Performance of Floristic Quality Assessment in Massachusetts Forested Wetlands

PERFORMANCE OF FLORISTIC QUALITY ASSESSMENT IN MASSACHUSETTS FORESTED WETLANDS

A Thesis Presented

by

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DEDICATION

I dedicate this thesis to my Grandfather, Charles Gorss, who has been a role model for my decision to pursue my passion for wetland science and environmental conservation. I would also like to dedicate this thesis to my family, friends, fellow graduate students, and faculty who have supported and encouraged me every step along this journey.

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ABSTRACT

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In order to combat the loss of valuable wetland functions and services, federal, state and tribal governments must have the tools to accurately assess and monitor the condition of wetland ecosystems. One particular method of wetland assessment is Floristic Quality Assessment (FQA), which has been growing in popularity throughout the United States since its creation in the 1970s. FQA relies on vegetative indicators of human disturbance to assess the integrity of an ecosystem. FQA calculations are based on Coefficients of Conservatism (C-scores), professionally-assigned scores ranging from 0-10 that denote a local species' tolerance to anthropogenic disturbance. Despite increasing interest in the use of FQA, few studies have thoroughly tested the performance of FQA, especially in New England. We used the Conservation Assessment and Prioritization System (CAPS), a landscape-based, coarse-scale assessment method, as a basis for evaluating FQA's performance in Massachusetts's forested wetlands. Our objective was to use CAPS Index of Ecological Integrity (IEI) scores (a form of generalized stressor gradient) to evaluate the performance of a variety of FQA indices (biological condition gradients), using C-scores from 7 states in the Northeast, and 2 ecoregions in Massachusetts. Based on our calculations of r-squared, and Spearman's rank analysis, we

determined that FQA and C-scores have a moderate to weak relationship with the CAPS index of ecological integrity. Of the 12 indices and metrics we tested, the index with the strongest relationship to the IEI stressor gradient was mean Coefficient of Conservatism. Based on this research a number of suggestions are proposed for improving FQA as it applies to wetland assessment.

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Review the factors that determine where species occur (i.e., abiotic vs biotic; including concepts of top-down vs bottom-up and trophic cascades). What determines ecological niches - why do some species show up everywhere and others are narrowly constrained to certain environments:

Plants, whether aquatic, emergent, shrub or tree, serve as the primary-producing foundations of all wetland ecosystems. Several biotic and abiotic factors determine where species are likely to grow. Habitat requirements that determine species' range include hydrology, sunlight, soil chemistry, competition, and the extent of grazing. Individual plants are adapted to deal with these biotic and abiotic factors differently; over many thousand years of evolution, these adaptations have allowed plants to create diverse communities within specific habitat types. A plant's niche, or role in its environment, is dependent on its adaptations and interactions with the unique abiotic and biotic factors that define that environment.

Vegetation composition and biomass are greatly influenced by interactions with grazers and other organisms in a food web. The interactions between wetland plants and higher trophic levels can generally be described through top-down and bottom-up trophic cascades. A bottom-up effect happens where a change to the bottom trophic level (vegetation) produces a correlated change in the trophic levels above it (Hoekman, Winston, & Mitchell, 2009; Leibold, Chase, Shurin, & Downing,1997). For example, a prosperous year for the plant *Typha latifolia* might

provide more nutrients and food for herbivores that graze on it. Those herbivores benefit from the extra nutrients, reproduce, and then become preyed upon by higher trophic levels. The positive impacts spread up the food chain. By contrast, trophic cascades happen when a change is made to higher trophic levels, resulting in an alternating pattern of increasing and decreasing population levels of each lower trophic level (Hoekman, Winston, & Mitchell, 2009; Leibold, Chase, Shurin, & Downing,1997). For example, an increase in raptor populations may cause heavy predation that reduces the number small rodents. Fewer rodents would be available to consume the vegetation layer, resulting in a more prosperous herbaceous community. This trend reverses with a decrease in raptor populations. If a new tertiary consumer or disease negatively impacts raptor populations, rodent populations would increase from lack of predation. The herbaceous community would then be consumed to a higher degree (Leibold, Chase, Shurin, & Downing,1997). These trophic interactions directly affect vegetation biomass, but the degree to which the different trophic levels change is variable (Herendeen, 2004).

Competition plays a role in plant composition, just as the effects of trophic interactions do. In the aforementioned situation of raptor population decrease, one would expect to see an associated decrease in vegetative species specifically targeted by the rodents (Hoekman, Winston, & Mitchell, 2009). In the same situation, other species such as *Toxicodendron radicans* that are less palatable may see population increases from this trend, as its competition for space or some other resource is eaten. Plants have adapted to compete for space, sunlight, and nutrient resources within their ecological niches (Grimes, 2001). When space is thus created in the plant community, the plants that have adapted to grow quickly and have large seed banks will be the first to propagate themselves in that area (Grimes, 2001). Invasive species are often highly competitive in

these circumstances for their ability to reproduce and grow quickly in a wide range of environments (Dirks et al., 2017; Čuda et al., 2015). In native communities, plant competition is driven by factors such as long-term population growth rates, specific resource requirements, defenses against natural enemies, and dispersal techniques (Martorell & Freckleton, 2014). Plant composition over time is largely influenced by these competitive adaptations.

In addition to competition and food web interactions, hydrology is a highly defining aspect of plant survival and composition specifically in wetland ecosystems (Mitsch & Gosselink, 2007). The unique hydrology of each wetland is characterized by the length and duration of saturation, which often changes with season. The composition of any given wetland is somewhat predictable, based on our knowledge of which plants are better adapted to the varying levels of saturation. Some wetland plants, called obligate species, are so uniquely adapted to a heavily saturated environment that they do not grow outside wetlands (Tiner, 1999). Besides the obligates, species growing in wetlands can be classified as facultative-wet (60-90% occurrence), facultative (50% occurrence), or facultative-upland (10-30% occurrence), depending on their documented frequency of occurrence in wetlands or uplands (Tiner, 2005). Various physiological adaptations of wetland vegetation to waterlogging include the development of arenchyma, adventitious roots, lenticels, oxidized rhizospheres, and hydrophobic leaves. Many of these adaptations, such as the spongy air-filled arenchyma tissue, are designed specifically to facilitate the flow of oxygen to the roots in waterlogged soil (Mitsch & Gosselink, 2007). Wetland plants can be further defined as either tolerators or regulators, depending on whether they have these adaptations to tolerate saturation or whether they have adapted to actively avoid saturated conditions. Plants that regulate themselves are more likely to be long-term inhabitants of wetland

environments, where tolerators are more likely to be found on the fringes of a wetland (Mitsch & Gosselink, 2007). Any plant's range and ability to survive in certain niches and communities is heavily dependent on these adaptations.

Hydrologic changes to wetland ecosystems are generally reflected in the soil chemistry. The soil that wetland plants are adapted to live in may contain high percentages of clay, carbon, and other dissolved minerals such as iron, manganese, sulfur, sodium, or calcium relative to upland soil (Mitsch & Gosselink, 2007). The community of plants in any given wetland responds to these unique soil conditions. For example, the salt marsh grass *Spartina alterniflora* has adapted to excrete excess salt from the glands in its leaves (Mitsch & Gosselink, 2007). The soil in bogs has lower levels of nitrogen than other wetland communities. Carnivorous bog species such as *Sarracenia purpurea* have adapted to get their nutrients by consuming insects and other allochthonous resources originating from outside the wetland (Mitsch & Gosselink, 2007). Just as plant species occur in areas they are adapted to hydrologically, they also occur where they are adapted to the soil anomalies. However, native wetland composition may be disrupted if a non-native species with generalist tendencies is able to propagate itself in wetland areas with a very specific soil chemistry. *Phragmites australis,* for example, is adapted to grow in a wide range of wet and anthropogenically disturbed habitats, with a tendency to overrun and crowd out native species (Armstrong, Jones & Amstrong, 2006). The result of non-native invasion is often a decrease in biodiversity (creation of a monoculture) and a drastic change in native species distribution and composition (Gordon, 1998). Other invasives, such as the *Melaleuca* tree in Florida, can even change the hydrology of the wetland (Gordon, 1998).

In wetlands where natural features include extreme or variable hydrology and soil chemistry, upland plant species are less competitive for space and nutrients. The plant species that are able to compete for space and resources across habitats, upland as well as wetland, are called generalists (Mitsch & Gosselink, 2007). Alternately, specialists are those that have spent hundreds of years adapting to the unique environments, such as wetlands. All plants, whether generalist or specialist, must strike a balance between survival and propagation. Generalist vegetation will have focused energy to inhabit a wide range of environments, but may not have the competitive edge to thrive in a specialized environment such as wetlands (Grimes, 2001). The wide range of specialist adaptations has created great diversity in wetland plant form and function.

Plant survival in wetland communities is driven by very specific physiological adaptations to biotic and abiotic stressors, and the availability of resources. All of these factors determine where species occur in wetlands and the uplands surrounding them. Wetland scientists have mapped, classified, and documented just about every wetland plant species in Massachusetts. We now have a pretty clear idea of what grows where, and with what percent chance of occurrence they will be found there. Through these efforts, it is now understood that the ranges and abilities of each plant species are heavily dependent on its interactions with the environment and years of evolution that have enabled it to survive to this day.

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1.2 Scott Jackson's Comprehensive Exam Question (11/30/17)

1.2.1. Review the provisions of the federal Clean Water Act that create the responsibility for states to assess and monitor wetlands.

Wetland functions and values are often difficult to define in financial terms (Tiner, 2005). This can make communicating their worth for the purpose of lawmaking more challenging. Still, the ecosystem services they provide and their benefits to downstream water resources are well documented (Tiner, 2005; Mitsch & Gosselink, 2007). For this reason, wetlands are protected ecosystems on a national and global scale. The laws and enforcements that protect U.S. wetlands have changed dramatically in the past seven decades. Understanding the evolution of wetland protection can help us come to terms with why wetland assessment is so important in 2017.

The first major water protection act of its kind in the United States was the Federal Water Pollution Control Act. It was created in 1948, in response to aquatic point-source pollution threats to water resources, and human health (US EPA, "History of the Clean Water Act"). However, it lacked solid measures of enforcement and specificity in regulations, so it was heavily amended in 1972 (US FWS, 2013). With the 1972 amendments, the act was renamed the "Clean Water Act (CWA). The Clean Water Act did not explicitly protect wetlands, but rather regulated the addition of pollutants to "navigable waters." Jurisdiction would eventually come to include tributaries and wetlands if they could be proven to significantly affect downstream navigable waters (U.S. EPA, 2008).

Much has been added to the CWA since 1972. In the years between 1972 and 1990, the EPA's authority to implement pollution control programs increased (US EPA, "History of the Clean Water Act"). The Clean Water State Revolving Fund came along with changes in 1987, allowing states to finance their highest priority water quality needs with funding from the federal government. This financial cooperation set the stage for greater cooperation between states and the EPA when it came to regulation and monitoring water resources (US EPA, "History of the Clean Water Act"). In 2003, The EPA published a guidance document entitled "Elements of a State Water Monitoring and Assessment Program" to inform state monitoring and assessment programs. This was a way for the federal government to standardize methods of assessment, and ensure that the data collected by states were useful and of high quality (US EPA, "History of the Clean Water Act"). This strategy initially addressed only major waterways. Wetlands were not immediately included in assessment and monitoring programs.

In 1985, the supreme court ruled unanimously in favor of including adjacent wetlands as jurisdictional under the term "navigable waters", in the case of *The United States Vs. Riverside Bayview Homes*. In another court ruling in 2001, Solid Waste Agency of Northern Cook County v. U.S. Army Corps of Engineers, the supreme court ruled that isolated, non-navigable wetlands could not be considered jurisdictional under the CWA, due solely to their importance for migratory birds.

In the 2006 Supreme Court ruling of *Rapanos Vs. the United States*, the federal government was tasked with determining whether or not certain wetlands and tributaries could be protected as "waters of the united states". The decision from this court case served to further restrict jurisdiction over wetlands; it stated that before certain wetlands could be considered for coverage, the EPA and the Corps would need to establish whether a "significant nexus" existed between a wetland, separately or in combination with other, similarly situated wetlands, and the physical, chemical, and biological integrity of downstream navigable waters (U.S. EPA, 2008). Wetlands that were proven on a case by case basis to have a significant nexus could be granted protection under the CWA. In 2015, the "Waters of the United States" (WOTUS) rule sought to provide guidance on establishing a wetlands significant nexus. This WOTUS rule expanded CWA applications to include intermittent headwater streams and isolated wetlands, as long as a significant nexus could be established (EPA, *"Waters of the United States "WOTUS" Rulemaking")*. However, this rule has been stayed by a U.S. Court of Appeals.

Since the mid 1990's, wetland regulations have been increasingly delegated to the states and interstate agencies, such as the New England Interstate Water Pollution Control Commission, from the federal government. As of 2017, reporting on wetland condition for the purpose of regulation and protection is carried out on a state-to-state basis under the supervision of EPA (Votteler & Muir, 2002). There are multiple state-level wetlands protection initiatives that are separate from the clean water act. States may add additional protections to wetlands within their borders, but never detract from Federal wetland laws. In some states, towns include higher levels of wetland protection through their bylaws. The necessity for states to carry out the assessment

and monitoring of their wetlands and waterways is delegated within the following key provisions within the Clean Water Act:

Section 101(a): Introduces Designated Uses. Also introduces the national goal that wherever possible, these designated uses (one or many) should be attainable.

Section 106(a): The EPA is responsible for making sure that individual states are monitoring the quality of navigable waters, compiling and analyzing data on water quality, so that it can be reported and additional funds granted. Wetlands are included in this definition of 'waters of the United States'.

Section 303: States must develop and implement water quality standards (including those of jurisdictional wetlands) for all listed waters. These standards are used to designate the use of each water body, as well as establish criteria and requirements to protect and maintain the health of the water body. When it comes to designated uses, water bodies may fall under one or more of the following categories:

1. Protection and propagation of fish, shellfish and wildlife (aquatic life use)
2. Recreation
3. Public drinking water supply
4. Agricultural, industrial, navigational and other purposes.

Wetlands usually only qualify for the first, valuable mostly for supporting aquatic life use.

Section 303(d): States are required to assess their waters, and create a list of threatened and impaired waters (that are below the water quality standards established by that state). This section also has states develop Total Maximum Daily Loads (TMDLs) that each impacted waterbody on the list can accommodate before surpassing the water quality standards of that state.

Section 305(b): Used in conjunction with 303(d), requires states to submit a report of the findings on water quality to the EPA every 2 years.

Section 401: Limiting discharge into navigable waters (including jurisdictional wetlands): The State, or interstate organization that is responsible for the waters affected by any project that is discharging material into a navigable water of the United States, must issue a certificate for that project. State issuance of a 401 certificate indicating that a project meets water quality standards is required before the ACOE can issue a section 404 permit.

Section 404: Dredge and Fill regulations for waters of the united states including wetlands. Enforced by the EPA and ACOE jointly. General and individual permits are issued by the ACOE in consultation with EPA, states and fish and wildlife agencies. Activities that are covered by general permits for minimal adverse effects require no individual review; other projects are reviewed by the EPA, ACOE, or States depending on the category of activity.

States have the following roles to play under section 404:

1. States may consult with the EPA on the ACOE issuance of state program general permits
2. Water quality certification
3. Program assumption: This allows states to assume administration of the Clean Water Act under section 404. This removes ACOE involvement, but the EPA still reviews state activity to make sure it is in compliance with the CWA.

(US *EPA, 1995;* US FWS, 2013; US EPA, "History of the Clean Water Act")

1.2.2 How does wetland assessment and monitoring compare to the more established approaches for monitoring the condition of water bodies (lakes, ponds, rivers and streams)?

In reference to sections 101(a) and 303(d) of the Clean Water Act, when setting water quality standards for wetlands, "designated uses" generally only fall under the "Protection and propagation of fish, shellfish and wildlife" (US EPA, "Standards for Water Body Health"). Numerous wetland functions and values such as water storage, slow-water release, nutrient retention and cycling, sediment retention, etc., are not listed as designated uses in section 303b. (Tiner, 2005). Wetlands are generally not deep or clear enough to be used for recreation the same way lakes and large bodies of water are used. Their often mucky and highly vegetated states prevent most from being used for public drinking water supply. Agriculture, industry and navigation similarly conflict with the nature of wetlands, where stability of soil and consistency of water depth are desired. Of the listed designated uses then, only wildlife propagation and protection apply to wetland ecosystems (US EPA, "*Impaired Waters and TMDLs: Identifying and Listing Impaired Waters")*.

In 2005 the EPA drafted a reference document to help quantify the designated uses of wetlands, further exploring the overarching wildlife propogation use. This document was titled "Use of Biological Information to Better Define Designated Aquatic Life Uses in State and Tribal Water Quality Standards: Tiered Aquatic Life Uses". Tiered Aquatic Life Use, shortened to "TALU", is a model that has been developed to express the quantity and condition of biological

resources within a waterbody (U.S. EPA, 2005). For a wetland, this can help to quantify the aquatic life use that would otherwise be difficult to account for. Specific goals of TALU, as listed by the EPA's guidance document include:

- "Set ecologically-based aquatic life goals for water bodies
- Establish a consistent approach for identifying attainable, incremental restoration goals that are grounded in the concept of biological integrity
- Provide a framework that better relates traditional water quality criteria and biological criteria in determining use attainment, thus strengthening stressor/response models implicit in designated uses and criteria in water quality standards
- Better link monitoring and assessment with water quality standards
- Prioritize management actions that result in more effective use of resources." (U.S. EPA, 2005)

TALU relies on a biological condition gradient (BCG) to reflect the response of an aquatic community to individual or multiple sources of stress. The BCG is a model that establishes and graphically displays relationships between biological condition and environmental stressors. The BCG concept is meant to be a starting point for states to think about how to use available scientific information to better define aquatic life uses for their water bodies (Fig. 1.1) U.S. EPA, 2005). Through this framework, wetland designated uses can be better communicated to a wider audience. The EPA highly encourages states to incorporate biological information into their setting of water quality standards, and supports the use of TALU in quantifying designated uses of wetlands and other water bodies. The illustrated conceptual model of the biological condition gradient from the EPA's guidance document on TALU is depicted below:

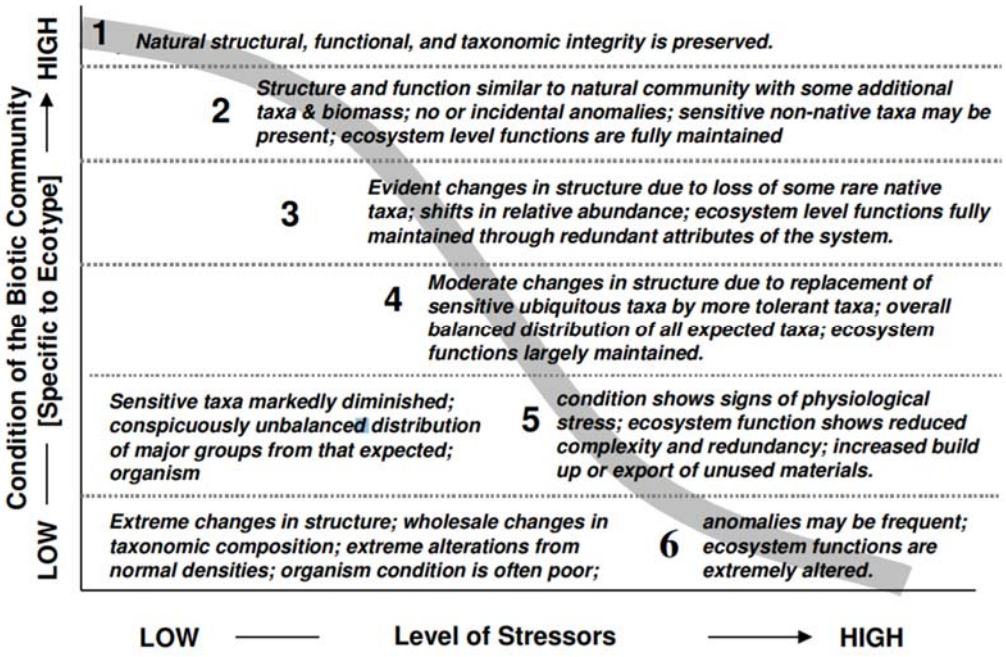

Figure 1.1: EPA Biological Condition Gradient (U.S. EPA, 2005)

1.2.3 What guidelines has EPA provided to states on wetlands assessment and monitoring?

In 2006 the EPA published an updated document to provide clarification and specifics to how the state water monitoring and assessment requirements applied to wetlands. Specific guidelines it offers to states on how to create and maintain wetland assessment and monitoring programs (US EPA, 2006) are outlined below.

According to the Application of Elements of a State Water Monitoring Program (as updated in 2006), there are 10 elements that make up a successful monitoring program and they include the following:

Monitoring Program Strategy: EPA recommends multiple agencies include wetland monitoring as an additive to pre-existing water monitoring strategies.

Monitoring Objectives: The EPA notes that individual objectives will determine the nature of the monitoring design, but should include one or all of the following:

- Establish a baseline wetland condition
- Evaluate the environmental consequences of a federal action or group of actions
- Evaluate the performance of wetland restoration projects
- Evaluate the cumulative effects of wetland loss and/or restoration, and develop watershed plans for the recovery of impaired waterbodies
- Refine or create wetland specific water quality standards

Monitoring Design: 3 generally accepted sampling designs for the monitoring and assessment of wetlands. There is emphasis placed on the procedure for setting a reference wetland condition.

- A census that entails examining every unit in the population of interest
- Probability sampling/ survey
- Use best professional judgment to target sampling within specific wetlands for purposes of comparison (RAM's use this method often)

Core and Supplemental Indicators (and Methods): Should be based on 3 levels of technical approach, and indicators should reflect identified monitoring objectives.

- Level 1 - Landscape Assessment
- Level 2 - Rapid Assessment
- Level 3 - Intensive Site Assessment

Quality Assurance: Assessments should include peer-reviewed QA/QC/QAPP according to EPA policy, in order to maintain scientific integrity of data.

Data Management: Data should be recorded and collected using reliable and accessible online databases. It is highly recommended that GIS or another landscape software be used in the storage and collection of data.

Data Analysis/Assessment: Procedures should be well documented and backed up with physical data collection sheets if possible.

Reporting: It is incumbent upon the state that it produces timely and complete water quality and wetland condition reports.

Programmatic Evaluation: "The State, in consultation with its EPA Region, conducts periodic reviews of each aspect of the monitoring program to determine how well the program serves its water quality decision needs for all State waters, including all water body types."

General Support and Infrastructure Planning: States should identify current and future resources needed to continue monitoring wetlands.

(US EPA, 2006)

These 10 program elements are the same as those listed for general waters of the United States. There are many additional texts that offer EPA guidance on how to best assess the quality of water bodies. However, since there has always been greater importance placed on improving water quality for human health and consumption, there often are fewer resources available for federal wetland assessment than for other water resource assessment.

1.2.4 Why we're doing what we're doing

Wetland laws are constantly changing and evolving. Federal wetland protection has become a polarized issue between those who wish to protect wetland ecosystems, and those who feel like the government has too much "red tape" over environmental regulations (Hopkinson, 2015). The 2017 presidential administration is on the verge of cutting back on a variety of environmental regulations. One ruling targeted by the Trump administration is the "Waters of the United States" Rule . Set in place in 2015 by the Obama administration, this interpretation of the Clean Water Act clarifies protection for intermittent streams and small wetlands that feed into larger waterways. If this rule were to be rescinded, it would be a step backwards for wetland protection and the management of clean water throughout the country. However, several states have regulations that are stricter than those at the Federal level. It will be curious to see what the effect of weakened Federal protections will have on State and town-level wetland protections and bylaws.

The general frustration with current wetland protection laws may stem from a basic misunderstanding of wetland functions and values, or confusion surrounding the various levels of regulation surrounding wetlands. Wetland protection also comes at the financial cost of landowners who are unable to build up their property if their project intersects a wetland or its buffer zone (Hopkinson, 2015). Should these broader reaching wetland protection laws be reduced, it would noticeably benefit organizations that see short-term monetary profits in the conversion of wetlands to farming and housing land uses.

The protection of this country's wetlands and waterways is more important than ever. Our nation's wetlands are a valuable ecological resource. In order to accurately account for such resources, we must be able to have reliable methods of assessing the current conditions of what we have. According to the Clean Water Act, it is still required that states report on the condition of their state's wetlands. It is important that states incorporate a scientific understanding of the biota in these assessments and reports. The research we are doing into the performance of wetland assessment methodologies will establish a better connection between ecological science and the regulations that protect wetland resources. This research is also meant to facilitate better communication between the state and federal government on the status of the nation's wetlands, and how we can protect them for future generations.

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1.3 Kevin McGarigal's Comprehensive Exam Question

The goal of this comprehensive question was to thoroughly review the statistical analyses of a single paper. Here I state the statistical procedure and which inference framework it belongs to. Then I concisely summarize the methods, identifying variables and spelling them out in clear detail.

This paper used a parametric frequentist framework to analyze the relationship between a new index of wetland condition called OIWI (that uses invertebrate biological indicators to assess wetland condition), against an established index of wetland integrity abbreviated (RIRAM) as well as the proportion of impervious surface area (ISA).

The process of calculating OIWI involves characterizing the population of Odonates (dragonflies and damselflies) in a wetland, and calculating their mean conservatism scores to determine wetland condition. The RIRAM index relies on 10 empirically calculated metrics based on ecosystem, water, and vegetation characteristics. The proportion of impervious surface area (ISA) is calculated within a 305m buffer zone around the perimeter of a wetland study site. The OIWI scores for wetland sites in this study were regressed against RIRAM and ISA, in order to compare their performance.

In their simple linear model comparison of OIWI to RIRAM & ISA, the authors found that the residual errors of the data were not normal. In order to get a better idea of what the margin of error for the R^2 values might be, they bootstrapped the data with resampling, reporting the R^2 for

model fiit 1000 times. They reported the range of R^2 values produced from these bootstrap iterations, as model fit estimations for OIWI/RIRAM and OIWI/ISA comparisons.

- **The observational unit for the analysis?**

The observational units are the individual freshwater wetlands in Rhode Island.

- **How the observational units are distributed in space and/or time (i.e. study design or sampling layout)?**

The study uses Odonate sampling data from wetland sites in Rhode Island, collected between 1999 and 2004. A subset of these original sampling sites was selected for use in this study, spanning of a gradient of surrounding land-use intensity. Sites along this gradient were chosen only if they had at least 10 specimen occurrences of Odonata over the span of time they were sampled.

- **The sample size?**

This study had a sample size of 51 wetlands.

- **The dependent variable? and was it transformed or standardized for the analysis, and if so how?**

The dependent variable is the OIWI. It is not transformed or standardized.

- **The independent variables (include scale/measurement)?**

The independent variable is wetland condition, as expressed by the RIPRAM and ISA gradients. RIRAM is calculated a scale between 0-100, measuring metrics of buffer integrity, functional integrity, and in-wetland stress. ISA is measure of the proportion (scale of 0-1) of impervious area within a 305m buffer of each polygonal unit of wetland that was studied.

What is the deterministic model is for a statistical procedure? (If the procedure was parametric, what was the assumed error distribution? If it was nonparametric, it didn't have an error distribution.)

The deterministic model here is is a linear function of wetland condition: (y=ax+b), where x is the independent variable wetland condition (RIRAM index), and y is the dependent variable (OIWI index). The slope estimate is denoted by a, and the intercept by b.

No assumptions about the error distributions were made by the authors, which infers that their statistical methods are non-parametric. They tested the error distribution of their data, and found that they were heterogeneous and not-normal, but normality/non-normality was not assumed ahead of time. On the other hand, they made overarching assumptions about OIWI, based on its fit to this linear model. Essentially, they assumed that an underlying linear model explains the relationship between the OIWI index and wetland condition. This indicates that their inference framework was parametric and frequentist in nature.

Briefly discuss the inferences made from this model:

- **Fit and report of model parameters?**

Linear model fit was reported with ranges of R^2 values produced from the bootstrapped models of their data. They reported the best linear fit (highest range of R^2 values) to be between OIWI and RIPRAM (R^2 = 0.537 - 0.803). OIWI had a weak, negative linear correlation with ISA.

- **Test significance and of what, the entire model and/or each parameter?**

Significance of these R-squared values was not tested in this study.

- **Use model selection to weigh strength of evidence for alternative models?**

Model selection was not explored.

- **Use the model to make predictions and for what purpose?**

The model was not used to make predictions.

Discuss the strengths and limitations of each approach with regards to the reliability of the inferences.

The inferences drawn from this research were solely about model fit. The use of R^2, as a coefficient of determination, is accepted as a reliable way of determining model fit. The use of bootstrapping to repeatedly calculate model fit output from the resampled data, gives the the authors a range of estimates for R^2 in, which is . This was a simple, yet effective way of measuring the relationships between the 3 indices of biological and ecological integrity. However, a higher sample size than 51 could have improved their estimation of R^2.

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CHAPTER 2

PERFORMANCE OF FLORISTIC QUALITY ASSESSMENT IN MASSACHUSETTS FORESTED WETLANDS

Abstract

In order to combat the loss of valuable wetland functions and services, federal, state and tribal governments must have the tools to accurately assess and monitor the condition of wetland ecosystems. One particular method of wetland assessment is Floristic Quality Assessment (FQA), which has been growing in popularity throughout the United States since its creation in the 1970s. FQA relies on vegetative indicators of human disturbance to assess the integrity of an ecosystem. FQA calculations are based on Coefficients of Conservatism (C-scores), professionally-assigned scores ranging from 0-10 that denote a local species' tolerance to anthropogenic disturbance. Despite increasing interest in the use of FQA, few studies have thoroughly tested the performance of FQA, especially in New England. We used the Conservation Assessment and Prioritization System (CAPS), a landscape-based, coarse-scale assessment method, as a basis for evaluating FQA's performance in Massachusetts's forested wetlands. Our objective was to use CAPS Index of Ecological Integrity (IEI) scores (a form of generalized stressor gradient) to evaluate the performance of a variety of FQA indices (biological condition gradients), using C-scores from 7 states in the Northeast, and 2 ecoregions in Massachusetts. Based on our calculations of r-squared, and Spearman's rank analysis, we determined that FQA and C-scores have a moderate to weak relationship with the CAPS index of ecological integrity. Of the 12 indices and metrics we tested, those with the strongest relationship to the IEI stressor gradient were mean Coefficient of

Conservatism. Based on this research a number of suggestions are proposed for improving FQA as it applies to wetland assessment.

2.1 Introduction

The beneficial natural functions that wetlands provide to society and the environment are well documented. Even so, wetland loss and degradation is an ongoing struggle in the United States. At all levels of government (federal, state/tribal and local), regulations are in place to conserve and restore wetlands. The process of mitigating wetland loss starts first with identifying which wetlands are most in need of conservation or restoration, then continues with monitoring to track changes to those wetlands over time. This process is supported through frequent reporting on wetland function and condition. Mandates for states and tribes to report on wetland condition are included in federal regulation through the Clean Water Act (CWA) of 1972.

2.1.1 The Clean Water Act

Section 303 of the Clean Water Act mandates that there be state/tribal Water Quality Standards for jurisdictional "Waters of the United States" (United States, 1972; US EPA 2001). These water quality standards are based on a list of "Designated Uses", which exist to help identify the values and uses for each water body and set water quality standards necessary to support those designated uses. Section 305(b) requires state and tribal governments to report on the status of their waters of the U.S. with regard to whether they are meeting water quality standards. State and tribal reports are due on a regular bases, usually every 2 years. Section 303(d) mandates that States and Tribes specifically identify waters that fail to meet their water quality

standards. In order to meet the requirements of 305(b) and 303(d), state and tribal governments need to institute assessment and monitoring programs for Waters of the United States.

The federal Environmental Protection Agency's (EPA) initial efforts on water quality assessment and monitoring programs focused on water bodies (lakes, ponds, rivers and streams). More recently, these monitoring and assessment programs include wetlands as well. Since the mid 1990's, it has been the programmatic responsibility of states and tribes to perform wetland monitoring and assessment required under the CWA (Votteler & Muir, 2002).

In their recommendations for state and tribal assessment and monitoring programs, the EPA provides a 3-tiered approach to assessing wetlands. Level 1 is coarse-scale assessment based on the surrounding landscape. Level 2 is rapid, site-based assessment. Level 3 is intensive, site-based assessment. Most states draw on one or more of these levels to fulfill their requirements under the CWA (US *EPA, 2001; US EPA 1995; Stetson, 2008).*

2.1.2 Designated Uses

The water quality standards required through section 303 of the CWA are used to identify designated uses for each water body, as well as establish criteria and requirements to protect and maintain the health of the water body. Sections 101 and 303 of the CWA address Designated Uses. Water bodies may fall under one or more of the following designated uses:

1. Protection and propagation of fish, shellfish and wildlife (aquatic life use)
2. Recreation

3. Public drinking water supply
4. Agricultural, industrial, navigational and other purposes.

(US *EPA, 1995;* USFWS, 2013; US EPA, "History of the Clean Water Act")

The designated use most commonly applied to wetlands is "Protection and propagation of fish, shellfish and wildlife" but is generally simplified to "aquatic life use" (US EPA, "Standards for Water Body Health"). Numerous wetland functions and values such as water storage, slow-water release, nutrient retention and cycling, sediment retention, etc., are important wetland functions but not listed as designated uses in sections 101a and 303b. (Tiner, 2005). Wetlands are generally not used for recreation, drinking water, or the other listed purposes like rivers, lakes and other water bodies are. (US EPA, "*Impaired Waters and TMDLs: Identifying and Listing Impaired Waters")*.

2.1.3 Tiered Aquatic Life Use and the Biological Condition Gradient

In 2005 the EPA drafted a reference document to help quantify the designated uses of wetlands, further explaining the common designation for wildlife propagation use. This document was entitled "Use of Biological Information to Better Define Designated Aquatic Life Uses in State and Tribal Water Quality Standards: Tiered Aquatic Life Uses". Tiered Aquatic Life Use (TALU) is a model that expresses the quantity and condition of biological resources within a waterbody (U.S. EPA, 2005). For wetlands, this can help to quantify the aquatic life use that would otherwise be difficult to account for through traditional Designated Uses (U.S. EPA, 2005).

TALU relies on a biological condition gradient (BCG) to reflect the response of an aquatic community to individual or multiple sources of stress. A BCG is a model that assumes that there is a direct relationship between biological condition and environmental stressors. The BCG concept (Fig. 1.1) is available to states when thinking about how to use available scientific information to better define aquatic life uses for their water bodies and wetlands (U.S. EPA, 2005). The EPA highly encourages states to incorporate biological information into setting water quality standards, and supports the use of TALU in quantifying designated uses of wetlands and other water bodies.

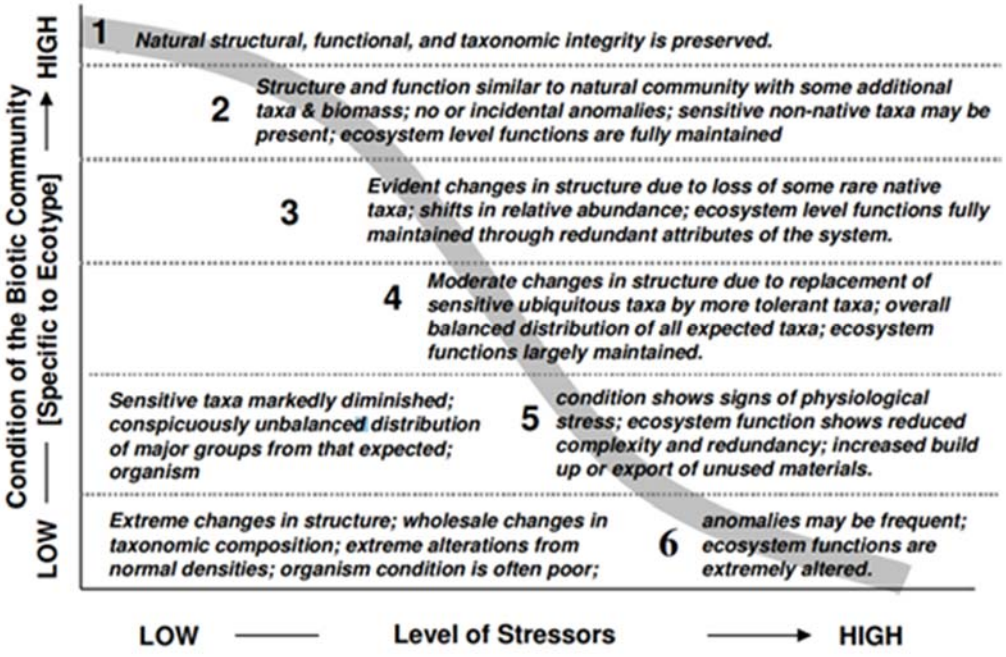

Figure 2.1: EPA Biological Condition Gradient (U.S. EPA, 2005)

2.1.4 Floristic Quality Assessment

Floristic Quality Assessment (FQA) has been proposed as a method to measure an index of biological condition, reflecting the response of an ecosystem's biological community to anthropogenic stressors. The FQA method is a vegetation-based, Level 2, rapid assessment with a long history of use in Central and Southeastern United States. The method was created in the 1970's to assess the condition of prairies, and is now being used to evaluate other ecosystems, including wetlands (Medley & Scozzafava, 2009). The FQA process for ecosystem assessment involves calculating an index of biological condition based on vegetation composition. It is assumed that this index of biological condition reflects a general stressor gradient based on anthropogenic disturbance in the surrounding landscape. Individual plant species are assigned a C-score, a number ranging from 0-10. Low C-scores indicate generalist species, as well as species that are tolerant to disturbed habitat. High C-scores indicate specialist species that are relatively intolerant of disturbed habitat. C-scores are determined by individual botanists (Bried, et al., 2012), groups of professional botanists (Chamberlain & Ingram, 2012) or combinations of botanist and state managers (Bried et al., 2012). The different states may have different C-scores for the same species, depending on who assigned the score, and how that plant interacts with the local environment (Bried et al., 2011). The distribution of high and low C-scores is assumed to be linear relative to a general stressor gradient (Chamberlain et al., 2016). In all the applications of FQA that we are aware of, non-native species are given scores of 0. There are a variety of Floristic Quality indices that use vegetation to calculate indices of biological condition, based on plant species composition and C-scores (Wilhelm & Masters, 1995). Some FQA indices exclude non-natives all together in their calculation of biological integrity.

2.1.5 Increased Interest in FQA for wetland assessment

Floristic Quality Assessment was originally intended to quantify an area's fidelity to its natural composition of flora (Wilhelm & Masters, 1995). Alternative uses of FQA's approach have been explored by scientists and managers to assess the condition of a variety of ecosystems. Research evaluating FQA performance has been concentrated mainly in the Central and Eastern regions of the United States, such as Michigan (Bourdaghs et al. 2006), North Dakota (Hargiss et al., 2017), Indiana (Rothrock and Homoya, 2005), Illinois (Bowles et al. 2006; Matthews et al,, 2005, 2015, Spyreas et al., 2012), Mississippi (Ervin, et al. 2006), Ohio (Lopez and Fennessy, 2002), Oklahoma (Jog et al., 2017), Florida (Cohen et al., 2004), upstate New York (Bried et al. 2013, Wentzell et al., 2016), Pennsylvania and the Mid-Atlantic Region (Chamberlain, et al. 2016, 2012; Nichols et al., 2006; Spyreas et al. 2012; Miller and Wardrop, 2005). Only recently have C-scores been developed for New England states (Bried et al. 2012). In addition to evaluating ecosystem condition, FQA has also been used in combination with historical data to measure changes in an area due to natural or anthropogenic disturbances (Maginel et al., 2016;), and to detect successional changes (Spyreas et al., 2012). FQA has been used in prairies and fields (Bowles & Jones, 2006; Spyreas et al., 2012; Spyreas, 2016), forests (Francis et al., 2000) forested wetlands (Bell et al., 2017; Nichols et al., 2006), emergent wetlands (Cohen et al. 2004), and a variety of other wetland types (Rothrock and Homoya, 2005; Hargiss et al., 2017; Bourdaghs et al., 2006; Lopez and Fennessy, 2002; Jog et al., 2017; Chamberlain, et al. 2016).

2.1.6 Criticism of and Support for FQA

As the FQA approach has been applied to an expanding list of ecosystem types and regions questions have been raised about some of its assumptions and its overall effectiveness (Bourdaghs et al., 2006). General support for FQA is related to its inexpensive, rapid methodology of biological assessment, and it's similar performance to more complex approaches (Bried et al. 2013). The FQA method was originally designed to eliminate a degree of subjectivity from analyses of ecosystem condition that had historically been more qualitative than quantitative (Wilhelm & Masters, 1995; Andreas et al., 2004), but now criticized for being too subjective, specifically with regard to the assignment of C values (Spyreas, 2016; Wentzell et al., 2016; Bourdaghs et al., 2006). Other concerns include the sensitivity of FQA indices to changes in C-score assignment (Spyreas, 2016), and undervaluation of non-native species (Matthews et al., 2015). There are a number of different FQA indices from which to choose, but also a lack of official guidance on which to use in various situations.

2.1.7 Stressor Gradient

According to the BGC model set forth by the EPA (U.S. EPA, 2005), the degree of the biological response should exhibit a relationship with some quantifiable gradient of stress (Bell et al., 2017). Other research investigating the efficacy of Floristic Quality Assessment used similar measures to quantify the Generalized Stressor Gradient (GSG). These stressor indices generally involved characterization of land use and land cover within a circular window around study sites. For the purpose of this research, we propose the use of the Conservation Assessment and Prioritization System (CAPS) as an expression of that stressor gradient. CAPS is a sophisticated

landscape modeling system that includes metrics for many anthropogenic stressors that affect wetlands (Jackson et al., 2017).

2.1.8 Research Objectives

The purpose of this study was to investigate the relationship between FQA indices of biological condition for forested wetlands and a generalized stressor gradient based on a level 1 evaluation of land use and landscape characteristics. I assessed the performance of Floristic Quality Assessment scores by comparing them against the CAPS characterization of the stressor gradient affecting forested wetlands in New England and New York.

The objectives of this research are:

1. Evaluate the performance of various FQA metrics and indices, and identify those that provide the best assessment of wetland condition.
2. Evaluate how variation in C-scores across states and ecoregions effects FQA performance
3. Determine whether particular plant species were more or less informative as an indicator species.

2.2 Materials & Methods

2.2.1 Generalized Stressor Gradient

The Conservation Assessment and Prioritization System (CAPS) is a level 1, coarse-scale assessment methodology developed at the University of Massachusetts, Amherst. CAPS is a

computer program and approach for conducting landscape-based assessments of ecological integrity for various natural communities, including wetlands (McGarigal et al., 2011). Ecological integrity is defined as the long-term capability of an ecological community to sustain its composition, structure and function, and thus also its resiliency to stress. The CAPS system identifies the developed and undeveloped elements of the landscape on a computer-based map and evaluates each point in the landscape for a number of stressors and landscape characteristics (Table 2.1). The results are then used by the program to calculate an Index of Ecological Integrity (IEI) for each point in the landscape relative to other points of the same ecosystem type within a specified geographic extent. Most of the metrics used by CAPS are models of anthropogenic stressors, essentially qualifying the CAPS IEI as a generalized stressor gradient. Low IEI scores represent high stress and high IEI scores representing low stress. In addition to IEI, we regressed the FQA indices against a simpler metric, habitat loss. This metric is calculated in the CAPS model by measuring the intensity of habitat loss caused by all forms of development, including agriculture, in the neighborhood surrounding the focal cell weighted by a logistic function of Euclidean distance.

Table 2.1
CAPS weighted metrics of stressors and landscape characteristics (McGarigal et al., 2011)

Metric	Description	Weight (percent)
Habitat loss	Measures the intensity of habitat loss caused by all forms of development in the neighborhood surrounding the focal cell, based on a logistic function of Euclidean distance.	8.9
Watershed habitat loss	Measures the intensity of habitat loss caused by all forms of development in the neighborhood upstream from the focal cell, based on the aquatic distance from the focal cell using on a time-of-flow model.	5.0
Road traffic intensity	Measures the intensity of road traffic (based on measured road traffic rates) in the neighborhood surrounding the focal cell, based on a logistic function of distance.	8.9
Mowing & plowing intensity	Measures the intensity of agriculture in the neighborhood surrounding the focal cell, based on a logistic function of distance.	5.0
Microclimatic alteration	Measures the adverse effects of induced (human-created) edges on the integrity of patch interiors. The metric is based on the "worst" edge effect among all adverse edges in the neighborhood surrounding the focal cell, where each adverse edge is evaluated using a "depth-of-edge" function in which the "effect" is scaled using a logistic function of distance.	5.0
Road salt	Measures the intensity of road salt application in the watershed above an aquatic focal cell weighted by road class and the modeled "influence value" for each cell, which is the aquatic distance from the focal cell based on a time-of-flow model.	5.0
Road sediment	Measures the intensity of road sediment production in the watershed above an aquatic focal cell weighted by road class (i.e., size, substrate, gradient) and the modeled "influence value" for each cell, which is the aquatic distance from the focal cell based on a time-of-flow model.	5.0
Edge predators	Measures the intensity of development associated with sources of human commensal mesopredators (e.g., raccoons, skunks) in the neighborhood surrounding the focal cell, based on a logistic function of distance to development classes.	5.0
Invasive plants	Measures the intensity of development associated with sources of non-native invasive plants in the neighborhood surrounding the focal cell, based on a logistic function of distance to development classes.	8.9
Invasive earthworms	Measures the intensity of development associated with sources of non-native invasive earthworms in the neighborhood surrounding the focal cell, based on a logistic function of distance to development classes.	5.0
Similarity	Measures the amount of similarity between the ecological setting at the focal cell and those of neighboring cells, weighted by a logistic function of distance.	8.9
Connectedness	Measures the disruption of habitat connectivity caused by all forms of development between each focal cell and surrounding cells as well as the "resistance" of the surrounding undeveloped landscape.	18.8
Aquatic connectedness	An aquatic version of the connectedness metric, measuring connectivity along streams and rivers. Aquatic connectedness includes the resistance from culverts, bridges and dams for organisms that are primarily aquatic.	2.0
Tidal restriction	Measures the magnitude of alteration to the tidal hydrology of the focal cell due to tidal restrictions.	8.9

2.2.2 Data Collection

Our study was designed to use a large data set, available to us from previous research on CAPS IEI performance (Jackson et al., 2017). We used vegetative survey data from 370 forested wetland sites throughout Massachusetts. Sites were located in the Westfield, Taunton, Millers, Concord, Chicopee, and Housatonic river watersheds (Fig. 2.2). Data were collected from 2008-2015 by teams of professional botanists. Sites were targeted for field work across the gradient of IEI values. These sites were grouped by IEI, which was broken down into deciles, and then sorted into numbered “bins”. Bin labels were randomly sorted, so that field managers and botanists selecting points to survey wouldn’t know from which IEI bin a sample point came based on its name.

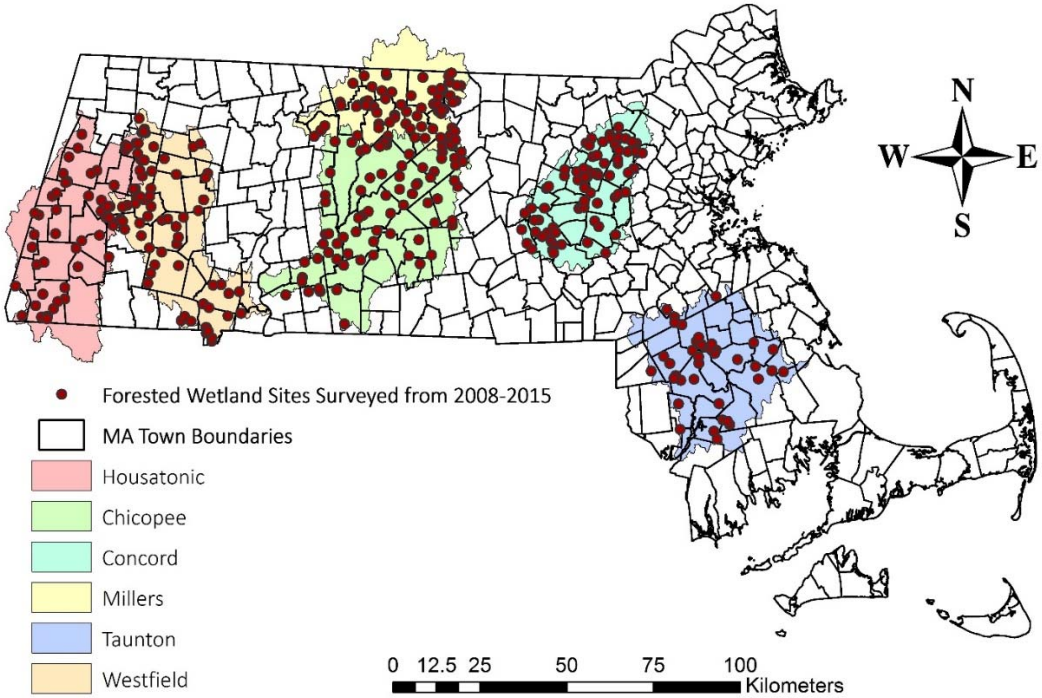

Figure 2.2: Map of sampling locations in forested wetlands in Massachusetts, surveyed from the years 2008-2015. Sites located in the Westfield, Taunton, Millers, Concord, Chicopee, and Housatonic river watersheds.

Botanists characterized the vegetation at each site by using a line-point intercept system. At each wetland site, botanists set up four 30m transects beginning at the center of the sampling point and extending in each of the four cardinal direction (North, South, East & West). Beginning at the 5m mark, the botanists walked each transect, stopping at 1m intervals along the way, and tallied plant species that intercept the line at each meter mark. After walking the transects, field botanists did an "area search": a walk around the plot to account for any species missing from the transects. Species found during the area search were given an abundance of 0.01

2.2.3 Floristic Quality Assessment

Plant data collected over these field seasons gave us the information on species composition and abundance needed to calculate indices of Floristic Quality identified from the literature (Table 2.2). Many different Floristic Quality indices have been adapted to assess site condition. One of our objectives was to determine which of these indices correlated most closely with a general stressor gradient (GSG).

We tested the correlation of 12 indices and metrics against the GSG, eight of which were equations that used C-scores to produce an FQA index of biological condition. An index here is defined as a calculation that we can interpret directly as some measure of condition, and a metric is a separate calculation that measures some aspect of the site, but does not translate directly to condition. The other four were either simple species richness or nativity metrics. The first two indices, Total FQA (FQA1) and Native FQA (FQA2), are original FQA indices developed by Swink and Wilhelm (Swink & Wilhelm, 1979). The first index, FQA1, includes non-native species in its calculation, and FQA2 excludes them. Adjusted FQA (FQA3) is weighted by the percent of native species at a site, and was developed by Miller & Wardrop in 2006. Mean C-score (FQA4), also called "Mean C," is also an original index of floristic quality (Swink & Wilhelm, 1979), and was calculated in our study as FQA4 and FQA5, with the inclusion and exclusion of non-native species respectively. Metrics FQA6 through FQA8 are simple species richness and site nativity metrics reported from the Universal FQA calculator (Freyman, 2016). The FQA9 index produces a score weighted by frequency of each species at a site (Cohen et al., 2004). The FQA10 and FQA11 indices are weighted by the species abundance (Bell 2017, and Rocchio 2007). The last index that we included in this research, Relative Non-native Cover

(FQA12), was developed specifically for this study, to investigate the potential of using non-native species abundance as an index of biological condition.

2.2.4 Coefficients of Conservatism

In order to gauge how variation in C-scores might affect the performance of FQA, we calculated the eight FQA indices using state-specific C-scores from New York and New England (Massachusetts, Connecticut, Rhode Island, New York, Vermont, New Hampshire, and Maine) (NEIWPCC). Additionally, we calculated these FQA indices using two EPA level 3 ecoregional scores in Massachusetts (ecoregion 58: Northeastern Highlands and ecoregion 59: Northeastern Coastal Zone) (US EPA, 2013). Plants in these ecoregions are assigned C-scores that are more localized than whole-state C-scores (NEIWPCC).

In order to further investigate how individual variability in C-score assignment affected FQA performance, four of our professional botanists assigned C-scores to plants based on their own personal knowledge of plants occurred in forested wetlands. Their instructions were to assign C-scores based on each species' individual sensitivity to anthropogenic disturbance. An average of these botanists' scores was then calculated for each plant species. These C-scores were used to calculate FQA4, one of the indices that performed best when state and ecoregional C-scores were used.

Table 2.2: Equations and Metrics used to Calculate and Compare Floristic Quality with a General Stressor Gradient.

Index/Metric Name	Descriptive Name	Equation	Description / Notes	Source
FQA1	"Total FQA"	$\bar{C} \times \sqrt{N}$	All species	Swink & Wilhelm 1979
FQA2	"Native FQA"	$\overline{Cn} \times \sqrt{Nn}$	Native Species only	Swink & Wilhelm 1979
FQA3	"Adjusted FQA"	$\left(\frac{\bar{C}}{10} \quad \frac{\sqrt{Nn}}{\sqrt{N}}\right) \times 100$	Weighted by percent native / non-native	Miller & Wardrop 2006
FQA4	"Total Mean C"	$\bar{C}$	Mean C-score for each site	Swink & Wilhelm 1979
FQA5	"Native Mean C"	$\overline{Cn}$	Mean C-score for native species only at each site	Swink & Wilhelm 1979
FQA6	"Total Species Richness"	N	Species Richness	Universal FQA: Freyman, 2016
FQA7	"Native Species Richness"	Nn	Species Richness of native species only	Universal FQA: Freyman, 2016
FQA8	"Percent Native"	$\frac{Nn}{N}$	Percent native species	Ervin et al. 2006
FQA9	"Frequency-Weighted FQA"	$\frac{\sum(C \times RF)}{N}$	Weighted by abundance.	Cohen et al. 2004
FQA10	"Cover-Weighted Mean C"	$\frac{\sum(C \times MC)}{TC}$	Weighted by its proportion of cover.	Bell et al. 2017
FQA11	"Cover-Weighted FQA"	$\frac{\sum(C \times MC)}{TC} * \sqrt{N}$	Weighted by its proportion of cover.	Bell et al. 2017
FQA12	"Relative Non-native Cover"	$\frac{TCa}{TC}$	The sum of non-native species abundance divided by the sum of all species abundance	Produced for this study

C = C-score for species at each site, *Cn* = Native C-score for species at each site, *N* = Species Richness (Number of species at each site), *Nn* = Native Species Richness(Number of native species at each site), *A* = Non-Native Species Richness (Number of non-native species at each site), *RF* = Relative Frequency (Species abundance divided by total abundance at each site), *MC* = Mean Cover of a species (Species abundance), *TC* = Total Cover (Total abundance of all species), *TCa* = Total Non-native Cover (Total abundance of all non-native species).

2.2.5 Statistical Analysis

The eight FQA indices were used to calculate site scores for each of the 370 forested wetlands used in this study. In order to express the relationship between each site's FQA score and its stressor gradient, the site scores (the dependent variable, y) were regressed against IEI scores (the independent variable, x) in a scatter plot. We expected this relationship between index and condition to be linear (Chamberlain et al., 2016), so a best fit line (FQA = a + b*IEI) was used to visualize this relationship. The best fit line expresses the strength of each relationship, reported with an r-squared value that measures both tightness of fit and slope. Significance values were not reported because we were not looking for statistically significant outcomes, only investigating the nature of the relationships between FQA and IEI. Instead, we compared differences in the reported r-squared values to find the best and worst performing indices of FQA.

2.3 Results

2.3.1 Floristic Quality indices

The FQA index which demonstrated the strongest relationship with the general stressor gradient (IEI), specifically when Massachusetts C-scores were used, was Adjusted FQA (FQA3), with an r-squared value of 0.252 (Fig. 2.3). The second strongest relationship was expressed by Mean C (FQA4), with an r-squared value of 0.245 (Fig. 2.3). In decreasing order of relationship strength with IEI, the next indices that showed a noticeable relationship with IEI include Percent Native (FQA8) with r-squared values of 0.19 (Fig. 2.4) Native Mean C (FQA5) with an r-squared of 0.176, Total FQA (FQA1) with an r-squared value of 0.131, Native FQA (FQA2) with an r-

squared value of 0.098. The other metrics and indices had either weak or no relationship to the GSG (Tables 2.3.1 and 2.3.2). Slope estimates and 95% confidence intervals were graphed along with each scatterplot. Tight slope estimates give us confidence that our calculation of the best fit linear relationship between FQA scores and the condition gradient are accurate. Wide 95% confidence intervals indicate that there is a lot of stochastic noise in the linear relationship.

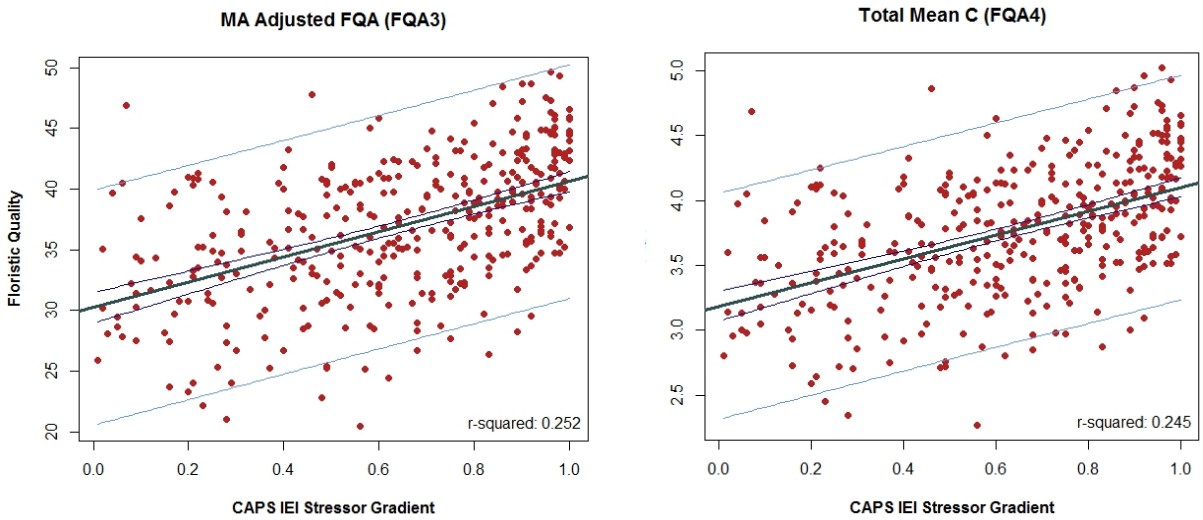

Figure 2.3: Performance of Adjusted FQA (FQA3) and Mean C (FQA4) using MA C-Scores. Model fit is reported with R^2. Slope estimates in black, 95 % confidence intervals in blue.

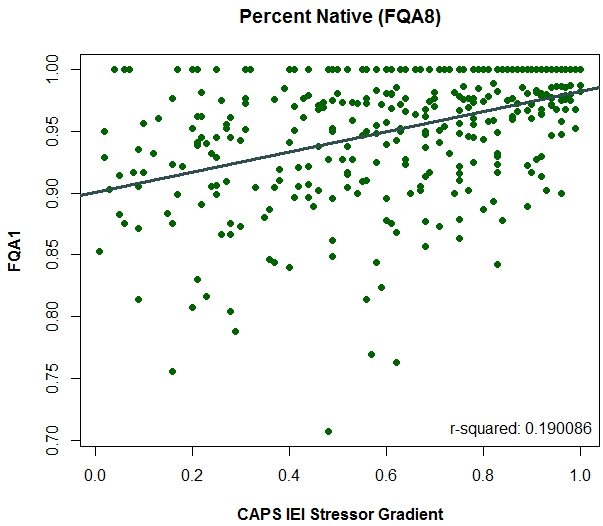

Figure 2.4: Performance of the metrics Percent Native (FQA8) using MA C-Scores. Model fit is reported with R^2. Slope estimate in black.

Table 2.3.1: Coefficient of determination (R^2) in the linear relationship between 12 different indices of forested wetland condition (FQA1-FQA12) and a generalized stressor gradient for 370 forested wetlands in Massachusetts. R^2 values for the eight indices involving Coefficient of Conservatism scores (C-scores) assigned to each plant species were derived separately using C-scores published for each of the New England states. In Massachusetts, C-scores were also assigned separately by ecoregion (MA_eco).

State	FQA4	FQA3	FQA5	FQA1	FQA2	FQA10	FQA11	FQA9
MA	0.245	0.252	0.176	0.131	0.098	0.024	0.008	0.008
MA_eco	0.216	0.186	0.149	0.112	0.078	0.143	0.089	0.007
CT	0.333	0.33	0.321	0.216	0.159	0.104	0.098	0.008
RI	0.226	0.233	0.143	0.128	0.088	0.007	0.002	0
NY	0.185	0.216	0.19	0.064	0.11	0.145	0.124	0.013
NH	0.202	0.213	0.096	0.106	0.059	0.094	0.056	0.004
VT	0.095	0.118	0.011	0.065	0.028	0.071	0.045	0.009
ME	0.129	0.152	0.018	0.062	0.024	0.06	0.041	0.002

Table 2.3.2: Performance (reported with R^2 values) of Floristic Quality Metrics (and one index – FQA12) that do not involve C-scores, and thus do not vary with State C-score assignment.

State	FQA6	FQA7	FQA8	FQA12
All C-scores	0.002	0.013	0.19	0.072

2.3.2 Coefficient of Conservatism Scores

The use of Connecticut C-scores produced stronger relationships to the condition gradient than the Massachusetts scores. New York state C-scores produced higher correlations than Massachusetts C-scores when used in the cover-weighted FQA indices. In all cases however, the relationships between IEI and cover-weighted scores were relatively weak. The overall strongest correlation between FQA and IEI resulted from using Connecticut state C-scores and Mean C (FQA4) (Fig. 2.5).

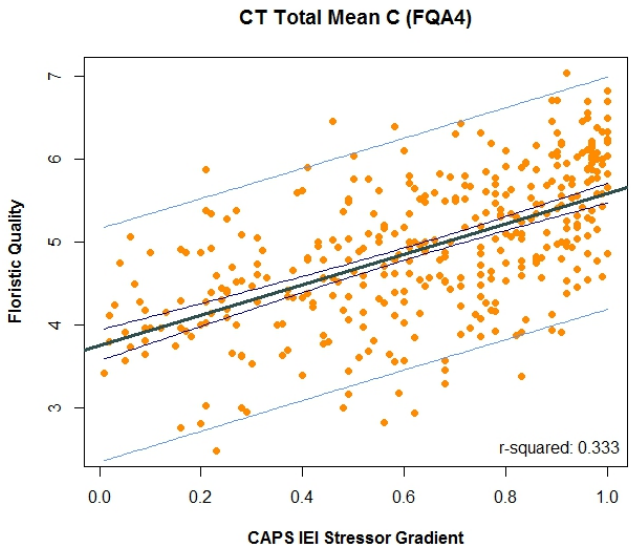

Figure 2.5: Performance of C (FQA4) using CT C-Scores. Model fit is reported with R^2. Slope estimates in black, 95 % confidence intervals in blue.

Our results showed that the use of ecoregional C-scores showed little or no improvement on the performance of FQA indices in Massachusetts, with the exception of cover-weighted indices. Slight improvements were seen (a difference in r-squared of .02) when ecoregional scores were used in FQA1: Total Floristic Quality. FQA indices 10 and 11 both saw increases in performance

when used with ecoregional scores, although these relationships were all relatively weak. For most FQA indices, statewide C-scores yielded better results than ecoregional scores.

The use of some of our botanists' assigned C-scores in the calculation of Mean C (FQA4) at each site resulted in an increase in the relationship between FQA4 and IEI (Table 2.4). The average of their species C-scores also resulted in an increase in Mean C (FQA4) r-squared scores. Using these individually-assigned and group averaged C-scores, we were able to take a closer look at individual variability in C-score assignment (Table 2.5 and 2.7) and see how that variability affects overall FQA indices (Table 2.6).

Table 2.4: Botanist C-score performance (measured with r-squared) with Mean C (FQA4), one of the two top-performing FQA indices.

	Avg Botanist C-Scores	Botanist 1	Botanist 2	Botanist 3	Botanist 4
FQA4	0.319	0.342	0.236	0.28	0.201

Table 2.5: Correlation Matrix of variation between C-scores assigned by each botanist. This shows us how similar the assignments of C-scores for each species are between botanists.

	Bot. 1	Bot. 2	Bot. 3	Bot. 4	Avg
Bot. 1	1	0.560401101	0.614733161	0.544367189	0.861773645
Bot. 2		1	0.394636066	0.61531672	0.777635049
Bot. 3			1	0.469702763	0.791852065
Bot. 4				1	0.788189246
Avg					1

Table 2.6: Correlation Matrix of Mean C (one of the top 2 performing FQA indices) comparing state-assigned C-score assignment.

	MA_eco	MA	CT	ME	NH	NY	RI	VT
MA_eco	1	0.9	0.89	0.83	0.94	0.83	0.93	0.89
MA		1	0.91	0.79	0.88	0.88	0.86	0.81
CT			1	0.67	0.82	0.8	0.89	0.72
ME				1	0.84	0.66	0.75	0.82
NH					1	0.8	0.89	0.88
NY						1	0.77	0.78
RI							1	0.81
VT								1

Table 2.7: Correlation Matrix of Mean C (one of the top 2 performing FQA indices) comparing our botanist-assigned C-score assignment.

	Bot.2	Bot.1	Bot.4	Bot.3
Bot.2	1	0.79	0.87	0.79
Bot.1		1	0.73	0.9
Bot.4			1	0.69
Bot.3				1

2.3.3 Native and Non-Native Species inclusion

Calculations of FQA, when done with native species only and using MA C-scores, had similar or weaker relationships with site condition than when all species were included (Table 2.3.1). In the case of Total vs Native FQA (FQA1 & 2), the removal of non-native species from the calculations resulted in a decrease in r-squared from 0.131 to 0.098. In the case of Total vs Native Mean C (FQA4 & 5), the removal of non-native species from the calculations resulted in a decrease in r-squared from 0.245 to 0.176. The metrics Relative Non-Native cover (FQA12) resulted in small positive relationships with IEI, with r-squared value of 0.072. The overall strongest FQA metric

(FQA3) for MA and the other states was a version of FQA that was weighted by the percentage of native species.

2.3.4 Habitat Loss Metric

In order to assess the ensure the validity of our CAPS IEI gradient, we also wanted to explore the relationship between FQA and a simple metric of anthropogenic disturbance similar to those used in other studies. When compared against habitat loss as the GSG, FQA scores performed comparably or worse than they did when compared against our CAPS IEI gradient.

3.5 Variation in Species C-scores

In order to investigate potential areas for improvement of C-score assignment, we calculated the frequency of occurrence for some of the plants with highly frequencies of occurrence to examine how variable the C-scores are. Frequency of occurrence was calculated by dividing the number of sites at which a species occurs by the total number of sites in the study. The top 10 plant species in this study with the highest frequency of occurrence, and their ranges of possible C-scores in New York and New England are shown in Table 2.8.

Table 2.8: Species frequencies of occurrence, and corresponding state-assigned C-scores (MA, CT, ME, NH, NY, RI, and VT).

Taxon	Frequency	Total Sites	Frequency of Occurrence	MA	CT	ME	NH	NY	RI	VT	C-Score Difference
Acer rubrum	367	371	0.99	2	3	2	3	2	3	2	1
Ilex verticillata	332	371	0.89	5	5	3	3	4	3	7	4
Maianthemum canadense	297	371	0.8	3	5	2	2	4	3	3	3
Pinus strobus	284	371	0.77	2	3	2	2	4	5	4	3
Vaccinium corymbosum	252	371	0.68	4	5	4	4	5	4	7	3
Rubus hispidus	244	371	0.66	2	4	2	2	5	3	4	3
Aralia nudicaulis	221	371	0.6	3	5	4	4	3	5	4	2
Fraxinus americana	210	371	0.57	4	7	2	5	3	5	3	5
Coptis trifolia	210	371	0.57	6	8	4	5	5	7	4	4

2.4 Discussion

2.4.1 Floristic Quality indices

One of our two best performing FQA indices was the Adjusted FQA (FQA3). Adjusted FQA was created by Miller and Wardrop in 2006. To test its performance, they regressed Adjusted FQA against an index of human disturbance based on an assessment of surrounding land use, buffer characteristics, and an assessment of on-site stressors. Miller and Wardrop reported an r-squared score of -0.87 from this correlation. The relationship between Adjusted FQA and our stressor gradient produced much lower r-squared scores of 0.252 with MA C-scores and r-squared scores of 0.33 with CT C-scores. Since its first appearance in 2006, Adjusted FQA has appeared in few studies that comparatively test the different Floristic Quality Indices (Chamberlain and Brooks, 2015). Despite its recent creation, Adjusted FQA is featured as one of options in the NEIWPCC online FQA calculator (Freyman, 2016).

The other FQA index that performed relatively well in our study was Mean C (FQA4), one of the two original Floristic Quality Assessment indices created by Swink & Wilhelm (1979). Mean C-scores have been commonly used to determine floristic quality (Miller and Wardrop 2006; Spyreas, 2016; Miller and Wardrop, 2006; Bried et al., 2013; Chamberlain et al., 2015; Bell et al, 2017). However, the use of Mean C has also been criticized for not being informative enough. Miller and Wardrop in 2006 warned that the use of Mean C-scores alone was misleading. In their research adapting FQA to wetland assessment in Pennsylvania, they found many sites with similar Mean C-scores that had a wide range of disturbance ranks, FQA index scores, and native species richness. However, this index has been applauded for its consistent performance in other studies. Chamberlain and Brooks (2015) praised Mean C, saying that it lacked influence from sample size and species richness that may cause bias in Total FQA (FQA1). In 2016, Spyreas showed that Mean C allowed for consistent floristic quality analysis among varying plot sizes, species detectability, rates of species misidentification, and sample year in their analysis. Our Mean C-scores are relatively consistent with varying C-score assignments as well, but we did not test the degree of that consistency. We also found that similar Mean C-scores can be found in sites with wide ranging IEI ecological integrity scores.

In contrast to some of these better-performing FQA indices, specific indices that performed poorly against the generalized stressor gradient in our study included FQA2, FQA10, FQA11, and FQA9. Of those listed, FQA2, FQA11, and FQA9 include some measure of species richness in their equations, which may be affecting their performance. Miller and Wardrop in 2006 noted that Total FQA (FQA1), Mean C (FQA4) and Adjusted FQA (FQA3) were highly sensitive to species

richness, and that sites with larger species richness values always scored higher than those that had fewer species present. Maginel et al., in 2016 addressed how Total FQA (FQA1) was strongly influenced by the calculation of species richness, and found that their own calculations of Mean C were unable to differentiate between burned/unburned sites when species richness was high.

The relationship between species richness and FQA is influenced by size of the assessment area (Matthews et al. 2005). In their research, they found that FQA is affected by spatial landscape attributes such as isolation and area (which impact species richness), weakening the relationship between disturbance and plant species composition. This was the same reasoning explained by Bried et al., in 2012, when they suggested that variability in species richness is misleading because it is influenced by other factors unrelated to human disturbance such as area size, seasonality, and sampling effort. Bourdaghs et al. in 2006 also noted that size of the site matters, and that FQA reliability increases with sampling area.

Many of the indices that performed poorly in our study (FQA9, FQA10, and FQA11) were weighted by abundance (percent cover or frequency). The cover-weighted Mean C (FQA10) and cover-weighted FQA (FQA11) show very weak relationships with the stressor gradient. However, in 2016 these two indices were tested by Bell, et al., and the cover-weighted Mean C correlated very closely with their calculation of Ecological Integrity Assessment scores. The Ecological Integrity Assessment used in Bell et al. is different from our CAPS-based Index of Ecological Integrity. The assessment used by Bell et al. measured various ecological factors including landscape, buffer, size, vegetation, hydrology, and soil metrics. Another index, cover-weighted FQA (FQA11) did not perform as well, and they instead recommended the use of Mean C or

Weighted Mean C for wetland assessment purposes (Bell et al., 2016). In a study by Cohen et al. (2004), when abundance-weighted FQA (FQA9) was fit to a model comparing it to a different landscape disturbance index, it performed comparatively well with Mean C. Due to the extra effort involved with calculating frequency in FQA9, they ended up excluded it from their subsequent analyses (Cohen et al., 2004). It is difficult to understand how indices that only account for the presence/absence of species could out-perform indices that take into account species abundance, if the plants used in FQA are good indicators of ecosystem condition. The failure of cover or abundance-weighted indices might indicate some fundamental problem with the current approach for assigning C-scores. For example, if large quantities of uninformative species were present and weighted by their abundance, it would add noise to the relationship between FQA and the stressor gradient.

2.4.2 Coefficient of Conservatism Scores

The process of calculating Floristic Quality Assessment (FQA) is based on the assignment of C-scores for plant species growing in a region of interest. The assignment of C-scores has often been regarded as subjective, and an important weakness of the FQA process. Bowles and Jones (2006) considered C-scores to be highly biased. In their research with fire-managed systems, they suggested that the inability of FQA and Mean C values to detect negative changes at their sites was most likely due to deficiencies with C scores.

FQA operates under the assumption that all species are indicators, and each should have a single assigned C value. When Nichols et al. investigated FQA's ability to detect human disturbance in

Virginia hardwood flats, they concluded that the C-scores for woody plants do a particularly poor job at distinguishing condition. Matthews et al.(2005) measured the extent to which woody species were undervalued by graphing the relationships between species C-scores and the average of the C-scores of the species with which they co-occurred. A paper published in 2004 by Cohen et al. investigated the process of assigning C-scores to individual plant species. They tested the variability in C-scores assigned by 10 individual botanists vs. scores assigned by a panel of experts, reporting "significant disagreement between botanist opinions" (Cohen et al., 2004). The mean pairwise correlation between botanists was 0.62. We had the four professional botanists who collected data for our research independently assign C-scores to species found at sites used in this study. We also found the variation between C-score assignments to be high (Table 2.5 scores 0.394-0.861). In 2005, Rothrock and Homoya were also interested in researching the consistency of C-score assignment. They compared variation in C-scores between Illinois and Indiana, stating that in practice, variation in C-scores between neighboring states should have little impact on the calculation of FQA indices, but their research did not support this. They found that more than a third of species in Indiana diverged from Chicago species C-scores by 1-3 coefficients of conservatism, and that there was more variation in middle range scores (3-4) than extreme values in low/high scoring species. Rothrock and Homoya did not actually test how this variation in C-scores affected FQA index scores, but our research tested that relationship among C-scores from states in the Northeast. Our results indicate that FQA is highly sensitive to changes in C-scores.

A basic assumption of C-scores is that they become less indicative the further away you use them from their intended geographic range (Wilhelm and Masters, 1995). In order to improve the assignment of C-scores, Bried et al. (2012) recommended that C-scores be assigned by ecoregion

to improve their ecological accuracy. The justification for assigning ecoregional C-scores is that species' ranges and responses to ecological setting are not confined to state boarders (New England Interstate Water Pollution Control Commission, 2018). Under this assumption, we would expect that ecoregional C-scores would out-perform the general Massachusetts C-scores, and that Massachusetts C-scores would perform better when calculated in Massachusetts wetlands than C-scores from other states. Our results contradict these assumptions. We found that when ecoregional scores were used in place of MA scores, the only improvement in the relationship between FQA and the disturbance gradient occurred when cover-weighted indices were used. However, in our tests, cover-weighted indices performed quite poorly overall compared to the other indices. In all the other FQA indices, ecoregion-assigned C-score perform worse than overall Massachusetts Scores. Also contrary to general C-score assumptions is the fact that using Connecticut C-scores to evaluate Massachusetts forested wetlands resulted in higher correlations with the stressor gradient than when Massachusetts C-scores were used.

At both the state or ecoregional level, C-scores are assigned to plants on the premise that they indicate disturbance equally well across many ecosystem types (Wilhelm and Masters, 1995). Bell et al. (2017) warned that community specific studies are necessary to identify local quality thresholds for habitat quality. Further, the decision to have separate C-scores for different states/ecoregions acknowledges that vegetative communities are determined by more than just stressor gradients; the role of abiotic factors and ecological setting are also important. Difference between one ecoregion and another (or one state and another) are probably small compared to differences among ecosystems (e.g. salt marsh and freshwater tidal marsh, or salt marsh and forested wetland). FQA might be more effective if C-scores were assigned based on ecosystem, in

addition to ecoregion, as suggested by Jackson et. al, 2017. In that study, an empirically derived, vegetation-based index of biological integrity specifically developed for forested wetlands performed poorly when applied to shrub swamps.

2.4.3 Native and Non-Native Species Inclusion

Another interesting result of this study was how the various FQA indices performed relative to the inclusion/exclusion of non-native species in FQA calculations. We had three pair of indices that were calculated both with and without non-native plants. In the case of Total vs Native FQA (FQA1 and 2), the removal of non-native species from the calculations resulted in a decrease in r-squared from 0.131 to 0.098. In the case of Total vs Native Mean C (FQA4 and 5), the removal of non-native species from the calculations resulted in a decrease in r-squared from 0.245 to 0.176. This improvement in model fit with the inclusion of non-native species is consistent with other studies (Cohen et al. 2004; Miller and Wardrop, 2006; Francis et al., 2000).

Original calculations of Floristic Quality omitted non-native species from their analysis, because non-native species were deemed uninformative for defining natural areas (Wilhelm & Masters, 1995). Since then, many have chosen to include non-native species in FQA calculations when assessing site condition, assigning all non-native C values of zero. Miller and Wardrop, in their 2006 paper, describe the interaction between non-native species and wetland condition by discussing how the quantity of non-natives at any site influences the quality of that site, pointing out that poor quality sites invite the establishment of invasive plants. They concluded that non-native plants are always associated with a decrease in site quality, and that that decrease should be

accounted for in FQA calculations. Miller and Wardrop (2006) developed an "Adjusted FQA" index to be sensitive to non-native species richness versus total species richness. Adjusted FQA (FQA3) performed very well in that study when compared to a score of disturbance (r-squared = -0.87). In 2012, Spyreas acknowledged the informative nature of non-native species and their negative effects on floristic quality through strong invasions. Matthews et al., (2014) suggested that including non-natives in comparisons of FQA indices to a disturbance gradient resulted in slightly improved r-squared values. In 2004, Cohen et al., found that the use of nonnative taxa resulted in minor improvements in model fit of Total FQA (FQA1) and Mean C (FQA4) when compared to landscape development intensity.

2.4.4 References for evaluating FQA

Studies testing FQA performance have generally relied on calculations of landscape disturbance as the basis for testing FQA (Bried et al., 2013; Miller and Wardrop, 2006). There is some disagreement on the use of buffer zones and simple land-use analyses as the basis for evaluating FQA. Jog et al. (2016) stated that land use within a buffer zone does not accurately indicate biological integrity at the 200-m scale. They conceded that due to the limited nature of their study, larger scales might be more informative. When Lopez and Fennessey (2002) used a disturbance gradient that took land use intensity within a 100 ft. buffer into account, they found it had a strong relationship with FQA scores (p = -0.695). Bried et al. in 2012 used an estimation of buffer degradation within a 100 ft. buffer around the plot perimeter, coupled with a land use intensity analysis within 500 ft. to quantify disturbance. They also reported strong relationships between Mean C and this disturbance gradient.

In order to make sure the rigorous, Level-1 CAPS approach wasn't an overly complicated way to quantify the GSG (IEI scores), we also evaluated FQA index scores using a simple "habitat loss" metric, similar to GSGs used in other studies. The resulting relationships between FQA indices and habitat loss were not stronger than those between FQA indices and IEI. Therefore, we concluded that the relatively weak relationships that we found compared to other studies were not due to some shortcoming of the IEI-based GSG.

There are benefits and disadvantages to using each of the different levels of wetland assessment set forth by the EPA. Land-use data in Level-1 does not fully explain fine-scale, local interactions without site-level confirmation (Hargiss et al., 2017). Level-1 assessments are useful for understanding watershed-scale effects on the condition of a site, provide a broad understanding of the variety of stressors affecting a location, and can provide comprehensive assessments of all wetlands with a watershed. Level-3 assessments provide in-depth understanding of individual site condition and function, but are costly and often require multiple visits and taxonomic expertise that limit the ability to use them broadly across the landscape (Hargiss et al., 2017). Level-2 assessments tend to be less detailed/more subjective than level-3 assessment methods, but are also cheaper and faster to implement than the other two levels (Hargiss et al., 2017). Although it is generally infeasible to use rapid assessment methods (RAMs) to comprehensively assess all wetlands in a watershed or state, they are more suitable than intensive methodologies for assessing large numbers of sites.

These different levels of wetland assessment serve different purposes, and used individually are likely to give an incomplete assessment of wetland sites. When Level-2 assessments are validated using Level-1 assessments, any potential improvement of that Level-2 assessment brings it closer to replicating the landscape-level assessment. The Level-2 assessment is then at risk of becoming redundant with the results of the Level-1 assessment. To be useful as a complement to level-1 assessments, level-2 assessment protocols should be validated by, and calibrated to, Level-3 assessments" (US EPA, 2006).

We recommend that Level-2 rapid assessment methodologies be developed from, and tested against more comprehensive site-based assessment methodologies. The kind of circular logic that comes about when Level-2 methods are evaluated against the results of other simple assessment methods is evident in many studies that seek to create an index of wetland condition based on biological data. New indices are tested against a landscape level or rapid assessment methods that use stressor metrics as a surrogate for true indicators of condition. If we only consider indices valid if they replicate stressor gradients, then why do we need them, when quantifying human disturbance is so much easier and can provide comprehensive coverage for all wetlands? Research is needed to test the efficacy of FQA and other rapid assessment methodologies using Level-3, comprehensive assessment methodologies based on condition, rather than stressor, metrics.

2.4.5 Proposals for Improving FQA

A few improvements have been suggested for FQA in the literature. Bried et al. (2012) recommended that sites should be sampled throughout the growing season in order to account for

variability in species detectability. Chamberlain et al., in 2015, suggested that using only dominant species will provide similar Mean C-scores with less work. They defined dominant species using the 50/20 rule developed for wetland delineations, and found little change to Mean C-scores when all grammanoids were removed from the analysis. However, it is possible that use of only dominant plants would place more weight on widespread species that occur in both degraded and relatively pristine sites. An example of a ubiquitous and potentially uninformative species is red maple (*Acer rubrum*). This species was found at 99% of the forested wetland sites in our research. It accounts for >25% cover at 60 percent of our sites. As a widespread species in forested wetlands, red maple is tolerant of disturbance, but it is not necessarily indicative of disturbance.

Ervin et al.,(2006) suggested that a wetness index be used instead of coefficients of conservatism, so that places without established local C-scores would be able to calculate floristic quality. They created an index called FAQWet that replaced C-scores with a numeric system that rewarded plants found more frequently in wet environments. FAQWet performed similarly to FQAI in that study, but neither index was correlated to a high degree with indices of human disturbance. The Army Corps of Engineers cautions against the use of wetland indicator status as an indicator of site condition, because those categories were designed to show how often plants occur in wet areas, not to indicate the degree of wetness for a site.

2.4.6 Management Implications

One of the goals of this research was to evaluate the performance of various FQA metrics and indices, and identify those that provide the best assessment of wetland condition. The FQA indices that performed worse in our study also performed poorly in other studies (Miller and Wardrop, 2006; Bell et al., 2017, Chamberlain et al., 2016; Bried et al., 2013). Consistent poor performance suggests that Total FQA (FQA1), Native FQA (FQA2), Total Species Richness (FQA 6), and Native Species Richness (FQA7) should not be used for assessing wetland condition. These equations all involve species richness calculations, which may be impairing their performance. We also recommend against the use of Native Mean C (FQA5), because of the superiority of its counterpart, Total Mean C (FQA4). The abundance-weighted indices which performed poorly in this study, Frequency-weighted FQA (FQA9), Cover-weighted Mean C (FQA10) and Cover-weighted FQA (FQA11) have been shown to perform well in other studies (Cohen et al., 2004; Bell et al., 2017), so we believe that there may be potential to further develop abundance-weighted FQA indices even though they had little relationship with the GSG in our study.

Although none of the relationships between FQA indices and our GSG were particularly strong, the indices that performed better than others, Adjusted FQA (FQA3) and Mean C (FQA4), may be worthy of further development as rapid assessment methods. In some regions, plants were rated by the confidence botanists had in assigning C-scores (Freyman et al., 2016). Weighting C-scores by the confidence scores might improve the performance of these indices.

One option for improving abundance-weighted indices might be to identify and drop species that are relatively uninformative. These are species that show little or no relationship with human disturbance or stressors. For indices where species are weighted by abundance, uninformative species might cause excessive noise in the relationship between FQA and the stressor gradient, especially if those species are commonly encountered and often abundant. Alternatively, species C-scores can be weighted by how strong each plant is as an indicator of condition.

Our research suggests that the performance of the different FQA indices is dependent on the quality of C-scores. Improvement in C-score assignment has the potential to significantly improve the performance of FQA indices. Reliance on the professional, yet subjective, judgement of botanists for C-score assignment undermines confidence in the indices. The whole process would be strengthened by adopting an empirical approach for assigning and testing C-scores. Further, C-scores assignments should vary not just by geography but should also be developed independently for each major wetland system in which they will be deployed.

The noise that we see in our data could be due to natural variation, a design flaw in FQA indices, or the need for better approaches for assigning C-scores. This suggests shortcomings in current approaches to FQA and their ability to accurately assess site condition. The Floristic Quality Assessment method was designed as a tool for assessing the condition of sites through an evaluation of the vegetative community. Ervin et al., (2006) recommended that it would be more appropriate to use FQA for monitoring sites over time, as opposed to point-in-time sampling. Hargiss et al. (2017), concluded that FQA would be best used in combination with other levels of assessment, based on the needs of the surveyors. The authors concluded that it was necessary to

conduct in-depth assessments often enough to verify that the assessment levels are in agreement. Whether or not FQA performs well enough for use in a comprehensive wetlands assessment and monitoring program depends on how it will be used in combination with other assessment methodologies. At this point, it is unclear how FQA fits in with the EPA recommended three level approach to wetlands assessment.

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