles and the 10th and 90th percentiles for the annual means (every year in the flow record). MA44 is the 90th percentile minus the 10th percentile divided by the median of the annual means (dimensionless – spatial).
MA45	Skewness in the annual flows. MA45 is the mean of the annual flow means minus the median of the annual means divided by the median of the annual means (dimensionless – spatial).
ML1 – ML12	Mean (or median - Use Preference option) minimum flows for each month across all years. Compute the minimum daily flow for each month over the entire flow record. For example, ML1 is the mean of the minimums of all January flow values over the entire record (cubic feet per second – temporal).
ML13	Variability (coefficient of variation) across minimum monthly flow values. Compute the mean and standard deviation for the minimum monthly flows over the entire flow record. ML13 is the standard deviation times 100 divided by the mean minimum monthly flow for all years (percent – spatial).
ML14	Compute the minimum annual flow for each year. ML14 is the mean of the ratios of minimum annual flows to the median flow for each year (dimensionless – temporal).
ML15	Low flow index. ML15 is the mean of the ratios of minimum annual flows to the mean flow for each year (dimensionless – temporal).
ML16	Median of annual minimum flows. ML16 is the median of the ratios of minimum annual flows to

Code	Definition
	the median flow for each year (dimensionless – temporal).
ML17	Base flow. Compute the mean annual flows. Compute the minimum of a 7-day moving average flow for each year and divide them by the mean annual flow for that year. ML17 is the mean (or median - Use Preference option) of those ratios (dimensionless – temporal).
ML18	Variability in base flow. Compute the standard deviation for the ratios of 7-day moving average flows to mean annual flows for each year. ML18 is the standard deviation times 100 divided by the mean of the ratios (percent – spatial).
ML19	Base flow. Compute the ratios of the minimum annual flow to mean annual flow for each year. ML19 is the mean (or median - Use Preference option) of these ratios times 100 (dimensionless – temporal).
ML20	Base flow. Divide the daily flow record into 5-day blocks. Find the minimum flow for each block. Assign the minimum flow as a base flow for that block if 90 percent of that minimum flow is less than the minimum flows for the blocks on either side. Otherwise, set it to zero. Fill in the zero values using linear interpolation. Compute the total flow for the entire record and the total base flow for the entire record. ML20 is the ratio of total base flow to total flow (dimensionless – spatial).
ML21	Variability across annual minimum flows. Compute the mean and standard deviation for the annual minimum flows. ML21 is the standard deviation times 100 divided by the mean (percent – spatial).
ML22	Specific mean annual minimum flow. ML22 is the mean (or median - Use Preference option) of the annual minimum flows divided by the drainage area (cubic feet per second/square mile – temporal).
MH1 – MH12	Mean (or median - Use Preference option) maximum flows for each month across all years. Compute the maximum daily flow for each month over the entire flow record. For example, MH1 is the mean of the maximums of all January flow values over the entire record (cubic feet per second – temporal).
MH13	Variability (coefficient of variation) across maximum monthly flow values. Compute the mean and standard deviation for the maximum monthly flows over the entire flow record. MH13 is the standard deviation times 100 divided by the mean maximum monthly flow for all years (percent – spatial).
MH14	Median of annual maximum flows. Compute the annual maximum flows from monthly maximum flows. Compute the ratio of annual maximum flow to median annual flow for each year. MH14 is the median of these ratios (dimensionless – temporal).
MH15	High flow discharge index. Compute the 1 percent exceedence value for the entire data record. MH15 is the 1 percent exceedence value divided by the median flow for the entire record (dimensionless – spatial).
MH16	High flow discharge index. Compute the 10 percent exceedence value for the entire data record. MH16 is the 10 percent exceedence value divided by the median flow for the entire record (dimensionless – spatial).
MH17	High flow discharge index. Compute the 25 percent exceedence value for the entire data record. MH17 is the 25 percent exceedence value divided by the median flow for the entire record (dimensionless – spatial).
MH18	Variability across annual maximum flows. Compute the logs (log10) of the maximum annual flows. Find the standard deviation and mean for these values. MH18 is the standard deviation

Code	Definition
	times 100 divided by the mean (percent – spatial).
MH19	Skewness in annual maximum flows. Use the equation: $$MH19 = \frac{N2 \times sum(qm3)-3N \times sum(qm) \times sum(qm2) + 2 \times (sum(qm))3}{N \times (N-1) \times (N-2) \times stddev3}$$ Where: N = Number of years qm = Log10 (annual maximum flows) stddev = Standard deviation of the annual maximum flows (dimensionless – spatial).
MH20	Specific mean annual maximum flow. MH20 is the mean (or median - Use Preference option) of the annual maximum flows divided by the drainage area (cubic feet per second/square mile – temporal).
MH21	High flow volume index. Compute the average volume for flow events above a threshold equal to the median flow for the entire record. MH21 is the average volume divided by the median flow for the entire record (days – temporal).
MH22	High flow volume. Compute the average volume for flow events above a threshold equal to three times the median flow for the entire record. MH22 is the average volume divided by the median flow for the entire record (days - temporal).
MH23	High flow volume. Compute the average volume for flow events above a threshold equal to seven times the median flow for the entire record. MH23 is the average volume divided by the median flow for the entire record (days - temporal).
MH24	High peak flow. Compute the average peak flow value for flow events above a threshold equal to the median flow for the entire record. MH24 is the average peak flow divided by the median flow for the entire record (dimensionless – temporal).
MH25	High peak flow. Compute the average peak-flow value for flow events above a threshold equal to three times the median flow for the entire record. MH25 is the average peak flow divided by the median flow for the entire record (dimensionless – temporal).
MH26	High peak flow. Compute the average peak flow value for flow events above a threshold equal to seven times the median flow for the entire record. MH26 is the average peak flow divided by the median flow for the entire record (dimensionless – temporal).
MH27	High peak flow. Compute the average peak flow value for flow events above a threshold equal to 75th percentile value for the entire flow record. MH27 is the average peak flow divided by the median flow for the entire record (dimensionless – temporal).
FL1	Low flood pulse count. Compute the average number of flow events with flows below a threshold equal to the 25th percentile value for the entire flow record. FL1 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FL2	Variability in low pulse count. Compute the standard deviation in the annual pulse counts for FL1. FL2 is 100 times the standard deviation divided by the mean pulse count (percent – spatial).
FL3	Frequency of low pulse spells. Compute the average number of flow events with flows below a threshold equal to 5 percent of the mean flow value for the entire flow record. FL3 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH1	High flood pulse count. Compute the average number of flow events with flows above a threshold equal to the 75th percentile value for the entire flow record. FH1 is the average (or median - Use Preference option) number of events (number of events/year – temporal).

Code	Definition
FH2	Variability in high pulse count. Compute the standard deviation in the annual pulse counts for FH1. FH2 is 100 times the standard deviation divided by the mean pulse count (number of events/year – spatial).
FH3	High flood pulse count. Compute the average number of days per year that the flow is above a threshold equal to three times the median flow for the entire record. FH3 is the mean (or median – Use Preference option) of the annual number of days for all years (number of days/year – temporal).
FH4	High flood pulse count. Compute the average number of days per year that the flow is above a threshold equal to seven times the median flow for the entire record. FH4 is the mean (or median - Use Preference option) of the annual number of days for all years (number of days/year – temporal).
FH5	Flood frequency. Compute the average number of flow events with flows above a threshold equal to the median flow value for the entire flow record. FH5 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH6	Flood frequency. Compute the average number of flow events with flows above a threshold equal to three times the median flow value for the entire flow record. FH6 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH7	Flood frequency. Compute the average number of flow events with flows above a threshold equal to seven times the median flow value for the entire flow record. FH6 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH8	Flood frequency. Compute the average number of flow events with flows above a threshold equal to 25 percent exceedence value for the entire flow record. FH8 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH9	Flood frequency. Compute the average number of flow events with flows above a threshold equal to 75 percent exceedence value for the entire flow record. FH9 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
FH10	Flood frequency. Compute the average number of flow events with flows above a threshold equal to median of the annual minima for the entire flow record. FH10 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
	Note - 1.67-year flood threshold (Olden and Poff 2003) - For indices FH11, DH22, DH23, DH24, TA3, and TH3 compute the log10 of the peak annual flows. Compute the log10 of the daily flows for the peak annual flow days. Calculate the coefficients for a linear regression equation for logs of peak annual flow versus logs of average daily flow for peak days. Using the log peak flow for the 1.67 year recurrence interval (60th percentile) as input to the regression equation, predict the log10 of the average daily flow. The threshold is 10 to the log10 (average daily flow) power (cubic feet/second).
FH11	Flood frequency. Compute the average number of flow events with flows above a threshold equal to flow corresponding to a 1.67-year recurrence interval. FH11 is the average (or median - Use Preference option) number of events (number of events/year – temporal).
DL1	Annual minimum daily flow. Compute the minimum 1-day average flow for each year. DL1 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DL2	Annual minimum of 3-day moving average flow. Compute the minimum of a 3-day moving average flow for each year. DL2 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DL3	Annual minimum of 7-day moving average flow. Compute the minimum of a 7-day moving average flow for each year. DL3 is the mean (or median - Use Preference option) of these values

Code	Definition
	(cubic feet per second – temporal).
DL4	Annual minimum of 30-day moving average flow. Compute the minimum of a 30-day moving average flow for each year. DL4 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DL5	Annual minimum of 90-day moving average flow. Compute the minimum of a 90-day moving average flow for each year. DL5 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DL6	Variability of annual minimum daily average flow. Compute the standard deviation for the minimum daily average flow. DL6 is 100 times the standard deviation divided by the mean (percent – spatial).
DL7	Variability of annual minimum of 3-day moving average flow. Compute the standard deviation for the minimum 3-day moving averages. DL7 is 100 times the standard deviation divided by the mean (percent - spatial).
DL8	Variability of annual minimum of 7-day moving average flow. Compute the standard deviation for the minimum 7-day moving averages. DL8 is 100 times the standard deviation divided by the mean (percent - spatial).
DL9	Variability of annual minimum of 30-day moving average flow. Compute the standard deviation for the minimum 30-day moving averages. DL9 is 100 times the standard deviation divided by the mean (percent - spatial).
DL10	Variability of annual minimum of 90-day moving average flow. Compute the standard deviation for the minimum 90-day moving averages. DL10 is 100 times the standard deviation divided by the mean (percent - spatial).
DL11	Annual minimum daily flow divided by the median for the entire record. Compute the minimum daily flow for each year. DL11 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DL12	Annual minimum of 7-day moving average flow divided by the median for the entire record. Compute the minimum of a 7-day moving average flow for each year. DL12 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DL13	Annual minimum of 30-day moving average flow divided by the median for the entire record. Compute the minimum of a 30-day moving average flow for each year. DL13 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DL14	Low exceedence flows. Compute the 75 percent exceedence value for the entire flow record. DL14 is the exceedence value divided by the median for the entire record (dimensionless – spatial).
DL15	Low exceedence flows. Compute the 90 percent exceedence value for the entire flow record. DL14 is the exceedence value divided by the median for the entire record (dimensionless – spatial).
DL16	Low flow pulse duration. Compute the average pulse duration for each year for flow events below a threshold equal to the 25th percentile value for the entire flow record. DL16 is the median of the yearly average durations (number of days – temporal).
DL17	Variability in low pulse duration. Compute the standard deviation for the yearly average low pulse durations. DL17 is 100 times the standard deviation divided by the mean of the yearly average low pulse durations (percent – spatial).
DL18	Number of zero-flow days. Count the number of zero-flow days for the entire flow record. DL18 is the mean (or median - Use Preference option) annual number of zero flow days (number of

Code	Definition
	days/year – temporal).
DL19	Variability in the number of zero-flow days. Compute the standard deviation for the annual number of zero-flow days. DL19 is 100 times the standard deviation divided by the mean annual number of zero-flow days (percent – spatial).
DL20	Number of zero-flow months. While computing the mean monthly flow values, count the number of months in which there was no flow over the entire flow record (percent – spatial).
DH1	Annual maximum daily flow. Compute the maximum of a 1-day moving average flow for each year. DH1 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH2	Annual maximum of 3-day moving average flows. Compute the maximum of a 3-day moving average flow for each year. DH2 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH3	Annual maximum of 7-day moving average flows. Compute the maximum of a 7-day moving average flow for each year. DH3 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH4	Annual maximum of 30-day moving average flows. Compute the maximum of 30-day moving average flows. Compute the maximum of a 30-day moving average flow for each year. DH4 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH5	Annual maximum of 90-day moving average flows. Compute the maximum of a 90-day moving average flow for each year. DH5 is the mean (or median - Use Preference option) of these values (cubic feet per second – temporal).
DH6	Variability of annual maximum daily flows. Compute the standard deviation for the maximum 1-day moving averages. DH6 is 100 times the standard deviation divided by the mean (percent – spatial).
DH7	Variability of annual maximum of 3-day moving average flows. Compute the standard deviation for the maximum 3-day moving averages. DH7 is 100 times the standard deviation divided by the mean (percent – spatial).
DH8	Variability of annual maximum of 7-day moving average flows. Compute the standard deviation for the maximum 7-day moving averages. DH8 is 100 times the standard deviation divided by the mean (percent – spatial).
DH9	Variability of annual maximum of 30-day moving average flows. Compute the standard deviation for the maximum 30-day moving averages. DH9 is 100 times the standard deviation divided by the mean (percent – spatial).
DH10	Variability of annual maximum of 90-day moving average flows. Compute the standard deviation for the maximum 90-day moving averages. DH10 is 100 times the standard deviation divided by the mean (percent – spatial).
DH11	Annual maximum of 1-day moving average flows divided by the median for the entire record. Compute the maximum of a 1-day moving average flow for each year. DL11 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DH12	Annual maximum of 7-day moving average flows divided by the median for the entire record. Compute the maximum daily average flow for each year. DL12 is the mean of these values divided by the median for the entire record (dimensionless – temporal).
DH13	Annual maximum of 30-day moving average flows divided by the median for the entire record. Compute the maximum of a 30-day moving average flow for each year. DL13 is the mean of these values divided by the median for the entire record (dimensionless – temporal).

Code	Definition
DH14	Flood duration. Compute the mean of the mean monthly flow values. Find the 95th percentile for the mean monthly flows. DH14 is the 95th percentile value divided by the mean of the monthly means (dimensionless – spatial).
DH15	High flow pulse duration. Compute the average duration for flow events with flows above a threshold equal to the 75th percentile value for each year in the flow record. DH15 is the median of the yearly average durations (days/year – temporal).
DH16	Variability in high flow pulse duration. Compute the standard deviation for the yearly average high pulse durations. DH16 is 100 times the standard deviation divided by the mean of the yearly average high pulse durations (percent – spatial).
DH17	High flow duration. Compute the average duration of flow events with flows above a threshold equal to the median flow value for the entire flow record. DH17 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH18	High flow duration. Compute the average duration of flow events with flows above a threshold equal to three times the median flow value for the entire flow record. DH18 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH19	High flow duration. Compute the average duration of flow events with flows above a threshold equal to seven times the median flow value for the entire flow record. DH19 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH20	High flow duration. Compute the 75th percentile value for the entire flow record. Compute the average duration of flow events with flows above a threshold equal to the 75th percentile value for the median annual flows. DH20 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH21	High flow duration. Compute the 25th percentile value for the entire flow record. Compute the average duration of flow events with flows above a threshold equal to the 25th percentile value for the entire set of flows. DH21 is the average (or median - Use Preference option) duration of the events (days – temporal).
DH22	Flood interval. Compute the flood threshold as the flow equivalent for a flood recurrence of 1.67 years. Determine the median number of days between flood events for each year. DH22 is the mean (or median - Use Preference option) of the yearly median number of days between flood events (days – temporal).
DH23	Flood duration. Compute the flood threshold as the flow equivalent for a flood recurrence of 1.67 years. Determine the number of days each year that the flow remains above the flood threshold. DH23 is the mean (or median - Use Preference option) of the number of flood days for years in which floods occur (days – temporal).
DH24	Flood-free days. Compute the flood threshold as the flow equivalent for a flood recurrence of 1.67 years. Compute the maximum number of days that the flow is below the threshold for each year. DH24 is the mean (or median - Use Preference option) of the maximum yearly no-flood days (days – temporal).
TA1	Constancy. Constancy is computed via the formulation of Colwell (see example in Colwell, 1974). A matrix of values is compiled where the rows are 11 flow categories and the columns are 365 (no February 29th) days of the year. The cell values are the number of times that a flow falls into a category on each day. The categories are: log(flow) < .1 x log(mean flow), .1 x log(mean flow) <= log(flow) < .25 x log(mean flow) .25 x log(mean flow) <= log(flow) < .5 x log(mean flow) .5 x log(mean flow) <= log(flow) < .75 x log(mean flow)

Code	Definition
	.75 x log(mean flow) <= log(flow) < 1.0 x log(mean flow) 1.0 x log(mean flow) <= log(flow) < 1.25 x log(mean flow) 1.25 x log(mean flow) <= log(flow) < 1.5 x log(mean flow) 1.5 x log(mean flow) <= log(flow) < 1.75 x log(mean flow) 1.75 x log(mean flow) <= log(flow) < 2.0 x log(mean flow) 2.0 x log(mean flow) <= log(flow) < 2.25 x log(mean flow) log(flow) >= 2.25 x log(mean flow) The row totals, column totals, and grand total are computed. Using the equations for Shannon information theory parameters, constancy is computed as: $1- \frac{\text{(uncertainty with respect to state)}}{\text{log (number of state)}}$ (dimensionless – spatial).
TA2	Predictability. Predictability is computed from the same matrix as constancy (see example in Colwell, 1974). It is computed as: $1- \frac{\text{(uncertainty with respect to interaction of time and state - uncertainty with respect to time}}{\text{log (number of state)}}$ (dimensionless – spatial).
TA3	Seasonal predictability of flooding. Divide years up into 2-month periods (that is, Oct-Nov, Dec-Jan, and so forth). Count the number of flood days (flow events with flows > 1.67-year flood) in each period over the entire flow record. TA3 is the maximum number of flood days in any one period divided by the total number of flood days (dimensionless – temporal).
TL1	Julian date of annual minimum. Determine the Julian date that the minimum flow occurs for each water year. Transform the dates to relative values on a circular scale (radians or degrees). Compute the x and y components for each year and average them across all years. Compute the mean angle as the arc tangent of y-mean divided by x-mean. Transform the resultant angle back to Julian date (Julian day – spatial).
TL2	Variability in Julian date of annual minima. Compute the coefficient of variation for the mean x and y components and convert to a date (Julian day – spatial). Note - 5-year flood threshold (Olden and Poff 2003) – For TL3 and TL4, compute the log10 of the peak annual flows. Compute the log10 of the daily flows for the peak annual flow days. Calculate the coefficients for a linear regression equation for logs of peak annual flow versus logs of average daily flow for peak days. Using the log peak flow for the 5-year recurrence interval (80th percentile) as input to the regression equation, predict the log10 of the average daily flow. The threshold is 10 to the log10 (average daily flow) power (cubic feet per second).
TL3	Seasonal predictability of low flow. Divide years up into 2-month periods (that is, Oct-Nov, Dec-Jan, and so forth). Count the number of low flow events (flow events with flows <= 5 year flood threshold) in each period over the entire flow record. TL3 is the maximum number of low flow events in any one period divided by the total number of low flow events (dimensionless – spatial).
TL4	Seasonal predictability of non-low flow. Compute the number of days that flow is above the 5-year flood threshold as the ratio of number of days to 365 or 366 (leap year) for each year. TL4 is the maximum of the yearly ratios (dimensionless – spatial).
TH1	Julian date of annual maximum. Determine the Julian date that the maximum flow occurs for each year. Transform the dates to relative values on a circular scale (radians or degrees). Compute the x and y components for each year and average them across all years. Compute the mean angle as the

Code	Definition
	arc tangent of y-mean divided by x-mean. Transform the resultant angle back to Julian date (Julian day – spatial).
TH2	Variability in Julian date of annual maxima. Compute the coefficient of variation for the mean x and y components and convert to a date (Julian days - spatial).
TH3	Seasonal predictability of nonflooding. Computed as the maximum proportion of a 365-day year that the flow is less than the 1.67-year flood threshold and also occurs in all years. Accumulate nonflood days that span all years. TH3 is maximum length of those flood-free periods divided by 365 (dimensionless – spatial).
RA1	Rise rate. Compute the change in flow for days in which the change is positive for the entire flow record. RA1 is the mean (or median - Use Preference option) of these values (cubic feet per second/day – temporal).
RA2	Variability in rise rate. Compute the standard deviation for the positive flow changes. RA2 is 100 times the standard deviation divided by the mean (percent – spatial).
RA3	Fall rate. Compute the change in flow for days in which the change is negative for the entire flow record. RA3 is the mean (or median – Use Preference option) of these values (cubic feet per second/day – temporal).
RA4	Variability in fall rate. Compute the standard deviation for the negative flow changes. RA4 is 100 times the standard deviation divided by the mean (percent – spatial).
RA5	Number of day rises. Compute the number of days in which the flow is greater than the previous day. RA5 is the number of positive gain days divided by the total number of days in the flow record (dimensionless – spatial).
RA6	Change of flow. Compute the log10 of the flows for the entire flow record. Compute the change in log of flow for days in which the change is positive for the entire flow record. RA6 is the median of these values (cubic feet per second – temporal).
RA7	Change of flow. Compute the log10 of the flows for the entire flow record. Compute the change in log of flow for days in which the change is negative for the entire flow record. RA7 is the median of these log values (cubic feet per second/day – temporal).
RA8	Number of reversals. Compute the number of days in each year when the change in flow from one day to the next changes direction. RA8 is the average (or median - Use Preference option) of the yearly values (days - temporal).
RA9	Variability in reversals. Compute the standard deviation for the yearly reversal values. RA9 is 100 times the standard deviation divided by the mean (percent – spatial).

Environmental Flow Component	EFC Code	HIP Sub-component
October Low Flow	1	ML
November Low Flow	2	ML
December Low Flow	3	ML
January Low Flow	4	ML
February Low Flow	5	ML
March Low Flow	6	ML
April Low Flow	7	ML
May Low Flow	8	ML
June Low Flow	9	ML
July Low Flow	10	ML
August Low Flow	11	ML

September Low Flow	12	ML
Extreme low peak	13	ML
Extreme low duration	14	DL
Extreme low timing	15	TL
Extreme low freq.	16	FL
High flow peak	17	MH
High flow duration	18	DH
High flow timing	19	TH
High flow frequency	20	FH
High flow rise rate	21	RA
High flow fall rate	22	RA
Small Flood peak	23	MH
Small Flood duration	24	DH
Small Flood timing	25	DH
Small Flood freq.	26	FH
Small Flood rise rate	27	RA
Small Flood fall rate	28	RA
Large flood peak	29	MH
Large flood duration	30	DH
Large flood timing	31	TH
Large flood freq.	32	FH
Large flood rise rate	33	RA
Large flood fall rate	34	RA

APPENDIX 4: VALUE ADDED ATTRIBUTES IN THE NORTHEAST AQUATIC HABITAT CLASSIFICATION SYSTEM

Table 1. Value Added Attributes from USGS NHD-Plus

Value Added Attributes from USGS NHD-Plus

GNIS_Name	Feature Name from the Geographic Names Information System
LengthKM	Feature length in kilometers
FType	NHD Feature Type
HUC_8	8-digit Hydrologic Unit Code, also known as Subbasin code (formerly known as catalog unit code)
HU_8_Name	Text name of Subbasin
Precip	Mean annual precipitation in mm
Temp	Mean annual temperature in degrees centigrade * 10
AreaWtMAP	Area Weighted Mean Annual Precipitation at bottom of flowline in mm
AreaWtMAT	Area Weighted Mean Annual Temperature at bottom of flowline in degree C * 10
CumDrainag	Cumulative drainage area in square kilometers(sq km) at bottom of flowline
MAFlowU	Mean Annual Flow in cubic feet per second (cfs) at bottom of flowline as computed by Unit Runoff Method
MAFlowV	Mean Annual Flow (cfs) at bottom of flowline as computed by Vogel Method
MAVelU	Mean Annual Velocity (fps) at bottom of flowline as computed by Unit Runoff Method
MAVelV	Mean Annual Velocity (fps) at bottom of flowline as computed by Unit Runoff Method
IncrFlowU	Incremental Flow (cfs) for Flowline as computed by the Unit Runoff Method
MaxElevSmo	Maximum elevation (smoothed) in meters
MinElevSmo	Minimum elevation (smoothed) in meters
Slope	Slope of flowline (m/m)
StreamLeve	Stream level
StreamOrde	Strahler stream order
Local and Cumulative 1992 NLCD Land Cover	11. Open Water 12. Perennial Ice/Snow 21. Low Intensity Residential 22. High Intensity Residential 23. Commercial/Industrial/Transportation 31. Bare Rock/Sand/Clay 32. Quarries/Strip Mines/Gravel Pits 33. Transitional 41. Deciduous Forest 42. Evergreen Forest 43. Mixed Forest 51. Shrubland 61. Orchards/Vineyards/Other 71. Grasslands/Herbaceous 81. Pasture/Hay 82. Row Crops 83. Small Grains 84. Fallow 85. Urban/Recreational Grasses 91. Woody Wetlands 92. Emergent Herbaceous Wetlands

Table 2. Value Added Attributes Calculated by The Nature Conservancy

Value Added Attributes Calculated by The Nature Conservancy

GEOLOGY (local and cumulative watershed)		
%Acidic sed/metased	100	TNC regional bedrock and surficial geology dataset, compiled from state sources.
%Acidic shale	200	TNC regional bedrock and surficial geology dataset, compiled from state sources.
%Calcareous sed/metased	300	TNC regional bedrock and surficial geology dataset, compiled from state sources.
%Mod calcareous sed/metased	400	TNC regional bedrock and surficial geology dataset, compiled from state sources.
%Acidic granitic	500	TNC regional bedrock and surficial geology dataset, compiled from state sources.
%Mafic/intermediate granitic	600	TNC regional bedrock and surficial geology dataset, compiled from state sources.
%Ultramafic	700	TNC regional bedrock and surficial geology dataset, compiled from state sources.
%Coarse sediments	800	TNC regional bedrock and surficial geology dataset, compiled from state sources.
%Fine sediments	900	TNC regional bedrock and surficial geology dataset, compiled from state sources.
REGIONAL ELEVATION CLASS (put USGS NHD Plus continuous minimum elevation data into major class as follows:)		
coastal zone	1	<20ft
low elevation	2	20-800'
mid-to-lower elevation transitional	3	800-1700'
mid-to-upper elevation transitional	4	1700-2500
high elevation	5	2500-3600
subalpine/alpine	6	> 3600'
NETWORK POSITION		
upcomid		mainpath upstream reach
downcomid		mainpath downstream reach comid
Lake Connected (up or down)	9	
Downstream Size Class 2	2	
Downstream Size class 3a	3a	
Downstream Size class 3b	3b	
Downstream Size class 4	4	
Downstream Size class5	5	
LOCAL WATERSHED LANDFORMS		
mean local watershed slope		continuous, based on sampling USGS NED 30m digital elevation model 2004.
%Summit/ridgetop	1	as modeled by TNC Ecological Land Units
%Cliff/steep slope	2	as modeled by TNC Ecological Land Units
%Sideslope	3	as modeled by TNC Ecological Land Units
%Cove/Foodslope	4	as modeled by TNC Ecological Land Units
%Hill/Valley/gentle slope	5	as modeled by TNC Ecological Land Units
%Dry flats	6	as modeled by TNC Ecological Land Units
%Wetflats	7	as modeled by TNC Ecological Land Units
%Open water	8	as modeled by TNC Ecological Land Units
FLOW STABILITY (calculate mean USGS Baseflow Index grid value for each reach + put major class as follows:)		
mean local watershed		continuous, based on sampling USGS Base-flow index grid for the conterminous United States 2003 D. Wolock
Flashy	1	0-.4
Intermediate	2	.4-.65
Very Stable	3	>.65
AREA WEIGHTED MEAN ANNUAL TEMP CLASS (degree C * 10) (put USGS NHD Plus continuous data into major class as follows:)		
	1	1. 0-15
	2	2. 15-30
	3	3. 30-45
	4	4. 45-60
	5	5. 60-75
	6	6. 75-90
	7	7. 90-105
	8	8. 105-120
	9	9. 120-135
	10	10. 135-150
HIGHER STRATIFICATION		
Freshwater Ecoregion		
Ecological Drainage Unit		