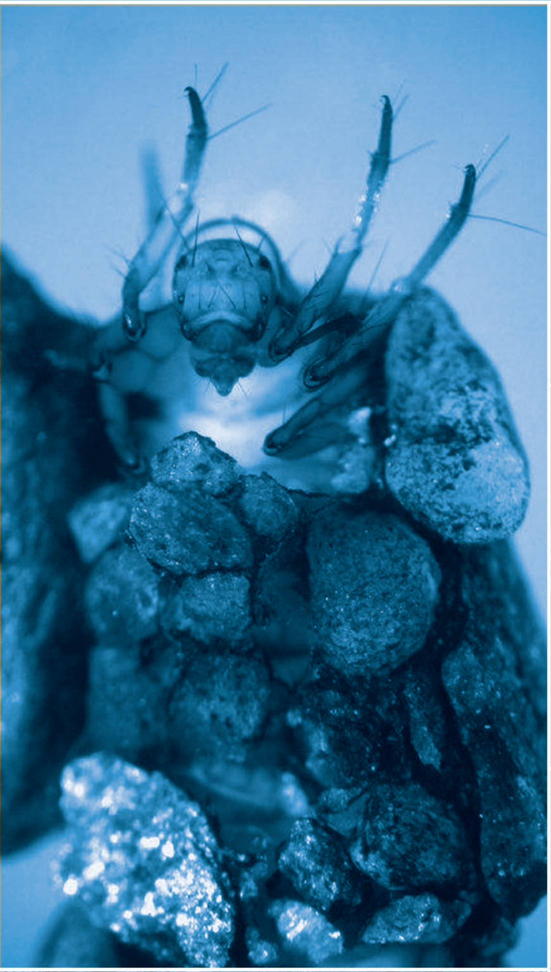
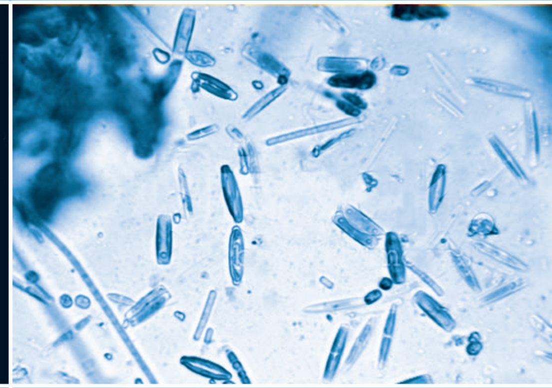
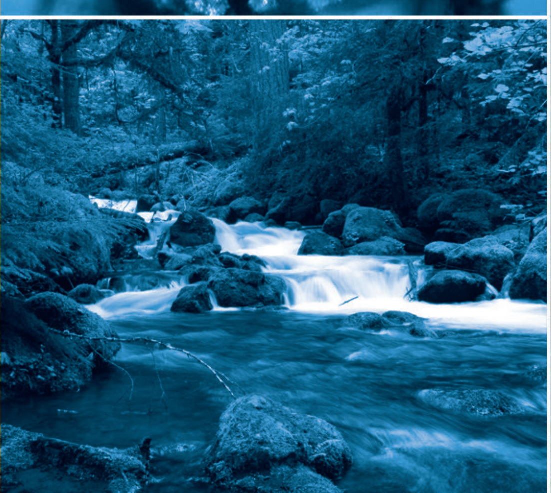
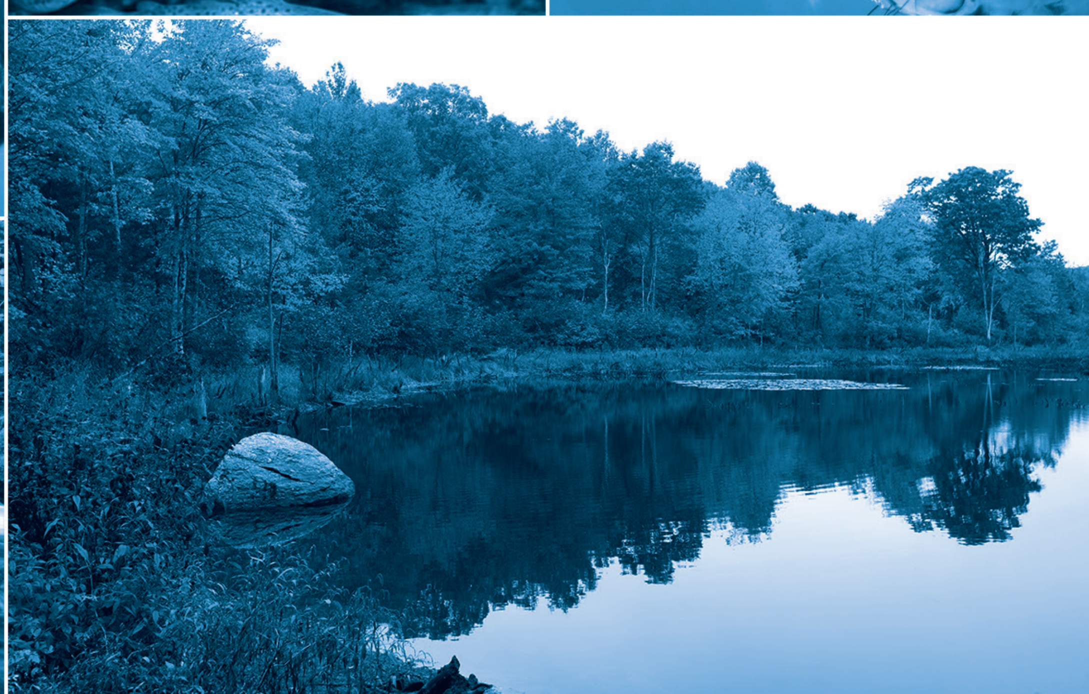
EPA 842-R-16-001

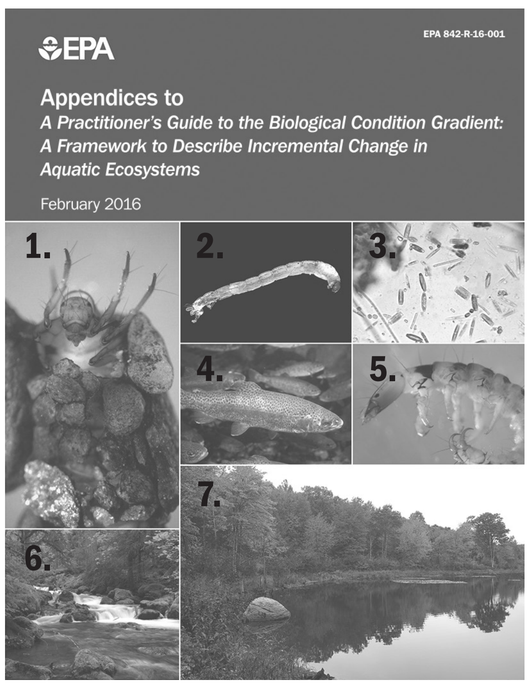
Appendices to
A Practitioner's Guide to the Biological Condition Gradient: A Framework to Describe Incremental Change in Aquatic Ecosystems

February 2016







Front cover sources:

1. Vermont Department of Environmental Conservation
2. Christophe Quintin, flickr
3. ©istockphoto.com
4. USEPA
5. Vermont Department of Environmental Conservation
6. ©istockphoto.com
7. USDA NRCS

Appendices to

A Practitioner's Guide to the Biological Condition Gradient: A Framework to Describe Incremental Change in Aquatic Ecosystems

Note: For a complete list of the acronyms in this document, see the "Abbreviation and Acronym" list with Chapters 1–6 of this document.

Contents

Appendix A: Generalized Stress Axis Support Materials

Appendix A includes a series of tables that provide a conceptual Generalized Stress Axis (GSA) construct, as well as examples of pressure and stressor indicators for key environmental processes and elements; the stressors that are produced when these processes and elements are altered by human disturbance; and possible mechanisms of stressor action on the aquatic biota and habitats. As described in Chapter 5, indicators of pressure and stressors can be applied to quantify the GSA. Scenarios for two climatic regions of the U.S., humid-temperate and arid, are included. The tables are listed below.

A-1. Conceptual scenarios for stress related changes in the major environmental factors that influence biological condition. The scenarios describe potential changes in the factors in two regions of the United States: a humid-temperate and an arid region.

A-2. Potential impacts of climate change at low/medium and high stress levels.

A-3. Potential indicators for stressors associated with altered flow, material transport, channel structure, and riparian/watershed structure.

A-4. Examples of some fundamental environmental processes and materials (e.g., flow, material transport, channel structure, riparian and watershed structure, biological interactions) that can be altered by human-induced disturbances (i.e.., pressure indicators) and create stressors.

Background

This appendix is not a comprehensive compendium of indicators, but it has drawn upon existing sources of information such as the EPA Recovery Potential Screening (RPS) method (Norton et al. 2009) and online resource tools.[1] The RPS website provides step-by-step instructions for evaluating the recoverability of degraded watersheds based on user-selected and weighted ecological, stressor, and social indicators. The site contains reference materials on recovery potential indicators, including their definitions, relevance to restorability, data sources, measurement methods, and relevant points from the technical literature. This online resource also includes a master list of indicators, including indicators for pressures, alteration in ecosystem processes and elements, and stressors, all of which can be applied to development of the Biological Condition Gradient (BCG) x-axis, the GSA. Associated with the RPS site is the Watershed Index Online (WSIO), a national watershed indicator library and online comparative watershed analysis tool that houses the data for hundreds of ecological, stressor, and social indicators compiled nationally on the HUC12 watershed scale.[2] At the WSIO site, state-specific RPS tools for the lower 48 states containing over 200 WSIO indicators at the 12-digit hydrologic unit code (HUC12) scale can also be downloaded[3] (HUC12 watersheds average 35 square miles in area).

Whereas these online resources have derived many of their metrics from commonly used geospatial data sources such as land cover, transportation, and impaired waters data sets, they have gone into considerably more detail and variation than basic "% in the watershed" statistics in order to provide

[1] More information is available at: http://www.epa.gov/rps. Accessed February 2016.

[2] Watershed Index Online is available at http://www.epa.gov/watershed-index-online. Accessed February 2016.

[3] See statewide tools at http://www.epa.gov/watershed-index-online/watershed-index-online-wsio-download-statewide-tools. Accessed February 2016.

greater choices for the most locally relevant or issue-specific metrics to use. For example, the national data's variations on impervious cover statistics include riparian zone totals, proportion of different impervious cover densities in each watershed, and percent by proximity of impervious cover to surface waters versus merely percent in the watershed overall. Summary metrics by watershed also address the extent of 303(d)-listed (impaired) waters per watershed by major pollutant, occurrence of existing total maximum daily loads, and other management-relevant metrics. These social indicators also could potentially provide a third axis, also useful to implementing the BCG in practice, that provides insights on social factors that may make improving biological condition more (or less) easy to accomplish.

Additionally, the U.S. Environmental Protection Agency's (EPA's) Causal Analysis/Diagnosis Decision Information System, or CADDIS, is a website developed to help scientists and engineers in the regions, states, and tribes conduct causal assessments in aquatic systems.[4] It is organized into five volumes:

- *Volume 1: Stressor Identification* provides a step-by-step guide for identifying probable causes of impairment in a particular system, based on EPA's Stressor Identification process. Those interested in conducting a complete causal assessment, learning about different types of evidence, or reviewing a history of causal assessment theory, should start with this volume.
- *Volume 2: Sources, Stressors & Responses* provides background information on many common sources, stressors, and biotic responses in stream ecosystems. Those interested in viewing source- and stressor-specific summary information (e.g., for urbanization, physical habitat, nutrients, metals, pH, and other stressors), should start with this volume.
- *Volume 3: Examples & Applications* provides examples illustrating different steps of causal assessments. Those interested in reading completed causal assessment case studies, seeing how Stressor Identification worksheets are completed, or examining example applications of data analysis techniques, should start with this volume.
- *Volume 4: Data Analysis* provides guidance on the use of statistical analysis to support causal assessments. Those interested in learning how to use data in a causal assessment, should start with this volume.
- *Volume 5: Causal Databases* provides access to literature databases and associated tools for use in causal assessments. Those interested in applying literature-based evidence to a causal assessment, should start with this volume.

The conceptual diagrams in the EPA Causal Analysis CADDIS Volume 2 can provide a starting point description of how human activities can lead to stressors and biological effects.

[4] For more information, visit: http://www3.epa.gov/caddis/ssr_home.html. Accessed February 2016.

Appendix A-1. Conceptual scenarios for stress related changes in the major environmental factors that influence biological condition (see Chapter 5, Figure 19). The scenarios describe potential changes in the factors in two regions in the United States: a humid-temperate and an arid region. The stressor levels are qualitative and used only to describe relative differences in magnitude. Both local and watershed scale factors are important for determining the condition of streams. Note that under the BCG conceptual model, alterations to the factors of flow regime, water quality, energy source, and physical habitat structure represent increased stressors or categories of stressors (attributes of the GSA (BCG x-axis)), whereas alterations to biotic interactions are included in the BCG y-axis attributes (e.g., attribute VI (non-native or intentionally introduced species)).

			BCG X-Axis			BCG Y-Axis
	Stressor Level	**Flow Regime**	**Water Quality**	**Energy Source**	**Physical Habitat Structure**	**Biotic Interactions**
HUMID-TEMPERATE SCENARIO	**No/ Low**	Within the naturally occurring range, includes floods & low flows at natural rates, intervals, and extent; High connectivity with ground water maintained	Within the naturally occurring range, or with only minimal increase in nutrients & sediments, including flood-related turbidity & summer warming (linked to season and antecedent moisture conditions); no point sources of nutrients or toxic substances; usually cool or cold & dissolved oxygen (DO) saturated Within the naturally occurring range, typically rare, and no materials in amounts toxic to aquatic biota	Within the naturally occurring range expected for a stream with a particular channel width; smaller systems typically dominated by riparian woody vegetation, unless naturally autochthonous	Within the naturally occurring range for a stream of a particular size & slope; typically, large woody debris (LWD) abundant; coarse substrate; overhanging vegetation and undercut banks are present	Within the naturally occurring range expected, e.g., anadromy and potamodromy;[5] beavers common; aquatic invasive species (AIS) non-detrimental; anomalies (e.g., deformities, erosion, lesions, and tumors (DELT)) due to disease or parasitism absent or infrequent; no or insignificant historical range changes

[5] The terms "anadromy" and "potamodromy" relate to long distance river migrants. See glossary.

	Stressor Level	Flow Regime	BCG X-Axis Water Quality	Energy Source	Physical Habitat Structure	BCG Y-Axis Biotic Interactions
HUMID-TEMPERATE SCENARIO	**Medium**	Flashy; greater maxima and minima; increased drought frequency; some water withdrawals; low to moderate amount of wetland drainage; damming may reduce annual floods and droughts; some groundwater extraction close to the floodplain	Enriched with nutrients and ions; turbidity may increase, moderate diel warming; small DO sags may occur but these rarely violate criteria; point sources of nutrients or toxic substances minor or if they exist are treated; fish kills rare Suspended and dissolved materials in amounts rarely toxic to aquatic biota, but mercury or persistent organic contaminants may become of chronic concern to top piscivores due to bioaccumulation; sediment contamination may be detectable but not causing effects in benthic biota	Autochthonous production higher than expected in lower order streams; filamentous algae may be present	Reduced amounts of LWD in channel; fines slightly to moderately more abundant than expected from stream power; pool substrate moderately embedded; reduced extent of undercut banks, overhanging vegetation, and habitat complexity; some loss of pool volume and pool/riffle proportions may be altered	Altered fish age structure from fishing and stocking may change predation and competition with a significant effect on native populations; expected beaver populations diminished; higher than expected occurrence of DELT anomalies due to parasitism or disease; sensitive AIS may dominate, tolerant AIS may be present; minor to moderate historical range alterations; cosmopolitan species may extend distributions further upstream

	Stressor Level	Flow Regime	BCG X-Axis Water Quality	Energy Source	Physical Habitat Structure	BCG Y-Axis Biotic Interactions
HUMID-TEMPERATE SCENARIO	**High**	Flashy; highly altered low regimes lead to greater risk of drought/flood; mostly or entirely human controlled runoff in urban and agricultural areas; water withdrawals & impoundments, if present, fundamentally alter the nature of the ecosystem	Highly enriched, turbid, warm; large diel DO & temperature changes; chemical and point sources of nutrients or toxic substances inadequately treated or overwhelmed by untreated diffuse toxic pollution. Dams when present produce altered thermal regime and nutrient dynamics Dissolved, suspended, or sediment-associated materials may reach concentrations that are chronically or acutely toxic to biota or can affect growth & reproduction; high to extreme sediment contamination; anomalies when associated with toxic impacts are abundant & serious; fish consumption advisories serious. Pharmaceuticals and/or personal care products present in effluent at concentrations high enough to be routinely detected in water and/or tissue	Secondary production sustained mostly by autochthonous or imported fine particulate organic matter or dissolved organic matter; water may be too turbid for benthic filamentous algae to develop. Strong diel periodicity in dissolved oxygen concentrations, including night-time anoxia	Simplified or manmade, straightened and/or leveed; wood, undercut banks, & overhanging vegetation absent or non-functioning; rubble & trash common, substrates highly armored or embedded; sedimented with sand or silt; aquatic macrophytes missing or extremely rare; riparian habitats reduced or destroyed; dam impoundments often present	Dominated by transient fishes or tolerant AIS; historically common keystone species extirpated and once-common species now threatened, endangered, or extirpated from large portions of their historical ranges due to changes in predation and competition relationships and/or alteration of physical habitat structure by stocked fish; beavers transient or absent

	Stressor Level	Flow Regime	BCG X-Axis Water Quality	Energy Source	Physical Habitat Structure	BCG Y-Axis Biotic Interactions
ARID SCENARIO	**No/ Low**	Within the naturally occurring range, or only slightly altered, includes floods & low flows at natural rates, intervals, and extent; floods flashy; annual scouring flows; high connectivity with ground water	Within the naturally occurring range, with only minimal increase in nutrients & sediments, includes flood turbidity & summer warming; depending on soils, may be naturally saline or alkaline; relative ionic concentration affected by evaporation (Griffith 2014); enriched where beaver present; ash from 5–20 year fire cycles; no point sources Within the naturally occurring range, typically rare, but may be natural sources of arsenic & selenium; No toxics in amounts toxic to aquatic biota	Within the naturally occurring range, varies with channel width, typically dominated by riparian woody vegetation in small unconstrained channels; heterotrophic & autochthonous in wider systems	Within the naturally occurring range, varies with geology, substrate, flow, size, slope, soil, latitude, elevation, & orography; relatively stable riparian vegetation, LWD in flats	Within the naturally occurring range expected e.g., potamodromy; AIS absent or non-detrimental. Intermittent and ephemeral invertebrate species traits are linked to flow permanence, drought and flood cycles (Bonada et al. 2007)
	Medium	Altered, increasingly flashy; increased drought frequency; some water withdrawals and wetland drainage; flow alterations mitigated to some extent by environmental flow releases; some groundwater extraction close to the floodplain; dams may be present	Enriched, warmer & saltier, turbid at low flows, small DO sags; point sources if present with treatment. No fish kills No acute toxicity is observed, but chronic toxicity is possible due to bioaccumulation. Fish consumption advisories likely for sensitive populations. Low concentrations of personal care products and pharmaceuticals observed in wastewater receiving waters	Mostly allochthonous, but increasingly autochthonous in narrow streams; wide streams heterotrophic or autochthonous with increasing amounts of filamentous algae	Minor amounts of incision, widening, or shallowing; reduced LWD in channel; fines greater than expected from stream power; bed coarsening from upstream dams; pool substrate increasingly embedded; reduced aquatic macrophytes, undercut banks, & overhanging vegetation	Altered fish age structure from fishing and stocking may change predation and competition with a significant effect on native populations; AIS more common and beginning to reduce competitors & prey; potamodromy reduced (Death et al. 2009)

	Stressor Level	BCG X-Axis Flow Regime	Water Quality	Energy Source	Physical Habitat Structure	BCG Y-Axis Biotic Interactions
ARID SCENARIO	**High**	Human controlled; large inter-basin transfers; ground water overdrawn; effluent dominated streams below cities. Highly altered drought/flood regime; droughts yield more dry channels; withdrawals & dams severely alter nature of the ecosystem	Highly enriched, turbid, warm; large diel DO changes; effluent dominated; point sources may be inadequately treated or overwhelmed by untreated diffuse toxic pollution; dams produce altered thermal regime Toxics may be present in chronic or acutely toxic amounts; bioengineered chemicals can affect growth & reproduction; high to extreme sediment contamination; fish consumption advisories serious. Pharmaceuticals and/or personal care products present in effluent, water, and/or tissue	Mostly autochthonous or imported fine particulate or dissolved organic matter; filamentous algae common if turbidity allows it	Largely manmade, straightened, and/or leveed; little or no LWD, undercut banks, or overhanging vegetation; highly sedimented with sand or silt; sand bottom streams/rivers dominated by silt; construction rubble & trash common; aquatic macrophytes missing or extremely rare; riparian habitats reduced or destroyed	Assemblages dominated by tolerant species including tolerant AIS; once-common species now threatened, endangered, or extirpated from large portions of their historical ranges due to changes in predation, competition, and/or behaviors by stocked fish; potamodromy rare and erratic

Appendix A-2. Potential impacts of climate change at low/medium and high stress levels.

		BCG X-Axis				BCG Y-Axis
	Stressor Level	**Flow Regime**	**Water Quality**	**Energy Source**	**Physical Habitat Structure**	**Biotic Interactions**
HUMID SCENARIO	**Low to Medium**	Depending on the region, climate change likely to increase drought frequency and duration (leading to greater intermittent flows); greater incidence of intermittency in headwaters; increased frequency and magnitude of floods due higher incidence of intense storms; overall higher variability in streamflow	Degraded water quality due to climate change impacts may be observed as a result of increased loading due to flood frequency and duration. Low flows may result in higher water temperatures and episodic low oxygen levels, increasing rates of some geochemical processes. Lower nutrient loading and export rates expected during low flow Climate change impacts may result from watershed loading during floods. Low flows may result in episodic low oxygen levels, increasing rates of some geochemical processes	Climate change impacts (i.e., extended drought cycles) may negatively impact riparian systems leading to reduced shading, disruptions in input of CWD and particulate organic matter (POM), greater autochthonous production	Climate change is expected to reduce the extent of coldwater habitat, reduce stream shading as macrophytes, overhanging vegetation, and riparian vegetation are reduced. Increases in flood frequency and intensity will alter channel structure except in confined channels or bedrock dominated systems. Increased potential for increased debris flows in mountain environments with increased fire frequency (Braune et al. 2008; Cannon et al. 2010)	Climate change will result in range contractions of coldwater species and expansion of cool and warm-water species. Biodiversity shifts are likely due to shifts in energy sources, availability of nutrients, and changes in food webs at low flow. Invertebrates may experience increased voltinism. Increased incidence of drought may accelerate incidence of intermittency leading to reduction in secondary production (Freeman et al. 2007). The increased length of the growing season will increase yearly algal and cyanobacterial biomass and production

	Stressor Level	Flow Regime	BCG X-Axis Water Quality	Energy Source	Physical Habitat Structure	BCG Y-Axis Biotic Interactions
HUMID SCENARIO	**High**	Climate change expected to lead to increased frequency and duration of drought in some regions, leading to increased incidence of intermittency; increased incidence of extreme storms	Climate change impacts expected as a result of increased drought and flood frequency and duration likely will exacerbate anthropogenic disturbances. Low flows may result in low oxygen levels and increased water temperature, increasing rates of some geochemical processes In some regions, climate change impacts expected as a result of increased drought frequency and severity. Likely acceleration of mercury methylation and increased bioavailability of toxic metals. Mobilization of toxic chemicals likely during floods	Climate change impacts (i.e., extended drought cycles) may negatively impact riparian systems reducing or eliminating sources of CWD and POM	Effects of climate change due to intense storms and more frequent and intense drought may exacerbate effects of anthropogenic disturbances	Climate change impacts are less evident due to simplified assemblages and reduced richness (Durance and Ormerod 2007, 2009). Increased incidence of intermittency in headwaters may further simplify assemblage structure and lead to further reductions in secondary production (Freeman et al. 2007). Temperature sensitive taxa also are vulnerable to organic pollution (Hamilton et al. 2010)
ARID SCENARIO	**Low to Medium**	Climate change likely to increase drought frequency, magnitude of floods. Winter precipitation is predicted to increasingly fall as rain, increasing likelihood of rain-on-snow events, leading to less infiltration, and lower snowpacks. Earlier snow melt and lower and longer summer baseflows will shift seasonal flow patterns. Fire frequency and intensity may influence flow regime, with post-fire peaks observed (Benda et al. 2003)	Climate change impacts may be observed as a result of increased drought and flood frequency and duration. Low flows may result in low oxygen levels, increasing rates of some geochemical processes. Post-fire floods may contribute additional sediment to channel; impacts of fire on riparian vegetation may produce elevated temperatures (Mahlum et al. 2011) Climate change impacts may result from watershed loading during floods. Low flows may result in low oxygen levels, increasing rates of some geochemical processes	Climate change impacts (i.e., extended drought cycles) may negatively impact riparian systems, disrupting input and retention of LWD and POM. Increased frequency and duration of channel drying will reduce litter breakdown rates (Corti et al. 2011; Datry et al. 2011)	Climate change is expected to reduce the extent of coldwater habitat, reduce stream shading as macrophytes, overhanging vegetation, and riparian vegetation are reduced. Post-fire flood sediment contributions may alter riverine habitats	Climate change may result in range expansion and contractions, with expected losses in coldwater habitat and species and expansion of cool and warm-water species (Comte et al. 2013). Desiccation-sensitive taxa drop out with increased intermittence (Arscott et al. 2010). Invertebrate density, richness, and condition metrics decline with reduced taxon richness lower in response to reduced discharge

	Stressor Level	Flow Regime	BCG X-Axis Water Quality	Energy Source	Physical Habitat Structure	BCG Y-Axis Biotic Interactions
ARID SCENARIO	**High**	Climate change expected to lead to increased frequency and duration of drought, and increased incidence of extreme storms	Climate change impacts expected as a result of increased drought and flood frequency and duration likely will exacerbate anthropogenic disturbances. Low flows may result in low oxygen levels and increased water temperature, increasing rates of some geochemical processes In some regions, climate change impacts expected as a result of increased drought frequency and severity. Likely acceleration of mercury methylation and increased bioavailability	Climate change impacts (i.e., extended drought cycles) may negatively impact riparian systems reducing or eliminating sources of CWD and POM	Effects of climate change due to intense storms and more frequent and intense drought likely to exacerbate effects of anthropogenic disturbances	Climate change exacerbates anthropogenic disturbances. Reduced discharge has less effect on invertebrate community structure and function (Death et al. 2009)

Appendix A-3. Potential indicators for stressors associated with altered flow, material transport, channel structure, and riparian/watershed structure. The indicators of stress will differ depending on which environmental factor is being considered (e.g., flow regime, water quality, energy, physical habitat structure, biotic interactions). Ideally, indicator scores are benchmarked to undisturbed or minimally disturbed conditions for a given region or channel type, and stressors represent a departure from the natural range of variation. Biotic responses can be related to these stressor indicators (shaded column).

Major factor	Potential Stressor Indicators Listed for Different Environmental Process				Biological Responses
	Flow Alteration	**Material Transport**	**Channel Structure**	**Riparian / Watershed Structure**	**Biotic Structure/ Function**
Flow Regime	• Frequency of low and high flow events • Annual flow variability • Flashiness (change in flood peak and duration) • Base flow rate • Stream power • Timing of peak flow	• Stream power • Flashiness (changes in flood peak and duration) • Particle size distribution • LWD dam density • LWD transport • Organic matter (OM) transport • Erosion rates • Streambed stability • Turbidity	• Discharge • Hydraulic storage in catchment • Hydraulic storage in flood plain • Change in transient storage capacity • Pool-riffle structure • Erosion rates • Evidence of active erosion • Streambed stability • Nutrient spiraling rate	• Peak flow • Flood-mediated sediment, OM, and nutrient deposition in floodplain	• Macrophyte density • Dominance of flow-dependent taxa • Periphyton biomass • Fish species traits associated with flow regime (e.g., rheophils and nonguarding lithophils & lithopelagophils are replaced by residents, generalists, and polyphils) • Invertebrate traits associated with flow regime (= low crawling rate, short adult life span, erosional rheophily, med size at maturity, cool/cold thermal preference
Water Quality	• Temperature regime • DO regime • Flooding linked to sediment & nutrient loads • Flooding linked to contaminant loading	• Sediment loading rates • Sediment bound metals, contaminants • Nutrient concentrations	• DO regime • Temperature regime • No direct indicators	• Sediment/nutrient loading rates • Water temperature • Contaminant loading from upland or riparian • Contaminant bound sediment from roads, parking lots	• Increased biochemical oxygen demand • Increased nitrification/denitrification • Algal bloom • Rates of critical biogeochemical processes (e.g., mercury methylation)

Major factor	Potential Stressor Indicators Listed for Different Environmental Process				Biological Responses
	Flow Alteration	Material Transport	Channel Structure	Riparian / Watershed Structure	Biotic Structure/ Function
Energy Source	• OM decomposition rates • OM input and retention	• Particulate organic matter/Dissolved organic matter concentrations • Nutrient concentrations compared to natural	• LWD retention • OM retention • Solute retention	• OM quantity and composition • Primary production rates • Metabolism	• Food web alteration
Physical Habitat Structure	• % fines • Armoured substrate • Pool/riffle sequence • OM and LWD input and retention • Flood height • Width:depth	• LWD density • % fines • % embeddedness • Bed stability	• Pool-riffle sequence • Large woody debris volume • LWD input and retention • Bank erosion rates	• Floodplain connectivity • LWD storage • OM storage • Riparian fragmentation	• Macrophytes/algal mats • AIS dominance of species that change structure (e.g., macrophytes, carp, zebra/quagga mussels) • Amount of overhead cover for fish • Noxious weeds
Biotic Condition & Interactions (Responses)	• % rheophilic fish taxa • Biotic condition scores (invertebrates) • Invertebrate diversity and functional feeding	• Dissolved and particulate nutrient uptake • Primary/secondary production rates • Invertebrates in drift	• Assemblage structure • Primary and secondary production • Rheophilic species • Migrator fish • Spawning habitat	• Benthic metabolism rates • Food web structure compared to natural • Riparian buffer fragmentation	• AIS (e.g., native game fish decline, hatchery fish increase) • Native fish/benthos and riparian vegetation and birds relative to aliens • Sensitive specialists compared to tolerant generalists (birds, fish, invertebrates, plants) • Fish disease and anomaly rate • Benthic metabolism, decomposition rates

Appendix A-4. Examples of some fundamental environmental processes and materials (e.g., flow, material transport, channel structure, riparian and watershed structure, biological interactions) that can be altered by human-induced disturbances (e.g., pressure indicators) and create stressors (Figures 20, 21, and 26). Possible mechanisms for stressor production resulting from the altered processes are listed, as well as examples of management actions to reduce the stress.

	Pressure Indicators	Mechanism for Stressor Production	Management Actions to Reduce Stress
FLOW ALTERATION	% impervious area	Acceleration of water flow; reduction of infiltration and groundwater recharge; hot pavement and reduced retention times increases stream temperatures	Reduce impervious surface; install pervious pavements
	Road density	Increases impervious surface; increases number of roads crossings streams and culverts	Implement low-impact development strategies
	% urban area	Reduces groundwater infiltration; increases peak flows; decreases peak flow duration	Plant trees in headwaters; urban forests and parks; restore urban streams
	Population-density	Increases impervious surface	Restore riparian and floodplain vegetation
	Storm sewer miles	Increased potential for stormwater overflows	Restore natural flow regime; move storm sewers out of stream beds
	# diversions per catchment; quantity of water diverted	Decreased base flow; increased intermittency in headwater streams	Use appropriate culvert type and size
	# of dams; cumulative volume of water in reservoirs	Impacts natural flow regime by depressing flood height & duration; maintains base flow downstream but may prevent fish migration upstream	Restore connectivity to floodplain; fish ladders
	Proportion of river length channelized	Reduced base flow	Install flood retention structures
ALTERATION OF MATERIALS TRANSPORTED	# dams, # diversions	Flow alteration; increased sediment retention upstream; sediment pulses when gates opened	Restore natural flow regime
	Point source discharge constituent levels; # point source discharges per catchment; density of point source discharges	Increased discharge of pollutants, including excess nutrients, toxic materials, and particulates from point sources; enhanced flow increases erosion	Water quality management actions (e.g., permits); sediment retention basin
	Length (in km) of intact riparian buffers	Erosion of surface solutes, sediments, and warmer water; reduced riparian canopy increases algal biomass, increases nutrient uptake, enhances benthic production	Riparian restoration
	% impervious surfaces, population density	Eroded material adds carbon and nutrients that increase biological activity; increased contaminants	Low-impact development

	Pressure Indicators	Mechanism for Stressor Production	Management Actions to Reduce Stress
ALTERATION OF MATERIALS TRANSPORTED	# road crossings per catchment; road density	Increased road density and numbers of road crossings may result in episodic, high volume flow events that erode stream banks and produce increased pollutant (e.g., metals, oils) and sediment loading	Road maintenance; retention ponds; riparian habitat restoration; road placement
	% row crops	Nutrient and pesticide applications and irrigation contributes to contaminated runoff and increased sedimentation	Buffer strips
	Atmospheric deposition	Increased nitrogen, carbon, and sulfur loading	Substitute cleaner fuels; implement "clean" combustion technologies; implement non combustion energy production and use (e.g., solar, wind)
	CAFO size and density	Manure results in increased nutrient, pathogen, pesticide, antibiotic, and sediment loading	Composting; appropriate manure application; retention structures; reduced antibiotic and pesticide management practices
	Area (in km^2) of tile drains	Enhance discharge especially during storms	Two-stage ditch
	Area of catchment logged; length of logging roads	Enhanced surface runoff from bare soils	Buffer strips; use temporary road crossings in winter; select cutting replacing clear-cutting
	# mines per area; area of valley fill	Enhanced windborne fines and surface-derived sediments, salts and metals, acid drainage Elimination of natural steams and forest habitats	Sediment retention basins Alter mining practices (e.g., mountain top removal and valley fill practices to traditional underground mining practices)
	# acres of irrigated cropland with no BMPs implemented	Increase in surface runoff, sedimentation, eutrophication, with higher levels of pesticides, herbicides, solutes leached from soils (e.g., salts, selenium)	Drip irrigation based on soil moisture levels
	# quarries per catchment	Sediment loading	Buffer strip
	Acres of drained wetlands	Storm storage	Retention basin
	Length of armored channel	Increased stream power; reduced erosion potential	Restore natural channel

	Pressure Indicators	Mechanism for Stressor Production	Management Actions to Reduce Stress
CHANGE IN CHANNEL STRUCTURE	Length (in km) of channelized stream	Flow alteration; habitat loss	Restore natural channel shape and flow regime
	Stream length stabilized by riprap/concrete	Hardening of shoreline alters flow and erosion processes and simplifies habitat	Restore natural streambanks
	# dams; volume in reservoirs	Solute, sediment transport interrupted	Remove dams where appropriate
	# diversions; volume of water diverted	Flow regime altered; base flow impacted	Restore natural flow regime
	Culvert density	Flow disruption	Install appropriate culvert type/size
	Density of road crossings	Riparian alteration; sedimentation; flow alteration	Mitigate dust, employ proper drainage tactics
	Presence of valley fills; extent of valley fill	Direct engineering activities; elimination of channel; water quality impacts	Stop activity; employ retention basins
	Length of levees per catchment	Connection to floodplain disrupted	Reconnect floodplain
	Length of intact riparian zone	Loss of shading; loss of OM	Restore natural riparian vegetation
	Evidence of snagging of LWD	Habitat loss	Restore natural wood structures
	Evidence of connectivity disruption or artificial connections established	Habitat loss; vector for non-native invasive species (NIS) established	Restore connectivity or employ structures to remove connectivity
CHANGE TO RIPARIAN AND WATERSHED STRUCTURE	Fragmentation of riparian zone	Loss of natural vegetation cover leading to habitat loss, increased sediment/nutrient input	Restore vegetation cover
	Riparian width	Reducing, disturbing, or completely removing riparian cover increases sedimentation and reduces habitat and other effects	Restore natural vegetation type and extent
	% shading	Increased solar insolation leading to greater algal biomass results in greater daytime photosynthesis and night-time respiration	Plant trees in riparian zone
	Levees	Floodplain disconnect from river and prevent replenishment of flow, nutrients, and sediments	Restore natural channel form; engineer flow, nutrients and sediment delivery to mimic natural regime
	Tile number-drains/ditches	Altered flow regime; increased nutrient input	Two-stage ditch
	Length (in km) of streamside roads	Sedimentation	Install buffer strip; pave road near stream
	Surface area of off-stream ponds or wetlands	Loss of connectivity; loss of flood storage	Restore connectivity
	Area of valley bottom grazing	Manure from cattle adds carbon and nutrients that increase in-stream biological activity; cattle moving through streambed results in habitat modification	Fence pasture; bridges for cattle to cross streams; manage manure
	Area of aggregate mining	Increased fine sediment loading	Install sediment basin

	Pressure Indicators	Mechanism for Stressor Production	Management Actions to Reduce Stress
CHANGE IN BIOLOGICAL CONDITION AND ACTIVITY	# NIS	Stocking programs; accidental introduction from aquaculture facility	Education; enhanced facility inspections; programs to extirpate NIS
	$ of baitfish sales	Habitat modification or negative biotic interactions by invasive plants and fish	Education; boat inspections
	# fishing licenses issued	Over harvest	Harvest limits; education; reduce number of licenses
	# of dams and reservoirs	Prevent fish migration upstream	Restore connectivity fish ladders

Appendix A References

Arscott, D.B., S.T. Larned, M. Scarsbrook, and P. Lambert. 2010. Aquatic invertebrate community structure along an intermittence gradient: Selwyn River, New Zealand. *Journal of the North American Benthological Society* 29(2):530–545.

Benda, L., D. Miller, P. Bigelow, and K. Andras. 2003. Effects of post-wildfire erosion on channel environments, Boise River, Idaho. *Forest Ecology and Management* 178:105–119.

Bonada, N., M. Rieradevall, and N. Prat. 2007. Macroinvertebrate community structure and biological traits related to flow permanence in a Mediterranean river network. *Hydrobiologia* 589:91–106.

Braune, E., O. Richter, D. Sondgerath, and F. Suhling. 2008. Voltinism flexibility of a riverine dragonfly along thermal gradients. *Global Change Biology* 14:470–482.

Cannon, S.H., J.E. Gartner, M.G. Rupert, J.A. Michael, A.H. Rea, and C. Parrett. 2010. Predicting the probability and volume of post-wildfire debris flows in the intermountain western United States. *Geological Society of America Bulletin* 122(1/2):127–144. doi: 10.1130/B26459.1.

Comte, L., L. Buisson, M. Daufresne, and G. Grenouillet. 2013. Climate-induced changes in the distribution of freshwater fish: Observed and predicted trends. *Freshwater Biology* 58:625–639.

Corti, R., T. Datry, L. Drummond, and S.T. Larned. 2011. Natural variation in immersion and emersion affects breakdown and invertebrate colonization of leaf litter in a temporary river. *Aquatic Sciences* 73(4):537–550. doi: 10.1007/s00027-011-0216-5.

Datry, T., R. Corti, C. Claret, and M. Philippe. 2011. Leaf litter decomposition along a gradient of flow permanence in a French temporary river: The memory of drying. *Aquatic Sciences* 73:471–483. doi: 10.1007/s00027-011-0193-8.

Death, R.G., Z.S. Dewson, and A.B.W. James. 2009. Is structure or function a better measure of the effects of water abstraction on ecosystem integrity? *Freshwater Biology* 54(10):2037–2050.

Durance, I., and S.J. Ormerod. 2007. Climate change effects on upland stream macroinvertebrates over a 25-year period. *Global Change Biology* 13:942–957.

Durance, I., and S.J. Ormerod. 2009. Trends in water quality and discharge confound long-term warming effects on river macroinvertebrates. *Freshwater Biology*. 54(2):388–405. doi:10.1111/j.1365-2427.2008.02112.x.

Freeman, M.C., C.M. Pringle, and C.R. Jackson. 2007. Hydrologic connectivity and the contribution of stream headwaters to ecological integrity at regional scales. *Journal of the American Water Resources Association* 43(1):5–14.

Griffith, M.B. 2014. Natural variation and current reference for specific conductivity and major ions in wadeable streams of the conterminous USA. *Freshwater Science* 33(1):1–17.

Hamilton, A.T., J.D. Stamp, and B.G. Bierwagen. 2010. Vulnerability of biological metrics and multimetric indices to effects of climate change. *Journal of the North American Benthological Society* 29(4):1379–1396.

Mahlum, S.K., L.A. Eby, M.K. Young, C.G. Clancy, and M. Jakober. 2011. Effects of wildfire on stream temperatures in the Bitterroot River Basin, Montana. *International Journal of Wildland Fire* 20:240–247.

Norton, D.J., J.D. Wickham, T.G. Wade, K. Kunert, J.V. Thomas, and P. Zeph. 2009. A method for comparative analysis of recovery potential in impaired waters restoration planning. *Environmental Management* 44:356–368.

Appendix B: Examples of Development of the Biological Condition Gradient for Large Rivers, Coral Reefs, and Estuaries

This appendix includes examples of work underway on development and application of the conceptual Biological Condition Gradient (BCG) framework to large rivers, estuaries, and coral reefs. These examples illustrate how the BCG framework may be refined for different aquatic systems. The case studies are included here to generate discussion and share information among state water quality program managers and scientists interested in applying to BCG to water bodies other than streams and wadeable rivers. The author's name and affiliation are included for each case study. Contact information is included for the primary author.

The following case examples are included:

B1. Upper Mississippi River: Development of a Biological Condition Gradient for Fish Assemblages of the Upper Mississippi River and a "Synthetic" Historical Fish Community

Ed Rankin, Midwest Biodiversity Institute, Columbus, Ohio[6]

B1.1 Background

As with streams and wadeable rivers, the BCG framework may also be applied to non-wadeable rivers to help assess attainment of aquatic life use (ALU) goal conditions, identify high quality waters, set incremental biological goals for environmental improvements, and track progress in achieving the improvements. Typically, regional reference conditions are used to empirically derive numeric biological thresholds to assess ALU attainment for streams and rivers where sufficient reference sites exist (Hughes et al. 1986; Stoddard et al. 2006). However, for more complex and larger river systems (e.g., large and great rivers), the extent and types of historical alterations of these waters makes the regional reference condition approach difficult (Angradi et al. 2009a).[7]

Large and great rivers are frequently modified by dams, levees, flow controls, water diversions, water withdrawals, and chemical impacts (e.g., effluents, runoff). Conditions that are considered comparable to undisturbed and minimally disturbed conditions (Stoddard et al. 2006), do not exist for these systems, and the Upper Mississippi River (UMR) (Figure B1-1) has a long history of such human alteration (Alexander et al. 2012). Biological multimetric indices used in biological assessments are typically derived using data from least disturbed reference sites and/or stressor response data. Three multimetric indices applicable to large rivers include: the Great River Fish Index (GRFIn) (Angradi et al. 2009); the Fish Assessment Community Index (FACI) (Emery et al. 2007), and the Ohio Continuous Index of Biological Integrity (Ohio CIBI) (Rankin 2010). The GRFIn is based on a stressor derived reference condition, the FACI is based on a method that uses all the data in a continuous scaling approach for calibration (Blocksom 2003), and the Ohio CIBI is based on a regional reference site approach (least disturbed).

This case study explores a synthetic modeling approach (Armitage et al. 2009) for using historic data to model a quantitative description of BCG level 1 and 2 conditions—incorporating historical ecology (e.g., McClenachan et al. 2015) with commonly used assessment methods. BCG level 1 and 2 conditions may currently not be achievable in many large rivers, but knowing the characteristics of a biotic community that would be supported under these conditions may assist in defining incremental and sustainable biological goals for water quality improvements. The synthetic modeling approachis applied here in conjunction with the GRFIn, FACI, and Ohio CIBI to examine how historical data can help define a trajectory toward restoration of all, or elements of, a historic fish assemblage in a highly modified riverine system.

[6] ERankin@mwbinst.com

[7] The definition of great rivers has differed among investigators, but Angradi et al. (2009b) considered great rivers as those with catchments > 1,000,000 km^2, which would include the three mid-continent rivers included in the Environmental Monitoring and Assessment Program (EMAP) Great Rivers Evaluation (GRE), the Mississippi, Missouri, and Ohio Rivers.

B1.2 Upper Mississippi River Biological Assessment Initiative

Currently, ALU assessments are conducted independently by each of the five states bordering the UMR,[8] using different methods and assessing against different thresholds. Yoder et al. (2010) summarized the variety of methods used to assess ALU attainment by Illinois, Iowa, Minnesota, Missouri, and Wisconsin. Furthermore, ALU assessments are based primarily on chemical and physical water quality data collected at widely separated fixed stations. The Upper Mississippi River Basin Association (UMRBA), through its Water Quality Task Force (WQTF), sponsored a project in 2009–2011 to develop a Clean Water Act (CWA) Biological Assessment Implementation Guidance Document for the interstate UMR. This document (Yoder et al. 2011) focused on how to integrate UMR-specific biological assessment approaches into the water quality management programs of the UMR states. It also provided technical methods on conducting biological assessments of the UMR and guidance on how to integrate these methods into the water quality management programs of the UMR states. Additionally, a detailed analysis of existing biological assessment data collected by EPA's Great Rivers Evaluation (GRE) and the U.S. Army Corps of Engineers' Long-Term Resource Monitoring programs was undertaken (Miltner et al. 2011) to develop biological assessment thresholds for the UMR.[9]

Historical knowledge about large and great river fish assemblages is essential in the development of contemporary measures of biological condition and a determination of what thresholds might be attainable. Such information can be obtained from the accounts of pioneering naturalists, settlers, or from early fisheries accounts about these systems (e.g., Trautman 1981; Steuck et al. 2010). Native American middens and fossils and subfossils can also provide historical evidence of fish species occurrence and distribution in these systems (Lyman 2006; Humphries and Winemiller 2009). A description of the historical changes that have occurred in the UMR fish assemblage is available in Pitlo and Rasmussen (2004). These accounts provide important insights about the great river fish assemblages that occurred prior to the extensive alterations of the 19th and 20th centuries.

Some large and great river data sets in the Midwest United States now have at least a 20-year accumulation of fish assemblage data paired with chemical, physical, habitat, and other stressor data. Gradients of ecological sensitivity can be extracted for many fish species from these data sets by examining probabilities of occurrence along chemical, physical, and biological stressor gradients. The combination of contemporary data on species distributions along stressor gradients can be combined with historical accounts of rare, extirpated, or even extinct species to reconstruct "synthetic" historic fish assemblages (Armitage et al. 2009). The resulting model can then be used to "back cast" assemblage condition-stressor relationships to better inform goal setting for a river system.

---

[8] Iowa, Illinois, Missouri, Minnesota, Wisconsin

[9] The UMR in this study extended from the confluence with the Ohio River at Cairo, IL upstream to the upstream most lock and dam in Minneapolis, MN.

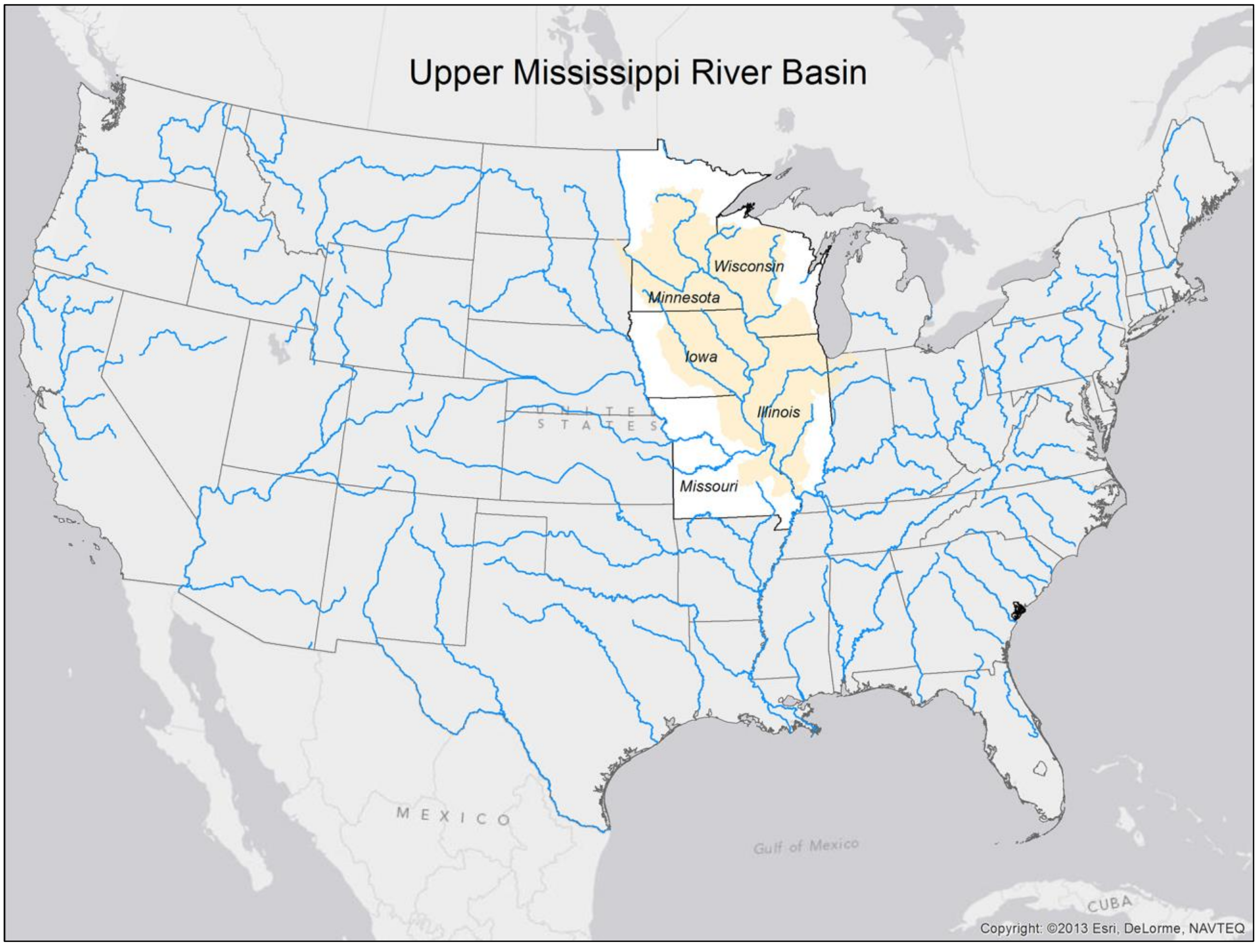


Figure B1-1. Map of the UMR Basin.

B1.3 Methodology for Developing a Synthetic Fish Assemblage

To extrapolate fish species and abundances during a historical pre-disturbance[10] time period in the UMR, large river data sets from the Midwest were used to estimate (1) the frequency of occurrence of a species by biological condition range based on existing fish IBIs; and, (2) the relative catch rates (numbers/km) using boat electrofishing methods for each species. This information was then combined with the historical fish distribution information of Steuck et al. (2010) that includes the historical and present occurrence of fish species in the upper impounded reach, lower impounded reach, and unimpounded reach of the UMR. Known life history information and descriptions of UMR fish populations from historical records (e.g., Carlander 1954) were used to derive extrapolated catch frequencies and abundances that likely occurred prior to the major alterations such as creation of impoundment by navigational dams and modification of the open river reach by levees and wing dams. These frequencies and estimates of abundance were then used to create a "pool" of fish to "sample" using a random selection process. Ten iterations were performed for each of the three UMR reaches, and the data were used to calculate the FACI, the GRFIn for the impounded and open river reaches of the UMR, and the Ohio CIBI for boatable rivers (Table B1-1). The steps in this process are summarized in Table B1-2. In addition to the modeling of pre-settlement conditions, researchers also used early fish

[10] "Pre-disturbance" conditions reflect free-flowing conditions during which early fish distribution patterns were recorded by pioneering naturalists and early settlers, as well as investigated using evidence in Native American middens.

data from large rivers of Ohio that had very poor conditions to model a "very poor" fish assemblage that might have occurred during the 1960s–1980s prior to CWA point source pollution control mandates. For the calculation of the GRFIn, researchers used the current and historical data to recalculate metric ceiling and floor values (95th and 5th percentiles).

Table B1-1. Fish indices applied in the UMR Basin.

Index Acronym	Description	Citation
FACI	Fish Assessment Community Index—A multimetric fish assemblage index developed from sites sampled during the Regional EMAP Large Rivers project	Emery et al. (2007)
GRFIn	Great River Fish Index—A multimetric index created by EMAP-GRE, developed specifically for the UMR and the lower Missouri River	Angradi et al. (2009a)
CIBI	Continuous Index of Biotic Integrity—A continuous scoring form of the Ohio Fish IBI created to improve the scoring sensitivity of the original Ohio IBI and to provide the ability to score historical fish assemblages not truncated to current conditions	Rankin (2010)

Table B1-2. Steps in the development of a synthetic fish assemblage that approximates pre-settlement conditions in the UMR.

Step	Activity Description
1	Compile historical fish assemblage list for reaches of the UMR (i.e., upper impounded, lower impounded, and open river reaches).
2	Use existing data to determine response in abundance and probability of occurrence of each species at biological condition ranges (Very Poor, Poor, Fair, Good, Excellent; for regionally relevant fish IBIs).
3	Estimate typical relative abundance in catch when trend is extrapolated to pre-settlement conditions (use trends from step 2 along with life history information, historical descriptions of occurrence, abundances recorded elsewhere, etc.). Do separately for upper impounded, lower impounded, open river.
4	For rare, extirpated, or extinct species, estimate abundance during pre-settlement periods using historical descriptions, life history information, abundances recorded elsewhere, etc. Do separately for upper impounded, lower impounded, open river.
5	Create "population" of > 100,000 fish for "sampling" by multiplying for each species by the probability of occurrence x the average estimated abundance x 1000.
6	Begin random selection process for "fishing" historical "synthetic" pool of fish—10 iterations for each reach of UMR.
7	Randomly select among best large rivers in Ohio/Indiana data to define maximum abundance and species richness for each iteration; cap richness at randomly selected site +5 and abundance at relative number/km + 500
8	For each iteration, randomly select, without replacement, individuals until species and abundance caps reached.
9	For each iteration a "sampled" assemblage is created, which is scored with appropriate GRFIn, FACI, and Ohio CIBI.

B1.3.1 Inferring Stressor Levels from Species Assemblages

Miltner et al. (2011) employed multivariate and correlative measures using the existing GRFIn and FACI indices and other measures to identify limiting stressors to the fish assemblages in the UMR (i.e., a "top-down" approach). For this case study, an alternative approach used information about individual species' responses to stressors gained from broad-scale studies of species sensitivities. The approach used information to infer which stressors were most limiting, to understand the limiting nature of stressors, and to predict species occurrences and distributions. By examining the inferred stressor levels during historical periods, one can begin to understand which stressor or stressors might be limiting rare species and estimate the feasibility of restoration from current conditions.

In employing this approach, Weighted Stressor Values (WSVs; Meador et al. 2008) were determined for each species in the fish assemblage databases for boat electrofishing sites for purpose of ranking the relative tolerance of fish species to different stressors. Tolerance Indicator Values (TIVs), which are the ordinal ranks of WSVs (1–10) for each species and stressor (Meador and Carlisle 2007) were also derived to place stressors on the same numerical scale.[11] These values were then summed across all sites and divided by the total abundance at all sites to arrive at a WSV.[12] The TIVs were used to infer the stressor level at a site based on the biological assemblage data that were collected, for example, where there was incomplete stressor data. Grand mean TIV values were calculated by creating a mean across all species weighted by the abundance of that species at a site. One goal of this analysis was to estimate how stressor conditions varied between current and historical time periods.

B1.3.2 Assumptions

If sampling were to occur in a riverine habitat with conditions close to "as naturally occurs," it is assumed that sampling would occur along the main channel border and the samples collected would include currently rare or extirpated species from the backwater and side channel habitats. The same assumption has been made by others to conclude that such sampling has been typically representative of the conditions in the backwaters and secondary channels (Angradi 2006; Thorp 1992).

B1.4 Extrapolation of Fish Assemblages to Historical Conditions in the Upper Mississippi River

Researchers described historical condition in the UMR and extrapolated it to approximate BCG level 1 and 2 conditions in order to determine the potential to restore UMR fish assemblages towards this condition. The principal concept is illustrated in Figure B1-2. The dark blue points in the BCG levels 3–5 range represent the existing conditions in the UMR along a generalized stressor gradient. This stressor gradient represents the cumulative stressor load that influences the current condition of the UMR. The green and grey points in the BCG level 1–2 range reflect pre-settlement and immediate post-settlement conditions in the UMR prior to its alteration for commercial navigation.

[11] WSVs are derived for individual stressor variables (e.g., DO, pH, ammonia-nitrogen) as an average (or maximum for toxicants) weighted by the abundance of a species at each site

[12] Calculating TIV scores standardizes WSVs measured on different scales and allows averaging of the TIVs to create a cumulative grand stressor rank across major stressor categories.

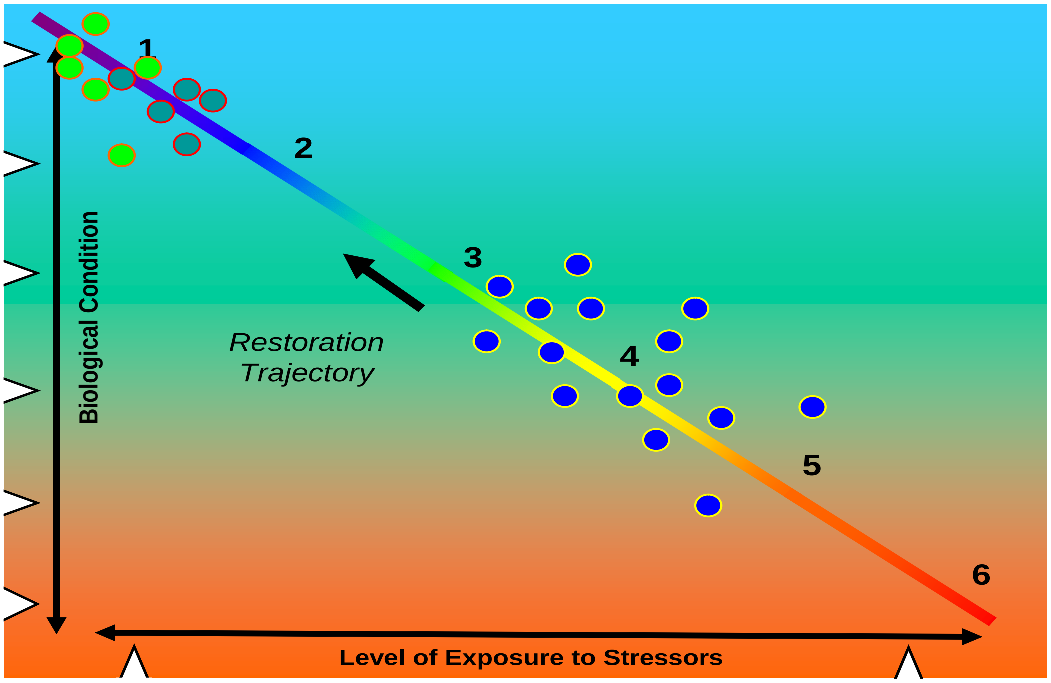


Figure B1-2. The BCG is used to depict the position of current-day UMR fish assemblage as measured by the GRFIn (dark blue points) compared to the "as naturally occurs" pre-impoundment historical condition (green points) that was approximated by the synthetic model. A restoration trajectory is apparent between these conditions. The BCG can be used to define incremental biological improvements along that trajectory.

Using the template of the BCG, UMR fish species and/or other suggested measures were assigned to each of the 10 BCG attributes (Rankin and Yoder 2011). The species assignments provide the probability of capture and average relative abundance of each species with extrapolations to historical conditions.

B1.4.1 Attribute I. Historically Documented, Sensitive, Long-lived, or Regionally Endemic Taxa

Attribute I of the BCG is perhaps among the most influential in defining the characteristics of the UMR as it naturally occurred. This attribute contains information not only about a species occurrence, but also age/size distributions of the long-lived species such as paddlefish, sturgeon, and muskellunge. Nearly all of the species in BCG attribute I are sensitive to pollutants, habitat loss, and other alterations. Some species, such as American eel, are tolerant of pollutants, but they were included because of their life history requirements and to reflect ecological connectance. Because of their migratory habits, the regular occurrence of this species would reflect that the UMR was well connected to the Gulf of Mexico. For this study, the distinquishing characteristics for species assigned to this attributed include: rare, endemic and long lived.

B1.4.2 Attribute II. Highly Sensitive Taxa

The UMR species assigned to attribute II reflect a high level of sensitivity to stressors influencing large Midwest rivers (Table B1-3). While many of these species still occur in the UMR, many are currently rarely observed but would likely be more common in the UMR under reduced levels of stressors (e.g., reduced nutrient enrichment, unimpounded habitats, or connected to backwater and side-channel habitats, Mac et al. 1998; Fremling et al. 1989). This group especially reflects sensitivity to habitat degradation and loss of connectivity (Mac et al. 1998).

Table B1-3. Sample list of fish species collected or reported from the UMR, by state for the first six BCG attributes.

Species	State Endangered, Threatened, Extirpated, or Special Concern Designation					BCG Species Attributes					
						I	II	III	IV	V	VI
	MN	WI	IA	IL	MO	Rare, Endemic, Long-lived	Highly Sensitive	Intermediate Sensitive	Intermediate Tolerance	Tolerant	Non-native
Silver lamprey								X			
American brook lamprey			T				X*				
Chestnut lamprey								X			
Paddlefish						X					
Lake sturgeon	SC	SC	E	E	E	X					
Shovelnose sturgeon							X				
Pallid sturgeon			E	E	E	X					
Alligator gar				E		X					
Shortnose gar									X		
Spotted gar							X				
Longnose gar									X		
Bowfin									X		
Goldeye		E						X			
Mooneye								X			
Skipjack herring	SC	E					X				
Gizzard shad										X	
Threadfin shad								X			
Alabama shad							X				
Central mudminnow									X*		
Grass pickerel			T						X		
Northern pike								X			

E=endangered
T=threatened
SC=special concern
EX=extirpated
X=generally expected to occur; those species with an * associated with nearby smaller tributaries

B1.4.3 Attribute III. Intermediate Sensitive Taxa

Intermediate sensitive species persist through the initial stages of increasing levels of stress. However, they are generally at their highest abundances when stress is the lowest. These species are generally numerically predominant in natural fish assemblages historically.

B1.4.4 Attribute IV. Intermediate Tolerant Taxa

These are fish species that are not sensitive to moderate levels of most stressors and can become predominant as more sensitive species (attribute I, II, and III species) are reduced with increasing stress. Their mere presence suggests little about stressor levels at low to moderate levels of stress; however, combined with the absence or reduction of attribute I–III species, they can be indicative of high stressor levels. Attribute I fish species in the UMR are particularly sensitive to the loss of habitat, particularly floodplain and backwater spawning and nursery habitats (Etnier and Starnes 1993).

B1.4.5 Attribute V. Tolerant Taxa

These species are especially tolerant to most stressors and will persist at increasing levels of stressors above intermediate levels. At the very highest stressor levels, however, most of these species will be reduced in abundance.

B1.4.6 Attribute VI. Non-native or Intentionally Introduced Species

These are fish species that have either been introduced (intentionally or otherwise), and some are now resident in the UMR. Certain of these species (e.g., silver and bighead carp) are potentially more deleterious than others because of their disruption of the food web (Freedman et al. 2012). Most are moderately to highly tolerant of chemical and physical stressors.

B1.4.7 Attribute VII. Organism Condition

Attribute VII measures the condition of individual organisms. Several commonly used biological metrics can be used to gauge the condition of this attribute for fish in UMR large rivers (Table B1-4). Most large river fish assemblage programs use external anomalies to measure the degree of exposure to pollution. Data on multiple year classes for species are generally available from size measurements, and a good distribution of large, older year classes is generally indicative of good conditions. This would also translate to a high diversity by numbers and weights and high indices that reflects these characteristics, such as the Modified Index of Well-Being (MIwb), which is readily available provided that biomass data are collected. The MIwb should be used as a complimentary index with a fish IBI (Yoder and Smith 1999).

Table B1-4. Candidate measures of organism condition for attribute VII in the UMR BCG.

Name	Description
External Anomalies	*Incidence of erosions, lesions, tumors, and deformities observed on fish in a sample. A low incidence reflects sublethal and indirect stressors (e.g., excessive diel dissolved oxygen (DO) variations); high incidence can reflect toxic conditions.*
Multiple Year Classes	*Populations of all expected year classes should exist for all species in attribute groups I–III.*
High diversity based on numbers and weight	*Use the MIwb and its subcomponents based on numbers and weight.*

B1.4.8 Attribute VIII. Ecosystem Function

Great rivers in natural or close to natural conditions (e.g., undisturbed to minimally disturbed conditions) support complex ecosystem functions that result in high diversity and abundance across the various trophic guilds (Mac et al. 1998). Structural measures can potentially be used as surrogates to infer the intactness of ecosystem function (Table B1-5). Work is underway to explore use of candidate surrogate measures (Table B1-5).

Table B1-5. Candidate measures to infer ecosystem functioning in the UMR.

Metric	Description
Invertivores	*The historical UMR was characterized by large numbers of specialized invertivores that fed on the high diversity and production of aquatic invertebrates in multiple habitat types.*
Top Carnivores	*The historical UMR supported a high diversity and biomass of top carnivores that fed on abundant forage fish and other organisms supported by efficient energy cycling through the system.*
Omnivores	*Omnivores were not predominant in the mainstem UMR given the abundant insects and mussel assemblages that occurred in the river. A shift to predominance by omnivores would reflect an alteration to nutrient inputs and cycling.*

B1.4.9 Attribute IX. Spatial and Temporal Extent of Detrimental Effects

Attribute IX is especially important for temperate floodplain rivers and the UMR in particular. The extent of direct alterations to the UMR from the navigational impoundments and leveeing and wing dams in the open river reach has been system-wide and affects the entirety of the interstate UMR. These modifications have disconnected much of the mainstem from its former backwaters, modified the flow regime, and altered the original riverine habitats to a more lentic (upper impounded) or channelized (open river) condition. The modification of the original forested and wetland-dominated landscape by row cropping has affected flow, habitat, and water quality. These stressors, especially habitat and flow, are currently limiting to the recovery potential of the UMR fish assemblages. However, this study does not explore approaches to quantify this attribute.

B1.4.10 Attribute X. Ecosystem Connectance

Attribute X relates directly to the ability of fish species in the UMR main channel to move laterally into and out of adjacent backwaters, sloughs, and oxbows that were once characteristic of the UMR. The periodic but regular inundation of the floodplain to which many attribute I–III species are adapted has been altered by impoundment or leveeing of the UMR. Many of the sensitive species that are now rare or extirpated were associated with these connected, but off-channel habitats. While fish can move upstream and downstream, the ease with which this now takes place has been modified by the navigational dams and the alteration of flows and riverine habitat. As with attribute VIII, work is underway to explore use of biological information and attribute definitions as surrogate biological measure for this attribute.

B1.5 Synthetic Assemblage Results

The basis for deriving a synthetic historical fish assemblage is the observation that the probability of capture and average abundance of a species is related to the array of stressors present in a reach and is reflected in the biological indices used in this study (e.g., GRFIn, Ohio CIBI, FACI). Researchers have used this information to derive probabilities of capture and extrapolated abundances for the historical period prior to impoundment and a time period with poor to very poor water quality conditions caused by untreated wastewater discharges (e.g., 1960s). Figure B1-3 illustrates changes in the probability of capture, relative abundance, and abundance by capture rate from Ohio data for three key riverine species in the UMR: the blue sucker, river darter, and black buffalo. The extrapolation to historical data used to derive the pool of potential fish for the historical IBI was developed using the trend of actual data, the historical reports of occurrences and distribution (Steuck et al. 2010), and life history information and other historical sources that describe the general occurrence of these species in large Midwest rivers prior to the anthropogenic impacts of the past two centuries. Because natural species distributions vary geographically, the modeling was done separately for each of the three reaches of the UMR (upper impounded, lower impounded, and open river) as defined by Miltner et al. (2011).

The GRFIn indices, the regional FACI score, and the Ohio CIBI were then calculated using the synthetic data for the impounded and open river reaches of the UMR. The Ohio CIBI scoring ranges were not limited by existing conditions, but assumed that species richness metrics were greater in the past and allowed for higher scoring than current existing species richness levels. As expected, the synthetic data resulted in higher GRFIn, FACI, and CIBI scores than the sampled data (Figure B1-4). Estimates of abundance for these metrics were based on abundances observed at the best existing UMR sites and, for rare species, were based on extrapolations based on species life history knowledge and historical descriptions of abundances when available.

B1.5.1 Initial Reconstruction of Environmental Conditions to Match Biological Condition Gradient Levels 1–2

The synthetic fish assemblages are intended to approximate levels 1–2 of the BCG for the UMR during pre-settlement periods. Environmental conditions were inferred based on the grand ranking of TIV scores for the synthetic and existing data and plotted against the GRFIn and FACI for the impounded and open river reaches (Figure B1-5). Despite some overlap in terms of the extrapolated stressor levels with some of the recent data, particularly from sections of the unimpounded reaches of the UMR, there is a substantial degree of separation between the stressor levels approximating the historical and the present day assemblages. The synthesized data representing the 1960s are among the lowest FACI and GRFIn scores and coincide with the higher ranges of the extrapolated stressor ranking.

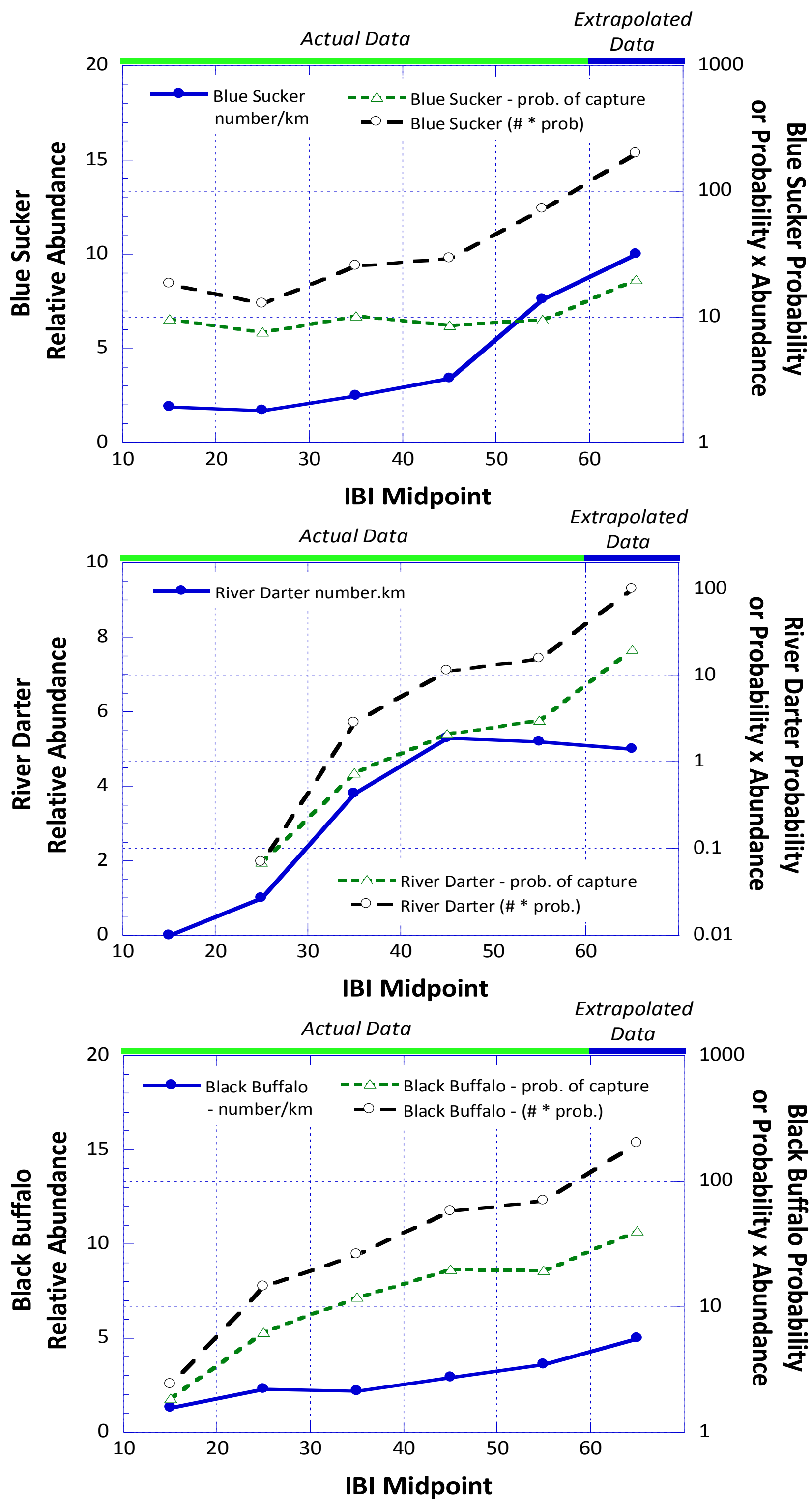
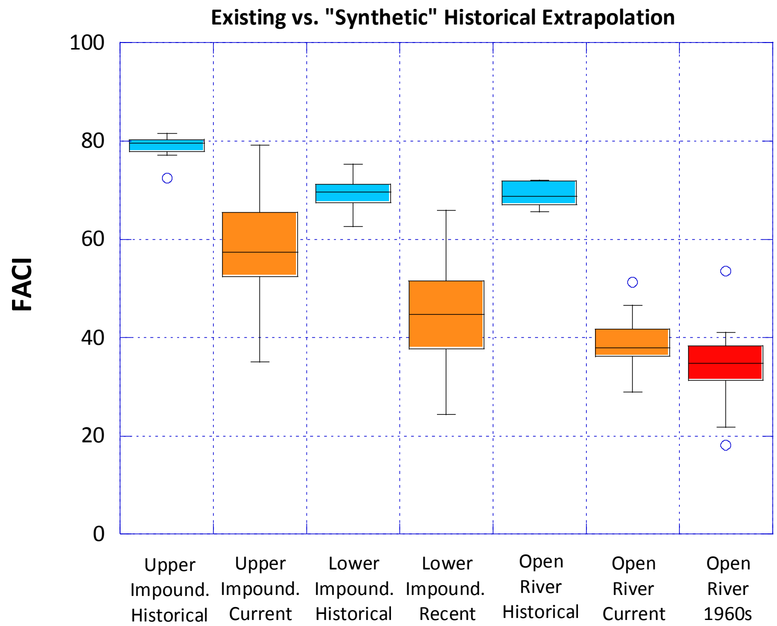


Figure B1-3. Plots of relative abundance, probability of capture, and abundance x probability vs. IBI midpoint for three riverine fish species: blue sucker, river darter, and black buffalo. Actual abundance data and probability of capture data generated from data on boatable sites from Ohio and Indiana; extrapolated data estimated using best professional judgment based on trends in actual data, data on historical distributions in the UMR (Steuck et al. 2010), and life history information and other historical information.



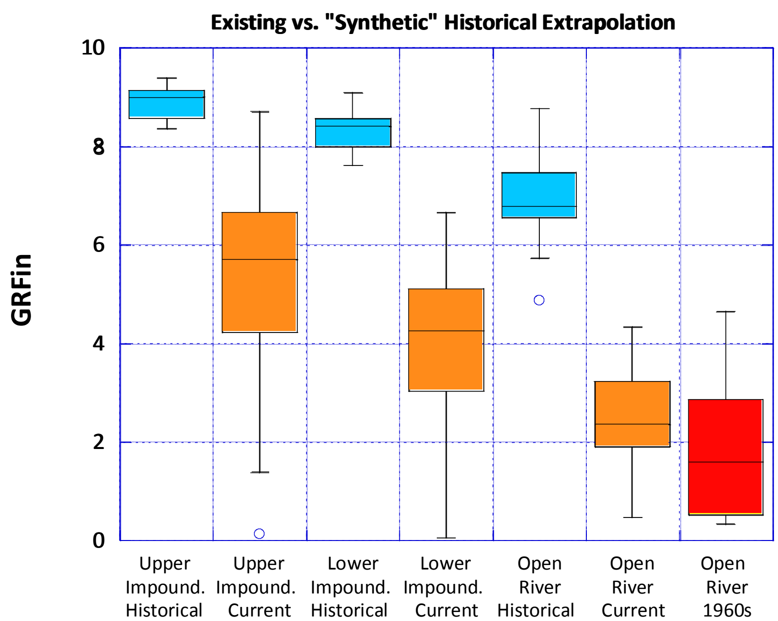


Figure B1-4. Box-and-whisker plots of FACI scores (top) and GRFIn scores (bottom) for historical "synthetically" derived fish assemblages (blue) and present-day data (orange) for the upper impounded, lower impounded, and the open river reaches of the UMR. The red shaded box are synthetically derived scores in the open river during the 1960s prior to CWA mandated point source controls.

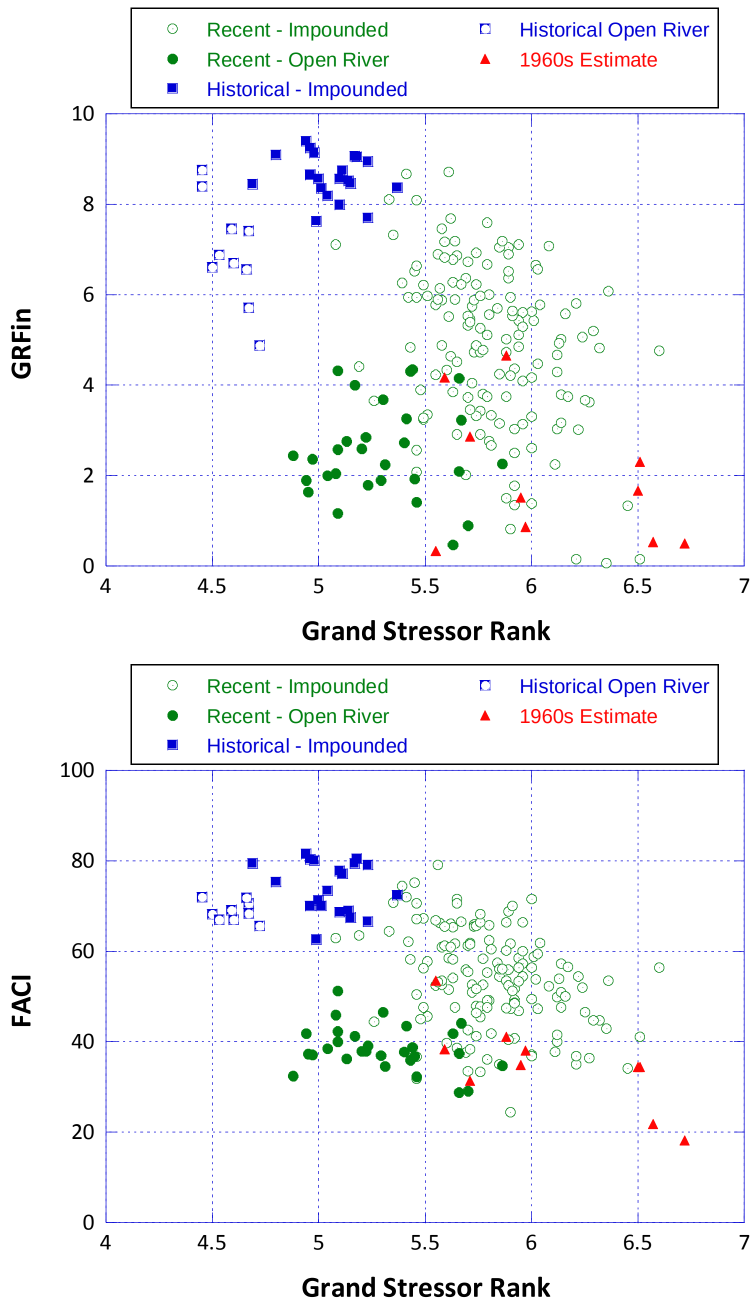


Figure B1-5. Plots of Grand Stressor Rank based on the average of TIV ranks for species collected at sites vs. the GRFIn (top) and FACI (bottom) for historical synthetic data (blue), recent actual data (green), and synthetic 1960s era data (red triangles) in the impounded and open river reaches of the UMR.

B1.5.2 Using the Biological Condition Gradient to Establish Attainable Biological Thresholds

The derivation of biological thresholds to assess ALU attainment using an Environmental Monitoring and Assessment Program (EMAP) data set focused on the statistical assumptions and consequences of using several different methods and approaches (Miltner et al. 2011). Researchers in this analysis suggest that such a statistical approach should be linked explicitly to biological inferences and narratives about their position along the BCG for improved communication with decisionmakers and stakeholders.

The effort to develop a synthetic historical fish assemblage was done to provide a foundation for evaluating the attainability of various biological thresholds. This approach helps define biological thresholds that can be interpreted relative to the CWA biological integrity objective. This framework assists in understanding current conditions along a gradient of stress and reduces the risk of setting thresholds based on an assumption that the current condition of a waterbody represents its full ecological potential when it may be significantly degraded (Humphries and Winemiller 2009). Executing this approach requires that the key measures of biological condition be linked to stressor gradients so that evaluation of the recovery potential of a large river, or segment within the river, can be performed. The statistical approaches conducted in the UMR thresholds analysis (Miltner et al. 2011) were designed to offer an analysis of various ways of developing biological thresholds and then evaluating the attainability of these thresholds.

One result was readily apparent in this analysis—all three UMR reaches historically had similar levels of attribute I, II and III species, i.e., the rare, endemic, long lived species;highly sensitive species; and intermediate sensitive species. This was especially important for the open river reach, because a variant of the GRFIn was derived and calibrated to current conditions in this UMR reach, which is highly modified, thus setting expectations at its current level of alteration. The open river reach is the most highly modified of all three reaches examined, and this is especially reflected by a higher proportion of tolerant and exotic individuals (Figure B1-6). The comparable levels of historic condition also suggest that the impounded GRFIn and stressor gradients could be extended to the open river. Presently, the open river has been treated separately from the impounded sections of the UMR in the derivation and calibration of the GRFIn.

To relate the BCG attributes to the current indices, the number of species in each attribute were compared to the GRFIn and FACI indices. Both indices showed a significant correlation with attribute III (Figure B1-7 and Figure B1-8, top). There was, however, no apparent correlation between BCG attributes I and II and either the GRFIn or the FACI (Figure B1-7 and Figure B1-8, middle and bottom). The lack of correlation with BCG attributes I and II may well be explained by the large difference between historical and existing conditions in the UMR – and that these more sensitive species appear to have been eliminated from the UMR. It could be that populations of these species in the main channel samples might be more related to the losses in connectivity with the side channel habitats than with the conditions in the main channel itself. The presence of the intermediate sensitive species though provides some promise that restoration of the more sensitive species may be possible if conditions improve and/or parts of the river are reconnected.

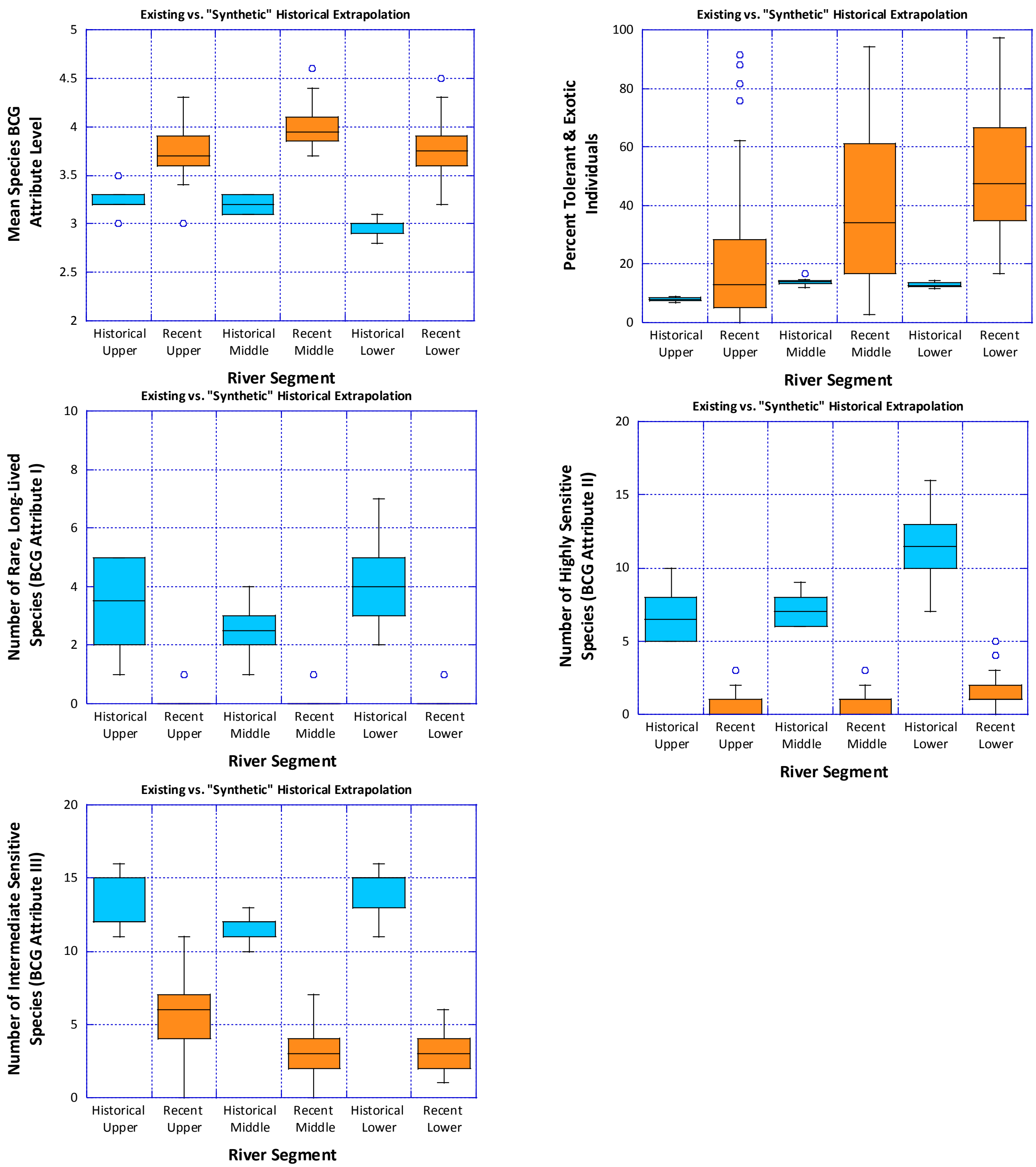


Figure B1-6. Box-and-whisker plots of BCG attributes for recent data (orange) and historical synthetic data (blue) for the upper impounded (river mile (RM) 523–812), lower impounded (RM 523–196), and open river (RM 196–0) UMR for mean species BCG attribute (upper left), mean tolerant and exotic species (upper right), number of rare, long-lived species (I) (middle, left), highly sensitive species (II) (middle, right), and intermediate sensitive species (III) (bottom, left).

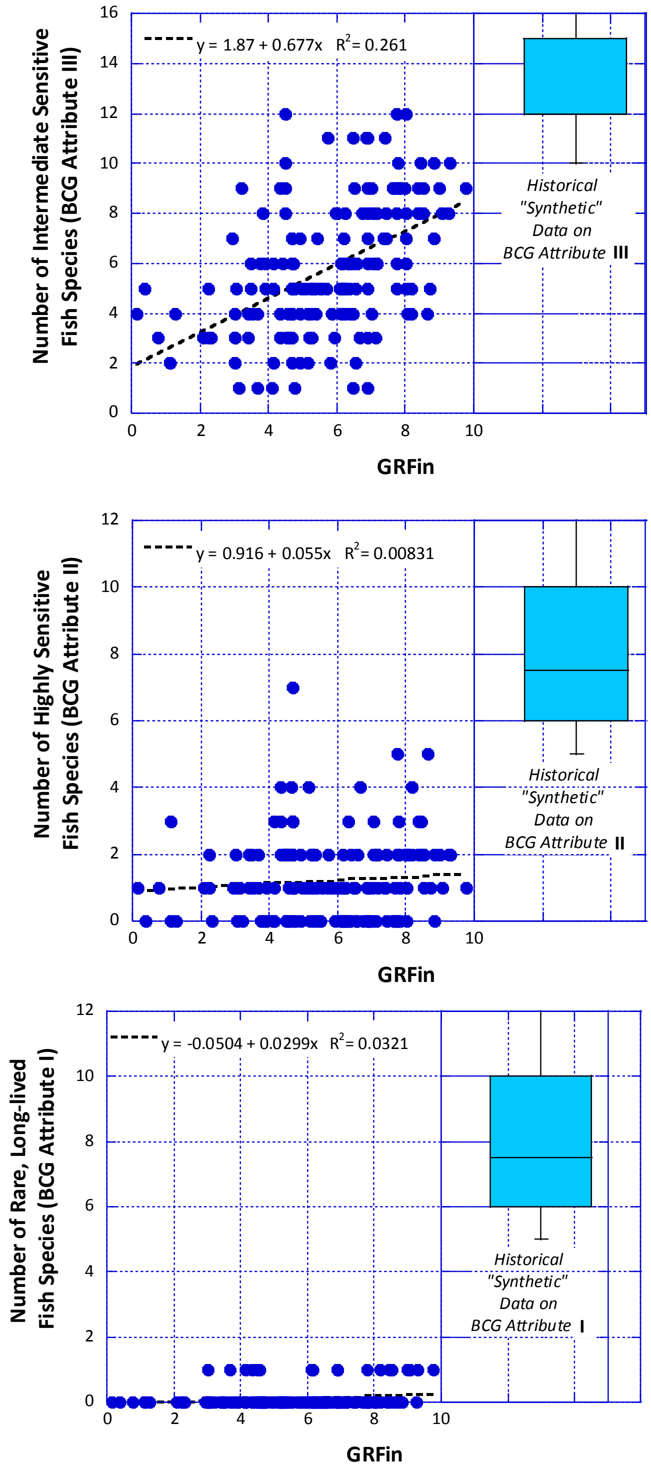


Figure B1-7. Scatter plots of the GRFIn index for existing data vs. BCG attributes III (top), II (middle), and I (bottom) from these data. Historical distribution of synthetic data for these attributes for the UMR is illustrated with a box plot to the right of each box.

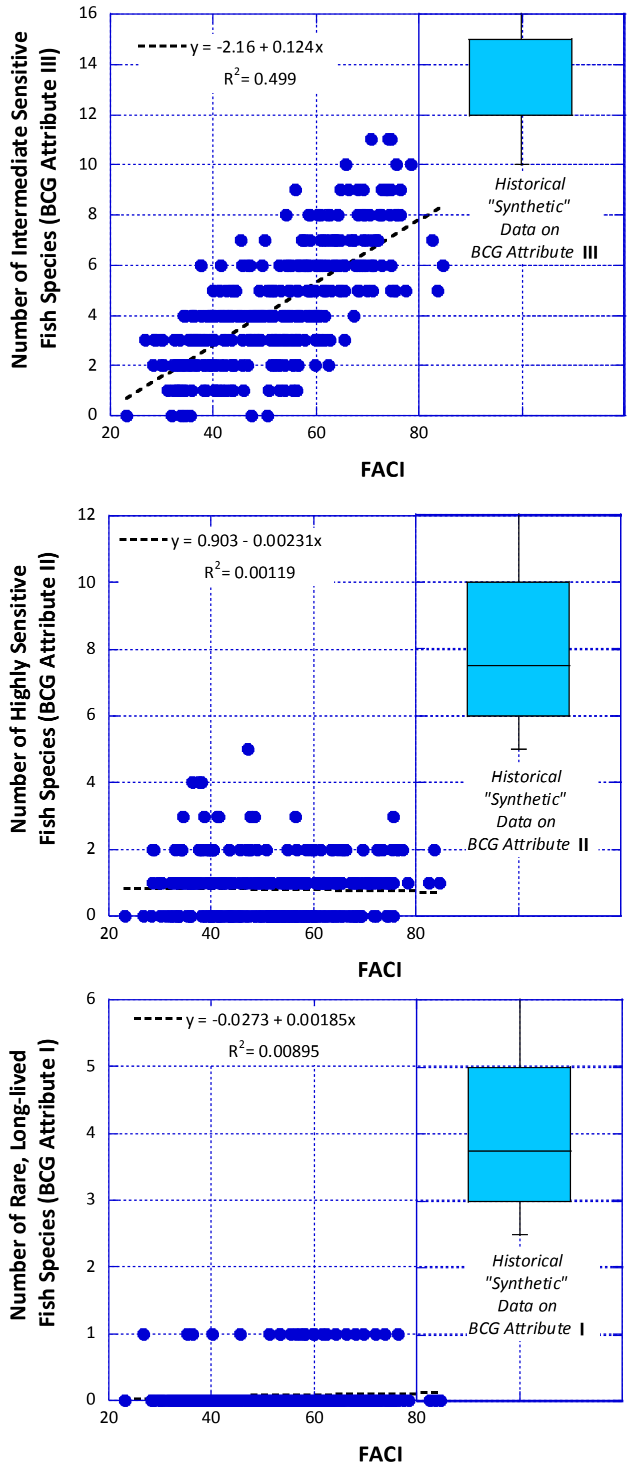


Figure B1-8. Scatter plots of the FACI index for existing data vs. BCG attributes III (top), II (middle), and I (bottom) from these data. Historical distribution of synthetic data for these attributes for the UMR is illustrated with a box plot to the right of each box.

B1.5.3 Using the Biological Condition Gradient and Biological Indices to Derive Aquatic Life Use Thresholds for the Upper Mississippi River

BCG attributes were used to derive cutoffs for the GRFIn, FACI, and Ohio CIBI based on best professional judgement using the relationship between the BCG attributes and index scores. Figure B1-9 shows plots of the GRFIn, FACI, and Ohio CIBI versus the number of combined BCG attribute I, II and III species at each UMR site. The BCG I, II, III species were combined to approximate the fish assemblage of the UMR that may be achieved, in part or in whole, with WQ improvements, mitigation and/or restoration efforts in the future. At a minimum, this "synthetic" assemblage can be used to inform efforts to improve conditions and restore a more naturally functioning and connected river system. The synthetic data are coded with blue squares to distinguish it from the present-day sampling data (green circles), and the open river reach is coded with solid orange circles. The highly degraded synthetic results are also included as red triangles in order to have the full breadth of the BCG represented. All three of the indices reflect a positive relationship with the number of BCG attribute I, II and III species (Figure B1-9) based on a locally weighted regression that minimizes the effect of outliers. The breaks in these curves illustrate patterns in the relationships that can be used to support various options for selecting tiered impairment thresholds. The break in the curves in these relationships with the weighted BCG is aided by the availability of the synthetic data to complete the curves (Figure B1-9) and is informative if higher aquatic life thresholds are desired. The tighter relationship between the FACI, the Ohio CIBI, and BCG attribute I, II and III species compared to the GRFIn is likely related to the similar metrics in these indices and a broader geographic basis for their derivation. The GRFIn is designed to maximize the association with a derived stressor gradient (Angradi et al. 2009a). The Ohio CIBI provides a way to separate high and low performing sites beyond the current range of that index based on contemporary conditions. Actual data from the UMR are also lacking for the time period when point source pollution stressors were the most severe (1950s–1970s), which presumably resulted in assemblages characteristic of BCG level 6. The availability of such data, which were synthesized the same way as were the historical conditions, should enhance change point analyses and make the indices more sensitive to the extremes of the disturbance gradient (e.g., the Ohio CIBI). Although the CIBI was originally calibrated for smaller large rivers, the application of the method can better illuminate change points since the other indices were calibrated to accommodate estimates of historical assemblage condition.

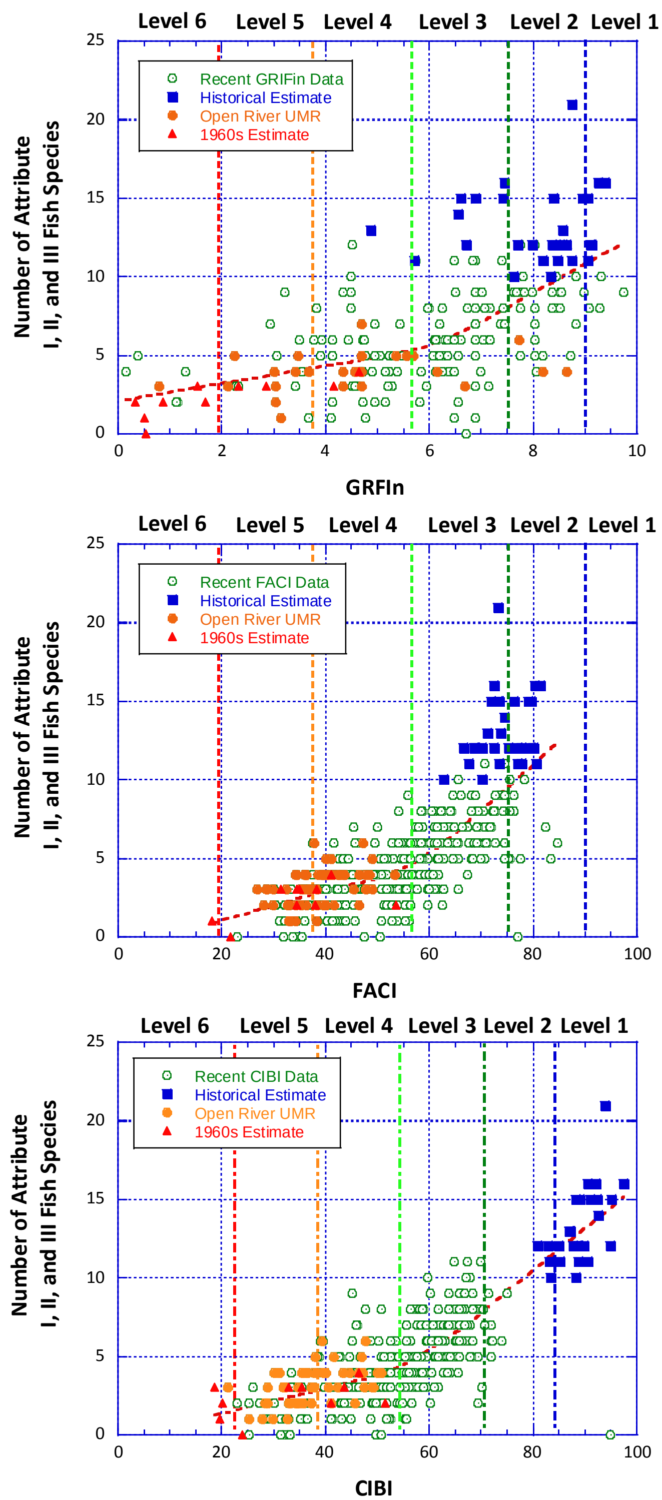


Figure B1-9. Plot of the GRFIn (top), FACI (middle), and CIBI (bottom) sectioned by BCG level vs. the number of BCG attribute I/II/III species (common, sensitive) for sites in the UMR. Historical calculations were not available for the GRFIn, but were extrapolated from correlations with the FACI.

B1.6 Conclusions

The UMR, like many Midwest rivers, has been subjected to a series of perturbations that have accelerated greatly with European settlement beginning in the 19th century. Some of the historically common UMR species are now rare, but most remain present if even in limited numbers and distribution. Present-day conditions, however, have improved since the zenith of gross pollution from untreated industrial and human wastewater sources during the late 19th and the first half of the 20th century.

Linking existing UMR biological indices to the BCG levels and individual attributes may help strengthen the technical basis and ability to communicate the rationale for setting appropriate and attainable biological thresholds for the UMR. BCG levels 1 and 2 represent undisturbed or minimally disturbed conditions and can be characterized based on historical data and records when these conditions no longer exist. BCG levels 3 and 4 represent biological assemblages that have been subject to increasing levels of stress, but which still include some representative species that would be expected under undisturbed or minimally disturbed conditions. As an initial first step, linkage between specific BCG attributes, the GRFIn, FACI, and the Ohio CIBI were examined and BCG level thresholds proposed.

The BCG and individual attributes can also be also used to provide a narrative backup to the statistically derived impairment thresholds of Miltner et al. (2011). The distance between the present-day conditions and the "as naturally occurs" conditions that once existed in the UMR leaves much room for restoration, but restoration also requires an awareness about the status of present-day UMR fish assemblages with respect to the currently available indices such as GRFIn, FACI, and the Ohio CIBI. The fact that many of the historically common fish species are still present indicates that habitats still exist to support at least relict populations of BCG attribute I–III species.

Appendix B1 References

Alexander, J.S., R.C. Wilson, and W.R. Green. 2012. *A Brief History and Summary of the Effects of River Engineering and Dams on the Mississippi River System and Delta*. U.S. Geological Survey Circular 1375. U.S. Geological Survey, Lincoln, NE.

Angradi, T.R., ed. 2006. *Environmental Monitoring and Assessment Program, Great River Ecosystems Field Operations Manual*. EPA/620/R06/002. U.S. Environmental Protection Agency, Office of Research and Development, Washington, DC. http://www.epa.gov/emap/greatriver/fom.html. Accessed February 2016.

Angradi, T.R., M.S. Pearson, T.M. Jicha, D.L. Taylor, D.W. Bolgrien, M.A. Moffett, K.A. Blocksom, and B.H. Hill. 2009a. Using stressor gradients to determine reference expectations for great river fish assemblages. *Ecological Indicators* 9(4):748–764.

Angradi, T.R., D.W. Bolgrien, T.M. Jicha, M.S. Pearson, B.H. Hill, D.L. Taylor, E.W. Schweiger, L. Shepard, A.R. Batterman, M.F. Moffett, C.M. Elonen, and L.E. Anderson. 2009b. A bioassessment approach for mid-continent great rivers: The Upper Mississippi, Missouri, and Ohio (USA). *Environmental Monitoring and Assessment* 152:425–442.

Armitage, B.J., R. Mueller, and E.T. Rankin. 2009. An Assessment of Threats to the Biological Reference Expectations for Great River Fish Assemblages. Ecological Indicators Condition of the Wabash River Aquatic Ecosystem of Indiana. In two parts. Prepared for The Indiana Nature Conservancy, Indianapolis, IN.

Blocksom, K.A. 2003. A performance comparison of metric scoring methods for a multimetric index for Mid-Atlantic highlands streams. *Journal of Environmental Management* 31:670–682.

Carlander, H.B. 1954. *A History of Fish and Fishing in the Upper Mississippi River*. UMRCC Special Publication.

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.

Emery, E., R. Tewes, R. Argo, and J. Thomas. 2007. *Development of a Probability Based Monitoring and Assessment Strategy for Select Large Rivers within U.S. EPA Region 5*. Report in fulfillment of EPA Grant RM-83169201. ORSANCO, Cincinnati, OH.

Etnier, D.A., and W.C. Starnes. 1993. *The Fishes of Tennessee*. University of Tennessee Press, Knoxville.

Freedman, J.A., S.E. Butler, and D.H. Wahl. 2012. *Impacts of Invasive Asian Carps on Native Food Webs*. Final project report to Illinois–Indiana Sea Grant. University of Illinois, Sullivan.

Fremling, C.R., J.L. Rasmussen, R.E. Sparks, S.P. Cobb, C.F. Bryan, and T.O. Claflin. 1989. Mississippi River fisheries: A case history. Pages 309–351 in D.P. Dodge, editor. *Proceedings of the International Large River Symposium*. Canadian Special Publication of Fisheries and Aquatic Sciences 106, Ottawa, Ontario.

Hughes, R.M., D.P. Larsen, and J.M. Omernik. 1986. Regional reference sites: A method for assessing stream potential. *Environmental Management* 10:629–635.

Humphries, P., and K.O. Winemiller. 2009. Historical impacts on river fauna, shifting baselines, and challenges for restoration. *BioScience* 59(8):674–684.

Kopf, R.K., C.M. Finlayson, P. Humphries, N.C. Sims, and S. Hladyz. 2015. Anthropocene baselines: Assessing change and managing biodiversity in human-dominated aquatic ecosystems. *BioScience* 65(8):798–811.

Lyman, R.L. 2006. Paleozoology in the service of conservation biology. *Evolutionary Anthropology* 15:11–19.

Mac, M.J., P.A. Opler, C.E. Puckett Haecker, and P.D. Doran. 1998. *Status and Trends of the Nation's Biological Resources*. 2 vols. U.S. Department of the Interior, U.S. Geological Survey, Reston, VA.

McClenachan, L., A.B. Cooper, M.G. McKenzie, and J. A. Drew. 2015. The importance of surprising results and best practices in historical ecology. *BioScience* 65(9):1–8.

Meador, M.R., and D.M. Carlisle. 2007. Quantifying tolerance indicator values for common stream fish species of the United States. *Ecological Indicators* 7(2):329–338.

Meador, M.R., D.M. Carlisle, and J.F. Coles. 2008. Use of tolerance values to diagnose water-quality stressors to aquatic biota in New England streams. *Ecological Indicators* 8(5):718–728.

Miltner, R.J., C.O. Yoder, and E.T. Rankin. 2011. *Preliminary Analysis of Biological Assessment Thresholds for Determining Aquatic Life Use Attainment Status in the Upper Mississippi River Mainstem*. MBI Technical Report/2011-5-1. Upper Mississippi River Basin Association, St. Paul, MN. http://www.midwestbiodiversityinst.org/publications. Accessed February 2016.

Pitlo, J., Jr., and J.L. Rasmussen. 2004. *A Compendium of Fishery Information on the Upper Mississippi River*. 3rd ed. UMRCC Special Publication. Conservation Committee.

Rankin, E.T. 2010. *Calibration of the Ohio IBI and ICI Using Continuous Scoring Methods*. Report to Ohio EPA, Division of Surface Water. Ohio University, Voinovich School for Leadership and Public Affairs, Athens, OH.

Rankin, E.T., and C.O. Yoder. 2011. *Improving Water Quality Standards and Assessment Approaches for the Upper Mississippi River: UMR Clean Water Act Biological Assessment Implementation Guidance: Development of a Biological Condition Gradient for Fish Assemblages of the Upper Mississippi River and Development of a "Synthetic" Historical Fish Community*. MBI Technical Report/2011-5-2. Submitted to UMRBA WQTF. 24 pp. http://www.umrba.org/wq.htm. Accessed February 2016.

Rankin, E.T., and C.O. Yoder. 2012. *Improving Water Quality Standards and Assessment Approaches for the Upper Mississippi River: Development of a Biological Condition Gradient for Fish Assemblages of the Upper Mississippi River and Development of a "Synthetic" Historical Fish Community.* MBI Technical Report/2011-5-2. Upper Mississippi River Basin Association, St. Paul, MN. http://www.midwestbiodiversityinst.org/publications. Accessed February 2016.

Steuck, M.J., S. Yess, J. Pitlo, A. Van Vooren, and J. Rasmussen. 2010. *Distribution and Relative Abundance of Upper Mississippi River Fishes*. Upper Mississippi River Conservation Committee, Onalaska, WI.

Stoddard, J.L., D.P. Larsen, C.P. Hawkins, R.K. Johnson, and R.H. Norris. 2006. Setting expectations for ecological condition of running waters: The concept of reference condition. *Ecological Applications* 16:1267–1276.

Thorp, J.E. 1992. Linkage between islands and benthos in the Ohio River, with implications for riverine management. *Canadian Journal of Fisheries and Aquatic Sciences* 49:1873–1882.

Trautman, M. 1981. *The Fishes of Ohio*. Revised edition. The Ohio State University Press, Columbus.

Yoder, C.O., and M.A. Smith. 1999. Using Fish Assemblages in a State Biological Assessment and Criteria Program: Essential Concepts and Considerations. In *Assessing the Sustainability and Biological Integrity of Water Resources using Fish Communities*, T.P. Simon (ed.), pp. 17–56. CRC Press, Boca Raton, FL.

Yoder, C.O., V.L. Gordon, N.B. Kale, and D.K. Hokanson. 2010. *Improving Water Quality Standards and Assessment Approaches for the Upper Mississippi River: UMR Clean Water Act Biological Assessment Implementation Guidance—Background and Scoping Report*. Upper Mississippi River Basin Association, St. Paul, MN.

Yoder, C.O., R.J. Miltner, V.L. Gordon, E.T. Rankin, N.B. Kale, and D.K. Hokanson. 2011. *Improving Water Quality Standards and Assessment Approaches for the Upper Mississippi River: UMR Clean Water Act Biological Assessment Implementation Guidance*. Upper Mississippi River Basin Association, St. Paul, MN. http://www.umrba.org/publications.htm#wq. Accessed February 2016.

B2. Narragansett Bay: Development of a Biological Condition Gradient for Estuarine Habitat Quality

Emily Shumchenia, PhD, E&C Enviroscape, LLC[13]; Giancarlo Cicchetti, PhD, and Marguerite C. Pelletier, PhD, EPA Office of Research and Development, National Health and Environmental Effects Research Laboratory–Atlantic Ecology Division, Narragansett, Rhode Island

B2.1 Background

Estuarine waters are affected by a variety of stressors acting at several scales: localized point sources of contaminants; widespread or diffuse nonpoint sources of contaminants such as nutrients; and global impacts such as climate change. Consequently, these valued ecosystems are greatly affected by the cumulative impacts of multiple stressors. Over time, this has led to "severe, long-term degradation of near-shore marine systems worldwide" (Lotze et al. 2006). As such, it is critical to have a way to interpret biological condition consistently and independently of assessment methods for estuaries and other coastal systems.

Biological condition integrates the effects to living organisms from exposure to stressors. A biological assessment is an effective tool in managing cumulative impacts. Many different biological assessment methods and biological indices to quantify biological condition have been developed and applied by scientists, local resource managers, states, and federal agencies. Most assessments evaluate changes in quality or quantity of ecologically or economically valued habitats, communities, or species relative to a defined reference condition. These assessments, when applied in different estuaries, often evaluate very different aspects of biology and use different reference conditions, usually for the good reason that biology itself differs among estuaries.

Few tools or frameworks exist to evaluate and manage the gradual degradation of estuaries and estuarine functions along with the ecological and social benefits they provide over decades. A method to interpret biological condition consistently, regardless of location, time, or assessment method, would allow scientists and water quality managers to compare assessments of aquatic resource condition more uniformly and directly, and communicate more clearly to the public both the current status of aquatic resources and their potential for restoration (Davies and Jackson 2006; USEPA 2011).

B2.2 Development of an Estuarine Biological Condition Gradient Framework

The estuarine BCG approach was initially proposed and launched at a 2005 workshop hosted by EPA (Office of Water and Region 1) in Providence, Rhode Island. Concepts were developed further at workshops in Maine during the winter of 2006 and spring of 2007. The approach was solidified when the EPA Office of Water, Region 1, and Office of Research and Development co-sponsored a November 2008 workshop in Narragansett, Rhode Island, inviting many national estuarine experts and managers. The goal of these efforts was to develop and refine a nationally consistent, integrative estuarine BCG framework to enable meaningful comparisons among metrics and water bodies.

[13] emily.shumchenia@gmail.com

The proposed estuarine framework considered structure, function, condition, connectivity, and non-native species in water bodies at multiple scales, including the species- and habitat-scale (e.g., seagrass health measures, benthic faunal indices), as well as the whole-estuary scale (e.g., measures of the estuarine mosaic of living habitats). The proposed estuarine framework drew from existing ideas on estuarine biological assessment:

- *The Biological Condition Gradient: A Descriptive Model for Interpreting Change in Aquatic Ecosystems* (Davies and Jackson 2006)
- *Estuarine and Coastal Marine Waters: Bioassessment and Biocriteria Technical Guidance* (EPA 2000)

The attributes, potential metrics, and condition levels developed for the estuarine BCG are shown in Table B2-1. Table B2-1 is intended to assist with organization of metrics, but users should recognize that applicable indicators can be estuary-specific, in terms of what data are collected by estuary programs and how often biological monitoring occurs.

Table B2-1. Attributes and potential metrics developed at the 2008 BCG workshop (left two columns) paired with examples of narrative for BCG levels (right 6 columns).

Attribute	Potential Metrics and Descriptions	Examples of BCG Level Narratives (based on Davies and Jackson 2006 and recommendations of a panel of experts)					
		1	2	3	4	5	6
Structure and Compositional Complexity	Measures of water body, community, or habitat structure and complexity, also recognizing loss of habitats or species due to human activities. Examples include macroinvertebrate or fish indices, phytoplankton or zooplankton community measures, epifaunal measures, biotope measures, presence/quantity of sensitive taxa or biotopes, measures of seagrass and macroalgae	Community composition is as naturally occurs except for global extinctions; patterns of primary production, biotope measures, and communities with large, long-lived and sensitive species are as naturally occurs	Slight changes in natural occurrences of biotopes or patterns of primary production; minimal changes in abundances of sensitive or tolerant species	Evident changes in biological metrics; decreases in sensitive species and increases in tolerant species; some evident changes in patterns of primary production; decreases in sensitive habitat area and changes to biotope measures are also evident	Moderate changes in biological metrics; some sensitive, large, or long-lived taxa may be markedly diminished or absent; increases in tolerant species; many evident changes in patterns of primary production; biotope measures significantly altered with replacement of natural habitats/biotopes by tolerant or non-naturally occurring components	Sensitive, large and/or long-lived taxa are markedly diminished or absent, with a dominance in abundance of tolerant taxa; significant shifts in species diversity, size, and densities of remaining species; biotope measures significantly altered; most sensitive natural habitats/biotopes lost with replacement by tolerant or non-naturally occurring components	Sensitive, large and/or long-lived taxa are absent, with extremes in abundance of tolerant taxa; extreme shifts in species diversity and in size spectra of remaining organisms; extreme alteration of natural biotope measures with complete loss of sensitive habitats

Attribute	Potential Metrics and Descriptions	Examples of BCG Level Narratives (based on Davies and Jackson 2006 and recommendations of a panel of experts)					
		1	2	3	4	5	6
Condition	Measures of the condition ("health") of water bodies, habitats, or species. Also includes measures of resiliency. Examples include harmful algal blooms, disease outbreaks, outbreaks of other harmful taxa, measures of habitat or biotope health such as seagrass condition or wetland condition, fish pathology or shellfish bed condition, measures of reproductive success	Diseases, harmful algal blooms, other outbreaks, measures of reproductive success, and condition/health measures are as naturally occurs	Diseases, harmful algal blooms, other outbreaks, measures of reproductive success, and condition/health measures are as naturally occurs	Incidences of diseases, harmful algal blooms, and other outbreaks are infrequent; reproductive success and condition/health measures are within the range of variability for naturally occurring characteristics	Incidences of diseases, harmful algal blooms, and other outbreaks may be slightly higher than expected; reproductive success and condition/health measures may be slightly lower than expected	Incidences of diseases, harmful algal blooms, and other outbreaks are increasingly common; reproductive success and condition/health measures are significantly lower than expected	Diseases, harmful algal blooms, and other outbreaks are common and serious, reproduction is minimal except for extremely tolerant groups, and condition/health measures are extremely low
Function	Measures of energy flow, trophic linkages and material cycling, including proxy or snapshot metrics that correlate to functional measures. Examples include photosynthesis: respiration ratios, benthic: pelagic production rates, chlorophyll a concentrations, benthic bioturbation, and form/ biomass of primary production	Energy flows, material cycling, and other functions are as naturally occur, typically characterized by complex interactions and many complex trophic links supporting large, long-lived organisms	Energy flows, material cycling, and other functions are as naturally occur, typically characterized by complex interactions and many complex trophic links supporting large, long-lived organisms	Virtually all functions are maintained through operationally redundant system attributes; minimal changes to some indicative functions	Most functions are largely maintained through operationally redundant system attributes, though there is evidence of loss of complexity, efficiency, and shifts in some rates	Losses of some ecosystem functions are apparent, manifested as changes in energy and material flows, functional rates, or reduced complexity	Most functions show extensive and persistent disruption: shifts in primary production, microbial dominance, highly simplified trophic structure, extreme shifts in energy and material processing rates, extensively reduced complexity

Attribute	Potential Metrics and Descriptions	Examples of BCG Level Narratives (based on Davies and Jackson 2006 and recommendations of a panel of experts)					
		1	2	3	4	5	6
Connectivity	Metrics of exchange or migrations of biota between adjacent water bodies or habitats; may be strongly affected by factors adjacent to or larger than the immediate study area. Proxies may be used as measures, including habitat landscape metrics, anadromous fish data, or hydrological measures	System is highly connected in space and time*; exchanges, migrations, and recruitment from adjacent water bodies or habitats are as naturally occurs. *Note that some systems are naturally closed off, and this is the level 1 state.	Ecosystem connectivity is unimpaired	Slight loss in connectivity in space or time, with slight decreases in exchanges, migrations, or recruitment from adjacent water bodies or habitats	Some loss in connectivity with adjacent water bodies or habitats, but alternative pathways prevent complete disconnects or other failures	Significant loss in ecosystem connectivity with adjacent water bodies or habitats is evident; alternative pathways do not exist for some taxa; some near-complete disconnects exist; significant reductions in naturally occurring biotopes	Complete loss in ecosystem connectivity in at least one dimension (either spatially or temporally) lowers reproductive or recruitment success or prevents migration or exchanges with adjacent water bodies or habitats; disconnects or other failures are frequent; most naturally occurring biotopes are eliminated
Non-Native Taxa	Metrics of non-native species, including intentionally introduced species. May include measures of the impact of introduced and non-native species. Examples include estimated numbers of species or individuals, biomass measures of natives and non-natives, or replacements of native species	Non-native taxa are absent, or if minimally present do not affect native biota or natural processes	Non-native taxa are present, but occurrence has a non-detrimental effect on native taxa or natural processes	Non-native taxa may be prominent in some assemblages (e.g., crustaceans, algae, bivalves, fishes); native taxa may be reduced	Some replacement of sensitive native taxa with functionally diverse assemblages of non-natives	Some assemblages (e.g., crustaceans, algae, bivalves, fishes, epifauna) are dominated by tolerant non-native taxa	Non-native taxa are often dominant and may be the only representative of some assemblages (e.g., crustaceans, algae, bivalves, fishes, epifauna)

While conceptually relying on basic BCG principles, the estuarine framework creates a flexible approach that can be applied to different individual estuaries. The framework promotes a system-level examination of the estuary and modifies attributes from freshwater descriptions to apply at different scales of assessment. For example, it expands the organism condition attribute to include habitat condition. It also begins to develop the BCG attributes for ecosystem function and connectance, which were not quantitatively defined for application to streams. The framework is designed so that the estuary can be conceptualized as a functioning system. Thus, the promise of the BCG framework is that once it is calibrated for a specific estuary, it can be used to assess overall estuarine condition in the past and present, and it can be used to develop visions for desired future conditions.

To address these challenges, since the 2008 Narragansett workshop, an estuarine BCG work group composed of scientists from EPA and the State of Rhode Island proposed a series of "action steps" that guide coastal scientists and managers through the process of developing a BCG (Cicchetti et al. 2016). Each step delivers a product or set of products of use to managers, but each step can be applied, or not applied, as best meets the goals of individual programs. The first action steps of the framework do not involve an actual BCG per se, but apply accepted management decision tools to lead management groups through evaluation of environmental problems. The next steps integrate the actual BCG into solving these problems by developing reference conditions, narratives for BCG levels, preliminary biological assessments, and broad goals. In the final stages a rigorous and quantitative BCG is developed through expert consensus, and it can support development of quantitative biological thresholds, potentially other regulatory and stressor-response thresholds, a variety of non-regulatory actions, and monitoring for effectiveness of management actions. This flexible framework allows scientists and managers to develop these steps using any method or sequence that would best address specific needs. Stages and action steps of estuarine BCG development are:

Management steps—clarifying needs and directions of work using larger frameworks such as Structured Decision Making or Drivers-Pressures-State-Impact-Response:

1. Identify management clients and stakeholders
2. Collaborate to define management goals, visions, and objectives
3. Determine the biological attributes, measures, and stressors most relevant to management objectives

BCG development steps for non-regulatory management—setting targets, communicating, motivating:

4. Delineate and classify the water body and watershed of interest
5. Organize and analyze existing data for the identified measures; collect new data, if needed
6. Define a "minimally disturbed" reference condition for the measures
7. Develop narrative descriptions of the biology expected at each BCG level; assist management partners

Additional BCG development steps for regulatory management—determining impairment, setting thresholds for actions, linking measures to stressors, monitoring for change:

8. Convert narrative descriptions to quantitative measures and thresholds for BCG levels
9. Develop a stressor gradient and stressor-response relationships
10. Organize, interpret, and report results
11. Develop decision-support, communication, and monitoring tools; assist management partners

In past experience presenting estuarine BCG concepts to national experts, scientists tended to focus their attention on step 6 above. The scientists' focus on the integrity of the data and approach to honing the definition of "minimally disturbed" as a reference condition for estuaries was extremely valuable for making progress toward an estuarine pilot.

B2.3 Establishing Reference Conditions in Historically Disturbed Environments

BCG level 1 has been interpreted as "as naturally occurs" conditions in absence of anthropogenic disturbance. Participants in the two estuarine BCG development workshops (Cicchetti and Pryor 2010; Cicchetti 2010) found that defining "natural" condition for estuaries was a challenge because very few (if any) undisturbed, or pristine, sites exist in coastal ecosystems today (Bald et al. 2005; Muxika et al. 2007). A more practical reference level that might be used in a BCG is a "minimally disturbed" condition that represents an ecological state "in the absence of significant human disturbance" (Stoddard et al. 2006) and has been considered as comparable to a BCG level 2 by stream biologists (Davies and Jackson 2006). These types of sites may still be difficult to locate in a modern estuary. For this reason, it may be desirable to use historical data to describe a minimally disturbed reference level. Historical baselines are not without their complications either, as (1) human impact pre-dates modern science in essentially all U.S. and European watersheds, and thus quantitative data are limited (Borja et al. 2011); (2) they are difficult to calibrate with current ecosystem status; (3) ecosystems were as dynamic in the past as they are today; and (4) climate change and the degree of anthropogenic influence can render these baselines unattainable (Samhouri et al. 2011). Careful definition and anchoring of reference conditions is needed to avoid shifting baselines (Pauly 1995) where societal and scientific perceptions of what is "good" or "normal" are based on the expectations developed during a human lifetime. As a result, the "best" conditions that remain in an area or region can be misinterpreted as "minimally disturbed" two or three decades later (Papworth et al. 2009).

Where historical quantitative stressor/response data are available, management and/or restoration efforts have been quite successful in utilizing a biological condition-type approach. While not explicitly a "BCG", scientists and managers in the Chesapeake Bay, Buzzards Bay, Tampa Bay estuary, and Puget Sound have taken similar historical biological assessment approaches that are certainly conducive to organization in the estuarine BCG framework. The Chesapeake Bay Program (CBP) is a well-known pioneer in setting restoration targets. CBP set numerous conservation and restoration goals in its 2000 report for ecosystem components ranging from oysters to nutrients and sediments, many of which are based on historical baselines (CBP 2000). In Buzzards Bay, the Buzzards Bay Coalition works with scientists and land use experts to examine the best available current and historical information for indicators in three categories: pollution, watershed health, and living resources (BBC 2011). The Coalition's work has suggested that Buzzards Bay is currently functioning at half its ecological capacity, therefore affecting the local economy and quality of life (BBC 2011). The Tampa Bay Estuary Program (TBEP) used historical seagrass cover, light attenuation, and chlorophyll-a data to set restoration targets for seagrass recovery (Greening and Janicki 2006). Following the establishment of restoration targets, average gains of seagrass of 142–202 acres per year occurred between 1988 and 1996 (Greening and Janicki 2006). TBEP also used a historical reference-based approach to set acreage targets for other intertidal habitats, again resulting in significant gains (Cicchetti and Greening 2011). On the west coast, the Puget Sound Partnership has used several types of reference levels, including historical baselines from the 19th century, to set restoration targets for eelgrass, wetlands, bald eagles, and resident southern killer whales (Samhouri et al. 2011). Both TBEP and the Puget Sound Partnership use a type of indicator "report card" to track changes in indicators with respect to each reference level and a response gradient (i.e., the grades on a report card).

The first formal case study of the estuarine BCG (Shumchenia et al. 2015) compiled a biological stressor-response gradient in time rather than space for Greenwich Bay, a sub-embayment of Narragansett Bay (Figure B2-1). In order to implement this concept, a long record of ecological data was needed. Anecdotal observations of the ecosystem were available as far back as the 1600s for this embayment.

After assembling ecological histories for seagrass extent, benthic communities, and primary productivity/shellfish, a "minimally disturbed" *range of conditions* was anchored by observations prior to 1850. Like the broader Narragansett Bay and many estuaries in the U.S., the relative importance of environmental stressors changed over time, but even qualitative descriptions of the biological indicators' status provided useful information for defining condition levels. This BCG demonstrated that stressors rarely acted alone and that declines in one biological indicator influenced the declines of others. For example, in Greenwich Bay the loss of eelgrass was linked to the loss of scallops. Documenting the timeline of changing stressors helps demonstrate that management actions of the past may no longer be appropriate or effective for managing the current stressor landscape.

Assembling this pilot example of the estuarine BCG for a small but data-rich embayment was the first step toward testing the framework at the estuary- and watershed-scale. To demonstrate the value of the estuarine BCG framework in implementing ecosystem-based management beyond local water quality and habitat management issues, a broader scale application was necessary.

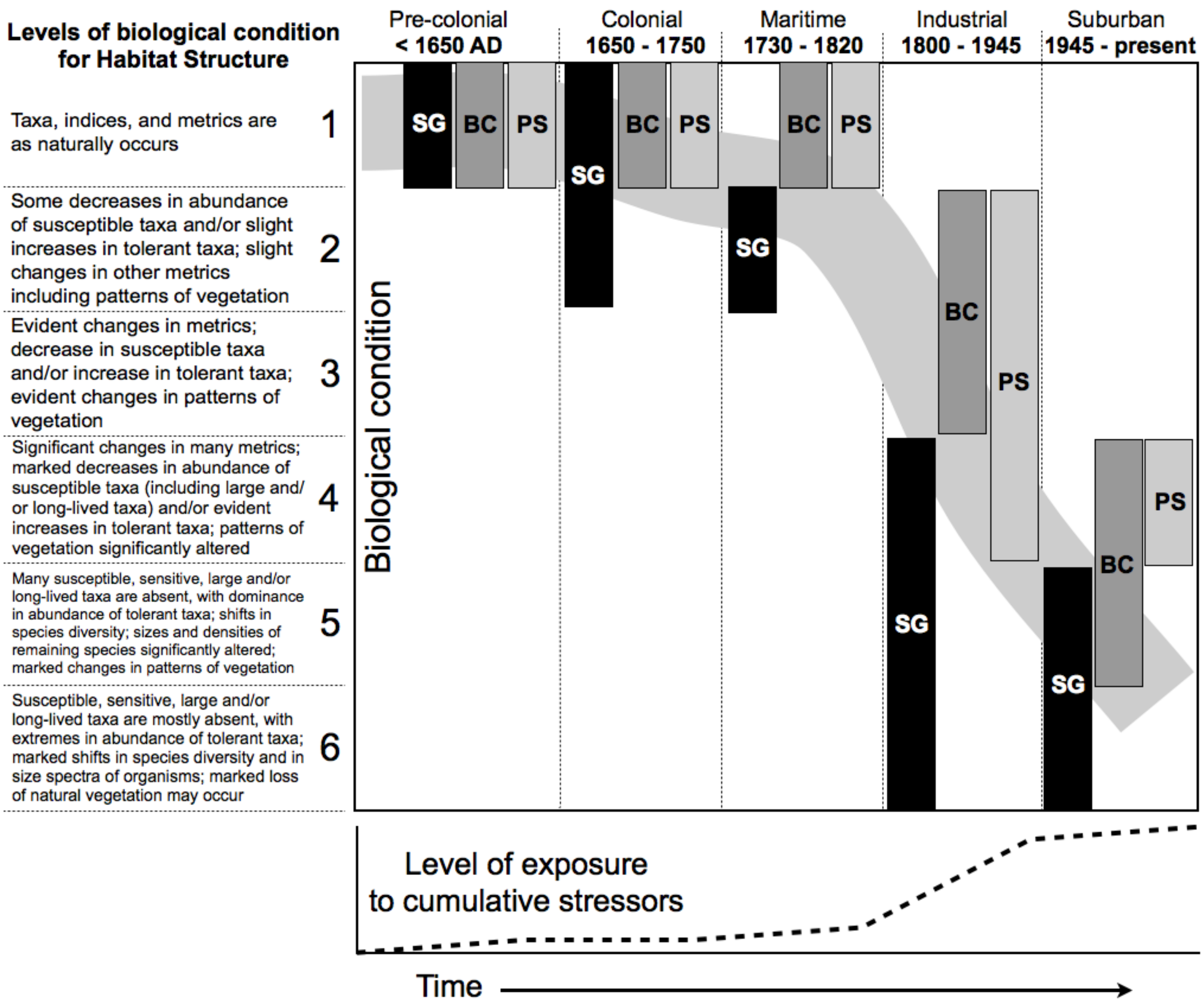


Figure B2-1. A chart developed for an estuarine BCG case study in Greenwich Bay, showing a generalized stressor gradient through time from the 1600s to present (x-axis), biological condition levels (y-axis), and the responses of three structural attributes: SG = seagrass; BC = benthic community; and PS = primary productivity and shellfish.

B2.4 Measuring Overall Estuary Condition

Estuaries and near-coastal systems are influenced not only by stressors and processes within the system, but also by watershed and oceanic pressures. The overall condition of the estuary derives from the conditions of all of these internal and external components, their connections, and their combined functions. Assembling a conceptual ecosystem model incorporating the physical, chemical, and biological processes that structure the particular system of interest helps assess overall estuary condition, but it is a complex undertaking.

Several methods and proxies have therefore been developed to evaluate overall condition of the estuary or coast, all of which could be incorporated into a BCG approach via comparisons to naturally occurring or minimally disturbed conditions. Robustness of the assessment will improve as spatial and temporal coverage increases and as more assemblages and habitats are considered. Ideally, the biological measures chosen should cover the entire estuarine gradient and incorporate multiple components of the estuary (e.g., intertidal and subtidal; primary and secondary production; benthos and nekton). Approaches that assess overall condition, which can be combined, include (see Cicchetti et al. 2016 for details):

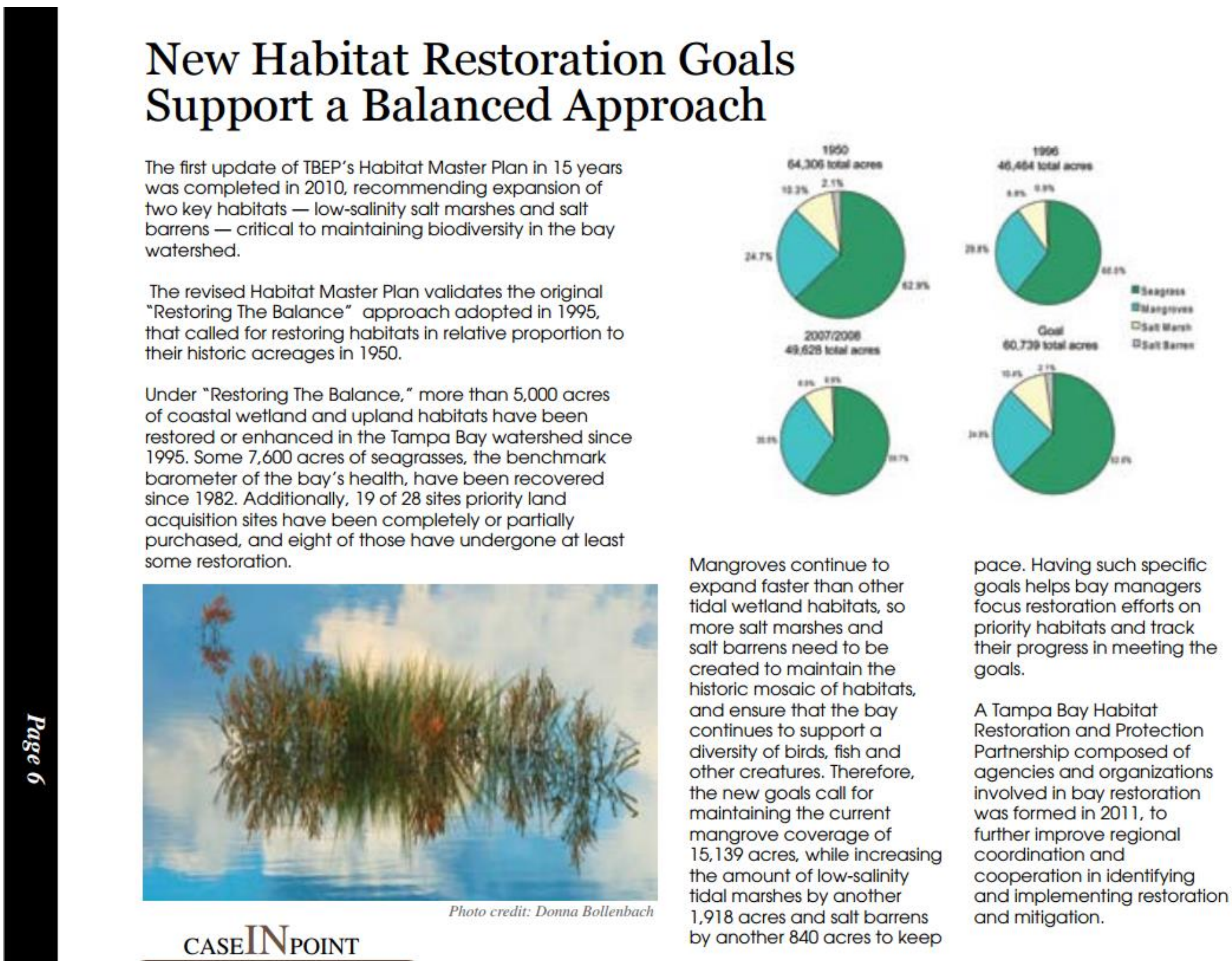
1. Use of structural measures including presence of keystone species or other indicator species; numbers of species, groups of species, communities, or habitats; or the extent, composition, or arrangement of living habitats, or biotopes
2. Use of measures of ecosystem function and connectivity, especially those that derive from complex interactions in the entire estuary such as energy flows, trophic webs and linkages, carbon or nutrient fluxes, production of diverse biomass, nutrient processing, or resilience to changes
3. Use of both biological indicators and stressor values, such as in the Greenwich Bay case study (Shumchenia et al. 2015). Information from four biological indicators and several attributes was synthesized with information from a generalized stressor gradient and additional information on specific stressors to describe the current state of the estuary together with the significant events and processes that have shaped it over time

A variety of methods and proxies can be effective, and they are generally selected to best address the unique features, needs, and available data that characterize an individual water body. For our Narragansett Bay case study, the relatively large amount of spatial and temporal data addressing benthic habitats, as well as documented linkages between habitat quality and water quality, were the impetus for choosing a habitat mosaic approach.

B2.5 Habitat and Biotope Mosaics

Productive estuaries in a natural state are composed of a mosaic of living habitats (Henningsen 2005) or *biotopes*, including seagrass beds, oyster reefs, mussel reefs, salt marshes, mangrove forests, clam flats, and specific soft-bottom benthic communities. The "Habitat Mosaic" approach recognizes that anthropogenic stress to an estuary leads to destruction of these living habitats and replacement with other habitats. The method considers the distribution of these living habitats to be a central part of estuarine biology. A further assumption is that a mosaic of biotopes that most resembles the mosaic that would naturally occur in an estuary will improve biological integrity and provide greatest benefit for the native communities of organisms that have evolved in that setting over millennia. An assessment using this approach compares acreage data from a time period of interest to acreage data from one or more time periods in the past, and several quantitative measures are available.

A contribution from the TBEP was to apply this biotope mosaic concept to the development of biological assessment based on estuary-wide changes to quantity (acres) and distributions (relative proportions) of habitats over time. Ecological priorities for Tampa Bay were to "Restore the Historic Balance" of critical habitats in percent compositions of biotope mixes relative to an undisturbed historic benchmark, as well as to restore total acres of all living habitats, to the extent possible (Figure B2-2). Tampa Bay stakeholders and the public were invested in the quantity and diversity of valued habitats, and the concept of "Restore the Historic Balance" resonated with this community. The appealing visual aspects of this method proved effective at communicating estuarine condition and developing stakeholder visions and goals, which led to management actions and environmental results. This method can be used together with other approaches as an important component in the management of estuaries, linking environmental goals to biotope acres and biotope metrics under the BCG framework.



New Habitat Restoration Goals Support a Balanced Approach

The first update of TBEP's Habitat Master Plan in 15 years was completed in 2010, recommending expansion of two key habitats — low-salinity salt marshes and salt barrens — critical to maintaining biodiversity in the bay watershed.

The revised Habitat Master Plan validates the original "Restoring The Balance" approach adopted in 1995, that called for restoring habitats in relative proportion to their historic acreages in 1950.

Under "Restoring The Balance," more than 5,000 acres of coastal wetland and upland habitats have been restored or enhanced in the Tampa Bay watershed since 1995. Some 7,600 acres of seagrasses, the benchmark barometer of the bay's health, have been recovered since 1982. Additionally, 19 of 28 sites priority land acquisition sites have been completely or partially purchased, and eight of those have undergone at least some restoration.

Photo credit: Donna Bollenbach

Mangroves continue to expand faster than other tidal wetland habitats, so more salt marshes and salt barrens need to be created to maintain the historic mosaic of habitats, and ensure that the bay continues to support a diversity of birds, fish and other creatures. Therefore, the new goals call for maintaining the current mangrove coverage of 15,139 acres, while increasing the amount of low-salinity tidal marshes by another 1,918 acres and salt barrens by another 840 acres to keep pace. Having such specific goals helps bay managers focus restoration efforts on priority habitats and track their progress in meeting the goals.

A Tampa Bay Habitat Restoration and Protection Partnership composed of agencies and organizations involved in bay restoration was formed in 2011, to further improve regional coordination and cooperation in identifying and implementing restoration and mitigation.

Page 6

CASE IN POINT

Figure B2-2. TBEP graphic describing "Restore the Balance" (TBEP 2012).

B2.6 Using Biological Condition Gradient Concepts to Re-Assess Narragansett Bay Benthic Habitat Quality

This section is from "A re-assessment of Narragansett Bay benthic habitat quality between 1988 and 2008" by Emily J. Shumchenia, Marisa L. Guarinello, and John W. King; submitted to Estuaries and Coasts.[14]

There are currently efforts to measure whole estuary condition in Narragansett Bay over the past several decades. The structural measures of benthic biotope extent, composition and arrangement, are used together with information about two apparently dominant stressors over this time period: anthropogenic nutrient inputs and warming. Although an explicit estuarine BCG has not yet been constructed using these data, the framework that is used to present and interpret these data uses BCG concepts. Furthermore, this scientific analysis is viewed as an important first step toward constructing a BCG explicitly. The acceptance of the interpretation of BCG data by the scientific community is an important pre-cursor to estuarine BCG construction, and ensures it will be credible in presentations to local management and regulatory agencies.

B2.6.1 Introduction to Narragansett Bay

Narragansett Bay is the second-largest estuary on the east coast of the U.S. (328 km^2) and has the most densely populated watershed, shared between the states of Rhode Island and Massachusetts. Narragansett Bay is best conceptualized as a bay with several relatively distinct regions and a general north to south gradient of enrichment patterns from the heavily populated and narrow Providence River Reach, to the more ocean-influenced Open Bay consisting of the East and West Passages on either side of Conanicut Island (Valente et al. 1992; Nixon et al. 2009; Raposa 2009). In addition, a few Shallow Embayments (e.g., Greenwich Bay and associated coves) form distinct regions of the Bay. Overall, Narragansett Bay is known as a phytoplankton-based temperate ecosystem with a mean depth of 8.6 m and a mean flushing rate of 26 days (Pilson 1985; Nixon et al. 1995). Freshwater input is relatively low (100 $m^3 s^{-1}$), with the result that the mid-bay is generally well mixed (Nixon et al. 2005). Salinity follows a down-bay gradient from 20 psu at the head to 32 psu at the mouth of Narragansett Bay. The annual temperature varies from about 0 to 24°C. Sediments of Narragansett Bay are mainly clayey silt and sand-silt-clay (McMaster 1960).

Narragansett Bay lies near the boundary between the Gulf of Maine/Bay of Fundy (also known as Acadian) ecoregion to the north and the Virginian ecoregion to the south (Spaulding et al. 2007). This boundary is defined by the departure of the Gulf Stream from the coast across the northwestern Atlantic, with generally more cold-water species north of Cape Cod. Recent shifts in marine species distribution and abundance near this boundary are driven in part by climate change (Oviatt 2004; Collie et al. 2008; Pinsky et al. 2013). Therefore, Narragansett Bay, which, like most estuaries is likely experiencing climate-change-related ecological changes (Nixon et al. 2009), provides an excellent case study for other estuaries that may experience climate-driven ecological shifts and oligiotrophication in the future.

[14] The text of this section has been altered to include heading, table, and figure numbering consistent with the rest of the chapter.

B2.6.2 Why Benthic Biotopes?

Soft sediment benthic biotopes, i.e., abiotic environments and associated assemblage of species (Connor et al. 2004; Costello 2009; Davies et al. 2004), are particularly useful for monitoring patterns of organic enrichment in time and space (Pearson and Rosenberg 1978) because they are effective integrators of cumulative stressors such as eutrophication and hypoxia (Pearson and Rosenberg 1978; Valente et al. 1992; Germano et al. 2011). The structure of surface sediments and the composition, or successional stage, of benthic communities are linked to the degree of organic loading to a water body (Rosenberg 2001) and readily indicate recent (weeks to months) water quality conditions (Cicchetti et al. 2006; Shumchenia and King 2010a). Comprehensive characterizations of benthic biotopes at the whole estuary scale are rare because of the high level of detail required to populate biotope classifications. Benthic biotopes that have been defined for a sub-embayment of Narragansett Bay include '*Ampelisca* on shallow mud' and '*Spiochaetopterus* on deep coarse sand' (Shumchenia and King 2010b). These coastal marine biotopes comprise mosaics at the landscape scale (Boström et al. 2011) and are ecologically meaningful units for conservation and management purposes (Salomidi et al. 2012). Biotope mosaics are interrelated and functionally connected such that a change to one biotope may affect others, as well as the entire ecosystem (Boström et al. 2011). To date, studies to characterize patterns and quantitative change over time in benthic biotope mosaics have been mostly limited to seagrass, mangrove and saltmarsh ecosystems due to the utility of aerial photography in these environments (Boström et al. 2011; Pittman et al. 2011; Cicchetti and Greening 2011; Zajac 2008). The composition of a biotope mosaic and how it changes over time may indicate degradation or recovery of an ecosystem (Dunning et al. 1992; Wiens et al. 1993; Pittman et al. 2007), and thus monitoring of biotope mosaics can help assess the effects of human alterations and multiple stressors on coastal marine ecosystems (Cicchetti and Greening 2011). However, benthic biotope characterization for whole-estuary assessments by traditional sampling methods is incredibly labor-intensive due to the collection, sorting, and identification of benthic samples.

One method that has increased the efficiency of benthic assessments is sediment profile imagery (SPI). SPI is a rapid reconnaissance technique that delivers clear images of benthic biotopes regardless of water column turbidity (Germano et al. 2011). Ideally, SPI images capture an area including the sediment-water interface and up to 20 cm below, i.e., the most biologically active zone of the sediment column.

In 1988, SPI was used in the first comprehensive survey of benthic habitat quality in Narragansett Bay (Figure B2-3) in the context of organic enrichment from wastewater treatment facility (WWTF) discharges (Germano and Rhoads 1988; Valente et al. 1992). The 1988 SPI study provided the first in-situ snapshot of benthic processes in Narragansett Bay soft sediments. In fact, most researchers were unaware and "surprised" by the proportion of the bottom that had been exposed to high levels of organic deposition and low concentrations of dissolved oxygen (DO) (Granger et al. 2000). Many of the sites identified as having excessive organic enrichment and degraded benthic habitat were in the Providence River Reach or Shallow Embayment sub-regions of the Bay; sites near WWTF outfalls, in coves, or other spatially constricted areas that received effluent (refer to Figure B2-3; Valente et al. 1992).

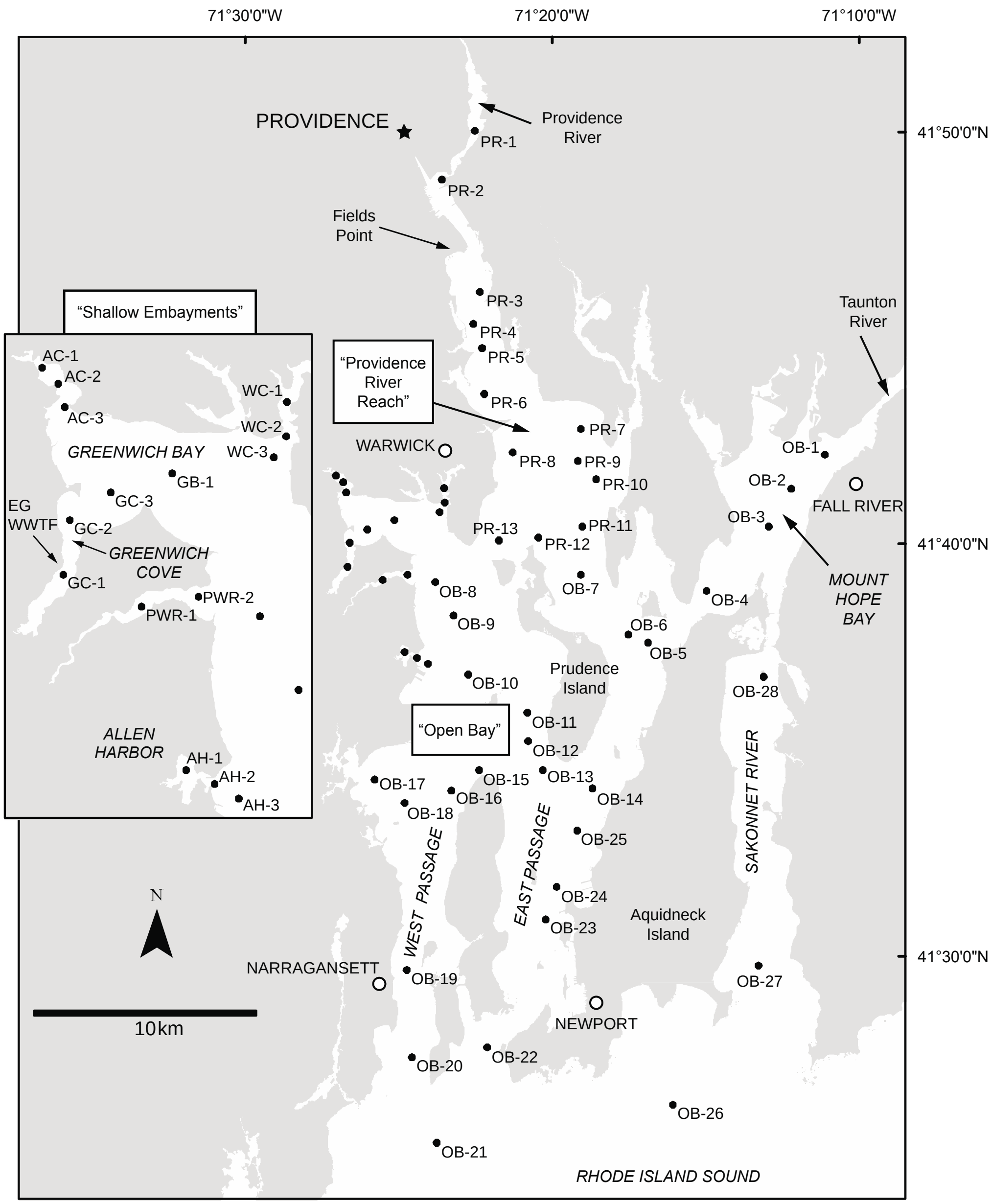


Figure B2-3. Locations within Narragansett Bay, Rhode Island USA, where sediment profile images were taken in 1988 and 2008. Three sub-regions used in the analyses are highlighted: Providence River Reach stations (PR-); Open Bay (OB-); Shallow Embayments (inset). Note locations of Fields Point and East Greenwich (EG) Waste WWTF at labels

B2.6.3 Changes in the Narragansett Bay Stress Gradient

Since the 1988 study, there has been a great deal of human intervention and human-mediated change in Narragansett Bay and its watershed. These human actions translate directly to changes in the stress gradient experienced by habitats and biota.

Human population in the watershed has increased by about 200,000 since the early 1980s to a total of about 2 million people (Nixon et al. 2008), increasing both impervious surfaces and WWTF loads. Between 1980 and 1995, the Field's Point WWTF in Providence (responsible for approximately 55% of total effluent discharged directly to the Bay) transitioned from being considered by the U.S. Environmental Protection Agency one of the worst in the country to one of the best (Nixon and Fulweiler 2012). The Field's Point plant initiated secondary treatment of its sewage in June of 1988, just months before the benthic habitat quality assessment took place. In the 1990s, seasonal nutrient- and stratification-driven hypoxia was discovered in upper Narragansett Bay, and has since been monitored by state and academic programs (Deacutis 2008; Codiga et al. 2009). In 2003, a large fish kill occurred in Greenwich Bay due to a confluence of unique hydrological conditions and organic loading that caused severe hypoxia and anoxia (RIDEM 2003). The fish kill resulted in media and political attention, and in 2004, the Rhode Island Department of Environmental Management (RIDEM) issued a statutory mandate to eleven WWTFs within the upper Narragansett Bay watershed to reduce summer season nitrogen discharges to the Bay between 48%–65% with respect to 1995–1996 levels (RIDEM 2005). As of January 2008, the Field's Point WWTF had not yet implemented nitrogen removal procedures and five smaller WWTFs had implemented biological nitrogen removal upgrades (RIDEM 2008).

Most monitoring efforts since the 2003 fish kill and 2004 nutrient reduction mandate have focused on DO data to evaluate compliance with water quality standards (RIDEM 2005) and highlight the summer recurrence of bottom water hypoxia in upper- and mid-Narragansett Bay (Bergondo et al. 2005; Deacutis et al. 2006; Melrose et al. 2007; Codiga et al. 2009). Despite recommendations for monitoring at four "critical boundaries" to detect long-term changes to benthic enrichment of the Bay bottom (Germano and Rhoads 1988), only limited-term temporal studies of the benthos have been conducted since (e.g., Cicchetti et al. 2006; Calabretta and Oviatt 2008; Shumchenia and King 2010a). However, analyses of the available data do suggest a shift in benthic community composition due to organic enrichment between the 1950s and 1980s (Frithsen 1990). SPI surveys conducted for purposes other than benthic habitat quality assessment and/or at different sites and times of year than the 1988 study suggest a potential decline in benthic habitat conditions in the Providence River Reach between 1975 and 1988 (Myers and Phelps 1978) and another potential slight decline in the upper Bay between 1988 and 1994 (Diaz 1995).

Exactly 20 years after the 1988 study (i.e., August 2008), and using the same SPI techniques, the same sites were revisited to reassess benthic biotope status in the context of environmental changes in the intervening years, including recent improvements in wastewater treatment aimed at curtailing organic loading. Using the same image analysis approach (Germano et al. 2011) on both 1988 and 2008 data sets, the abiotic and biotic features of the surface and near-surface environment were classified into benthic biotopes to compare biological and physical processes between surveys. To assess benthic condition throughout the estuary, SPI results were analyzed using a biotope mosaic assessment approach (Cicchetti and Greening 2011). The spatial distribution, composition and diversity of benthic biotopes throughout Narragansett Bay in 1988 and 2008 were compared and related to any observed trends in biotope condition to changes in organic loading. Whether or not oligotrophication of Narragansett Bay will result from climate changes and anthropogenic nutrient reductions already set in motion, consistent, comparable, and frequent monitoring of the benthos will be necessary for scientists and managers to be able to track, investigate, and respond to these stressors.

B2.6.4 Methods

In planning the 2008 data collection and analysis, similarities with the 1988 study were maximized. The 2008 survey was conducted during the same time of year as the 1988 study (mid-August), over roughly the same number of consecutive days (four in 2008; five in 1988) and during the same tidal stage (neap). Planning in this way helped to ensure an improved ability to compare images between years versus previous SPI comparisons in Narragansett Bay (Germano and Rhoads 1988; Diaz 1995). Despite these precautions, some limitations need to be considered. These limitations were apparent in both the data collection and analysis phases of the study and are described more fully below.

Station locations were originally chosen to define Bay-wide trends in benthic habitat quality. The majority of stations were located in depths ranging from 5 to 20 m and in unconsolidated sediments.

At each station an Ocean Imaging Systems Inc. (Falmouth, MA) digital sediment profile camera was deployed for three to five replicate drops. The coordinates of the first camera drop at each station were compared with the 1988 target location.

Analysis of 2008 Images and Re-analysis of 1988 Images

Original, analog printed black and white images from the 1988 survey were obtained from archives at the Graduate School of Oceanography, University of Rhode Island, and scanned into digital format at a resolution of 300 pixels per inch. The 2008 digital images were uploaded to the analysis computer in .jpg format at 335 pixels per inch. Because of differences in image quality between the two surveys (i.e., analog vs. digital, black and white vs. color, scanned vs. raw digital), there were limitations to the types of quantitative measurements that could be compared between the sets of images. For example, the measurement of the apparent redox potential discontinuity (aRPD) depth is a key indicator of benthic habitat quality and a significantly contributing variable of the multi-parameter Organism-Sediment Index (OSI) that was used to summarize conditions at each station in 1988 (Valente et al. 1992). The depth of the aRPD is often "over-estimated" in black and white images, and thus comparisons between black and white and color images would be particularly susceptible to error (Diaz 1995). Therefore, OSI and aRPD values between 1988 and 2008 could not be compared. The original 1988 images were re-analyzed using the same method as the 2008 images. This analysis approach relied on a biotope classification derived from the abiotic and biotic surface and subsurface features visible in the SPI images.

All SPI photos were imported into Adobe Photoshop CS6 and image brightness and contrast were adjusted manually to increase the detectability of habitat features such as tubes and burrows. For each station in each survey, each available replicate was examined and the image with the least disturbance and deepest prism penetration was selected for analysis. Information on sediment grain size, surface features, and subsurface features was recorded following the protocol described in Rhoads and Germano (1982; 1986).

Sediment Grain Size

Sediment descriptors represented the major modal class for each image. Organic-rich mud, mud, sandy mud, sand and gravel were distinguishable classes. Coarse-grained sediments were indicated by shallow prism penetration and suggested physically dominated habitats. Fine-grained sediments were indicated by deeper prism penetration and suggested that the seafloor environment was depositional and less-frequently physically disturbed. Organic-rich muds were characterized by fine-grained sediments and very deep prism penetration, little to no visible surface oxidation, and minimal surface disturbance or roughness.

Surface Features

Surface descriptors included both biogenic and physical features, such as amphipod and worm tubes, epifauna (e.g., snails, crabs), shells, macroalgae, bacterial mats (e.g., *Beggiatoa sp*.), feeding pits/mounds, bedforms, and roughness. The presence of these features was noted to indicate any recent disturbance and the degree and nature of biological activity at each station.

Subsurface Features

The presence of burrows, infaunal feeding voids, infauna and gas voids were recorded in each image. Subsurface features such as burrows, feeding voids, and infauna indicated biologically active environments, whereas the presence of gas-filled voids at depth indicated high rates of methanogenesis and anaerobic respiration (Rhoads and Germano 1986) associated with high rates of organic matter decomposition.

Benthic Biotope Classification

From the abiotic and biotic surface and subsurface descriptors, each image from 1988 and 2008 was assigned to a benthic habitat/biotope based on a SPI classification scheme previously described for Narragansett Bay (Diaz 1995). Using guidance from the Coastal and Marine Ecological Classification Standard (FGDC 2012) to classify an image, the sediment, surface and subsurface descriptors were summarized into a short phrase such as '*Ampelisca spp.* beds of low to high density, with other small tube-building and shallow burrowing fauna on organic-rich mud.' These descriptions were kept consistent among images and grouped by dominant biota and/or sediment type into eight biotopes (Table B2-2).

Table B2-2. Descriptions of the eight biotopes based on observed dominant biota and sediment type to which sediment profile images were assigned.

Biotope	Code
Ampelisca spp. beds of low to high density, occasionally with other small tube-building and shallow-burrowing fauna on substrates ranging from organic-rich muds to sand. *Beggiatoa spp*. or *Mulinia lateralis* may be present.	AM
Organic-rich muds with various mixes of tolerant species such as *Beggiatoa spp*., tube-building polychaetes, and shallow burrowing fauna.	UN.SF
Burrowing fauna on mud with shell hash. Tube-building polychaetes or deep burrowing fauna may be present.	UN.SH
Burrowing fauna on mud. Tube-building polychaetes, larger tube-builders such as *Chaetopterus*, or deep burrowing fauna may be present.	UN.SI
Burrowing and tube-building fauna on sandy mud.	UN.SS
Crepidula bed on mud. Mobile crustaceans, gastropods or *Beggiatoa spp*. may be present.	SH.SI
Very coarse sands with shell hash. Rafting macroalgae or *Crepidula* beds may be present.	SH.SA
Hard sands with epibenthic sponges, rafting or attached macroalgae, and/or mobile gastropods.	SA

Comparison of 1988 and 2008 Surveys Using Biotope Mosaic Approach

To assess the influence of survey repositioning on interpreted biotope change, the relationship between biotope change and the 2008 field distance from the 1988 target location was examined. Sites that were classified as 'changed' were a mean 60.7 m from the target location (n=25) and sites classified as 'no change' were a mean 56.5 m from the target location (n=15), indicating that repositioning likely had little to do with change detection.

Once images were assigned to one of eight biotopes, biotope diversity Bay-wide and within each Bay region (Providence River Reach, Shallow Embayments and Open Bay) was assessed for each survey. The ratios of biotopes Bay-wide and among Bay regions were measured to assess the composition and spatial structure of the benthic biotope mosaic. These ratios were calculated using only the subset of stations that could be directly compared between years. If a biotope changed between 1988 and 2008, the type of biotope change was noted.

B2.6.5 Results

Fifty-two of the original 56 stations were resampled in 2008. Between the two surveys, 38 total stations could be directly compared at the biotope level. Change between 1988 and 2008 was examined using a biotope mosaic approach, where the proportions and spatial arrangement of biotopes Bay-wide and among three Bay regions was characterized.

The composition and spatial arrangement of benthic biotopes differed markedly between the 1988 and 2008 surveys despite identical levels of benthic biotope diversity. Over half (58%) of the stations visited changed biotope between 1988 and 2008, with the bulk of that change occurring in the Providence River Reach and Shallow Embayments. These spatial patterns of benthic biotope change provided information about where to focus attention for this analysis and suggested focal areas for monitoring future changes.

B2.6.6 Ampelisca Biotopes Mark "Critical Boundaries" in the Stress Gradient

Biotopes dominated by *Ampelisca spp.* tubiculous amphipods increased > 5-fold between 1988 and 2008, and expanded into the more urban, anthropogenically-stressed Providence River Reach (Figure B2-4).

The prominence of ampeliscid amphipods (e.g., *Ampelisca spp*.) within Narragansett Bay benthic biotopes dates back at least to an early detailed benthic study of Greenwich Bay (Stickney and Stringer 1957). It has been suggested that ampeliscids are organic enrichment opportunists (McCall 1977). There is also debate as to whether ampeliscids serve as indicators of impending hypoxia (Levin et al. 2009) or of improving conditions (Diaz et al. 2008; Rhoads and Germano 1986). A recent study in Greenwich Bay (i.e., one of the Shallow Embayments in this study) of benthic response to water quality changes on the order of weeks to months showed that ampeliscids colonized quickly and indicated improving water quality (Shumchenia and King 2010a). Ampeliscid tube structures have been associated with increased biogenic activity and oxygen penetration into the sediment (Diaz et al. 2008) and increased hard clam abundance (MacKenzie et al. 2006), but have also contributed to the exclusion of other tube-dwelling species (Santos and Simon 1980). In Jamaica Bay, New York, amphipod productivity was so high that it was likely more than sufficient to support the entire local winter flounder (*Pleuronectes americanus*) population, with *Ampelisca abdita* making up 88% of the diet of juveniles (Franz and Tanacredi 1992). In Narragansett Bay, winter flounder juveniles have been most abundant in the Providence River Reach and Shallow Embayments recently (Meng et al. 2005) so increases in *Ampelisca* biotopes in these Bay regions may have already benefited this important fish species. Ampeliscids do require large quantities of organic matter to sustain "mat" densities (Franz and Tanacredi 1992; McCall 1977), which signals eutrophic conditions. It was estimated that 500 g carbon m^{-2} yr^{-1} is required to maintain *Ampelisca spp.* tube mats in Boston Harbor, approximately 60 miles to the north (Diaz et al. 2008). Assuming this relationship is relevant to Narragansett Bay, there has likely been enough carbon produced historically via primary productivity on an annual basis in the Providence River (mean of 559 g C m^{-2} yr^{-1}), Greenwich Bay (mean 219–254 g C m^{-2} yr^{-1}), and the Open Bay north of Prudence Island (mean of 517 g C m^{-2} yr^{-1}) to support these communities (Oviatt et al. 2002). Given that primary productivity historically comprised an estimated 80% of the total organic carbon input to the Bay (Nixon et al. 1995), it is likely that there are sufficient amounts of organic matter stored in the sediments on which these ampeliscids thrive.

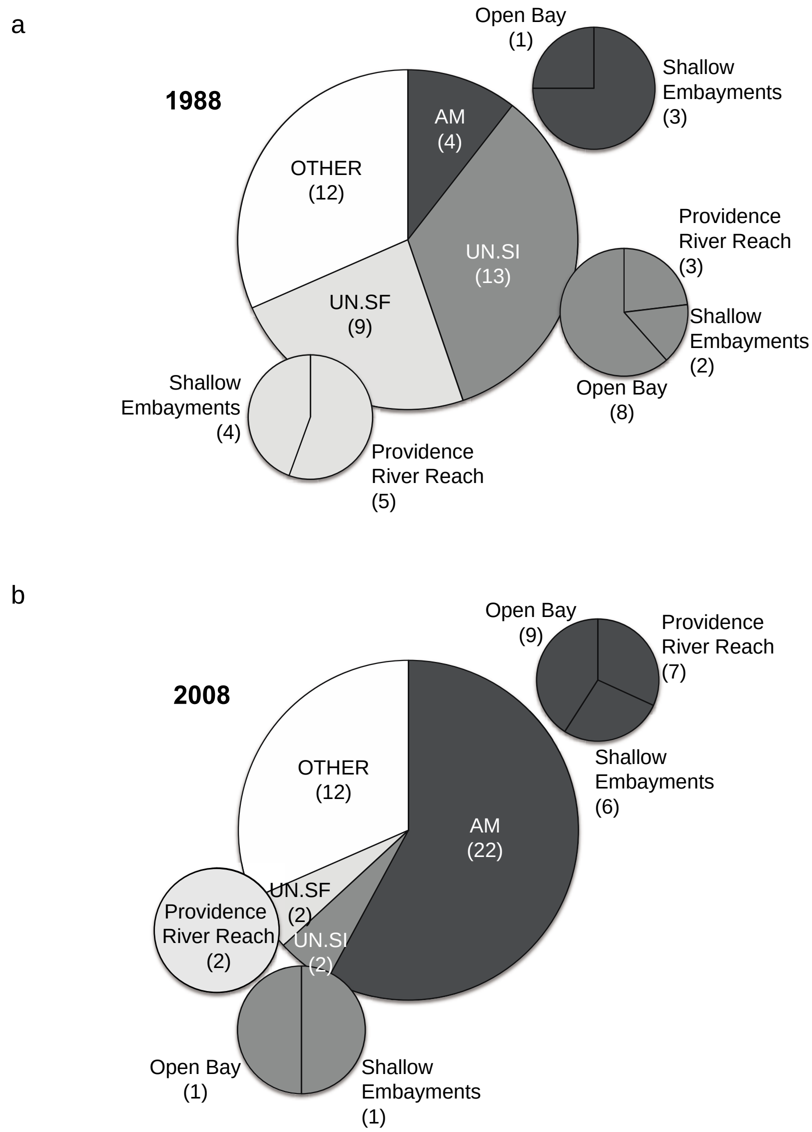


Figure B2-4. Composition and spatial arrangement of the Narragansett Bay benthic biotope mosaic in 1988 (a) and 2008 (b) AM = 'Ampelisca spp. beds'; UN.SI = 'burrowing fauna on mud'; UN.SF = 'organic rich muds with various mixes of tolerant species'.

Dense *Ampelisca spp*. communities in areas with high organic input and good water quality have been previously observed within Narragansett Bay and in Boston Harbor (Stickney and Stringer 1957; Diaz et al. 2008). The cessation of primary sewage discharges to Boston Harbor (Massachusetts, USA) in the early 1990s appears to have "set the stage" for "widespread increases" in *Ampelisca spp*. throughout the harbor. Prior to 1992, organic loading was high but water quality may have been too poor to allow *Ampelisca spp*. to thrive (Diaz et al. 2008). In the late 1990s and early 2000s, several years after the sewage outfall relocation, subsequent declines in *Ampelisca spp*. tubes were associated with the reductions in organic loadings to the harbor and the eventual depletion of sediment organic inventories (i.e., surface sediment total organic carbon) (Diaz et al. 2008). Unlike in Jamaica Bay, reductions in organic matter and lower numbers of *Ampelisca spp*. have apparently had a positive effect on fish species, as the recreational fishing community has made note of significant recent increases in winter flounder populations in Boston Harbor (Powers 2015).

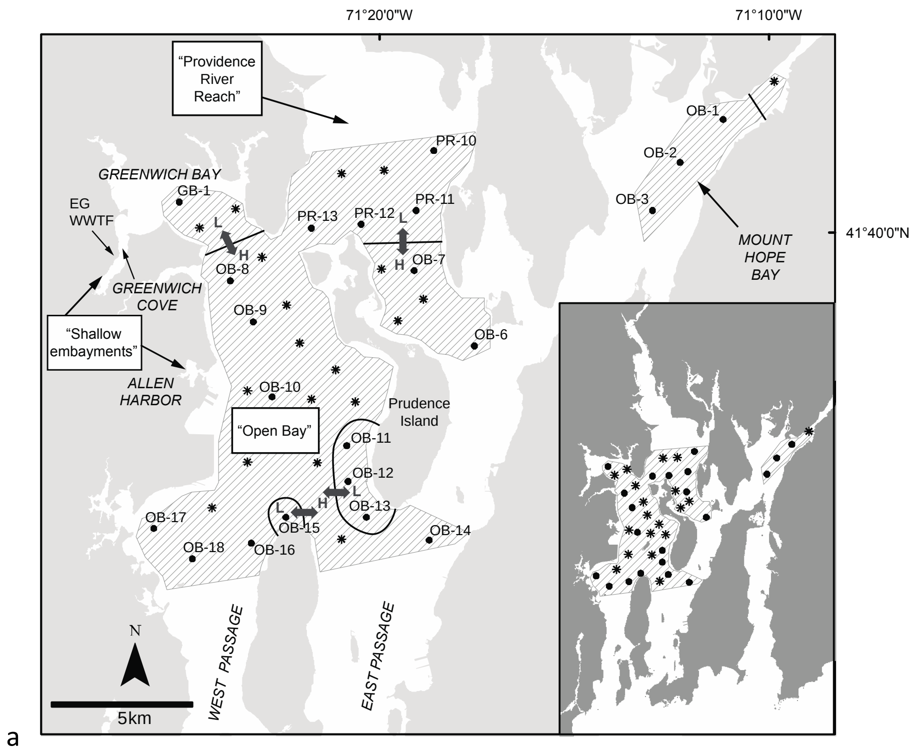
It is possible that a pattern similar to the Boston Harbor example is currently occurring in Narragansett Bay. In 1988 we observed conditions that did not favor widespread *Ampelisca* biotopes: stations with high organic loading and surface sediments that indicated poor water quality conditions (Valente et al. 1992). Between 1988 and 2008, conditions theoretically became increasingly favorable for *Ampelisca* biotopes: management strategies to reduce organic loadings and improve water quality were initiated. In 2008, an increase in the proportion of *Ampelisca* biotopes Bay-wide was observed, and especially in areas where organic loading was known to be previously high. Water quality monitoring programs continue to record hypoxic events,[15] but it is possible that hypoxia occurs now over a smaller area, with less frequency and/or intensity than previous events (the first Bay-wide DO monitoring program did not begin until 1999; Prell et al. 2004). Regardless, because continuous monitoring of benthic biotopes did not occur between 1988 and 2008, it is impossible to determine where Narragansett Bay is "located" along the similar trajectory observed in Boston Harbor. However, if future surveys classify and analyze the proportions of benthic biotopes Bay-wide, Bay-wide benthic biotope quality could be:

1. Staying the same—i.e., *Ampelisca* biotopes are maintaining position via existing sediment organic matter inventories under good or improving water quality
2. Improving—i.e., a decrease in *Ampelisca* biotopes coupled with increases in 'burrowing fauna on mud,, any other benthic biotopes, or even the appearance of new benthic biotopes, as well as a benthic biotope mosaic showing *Ampelisca* biotopes remaining only in regions with formerly high organic loading
3. Declining—i.e., a decrease in *Ampelisca* biotopes coupled with an increase in 'organic rich muds with various mixes of tolerant species' and a benthic biotope mosaic similar to 1988

B2.6.7 Recommendations for Future Benthic Biotope Monitoring in Narragansett Bay

In the initial report of the 1988 SPI survey (Germano and Rhoads 1988), the authors propose a benthic monitoring strategy aimed specifically at tracking the future movement of Bay gradients in organic enrichment and any associated changes to benthic biotope (habitat) quality. They suggest that future monitoring should focus on four critical boundaries separating high quality benthic habitats from organically-enriched areas: the southern edge of the Providence River Reach, the mouth of Greenwich Bay, the mouth of the Taunton River, and the southwest edge of Prudence Island. Rather than monitor the full Bay-wide suite of original stations, the proposed strategy concentrated station locations around the four critical boundaries, revisiting 21 original stations and adding 19 new locations to increase the detectability of gradient movement (Figure B2-5a). The direction of movement of these boundaries in habitat quality would indicate expanding zones of enrichment (boundaries move down-Bay, away from shore) or decreases in enrichment (boundaries move up-Bay, shoreward).

[15] See http://www.dem.ri.gov/bart. Accessed February 2016.



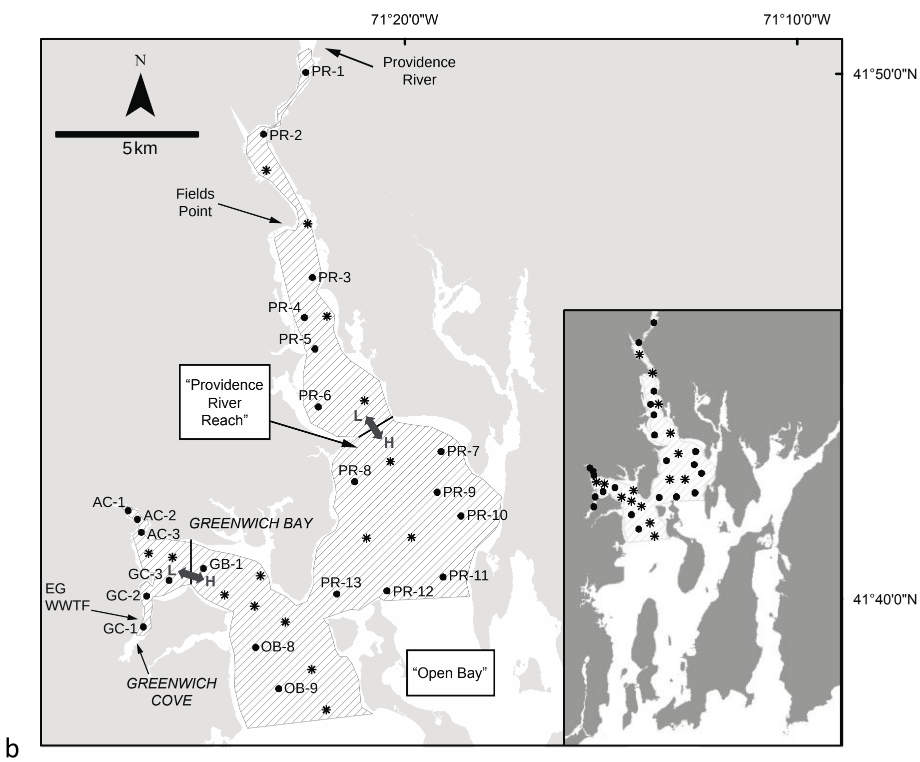


Figure B2-5. Proposed benthic biotope monitoring strategy for detecting changes in organic enrichment and habitat quality from the (a) original 1988 study and (b) modified considering the results of this study. Closed circles = existing stations; star symbols = proposed new stations. Hatched area = monitoring focal area.

By re-sampling the full Bay-wide suite of original 1988 stations, the results of this study have shown that these gradients moved up-Bay and shoreward as of 2008. To account for these changes, an updated strategy proposes moving the critical boundaries further up-Bay and shoreward, and similarly adds stations to a number of original locations to improve the detectability of future changes (Figure B2-5b). In this updated monitoring strategy, 22 original stations would be revisited and 15 new locations would be added to detect any further movement of the habitat quality gradient up-Bay and shoreward. This new monitoring strategy focuses on the Providence River Reach, Greenwich Bay, and the upper portion of the West Passage. A focus on these areas of Narragansett Bay targets biotopes in close proximity to the major WWTFs discharging directly to the Bay. Due to the rapid nature of SPI data acquisition, this survey plan could likely be completed in 1–2 days. One monitoring plan option could be to visit these 37 stations annually in August during neap tide, and then visit the full suite of ~50 original stations every 5 years in order to provide a Bay-wide perspective at a lesser interval. Using the benthic biotope classification developed for this study, future results can be entered into the time series and interpreted in the context of the shifting critical boundaries of organic enrichment in the Bay.

B2.6.8 Conclusions

The *Ampelisca* biotope may be the most important biotope to track in Narragansett Bay. *Ampelisca* biotopes appear to follow the "critical boundaries" of organic enrichment, and *Ampelisca* tubes can exist in such dense aggregations that they are likely important prey sources for the demersal fish of Narragansett Bay. Demersal fish species in Narragansett Bay have declined in number over the past 47 years and especially since 1980, concurrent with increases in water temperature and decreases in chlorophyll concentrations (Collie et al. 2008). When the critical boundaries of organic enrichment are in the more shallow, protected (constricted) regions of the Bay as in 2008, robust *Ampelisca* biotopes may serve as critical habitats for juvenile demersal fish such as winter flounder. When the critical boundaries of organic enrichment existed in deeper, less protected waters as in 1988, fewer *Ampelisca* biotopes were observed. Therefore, benthic biotope composition and quality in 1988 may have represented poorer conditions for the protection and growth of juvenile demersal fish. With future warming and decreasing anthropogenic nutrient inputs, *Ampelisca* biotopes should be monitored more frequently as potential indicators of patterns in organic enrichment and important fish habitat.

Overall, biotope mosaics can be very useful for tracking habitat change if the data are available to develop such analyses. This case study demonstrated that continuous-coverage habitat data are not required in order to take a mosaic approach. The proportions of key habitats Bay-wide and the changes in their spatial arrangements were measurable from point-sample observations. Two observation periods (i.e., 1988 and 2008) do not enable trends in Bay-wide habitat quality to be gleaned, so a conceptual model was developed for what future observations of biotope type, proportion, and arrangement could mean for overall estuarine habitat quality. This type of conceptual model is essentially a BCG, and could help managers respond quickly according to the latest monitoring results. Work will continue to better develop these data in a more explicit estuarine BCG format for use as a communications tool.

B2.7 Status of This Effort

Work continues with the Narragansett Bay Estuary Program (NBEP) to apply the results of this latest case study, as well as concepts developed in the Greenwich Bay case study, to their next State of the Watershed Report, due to be released in 2016. These data will support the reporting of a benthic habitat quality indicator for Narragansett Bay and a proposed program for monitoring future changes in benthic habitat quality. While not explicitly using the BCG framework, NBEP is considering a habitat mosaic approach with two other habitat types to be included—salt marsh and seagrass. The estuarine BCG work group will contribute a short case example of the BCG approach for these three habitat quality indicators in the NBEP State of the Watershed Report to introduce the estuarine BCG framework, highlight similarities between it and the existing NBEP indicator framework, and demonstrate its potential usefulness for management. A large amount of quantitative data and analyses have been assembled by the NBEP for their latest State of the Watershed Report. It is possible that the NBEP community of partners will be interested in further developing BCG concepts, as well as potentially developing quantitative thresholds for ecosystem metrics to stressor levels in the near future.

The broader theme of linking benthic habitat quality to common estuarine stressors (e.g., organic enrichment, climate change) will continue to be explored through partners including the Narragansett Atlantic Ecology Division[16] lab, the Southeast New England Program,[17] NBEP, and the Buzzards Bay Estuary Program. The Narragansett Bay and Buzzards Bay estuaries share similar ecosystem attributes and have experienced similar stressor histories, but have different supporting data sets, monitoring programs, and scientific studies. This regional umbrella allows exploration and testing of the ability of the estuarine BCG framework to offer common measures of biological condition among these ecosystems.

Appendix B2 References

Bald, J., A. Borja, I. Muxika, J. Franco, and V. Valencia. 2005. Assessing reference conditions and physico-chemical status according to the European Water Framework Directive: A case-study from the Basque Country (Northern Spain). *Marine Pollution Bulletin* 50:1508–22.

BBC. 2011. *State of Buzzards Bay*. Buzzards Bay Coalition, New Bedford, MA. http://www.savebuzzardsbay.org/document.doc?id=645. Accessed January 2016.

Bergondo, D.L., D.R. Kester, H.E. Stoffel, and W.L. Woods. 2005. Time-series observations during the low sub-surface oxygen events in Narragansett Bay during summer 2001. *Marine Chemistry* 97:90–103.

Borja, Á., and D. Dauer. 2008. Assessing the environmental quality status in estuarine and coastal systems: Comparing methodologies and indices. *Ecological Indicators* 8:331–337.

Borja, Á., D.M. Dauer, and A. Grémare. 2011. The importance of setting targets and reference conditions in assessing marine ecosystem quality. *Ecological Indicators* 12:1–7.

Boström, C., S.J. Pittman, C. Simenstad, and R. Kneib. 2011. Seascape ecology of coastal biogenic habitats: Advances, gaps, and challenges. *Marine Ecology Progress Series* 427:191–217.

Calabretta, C.J., and C.A. Oviatt. 2008. The response of benthic macrofauna to anthropogenic stress in Narragansett Bay, Rhode Island: A review of human stressors and assessment of community conditions. *Marine Pollution Bulletin* 56:1680–1695.

[16] Narragansett Atlantic Ecology Division is part of EPA Office of Research and Development.

[17] The Southeast New England Program is part of EPA Region 1.

CBP. 2000. *Chesapeake 2000 Agreement*. Chesapeake Bay Program, Annapolis, MD. http://www.chesapeakebay.net/documents/cbp_12081.pdf. Accessed January 2016.

Cicchetti, G. 2010. *Summary of a Technical Workshop Held October 28 and 29, 2009: Developing a Biological Condition Gradient for Narragansett Bay*. Report AED-10-088. U.S. Environmental Protection Agency, Atlantic Ecology Division.

Cicchetti, G., S. Latimer, S. Rego, G. Nelson, J. Bergen, and L. Coiro. 2006. Relationships between near-bottom dissolved oxygen and sediment profile camera measures. *Journal of Marine Systems* 62:124–141.

Cicchetti, G., and M. Pryor. 2010. *Summary of the Estuarine BCG Workgroup November 2008 Workshop, and a Proposed Organizing Framework for Bioassessment of Estuaries*. Report AED-10-087. U.S. Environmental Protection Agency, Atlantic Ecology Division.

Cicchetti, G., and H. Greening. 2011. Estuarine biotope mosaics and habitat management goals: An application in Tampa Bay, FL, USA. *Estuaries and Coasts* 34:1278–1292.

Cicchetti, G., M.C. Pelletier, M. Pryor, S.K. Jackson, P. Bradley, S.P. Davies, C.F. Deacutis, K.J. Rocha, D.L. Santavy, and E.J. Shumchenia. 2016. *A Biological Condition Gradient Framework for Management of Estuaries and Coasts*. EPA/600/R-15/287. U.S. Environmental Protection Agency, Office of Research and Development, Atlantic Ecology Division, Narragansett RI.

Codiga, D.L., H.E. Stoffel, C.F. Deacutis, S. Kiernan, and C.A. Oviatt. 2009. Narragansett Bay hypoxic event characteristics based on fixed-site monitoring network time series: Intermittency, geographic distribution, spatial synchronicity, and interannual variability. *Estuaries and Coasts* 32:621–641.

Collie, J.S., A.D. Wood, and H.P. Jeffries. 2008. Long-term shifts in the species composition of a coastal fish community. *Canadian Journal of Fisheries and Aquatic Sciences* 65:1352–1365.

Connor, D.W., J.H. Allen, N. Golding, L. Kerry, L.M. Lieberknecht, K.O. Northen, and J.B. Reker. 2004. The Marine Habitat Classification for Britain and Ireland Version 04.05. In JNCC (2015) The Marine Habitat Classification for Britain and Ireland Version 15.03. http://jncc.defra.gov.uk/MarineHabitatClassification. Accessed January 2016.

Costello, M.J. 2009. Distinguishing marine habitat classification concepts for ecological data management. *Marine Ecology Progress Series* 397:253–268.

Davies, C.E., D. Moss, and M.O. Hill. 2004. *EUNIS Habitat Classification Revised 2004*. Report to European Environment Agency, European Topic Center on Nature Protection and Biodiversity.

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.

Deacutis, C.F. 2008. Evidence of Ecological Impacts from Excess Nutrients in Upper Narragansett Bay. In *Science for Ecosystem-based Management: Narragansett Bay in the 21st Century*, eds. A. Desbonnet and B.A. Costa-Pierce, pp. 349–382. Springer, New York.

Deacutis, C.F., D.W. Murray, W. Prell, E. Saarman, and L. Korhun. 2006. Hypoxia in the upper half of Narragansett Bay, RI, during August 2001 and 2002. *Northeastern Naturalist* 13:173–198.

Diaz, R.J. 1995. *Benthic Habitat Classification of Selected Areas of Narragansett Bay and the Providence River, Rhode Island*. Prepared for Normandeau Associates, Inc., Bedford, NH by R.J. Diaz and Daughters, Ware Neck, VA.

Diaz, R.J., D.C. Rhoads, J.A. Blake, R.K. Kropp, and K.E. Keay. 2008. Long-term trends of benthic habitats related to reduction in wastewater discharge to Boston Harbor. *Estuaries and Coasts* 31:1184–1197.

Dunning, J.B., B.J. Danielson, and H.R. Pulliam. 1992. Ecological processes that affect populations in complex landscapes. *Oikos* 65:169–175.

FGDC. 2012. Coastal and Marine Ecological Classification Standard (CMECS) version IV. FGDC-STD-018-2012. Federal Geographic Data Committee. https://coast.noaa.gov/digitalcoast/publications/cmecs. Accessed February 2016.

Franz, D.R., and J.T. Tanacredi. 1992. Secondary production of the amphipod Ampelisca abdita Mills and its importance in the diet of juvenile winter flounder (*Pleuronectes americanus*) in Jamaica Bay, New York. *Estuaries* 15:193–203.

Frithsen, J.B. 1990. *The Benthic Communities within Narragansett Bay*. Technical report #NBP-90-28 of the Narragansett Bay Estuary Program.

Germano, J.D., and D.C. Rhoads. 1988. *Narragansett Bay Sediment Quality Survey: August 1988.* Technical report #NBP-89-23 of the Narragansett Bay Estuary Program.

Germano, J.D., D.C. Rhoads, R.M. Valente, D.A. Carey, and M. Solan. 2011. The use of sediment profile imaging (SPI) for environmental impact assessments and monitoring studies: Lessons learned from the past four decades. *Oceanography and Marine Biology: An Annual Review* 49:235–298.

Granger, S.L., M. Brush, B.A. Buckley, M. Traber, M. Richardson, and S.W. Nixon. 2000. An assessment of eutrophication in Greenwich Bay. In *Restoring Water Quality in Greenwich Bay,* ed. M. Schwartz. Rhode Island Sea Grant, Narragansett RI.

Greening, H., and A. Janicki. 2006. Toward reversal of eutrophic conditions in a subtropical estuary: Water quality and seagrass response to nitrogen loading reductions in Tampa Bay, Florida, USA. *Environmental Management* 38:163–78.

Henningsen, B. 2005. The maturation and future of habitat restoration programs for the Tampa Bay estuarine ecosystem. ed. S.F. Treat. In *Proceedings of the Tampa Bay Area Scientific Information Symposium, BASIS 4*, Tampa Bay Regional Planning Council, St. Petersburg, Florida, October 27–30, 2003, pp. 165–170.

Levin, L.A., W. Ekau, A.J. Gooday, F. Jorissen, J.J. Middelburg, W. Naqvi, C. Neira, N.N. Rabalais, and J. Zhang. 2009. Effects of natural and human-induced hypoxia on coastal benthos. *Biogeosciences Discussions* 6:3563–3654.

Lotze, H.K., H.S. Lenihan, B.J. Bourque, R.H. Bradbury, R.G. Cooke, M.C. Kay, S.M. Kidwell, M.X. Kirby, C.H. Petersen, and J.B.C. Jackson. 2006. Depletion, degradation, and recovery potential of estuaries and coastal seas. *Science* 312: 1806–1808.

MacKenzie, C.L., Jr., R.A. Pikanowski, and D.G. McMillen. 2006. Ampelisca amphipod tube mats may enhance abundance of northern quahogs Mercenaria mercenaria in muddy sediments. *Journal of Shellfish Research* 25:841–847.

McCall, P.L. 1977. Community patterns and adaptive strategies of the infaunal benthos of Long Island Sound. *Journal of Marine Research* 35:221–266.

McMaster, R.L. 1960. Sediments of Narragansett Bay system and Rhode Island Sound, Rhode Island. *Journal of Sedimentary Petrology*, *30*(2), 249–274.

Melrose, D.C., C.A. Oviatt, and M.S. Berman. 2007. Hypoxic events in Narragansett Bay, Rhode Island, during the summer of 2001. *Estuaries and Coasts* 30:47–53.

Meng, L., G. Cicchetti, and S. Raciti. 2005. Relationships between juvenile winter flounder and multiple-scale habitat variation in Narragansett Bay, Rhode Island. *Transactions of the American Fisheries Society* 134:1509–1519.

Muxika, I., A. Borja, and J. Bald. 2007. Using historical data, expert judgement and multivariate analysis in assessing reference conditions and benthic ecological status, according to the European Water Framework Directive. *Marine Pollution Bulletin* 55:16–29.

Myers, A.C., and D.K. Phelps. 1978. *Criteria of Benthic Health: A Transect Study of Narragansett Bay, Rhode Island*. Prepared for the U.S. Environmental Protection Agency, Ecosystem Research Laboratory, Narragansett, RI.

Nixon, S.W., S.L. Granger, and B.L. Nowicki. 1995. An assessment of the annual mass balance of carbon, nitrogen, and phosphorus in Narragansett Bay. *Biogeochemistry* 31:15–61.

Nixon, S.W., B.A. Buckley, S.L. Granger, L. Harris, A.J. Oczkowski, and L. Cole. 2005. *Anthropogenic nutrient inputs to Narragansett Bay: A Twenty-five Year Perspective*. Narragansett Bay Commission and Rhode Island Sea Grant, 52 pp.

Nixon, S.W., B.A. Buckley, S.L. Granger, L.A. Harris, A.J. Oczkowski, R.W. Fulweiler, and L.W. Cole. 2008. Nitrogen and Phosphorus Inputs to Narragansett Bay: Past, Present, and Future. In *Science for Ecosystem-based Management: Narragansett Bay in the 21st Century*, eds. A. Desbonnet and B.A. Costa-Pierce, pp. 101–175. Springer, New York.

Nixon, S.W., R.W. Fulweiler, B.A. Buckley, S.L. Granger, B.L. Nowicki, and K.M. Henry. 2009. The impact of changing climate on phenology, productivity, and benthic-pelagic coupling in Narragansett Bay. *Estuarine, Coastal and Shelf Science* 82:1–18.

Nixon, S.W., and R.W. Fulweiler. 2012. Ecological footprints and shadows in an urban estuary, Narragansett Bay, RI (USA). *Regional Environmental Change* 12:381–394.

Oviatt, C.A., A. Keller, and L. Reed. 2002. Annual primary production in Narragansett Bay with no bay-wide winter–spring phytoplankton bloom. *Estuarine, Coastal and Shelf Science* 54:1013–1026.

Oviatt, C.A. 2004. The changing ecology of temperate coastal waters during a warming trend. *Estuaries* 27: 895–904.

Papworth, S.K., J. Rist, L. Coad, and E.J. Milner-Gulland. 2009. Evidence for shifting baseline syndrome in conservation. *Conservation Letters* 2:93–100.

Pauly, D. 1995. Anecdotes and the shifting baseline syndrome of fisheries. *Trends in Ecology and Evolution* 10:430.

Pearson, T.H., and R. Rosenberg. 1978. Macrobenthic succession in relation to organic enrichment and pollution of the marine environment. *Oceanography and Marine Biology: An Annual Review* 16:229–311.

Pilson, M.E.Q. 1985. On the Residence Time of Water in Narragansett Bay. *Estuaries*, *8*(1):2.

Pinsky, M.L., B. Worm, M.J. Fogarty, J.L. Sarmiento, and S.A. Levin. 2013. Marine taxa track local climate velocities. *Science* 341: 1239–1242.

Pittman, S.J., C. Caldow, S.D. Hile, and M.E. Monaco. 2007. Using seascape types to explain the spatial patterns of fish in the mangroves of SW Puerto Rico. *Marine Ecology Progress Series* 348:273–284.

Pittman, S.J., R. Kneib, and C. Simenstad. 2011. Practicing coastal seascape ecology. *Marine Ecology Progress Series* 427:187–190.

Powers, R. 2015. Boston blackbacks. *On The Water New England Edition*. pp 50–55.

Prell, W., E. Saarman, D.W. Murray, and C.F. Deacutis. 2004. *Summer-Season, Nighttime Surveys of Dissolved Oxygen in Upper Narragansett Bay (1999–2003)*.

Raposa, K.B. 2009. Ecological Geography of Narragansett Bay. In An Ecological Profile of the Narragansett Bay Estuarine Research Reserve, eds. K.B. Raposa and M.L. Schwartz, 77–88. Rhode Island Sea Grant, Narragansett RI.

Rhoads, D.C., and J.D. Germano. 1982. Characterization of organism-sediment relationships using sediment profile imaging: An efficient method of Remote Ecological Monitoring of the Seafloor (REMOTS System). *Marine Ecology Progress Series* 8: 115–128.

Rhoads, D.C., and J.D. Germano. 1986. Interpreting long-term changes in benthic community structure: A new protocol. *Hydrobiologia* 142:291–308.

RIDEM. 2003. *The Greenwich Bay Fish Kill – August 2003: Causes, Impacts and Responses*. Rhode Island Department of Environmental Management, Providence, RI.

RIDEM. 2005. *Plan for Managing Nutrient Loadings to Rhode Island Waters*. Rhode Island Department of Environmental Management, Providence, RI.

RIDEM. 2008. *Rhode Island Wastewater Treatment Facility Information*. Rhode Island Department of Environmental Management, Providence, RI.

Rosenberg, R. 2001. Marine benthic faunal successional stages and related sedimentary activity. *Scientia Marina* 65:107–119.

Salomidi, M., S. Katsanevakis, A. Borja, U. Braeckman, and D. Damalas. 2012. Assessment of goods and services, vulnerability, and conservation status of European seabed biotopes: A stepping stone towards ecosystem-based marine spatial management. *Mediterranean Marine Science* 13:49–88.

Samhouri, J.F., P.S. Levin, C.A. James, J. Kershner, and G. Williams. 2011. Using existing scientific capacity to set targets for ecosystem-based management: A Puget Sound case study. *Marine Policy* 35:508–518.

Santos, S.L., and J.L. Simon. 1980. Response of soft-bottom benthos to annual catastrophic disturbance in a south Florida estuary. *Marine Ecology Progress Series* 3:347–355.

Shumchenia, E.J., M.L. Guarinello, and J.W. King. Submitted. A re-assessment of Narragansett Bay benthic habitat quality between 1988 and 2008. In press. *Estuaries and Coasts*.

Shumchenia, E.J., and J.W. King. 2010a. Evaluation of sediment profile imagery as a tool for assessing water quality in Greenwich Bay, Rhode Island, USA. *Ecological Indicators* 10:818–825.

Shumchenia, E.J., and J.W. King. 2010b. Comparison of methods for integrating biological and physical data for marine habitat mapping and classification. *Continental Shelf Research* 30:1717–1729.

Shumchenia, E.J., M.C. Pelletier, G. Cicchetti, S. Davies, C.E. Pesch, C.F. Deacutis, and M. Pryor. 2015. A biological condition gradient model for historical assessment of estuarine habitat structure. *Environmental Management* 55:143–58.

Spalding, M.D., H.E. Fox, G.R. Allen, N. Davidson, Z.A. Ferdaña, M. Finlayson, B.S. Halpern, M.A. Jorge, A. Lombana, S.A. Lourie, K.D. Martin, E. McManus, J. Molnar, C.A. Recchia, and J. Robertson. 2007. Marine Ecoregions of the World: A Bioregionalization of Coastal and Shelf Areas. *BioScience 57*(7):573.

Stickney, A.P., and L.D. Stringer. 1957. A study of the invertebrate fauna of Greenwich Bay, Rhode Island. *Ecology* 38:111–122.

Stoddard J.L., D.P. Larsen, C.P. Hawkins, R.K. Johnson, and R.H. Norris. 2006. Setting expectations for the ecological condition of streams: The concept of reference condition. *Ecological Applications* 16:1267–1276.

TBEP. 2012. *A Tampa Bay Estuary Program Progress Report 2012*. Tampa Bay Estuary Program. http://www.tbep.org/pdfs/tbep_state_of_bay_2012_ptr_reduced.pdf. Accessed January 2016.

USEPA. 2000. *Estuarine and Coastal Marine Waters: Bioassessment and Biocriteria Technical Guidance*. EPA-822-B-00-024. U.S. Environmental Protection Agency, Washington, DC.

USEPA. 2005. DRAFT: *Use of Biological Information to Better Define Designated Aquatic Life Uses: Tiered Aquatic Life Uses*. EPA-822-R-05-001. U.S. Environmental Protection Agency, Washington, DC.

USEPA. 2011. *A Primer on Using Biological Assessments to Support Water Quality Management*. EPA 810-R-11-01. U.S. Environmental Protection Agency, Office of Science and Technology and Office of Water, Washington, DC.

Valente, R.M., D.C. Rhoads, J.D. Germano, and V.J. Cabelli. 1992. Mapping of benthic enrichment patterns in Narragansett Bay, Rhode Island. *Estuaries* 15:1–17.

Wiens, J.A., N.C. Stenseth, B. Van Horne, and R.M. Ims. 1993. Ecological mechanisms and landscape ecology. *Oikos* 66:369–380.

Zajac, R.N. 2008. Challenges in marine, soft-sediment benthoscape ecology. *Landscape Ecology* 23:7–18.

B3. Caribbean Coral Reefs: Benchmarking a Biological Condition Gradient for Puerto Rican Coral Reefs

Patricia Bradley[18] and Deborah Santavy, PhD., USEPA Office of Research and Development, National Health and Environmental Effects Research Laboratory–Gulf Ecology Division, Gulf Breeze, Florida

B3.1 Background

More than half of the U.S. population lives in coastal counties—areas that border oceans, bays, and estuaries (NOAA 2014). In the states of Florida and Hawai'i and the territories of Puerto Rico, U.S. Virgin Islands (USVI), Guam, American Samoa, and the Commonwealth of the Northern Mariana Islands, nearly everyone lives within 60 miles of the coast, causing impacts to coral reef ecosystems from development, fishing pressure, and climate change.

Several federal agencies (e.g., the National Oceanic and Atmospheric Administration (NOAA), the National Park Service, the U.S. Fish and Wildlife Service, and EPA), together with state and territorial environmental and natural resource agencies, are responsible for the management and protection of U.S. coral reefs.

The Coral Reef Conservation Act of 2000 (16 USC § 6401 2000) sets forth the requirement for a national monitoring program to promote the understanding, conservation, and sustainable use of coral reef ecosystems. The President's Ocean Action Plan (The White House 2004) directed EPA to develop biological criteria and assessment methods for states and territories to evaluate the condition of coral reefs and surrounding marine water quality (Bradley et al. 2008, Bradley et al. 2010). Currently, EPA Region 2 is working with Puerto Rico Environmental Quality Board and USVI Department of Planning and Natural Resources to revise the territorial water quality standards to be more explicitly protective of coral reefs and other aquatic habitats, including incorporating ALU language into the water quality standards and using the BCG to develop narrative and numeric coral reef biological criteria. Development of a coral reef BCG is considered a first step towards coral reef biological criteria. The model can be used to assess baseline condition. Use of the BCG in this manner could help avoid predicaments associated with shifting baselines (Pauly 1995) by tracking biological effects of gradual regional or global stresses (such as coastal/ocean acidification or rising sea surface temperatures).

In Puerto Rico in 2011, EPA assembled a panel of U.S. Caribbean coral reef and fisheries experts (Bradley et al. 2014). During a series of three workshops and numerous webinars over the following 5 years, the experts described the aquatic assemblages under natural conditions; identified the predominant regional stressors; described the BCG, including the theoretical foundation and observed assemblage response to stressors; developed increasingly quantitative decision rules; and began to calibrate the conceptual model, populating it with monitoring data.

This case study summarizes the status of BCG model development by covering the following topics:

1) Development of a Conceptual BCG Framework for Coral Reefs in the Caribbean (workshop 1)
2) Development of Quantitative BCG Models for Two Coral Reef Assemblages (workshops 2 and 3)

[18] Patricia Bradley retired from USEPA in October 2015. Since that time, she has joined Tetra Tech, Inc. She can be reached at: patbradley@comcast.net.

   a. Fish Expert Breakout Group
   b. Benthic Macroinvertebrates Breakout Group
   c. Bringing the Two Assemblage Breakout Groups Together

3) Enhancing Monitoring Design and Protocols

B3.2 Development of a Conceptual Model

The first workshop provided proof of concept that the BCG can be adapted for coral reef ecosystems (Bradley et al. 2014). The panel of experts evaluated and ranked coral reef condition from photographs and videos collected during EPA's 2010 and 2011 coral reef assessment surveys in shallow waters (< 12 m deep) of southwestern Puerto Rico. The biological assemblages considered were stony corals, fishes, sponges, gorgonians, and benthic macroinvertebrates. Participants examined the visual media, rated the condition of various coral reefs and provided rationale for their ratings (Figure B3-1). Descriptions of characteristics relative to natural and degraded ecological condition were captured during facilitated discussions.

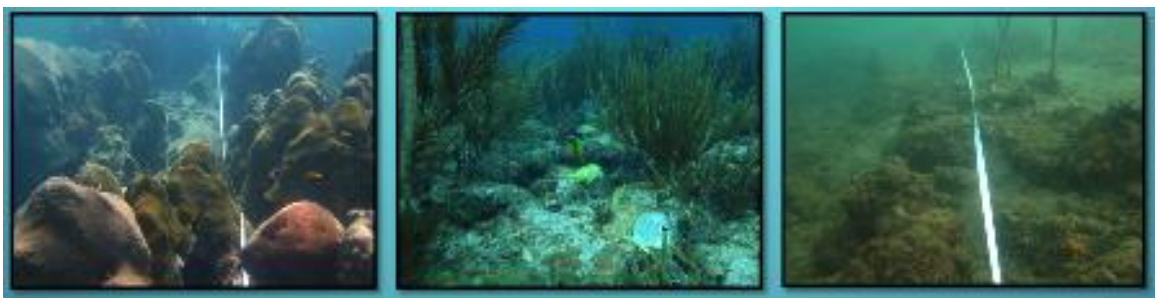


Figure B3-1. Photos from EPA coral reef sites reflect a range of coral reef conditions, from good (left) to intermediate quality (middle), to severely degraded (right).

The experts proposed a preliminary narrative BCG with four levels of condition (very good–excellent; good; fair; and poor), and associated physical and biological attributes (Table B3-1).

Table B3-1. Narrative condition levels and associated BCG attributes (Bradley et al. 2014).

Condition level	Attribute descriptions
Very Good Excellent **(approximate BCG Level 1–2)**	Physical structure: High rugosity or 3D structure; substantial reef built above bedrock; many irregular surfaces provide habitat for fish; very clear water; no sediment, flocs or films
	Corals: High species diversity including rare; large old colonies (*Orbicella*) with high tissue coverage; balanced population structure (old and middle-sized colonies, recruits); *Acropora* thickets present
	Gorgonians: Gorgonians present but subdominant to corals
	Sponges: Large autotrophic and highly sensitive sponges abundant
	Fish: Populations have balanced species abundances, sizes, and trophic interactions
	Large vertebrates: Large, long-lived species present and diverse (turtles, eels, sharks)
	Other invertebrates: *Diadema,* lobster, small crustaceans, and polychaetes abundant; some large sensitive anemone species present
	Algae: Crustose coralline algae abundant; turf algae present but cropped and grazed by *Diadema* and herbivorous fish; low abundance of fleshy algae
	Condition: Low prevalence of disease and tumors; mostly live tissue on colonies

Condition level	Attribute descriptions
Good (approximate BCG Level 3)	Physical structure: Moderate to high rugosity; moderate reef built above bedrock; some irregular cover for fish habitat; water slightly turbid; low sediment, flocs or films on substrate
	Corals: Moderate coral diversity; large old colonies (*Orbicella*) with some tissue loss; varied population structure (usually old colonies, few middle aged and some recruits); *Acropora* thickets may be present; rare species absent
	Gorgonians: Gorgonians more abundant than levels 1–2
	Sponges: Autotrophic species present but highly sensitive species missing
	Fish: Decline of large apex predators (e.g., groupers, snappers) noticeable; small reef fishes more abundant
	Large vertebrates: Large, long-lived species locally extirpated (turtles, eels)
	Other invertebrates: *Diadema,* lobster, small crustaceans, and polychaetes less abundant than levels 1–2; large sensitive anemone species absent
	Algae: Crustose coralline algae present but fewer than levels 1–2; turf algae present and longer, more fleshy algae present than levels 1–2
	Condition: Disease and tumor presence slightly above background level; more colonies have irregular tissue loss
Fair (approximate BCG Level 4)	Physical structure: Low rugosity; limited reef built above bedrock; erosion of reef structure obvious; water turbid; more sediment accumulation, flocs and films; *Acropora* usually gone or present as rubble for recruitment substrate
	Corals: Reduced coral diversity; emergence of tolerant species, few or no living large old colonies (*Orbicella*); *Acropora* thickets gone, large remnants mostly dead with long uncropped turf algae
	Gorgonians: Gorgonians more abundant than levels 1–3, replacing sensitive coral and sponge species
	Sponges: Mostly heterotrophic tolerant species and clionids
	Fish: Absence of small reef fishes (mostly Damselfish remain)
	Large vertebrates: Large, long-lived species locally extirpated (turtles, eels)
	Other invertebrates: *Diadema* absent; *Palythoa* overgrowing corals; crustaceans, polychaetes and sensitive anemones conspicuously absent
	Algae: Some coralline algae present but no crustose coralline algae; turf is uncropped, covered in sediment; abundant fleshy algae (e.g., *Dictyota*) with high diversity
	Condition: Higher prevalence of diseased corals, sponges, gorgonians; evidence of high mortality; usually less tissue than dead portions on colonies
Poor (approximate BCG Level 5–6)	Physical structure: Very low rugosity; no or little reef built above bedrock; no or low relief for fish habitat; very turbid water; thick sediment film and thick floc covering bottom; no substrate for recruits
	Corals: Absence of colonies, those present are small; only highly tolerant species with little or no live tissue
	Gorgonians: Small and sparse colonies; mostly small sea fans; often diseased
	Sponges: Heterotrophic sponges buried deep in sediment; highly tolerant species
	Fish: No large fishes; only a few tolerant species remain; lack of multiple trophic levels
	Large vertebrates: Usually devoid of vertebrates other than fishes
	Other invertebrates: Few or no reef invertebrates; high abundance of sediment dwelling organisms such as mud-dwelling polychaetes and holothurians
	Algae: High cover of fleshy algae (*Dictyota)*; complete absence of crustose coralline algae
	Condition: High incidence of disease and low or no tissue coverage on small colonies of corals, sponges, and gorgonians, if present

Building on the preliminary BCG framework, EPA worked with the expert panel following the first workshop to prepare a database with which to develop decision rules and to explore technical approaches to define the BCG Generalized Stress Axis (GSA). A brief discussion of these two topics follows.

B3.2.1 Coral Reef Database

Puerto Rico's coral reef ecosystems are a complex mosaic of habitats, including mangrove forests, seagrass beds, and coral reefs (Garcia-Sais et al. 2008), and the ecosystems support 69 shallow-water (< 40 m) scleractinian species, 260 fish species, 46 shallow-water alcyonarian species, and 500 species of benthic marine algal flora (Ballantine et al. 2008).

Since no single monitoring program collects all the information needed to develop the BCG, EPA is assembling coral reef assessment data from past Puerto Rico and USVI studies into a single database. The initial biological data set included data collected by NOAA and EPA for dominant assemblages (fish, stony corals, and macroinvertebrates). The data were normalized using the taxonomic naming protocols of the Integrated Taxonomic Information System. The standardized data are in the original format, a 'crosswalk' with translations in a standardized format and metadata with geospatial information following the ESRI standards.

These data have been uploaded into the STOrage and RETrieval (STORET)[19] Data Warehouse, an EPA data repository for water quality, biological, and physical data, that is used by state and territorial environmental agencies for reporting their water quality data under the CWA. Data storage in STORET makes the data available to the experts and others so they can easily view and interpret the results.

B3.2.2 Generalized Stress Axis

EPA and the expert panel discussed the concept of a GSA and focused on three stressors that should be considered for coral reefs: (1) land-based sources of pollution, (2) fishing pressure, and (3) global climate change-associated thermal anomalies.

Land-based Sources of Pollution

EPA began stressor axis work by applying the Landscape Development Intensity index (LDI) to demonstrate the link between land-based human activity and coral reefs in USVI (Oliver et al. 2011) (see Chapter 5). The LDI is an integrated measure of the intensity of human activities in a landscape or watershed, estimated by calculating the input of nonrenewable energy to different land use parcels. The premise that ecological communities are affected by cumulative human impacts in the surrounding watershed was shown for wetlands (Brown and Vivas 2005). The LDI index was demonstrated to be an effective landscape indicator of human impact on St. Croix corals and is being developed for Puerto Rico (Figure B3-2).

[19] More information about STORET is available at: http://www.epa.gov/storet/. Accessed February 2016.

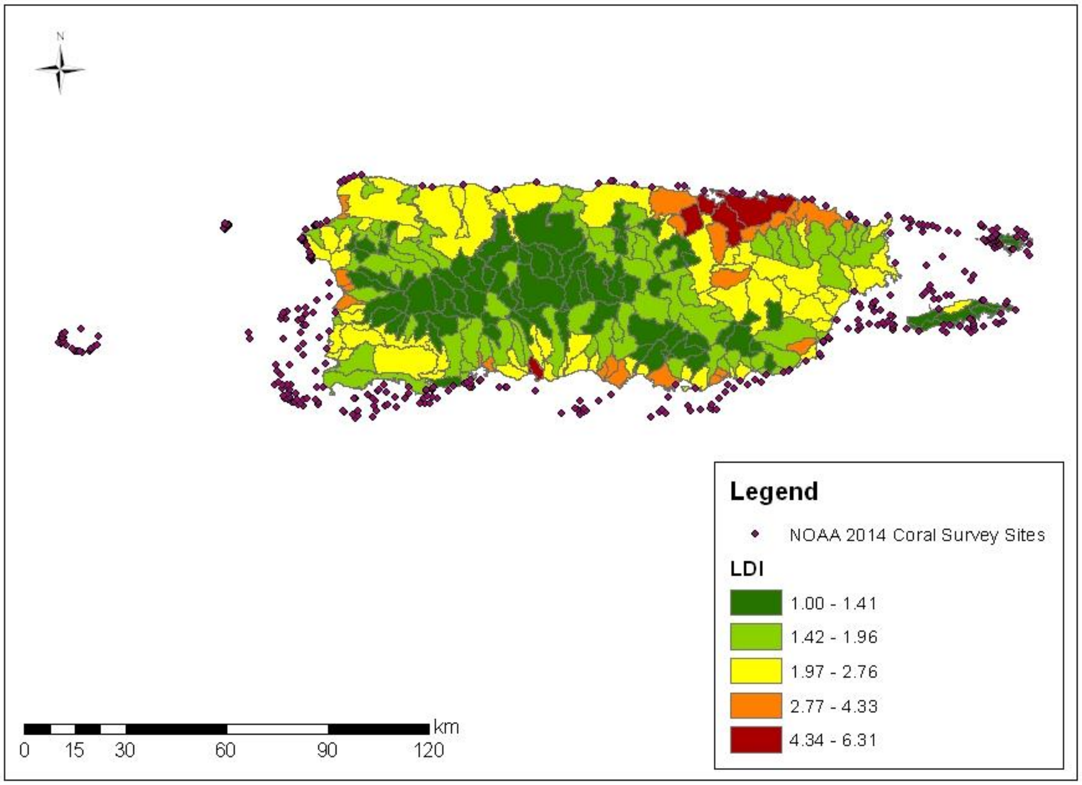


Figure B3-2. Puerto Rico 12-digit HUC watersheds and NOAA 2014 National Coral Reef Monitoring Program (NCRMP) coral stations. Watershed LDI values shown on a green–yellow–red continuum, where green indicates the lowest human activity and red indicates the highest.

Fishing Pressure

Over-fishing has dramatically altered the composition of biological communities on Caribbean coral reefs and seagrass beds. Large herbivores and carnivores such as turtles, groupers and sharks that were once abundant are now ecologically extinct (i.e., populations are so greatly reduced relative to past levels that the species no longer fulfills its former ecological/functional role). The reduction of these species has resulted in "trophic level dysfunction" (Steneck et al. 2004), with food chains now dominated by small fishes and invertebrates (Hay 1984, 1991; Knowlton et al. 1990; Jackson 1997). EPA is researching a modification of the approach developed by Ruiz Valentine (2013) to estimate fishing pressure from commercial, recreational, and artisanal (trap) fishers in Puerto Rico (Figure B3-3).

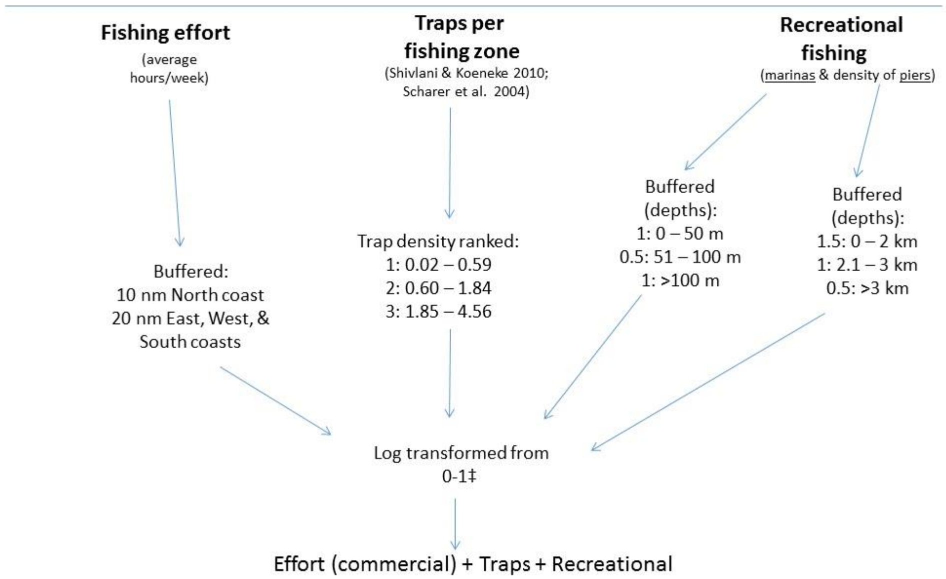


Figure B3-3. Fishing pressure in Puerto Rico (modified from Ruiz Valentine 2013).

Global Climate Change

The projected consequences of global climate change include shifts in ocean temperature, precipitation patterns, sea level rise, carbonate saturation equilibrium of calcite and aragonite, and other biogeochemical processes in the oceans. Most mass coral bleaching events have been associated with increased sea surface temperatures, which led to the long-term degradation of coral reefs worldwide (Brown and Suharsono 1990; Glynn 1991, 2000; Goreau et al. 1992; Goreau and Hayes 1994; Brown 1997; Goenega et al. 1989; Hoegh-Guldberg 1999; Wilkinson et al. 1999; Goreau et al. 2000; Reaser et al. 2000; Wilkinson 2000, 2004; Wellington et al. 2001; Hughes et al. 2003; Pandolfi et al. 2003; Sheppard 2003; Hoegh-Guldberg et al. 2007; Knowlton and Jackson 2008; Mora 2008). Increased carbon dioxide saturation rates in seawater reduce alkalinity and pH, which many believe will impact the survival and growth rates of calcium carbonate-secreting organisms (e.g., corals, bivalves, and calcareous algae) (Orr et al. 2005; Gattuso and Buddemeier 2000; Kleypas et al. 1999; Scavia et al. 2002). EPA has begun research to include threats from global climate change in the GSA, by using the NOAA Coral Reef Watch (CRW) thermal history experimental products to estimate the accumulation of thermal stress at significant levels to corals (Liu et al. 2012; Figure B3-4).

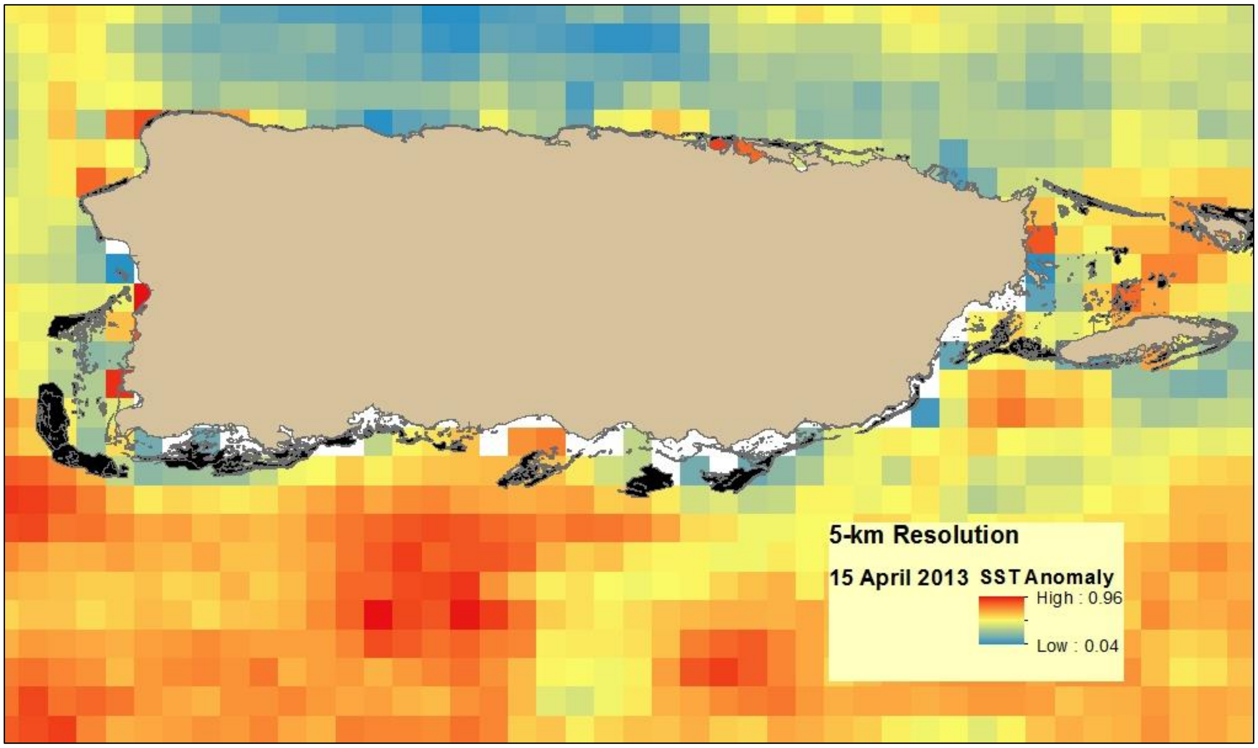


Figure B3-4. Sea Surface temperature (SST) anomaly: Example of NOAA's high-resolution (5-km^2) thermal anomaly products. SST Anomaly is the difference between daily SST and corresponding daily climatology. Daily climatology interpolates monthly mean SSTs, and as such detects cooler or warmer temperatures compared to long-term averages at specific locations (Source: NOAA CRW, Experimental Products, Thermal History).

B3.3 Development of Biological Condition Gradient Models for Two Coral Reef Assemblages

Similar to the freshwater streams efforts, the coral reef experts have been working in two breakout groups: mobile organisms (mainly fish) and sessile organisms (the benthos).

B3.3.1 Fish Breakout Group

The Fish Breakout Group assigned 128 species (fish observed during EPA's 2010 and 2011 surveys in Puerto Rico) to attributes.[20] The stressor categories that the experts considered most relevant to fish were land-based sedimentation and fishing pressure. For fishing pressure, the experts considered whether the species was subject to fishing pressure, the category of fishing (recreational, aquarium, or commercial), and whether the species was regulated. The experts did not assign any fish to attribute I, however they brainstormed fish species from the list of species observed historically in Puerto Rican coral reefs (n= 260) to be assigned to attribute I.

[20] See Table B3-5 in the Additional Data section after the references for this case study.

The experts used several indicators and metrics to distinguish BCG levels, including taxa richness; total biomass; sensitive taxa; density of damselfish, piscivores, and other fishes. Box plots showing the distributions of several metrics across the four assessed BCG levels are shown in Figures B3-5 and B3-6. Total taxa, richness of sensitive taxa, and total biomass were metrics often used by the panel (Figure B3-5). In contrast to taxa richness metrics, the percent density and dominance of damselfish increased at poorer BCG ratings (Figure B3-6). While several metrics are strongly associated with the panel's decisions, they relied on no single metric because most of the boxes show substantial overlap with adjacent BCG levels.

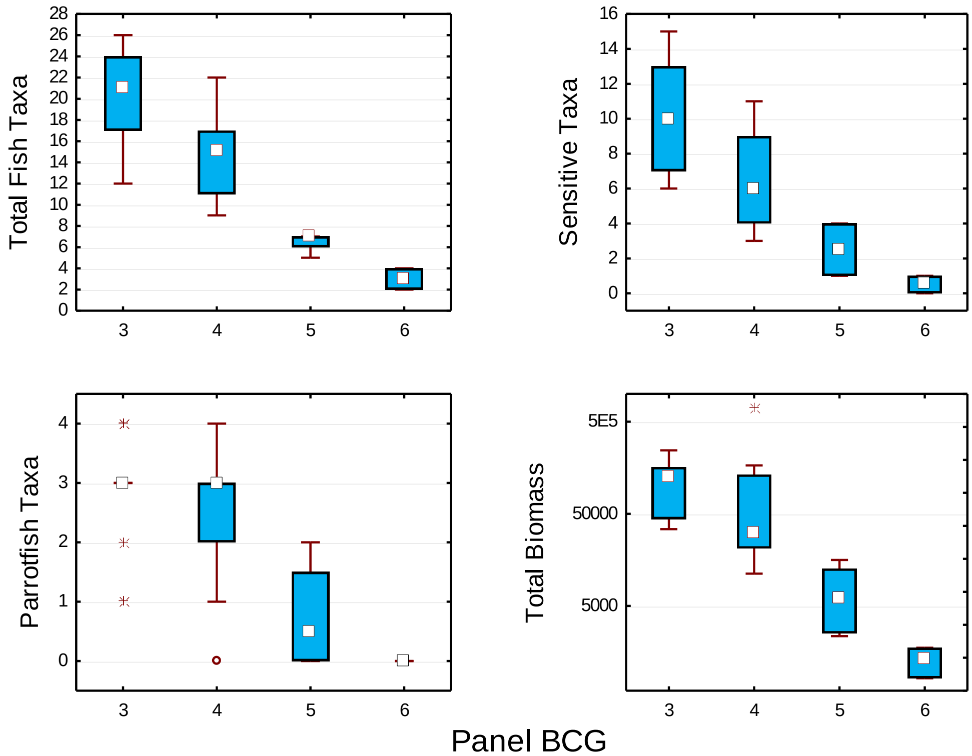


Figure B3-5. Box plots of fish metrics by panel BCG decision, total taxa, sensitive taxa (attributes II + III), parrotfish taxa, and total biomass.

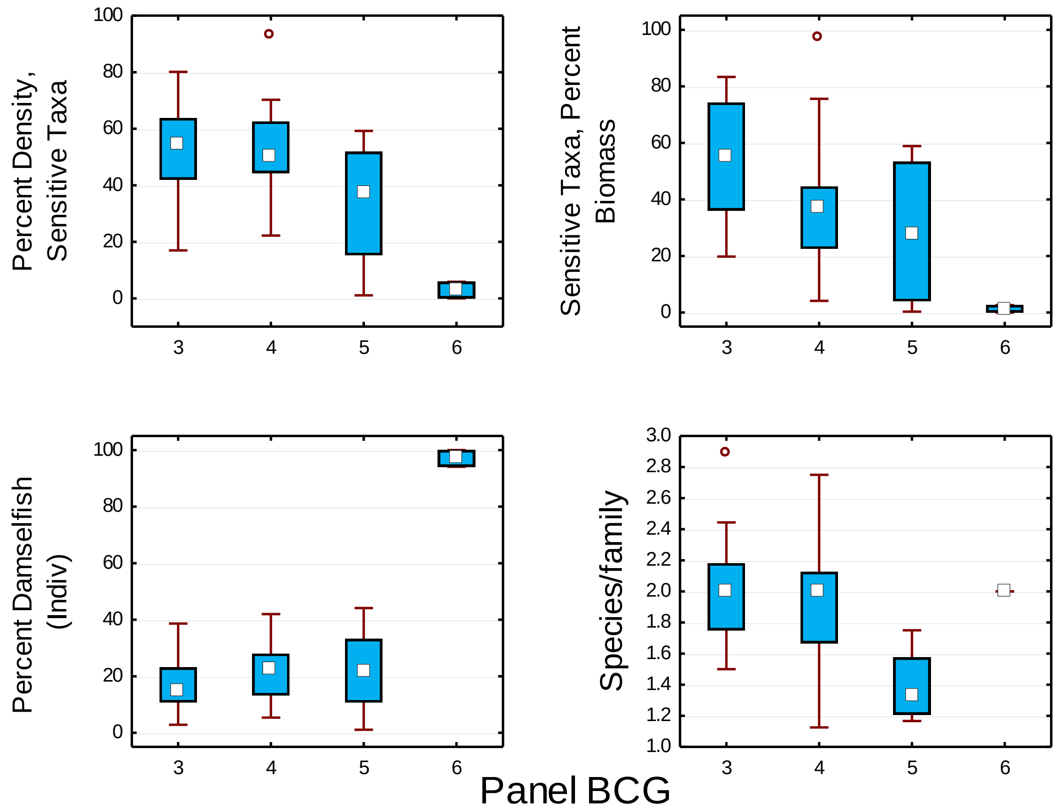


Figure B3-6. Box plots of fish metrics by panel BCG decision: Percent density of sensitive taxa (attributes II + III), percent biomass of sensitive taxa, and percent density of damselfish.

Quantitative rules were developed using the experts' narrative statements and the box plots to assign numbers to the narrative rules (Table B3-2).

Table B3-2. Reef Fish BCG Rules

	Narrative	Quantitative
BCG Level 3 (n = 15)		
Total taxa	Richness moderate to high	nt_total ≥ 15 (10–20)
Proportion all sensitive taxa	Small to moderate proportion of richness	nt_att23 ≥ 6 (4–8)
Total biomass	Fish biomass moderate to high	bio_total ≥ 35kg/100m^2 (30–40)
Piscivores	Presence of some snappers and other piscivores (proportion biomass)	pb_SP + pb_LP > 0
Within-family diversity (av # spp per family)	Within-family diversity not responsive	Not used
Parrotfish	Large body parrotfish present	nt_parrotfish2 ≥ 1 (0–2)
Damselfish	Damsels do not dominate catch	pd_damsels < 25% (20–30)
Groupers	Groupers present	nt_grouper > 0
Reef habitat rule	More stringent	Best 6 of 7 rules
Hardbottom habitat rule	Less stringent	Best 5 of 7 rules
BCG Level 4 (n = 17)		
Total taxa	Low to moderate diversity	nt-total ≥ 9 (4–14)
Proportion all sensitive taxa	Some sensitive taxa	nt_att23 ≥ 3 (1–5)
Total biomass	Low or higher	bio_total ≥ 11.5kg/100m^2 (7–15)
Within-family diversity (av # spp per family)	Within-group diversity of snappers, grunts, parrotfish, etc. declined	Not used (see level 3)
Parrotfish	Parrotfish present	Not used
Piscivores	At least one Snapper or piscivore present	Not used

During the calibration phase, the experts reviewed 11 sites, applying the fish rules (Table B3-2) to assign a BCG level to each site. Using the confirmation sites, the model correctly predicted nine (82% correct). There were, however, several issues that arose and could be further investigated. The issues are listed below with possible approaches for resolution.

Fish Size Distributions

Observations of juvenile and adult fish at a reef site might indicate that a full life cycle is supported at the site, inferring connectivity at the site for certain species. With observation of a single life stage, assessors were uncertain about the ability of the reef to support recruitment of juveniles or sustenance of adults. Therefore, in the BCG rating process, experts requested information about the size distribution (not just enumeration) of the fish observed. This information need is an example of how the BCG development process led to expert discussions and recommendations on monitoring protocols of coral reef systems in the future, including cross Federal agency coordination (see Section B3.4). The experts were familiar with critical sizes that might indicate single or multiple life stages and could relate the size and life-stage information to the integrity of the reef fish community.

During the field sampling, size was recorded in 5-cm intervals for all fish species, but association of juvenile and adult stages has not yet been completed for this data set. A listing of juvenile and adult size ranges for fish species might be available in the literature or might be created by the experts based on professional judgment. Enumeration of juvenile and adults (or size distribution) for future rating exercises would allow calculation of life-stage metrics for reef fish. The life stage metrics might allow better discrimination of BCG levels and connectivity. There was also discussion about modifying the attribute descriptions to reflect whether the fish were long lived—slow growth vs. short-lived—fast growth.

Correlations among Variables

Relationships among fish metrics, coral metrics, and environmental variables were suggested by the fish breakout group to increase their understanding of the fish community in response to stressors, environmental factors, and community interactions. A correlation matrix would be fairly quick to put together for the existing variables in the database, including relationships that were of particular interest to the experts (e.g., size of fish vs. size and number of fish; fish metrics and coral metrics).

Calibration to the Full Range of Conditions

Most of the consensus ratings for the sites in the data set were rated level 3 or 4. The distinction between levels 3 and 4 is often most difficult because these levels have vague narrative differences and metric distributions often overlap among samples rated 3 or 4. A few individual ratings were at level 2, and a few consensus ratings were made at 5 and 6. There were conceptual rules developed for level 2 and quantitative rules calibrated for levels 5 and 6. Confirmation ratings were only at levels 3 and 4, leaving the level 2, 5, and 6 rules un-validated. BCG level 1 was not expected to occur and was not described conceptually or with model rules.

A full range of biological conditions should be observed and rated to complete and confirm the BCG fish model. A detailed list of possible data to mine was developed that included databases from Puerto Rico, Florida, USVI, and the MesoAmerican Reef.

Fish Attributes

If the attributes are to become more useful, the fish need to be characterized first for frequency (commonness or rarity), and then for sensitivity and type of sensitivity. Commonness could be analyzed in the existing data set or with similar broad-scale surveys in the region. Sensitivity could be derived from literature, although toxicological approaches for reef fish are not common. In addition, connectivity attributes could be considered for each fish taxon. The experts recommended using the new NCRMP and other NOAA data to tease out the likelihood of seeing various fish species at a site. They suggested that the data collected by the Puerto Rico Aqueduct and Sewer Authority for CWA section 301(h) reporting purposes (fish species, habitat, water quality, sediment, coral species), could be used to assess for stressor response.

Fish Observation Protocols

The experts recommended an alternate observation method to collect data on presence of fish. Swimming along a transect causes fish to disperse and hide before they could be observed and identified. In the transect method, the fish counter does not look under gorgonians, sponges, etc., and it is not known how this may skew the data. An alternative method was suggested (Bohnsack and Bannerot 1986) that involved sitting (floating) still in one location along the transect for a period long enough that alarmed fish would return or emerge and be observable in the visible range (about 6.5 m). The experts recommended that a comparability study of the two methods be conducted concurrently at the same sites.

The fish experts also recommended revising the field method for rugosity since the rugosity measure did not always reflect what the experts observed in the videos. They felt that the NCRMP topographic complexity survey protocol would provide more information. NCRMP measures minimum/maximum depth and maximum vertical relief measurements within the entirety of a 25 m x 4 m transect. NCRMP also estimates topographic complexity within the transect area.

Defining Attribute X: Ecosystem Connectance

The fish group has initiated a discussion on attribute X. Connectivity between coral reefs, mangroves, sea grass beds, and lagoons provides a complex and dynamic mosaic that is well documented as a critical ecosystem attribute (Sale et al. 2008; Christensen et al. 2003; Aguilar-Perera and Appeldoorn 2007; McField and Kramer 2007; Mumby et al. 2004, 2008; Meynecke et al. 2008; Pittman et al. 2011). Higher densities of juveniles in mangroves and seagrass can be attributed to food availability, structural complexity, shade, and reduced predation (Beck et al. 2001; Adams et al. 2006; Dahlgren et al. 2006; Aguilar-Perera and Appeldoorn 2007) (Figures B3-7 and B3-8). Marine organisms may also make repeated migrations between habitats on various time scales, especially daily and seasonally (Sale et al. 2010). Ecosystem connectivity is therefore an important attribute to include in a coral reef conceptual model.

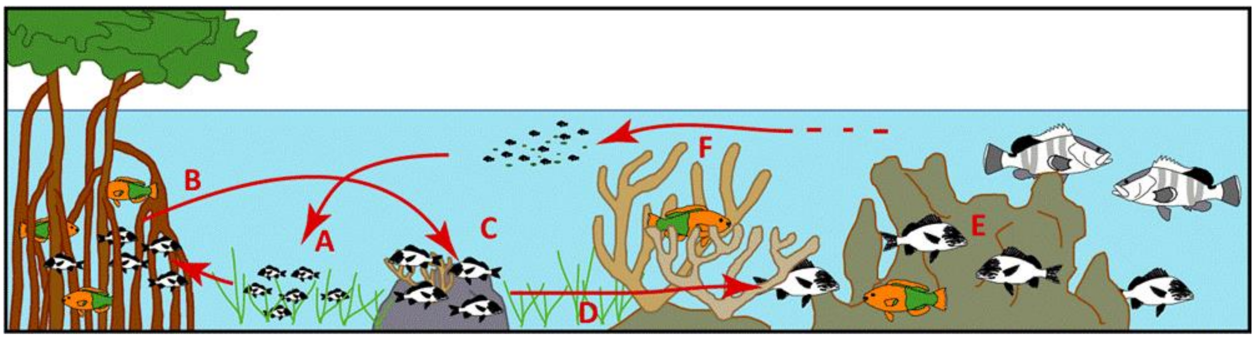


Figure B3-7. Mangroves present (modified from Mumby et al. 2004). Red letter "A" shows juvenile grunts, once reaching a given size in a seagrass bed, moving to mangroves (B). The mangroves serve as an intermediate nursery habitat before the fish migrate to patch reefs (C), and fish biomass is significantly enhanced on patch reefs (C), shallow fore reefs (D), and *Orbicella* reefs (E). Some fish (F), such as certain species of parrotfish, Scarus guacamaia, are dependent on mangroves and are not seen where mangroves are absent.

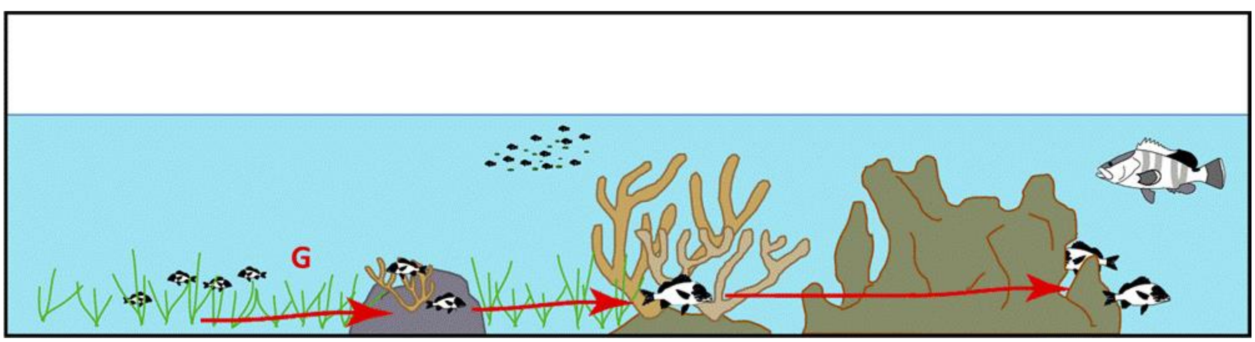


Figure B3-8. Mangroves Absent (Modified from Mumby et al. 2004). If the mangroves are not present, then fish move directly from the seagrass to the patch reefs, appearing on patch reefs (G) at a smaller size and at lower density, thus more vulnerable to predation.

The fish experts agreed that marine ecosystems are arrayed in space in response to gradients of:

- 3D structure
- depth
- water temperature
- salinity
- energy (wave regime, tide, currents, eddies)
- substrate type

The group felt that high-resolution reef bottom topography (LIDAR or other) was critically needed to allow for better estimation of connectivity. With high-resolution topography, features related to connectivity would be recognizable and quantifiable. High-resolution topography would also indicate elements of rugosity as well as connectivity, allowing characterization of broad-scale relief and a possible basis for classification of reefs. This project would require coordination among multiple agencies.

B3.3.2 Benthic Macroinvertebrate Breakout Group

The coral experts in the benthic group assigned 46 scleractinian and hydrozoan hard coral species found in the Western Atlantic to attributes II–VI based on their sensitivity and tolerance to human induced stressors. Studies documenting the tolerances of coral species to different anthropogenic stressors are very limited, so most of the assignments were based on professional consensus. The experts agreed that thermal anomalies and land-based stressors were the most critical threats to corals. They assigned each species to an attribute level separately, for elevated temperature exposure and sediment exposure.[21] The experts did not assign any species to attribute I, and only two species to attribute II. Some taxa were left unassigned when experts had no opinion.

Coral Reef Habitat Classification

The benthic experts wrestled with the fundamental issue of reef classification. Discussions centered on defining the expectations of which coral species should be found at each site during attempts to assign species to a BCG level. Reef traits proposed as important to consider for determining optimum species composition included habitat classification, geology, sea level change, sediment exposure, and decadal temperature anomalies (Hubbard 1997; Hubbard et al. 2009; Costa et al. 2009, 2013; Zitello et al. 2009). Coral reef environments have distinct horizontal and vertical zones created by differences in depth, wave and current energy, temperature, and light (Zitello et al. 2009). The experts considered several different reef classification systems in an attempt to incorporate most of the critical reef traits discussed (Adey and Burke 1976; Darwin 1874; Hubbard et al. 2009). They agreed to use the latest edition of NOAA's Benthic Habitat reef classification as guidance (Costa et al. 2009, 2013; Zitello et al. 2009), which classifies reef habitat by a hierarchical structure into reef types, geographic zones, and geomorphological structures (Table B3-3). The benthic experts agreed to only use the fore reef zone for assigning reefs to BCG condition levels. The fore reef is defined as the area along the seaward edge of the reef crest that slopes into deeper water on the barrier or fringing reef type (Costa et al. 2013). Features associated with a non-emergent reef crest (but still having a seaward-facing slope that is significantly greater than the slope of the bank/shelf) are also designated as fore reef. Experts agreed that fore reefs should be further divided into two habitats: coral-based bioherms dominated by *Orbicella* species and colonized hard bottoms with gorgonian plains (Williams et al. 2015). This approach should

[21] See Table B3-6 in the Additional Data section after the references for this case study.

provide a template for application to other well-defined coral reef habitats (e.g., deep fore reef/escarpment with coral reef coverage) to evaluate in the future.

Table B3-3. Benthic habitat classification scheme used to define discrete habitat classes (Adapted from Costa et al. 2013)

<table>
<tr><th>Reef Type</th><th>Reef Geographic Zones</th><th>Geomorphological Structures*</th></tr>
<tr><td>Barrier Reef</td><td>Shoreline Intertidal
Lagoon
Back Reef
Reef Flat
Reef Crest
Fore Reef
Bank/Shelf
Bank/Shelf Escarpment</td><td rowspan="3">Coral Reef & Hard bottom (Hard)
Aggregate Reef
Aggregated Patch Reefs
Individual Patch Reef
Pavement
Pavement with Sand Channels
Reef Rubble
Rhodoliths
Rhodoliths with Scattered Coral & Rock
Rock/Boulder
Spur & Groove
Unknown
Unconsolidated Sediment (Soft)
Mud
Sand
Sand with Scattered Coral & Rock
Unknown
Other Delineations
Artificial
Unknown</td></tr>
<tr><td>Fringing Reef</td><td>Shoreline Intertidal
Reef Flat
Reef Crest
Fore Reef
Bank/Shelf
Bank/Shelf Escarpment</td></tr>
<tr><td>Non-Emergent Reef Crest</td><td>Shoreline Intertidal
Bank/Shelf (shallow)
Fore Reef
Bank/Shelf (deep)
Bank/Shelf Escarpment</td></tr>
</table>

*Geomorphologic structures are not unique to any reef type or reef geographic zone.

A priori habitat classifications incorporated into sampling designs have important consequences for assigning sites to BCG condition levels. Natural conditions for different reef types are composed of different coral species, depending on environmental features such as depth, photosynthetic active radiation (PAR), water flow, and geology, which influence conditions optimal for different coral species. Linear reefs (Kendall et al. 2001) in ≤ 12 m depth and ≤ 3 miles from shore were targeted in the sampling design for the data used by the experts in BCG development. Linear reefs are not identified in any of the reef types or geographic zones normally used, and dead geologic reef structure could not be discerned from live biologically active coral reefs since the linear reef designation was determined by ship sonar. Detailed information on the environmental features mentioned above is essential for formulating expectations about the benthic assemblages in a natural state, and it should be captured in future monitoring efforts.

Benthic Attributes for Coral Reefs

The benthic experts decided that the attributes developed for the Freshwater Streams BCG (Chapter 1, Table 1), especially relying on attributes I–V, did not apply very well to coral reefs benthic communities. The experts believed that the tolerances of most of the hard coral species probably varied based on the individual anthropogenic stressor; when multiple ones were combined the total effects could be additive, neutral or deleterious but largely unknown. The information and metrics the experts used to evaluate and rate benthic community included: the amount and quality of reef structure; size and density of massive reef-building coral species; the amount of live coral tissue on coral skeletons; coral colony density; percent colony mortality; rugosity; fish species richness; density of gorgonians, sponges, and *Diadema*; and density of "weedy coral species" such as *Porites astreoides* and *Siderastrea siderea*. The experts believed two critical metrics were missing which they required to confidently evaluate the sites: presence and absence of different calcareous and fleshy algae and the percent live coral cover as a planar measurement. They also stated the presence of coral or gorgonian diseases would be helpful in understanding the health and resiliency of the coral community. Sampling videos were critical for experts to derive the information necessary to evaluate benthic condition using the BCG approach. Experts recommended that more emphasis should be directed toward improving the quality of and protocols for the underwater videos. The BCG evaluation rules employed more benthic assemblages than just hard corals or sessile benthic invertebrates. The benthic experts included habitat structure, other benthic assemblages, and algae. Narrative rules were developed that reflected the expert judgment on critical elements and processes recommended for evaluation of coral community condition (Table B3-4).

Table B3-4. Benthic BCG Narrative Rules. Note: quantitative rules have not yet been developed.

BCG Level 1–2 (minimally disturbed)	
Stony corals	> 45% live cover of coral in fore reef habitat Minimal recent mortality in large reef-building genera (*Orbicella, Pseudodiploria, Colpophyllia, Acropora, Dendrogyra*) Normal frequency distribution of colony sizes within each species size range to include large, medium, and small colonies (≥ 4 cm) and presence of recruits (≤ 4 cm) Species composition and diversity composed of sensitive, rare species (*Isophyllia, Isophyllastrea, Mycetophyllia, Eusmilia, Scolymia*) present in appropriate habitat type Very low or just background levels of disease, tissue and skeletal anomalies, and bleaching *Orbicella* (fore reef), *Acropora* (back reef, reef crest) colonies dominant reef structure within respective zones
Rugosity	High rugosity resulting from large living coral colonies, producing spatial and topographical complexity
Macroinvertebrates	*Diadema* abundant Reef macroinvertebrates (e.g., Lobsters, crabs) common and abundant Low levels of invertebrate coral predators (*Coralliophylia spp., Hermodice spp.*)
Algae	Minimal fleshy, filamentous, and cyanobacterial algae present Crustose coralline algae present, with some turf algae
Sponges	Phototrophic sponges dominate Low frequency of Clionid boring sponges
BCG Level 3	
Stony corals	> 25% live cover of coral in appropriate habitat Higher percentage of tissue loss with signs of recent mortality especially on large reef-building genera (*Orbicella, Pseudodiploria, Colpophyllia, Acropora, Dendrogyra*) Frequency distribution of colony sizes within each species size range starting to become skewed to include fewer medium and small colonies (≥ 4 cm) and lower number of recruits than expected (≤ 4 cm) Species composition and diversity: sensitive, rare species present in appropriate habitat Low to moderate levels of disease and bleaching *Orbicella* and *Acropora* colonies still dominant (within respective reef geomorphological zones)
Rugosity	Moderate to high rugosity or reef structure resulting from large living reef-forming and dead coral colonies, producing spatial complexity (or topographical heterogeneity)
Macroinvertebrates	*Diadema* present Reef macroinvertebrates (e.g., Lobsters, crabs) present
Algae	Minimal presence of fleshy, filamentous, and cyanobacterial algae cover Crustose coralline and turf algae present
Sponges	Phototrophic sponges present Low cover and abundance of Clionid boring sponges

BCG Level 4	
Stony corals	> 15% live cover of coral in appropriate habitat Moderate amount of recent mortality on reef-building genera (*Orbicella, Pseudodiploria, Acropora, Dendrogyra*) Mix of sizes: large colonies may be absent, primarily medium and small colonies; low amount of recruits Species composition and diversity: sensitive species may be absent (*Agaricia, Mycetophyllia, Colpophyllia*, etc.), more tolerant spp present (*Montastraea cavernosa, Siderastrea siderea, Porites astreoides*; at least some reef-building corals present but not dominant primarily *Orbicella*) Moderate to high levels of disease and potential bleaching on corals and sea fans/branching gorgonians
Rugosity	Usually lower rugosity due to old mostly dead coral structure
Macroinvertebrates	*Palythoa* may be present, but not dominant
Algae	Moderate to high amount of fleshy, filamentous, and cyanobacterial algae cover
Sponges	Moderate cover and abundance of Clionid boring sponges
BCG Level 5	
Stony corals	> 1% live cover of coral in appropriate habitat High tissue mortality on organisms present. Low amount of live tissue remains on mostly small colonies
Rugosity	Low rugosity comprised of mostly dead and eroded coral structure
Algae	Coral cover replaced by fleshy, filamentous, and cyanobacterial algae
Macroinvertebrates	*Palythoa* dominant
Sponges	Highest presence of Clionid boring sponges Non-phototrophic sponges dominant

Before proceeding to further develop BCG decision rules for assigning sites to BCG levels, the benthic experts identified several technical issues to address. A brief explanation of these issues follows, including summarizing expert comments and recommendations.

Correlations between Benthic Metrics and BCG Condition Levels

Exploration of correlative relationships between coral, macroinvertebrate, rugosity, sponge and gorgonian metrics with different environmental parameters and BCG condition levels could reveal more narrative and quantitative rules. Relationships revealed through this exercise need to be evaluated for biological relevancy and strength of association.

Characterizing Full Range of Conditions

Preferably, data used in the development of decision rules for defining BCG condition levels 1–6 should range from pristine or natural coral reef condition to severely degraded. Experts agreed that the natural or BCG level 1 probably does not exist in Puerto Rico, and if it does it would be in one specific reef system, Mona Island.[22] About 70% of the fore reef sites evaluated for the coral reef BCG calibration were in fair to poor condition or comparable to levels 4 and 5. There were about 30% of sites assigned to BCG level 3. No sites were assigned to BCG levels 1, 2, or 6. To anchor the BCG model in natural condition, the experts discussed what data would represent reference sites. Suggestions were made to use historical data or records from the 1970s or earlier to develop qualitative and quantitative descriptions for BCG level 1.

[22] Mona Island is a protected natural reserve of Puerto Rico located in the western part of the main island within the Mona Channel.

Additional Metrics

The benthic experts strongly recommended additional metrics be provided to adequately evaluate the condition of coral reef sites. The experts identified the absence of total percent live cover of coral by species for each site to be a critical information gap. They found population estimates of density and percent live tissue useful, but percent of live coral cover has been used over the last four decades and has been adopted as the standard. Experts indicated that using a metric for percent live coral cover was more intuitive for them to express their professional judgment. The expert panel also discussed the value of information on the health of coral and gorgonian colonies (e.g., prevalence of disease, bleaching, predation), an estimate of coral recruitment, a more robust estimate of rugosity, and observed mortality classified as whether it occurred recently or years ago. They wanted better characterization of the benthic environment, advising that it should include additional assemblages and not just corals, gorgonians, and sponges. They were adamant that algae (sub-classified as crustose coralline, fleshy, filamentous, or cyanobacteria), zoanthids, seagrass, and the type of bare substrate be incorporated quantitatively in any future monitoring programs. These data could lead to metrics more relevant for developing standardized rules to discern different BCG levels.

Additionally, the benthic experts urged monitoring water quality (temperature, pH, turbidity, chlorophyll a, nutrients, DO, etc.), sediment, and other significant environmental conditions at long-term fixed stations measured at the depth of coral communities, not just at the surface. These measurements could improve understanding of the responses of coral communities to different anthropogenic stressors and aid in developing the GSA.

Defining Attribute VII: Organism Condition

Organism condition is very important for maintaining coral reef integrity because increased levels of coral diseases and bleaching have been identified as major responses to anthropogenic disturbance, contributing to the decline of coral reefs. Stressors impacting corals and reef structure can range from local, regional, or global in scale (e.g., sea temperature anomalies, nutrient enrichment, sedimentation, sewage, herbicides, pesticides, and coastal acidification). Many coral reef assessments evaluate diseases, coral bleaching, amount and condition of tissue, and recent vs. old mortality, so there are data available for attribute VII.

In the second workshop, the coral experts recommended developing criteria for BCG attribute VII. Of particular importance would be: the amount of live coral/gorgonian tissue found on colonies as compared to the exposed bare skeleton, recent vs. old mortality, condition of tissue, and population demographics including size frequency distribution and recruitment. Organism anomalies in the BCG vary from naturally occurring incidence in levels 1 and 2 to higher-than-expected incidence in levels 3 and 4. In levels 5 and 6, colony tissue is reduced, the age structure of populations indicates premature mortality or unsuccessful reproduction, and the incidence of serious anomalies is high. As such, scleractinian corals' sessile nature makes them good candidates for assessing the impacts of exposure to anthropogenic stressors. A cadre of methods are amenable to exploring impacts of past exposures such as using sclerochronology to examine skeletal bands, a method that is analogous to examining tree rings.

B3.3.3 Bringing the Two Expert Assemblage Groups Together

The two expert groups (fish and benthos) worked independently of each other, to develop individual BCG models focused on their individual assemblages. During the third workgroup meeting the two sets of experts met together to share information on their progress and to evaluate fore reef sites together using both the narrative rules (benthic) (Table B3-4) and quantitative fish rules (Table B3-2). This was an exercise to attempt to understand how the emerging BCG models and decisions rules for the two assemblages could be combined.

Discussion outcomes and recommendations include:

- Experts noted that there was often not congruence between fish and benthic expert ratings. Fish communities are influenced by location (distance from shore), connectivity, physical structure of the reef, and fishing pressure. Some species are also strongly influenced by the characteristics of the living coral community—for food source and/or shelter. Benthic communities are influenced by reef type and history, depth, PAR, water quality, past geology, and other factors.
- The fish experts were able to predict fish species, size, abundance, and trophic level based on the habitat location and type, or to predict the habitat type based upon the fish species found at a site. The benthic experts had only a few condition predictors, which resulted in imprecise BCG rankings. The benthic experts used more intuition, gestalt, video evidence, and extensive knowledge of historical conditions to assign sites to BCG levels.
- In the multi-assemblage group it became clear to the coral experts that the fish experts had a process model upon which they heavily relied to assign sites to BCG levels. The benthic experts expressed their need to develop an analogous benthic model to the one fish experts used. All experts agreed that an analysis of time series data sets for coral and other important benthic species in the U.S. Caribbean could be employed to develop this model.
- Algae could serve as an early response signal to habitat degradation and would be an important component for determining thresholds or tipping points of BCG levels for coral benthic community assessments.
- The topographical complexity of the coral reef structure and substrate was considered in evaluation of site condition. Most of the experts recommended using more recently developed methods for rugosity, which use multiple heights (Dustan et al. 2013), instead of the older linear chain approach. Additional observations on the quality and quantity of specific substrate types could serve as an indicator for potential recruitment of corals, gorgonians, and other benthic organisms. For example, if the substrate is covered with diatoms or scuzzy filamentous algae, it is untenable for recruitment.
- The experts identified a fate and transport model for land-based stressors in the U.S. Caribbean as a critical need, which would require coordination among multiple agencies. Water chemistry and physical properties were not generally available in the data set used for site evaluation by the experts. However, even if available, those data might have been a poor representation of conditions tolerated by the fish community because of their movement and water dynamics. Fate and transport models would allow characterization of general stressor conditions at each sampling site. Sessile corals could prove to provide an excellent record of past and present exposures to different anthropogenic stressors as revealed in their skeletons. Once the model was calibrated in Puerto Rico, it could be applied elsewhere.

B3.4 Enhancing Monitoring Designs and Protocols

The experts identified gaps in available coral reef monitoring and assessment data sets that, if addressed, would enable more comprehensive and robust assessment of coral reef condition and support development of a more complete BCG model. The experts discussed the elements of a monitoring program that would ensure relevant metrics could be provided at different temporal and spatial scales for assessing coral reef conditions. They also discussed the need for a sampling protocol starting with a basic design (including measurements essential to obtain the basic required information), expanding to multiple tiers with increasing complexity and many more measurements to produce more detailed metrics. The tiered approach would allow selection of an appropriate assessment methodology ranging from a simple monitoring protocol for screening purposes to a more comprehensive protocol, dependent upon available funding, time, study objectives, and available personnel.

The experts recommended that the basic sampling unit, assessment unit, and replication approach be identified for coral reef surveys for screening or assessing general condition of reefs within a study area. The experts unanimously recommended that a probabilistic sampling design was appropriate for this purpose. The benthic group further recommended a stratified design by reef type, selected randomly from a sampling frame that excludes areas devoid of living coral reefs. It remains unresolved whether the most appropriate approach would be to use increased replication of shorter transects (10 m) or fewer transect replicates of longer length (25 m); this could be resolved by carefully reviewing the objectives of the study. The experts also considered that more stringent requirements be developed for setting up transects, with protocols defining how to select the transect placement and direction relative to the shore, as well as how to anchor lines down so they cannot move between the video documentation and the surveyors assessment. The experts believed the observed placement of the transect in the videos was not always representative of the typical reef habitat found at that site, or the data they expected to see while assigning a BCG level.

B3.5 Conclusion

Work continues on development of a quantitative decision model for the fore reef zone in Puerto Rico and will include application of the approach to other well-defined habitats (e.g., deep fore reef/escarpment with coral reef coverage). As part of the process, EPA is developing a GSA for Caribbean coral reef systems that includes land-based sources of pollution, fishing pressure, and global climate change associated temperature anomalies (see Chapter 5). EPA and U.S. Geological Survey are collaborating on a framework for attribute VII (organism condition) as it could be applied in a coral reef BCG, and the fish experts will continue to develop attribute X (ecosystem connectance). It is anticipated that at least one more workshop and a series of webinars will be needed to complete, test, and calibrate the quantitative decision model for the fore reef zone. Additionally, descriptions of BCG levels 1 and 2 will be formulated based on historic data and records, as well as evaluation of additional data sets provided by expert panel.

Additionally, EPA has developed a database that documents tolerance levels for 39 species of corals and 131 species of fish to different stressors, including thresholds, when known. The database was populated with environmental condition information and relevant citations of species' sensitivity/tolerance to stressors from the Encyclopedia of Life and Web of Science websites. The cited literature was organized into a species sensitivity library using EndNote. The plan is to expand the database to other assemblages (e.g., macroinvertebrates, reptiles, seagrasses, sponges, mangroves, etc.).

Appendix B3 References

Adams, A.J., C.P. Dahlgren, G.T. Kellison, M.S. Kendall, C.A. Layman, J.A. Ley, I. Nagelkerken, and J.E. Serafy. 2006. Nursery function of tropical back-reef systems. *Marine Ecology Progress Series* 318:287–301.

Adey, W., and R.B. Burke. 1976. Holocene bioherms (algal ridges and bank barrier reefs) of the eastern Caribbean. *Geological Society of America Bulletin* 87:95–109.

Aguilar-Perera, A., and R.S. Appeldoorn. 2007. Variation in juvenile fish density along the mangrove-seagrass-coral reef continuum in SW Puerto Rico. *Marine Ecology Progress Series* 348:139–148.

Ballantine, D., R. Appeldoorn, P. Yoshioka, E. Weil, R. Armstrong, J. Garcia, E. Otero, F. Pagan, C. Sherman, E. Hernandez-Delgado, A. Bruckner, and C. Lilyestrom. 2008. Biology and Ecology of Puerto Rican Coral Reefs. In *Coral Reefs of the USA,* ed. *B.M.* Riegl and R.E. Dodge, pp. 375–406. Springer, New York.

Beck, M.W., K.L. Heck Jr., K.W. Able, D.L. Childers, D.B. Eggleston, B.M. Gillanders, B. Halpern, C.G. Hays, K. Hoshino, T.J. Minello, R.J. Orth, P.F. Sheridan, and M.P. Weinstein. 2001. The identification, conservation, and management of estuarine and marine nurseries for fish and invertebrates. *BioScience* 51:633–641.

Bohnsack, J.A., and S.P. Bannerot. 1986. *A Stationary Visual Census Technique for Quantitatively Assessing Community Structure of Coral Reef Fishes*. NMFS Technical Report 41. National Oceanic and Atmospheric Association, National Marine Fisheries Service. Seattle, WA. 15 pp

Bradley, P., D.L. Santavy, and J. Gerritsen. 2014. *Workshop on Biological Integrity of Coral Reefs, August 21–22, 2012, Caribbean Coral Reef Institute, Isla Magueyes, La Parguera, Puerto Rico*. EPA/600/R-13/350. U.S. Environmental Protection Agency, Office of Research and Development, Atlantic Ecology Division, Narragansett, RI.

Brown, B.E. 1997. Coral bleaching: Causes and consequences. *Coral Reefs* 16:S129–S138.

Brown, B.E., and Suharsono. 1990. Damage and recovery of coral reefs affected by El Niño related seawater warming in the Thousand Islands, Indonesia. *Coral Reefs* 8:153.

Brown, M.T., and M.B. Vivas. 2005. Landscape development intensity index. *Environmental Monitoring and Assessment* 101:289–309.

Christensen, J.D., C.F.G. Jeffrey, C. Caldow, M.E. Monaco, M.S. Kendall, and R.S. Appeldoorn. 2003. Cross-shelf habitat utilization patterns of reef fishes in southwestern Puerto Rico. *Gulf and Caribbean Research* 14(2):9–27.

Costa, B.M., M.S. Kendall, K. Edwards, G. Kagesten, and T.A. Battista. 2013. *Benthic Habitats of Fish Bay, Coral Bay and the St. Thomas East End Reserve*. NOAA Technical Memorandum NOS NCCOS 175. Prepared by the NCCOS Center for Coastal Monitoring and Assessment Biogeography Branch, Silver Spring, MD.

Costa, B.M., L.J. Bauer, T.A. Battista, P.W. Mueller, and M.E. Monaco. 2009. *Moderate-Depth Benthic Habitats of St. John, U.S. Virgin Islands.* NOAA Technical Memorandum NOS NCCOS 105. NCCOS Center for Coastal Monitoring and Assessment Biogeography Branch, Silver Spring, MD.

Dahlgren, C.P., T. Kellison, A.J. Adams, B.M. Gillanders, M.S. Kendall, J.A. Ley, I. Nagelkerken, and J.E. Serafy. 2006. Marine nurseries and effective juvenile habitats: Concepts and applications. *Marine Ecology Progress Series* 312:291–295.

Darwin, C. 1874. *The Structure and Distribution of Coral Reefs*. 2nd ed. Smith Elder and Company, London, UK.

Dustan, P., O. Doherty, and S. Pardede. 2013. Digital reef rugosity estimates coral reef habitat complexity. *PlosOne* 8(2):e57386.

García-Sais, J., R. Appeldoorn, T. Battista, L. Bauer, A. Bruckner, C. Caldow, L. Carrubba, J. Corredor, E. Diaz, C. Lilyestrom, G. García-Moliner, E. Hernández-Delgado, C. Menza, J. Morell, A. Pait, J. Sabater, E. Weil, E. Williams, and S. Williams. 2008. The State of Coral Reef Ecosystems of Puerto Rico. In *The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States: 2008*. ed. J.E. Waddell and A.M. Clarke. NOAA Technical Memorandum NOS NCCOS 73.

Gattuso, J.P., and R.W. Buddemeier. 2000. Ocean biogeochemistry: Calcification and CO_2. *Nature* 407:311–313.

Glynn, P.W. 1991. Coral reef bleaching in the 1980s and possible connections with global warming. *Trends in Ecology and Evolution* 6:175–179.

Glynn, P. W. 2000. El Niño-Southern Oscillation Mass Mortalities of Reef Corals: A Model of High Temperature Marine Extinctions? In *Carbonate Platform Systems: Components and Interactions*, ed. E. Insalaco, P. W. Skelton, and T. J. Palmer, pp. 117-133. The Geological Society of London, London, UK.

Goenega, C., V.P. Vicente, and R.A. Armstrong. 1989. Bleaching induced mortalities in reef corals from La Parguera, Puerto Rico: A precursor of change in the community structure of coral reefs? *Caribbean Journal of Science* 25:59–65.

Goreau T.J., R.H. Hayes, J.W. Clark, D.J. Basta, and C.N. Robertson. 1992. Elevated Sea Surface Temperatures Correlate with Caribbean Coral Reef Bleaching. Chapter 9 in *A Global Warming Forum: Scientific, Economic, and Legal Overview,* ed. R.A. Geyer, pp. 225–255. CRC Press, Boca Raton, FL.

Goreau, T.J., and R.M. Hayes. 1994. Coral bleaching and ocean 'hot spots'. *Ambio* 23:176-180.

Goreau T., T. McClanahan, R. Hayes, and A. Strong. 2000. Conservation of coral reefs after the 1998 global bleaching event. *Conservation Biology* 14:5-15.

Hay, M.E. 1984. Patterns of fish and urchin grazing on Caribbean coral reefs: Are previous results typical? *Ecology* 65:446–454.

Hay, M.E. 1991. Fish-Seaweed Interactions on Coral Reefs: Effects of Herbivorous Fishes and Adaptations of Their Prey. In *The Ecology of Coral Reef Fishes*, ed. P.F. Sale, pp. 96–119. Academic Press, New York.

Hoegh-Guldberg, O. 1999. Climate change, coral bleaching and the future of the world's coral reefs. *Marine and Freshwater Research* 50:839–866.

Hoegh-Guldberg, O., P.J. Mumby, A.J. Hooten, R.S. Steneck, P. Greenfield, E. Gomez, C.D. Harvell, P.F. Sale, A.J. Edwards, K. Caldeira, N. Knowlton, C.M. Eakin, R. Iglesias-Prieto, N. Muthiga, R.H. Bradbury, A. Dubi, and M.E. Hatziolos. 2007. Coral reefs under rapid climate change and ocean acidification. *Science* 318:1737–1742.

Hubbard, D.K. 1997. Reefs as dynamic systems. In: *Life and Death of Coral Reefs*. Birkeland, C. (*Ed.*). Chapman and Hall Publishing, New York, NY. pp. 43–67.

Hubbard, D., W. Ramirez, D. Cuevas, T. Erickson, and A. Estep. 2009. Holocene reef accretion along the north side of Bahia Enriquillo (western Dominican Republic): Unique insights into patterns of reef development in response to sea-level rise. *Proceedings of the 11th International Coral Reef Symposium* 1:43-47.

Hughes, T.P., A.H. Baird, D.R. Bellwood, M. Card, S.R. Connolly, and C. Folke. 2003. Climate change, human impacts, and the resilience of coral reefs. *Science* 301(5635):929.

Jackson, J.B.C. 1997. Reefs since Columbus. *Coral Reefs* 16:S23–S32.

Kendall, M.S., C.R. Kruer, K.R. Buja, J.D. Christensen, M. Finkbeiner, R.A. Warner, and M.E. Monaco. 2001. *Methods Used to Map the Benthic Habitats of Puerto Rico and the U.S. Virgin Islands*. NOAA Technical Memorandum NOS NCCOS CCMA 152. Silver Spring, MD.

Kleypas, J.A., R.W. Buddemeier, D. Archer, J.P. Gattuso, C. Langdon, and B.N. Opdyke. 1999. Geochemical consequences of increased atmospheric carbon dioxide on coral reefs. *Science* 284:118–120.

Knowlton, N., J.C. Lang, and B.D. Keller. 1990. Case study of natural population collapse: post-hurricane predation on Jamaican staghorn corals. *Smithson Contributions to Marine Science* 31:1–25.

Knowlton, N., and J.B.C. Jackson. 2008. Shifting baselines, local impacts, and global change on coral reefs. *PLoS Biology* 6(2):e54.

Liu, G., C.M. Eakin, J.L. Rauenzahn, T.R.L. Christensen, S.F. Heron, J. Li, W. Skirving, A.E. Strong, and T. Burgess. 2012. NOAA Coral Reef Watch's Decision Support System for Coral Reef Management. In *Proceedings of the 12th International Coral Reef Symposium*, Cairns, Australia, July 9–13, 2012, pp.

McField, M., and P.R. Kramer. 2007. *Healthy Reefs for Healthy People: A Guide to Indicators of Reef Health and Social Well-being in the Mesoamerican Reef Region*. Healthy Reefs for Healthy People Initiative. 208 pp. http://www.gstcouncil.org/images/library/Healthy_Reef_24_complete-1.pdf. Accessed February 2016.

Meynecke, J.O., S.Y. Lee, and N.C. Duke. 2008. Linking spatial metrics and fish catch reveals the importance of coastal wetland connectivity to inshore fisheries in Queensland, Australia. *Biological Conservation* 141:981–996.

Mora, C. 2008. A clear human footprint in the coral reefs of the Caribbean. *Proceedings of the Royal Society of London* B. 275:767–773.

Mumby, P.J., A.J. Edwards, J.E. Arias-Gonzalez, K.C. Lindeman, P.G. Blackwell, A. Gall, M.I. Gorczynska, A.R. Harborne, C.L. Pescod, H. Renken, C.C. Wabnitz, and G. Llewellyn. 2004. Mangroves enhance the biomass of coral reef fish communities in the Caribbean. *Nature* 427:533–536.

Mumby, P.J., K. Broad, D.R. Brumbaugh, C.P. Dahlgren, A.R. Harborne, A. Hastings, K.E. Holmes, C.V. Kappel, F. Micheli, and J.N. Sanchirico. 2008. Coral reef habitats as surrogates of species, ecological functions, and ecosystem services. *Conservation Biology* 22:941–951.

NOAA. 2014. National Coastal Population Report: Population Trends from 1970 to 2020. http://oceanservice.noaa.gov/facts/population.html. Accessed August 17, 2016.

NOAA. 2016. *NOAA Coral Reef Watch, Experimental Products, Thermal Histor*y. http://coralreefwatch.noaa.gov/satellite/product_exp.php. Accessed February 2016.

Oliver, L.M., J.C. Lehrter, and W.S. Fisher. 2011. Relating landscape development intensity to coral reef condition in the watersheds of St. Croix, U.S. Virgin Islands. *Marine Ecology Progress Series* 427:293–302.

Orr, J.C., V.J. Fabry, O. Aumont, L. Bopp, S.C. Doney, R.A. Feely, A. Gnanadesikan, N. Gruber, A. Ishida, F. Joos, R.M. Key, K. Lindsay, E. Maier-Reimer, R. Matear, P. Monfray, A. Mouchet, G. Raymond, R.G. Najjar, G.K. Plattner, K.B. Rodgers, C.L. Sabine, J.L. Sarmiento, R. Schlitzer, R.D. Slater, I.J. Totterdell, M.F. Weirig, Y. Yamanaka, and A. Yool. 2005. Anthropogenic ocean acidification over the twenty-first century and its impact on calcifying organisms. *Nature* 437:681–686.

Pandolfi, J. M., R. H. Bradbury, E. Sala, T. P. Hughes, K. A. Bjorndal, R. G. Cooke, D. McArdle, L. McClenachan, M. J. H. Newman, G. Paredes, R. R. Warner, and J. B. C. Jackson. 2003. Global trajectories of the long-term decline of coral reef ecosystems. *Science* 301:955-958.

Pauly, D. 1995. Anecdotes and the shifting base-line syndrome of fisheries. *Trends in Ecology and Evolution* 10:430.

Pittman, S.J., G.F. Renchen, R. Clark, C. Caldow, D. Olsen, and R.L. Hill. 2011. Applications of estuarine and coastal applications in marine spatial planning. *Treatise on Estuarine and Coastal Science* 1:163–205.

Reaser, J.K., R. Pomerance, and P.O. Thomas. 2000. Coral bleaching and global climate change: Scientific findings and policy recommendations. *Conservation Biology* 14:1500–1511.

Ruiz Valentine, I. 2013. *Development and Testing of a Spatial Tool for the Assessment Anthropogenic Threats in Coastal and Marine Habitats in Puerto Rico*. Ph.D. thesis, University of Puerto Rico, Mayaguez, PR.

Scavia, D., J.C. Field, D.F. Boesch, R.W. Buddemeier, V. Burkett, D.R. Cayan, M. Fogarty, M.A. Harwell, R.W. Howarth, C. Mason, D.J. Reed, T.C. Royer, A.H. Sallenger, G. James, and J.G. Titus. 2002. Climate change impacts on U.S. coastal and marine ecosystems. *Estuaries* 25(2):149–164.

Sale, P.F., P. Jacob, and J.P. Kritzer. 2008. Connectivity: What it is, how it is measured, and why it is important for management of reef fishes. In: R. Grober-Dunsmore and B.D. Keller (*Eds*.). *Caribbean connectivity: Implications for marine protected area management*. Proceedings of a Special Symposium, 9–11 November 2006, 59th Annual Meeting of the Gulf and Caribbean Fisheries Institute, Belize City, Belize. Marine Sanctuaries Conservation Series ONMS-08-07. Silver Spring (MD): U.S. Department of Commerce, National Oceanic and Atmospheric Administration, Office of National Marine Sanctuaries, pp. 16–30.

Sale, P.F., H. Van Lavieren, M.C. Ablan Lagman, J. Atema, M. Butler, C. Fauvelot, J.D. Hogan, G.P. Jones, K.C. Lindeman, C.B. Paris, R. Steneck, and H.L. Stewart. 2010. *Preserving Reef Connectivity: A Handbook for Marine Protected Area Managers*. Connectivity Working Group, Coral Reef Targeted Research & Capacity Building for Management Program, UNU-INWEH.

Sheppard, C.R.C. 2003. Predicted recurrences of mass mortality in the Indian Ocean. *Nature* 425:294–297.

Steneck, R.S., J. Vavrinec, and A.V. Leland. 2004. Trophic-level dysfunction in kelp forest ecosystems of the Western North Atlantic. *Ecosystems* 7:523–552.

The White House. 2004. U.S. Ocean Action Plan: The Bush Administration's Response to the U.S. Commission on Ocean Policy. The White House, Washington, DC.

Wellington, G.M., P.W. Glynn, A.E. Strong, S.A. Navarrete, E. Wieters, and D. Hubbard D. 2001. Crisis on coral reefs linked to climate change. *EOS, Transactions of the American Geophysical Union* 82:1–6.

Wilkinson, C.R. 2000. World-wide coral reef bleaching and mortality during 1998: A Global Change Warning for the New Millennium? In *Seas of the Millennium: An Environmental Evaluation, Volume III, Global Issues and Processes*, ed. C. Sheppard, pp. 43–57. Elsevier, Amsterdam, The Netherlands.

Wilkinson, C. 2004. *Status of Coral Reefs of the World: 2004*. 2 volumes. Global Coral Reef Monitoring Network & Australian Institute of Marine Science, Townsville, AU.

Wilkinson C., O. Linden, H. Cesar, G. Hodgson, J. Rubens, and A.E. Strong. 1999. Ecological and socio-economic impacts of 1998 coral mortality in the Indian Ocean: An ENSO impact and a warning of future change? *AMBIO* 28:188-196.

Williams, S. M. P. J. Mumby, I. Chollett, and J. Cortés. 2015. Importance of differentiating Orbicella reefs from gorgonian plains for ecological assessments of Caribbean reefs. *Marine Ecology Progress Series* 530:93–101.

Zitello, A.G., L.J. Bauer, T.A. Battista, P.W. Mueller, M.S. Kendall, and M.E. Monaco. 2009. *Shallow-Water Benthic Habitats of St. John, U.S. Virgin Islands*. NOAA Technical Memorandum NOS NCCOS 96. Silver Spring, MD.

Appendix B3 Additional Information: Species Attribute Assignments Made during the Biological Condition Gradient Workshop

Table B3-5. Fish Species Attribute Assignments made during the BCG Workshop. Fish observed during EPA's 2010 and 2011 surveys in Puerto Rico were used for assignments to attributes II–VI. Since no attribute I species were observed during EPA's 2010 and 2011 surveys, fish experts brainstormed a list based on species known to be found in Puerto Rico.

Scientific Name	Common Name	Frequency (% of Samples)
I—Historically documented, sensitive, long-lived, or regionally endemic taxa		
Epinephelus itajara	Atlantic Goliath Grouper	Not Observed
Mycteroperca tigris	Tiger Grouper	Not Observed
Mycteroperca bonaci	Black Grouper	Not Observed
Epinephelus striatus	Nassau Grouper	Not Observed
Mycteroperca venenosa	Yellowfin Grouper	Not Observed
Lutjanus cyanopterus	Cubera Snapper	Not Observed
Scarus coelestinus	Midnight Parrotfish	Not Observed
Scarus coeruleus	Blue Parrotfish	Not Observed
Ginglymostoma cirratum	Nurse Shark	Not Observed
Carcharhinus perezii	Caribbean Reef Shark	Not Observed
Negaprion brevirostris	Lemon Shark	Not Observed
Sphyrna mokarran	Great Hammerhead Shark	Not Observed
Galeocerdo cuvier	Tiger Shark	Not Observed
Aetobatus narinari	Spotted Eagle Ray	Not Observed
Dasyatis americana	Southern Stingray	Not Observed
Lactophrys triqueter	Smooth Trunkfish	Not Observed
Acanthostracion polygonia	Honeycomb Cowfish	Not Observed
Lactophrys trigonus	Trunkfish	Not Observed
Acanthostracion quadricomis	Scrawled Cowfish	Not Observed
Caranx bartholomaei	Yellow Jack	Not Observed
II—Highly sensitive taxa		
Acanthurus coeruleus	Blue Tang	62%
Amblycirrhitus pinos	Red Spotted Hawkfish	0.7%
Anisotremus surinamensis	Black Margate	2.9%
Aulostomus maculatus	Trumpet Fish	8.6%
Cantherhines pullus	Orange Spotted Filefish	1.4%
Chaetodon sedentarius	Reef Butterflyfish	0.7%
Chromis cyanea	Blue Chromis	1.4%
Chromis multilineata	Brown Chromis	5.7%
Clepticus parrae	Creole Wrasse	0.7%
Elacatinus genie	Cleaner Goby	2.9%
Elacatinus oceanops	Neon Goby	0.7%
Gramma loreto	Fairy Baselet	1.4%
Haemulon chrysargyreum	Small-mouthed Grunt	2.9%
Heteropriacanthus cruentatus	Glasseye Snapper	0.7%
Holacanthus ciliaris	Queen Angelfish	0.7%
Malacoctenus triangulatus	Saddled Blenny	4.3%
Melichthys niger	Black Durgon	0.7%
Scarus guacamaia	Rainbow Parrotfish	0.7%
Scomberomorus regalis	Cero	1.4%
Serranus tigrinus	Harlequin Bass	12.9%
III—Intermediate sensitive taxa		
Acanthurus chirurgus	Doctorfish	38.6%
Alphestes afer	Mutton Hamlet	0.7%
Balistes vetula	Queen Triggerfish	7.1%

Scientific Name	Common Name	Frequency (% of Samples)
Bodianus pulchellus	Spotfin Hogfish	0.7%
Bodianus rufus	Spanish Hogfish	10%
Cephalopholis cruentata	Graysby	4.3%
Chaetodon capistratus	Foureye Butterflyfish	54.2%
Chaetodon striatus	Banded Butterflyfish	22.1%
Epinephelus adscensionis	Rock Hind	4.3%
Epinephelus guttatus	Red Hind	5%
Equetus punctatus	Spotted Drum	1.4%
Haemulon carbonarium	Caesar Grunt	15%
Haemulon macrostomum	Spanish Grunt	5%
Haemulon sciurus	Blue-striped Grunt	2.9%
Halichoeres garnoti	Yellowhead Wrasse	21.4%
Halichoeres maculipinna	Clown Wrasse	33.6%
Halichoeres radiatus	Pudding Wife	16.4%
Hypoplectrus aberrans	Yellowbelly Hamlet	0.7%
Hypoplectrus chlorurus	Yellowtail Hamlet	7.9%
Hypoplectrus indigo	Indigo Hamlet	1.4%
Hypoplectrus nigricans	Black Hamlet	0.7%
Hypoplectrus puella	Barred Hamlet	8.6%
Hypoplectrus randallorum	Tan Hamlet	1.4%
Hypoplectrus unicolor	Butter Hamlet	2.9%
Kyphosus sectator	Bermuda Sea Chubb	1.4%
Lutjanus apodus	Schoolmaster	29.3%
Lutjanus jocu	Dog Snapper	0.7%
Lutjanus mahogoni	Mahagony Snapper	5.7%
Microspathodon chrysurus	Yellowtail Damselfish	53.6%
Mulloidichthys martinicus	Yellow Goatfish	6.4%
Myripristis jacobus	Blackbar Soldierfish	11.4%
Odontoscion dentex	Reef Croaker	4.3%
Pareques acuminatus	Highhat	2.1%
Pempheris schomburgkii	Glassy Sweeper	2.9%
Pomacanthus arcuatus	Gray Angelfish	2.9%
Pomacanthus paru	French Angelfish	12.1%
Scarus taeniopterus	Princess Parrotfish	7.1%
Sparisoma chrysopterum	Redtail Parrotfish	2.1%
Sparisoma viride	Stoplight Parrotfish	59.3%
Sphyraena barracuda	Great Barracuda	1.4%
Stegastes planifrons	Threespot Damselfish	17.1%
Stegastes variabilis	Cocoa Damselfish	4.3%
Thalassoma bifasciatum	Bluehead Wrasse	86.4%
IV—Intermediate tolerant taxa		
Abudefduf saxatilis	Sergeant Major	21.4%
Acanthurus bahianus	Ocean Surgeonfish	70%
Anisotremus virginicus	Porkfish	18.6%
Canthigaster rostrata	Sharpnose Puffer	25%
Cephalopholis fulva	Coney	5.7%
Coryphopterus glaucofraenum	Bridled Goby	5.7%
Diodon hystrix	Porcupine Fish	0.7%
Gymnothorax moringa	Spotted Moray Eel	2.1%
Haemulon aurolineatum	Tomtate	3.6%
Haemulon flavolineatum	French Grunt	37.1%
Haemulon plumierii	White Grunt	6.4%
Holocentrus adscensionis	Squirrelfish	25.7%
Holocentrus rufus	Longspine Squirrelfish	10.7%
Lachnolaimus maximus	Hogfish	1.4%

Scientific Name	Common Name	Frequency (% of Samples)
Lutjanus analis	Mutton Snapper	2.1%
Lutjanus griseus	Gray Snapper	0.7%
Lutjanus synagris	Lane Snapper	5%
Monacanthus	Filefish	0.7%
Ocyurus chrysurus	Yellowtail Snapper	59.3%
Ophioblennius macclurei	Red Lipped Blenny	18.6%
Pseudupeneus maculatus	Spotted Goatfish	9.3%
Sargocentron vexillarium	Dusky Squirrelfish	1.4%
Scarus iseri	Striped Parrotfish	73.6%
Sparisoma aurofrenatum	Redband Parrotfish	85.7%
Sparisoma rubripinne	Yellowtail Parrotfish	21.4%
Stegastes adustus	Dusky Damselfish	42.1%
Stegastes diencaeus	Longfin Damselfish	41.4%
Stegastes leucostictus	Beaugregory	38.6%
Stegastes partitus	Bicolor Damselfish	66.4%
V—Tolerant taxa		
	Checkered Puffer	
Synodus foetens	Inshore Lizardfish	1.4%
VI—Non-native or intentionally introduced species		
Callogobius clitellus	Saddled Goby	0.7%
Pterois	Lionfish	1.4%
N—Taxa not assigned to an attribute		
Calamus calamus	Saucereye Porgy	1.4%
Carangoides ruber	Bar Jack	19.3%
Caranx crysos	Blue Runner	5.7%
Chaenopsis ocellata	Bluethroat Pickle Blenny	4.3%
Emblemariopsis	Dark Headed Blenny	0.7%
Eucinostomus gula	Silver Jenny	0.7%
Gerres cinereus	Yellowfin Mojarra	1.4%
Gobiidae	Gobies	0.7%
Gymnothorax	Common Moray Eel	2.1%
Haemulon melanurum	Cottonwick	0.7%
Haemulon parra	Sailors Choice	1.4%
Haemulon striatum	Striped Grunt	0.7%
Halichoeres bivittatus	Slippery Dick	54.3%
Halichoeres cyanocephalus	Yellowcheek Wrasse	0.7%
Halichoeres poeyi	Blackear Wrasse	0.7%
Holacanthus bermudensis	Blue Angelfish	0.7%
Malacanthus plumieri	Sand Tilefish	1.4%
Rypticus saponaceus	Greater soapfish	2.1%
Serranus baldwini	Lantern Bass	2.1%
Sphyraena borealis	Northern Sennett	0.7%
Stephanolepis hispida	Planehead Filefish	0.7%
Synodus intermedius	Sand Diver	2.1%

Table B3-6. Coral species assignments to BCG attributes by professional judgment of coral reef experts. Sediment tolerance was used as a surrogate for landscape stressors and elevated heat tolerance as a proxy for climate change stressors. The expected density at single site (distribution within a site) and frequency of occurrence (distribution among sites) were ranked from low to high.

BCG Attribute	% Stations Present[1]	Scientific Name	Sediment Tolerance	Heat Tolerance	Expected Density at Single Site	Expected Frequency of Occurrence
II	5.0	*Isophyllastrea rigida*	2 [2]	2 [2]	low	low
II	11.4	*Isophyllia sinuosa*	2 [2]	2 [2]	low	low
III	15.0	*Acropora cervicornis*	3	3	low	med
III	2.1	*Agaricia lamarcki*	3	2	med - low	med
III	17.9	*Colpophyllia natans*	3	3	med	med
III	6.4	*Dendrogyra cylindricus*	3 4	3	low	med
III	7.9	*Diploria labyrinthiformis*	3	3	med	hi
III	0.7	*Eusmilia fastigiata*	3	3	low	low
III	0.0	*Helioseris cucullata*	3	3	low	low
III	4.3	*Madracis decactis*	2 4	4	low	med
III	13.6	*Millepora complanata*	3	2	hi	low[3]
IV	17.1	*Acropora palmata*	4	3	med	med low
IV	0.0	*Acropora prolifera*	4	3	low	low
IV	35.0	*Agaricia agaricites*	4	2	med - hi	hi
IV	16.4	*Agaricia humilis*	4	2	med - hi	hi
IV	0.0	*Cladocora arbuscula*	4	4	low	low
IV	9.3	*Dichocoenia stokesi*	4	3	low	med
IV	0.7	*Madracis myriaster*	4	3	low	low
IV	0.0	*Meandrina jacksoni*	4	3	low	med
IV	15.7	*Meandrina meandrites*	4	3	low	med
IV	0.0	*Mussa angulosa*	4	2	low	low
IV	4.3	*Mycetophyllia aliciae*	4	3	low	med
IV	2.9	*Mycetophyllia ferox*	4	2 - 3	med	hi
IV	14.3	*Orbicella annularis*	4	2	med	hi
IV	52.1	*Orbicella faveolata*	4	2	hi	hi
IV	3.6	*Orbicella franksi*	4	2	low	hi
IV	10.0	*Porites furcata*	4	4 - 5	low	med
IV	26.4	*Porites porites*	4	4	med	hi
IV	0.0	*Scolymia cubensis*	4	4	low	low
IV	0.0	*Scolymia lacera*	4	4	low	low
V	0.0	*Favia fragum*	5	4	med	hi
V	0.0	*Manicina areolata*	5	5	low	low
V	57.1	*Millepora alcicornis*	5	2	med	hi
V	64.3	*Montastraea cavernosa*	5	4 - 5	med	hi
V	0.0	*Oculina diffusa*	5	4	low	low
V	91.4	*Porites astreoides*	5	5	hi	hi
V	7.1	*Porites divaricata*	5	4	med	low
V	40.0	*Pseudodiploria clivosa*	5	4	hi	hi
V	77.9	*Pseudodiploria strigosa*	5	4	hi	hi
V	3.6	*Siderastrea radians*	5	5	med	hi
V	92.9	*Siderastrea siderea*	5	4	med	hi
V	1.4	*Solenastrea bournoni*	5	4	low	low
V	31.4	*Stephanocoenia intersepta*	5	4	med	low
VI	0.0	*Tubastrea coccinea*				

BCG Attribute	% Stations Present[1]	Scientific Name	Sediment Tolerance	Heat Tolerance	Expected Density at Single Site	Expected Frequency of Occurrence
x	21.4	*|UNKNOWN|*				
x	9.3	*Agaricia fragilis*				
x	0.0	*Millepora squarrosa*		2	low	low
x	0.7	*Mycetophyllia daniana*			deep	
x	4.3	*Mycetophyllia lamarckiana*			deep	

[1] Total of 140 stations
[2] Only about 50% experts expressed an opinion
[3] Limited to shallow depths

B4. New England: Using the Biological Condition Gradient and Fish Index of Biotic Integrity to Assess Fish Assemblage Condition in Large Rivers

Chris Yoder, Research Director, Midwest Biodiversity Institute, Columbus, Ohio[23]

This case study examines the correspondence of BCG levels and attributes with a fish IBI developed for large, non-wadeable rivers in New England (termed "riverine IBI" hereafter) based on work conducted in 2002 through 2009. The principal objective was the development of a BCG-based fish IBI that could be used to assess and readily communicate the status of New England rivers. Intended applications include determining the existing status and quality of individual river reaches and the effectiveness of management efforts aimed at protecting and restoring native fish assemblages including diadromous species. Using the BCG to better visualize the "as naturally occurs" riverine fish assemblage proved essential to developing expectations for BCG levels 1 and 2, and describing the incremental changes predicted in BCG levels 3 through 6. The riverine IBI that was developed during 2002–2007 in Maine (Yoder et al. 2008) , and further tested and applied throughout New England in 2008–2009 (Yoder et al. 2015), served as the quantitative scale of measurement along the BCG.

The riverine fish fauna of New England has a unique make-up due to its comparative isolation from drainages to the west and north (Curry 2007). A narrative BCG for fish assemblages for New England was developed on the basis of expert judgment and historical knowledge of pre-settlement conditions. Numeric thresholds are proposed based on alignment of BCG attributes with the metrics from the riverine IBI. The resulting BCG-based IBI model can be used to communicate aquatic life condition (fish) in New England Rivers and, based on historic knowledge, describe the fish assemblage expected in an "as naturally occurs" condition. Thus, a site is assigned to a BCG level based on its IBI score.

The riverine IBI and the attendant BCG were initially developed for the cool-coldwater, moderate-high gradient riverine ecotype as it is the most common type throughout New England. Many New England rivers also support several diadromous fish species that comprised a significant ecological and commercial aspect of riverine fish assemblages, at least historically. To better address this important component of the BCG, a supplemental set of IBI metrics were developed to specifically measure the diadromous component of the fish assemblage. This was done for two purposes: (1) to use the diadromous metrics (termed "diadromous metrics" hereafter) as a distinct indicator of whether a river is supporting a fishery on the basis of unobstructed access to freshwater (anadromous) or salt water (catadromous) for spawning, and (2) to retain the function of the riverine IBI for assessing rivers where diadromous species are not expected to occur. Common diadromous fish include sea run Atlantic salmon (Salmo salar), Atlantic sturgeon (Acipenser oxyrhinchus), shortnose sturgeon (Acipenser brevirostrum), American eel (Anguilla rostrata), sea lamprey (Petromyzon marinus), striped bass (Morone saxatilis), and three species of Clupeidae known as river herring.[24] Historically these species comprised a significant component of a New England riverine fish assemblage, but their numbers have been significantly reduced since the mid to late 19th century. Restoration efforts are currently widespread and are focused on improving upstream and downstream passage for diadromous fish. The

[23] CYoder@MWBinst.com

[24] In New England, river herring include alewife (Alosa pseudoharengus), blueback herring (Alosa aestivalis), and American shad (Alosa sapidissima).

development of a BCG-based riverine IBI provides the opportunity to quantitatively determine how these restoration efforts affect the whole fish assemblage.

B4.1 Background

A systematic approach to the assessment of fish assemblages in the large, non-wadeable rivers of New England was initiated in 2002 for the purpose of developing a large rivers fish IBI. Data collection occurred first in Maine through 2007 and was then extended into the remainder of New England in 2008–2009 as part of a Regional EMAP (REMAP) project. The aggregate effort produced an extensive and detailed region-wide coverage of riverine fish assemblages and habitat (Figure B4-1). The resulting database was sufficient to develop and test a riverine IBI. This project also paralleled the early development and piloting of the BCG by EPA (Davies and Jackson 2006). An expert advisory panel was formed to provide advice about New England riverine fish assemblages and evaluate the correspondence between BCG attributes and riverine IBI metrics. This included using the BCG to describe the composition of the "as naturally occurs" riverine fish assemblages for the cool-coldwater, moderate-high gradient riverine ecotype that prevailed throughout most of New England prior to the extensive modification of rivers in the 18th and 19th centuries.

B4.2 Riverine Index of Biotic Integrity Development—Summary of Key Tasks

A systematic and tractable methodology for sampling riverine fish in New England did not exist when this project was conceived in 2001. As such, methodological issues had to be addressed first and then followed by the organization of species' autecological information, both of which are essential steps in the complementary development of a BCG and IBI. These were followed by the more traditional tasks of selecting and testing candidate IBI metrics, selecting final metrics, calibrating the metrics, and testing the riverine IBI across a gradient of conditions ranging from reference to highly impacted. The following are the major tasks that were accomplished starting with the initial efforts in Maine and then expanding to rivers throughout New England (referencing the project documents that deal with each):

1) Developing an effective and systematically employed sampling methodology (Yoder et al. 2006a) with first phase of development occurring for Maine rivers with applicability across New England taken into account;
2) Establishing a sufficient spatial and temporal database in Maine, then testing throughout New England (Yoder et al. 2006b, 2008);
3) Describing the autecology of the extant fish fauna to support deriving and testing candidate metrics in Maine but application considered across New England (Yoder et al. 2006b, 2008);
4) Differentiating major lotic ecotypes in Maine and New England (Yoder et al. 2006b, 2008);
5) Visualizing the expected New England fish assemblages along the BCG (Yoder et al. 2008);
6) Establishing reference condition for Maine and application throughout New England (Yoder et al. 2008);
7) Deriving a fish IBI for the moderate-high gradient riverine ecotype, first in Maine and then applied throughout New England (Yoder et al. 2008); and,
8) Testing the BCG and IBI initially developed for Maine with data sets that represent a range of conditions and stressors across New England (Yoder et al. 2015).

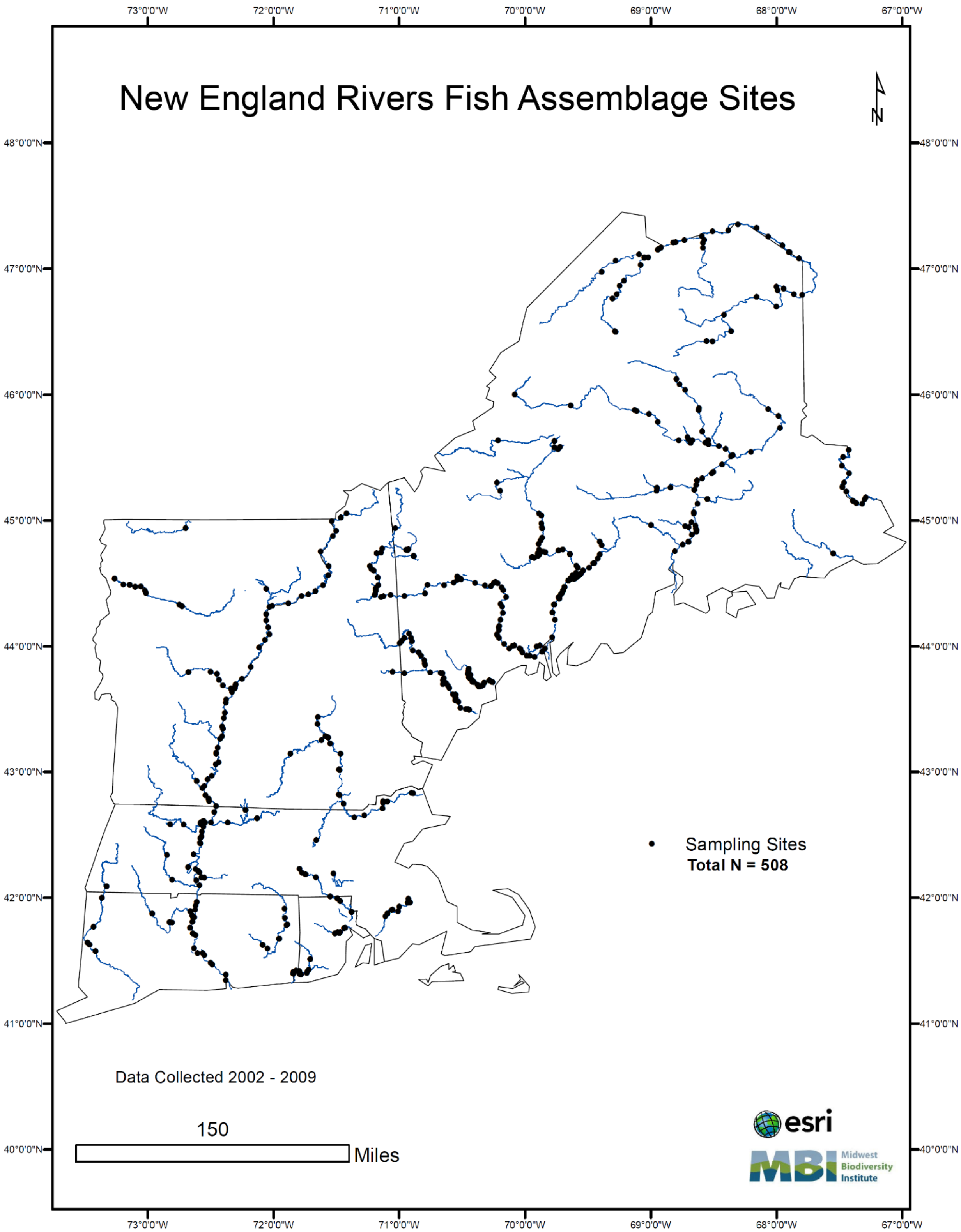


Figure B4-1. Locations of fish sampling in the non-wadeable rivers of New England for the Maine rivers and New England REMAP fish assemblage assessment projects, 2002–2009.

The IBI development process generally followed that described by Hughes et al. (1998), which has guided many leading examples in North America and elsewhere (see summary by Yoder and Kulik 2003), and Mebane et al. (2003). The tasks related to IBI derivation and testing and the BCG are summarized in the order in which they were accomplished.

B4.2.1 Task 1. Development of an Effective Sampling Methodology

A tractable sampling methodology did not exist for riverine fish assemblages in Maine when this project was first proposed. The lack of an approach was likely due to the status of rivers as "working rivers" that supported hydroelectric power production and the transport of logs, the latter of which rendered most rivers as inaccessible for fish sampling (log driving was discontinued in 1975). The approaches developed for sampling large rivers in the Midwestern U.S. in the 1970s and 1980s were applied and modified accordingly to suit the prevailing conditions in Maine. A perceived obstacle was the comparatively low conductivity of the water, which threatened to make electrofishing less effective. However, this was overcome by making adjustments to the equipment during an initial testing phase in 2002. Other aspects of the methodology were also established at this time (Yoder et al. 2006a).

B4.2.2 Task 2. Development of Sufficient Spatial and Temporal Database

The development of a database that is representative of the spatial and temporal aspects of riverine fish assemblages is an essential part of BCG and IBI development. The sampling conducted throughout Maine during 2002–2007 provided the database for developing the narrative BCG and numeric riverine IBI. The addition of data from rivers throughout New England in 2008–2009, coupled with the preceding years of sampling Maine rivers (Figure B4-1), provided a more complete stress gradient for testing the IBI and the BCG, which is illustrated by the specific assessment examples included herein.

B4.2.3 Task 3. Autecology of Extant New England Riverine Fish Fauna

Describing the autecology of the extant fish fauna is another essential step and includes information that is used to derive, select, and test candidate IBI metrics and to make BCG attribute assignments. Information about environmental tolerance, native status, habitat and flow preferences, thermal regime, foraging habitats, and reproductive habits were compiled[25] for 78 fish species known or suspected to occur in the non-wadeable rivers of New England Proper.[26] These classifications were compiled from a number of sources about native status, target fish classification, common riverine habitats where each species occurred, spatial occurrence in the New England region, thermal classification, environmental tolerance, foraging habits, reproductive habits, and predominant habitat residence. The most recent and geographically relevant sources, in combination with observations made during nine years of field studies, were used to make these assignments. This task fulfilled the breadth and type of information that Karr et al. (1986) described in the seminal guidance for developing fish IBIs. In addition to several new guilds that have appeared in contemporary IBIs of the past 10–15 years, a fluvial classification scheme based on the target fish community method of Bain and Meixler (2000, 2008) and a thermal classification scheme by Hokanson (1977) were used to better reflect attributes of the BCG for New England riverine fish assemblages.

---

[25] See http://www.midwestbiodiversityinst.org/reports/31/Maine%20Rivers%20IBI%20Appendix%20Table%20B4-6%2020160211.pdf. Accessed February 2016.

[26] New England Proper includes the rivers that drain directly to the Atlantic Ocean. It excludes the portions of the Lake Champlain, Hudson-Hoosic, St. Francois, and Lake Memphremagog drainages in Vermont and Massachusetts (David Halliwell, personal communication).

B4.2.4 Task 4. Differentiating Riverine Ecotypes

It was essential to determine naturally occurring and distinctive strata for classifying New England Rivers. The classification provides the basis for describing BCG levels 1 and 2 and predicting incremental changes from these conditions with increasing stress. In addition to commonly applied distinctions such as cold and warmwater assemblages, there are at least three riverine ecotypes in New England that are distinguished by baseline habitat characteristics and fish assemblage composition. These ecotypes are: moderate-high gradient riverine, low gradient riverine, and fresh-brackish water tidal habitats[27] (Yoder et al., 2008). Impounded habitats are viewed as a human-induced modification of moderate-high gradient riverine habitats, rather than as a distinct and naturally occurring ecotype. However, data from these modified habitats played an important role in testing the responsiveness of candidate metrics and the riverine IBI to human-made modifications of natural riverine habitat. Based on observations made in the field and in analyzing data for this and previous reports (Yoder et al. 2006a,b), there are distinctive differences in fish assemblage composition between moderate-high gradient riverine habitats and low gradient riverine habitats. The emphasis of this case study is on the moderate-high gradient riverine ecotype.

B4.2.5 Task 5. Expected Fish Assemblages along the Biological Condition Gradient

Developing an understanding of the natural fish assemblages that historically occurred in the non-wadeable rivers of New England is critical to determining their current status and potential for restoration. The BCG concept was employed for this task. Consistent with the BCG conceptual framework, the "as naturally occurs" fish assemblage represents the assemblage expected in an undisturbed/minimally disturbed condition for large, non-wadeable rivers in New England and corresponds with BCG levels 1 and 2. Restoring all New England rivers to such a condition may be impractical given the economically dependent activities and non-native species introductions that have substantially altered the fish assemblages in this region for more than two centuries. Nevertheless it is important to describe this condition because it serves as an essential anchor for the "upper levels" of the BCG and as an objective reference for assessing current conditions, providing more accurate understanding of what has been lost and, where possible, what can be restored. Description of the "as naturally occurs" fish fauna was based on historical observations by the first European settlers coupled with expert knowledge about how such assemblages were most likely organized based on current understanding of species autecology and distribution.[28]

In developing a BCG model for non-wadeable riverine fish assemblages, general information about the historical fish assemblages coupled with expert judgment was used in the process. This was accomplished through an expert advisory workgroup that included scientists from EPA, U.S. Fish and Wildlife Service, NOAA, State of Maine water quality and natural resource agencies, the Penobscot Indian Nation, and Trout Unlimited. One important outcome of the expert deliberations was the conclusion that the "as naturally occurs" fish assemblage in the moderate-high gradient riverine ecotype was largely comprised of native cool-coldwater species which are described as temperate stenotherms and mesotherms.[29] Based on these discussions and using the results of the initial sampling in Maine

[27] See http://www.midwestbiodiversityinst.org/reports/31/Maine%20Rivers%20IBI%20Appendix%20Table%20B4-6%2020160211.pdf. Accessed February 2016.

[28] See http://www.midwestbiodiversityinst.org/reports/31/Maine%20Rivers%20IBI%20Appendix%20Table%20B4-6%2020160211.pdf. Accessed February 2016.

[29] See http://www.midwestbiodiversityinst.org/reports/31/Maine%20Rivers%20IBI%20Appendix%20Table%20B4-6%2020160211.pdf. Accessed February 2016.

(Yoder et al. 2006a,b, 2008), the template for a BCG was developed for the cool-coldwater, moderate-high gradient riverine ecotype (Figure B4-2). This reflects a comparatively simple, qualitative method of visualizing what has happened in many instances to the "as naturally occurs" fish assemblage through time. Observed departures from level 1 attributes and characteristics are the result of historical modifications to water quality, habitat, flow regime, thermal regime, the native fauna via the widespread introduction of non-native species, and the loss of connectivity for diadromous species (Figure B4-2).

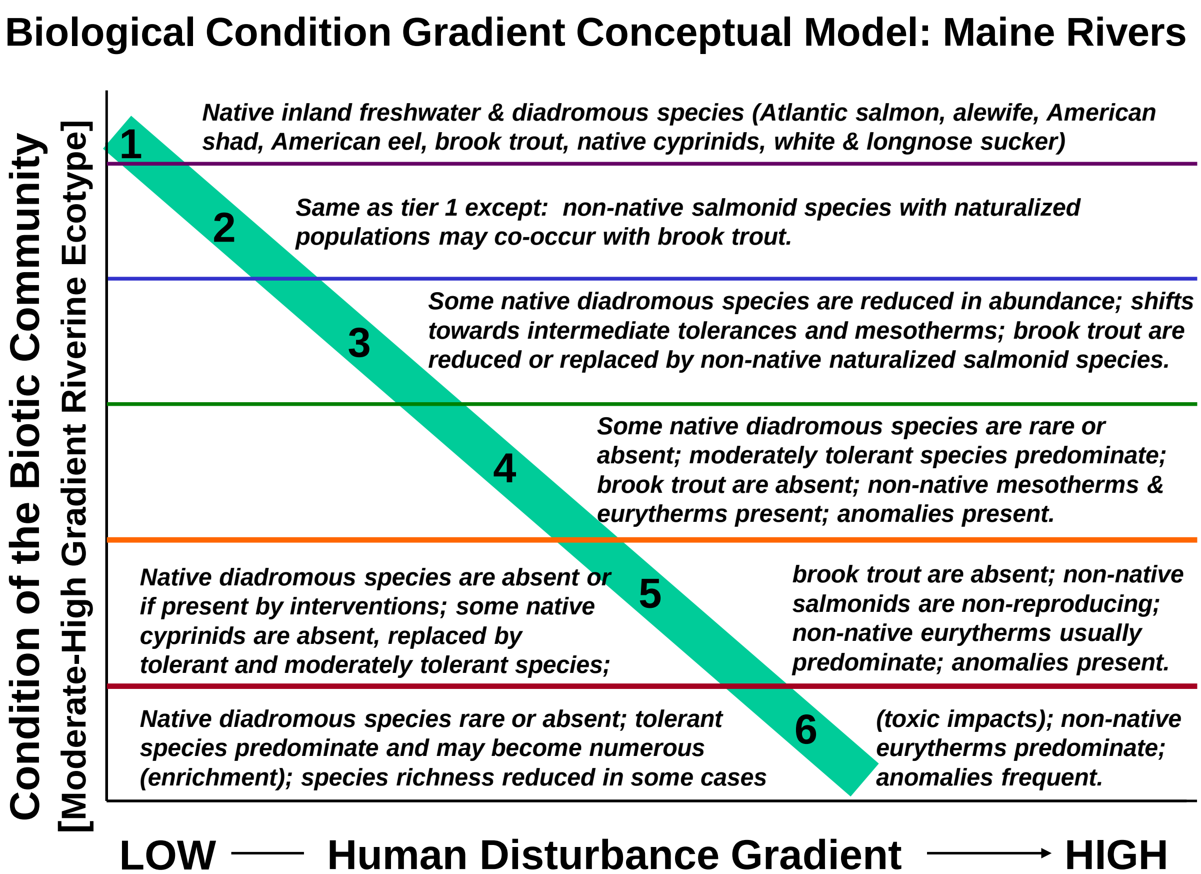


Figure B4-2. A BCG model for fish assemblages representative of cool-coldwater, moderate-high gradient riverine habitats in New England (after Yoder et al. 2008).

While the formal process of developing narrative and numeric decision rules through an expert driven fuzzy set modeling approach was not used, expert judgment played a role in defining the correspondence between BCG attributes and IBI metrics and informed deriving IBI thresholds for each BCG level. A more detailed description of the parallels and overlap between BCG attributes and metrics, and derivation of the riverine IBI and supplemental diadromous IBI (D-IBI) are described below.

B4.2.6 Task 6. Establishing Reference Condition Comparable to Biological Condition Gradient Levels 1 and 2

The techniques for screening and selecting reference sites has evolved significantly during the past 30 years from a mostly qualitative process first described by Hughes et al. (1986) and used by some of the pioneering developers of numeric biological criteria (e.g., Ohio EPA 1987; Barbour et al. 1996), to a more quantitative process (Stoddard et al. 2006) that is now used by EPA and an increasing number of states. How reference sites are selected and used to develop reference condition are essential components of how EPA evaluates the level or rigor of biological assessment programs (Yoder and Barbour 2009; USEPA 2013). While the majority of these efforts have focused on wadeable streams, there are now ample precedents for developing reference condition for large rivers (Hughes and Gammon 1987; Ohio EPA 1987; Lyons et al. 2001; Emery et al. 2003). The prevalence of legacy impacts in most non-wadeable rivers raises issues about the quality of the reference condition that contemporary sampling data represent. This is one reason why merging IBI development with the BCG framework is helpful. Ideally, reference condition is represented by undisturbed or minimally disturbed conditions, comparable to BCG levels 1 and 2. However, actually finding a contemporary example of BCG level 1, and sometimes level 2, has been elusive, especially in large rivers. BCG level 2 conditions have been observed in other ecological regions, though not present everywhere. Given this reality it then becomes important to understand how the relative states of "best" and "better" occur along the BCG within the domain of the riverine fish assemblage data across New England, so that the task of reconciling conceptual goals with societal realities can be dealt with more effectively. Articulating this framework now provides for a more accurate way of developing attainable thresholds later in the process.

Reference sites were selected using a combination of position in the landscape (with respect to point and nonpoint source stressors) and the intactness of the native fish fauna. The latter included selecting sites that lacked blackbasses (smallmouth and largemouth bass) and other non-native species based on knowledge about the negative impacts that these introductions have had on the native fish species that comprise the sensitive metrics of the riverine IBI (Whittier et al. 2000, 2001; Warner 2005; Yoder et al. 2008). Yoder et al. (2015) described the native status of the fish species that were either encountered in the 2002–2009 sampling or reasonably expected to have occurred in recent times.[30] The definitions of Halliwell (2005) were followed in describing the native status of fish species and in deriving candidate and final IBI metrics relative to native status. Hence, the presence of introduced species was a major factor in the selection and/or rejection of reference sites. New England rivers represent a unique situation in which all of the major river drainages are mostly contained within New England state boundaries, and all are coastal drainages discharging to Long Island Sound or the Gulf of Maine. As such, they have largely been isolated from adjacent drainages such as the St. Lawrence-Great Lakes and Hudson River drainages since post-glacial times (Curry 2007). This has influenced the character of the freshwater fish fauna with some species common to these adjacent drainages being historically absent. Examples are smallmouth and largemouth bass that are not indigenous to any New England river system, but which were introduced in the latter part of the 19th century becoming firmly established in several major river drainages to date (Warner 2005). A few select rivers in northern Maine have not yet been invaded by blackbasses or other introduced species, and these also tended to represent minimally impacted conditions in terms of landscape, habitat, thermal, and flow alterations. Hence, these were selected as the approximation of minimally disturbed reference for the derivation and testing of candidate IBI metrics. In addition to reflecting a minimum of anthropogenic chemical and physical impacts, they also reflected the absence of non-native species. Reference and a gradient of non-reference sites were selected to represent the full gradient of stress from BCG levels 1 and 2 (undisturbed/minimally disturbed conditions) to BCG level 6 (severely altered) as follows:

- "Minimally disturbed" reference sites lacking non-native species (BCG level 2).

[30] See http://www.midwestbiodiversityinst.org/reports/31/Maine%20Rivers%20IBI%20Appendix%20Table%20B4-6%2020160211.pdf. Accessed February 2016.

- Non-reference sites with conductivity ≥ 100 µS/cm.
- The remaining non-reference sites were partitioned by Qualitative Habitat Evaluation Index ranges: ≤ 50; 51–75; 76–90; > 90; this imparts a habitat gradient that reflects commonly occurring impacts throughout New England.

The resulting stress gradient was then used to evaluate the response of the candidate IBI metrics following the continuous calibration methodology of Mebane et al. (2003). This calibration method was first used in the Pacific Northwest, which has many similarities to the New England region, including depauperate cool-coldwater fish fauna impacted by similar stressors (i.e., thermal and flow alterations). A range of scores for each metric were defined for all 6 BCG levels based on the correspondence between the metric response and the narrative BCG level descriptions (Figure B4-2, Table B4-1). The riverine IBI and D-IBI were then derived based on the traditional IBI development approach, summing the scores for selected metrics into one index value (Yoder et al., 2008)).

Table B4-1. New England riverine IBI metrics with calibrated scoring equations and manual scoring adjustment criteria based on their initial application in Maine. Proportional (%) metrics are based on numbers unless indicated otherwise (for methods used to derive the scoring equations, see Yoder et al. 2008).

Metric	Scoring Equation	Scoring Adjustments	
		Score = 0	Score = 10
Native species richness	10 * (-0.2462 + (0.0828*numspec2)))	< 3 sp.	≥ 15 sp.
Native Cyprinid species (excluding fallfish)	(10 * (0.4457 + (0.0109*allcyp_ff) - (0.00005629 * (allcyp_ff ^{2}))))	Eq[31]	Eq
Adult white & longnose sucker abundance (biomass)	(10 * (0.3667 + (0.008*ws_lns_pb) - (0.000023592 * (ws_lns_pb^{2}))))	0	≥ 128 kg/km
%Native Salmonids	(10 * (0.9537 + (0.00000000039*nat_salm) - (0.000078892 * (nat_salm^{2}))))	0	≥ 20%
%Benthic Insectivores	10 * (0.010966*benth_pc_n)	0	≥ 91.2%
%Blackbass	10 - (10 * (-0.09684 + (0.5638*log10(blackbass))))	Eq	0
%Fluvial Specialist/Dependent	(10 * (0.2775 + (0.0073*fluv_pc_n)))	0%	Eq
%Macrohabitat Generalists	10 - (10 * (0.1017 + (0.0096*macro_gen)))	> 90%	Eq
Temperate stenothermic species	(10 * (0.7154 + (0.4047*(log10(steno)))))	0 sp.	> 5 sp.
Non-guarding lithophilic species	(10 * (0.2979 + (0.8975*log10(lith_ng))))	< 1	> 10
Non-indigenous species	10 - (10 * (0.1063 + (0.3271*Non-indigenous_sp) - (0.029*(Non-indigenous_sp^{2}))))	≥ 5	0
%DELT anomalies	10 - (10 * (0.8965 + (0.1074*log10(delta))))	Eq	0

B4.2.7 Task 7. Deriving an Index of Biotic Integrity for the Cool-Coldwater, Moderate-High Gradient Riverine Ecotype

The riverine IBI metrics that were initially tested and selected had to take into account two of the unique characteristics of New England fish fauna. First, there is an inherently low species richness, and, second, the fauna has been isolated from adjacent regions since post-glacial times (Curry 2007). As such, New England rivers naturally lack several species that are common to the same latitudes in adjacent

[31] No scoring adjustments are necessary; scoring determined by equation (Eq) across entire metric scoring range of 0–10.

drainages. Given that physical factors (e.g., flow regime, habitat, and thermal regime) and biological factors (e.g., native status) are important in this region, metrics that reflect those characteristics were derived and selected.

This task included two principal steps: (1) the selection and testing of candidate IBI metrics, and (2) the derivation of an IBI for the cool-coldwater, moderate-high gradient riverine ecotype. These tasks have ample and recent precedence in North America and elsewhere. A growing body of information is now available for non-wadeable rivers (Yoder and Kulik 2003), including the baseline factors common to New England (e.g., an appropriate thermal baseline for large rivers, metric testing and selection, and index development and testing (Hughes and Gammon 1987; Ohio EPA 1987; Lyons et al. 2001; Mebane et al. 2003; Emery et al. 2003)). Developing the metrics for the core riverine IBI involved sequential steps beginning with identifying candidate metrics, evaluating the responsiveness and relevance of those metrics along the BCG, and deriving indices comprised of the "best" set of metrics that represent the ecotype and other strata that are embedded within the process (Yoder et al. 2008).

The riverine IBI metrics and their calibration equations appear in Table B4-1. The riverine IBI is scored on a 0–100 scale making it amenable to scaling along the entirety of the BCG, rather than only a portion of the BCG as is common to the early IBIs that employed the ordinal calibration and 12–60 scale of Karr et al. (1986). Furthermore, the selection of riverine IBI metrics was based on emulating attributes of the BCG (Figure B4-2) that was developed prior to IBI development and calibration.

A supplemental set of four diadromous metrics were developed in 2011 to better reflect the diadromous component of the fish assemblage in rivers that have historically supported these species (Table B4-2). These supplement metrics are applied only where diadromous fish have historically been documented, thus it is not applied where natural barriers have historically prohibited their occurrence. It is added to the riverine IBI and the resulting index is termed the D-IBI.

Table B4-3 illustrates the match, or correspondence, between the metrics that comprise the riverine IBI and D-IBI and the BCG attributes. As discussed above, the first 12 metrics comprise the cool-coldwater, moderate-high gradient riverine fish IBI while the four supplemental diadromous metrics are specific to the diadromous part of the fish assemblage. A complete match between a metric and an attribute indicates that the species assigned to an IBI metric fits wholly within the definition of the BCG attribute. Three IBI metrics were complete in their match with a BCG attribute—the percentage of fish with deformities, erosion, lesions, and tumors (DELT) anomalies (attribute VII), the proportion of benthic insectivores (attribute VIII), and non-guarding lithophils (attribute VIII). A partial match indicates that an IBI metric includes species that occur in multiple BCG attributes. For example, the fluvial specialist/dependent IBI metric includes species that occur in BCG attributes II and III, thus making the correspondence of that metric for the two attributes partial. The supplemental diadromous metrics are a surrogate measure of BCG attribute X (ecosystem connectance), with some of the species also corresponding with attribute I (historically documented, sensitive, long-lived, or regionally endemic taxa.

B4.2.8 Task 8: Assessing New England Large Rivers

The BCG-based riverine IBI was used to assess the condition of large rivers across New England and also to determine regional scale, reach level, and site-specific stressors (Yoder et al. 2015). Different data sets were used for four different projects: a regional scale assessment, an intensive assessment of the Connecticut River mainstem, a comparison and ranking of major rivers using the riverine IBI and D-IBI. and a site-specific application. This projects were conducted to explore the utility of using a BCG-based index for large river biological assessments, with a focus on use of different data sets.

Regional Assessment of New England Large Rivers

The results of a regional scale analysis of condition by BCG level across New England is depicted in Table B4-4. Two types of data sets were used: a probabilistic regional data set (REMAP[32]) and a targeted sites data set. BCG level 2 sites were only identified in western and northern Maine (Figure B4-3; Table B4-4). These sites also showed a low incidence of stressors, intact habitat, and the absence of non-native species. The proportion of samples that reflected BCG levels 3 through 6 were not substantially different between the REMAP probabilistic and targeted results (< 5% difference) (Table B4-3).

The only difference between the two sampling designs was illustrated by the absence of BCG level 2 samples in the REMAP probabilistic data set for New England (Table B4-5). A total of 19 targeted sites in Table B4-5 had higher IBI scores than the highest scoring REMAP probability site and only 4 of the 27 highest scoring sites were REMAP probabilistic sites. Two of the 27 highest scoring REMAP sites occurred outside of Maine in the upper Connecticut River in northern New Hampshire. These are the highest quality sites and rivers in the New England region, and are potential candidates for additional protections.

Connecticut River Assessment

The Connecticut River mainstem was sampled in 2008 and 2009 from the Third Connecticut Lake in New Hampshire downstream to the "salt wedge" just upstream from I-95 in Connecticut. Probabilistic sites were selected from the 2008–2009 NRSA draw of sites for two levels of coverage with targeted sites added to fill in "gaps" to complete a longitudinal pollution survey design on the mainstem. Based on the BCG levels and the corresponding riverine IBI scores, an objective was to compare the estimates of condition between the two probabilistic sample draws (NRSA base and REMAP) and the intensive pollution survey design. Though the targeted design detected BCG level 2 conditions that the probabilistic design did not, the overall proportion of BCG levels for both monitoring designs was generally comparable (Table B4-4).

[32] Regional EMAP, USEPA.

Table B4-2. Supplemental diadromous metrics intended to represent the diadromous component of a riverine fish assemblage in New England (for methods used to derive the scoring equations, see Yoder et al. 2008)

Metric	Scoring Equation	Scoring Adjustments	
		Score = 0	Score = 10
Diadromous species richness	Score = 0.0318 + 0.227*(Diadromous Species Richness)	0	≥ 5 sp.
Number of American eel	Score = 0.0689 + 0.2*(Log Eel Rel. No.) + 0.0616*(Log Eel Rel. No.)	0	≥ 389/km
Number of Clupeidae	Score = 0.832*Log10(Rel. No. Clupeids)^ (0.269)	0	≥ 96/km
Number of diadromous fish (all diadromous species)	Score = 0.0522 + 0.168*(Log(Diad Rel. No.) + 0.0644^(Log(Diad Rel. No.))	0	≥ 560/km

Table B4-3. Comparison of BCG attributes with riverine fish IBI metrics (metrics 1–12) and four supplemental diadromous metrics (metrics 13–16) applicable to non-wadeable rivers in New England

	Riverine Fish IBI Metrics												Diadromous Metrics			
BCG Attribute	Indigenous Species	Fluvial Specialist/ Dependent	Temperate Stenotherms	%Native Salmonids	Adult White/Longnose Sucker Biomass	Native Cyprinids	%Macrohabitat Generalists	%Benthic Insectivores	Non-indigenous Species	% Black-**basses**	Non-guarding Lithophils	DELT Anomalies	*Diadromous Species*	*No. American Eel*	*No. Clupeidae*	*No. Diadromous Individuals*
I. Historically Documented, Sensitive, Long-lived, or Regionally Endemic Taxa	◖			◖	◖								◖	●		
II. Highly Sensitive Taxa		◖	●	●												
III. Intermediate Sensitive Taxa		◖				◖										
IV. Intermediate Tolerant Taxa					◖	◖	◖		◖	◖						
V. Tolerant Taxa						◖	●		◖							
VI. Non-native or Intentionally Introduced Species							◖		●	●						
VII. Organism Condition												●				
VIII. Ecosystem Function								●			●					
IX. Spatial and Temporal Extent of Detrimental Effects	●															●
X. Ecosystem Connectance				◖	◖				●				●	●	●	●

●- complete match between IBI metric and BCG attribute.
◖ - partial match between IBI metric and BCG attribute.

Table B4-4. The number and percentage of New England Large River (NELR) REMAP probabilistic and targeted samples arranged by corresponding BCG level for the riverine IBI

	NELR REMAP Probabilistic		NELR REMAP Targeted	
BCG Level	**Samples**	**Percent**	**Samples**	**Percent**
IBI				
Level 1: IBI ≥ 95	0	*0*	0	*0*
Level 2: IBI ≥ 80 and IBI < 95	0	*0*	12	*3.2*
Level 3: IBI ≥ 60 and IBI < 80	15	*10.1*	42	*11.3*
Level 4: IBI ≥ 40 and IBI < 60	48	*32.2*	127	*34.2*
Level 5: IBI ≥ 20 and IBI < 40	78	*52.3*	177	*47.7*
Level 6: IBI < 20	8	*5.4*	13	*3.5*
Totals	149	*100*	371	*100*

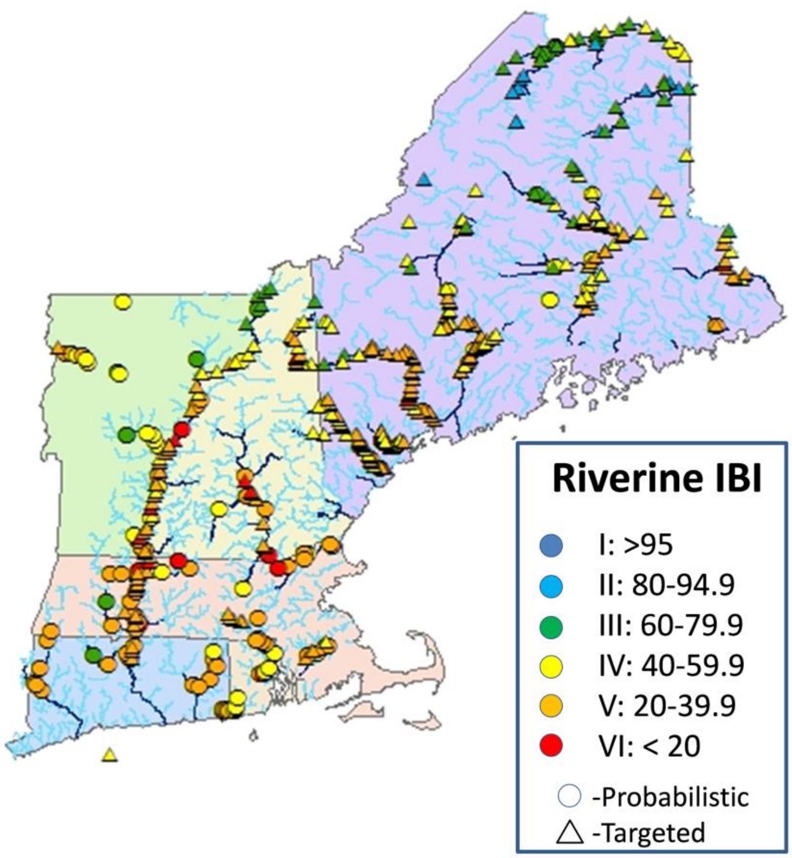


Figure B4-3. Map of non-wadeable fish sampling sites in New England with riverine IBI values colored coded by BCG levels 1–6.

Table B4-5. Riverine IBI and supplemental diadromous metrics and their BCG level equivalents with the method of estimation (e.g., regression equation, by eye). These were used in the mapping of the IBI metrics (after Yoder et al. 2015).

IBI Metric	Biological Condition Level						Equation/Method for BCG Cutoff Estimation
	BCG 1	BCG 2	BCG 3	BCG 4	BCG 5	BCG 6	
No. of Native Species	> 10	10	8–9	6–7	4–5	< 4	Numspec=1.83+0.018*IBI
No. Temperate Stenothermic Species	≥ 5	4	3	1–2	0	–	Stenotherms=-1.94+0.0713*IBI
No. of Non-Guarding Lithophilic Species	≥ 7	6	4–5	3	1–2	0	Lithophil NG=-1.36+0.0866*IBI
%. of Cyprinid Species*	≥ 58.2	> 47.3–58.2	> 32.8–47.3	> 18.3–32.8	> 3.8–18.3	≤ 3.8	% Cyprinids=-10.6+0.724*IBI
% Native Salmonids	≥ 4.20	> 3.22–4.20	> 1.91–3.22	> 0.59–1.91	0	–	% Nat. Salm.=-2.03+0.0656*IBI
% Benthic Insectivores	≥ 39.2	> 30–39.2	> 17.7–30.0	> 5.3–17.7	≤ 5.3	–	% Benth. Ins.=-19.3+0.616*IBI
% Black Bass	–	0	> 0–9.2	> 9.2–19.3	> 19.3–29.4	≥ 29.4	% Blackbass=39.5-0.505*IBI
% Fluvial Specialists and Dependents	≥ 96.8	> 86.3–96.8	> 68.7–86.3	> 43.9–68.7	> 1.4–43.9	≤ 1.4	% Fluvial Specialists= -182+141*log(IBI)
% Macrohabitat Generalists	≤ 0.6	> 0.6–9.4	> 9.4–24.2	> 24.2–45.0	> 45.0–80.5	> 80.5	%Macrohab. Gen.=234-118*log(IBI)
Adult White, Longnose Sucker Biomass	≥ 63.4	> 52.8–63.5	> 38.7–63.5	> 24.6–38.7	> 10.5–24.6	≤ 10.5	White, LN Sucker=-3.62+0.705*IBI
Non-Indigenous Species	0	1	2	3	4	≥ 5	By eye
% DELT Anomalies	0	> 0–0.30	> 0.30–0.50	> 0.50–1.0	> 1.0–2.0	> 2.0	Threshold by eye
Log American Eel Number/km	≥ 2.5	> 2.0–2.5	> 1.5–2.0	> 1.0–1.5	> 0.5–1.0	≤ 0.50	By eye
Log Diadromous Number/km	≥ 2.5	> 2.0–2.5	> 1.5–2.0	> 1.0–1.5	> 0.5–1.0	≤ 0.50	By eye
+Log Clupeid Number/km	≥ 2.5	> 2.0–2.5	> 1.5–2.0	> 1.0–1.5	> 0.5–1.0	≤ 0.50	By eye
Diadromous Species Richness	5	4	3	2	1	0	By eye

*excludes fallfish.

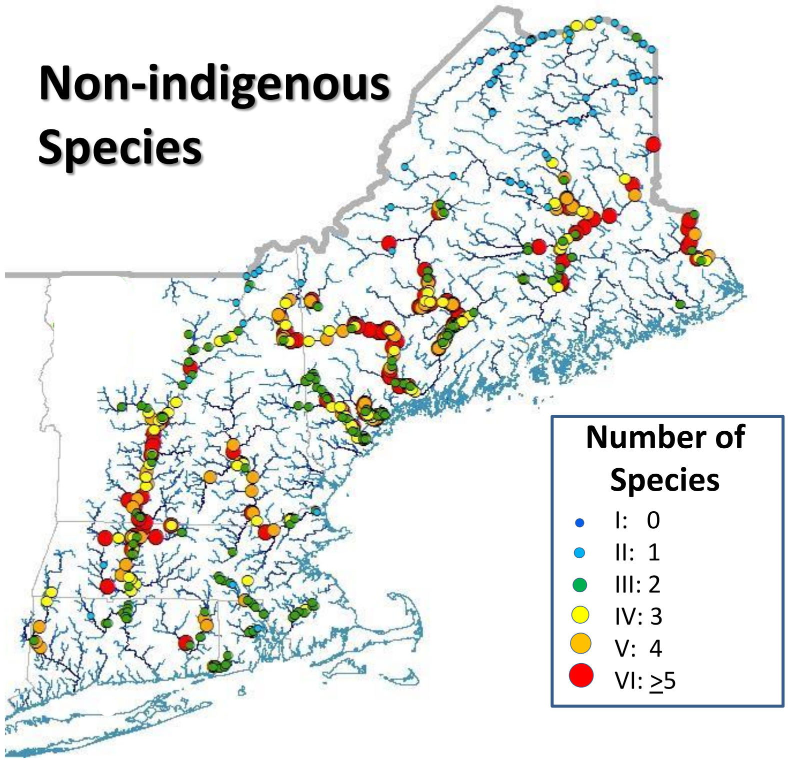
Correlations between Index of Biotic Integrity Metrics and Their Biological Condition Gradient Equivalents

Spatial patterns in riverine IBI metrics and their corresponding BCG level assignments across New England were examined as part of the process for indexing the IBI to the BCG. Maps of each IBI metric were color coded to its equivalent BCG level as depicted in Table B4-5. This allowed for the visualization of general patterns across New England and also to highlight site- or river reach-specific issues that might otherwise be obscured by regionally focused analyses and which could warrant more detailed follow up investigations. Two representative metrics are included in Figure B4-4 and each exposes gradients related to the degree of disturbance from the major stressors identified by Yoder et al. (2015). The non-native species metric, corresponding to BCG attribute VI (presence of non-native taxa), shows the extent of introductions in central and southern New England and their virtual absence in northern Maine (Figure B4-4), the latter of which is due primarily to both natural and artificial barriers to their ingress. Some non-native introductions such as smallmouth and largemouth bass have had deleterious effects on native Cyprinids and other indigenous species in both lakes (Whittier et al. 2000, 2001) and rivers (Yoder et al. 2008). At some locations more than five non-native species were collected, and these occurred in river reaches that are most impacted by hydrological alterations and chemical pollution. As such, the extent of non-native species introductions represent one of the major negative influences on the condition of the native New England riverine fish fauna.

The distribution and occurrence of temperate stenotherms (which are all native species) roughly mirrors that of non-native species (Figure B4-4). While this metric represents typically cool-coldwater fish species, it also represents BCG attributes II and III (sensitive and moderately sensitive taxa) since this metric is comprised of species that cannot tolerate significant alterations to the natural thermal regime, habitat, and/or flow regime. This same pattern generally held for the other riverine IBI metrics that are predicted to decrease with increasing stress (e.g., native species richness, % native Salmonids, and fluvial specialist/dependent species). The pattern exhibited by non-native species generally held for the metrics predicted to increase with increasing stress such as % blackbasses and macrohabitat generalists (Yoder et al. 2015).

Comparing New England Mainstem River Reaches along the Biological Condition Gradient

Sufficient data were available to make a comparative assessment of the status of 36 individual rivers across New England which was one of the primary objectives of the project (Figure B4-5). The riverine IBI reflects the status of the resident freshwater assemblage. The supplemental diadromous metrics highlight the ability of each individual river and/or river reach to support a diadromous fish assemblage. As discussed earlier, the diadromous metrics can be interpreted as an inferred measure of ecosystem connectance (e.g., free from fish passage barriers) since diadromous fish rely upon free-access to fresh and marine habitats to complete their natural life cycles. Box-and-whisker plots were used to display and rank each river by the 75th percentile value of the summed riverine IBI and diadromous metrics (i.e., the D-IBI) and by shading the boxes with the corresponding BCG color level based on the median (50th percentile) D-IBI value. Ranking each river by the 75th percentile of D-IBI values reflects the protection and/or restoration potential of each river or reach. The rankings were done according to the D-IBI which best incorporates all of the BCG attributes and the current quality of each river while revealing when the riverine IBI and/or D-IBI exhibit markedly different results. It also shows where the freshwater part of the assemblage is positioned along the BCG relative to the status of the diadromous part of the assemblage. The restoration of the diadromous part of the assemblage will potentially benefit the freshwater fish assemblage in coastal rivers—indirectly due to restoration of riverine habitat if dams are removed and directly through the influx of marine nutrients with diadromous fish (Saunders et al. 2006).



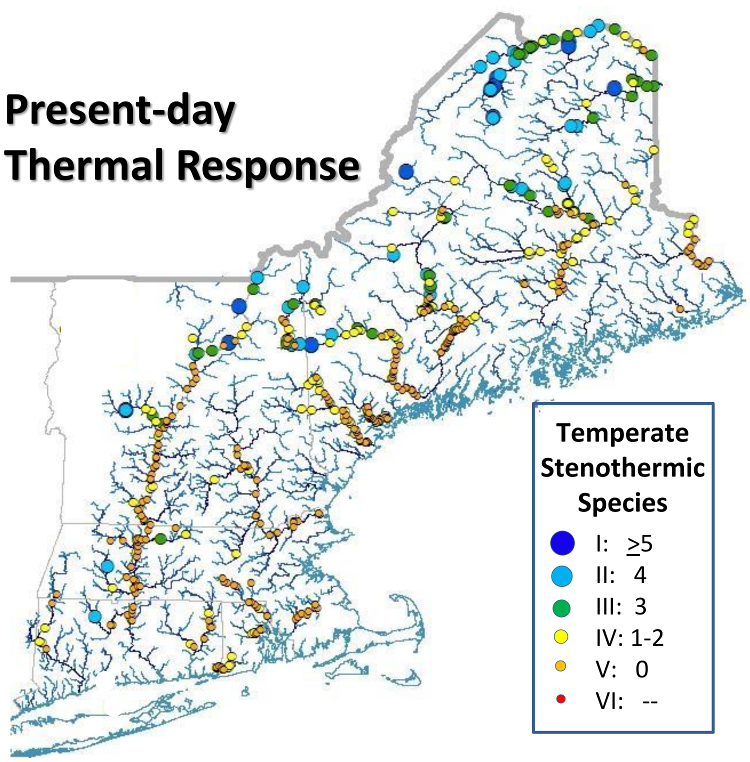


Figure B4-4. Number of non-indigenous (upper) and temperate stenothermic species (lower) at New England large river sites with symbols coded by the IBI metric value that corresponds to BCG levels 1–6.

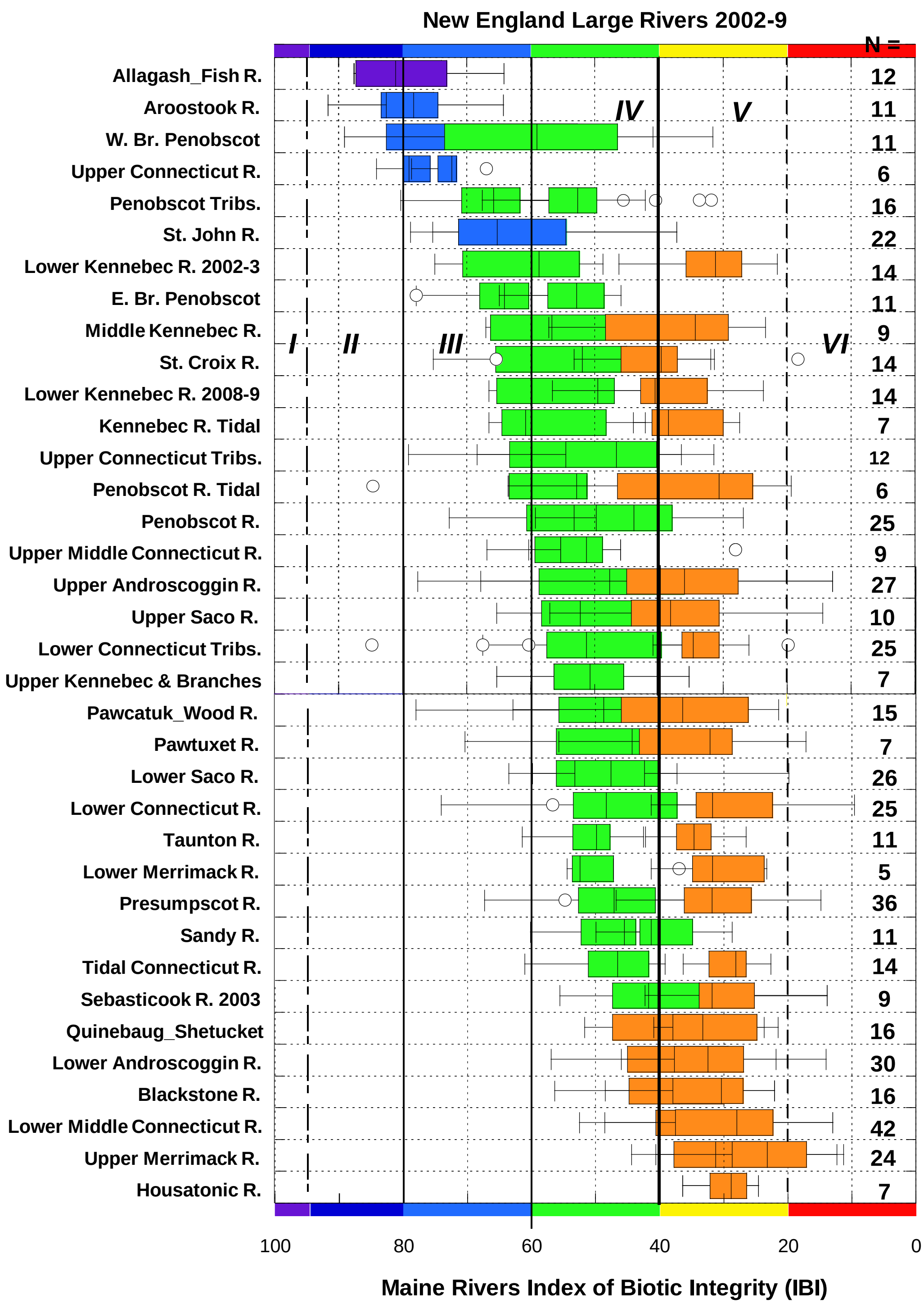


Figure B4-5. New England river reach D-IBI and riverine IBI box-and-whisker plots for all sites sampled during 2002–2009 in 36 major riverine segments in New England and ordered by the 75th percentile of the D-IBI. Fill color corresponds to the BCG range using the 50th percentile D-IBI or riverine IBI value for each river and reach.

Assessment of Reach-Level Impacts: Flow Modification

The Connecticut River downstream from the Turners Falls dam is affected by flow diversions to the Cabot hydropower project. An objective of the 2008–2009 Connecticut River intensive survey was to assess possible local scale effects from stressors such as habitat and flow modifications. River flows in an approximate 3.5 mile long reach of the Connecticut River are effectively "by-passed" with most of the flow being diverted into a canal that provides water to the Cabot hydroelectric generating station. A minimum flow of 120 cfs is maintained over the Turners Falls dam into the by-pass reach during low flow periods. The result is a very constricted wetted channel with the much wider physical channel lacking flows that are comparable to a typical New England moderate-high gradient river of this size. As a result, the habitat consisted almost entirely of pools with little or no flow velocity in the upper reach that is represented by the upstream most site (RM 67.9). Four sites were located within and immediately downstream from the 3.5 mile bypassed reach to assess potential effects of the diversion of flows (Figure B4-6). The riverine IBI, D-IBI, and selected IBI metric results for the four sites are shown in Figure B4-6. The riverine IBI at upstream-most site (RM 67.9) in the bypass reach revealed BCG level 6 (very poor) quality, and the second site (RM 66.9) was BCG level 4 (fair) for the IBI. This site is downstream from the partial return of flows from the Cabot station feed channel, which was positive for the fish assemblage. The D-IBI was one BCG level higher at three of the four sites, indicating a higher abundance of diadromous species in three samples. The IBI metric results generally reflected BCG levels 5 and 6 with sporadic exceptions. The results for %blackbasses and macrohabitat generalists were consistent with the high degree of flow alteration and the resulting negative influences of the flow diversion on habitat quality in the bypass reach. Simply increasing the minimum flows over the Turners Falls dam would result in improved IBI scores and an increase in the BCG level. The BCG framework provides a means to communicate information to stakeholders to better understand the gains, or losses, in management decisions.

Assessment of Reach-Level Impacts: Dam Removal

In an effort to document how the fish IBIs responded to the improved habitat and access to diadromous fish, the lower Sebasticook River has been sampled annually since 2009 as a follow-up to the removal of the Ft. Halifax dam at the mouth in Winslow, Maine. A baseline assessment of what was then an impounded riverine habitat was conducted at three sites upstream from the Ft. Halifax dam in 2003. The dam was removed in 2008 as part of a FERC relicensing agreement to improve access for river herring to their historic spawning areas in the Sebasticook River drainage. The Ft. Halifax dam removal was coupled with improved fish passage at two upstream dams. The results of sampling after the dam removal in 2009, 2010, and 2011 show increases in both the riverine IBI and D-IBI, but particularly so with the latter (Figure B4-7). The modest improvement in the riverine IBI reflected improved riverine habitat for resident freshwater species, but the capacity for additional improvement is limited by historic alterations in the flow and thermal regimes and the introduction of non-native species such as smallmouth and largemouth bass. Other introduced species such as northern pike and several sunfish species have occurred post-dam removal. The D-IBI showed a comparatively larger increase due to improved access by diadromous species and river herring in particular. These results show not only improved access, but the success of these species finding and reproducing in their historic spawning areas as the sampling measures the outmigration of the young-of-year of these species in the late summer and fall of each year. By including the D-IBI, the results are a better representation of the BCG level corresponding to the strong improvement in the diadromous species that comprise attribute X.

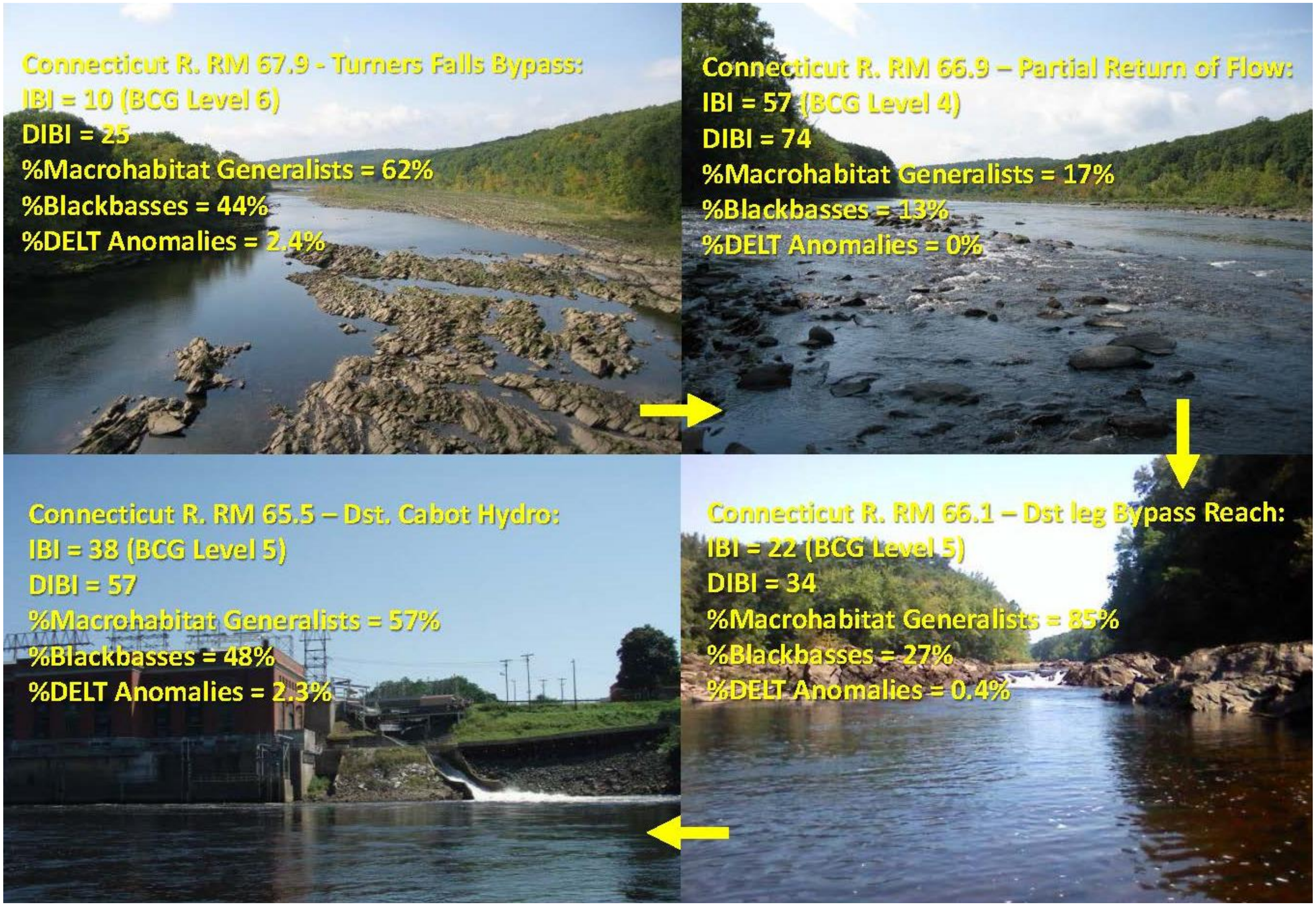


Figure B4-6. Fish sampling results in and downstream from the Turners Falls bypass reach in the Connecticut River and in the vicinity of the Cabot hydropower project in 2009 showing riverine IBI and D-IBI scores and selected metric results. Color shading in the cells corresponds to the BCG level for the IBI, D-IBI, and each IBI metric (see Table B4-5). (Green BCG level 3, Yellow BCG level 4, Orange BCG Level 5, Red BCG level 6).

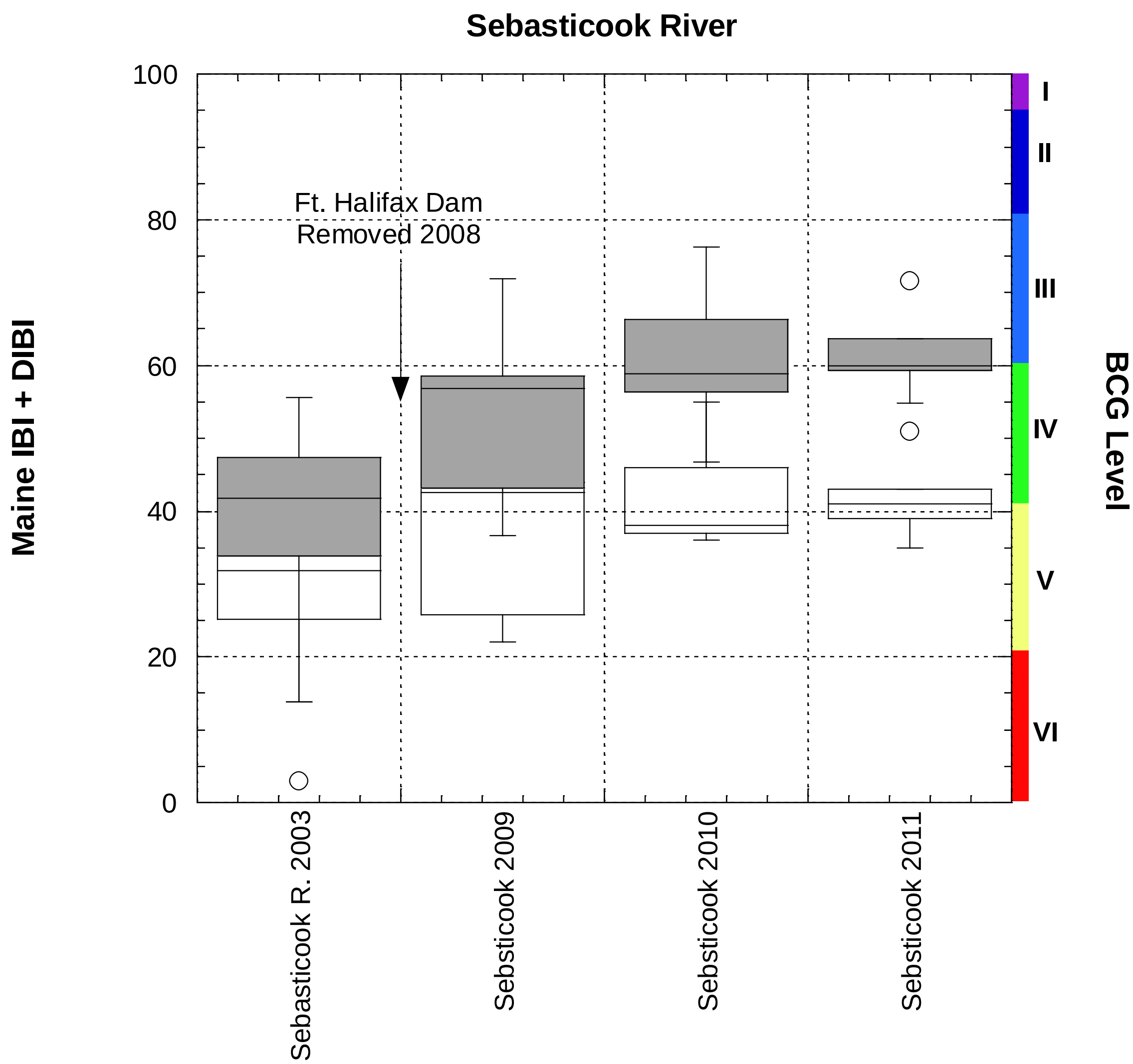


Figure B4-7. Box-and-whisker plots of IBI and DIBI values based on annual fish sampling results at three locations in the lower Sebasticook River upstream from the Ft. Halifax dam site in Winslow, ME. The Ft. Halifax dam was removed in 2008. The corresponding BCG levels are depicted on the y2 axis.

B4.3 Conclusions

The BCG was used as a conceptual foundation for the derivation of the New England riverine fish IBI and supplemental diadromous metrics. The BCG informed the selection of metrics and establishing incremental thresholds along a gradient of condition. The BCG framework spurred the derivation and use of the D-IBI, expanding the riverine IBI to include four additional metrics that reflect the presence and condition of diadromous fish species. In concert with the riverine IBI metrics, the D-IBI provide a more comprehensive assessment of the riverine fish community by including the diadromous characteristics of the New England riverine fish fauna. While the quantitative calibration of IBI metrics was done independently of the BCG, the choice of using a continuous calibration and the 0–100 scoring was done to emulate as closely as possible the full scale of the y-axis of the BCG. The results found throughout New England, and northern Maine in particular, suggest that this was a reasonable approach. Though there were no BCG level 1 sites found, level 2 conditions were observed where they would have been expected based on minimal levels of disturbance. Thus far, the resulting IBIs have utility in detecting the biological impacts from alterations to the flow regime, the thermal regime, and habitat, each of which is a key focus of contemporary restoration efforts in numerous New England rivers.

Appendix B4 References

Bain, M.B., and M.S. Meixler. 2008. A target fish community to guide river restoration. *River Research and Applications* 24:453–458.

Bain, M. B., and M. S. Meixler. 2000. *Defining a Target Fish Community for Planning and Evaluating Enhancement of the Quinebaug River in Massachusetts and Connecticut*. Final report by the New York Cooperative Fish and Wildlife Research Unit, Cornell University, Ithaca, NY to the New England Interstate Water Pollution Control Commission, Lowell, MA. 51 pp.

Barbour, M.T., J. Gerritsen, G.E. Griffith, R. Frydenborg, E. McCarron, J.S. White, and M.L. Bastian. 1996. A framework for biological criteria for Florida streams using benthic macroinvertebrates. *Journal of the North American Benthological Society* 15(2):185–211.

Curry, R.A. 2007. Late glacial impacts on dispersal and colonization of Atlantic Canada and Maine by freshwater fishes. *Quaternary Research* 67(2):225–233.

Danielson, T. 2006. *Standard Operating Procedure Methods for Sampling Stream and Wetland Algae*. Maine Department of Environmental Protection, Bureau of Land and Water quality, Division of Environmental Assessment, Augusta, ME. DEPLW0634. 18 pp.

Davies, S.P., and S.K. Jackson. 2006. The biological condition gradient: A descriptive model for interpreting change in aquatic ecosystems. *Ecological Applications* 16(4):1251–1266 + appendices.

Davies, S.P., L. Tsomides, J.L. DiFranco, and D.L. Courtemanch. 1999. *Biomonitoring Retrospective: Fifteen Year Summary for Maine Rivers and Streams*. Maine Department of Environmental Protection, Bureau of Land and Water quality, Division of Environmental Assessment, Augusta, ME. DEPLW1999-26. 192 pp.

Davies, S.P. and L. Tsomides. 1997. *Methods for Biological Sampling and Analysis of Maine's Inland Waters*. Maine Department of Environmental Protection, Bureau of Land and Water quality, Division of Environmental Assessment, Augusta, ME. 29 pp.

Emery, E.B., T.P. Simon, F.H. McCormick, P.A. Angermeier, J.E. DeShon, C.O. Yoder, R.E. Sanders, W.D. Pearson, G.D. Hickman, R.J. Reash, and J.A. Thomas. 2003. Development of a Multimetric Index for Assessing the Biological Condition of the Ohio River. *Transactions of the American Fisheries Society* 132:791–808.

Goldstein, R.M., and T.P. Simon. 1999. Toward a unified definition of guild structure for feeding ecology of North American freshwater fishes, pp. 97–122. *in* T.P. Simon (ed.), *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. CRC Press, Boca Raton, FL.

Halliwell, D.B. 2005. *Introduced Fish in Maine*. Focus on Freshwater Biodiversity, MABP Series. 12 pp.

Halliwell, D.B, R.W. Langdon, R.A. Daniels, J.P. Kurtenbach, and R.A. Jacobson. 1999. Classification of freshwater fish species of the northeastern United States for use in the development of indices of biological integrity, with regional applications, pp. 301-338. *in* T.P. Simon (ed.), *Assessing the Sustainability and Biological Integrity of Water Resources Using Fish Communities*. CRC Press, Boca Raton, FL.

Hartel, K.E., D.B. Halliwell, and A.E. Launer. 2002. *Inland Fishes of Massachusetts*. Massachusetts Audubon Press, Faunal Series No. 3, Lincoln, MA.

Hokanson, K.E.F. 1977. Temperature requirements of some percids and adaptations to the seasonal temperature cycle. *Journal of the Fisheries Research Board of Canada* 34(10):1524–1550.

Hughes, R.M., and J.R. Gammon. 1987. Longitudinal changes in fish assemblages and water quality in the Willamette River, Oregon. *Transactions of the American Fisheries Society* 116:196–209.

Hughes, R.M., D.P. Larsen, and J.M. Omernik. 1986. Regional reference sites: A method for assessing stream pollution. *Environmental Management* 10:629–635.

Hughes, R.M., P.R. Kaufmann, A.T. Herlihy, T.M. Kincaid, L. Reynolds, and D.P. Larsen. 1998. A process for developing and evaluating indices of fish assemblage integrity. *Canadian Journal of Fisheries and Aquatic Science* 55:1618–1631.

Karr, J.R., K.D. Fausch, P.L. Angermeier, P.R. Yant, and I.J. Schlosser. 1986. *Assessing Biological Integrity in Running Waters: A Method and its Rationale*. Illinois Natural History Survey Special Publication 5, 28 pp.

Langdon, R.B. 2001. A preliminary Index of Biotic Integrity for fish assemblages of small coldwater streams in Vermont. *Northeastern Naturalist* 8(2):219–232.

Lyons, J., R.R. Piette, and K.W. Niermeyer. 2001. Development, validation, and application of a fish-based index of biotic integrity for Wisconsin's large warmwater rivers. *Transactions of the American Fisheries Society* 130(6):1077–1094.

Mebane, C.A., T.R. Maret, and R.M. Hughes. 2003. Development and testing of an index of biotic integrity (IBI) for Columbia River basin and western Oregon. *Transactions of the American Fisheries Society* 132:239–261.

Nelson, J.S., E.J. Crossman, H. Espinosa-Pérez, L.T. Findley, C.R. Gilbert, R.N. Lea, and J.D. Williams. 2004. *Common and Scientific Names of Fishes from the United States, Canada, and Mexico*. Sixth edition, American Fisheries Society Special Publication 29. 386 pp.

Ohio Environmental Protection Agency. 1987. Biological Criteria for the Protection of Aquatic Life: *Volume II. Users Manual for Biological Field Assessment of Ohio Surface Waters*. Division of Water Quality Monitoring and Assessment, Surface Water Section, Columbus, Ohio.

Page, L.M., H. Espinoza-Perez, L.T. Findley, C.R. Gilbert, R.N. Lea, N.E. Mandarak, R.L. Mayden, and J.S. Nelson. 2013. *Common and Scientific Names of Fishes from the United States, Canada, and Mexico, 7th edition*. American Fisheries Society, Special Publication 34, Bethesda, Maryland.

Saunders, R., M.A. Hachey, and C.W. Fay. 2006. Maine's diadromous fish community: Past, present, and implications for Atlantic salmon recovery. *Fisheries* 31(11):537–547.

Stoddard, J.L., D.P. Larsen, C.P. Hawkins, R.K. Johnson, and R.H. Norris. 2006. Setting expectations for the ecological condition of streams: The concept of reference condition. *Ecological Applications* 16(4):1267–1276

USEPA. 2013. *Biological Assessment Program Review: Assessing Level of Technical Rigor to Support Water Quality Management*. EPA 820-R-13-001. Office of Water, Washington, DC. 144 pp.

Warner, K. 2005. Smallmouth bass introductions in Maine: history and management implications. *Fisheries* 30(11):20–26.

Whittier, T.R., D.B. Halliwell, and R.A. Daniels. 2001. Distribution of lake fishes in the northeast–IV. Benthic and small water column species. *Northeastern Naturalist* 8(4):455–482.

Whittier, T.R., D.B. Halliwell, and R.A. Daniels. 2000. Distribution of lake fishes in the northeast–II. The Cyprinidae (minnows). *Northeastern Naturalist* 7(2):131–156.

Yoder, C.O., and M.T. Barbour. 2009. Critical technical elements of state bioassessment programs: A process to evaluate program rigor and comparability. *Environmental Monitoring and Assessment*. DOI10.1007/s10661-008-0671-1 (accepted for publication).

Yoder, C.O. and B.H. Kulik. 2003. The development and application of multimetric biological assessment tools for the assessment of impacts to aquatic assemblages in large, non-wadeable rivers: A review of current science and applications. *Canadian Journal of Water Resources* 28(2):301–328.

Yoder, C.O., B.H. Kulik, and J.M. Audet. 2006a. *The Spatial and Relative Abundance Characteristics of the Fish Assemblages in Three Maine Rivers*. MBI Technical Report MBI/12-05-1. Grant X-98128601 report to U.S. Environmental Protection Agency, Region 1, Boston, MA. 136 pp. + appendices.

Yoder, C.O., B.H. Kulik, B.J. Apell, and J.M. Audet. 2006b. *2005 Maine Rivers Fish Assemblage Assessment: I. Northern Maine Rivers Results; II. Maine Rivers Fish Species Distribution Atlas; III. Toward the Development of a Fish Assemblage Index for Maine Rivers*. MBI Technical Report 12-06-1. Report to U.S. Environmental Protection Agency, Region 1, Boston, MA. 71 pp. + appendices.

Yoder, C.O., R.F. Thoma, L.E. Hersha, E.T. Rankin, B.H. Kulik, and B.R. Apell. 2008. *Maine Rivers Fish Assemblage Assessment: Development of an Index of Biotic Integrity for Maine Rivers*. MBI Technical Report 2008-11-2. Report to U.S. EPA, Region 1, Boston, MA. 69 pp. http://www.midwestbiodiversityinst.org/publications. Accessed February 2016.

Yoder, C.O., E.T. Rankin, and L.E. Hersha. 2015. *Development of Methods and Designs for the Assessment of the Fish Assemblages of Non-Wadeable Rivers in New England*. MBI Technical Report MBI/2015-3-3. U.S. EPA Assistance Agreement RM-83379101. U.S. EPA, Office of Research and Development, Atlantic Ecology Division, Narragansett, RI and U.S. EPA, Region 1, Boston, MA. 152 pp. http://www.midwestbiodiversityinst.org/publications. Accessed February 2016.

Whittier, T.R., and R.M. Hughes. 1998. Evaluation of fish species tolerances to environmental stressors in lakes in the northeastern United States. *North American Journal of Fisheries Management* 18(2):236–252.