American Fisheries Society Symposium 91:xxx–xxx, 2019

The Really Big Picture for Fish Habitat: The Conceptual Underpinnings and Vision for the National Fish Habitat Partnership's National Fish Habitat Assessment

Gary E. Whelan*
Michigan Department of Natural Resources-Fisheries Division
Post Office Box 30446, Lansing, Michigan 48909, USA

Abstract.—Over much of the history of fisheries management, fisheries biologists challenged with the conservation of degraded fisheries habitats have primarily focused on addressing the symptoms of habitat degradation as opposed to confronting the overarching processes and factors that control fish habitat condition. This is often attributable to the substantial amount of inaccessible or unorganized data that confound resource management decisions. The National Fish Habitat Partnership (NFHP) was formed in 2006 to provide a science-based, holistic, and voluntary-based approach to address the trillions of U.S. dollars in damages that have been inflicted on fish habitats in the United States. The NFHP uses a periodically measured, landscape-level national fish habitat assessment to identify intact systems that need conservation or protection and to assess the root causes of aquatic habitat degradation in altered systems. Categories of data and information contained within the NFHP national fish habitat assessment consist of hydrology, connectivity, water quality, material transport and recruitment, geomorphology, and aquatic organisms' effect on habitat and energy flow. These processes are critically important in controlling fish habitat condition in all types of aquatic systems, with the key differences being the relative importance and the rates in which the processes and factors operate. Data and information on fish and aquatic organisms and social data are the other components needed to build a comprehensive assessment and decision support framework for fish habitats in the United States. A framework for a model national fish habitat assessment (model assessment) is outlined herein, with each category described in measurable subcomponents that are actionable by fisheries biologists or other aquatic resource managers. Key variables for each process and factor, along with needed data and information for development of dose–response relationships and social data for societal importance indication, are also provided. Although much of the data to fully populate a model assessment are not available currently, it is important to establish a vision for the future. Many of the envisioned data necessary for a model assessment are available on a localized or regional basis to enable the detailed analyses to occur on those spatial scales, allowing the testing of the robustness of the framework. Once the model assessment is fully developed, aquatic resource managers will

* Corresponding author: whelang@michigan.gov

have a powerful tool to prioritize the trillions of dollars needed to conserve intact and rehabilitate degraded aquatic habitats to build self-sustaining and resilient fish communities. The tool will also help facilitate the NFHP's goals to maintain intact systems and to move degraded system processes and factors back to within 25% of the expected norms for those watersheds.

Introduction

A pressing challenge for aquatic resource managers is that we are swimming in billions of dollars (U.S.) of fish and aquatic community and fish habitat data in the United States that are unorganized and generally unavailable, except for some state and individual watershed efforts. To address this issue, the development of a complete and comparable picture of the condition of U.S. fisheries habitat that can assist in large-scale prioritization of habitat restoration efforts has been a long-time dream of fisheries biologists. This dream was not technically feasible until the 2000s with the advent of geographic information systems and powerful desktop computers. The need for such decision support tools is clear as many U.S. fisheries and their supporting populations are in decline, with about 20% of the biota that rely on aquatic systems considered imperiled or critically imperiled (The H. John Heinz III Center for Science, Economics and the Environment 2002). In addition, approximately 37% of fish fauna are considered at risk or vulnerable, with nearly 4 of every 10 native freshwater fishes at risk of extinction (Stein et al. 2000). Declines in at-risk species are found throughout the United States, with the highest proportion in Hawaii and the southeastern United States (The H. John Heinz III Center for Science, Economics and the Environment 2002). Miller et al. (1989) stated of the multiple causes involved with the extinction of America's freshwater fishes that physical habitat changes are the most common cause implicated (73% of extinctions), followed by introduced species effects (68%), chemical alteration of habitat (38%), hybridization (38%), and overharvest (15%). Similar trends have been noted in coastal systems in the Pacific Northwest, where 214 salmonid stocks representing genetically distinct populations are rare or threatened and another 106 populations have disappeared due to habitat alteration, dam construction, deforestation, and inappropriate uses of hatchery fish (Nehlson et al. 1991). Similar declines have been documented on the eastern seaboard of the United States (Waldham 2013). However, these initial large-scale fish habitat analyses cannot provide aquatic resource managers with sufficient detail to take actions on specific aquatic systems as the spatial scale was inappropriately large and available data and information were not specific to individual systems or watersheds. Furthermore, these analyses did not provide direct information on the impaired processes or functions to focus conservation actions.

In response to this clear need, the National Fish Habitat Partnership Board (Board) developed a National Fish Habitat Assessment Program (Assessment Program) designed to provide more concise aquatic habitat information to guide and inform conservation actions. The National Fish Habitat Partnership (NFHP), a voluntary and nonregulatory approach to conserving and improving fish habitat, was established in 2006 to bring new resources and attention to the decline in fish stocks linked to long-term aquatic habitat degradation. The Board's latest approved October 2017 vision for the Assessment Program is a comprehensive, comparable, and connected assessment of all U.S. fish habitat from the mountaintop to the shelf. Ultimately, the Assessment Program should provide a connected habitat portrait of the entire nation and, in the future, include waters that influence U.S. aquatic habitats from Canada and Mexico. The Assessment Program's purpose is to support the conservation, rehabilitation, and improvement of fish habitats by providing (1) fish habitat partnerships with national and regional data sets and accompanying analytical procedures that assess the condition of U.S. fish habitats using the best available data, and (2) the Board and stakeholders with communication products to highlight the status and importance of the nation's fish habitats.

The Assessment Program has produced two national assessments that provided the initial national fine-scale fish habitat portraits (National Fish Habitat Board 2010; Crawford et al. 2016). These assessments established the geospatial framework based on the National Hydrography Dataset Version 2 (NHD+; 1:100,000 scale) for inland waters and initiated the geospatial framework development for coastal areas. Both assessments used available standardized national data sets, consisting of landscape-scale land use and land cover data layers attributed to individual stream segments and estuaries. By the 2016 assessment, fish community data for the conterminous United States were developed from single pass electrofishing information from more than 39,000 stream reaches and from trawl information from selected coastal estuaries with a focus on the Gulf Coast (Crawford et al. 2016). Significant analytical improvements were made in the 2016 assessment, including inclusion of statistically defensible, dose–response relationships. Significant assessment gaps were present in the 2010 and 2016 assessments, including fish habitats in Alaska, lakes and most reservoirs, most coastal areas, and deepwater habitat from all coasts and Great Lakes. These habitats were not assessed due to insufficient staff resources along with the unavailability of NHD+ in Alaska; peer-reviewed spatial frameworks, particularly for lakes, reservoirs, and the inshore marine systems; consistently sampled fish community data; and analytical frameworks for the unassessed habitats. All of the Assessment Program data is housed at http://assessment.fishhabitat.org.

Although all landscape data layers (e.g., percent urban area, percent pasture and grazing) were attributed at the appropriate scale for assessed inland waters and estuaries in both assessments, most of the information was very difficult for aquatic resource managers to take direct action on. Additionally, most of the data layers have effects on a range of habitat-forming processes and functions, again making direct action difficult. For example, how can a fisheries manager effectively change the hydrologic alteration from the high percentage of impervious surfaces within urban areas, a key habitat condition driver in the 2010 and 2016 assessments, just one of many effects of urban land use on fish habitats. Given the lessons learned by aquatic resource managers from both assessments and the difficulty of influencing landscape-scale variables, the objective of this chapter is to provide a list of recommended variables to use and a template of what a model national fish habitat assessment (model assessment) would look like to allow it to be used as a decision support tool for aquatic resource managers. The fully implemented model assessment will provide fine-scale information to allow managers to take direct action on the causative habitat processes and factors, assuming sufficient resources are available, and to allow improved prioritization and allocation of habitat funding across large regions of or the entire United States.

Key Tenets and Assumptions for a Model National Fish Habitat Assessment

Aquatic habitat assessments must have a set of clear foundational concepts that provide analytical boundaries and set an overall direction for habitat assessment efforts. The following is a set of foundational tenets proposed for the model assessment:

- Information from the model assessment focuses on the controlling physical processes, watershed functions, and factors not the symptoms of degradation.
- Model assessment data provide clear direction on habitat conservation, rehabilitation, and improvement priorities to get the best return on incremental habitat investments, along with displaying locations where systems are intact and require conservation or protection from degradation.
- Model assessment uses spatially based information from all three fishery components as follows:
    - o fish and aquatic communities—to allow for dose–response relationships that detail the effects of variables on fish and aquatic organisms to be built,
    - o habitat—to build fish-community relationships and to focus conservation efforts, and
    - o people—to allow social information to be incorporated in the decision support system.
- Model assessment incorporates spatially based data on the key system constraints, such as physical geography controls (i.e., surficial geology, climate, and slope) on each system being evaluated that cannot be changed by management actions.
- The assessed variables are consistently measured across the landscape or at least have transfer or conversion functions to allow seamless data use between databases and sampling methods, can be influenced through management actions, and have a measurable effect on fish and other aquatic resources.
- Model assessment uses the expected value and range for all processes, factors, or variables as action criteria.
- The socioeconomics and societal values for waters should be incorporated spatially and used as another set of prioritization variables.
- Model assessment employs and mines existing data sets.
- Information should be available at any needed spatial scale or template (i.e., individual lakes, river segment(s), watershed(s), estuary complexes, congressional districts, and state boundaries).

Key Processes, Factors, and Functions

A model assessment focuses on key processes, factors, and functions, and these need to be robust enough to be useable for all aquatic habitats (i.e., rivers and streams, lakes and reservoirs, estuarine, and coastal systems). The Board's Science and Data Committee developed six processes, factors, and function categories that incorporate the concepts documented by the Instream Flow Council in Annear et al. (2004) and Locke et al. (2008). The following categories are recommended in the model assessment: connectivity; hydrology; geomorphology and bottom form; material recruitment and transport to include woody debris and sediment; water quality; and aquatic community effects on habitat and energy flow, including living habitat, invasive species, and size spectra of organisms that control energy flow.

Expected values

All model assessment processes, factors, and functions should either have (1) an expected condition estimate and associated variance based on a designated baseline date, preferably from an unaltered or least impaired period as possible; or (2) a range of estimated expected values leading to variance calculation in the future as new data and information become available. This will allow thresholds to be determined when intervention is required. These thresholds will focus habitat efforts on protection or conservation in systems that are within expected values and on rehabilitation or improvement for systems that are not within the expected values. The Board's Science and Data Committee recommends action be taken on any process or function variable that is 25% beyond the expected range or variation in a system. This threshold aligns with the estimates from Booth and Jackson (1997), Carlisle et al. (2010), Poff and Zimmerman (2010), Grantham et al. (2014), and Wagner and Midway (2014), who showed system changes with deviations of 10–50% for a range of parameters. In addition to measuring threshold points, it is equally important to measure rates of change in processes, factors, and functions temporally with the intent to target habitat efforts towards the most rapidly changing or threatened landscapes.

Model assessment category—hydrology

Hydrology in the model assessment is broadly defined in this chapter to include water discharge characteristics, water surface elevations, and velocity and current vectors to ensure applicability to all aquatic systems. Annear et al. (2004) states that the central character of rivers is driven by hydrology with clear contributions from the other processes and factors. This is equally true for all other types of aquatic habitats. This is a master variable that drives other key processes and factors that directly influence many others in the model assessment (Poff et al. 1997). The patterns of discharge or currents often dictate structure and function of systems more than the absolute amount of flow or currents (Arthington et al. 2006).

There are a broad range of potential ecologically useful variables to examine for hydrology, ranging from 67 in the Indicators of Hydrologic Alteration tool kit (Richter et al. 1996) to 171 in the Hydrologic Assessment Tool (Cade 2006). It is critical to set a baseline date for all systems to allow for a time-series analysis from that date, and the baseline date should be as unaltered or least impaired as possible, except for reservoirs, which are by definition man-made systems. While the many hydrologic variables in these tools provide insights into ecological questions and system changes over time, the focus should be on two sets of variables: the mean and the variance in the annual and daily hydrology and water surface elevation, and the timing of key hydrologic events. This focus generally follows the suggested approach or the analysis of hydrologic variables on aquatic communities in Poff and Allan (1995), Keough et al. (1999), Bunn and Arthington (2002), Sommer et al. (2004), Magilligan and Nislow (2005) , Helms et al. (2009), Konrad et al. (2008), Poff et al. (2006), Poff and Zimmerman (2010), and McManamay and Frimpong (2015).

Hydrology is one of the few processes with consistently measured data available as a result of the U.S. Geological Survey (USGS) national gaging system and its standardization. Assuming societal and legal approval allowing a watershed hydrology change, it is also one of the simplest processes to correct and rehabilitate. An example of how rapid changes can occur as a result of improvements in hydrologic conditions was demonstrated by Auer (1996) for Lake Sturgeon *Acipenser fulvescens* with a change in a hydropower project from hydropeaking operations to strict run-of-river operation. The Prickett Dam Hydroelectric Project on the Sturgeon River (Baraga County, Michigan, USA) operated daily with high flows, at or above bank-full flows, during generating periods and essentially no flow during off-generating periods, similar to extreme drought conditions. Lake Sturgeon, a state-listed threatened species, used this river for spawning and rearing and peaking flows resulted in poor recruitment. Once the flow regime from the powerhouse was changed to run of the river (i.e., inflows equaling outflows), successful spawning and recruitment of Lake Sturgeon was immediately documented by Auer (1996).

U.S. hydrology data continue to be limited at this time with key gaps regionally and for the state of Alaska, along with high altitude and small watersheds in other areas throughout the United States (Deweber et al. 2014). Miller et al. (2018) addressed many gaps in hydrology data for the conterminous United States using random forest modeling to estimate natural discharges from 1950 to 2015.

For each habitat type, the following category variables are recommended for the model assessment:

- *River and stream flows* using mean daily discharge data on an annual time step
  - o average monthly hydrograph and variance
  - o average and variance or range of the timing for and volume of peak and base flows
  - o duration and variance for or range of peak and base flows
- *River and stream flows* using instantaneous discharge data on a monthly timestep
  - o mean daily flow and variance or range
  - o mean daily flow fluctuation and its variance or range as defined by the difference in minimum and maximum flow
- *Lake and reservoir* levels using mean daily elevation data on an annual time step
  - o average monthly water surface elevation and variance
  - o average and variance of the timing for and volume of peak and lowest water surface elevation
  - o duration and variance for peak and lowest water surface elevation
- *Reservoir and impoundment* storage using mean daily data on a monthly time step
  - o monthly average storage change and variance
- *Lake and reservoir* levels using instantaneous elevation data on a monthly time step
  - o mean daily water surface elevation and variance or range
  - o mean daily water surface elevation fluctuation and its variance or range as defined by the difference in minimum and maximum water surface elevations
- *Coastal systems including the Great Lakes* current patterns using mean daily velocity and flow pattern daily data on a monthly time step
  - o average flow patterns and velocities, direction, and their variance
  - o average and variance of the timing for and volume of peak and lowest flow patterns and velocities
  - o annual duration and variance for peak and lowest flow patterns and velocities
- *Coastal systems including the Great Lakes* water surface elevations using mean daily data on a monthly time step
  - o average monthly water surface elevation and its variance
  - o average and variance of the timing for and volume of peak and lowest water surface elevation
  - o duration and variance for the peak and lowest water surface elevation
- *Wetlands*
  - o number and acreage of wetlands by type within watersheds to complete system water storage estimates along providing data on a key habitat class for many fish species

Model assessment category—connectivity

Highly connected systems are needed to move sediment and woody debris from the landscape. Connected systems are essential to maximize the productivity of aquatic systems by providing the best available habitat for all life stages and life history strategies of aquatic life (Schlosser 1991; Able 2005). Fish communities lose species and productivity when fragmented (Winemiller and Rose 1992).

Fish movement is critical to the movement of nutrients and energy from large to small water bodies, and this transport mechanism is well documented (Pacific: Cederholm et al. 1999; Gende et al. 2002; Naiman et al. 2002; Schlinder et al. 2003; Wipfli et al. 2003; Twining et al. 2017; Atlantic: Durbin et al. 1979; Twining et al. 2013; Waldham 2013; Great Lakes: Childress and McIntyre 2016; Jones and Mackereth 2016; Europe: Jonssen and Jonssen 2003). Large water body subsidies to smaller waters from migrating fish come in the form of excretion products, carcasses, eggs, fry and young-of-year fish, which are absolutely required to overcome the low productivity from nutrient and energy poor systems frequently found in interior systems.

Another benefit of system connectivity is as a recruitment refugia, allowing for genetically related fish populations to exist in separate parts of watersheds while ensuring connections to common reproductive habitat. Some part of the overall fish population will reproduce in any given year, despite any system catastrophes. Pacific coast examples of this benefit include Rainbow Trout *Oncorhynchus mykiss* and steelhead (anadromous Rainbow Trout) and Sockeye Salmon *O. nerka* and kokanee (lacustrine Sockeye Salmon). In the Great Lakes, nearly all the Great Lakes fish species historically used tributary streams for spawning with a subpopulation remaining resident in the tributary stream with examples, including Round Whitefish *Prosopium cylindraceum* and Smallmouth Bass *Micropterus dolomieu*.

Connectivity is one of the most feasible processes to correct, if impaired, and provides very quick results in most watersheds. This process also has a substantial amount of data available. There are major efforts at this time nationally to survey watersheds for their current connectivity status, including work done on northeastern dams (Martin and Apse 2011), northeastern road crossings (North Atlantic Aquatic Connectivity Collaborative, http://northatlanticlcc.org/products/north-atlantic-aquatic-connectivity-collaborative), southeastern barriers (http://southeastaquatics.net/sarps-programs/southeast-aquatic-connectivity-assessment-program-seacap/prioritization-connectivity-tools-and-other-resources/connectivity-resources), Great Lakes barriers (https://greatlakesconnectivity.org), California barriers (Statewide Barrier Inventory, www.cafishpassageforum.org/statewide-barrier-inventory), northwest barriers(www.streamnet.org/data/interactive-maps-and-gis-data/), and the Kenai River watershed, Alaska (https://kenaiwatershed.org/science-in-action/fish-barriers/culvert-assessment/). There are also a number of decision support models available, including the Northeast Aquatic Barriers Prioritization Tool (http://maps.freshwaternetwork.org/northeast/), the Chesapeake Bay Fish Passage Prioritization Tool (http://maps.freshwaternetwork.org/chesapeake/), the North Carolina Barrier Prioritization Tool (http://portal.ncdenr.org/web/cpt/), Fishwerks—Great Lakes Basin (https://greatlakesconnectivity.org), and FishPass—Anadromous Fish Optimization Tool (www.cafishpassageforum.org/fishpass). Given the amount of currently available and generally consistently measured information, the development of this layer of the model assessment is well along for the United States and will likely be available soon.

To assess connectivity, the following variables are recommended for the model assessment:

- *Natural barrier* determination using spatially referenced locations of waterfalls, high-gradient rapids, and shoots. The intent of the model assessment is to ensure natural connectivity patterns are maintained or re-established. Natural barrier data should include the slope, length, and height of each barrier to allow for additional hydrologic analysis of passability in extreme flow conditions along with the analysis of natural passage requirements of each target fish species.

- *Anthropogenic barrier* determination using spatially referenced locations of
  - o fisheries management barriers such as weirs
  - o dams
  - o culverts
  - o fishways, which are still barriers with fish passage delays
  - o concrete channels and conveyance channels that are often shallow, wide, and without any roughness, creating effective velocity and depth movement barriers
  - o water-quality impairments that will create fish avoidance conditions
  - o water-quantity impairments, such as large diversions that can create dewatered habitat downstream blocking fish movement

  Each barrier should have data collected on the slope, length, and height of each barrier, along with current velocity distribution information at and within the barrier. For fishways, information should include design parameters, attraction flow, and location on the structure, fallback estimates, and fish passage delay times to allow for a passability estimate to be generated.
- *Coastal velocity barriers* determined using spatially referenced locations of developments that have altered natural flow patterns, particularly in inshore and estuarine systems, including constructed navigation channels and structures, and tidal pond structures and gates. Each velocity barrier should have data collected on the length, height (only for tidal pond structures), and depth of each barrier along with any current velocity distribution information.

Model assessment category—material transport and recruitment

Watersheds are conveyance systems for the movement of soil and vegetative materials from the landscape, with rivers being the conveyors and lakes along with coastal areas being storage locations for these materials. Rivers, lakes, and coastal areas have temporary and long-term recruitment, storage, and internal processing rates for materials. These processes can be rapidly altered by hydrology and land use changes, either from natural or anthropogenic causes. Starting with Leopold et al. (1964), many geomorphologists have detailed the importance of the balance between input and movement rates of materials in transport. Destabilizing this balance leads to degraded river habitat (Collier et al. 1996; Van Steeter and Pitlick 1998; Kondolf et al. 2006). Rivers along with their associated fish habitat respond quickly to changes in this process. Fish habitat relies on this process working within geologic and climatic bounds.

Channel and shoreline processes, energy processing, and physical fish habitat rely on coarse organic material and large woody debris inputs being maintained within expected amounts for materials from the forest to the sea (Maser and Sedell 1996; Martin 2001; Gregory et al. 2003). In systems with large woody debris inputs, these are channel geomorphology controlling processes (Montgomery et al. 2003; Mutz 2003). Understanding the recruitment of materials to waters from riparian zones and ensuring the rate is appropriate for the forest type are key variables for fish and aquatic habitat for which measurable methods are available (Benda et al. 2003; Gurnell 2003; Piegay 2003).

Material transport and recruitment inputs and rates are possible to manage (Gregory et al. 2003; Kondolf et al. 2014; Rhienheimer and Yarnell 2017), but efforts will have a time delay, particularly if being rehabilitated on a watershed basis. This is an expensive process to correct, thus keeping systems within expected background values is critical. Currently, data on this process are only available locally or for specific point locations, with generalized data available through such sources as the USGS SPARROW (Spatially Referenced Regression on Watershed Attributes) models, which also serve as decision support models.

While sediment measurement is mostly standardized, woody debris and course organic material have not been nationally standardized, and efforts will be needed to make these data comparable nationally using the guidance in Gregory et al. (2003) to fully incorporate these data into the model assessment. It is expected that this category will take at least a decade to be incorporated into the model assessment, except in small-scale analyses.

To assess material transport and recruitment, the following variables are recommended for the model assessment:

- *All materials*
    - natural undisturbed annual rates and variances of sediment, woody debris, and coarse organic material recruitment and transport based on the forest type, geology, and climate for each watershed and coast
- *Sediment*
    - annual erosion rates and variance for each soil type in each watershed and coast
    - annual anthropogenic erosion rates and variance by land use
    - annual sediment transport and storage rates and variance for both in-channel and within-lake/reservoir/coastal storage areas. Data on how these rates are affected by land use changes is also desirable.
- *Woody debris*
    - annual riparian forest growth dynamics and expected variance, with an emphasis on the potential area of recruitment based on valley slope
    - annual woody debris recruitment rates and variance based on forest type and mortality rates
    - annual woody debris and course organic material transport and storage rates and variance for both in-channel and within-lake/reservoir/coastal storage areas. Estimated rate effects from land use changes are another desirable variable.

Model assessment category—water quality

While having appropriate quantities of water is clearly important to fish and aquatic communities, it must have the appropriate chemical characteristics to support all life stages and to maximize productivity of the expected aquatic community (Magnuson et al. 1979; Mainstone and Gulson 1990; Whitfield and Elliott 2005; Bain and Jia 2012; Hamilton et al. 2015; Payne et al. 2015; Speers-Roesch and Norin 2016). There is a substantial number of potential variables, and all of these are measured using standardized techniques such as the U.S. Environmental Protection Agency Clean Water Act standard methods (USEPA 2017). Guidance is available on the appropriate collection of river and stream temperature (Heck et al. 2018) and dissolved oxygen (Rounds et al. 2013). The more challenging issue is to develop the expected value for each variable with and without human land use changes. An additional quandary is which variables will provide largest return on investment.

Similar to material transport and recruitment, water quality is possible to manage, but efforts usually will have a significant time delay, particularly if being rehabilitated on a watershed basis. This category is often an expensive process to correct; thus, preventing degradation and keeping systems within expected values is critical. Currently, data are available broadly across the United States, but generally only for specific point locations within a watershed. More generalized data are available through such sources as SPARROW models. Given the fragmented nature of these data, it is not likely that this category can be fully incorporated into a model assessment for another decade, except in small-scale analyses.

The following water quality variables are recommended for the model assessment, with other variables, such as contaminant or chloride concentrations, being important in specific applications:

- *River, stream, and coastal temperature* is the key driving factor that structures fish and aquatic communities and should be measured similar to hydrologic factors, using both daily and annual time steps.
    - o annual summary temperature variables to measure using daily mean data on a monthly time step are average monthly temperature and variance, average and variance of the timing for the monthly maximum and minimum temperatures, and duration and variance for maximum and minimum temperatures for each month
    - o monthly summary temperature variables to measure on a daily time step are average daily temperature and variance, and average daily temperature flux (daily maximum minus the minimum temperature) and variance
- *Lake and reservoir temperature* using daily data on a monthly or seasonal time step.
    - o average monthly or seasonal temperature profile and variance
    - o average and variance of the timing for spring and fall turnover events
- *River, stream, and coastal dissolved oxygen*—Given the relationship of temperature and altitude to dissolved oxygen concentrations, the key variable to understand is percent saturation.
    - o annual summary dissolved oxygen variables to measure using mean daily data on a monthly time step are average monthly percent saturation and variance, average and variance of the timing for maximum and minimum percent saturation, and dura tion and variance for maximum and minimum percent saturation for each month
    - o monthly summary dissolved oxygen variables to measure on a monthly time step using daily data are average percent saturation temperature and variance, and average daily percent saturation fluctuation and its variance
- *Lake and reservoir dissolved oxygen*
    - o annual summary dissolved oxygen variables to measure on a monthly or seasonal time step using daily data are average monthly dissolved oxygen profile and variance, and annual average and variance of the timing for spring and fall turnover events
- *Nutrients*—While there are a host of potential nutrients to examine, the two key nutrients are total phosphorus and nitrogen, which control many of the primary production rates across fish habitats (Mueller and Helsel 1996 and Cloern 2001).
    - o annual summary nutrient variables to measure for phosphorus and nitrogen on a monthly or seasonal time step using daily mean data are average monthly loadings of total phosphorus and nitrogen and their variance, and average and variance of the amount and timing for maximum phosphorus and nitrogen loading events
- *pH*—The acidity of water controls a broad range of chemical processes both in freshwater and saltwater and is known to affect aquatic organisms from plankton to shellfish to salmonids (Gunn 1986; Ikuta et al. 2003; Logan 2010; Talmage and Gobler 2010; Mackenzie et al. 2014).

  - annual summary pH variables to measure using daily mean data on a monthly time step are average monthly pH and variance, average and variance of the timing for the monthly maximum and minimum pH, and duration and variance for maximum and minimum pH for each month
  - monthly summary pH variables to measure on a daily time step are average daily pH and variance, and average daily pH flux (daily maximum minus the minimum temperature) and variance
- *Salinity*—Nearly every estuarine and coastal species has very specific tolerances and ranges for salinity, which is a controlling factor for fish and aquatic species distributions in these environments (Peterson and Meador 1994; Boeuf and Payan 2001; Martino and Able 2003; Elliott et al. 2007). In nearshore and estuarine systems, it is highly influenced by inflow hydrology patterns.
  - annual summary salinity variables to measure for salinity on a monthly time step using daily mean data are average and variance of the monthly salinity, average and variance of the timing for maximum and minimum salinities, and duration and variance for maximum and minimum salinities
  - monthly summary salinity variables to measure on a daily time step are average daily salinity and variance for each month, and average daily salinity flux and its variance for each month

Model assessment category—geomorphology

Bottom shape or rugosity and system geomorphology or channel pattern, regardless of whether in rivers, lakes, reservoirs, or coastal systems, has a key influence shaping fish and aquatic communities, as it often controls current velocity distribution and sediment types (Imhof et al. 1996; Lamouroux et al. 2002; Walters et al. 2003; Kuffner et al. 2007; Dauwalter et al. 2008; Wright and Heyman 2008; D'Ambrosio et al. 2014; Trebilco et al. 2015). Geomorphology and bottom shape is directly related connected to and controlled by hydrology and material recruitment and transport with indirect relationships to connectivity and water quality. It is also a habitat factor that has been greatly altered by humans, whose actions generally simplify habitat.

Similar to material transport and recruitment and water quality, it is certainly possible to manage geomorphology, but efforts usually will have a significant time delay, particularly if being rehabilitated on a watershed basis. This category is often an expensive process to correct; thus, keeping systems from being degraded and within undisturbed expected values is critical. A current barrier is that data are only available for localized areas and broadly acceptable standardized approaches for collecting and analyzing this information are generally not available, except for rivers and streams. Additional research is needed to fully develop the methodology for this category for inclusion into the model assessment and is unlikely to be fully incorporated into the model assessment for another decade, except in small-scale analyses.

To evaluate this factor, it is important to understand what the unaltered bottom and channel shape was to comprehend what changes have been made. This is a significant information gap across the continent, which is not likely to be filled until far into the future. Understanding the current deviation of the geomorphic variables from expected background values is a focus of the model assessment in this category, as pointed out in Airoldi and Beck (2007). The geomorphic variables recommended for the model assessment are

- *Rivers and streams*
  - o channel sinuosity, pool-riffle-run ratios, bottom substrate, and bottom shape and depth diversity by river segment, as defined by NHD+ or, where available, National Hydrography Dataset High Definition (NHD+HD; 1:24,000 scale) for both undeveloped and current time periods
  - o amount of channel modification, including channelization
- *Lakes, reservoirs, and coastal systems*
  - o areal measures of rugosity, bottom shape, bottom substrate, and depth diversity for both undeveloped and current time periods
  - o anthropogenic affect and analysis require data on channelization and navigation channel construction (lineal distance and area); levee development (height and distance from thalweg); hardened shoreline amount in lineal distance; jetty numbers, mean size and variance, and density per kilometer; and dredged area and volume

Model assessment category—aquatic organisms effect on habitat and energy flow

This category measures the size distribution of all biota in the ecosystem, which controls energy flow in the system (Sheldon et al. 1972; Kerr 1974; Kerr 1977; Borgmann 1987), along with how organisms' control and form physical habitat. Organisms are known to control water chemistry and fish habitat through top-down and bottom-up control of system processes and factors.

The Great Lakes provide recent examples of both bottom-up and top-down control of ecosystems, as summarized below and detailed in Mills et al. (1994), Strayer (2009), and Lin and Guo (2016). Historically, the Great Lakes were dominated by Lake Trout *Salvelinus namaycush* (the apex predator), a diverse coregonid community that were key pelagic prey items for Lake Trout, inshore percids, and a diverse group of benthic fish. This community structure was altered by rapid declines in predators, particularly Lake Trout, in the 1940s, caused by a combination of intensive harvest, invasions, mortality by invasive and parasitic Sea Lamprey *Petromyzon marinus*, and the broad expansion of Alewife *Alosa pseudoharengus*, an invasive planktivorous fish that has high levels of thiaminase, which is harmful to Lake Trout recruitment. By the mid-1960s, Alewives comprised up to 95% of the fish biomass in the Great Lakes and were able to selectively alter the plankton size structure, which in turn changed water quality and primary production. The introduction of Pacific salmon into the Great Lakes in the mid-1960s reduced Alewife populations and moved plankton size distributions and primary producers toward pre-Alewife conditions. Bottom-up effects can be seen a few decades later by the changes incurred by zebra mussel *Dreissena polymorpha* and quagga mussel *D. bugensis* introductions into the Great Lakes in the late 1980s. These mussels filtered much of the suspended materials and plankton in Great Lakes waters, which in turn caused large Alewife reductions in Lake Michigan and a complete loss of this key pelagic prey in Lake Huron. This forage change resulted in pelagic predators, in particular Chinook Salmon *Oncorhynchus tshawytscha*, responding in the 2000s with lake-wide population declines in Lake Michigan and a collapse of Chinook Salmon populations in Lake Huron.

The biota and flora of a system provide direct physical habitat, as evidenced by oyster and coral reefs in inshore marine areas and submerged aquatic vegetation in all habitat types. One of key mechanisms biota and flora affect habitat is by reducing water velocities through increased bottom roughness, which in turn changes the deposition rates of suspended and bed load materials and the distribution of fish communities.

The particle-size spectrum hypothesis indicates that each system has an overall community size distribution that is the most effective at moving energy from one productivity level to the next (Sheldon et al. 1972; Kerr 1974; Borgmann 1982). When size spectra are disrupted by catastrophic events, anthropogenic alteration, or fish harvest, the system shows cascading ecosystem effects that change fish habitat through water quality and potentially physical habitat changes.

Another component of this category is the direct and indirect effects of invasive species. These species can directly alter fish habitat (e.g., Common Carp *Cyprinus carpio* increasing water turbidity [Weber and Brown 2009]) and particle size spectrum, as discussed above, with the Alewife and zebra and quagga mussel invasions of the Great Lakes and invasions of other inland lakes (Strayer 2009; Lin and Guo 2016). Invasive species can indirectly alter fish habitat through control measures such as barrier construction to stop Sea Lampreys in the Great Lakes from reaching spawning habitat, which in turn fragments watersheds.

Given the low amount of information available on total particle size distributions in ecosystems and the lack of measurement consistency, it will be difficult to make progress on this subvariable for decades. The movement of energy is a key structuring factor for fish habitat in each ecosystem and is likely one with the least amount of information on it, as it is very rare to have biomass data from bacteria to the largest fish or mammals in a system. Partial information on size structure of invertebrates and fish communities is more readily available, but unless excessive extraction or perturbation is an issue preventing the expression of expected size frequencies, it is a similarly difficult variable to affect. Extraction is a fairly straightforward issue to deal with, particularly by using size limits or total allowable harvest regulations.

Standardized invasive species distribution and population information is available for some invasive species in parts of their range. Information on the edges of range expansions is not likely to be fully available. Data on control measures are available as these actions permitted by regulatory entities although their direct effect on fish habitat may not be fully known in every instance, but overall invasive species effects are theoretically possible to manage in some instances at this time.

Information on living habitat as defined above has not yet been nationally standardized but is available at some level in some locations. There are clear methods available to rehabilitate degraded living habitats, so this is an area where progress can be made in the near time. Given the many data gaps in this category, it is not likely that these data can be included in the model assessment for at least a decade.

Again, the starting point is generating data for both expected and current conditions on each variable, as this difference is the key to rehabilitation or protection efforts. This is also one of the more difficult categories to obtain standardized data on, particularly for particle size distribution in a system.

The category variables recommended for the model assessment are

- *Particle size distribution*
    - o Mean monthly particle size distribution and variance; since this is likely to be lacking for some decades, the available invertebrate and fish community size structure and its variance would be a good starting place; even a consistently measured fish size distribution data set that is available for all or part of the fish community is a good initial measurement for this factor with an adjustment for gear bias
- *Living habitat distribution* using data on the seasonal location and density of living habitat.

   - Freshwater systems—annual areal estimates and variances for the dis tribution and density of submerged aquatic vegetation, mussel beds, submerged and emergent wetland complexes, and water resident trees such as bald cypress *Taxodium distichum*
   - Coastal systems—annual areal estimates and variances for the distribution and density of submerged aquatic vegetation, oyster, clam and mussel beds, coral reefs, submerged and emergent wetland complexes, and water resident trees such as mangroves
- *Invasive species*—By definition, this group of variables has no invasive species as the null baseline condition, but information should be developed to ensure that organisms were properly classified as introduced or invasive and not native to the system; additionally, first dates of occurrence should be noted.
   - seasonal location and density of aquatic invasive species along with the variation in the patch occurrence
   - seasonal effects of aquatic invasive species along with the variation in these effects
   - location and frequency of control measures, including physical, biological, and chemical measures

Fisheries and Aquatic Organism Data

To develop actionable strategies to protect, conserve, and improve fish habitat, a dose–response analysis between habitat variables and fish or aquatic communities, as detailed by Greene et al. (2015), Daniel et al. (2015), and Crawford et al. (2016), are required. The dose–response analysis includes a change-point analysis to determine greatest differences in community responses to stressors and a piecewise linear regression to allow threshold response determination (Qian and Cuffney 2011). The dose–response relationships use the stressor variables as predictor variables and measures of fish or aquatic community occurrence, abundance, or biomass as the response variable. Significant inflection points in the relationship provide information on where the level of the stressor has no effect and the value that reduces the fish or aquatic organism variables to their lowest level.

Consistently measured fisheries and aquatic organism data are needed to allow appropriate development of these relationships. Crawford et al. (2016) used survey data that targeted entire fish communities in rivers and streams using single-pass electrofishing data and trawl survey data in estuarine systems for the development of dose–response relationships for landscape-scale variables, as these were the only standardized data set available on a national scale. Ecological and habitat assemblages (e.g., lithophilic spawning guilds) for aggregated ecoregions are used to summarize data and improve samples sizes. Standardized fisheries and other aquatic organism data are needed for lakes, reservoirs, and coastal areas to enable dose–response relationship development. One possible approach for fish in lentic systems is collecting new data and using and accessing existing data from Bonar et al. (2009). This publication, currently being updated, details standardized methods for lakes and reservoirs and provides a compendium of data following their methods.

While a large amount of fisheries and aquatic organism data have been collected nationally, little is broadly available in centralized databases at this time, and even less of these data were collected in a standardized manner. The lack of standardization in collection methodologies is currently an acknowledged issue in many fisheries and natural resource agencies. Investment in developing transfer or conversion functions between gear types and methods, along with collecting new standardized information, is needed and will likely not be fully available for another decade. Investigators who worked on Crawford et al. (2016) faced communication and coordination issues in obtaining available data, resulting in data gaps, which also need to be improved for the model assessment and illustrate the need for centralized data repositories.

To fully build the needed dose–response relationships for all habitats, the fish and aquatic organism variables and data needed for the model assessment are

- the seasonal means and variances for fish and aquatic organism species composition variables, including species abundance, size distribution of the community or key species, and species movement dynamics (e.g., home range and life history stage-specific movements). These variables should be developed at the smallest spatial scale, which would be individual waters for lentic systems, each storm reach for lotic systems, and, for each classified unit, typically a specified grid size such as 30 m^2 or 1 km^2 for large lakes (e.g., Great Lakes) and estuarine and marine systems.
- mean annual timing of fish and aquatic organism movements or key activities along with the variance of these events
- the ecological and habitat fish and aquatic organism guilds

Socioeconomics Information

What society views and values as critical aquatic habitat are key pieces of information for aquatic resource managers and directly determine where habitat investments in conservation, rehabilitation, and improvement will occur. These beliefs are expressed by what society indicates the habitat priorities and issues are in each water, with the number of entities and plans that hold those views providing insights on the societal importance of that water and its habitat. Spatially recording each of the documented habitat issues and concerns from individual entities and planning documents for a particular location or water can make this information actionable by and informative to resources managers. One can reasonably assume the larger the number of entities and cumulative number of priorities that a water has is an indication of a higher societal profile and need.

Unlike many of the other processes, factors, and variables, there is a lot of available information in this category but little spatially attributed. Priorities are available from a wide range of entities, including state, federal, and tribal natural resource agencies to nongovernmental organizations (e.g., Trout Unlimited, The Nature Conservancy, and the Bass Anglers Sportsmen's Society) to local habitat rehabilitation groups (e.g., resource conservation authorities and watershed groups) to local and regional governments. These existing data need to be organized by aggregating the data and attributing it to the model assessment spatial framework. These data are available and can be immediately incorporated into a model assessment.

Another source of socioeconomic information is the estimated historic, current, and future economic value of fisheries and aquatic communities in each watershed. This information provides a clear indication of the importance of systems to society. Where available, economic values are usually restricted to extraction values or replacement values, with very few waters systems having existence or ecosystem service values. In inland areas where fishing or fisheries extraction are important, some valuation information is available from spot creel census data, but these data usually have a limited temporal and spatial range. Larger systems often have broad-ranging creel census information, such as the Great Lakes and coastal systems, to support harvest allocation work, and these data provide the ability to examine the fisheries value of these waters or locations or regions within these waters. Commercial fisheries on the coasts are additional sources of economic value information that can be used to support habitat prioritization efforts. The fragmented nature of these data makes them unlikely to be fully incorporated into the model assessment for at least another decade, except in small-scale analyses.

Protected lands are a piece of socioeconomic data that are available and should be included as a variable in the model assessment. This information provides key insights into public values for given systems and at least some information on ecosystem service values. These lands, as listed in order of protective class, include

1. Preserved lands and natural areas
    a. national and some state and tribal parks
    b. designated wilderness and natural areas
    c. some federal and state wild and scenic rivers
    d. some national and state wildlife refuges
2. Multiple use/purpose designations
    a. national and state forests
    b. Bureau of Land Management lands
    c. Native American and Alaska lands
    d. some national and state wildlife refuges
    e. some state parks
    f. some federal and state wild and scenic rivers
3. Private lands with conservation easements
4. Focused recreational use areas—state, county, and city parks
5. Private lands

These land designations provide an indication of the amount of management needed to protect and rehabilitate fish habitat, as lower numbers indicate protected areas with low rehabilitation costs. Protected lands are likely to have higher social values. This information is spatially available and can be immediately incorporated into the model assessment.

The final socioeconomic variables in the model assessment are the estimated costs of protecting/conserving intact systems and repairing damage to degraded waters. This information, which is generally not yet available nationally but is available on a local level in some areas, provides key information on the highest return on investments. Efforts should be made to develop national data sets on the costs to protect intact systems, through either direct purchase or easement acquisition, and on projects that are rehabilitating the processes and factors detailed above. Given the fragmented nature of these data, they are unlikely to be fully incorporated into a model assessment for a decade, except in small-scale analyses.

The socioeconomic variables, spatially attributed for each water or segment, for the model assessment are

- number of priorities and actions called for by individual entities and planning documents, with higher counts having higher scores or weighting
- economic value of the aquatic community, with the greater values having the higher scores or weighting
- area of each protected habitat, with the highest amount of Category 1 lands having the highest scores or weighting
- costs of habitat protection or rehabilitation measures with systems, with the lowest costs having the higher score or weighting

The best methods of incorporating and aggregating the above socioeconomic variables into water and system scores need further development.

Assessing Aquatic Habitat Condition

Model assessment habitat condition scoring for waters follows the methodology detailed in Esselman et al. (2013), Greene et al. (2015), Daniel et al. (2015), and Crawford et al. (2016) and is summarized in this section. After developing dose–response relationships between all stressor variables for each fish and aquatic organism aggregation, boosted regression tree analysis (Elith et al. 2008) is conducted to control the variance from natural variables and to allow the selection of the stressors that explain the largest amount of variance seen in the selected fish and aquatic organism assemblage values. The statistically significant stressor variables have their responses divided into effect scores using the threshold analysis of inflection points discussed previously, ranging from no effect to complete loss of fish and aquatic organism aggregation variable value. Crawford et al. (2016) used five effect scores from 5 for stressor levels of no effect to 1 for lowest fish community values documented at the stressor level. Values between 5 and 1 were evenly spaced along the response relationship.

To score waters, only statistically significant stressors are used for each fish and aquatic organism species assemblage for each ecological region. Scores are computed for each significant stressor by assemblage for each water or part of a water within an aggregated ecosystem using the appropriate dose–response relationship. Using the assumption that the stressor causing the most disturbance to the assemblage is the limiting factor for that assemblage group, the lowest score is used for that assemblage. Assuming that all assemblages are of equal value, the scores across all assemblages in the ecological region being evaluated are averaged to provide scores for each water or part of a water. As more information becomes available, scores for both stressors and assemblages should be weighted to improve the model realism.

As demonstrated in Crawford et al. (2016), there are different responses from assemblages to stressor spatial scales. Analyses should be done at both local (NHD+ segment or individual lake or estuary-shed) and network (cumulative effect of all waters upstream, including the local area) scales. The lowest stressor score, representing the most disturbance, between the spatial scales for a water or part of a water is used for final scoring.

The model assessment provides habitat scoring for all waters or parts of waters (e.g., river segment). Scores can then be aggregated by averaging individual water or segment scores for any spatial extent desired. When more data are available on how stressors effect network systems and aquatic species aggregations, scores should be weighted for each part of a system being analyzed. The socioeconomic data and information are one source of weighting system scores, with the highest socioeconomic score providing overall higher scores.

Conclusion

The attribution of the above listed data into the model assessment with the subsequent development of dose–response relationships, regression tree analysis of these relationships for overall habitat condition model building, scoring of waters and parts of waters, and incorporation of socioeconomic information as a key information weighting factor will provide aquatic resource managers with an information synthesis and decision support tool. When fully developed, the model assessment will determine the condition of any water from the mountains to the continent shelf, determine the expected outcome of changing a process or factor along with its effects on fish communities, and provide estimates of conservation or rehabilitation costs, including the societal value of a particular water. Waters can then be scaled and scored to allow for a much-improved mechanism to prioritize fish habitat conservation across a continental landscape. While the visualized data and the actual model assessment may be decades away on a continental framework, many of these data exist today or will be accessible soon for smaller geographic areas or for individual categories that can inform resource management decisions, if the data are gathered and displayed in a compelling fashion. For the first time, we can visualize a reliable mechanism to compile and organize the mass of available aquatic data and information spatially and have the analytical tools to develop all required supporting relationships to build a true decision support system to help direct the trillions of dollars of investment needed to protect, enhance, and conserve our fisheries habitats.

Acknowledgments

For the ideas and concepts behind this chapter, I owe a huge debt of gratitude to all of the past and current members of the NFHP Board Science and Data Committee along with the Inland and Coastal National Fish Habitat Assessment Teams, many of whose papers were cited in this chapter. In particular, I would like to thank Dana Infante (Michigan State University), Wes Daniel (U.S. Geological Survey), and Kristan Blackhart (National Oceanographic and Atmospheric Administration–National Marine Fisheries Service), along with the previous NFHP Inland and Coastal Assessment Leaders, for their many contributions. Without the long discussions on how to build a system that was not possible a generation ago but is attainable now, this chapter would not be possible or even conceived. I also need to acknowledge the Board and Board staff along with the primary funders of the assessments, the U.S. Fish and Wildlife Service, the National Oceanographic and Atmospheric Administration–National Marine Fisheries Service, and the U.S. Geological Survey, which have provided this unique opportunity to try to build this none like it in the world assessment. I owe all of them my deepest thanks.

References

Able, K. W. 2005. A re-examination of fish estuarine dependence: evidence for connectivity between estuarine and ocean habitats. Estuarine, Coastal and Shelf Science 64:5–17.

Airoldi, L., and M. W. Beck. 2007. Loss, status and trends for coastal marine habitats of Europe. Oceanography and Marine Biology: An Annual Review 45:345–405.

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

Arthington, A. H., 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.

Auer, N. A. 1996. Response of Lake Sturgeons to change in hydroelectric facility operation. Transactions of the American Fisheries Society 125:66–77.

Bain, M. B., and H. Jia. 2012. A habitat model for fish communities in large streams and small rivers. International Journal of Ecology [online serial] 2012:962071.

Benda, L., D. Miller, J. Sias, D. Martin, R. Bilby, C. Veldhyuisen, and T. Dunne. 2003. Wood recruitment processes and wood budgeting. Pages 49–74 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland.

Boeuf, G., and P. Payan. 2001. How should salinity influence fish growth? Comparative Biochemistry and Physiology Part C 130:411–423.

Bonar, S. A., W. A. Hubert, and D. W. Willis, editors. 2009. Standard methods for sampling North American Freshwater Fishes. American Fisheries Society, Bethesda, Maryland.

Booth, D. B., and C. R. Jackson. 1997. Urbanization of aquatic systems: degradation thresholds, stormwater detention, and the limits of mitigation. Journal of the American Water Resources Association 22:1077–1090.

Borgmann, U. 1987. Models on the slope of, and biomass flow up, the biomass size spectrum. Canadian Journal of Fisheries and Aquatic Sciences 44(Supplement 2):136–140.

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.

Cade, B. S. 2006. National hydrologic assessment tool (NATHAT). U.S. Geological Survey, Reston, Virginia.

Carlisle, D. M., J. Falcone, D. M. Wolock, M. R. Meador, and R. H. Norris. 2010. Predicting the natural flow regime: models for assessing hydrological alteration in streams. River Research and Applications 26:118–136.

Cederholm, C. J., M. D. Kunze, T. Murota, and A. Sibatani. 1999. Pacific salmon carcasses: essential contributions of nutrients and energy for aquatic and terrestrial ecosystems. Fisheries 24(10):6–15.

Childress, E. S., and P. B. McIntyre. 2016. Life history traits and spawning behavior modulate ecosystem level effects of nutrient subsidies from fish migrations. Ecosphere [online serial] 7(6):e01301.

Cloern, J. E. 2001. Our evolving conceptual model of the coastal eutrophication problem. Marine Ecology Progress Series 210:223–253.

Collier, M., R. H. Webb, and J. C. Schmidt. 1996. Dams and rivers: primer on the downstream effects of dams. U.S. Geological Survey Circular 1126.

Crawford, S., G. Whelan, D. M. Infante, K. Blackhart, W. M. Daniel, P. L. Fuller, T. Birdsong, D. J. Wieferich, R. McClees-Funinan, S. M. Stedman, K. Herreman, and P. Ruhl. 2016. Through a fish's eye: the status of fish habitats in the United States 2015. National Fish Habitat Partnership Board. Available: http://assessment.fishhabitat.org/. (April 2019).

Daniel, W. M., D. M. Infante, R. M. Hughes, Y. Tsang, P. C. Esselman, D. Weiferich, K. Herreman, A. R. Cooper, L. Wang, and W. W. Taylor. 2015. Characterizing coal and mineral mines as a regional source of stress to stream fish assemblages. Ecological Indicators 50:50–61.

Dauwalter, D. C., D. K. Splinter, W. L. Fisher, and R. A. Marston. 2008. Biogeography, ecoregions and geomorphology affect fish species composition in streams of eastern Oklahoma, USA. Environmental Biology of Fishes 82:237–249.

D'Ambrosio, J. L., L. R. Williams, M. G. Williams, J. D. Witter, and A. D. Ward. 2014. Geomorphology, habitat, spatial location influences on fish and macroinvertebrate communities in modified channels of an agriculturally-dominated watershed in Ohio, USA. Ecological Engineering 68:32–46.

Deweber, J. T., T. Tsang, D. M. Krueger, J. B. Whittier, T. Wagner, D. M. Infante, and G. Whelan. 2014. Importance of understanding landscape biases in USGS gage locations: implications and solutions for managers. Fisheries 39(4):155–163.

Durbin, A. G., S. W. Nixon, and C. A. Oviatt. 1979. Effects of the spawning migration of the Alewife, *Alosa pseudoharengus*, on freshwater ecosystems. Ecology 60:8–17.

Elith, J., J. R. Leathwick, and T. Hastie. 2008. A working guide to boosted regression trees. Journal of Animal Ecology 77:802–813.

Elliott, M., A. K. Whitfield, I. C. Potter, S. J. M. Blaber, D. P. Cyrus, F. G. Nordlie, and T. D. Harrison. 2007. The guild approach to categorizing estuarine fish assemblages: a global review. Fish and Fisheries 8:241–268.

Esselman, P. C., D. M. Infante, L. Wang, D. Wu, A. R. Cooper, and W. W. Taylor. 2013. An index of cumulative disturbance to river fish habitats of the conterminous United States from landscape anthropogenic activities. Ecological Restoration 29:133–151.

Gende, S. M., R. T. Edwards, M. F. Willson, and M. S. Wipfli. 2002. Pacific salmon in aquatic and terrestrial ecosystems. Bioscience 52:917–928.

Grantham, T. E., J. H. Viers, and P. B. Moyle. 2014. Systematic screening of dams for environmental flow assessment and implementation. Bioscience 64:1006–1018.

Greene, C. M., K. Blackhart, J. Nohner, A. Candelmo, and D. M. Nelson. 2015. A national assessment of stressors to estuarine habitats in the contiguous USA. Estuaries and Coasts 38:782–799.

Gregory, S. V., K. L. Boyer, and A. M. Gurnell, editors. 2003. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland.

Gunn, J. M. 1986. Behavior and ecology of salmonid fishes exposed to episodic pH depressions. Environmental Biology of Fishes 17:241–252.

Gurnell, A.M. 2003. Wood storage and mobility. Pages 75–92 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland.

Hamilton, P. B., I. G. Cowx, M. F. Oleksiak, A. M. Griffiths, M. Grahn, J. R. Stevens, G. R. Cavalho, E. Nicol, and C. R. Tyler. 2015. Population-level consequences for wild fish exposed to sublethal concentrations of chemicals—a critical review. Fish and Fisheries 17:545–566.

Heck, M. P., L. D. Schultz, D. Hockman-Wert, E. C. Dinger, and J. B. Dunham. 2018. Monitoring stream temperatures—a guide for non-specialists. U.S. Geological Survey Techniques and Methods, book 3, chapter A25. Available: https://pubs.er.usgs.gov/publication/tm3A25. (April 2019).

The H. John Heinz III Center for Science, Economics and the Environment. 2002. The state of the nation's ecosystems: measuring the lands, waters, and living resources of the United States. Cambridge University Press, Cambridge, UK.

Helms, B. S., J. E. Schoonover, and J. W. Feminella. 2009. Assessing influences of hydrology, physicochemistry, and habitat on stream fish assemblages across a changing landscape. Journal of the American Water Resources Association 45:157–169.

Ikuta, K., Y. Suzuki, and S. Kitamura. 2003. Effects of low pH on the reproductive behavior of salmonids fishes. Fish Physiology and Biochemistry 28:407–410.

Imhof, J.G., J. Fitzgibbon, and W.K. Annable. 1996. A hierarchical evaluation system for characterizing watershed ecosystems for fish habitat. Canadian Journal of Fisheries and Aquatic Sciences 53(Supplement 1):312–326.

Keough, J. R., T. A. Thompson, G. R. Guntenspergen, and D. A. Wilcox. 1999. Hydrogeomorphic factors and ecosystem responses in coast wetlands of the Great Lakes. Wetlands 19:821–834.

Kerr, S. R. 1974. Theory of size distribution in ecological communities. Canadian Journal of Fisheries and Aquatic Sciences 31:1859–1862.

Kerr, S. R. 1977. Structure and transformation of fish production systems. Canadian Journal of Fisheries and Aquatic Sciences 34:1989–1993.

Kondolf, G. M., A. J. Boulton, S. O'Daniel, G. C. Poole, F. J. Rahel, E. H. Stanley, E. Wohl, A. Bang, J. Carlstrom, C. Cristoni, H. Huber, S. Koljonen, P. Louhi, and K. Nakamura. 2006. Process-based ecological river restoration: visualizing three-dimensional connectivity and dynamic vectors to recover lost linkages. Ecology and Society [online serial] 11(2):5.

Kondolf, G. M., Y. Gao, G. W. Annandale, G. L. Morris, E. Jiang, J. Zhang, Y. Cao, P. Carling, K. Fu, Q. Guo, R. Hotchkiss, C. Peteuil, T. Sumi, H. Wang, Z. Wei, B. Wu, C. Wu, and C. Yang. 2014. Sustainable sediment management in reservoirs and regulated rivers: experiences from five continents. Earth's Future 2:256–280.

Konrad, C. P., A. M. D. Brasher, and J. T. May. 2008. Assessing streamflow characteristics as limiting factors on benthic invertebrate assemblages in streams across the western United States. Freshwater Biology 53:1983–1998.

Kuffner, I. B., J. C. Brock, R. Grober-Dunsmore, V. E. Bonito, T. D. Hickey, and C. W. Wright. 2007. Relationships between reef fish communities and remotely sensed rugosity measurements in Biscayne National Park, Florida, USA. Environmental Biology of Fishes 78:71–82.

Lamouroux, N., N. L. Poff, and P. L. Angermeier. 2002. Intercontinental convergence of stream fish community traits along geomorphic and hydraulic gradients. Ecology 83:1792–1807.

Leopold, L. B, M. G. Wolman, and J. P. Miller. 1964. Fluvial processes in geomorphology. W.H. Freeman, San Francisco, California.

Lin, P., and L. Guo. 2016. Do invasive quagga mussels alter CO_2 dynamics in the Laurentian Great Lakes? Science Reports [online serial] 6:39078.

Locke, A., C. Stalnaker, S. Zellmer, K. Williams, H. Beecher, T. Richards, C. Robertson, A. Wald, A. Paul, and T. Annear. 2008. Integrated approaches to riverine resource management: case studies, science, law, people, and policy. Instream Flow Council, Cheyenne, Wyoming.

Logan, C. A. 2010. A review of ocean acidification and America's response. Bioscience 60:819–828.

Jones, N. E., and R. W. Mackereth. 2016. Resource subsidies from adfluvial fishes increase stream productivity. Freshwater Ecology 61:991–1005.

Jonssen, B., and N. Jonssen. 2003. Migratory Atlantic Salmon as vectors for the transfer of energy and nutrients between freshwater and marine environments. Freshwater Biology 48:21–17.

Magilligan, F. J., and K. H. Nislow. 2005. Changes in hydrologic regime by dams. Geomorphology 71:61–78.

Magnuson, J. J., L. B. Crowder, and P. A. Medvick. 1979. Temperature as an ecological resource. American Zoologist 19:331–343.

Mainstone, C. P., and J. Gulson. 1990. The effects of water quality on freshwater fish populations. Final Report to National Rivers Authority, London.

Martin, D. J. 2001. The influence of geomorphic factors and geographic region on large woody debris loading and fish habitat in Alaska coastal streams. North American Journal of Fisheries Management 21:429–440.

Martin, E. H., and C. D. Apse. 2011. Northeast aquatic connectivity: an assessment of dams on northeastern rivers. The Nature Conservancy, Eastern Freshwater Program, Brunswick, Maine.

Martino, E. J., and K. W. Able. 2003. Fish assemblages across the marine to low salinity transition zone of a temperate estuary. Estuarine, Coastal and Shelf Science 56:969–987.

Maser, C., and J. R. Sedell. 1996. From the forest to the sea: the ecology of wood in streams, rivers, estuaries and oceans. St. Lucie Press, Boca Raton, Florida.

Mackenzie, C. L., G. A. Orondroyd, S. F. Curling, R. J. Ball, N. M. Whiteley, and S. K. Malham. 2014. Ocean warming, more than acidification, reduces shell strength in a commercial shellfish species during food limitation. PLOS (Public Library of Science) ONE [online serial] 9(1):e86764.

McManamay, R. A., and E. A. Frimpong. 2015. Hydrologic filtering of fish life history strategies across the United States: implications for stream flow alteration. Ecological Applications 25:243–263.

Mills, E. L., J. H. Leach, J. T. Carlton, and C. L. Secor. 1994. Exotic species and the integrity of the Great Lakes. Bioscience 44:666–676.

Miller, P. M., D. M. Carlisle, D. M. Wolock, and M. Wieczorek. 2018. A database of natural monthly streamflow estimates from 1950 to 2015 for the conterminous United States. Journal of the American Water Resources Association 54:1258–1269.

Miller, R. R., J. D. Williams, and J. E. Williams. 1989. Extinctions of North American fishes during the past century. Fisheries 14(6):22–38.

Montgomery, D. R., B. D. Collins, J. M. Buffington, and T. B. Abbe. 2003. Geomorphic effects of wood in rivers. Pages 21–48 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland.

Mueller, D. K., and D. R. Helsel. 1996. Nutrients in the nation's waters—too much of a good thing? U.S. Geological Survey Circular 1136.

Mutz, M. 2003. Hydraulic effects of wood in streams and rivers. Pages 93–108 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland.

Naiman, R. J., R. E. Bilby, D. E. Schlinder, and J. M. Helfield. 2002. Pacific salmon, nutrients, and the dynamics of freshwater and riparian ecosystems. Ecosystems 5:399–417.

National Fish Habitat Board. 2010. Through a fish's eye: the status of fish habitats in the United States 2010. Association of Fish and Wildlife Agencies, Washington, D.C.

Nehlson, W., J. E. Williams, and J. A. Lichatowich. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho and Washington. Fisheries 16(2):4–21.

Payne, N. L., J. A. Smith, D. E. van der Meulen, M. D. Taylor, Y. Y. Watanabe, A. Takahashi, T. A. Marzullo, C. A. Gray, G. Cadiou, and I. M. Suthers. 2015. Temperature dependence of fish performance in the wild: link with species biogeography and physiological thermal tolerance. Functional Ecology 30:903–912.

Peterson, M. S., and M. R. Meador. 1994. Effects of salinity on freshwater fishes in the coastal plain drainages in the southeastern U.S. Reviews in Fisheries Science 2(2):95–121.

Piegay, H. 2003. Dynamics of wood in large rivers. Pages 109–134 *in* S. V. Gregory, K. L. Boyer, and A. M. Gurnell, editors. The ecology and management of wood in world rivers. American Fisheries Society, Symposium 37, Bethesda, Maryland.

Poff, N. L., and J. D. Allan. 1995. Functional organization of stream fish assemblages in relation to hydrological variability. Ecology 76:606–627.

Poff, N. L., J. D. Allan, M. B. Bain, J. R. Karr, K. L. Pretegaard, 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., B. P. Bledsoe, and C. O. Cuhaciyan. 2006. Hydrologic variation with land use across the contiguous United States: geomorphic and ecological consequences for stream ecosystems. Geomorphology 79:264–285.

Poff, N. L., and J. K. Zimmerman. 2010. Ecological responses to altered flow regimes: a literature review to inform the science and management of environmental flows. Freshwater Biology 55:194–205.

Qian, S. S., and T. F. Cuffney. 2011. To threshold or not to threshold? That's the question. Ecological Indicators 15:1–9.

Rhienheimer, D. E., and S. M. Yarnell. 2017. Tools for sediment management in rivers. Pages 237–263 *in* A. C. Horne, J. A. Webb, M. J. Stewardson, B. Richter, and M. Acreman, editors. Water for the environment from policy and science to implementation and management, Academic Press, Cambridge, Massachusetts.

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

Rounds, S. A., F. D. Wilde, and G. F. Ritz. 2013. Dissolved oxygen (version 3.0). U.S. Geological Survey Techniques of Water Resources Investigations, book 9, chapter A6, section 6.2. Available: http://water.usgs.gov/owq/FieldManual/Chapter6/6.2_v3.0.pdf. (April 2019)

Schlinder, D. E., M. D. Scheuerell, J. W. Moore, S. M. Gende, T. B. Francis, and W. J. Palen. 2003. Pacific salmon and the ecology of coastal ecosystems. Frontiers in Ecology and Environment 1(1):31–37.

Schlosser, I. J. 1991. Stream fish ecology: a landscape perspective. Bioscience 41:704–712.

Sheldon, R. W., A. Prakash, and W. H. Sutcliffe, Jr. 1972. The size distribution of particles in the ocean. Limnology and Oceanography 17:327–340.

Sommer, T. R., W. C. Harrell, A. Mueller Solger, B. Tom, and W. Kimmerer. 2004. Effects of flow variation on channel and floodplain biota and habitats of the Sacramento River, California, USA. Aquatic Conservation: Marine and Freshwater Ecosystems 14:247–261.

Speers-Roesch, B., and T. Norin. 2016. Ecological significance of thermal tolerance and performance in fishes: new insights from integrating field and laboratory approaches. Functional Ecology 30:842–844.

Strayer, D. L. 2009. Twenty years of zebra mussels: lessons from the mollusk that made headlines. Frontiers of Ecology and Environment 7(3):135–141.

Stein, B. L., L. S. Kutner, and J. S. Adams, editors. 2000. Precious heritage: the status of biodiversity in the United States, Oxford University Press, New York.

Talmage, S. C., and C. J. Gobler. 2010. Effects of past, present, and future ocean carbon dioxide concentrations on the growth and survival of larval shellfish. Proceedings of the National Academy of Science of the United States of America 107:17246–17251.

Trebilco, R., N. K. Dulvy, H. Stewart, and A. K. Salomon. 2015. The role of habitat complexity in shaping the size structure of a temperate reef fish community. Inter-Research Marine Ecology Progress Series 532:197–211.

Twining, C. W., D. C. West, and D. M. Post. 2013. Historical changes in nutrient inputs from humans and anadromous fish in New England's coastal wetlands. Limnology and Oceanography 58:1286–1300.

Twining, C. W., E. P. Palkovacs, M. A. Friedman, D. J. Hasselman, and D. W. Post. 2017. Nutrient loading by anadromous fishes: species-specific contributions and effects of diversity. Canadian Journal of Fisheries and Aquatic Sciences 74:609–619.

USEPA (U.S. Environmental Protection Agency). 2017. Approved Clean Water Act chemical test methods. Available: www.epa.gov/cwa-methods/approved-cwa-chemical-test-methods. (May 2018).

Van Steeter, M. M., and J. Pitlick. 1998. Geomorphology and endangered fish habitats of the upper Colorado River: 1. Historic changes in streamflow, sediment load and channel morphology. Water Resources Research 34:287–302.

Wagner, T., and S. R. Midway. 2014. Modeling spatially varying landscape change points in species occurrence thresholds. Ecosphere [online serial] 5(11):145.

Waldham, J. 2013. Running silver: restoring Atlantic rivers and their great fish migrations. Lyons Press, Guilford, Connecticut.

Walters, D. M., D. S. Leigh, M. C. Freeman, B. J. Freeman, and C. M. Pringle. 2003. Geomorphology and fish assemblages in a Piedmont river basin, USA. Freshwater Biology 48:1950–1970.

Weber, M. J., and M. L. Brown. 2009. Effects of common carp on aquatic ecosystems 80 years after "carp as a dominant": ecological insights for fisheries management. Reviews in Fisheries Science 17:524–537.

Whitfield, A. K., and M. Elliott. 2005. Fishes as indicators or environmental and ecological changes within estuaries: a review of progress and some suggestions for the future. Journal of Fish Biology 61(sA):229–250.

Winemiller, K. O., and K. A. Rose. 1992. Patterns of life history diversification in North American fishes: implications for population regulation. Canadian Journal of Fisheries and Aquatic Resources 49:2196–2218.

Wipfli, M. S., J. P. Hudson, J.P. Caoette, and D. T. Charloner. 2003. Marine subsidies in freshwater ecosystems: salmon carcasses increase growth rates of stream-resident salmonids. Transactions of the American Fisheries Society 132:371–381.

Wright, D. J., and W. D. Heyman. 2008. Introduction to the special issue: marine and coastal GIS for geomorphology, habitat mapping and marine reserves. Marine Geodesy 31:1–8.