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Development and Application of a Fish-Based Index of Biotic Integrity for Lakes in Eastern South Dakota

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DEVELOPMENT AND APPLICATION OF A FISH-BASED INDEX OF BIOTIC INTEGRITY FOR LAKES IN EASTERN SOUTH DAKOTA

BY

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DEVELOPMENT AND APPLICATION OF A FISH-BASED INDEX OF BIOTIC INTEGRITY FOR LAKES IN EASTERN SOUTH DAKOTA

This thesis is approved as a creditable and independent investigation by a candidate for the Masters of Science degree in Wildlife and Fisheries Sciences and is accepting for meeting the thesis requirements for this degree. Acceptance of this does not imply that the conclusions reached by the candidates are necessarily the conclusions of the major department

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This thesis is dedicated to Daniel Ethington, a great father, grandfather, outdoorsman, and amazing man all around.

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ABSTRACT

DEVELOPMENT AND APPLICATION OF A FISH-BASED INDEX OF BIOTIC INTEGRITY FOR LAKES IN EASTERN SOUTH DAKOTA

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The Index of Biotic Integrity (IBI) was developed to summarize the impact of watershed and water quality degradation to biotic communities and help to provide a more complete picture of water quality changes that are not captured in traditional physical and chemical measurements. In 2014, nearly 15% of South Dakota lakes did not meet their designated beneficial uses based on water quality measures but the impacts to the fish communities are unknown. The goal of this study was to develop a fish-based IBI for eastern South Dakota lakes by addressing three specific objectives:

1) Determine whether an IBI could be developed for smaller lakes (surface area = 100 – 1,000 ha) using extant annual standardized fish community sampling data;
2) Evaluate whether the inclusion of small-bodied fishes improves the smaller lake IBI; and
3) Evaluate whether lentic IBIs should be developed for smaller and larger (>1,000 ha) lakes separately.

Extant fish community data was collected from annual surveys conducted by the South Dakota Department of Game, Fish and Parks between 2011 and 2015 and were either used alone or in combination with shoreline seine samples from 17 lakes. For the first two objectives, potential metrics were screened to select metrics that best described fish community responses to watershed degradation. The third objective was addressed by applying the IBI based on the results of the first two objectives to larger lakes. The IBI developed for smaller lakes based on extant data alone resulted in four final metrics, (e.g., proportion of Centrarchidae, percent insectivores, percent intolerant species, and proportion of Ictaluridae) and was not improved with the inclusion of smaller-bodied fish collection data. Further, there was no difference between the performance of IBIs developed for smaller and larger lakes. These results show that lentic IBIs may be developed based on extant data with no additional sampling required. Thus, a history of IBI trends may be calculated for eastern South Dakota lakes to evaluate fish community changes in response to watershed land-use over time. Such long-term data may be used to identify and prioritize interventions that may improve water quality and fish communities.

CHAPTER 1. INTRODUCTION

Anthropogenic disturbances within watersheds such as urbanization, grazing, and row crop agriculture have altered U.S. waters since European settlement, and the rates of such change have increased as human populations have grown over time (Bolstad and Swank 1997; Harding et al. 1998; Knox 2010). The United States' Clean Water Act of 1972 was enacted in response to degradation of the nation's waterways and calls for the maintenance and restoration of the chemical, physical, and biological integrity of surface waters. Determination of water quality historically has been primarily determined from measures of chemical and physical characteristics (Karr and Dudley 1981), but such assessments may not be an adequate reflection of the impact of human disturbance on the biotic integrity of those systems (Karr 1981, 1994; Beck and Hatch 2009).

A commonly used assessment of aquatic biological integrity is the Index of Biotic Integrity (IBI; Karr 1981). The IBI was first created using fish-based indices to numerically express the "health" of aquatic ecosystems. Fish are particularly advantageous as biological indicators for several reasons:

1) Fish communities are composed of a wide array of trophic guilds and can indicate watershed impairments to aquatic food webs;
2) Extensive knowledge of the life history of fishes can be used to predict individual species' responses to habitat degradation; and
3) Fish community responses to habitat degradation or improvement is readily understood by the public (Beck and Hatch 2009).

Monitoring IBIs over time can indicate impairments or improvements in water quality as fish communities respond. For example, Lyons (1992) showed that the number of top carnivore species declined as environmental degradation (i.e., water quality, siltation, increased turbidity, and channelization) increased in Wisconsin warmwater streams and rivers. Conversely, increased IBI scores in coldwater streams indicated shifts to disturbed states as species richness increased over a time period due to introductions and invasions of warmwater species (Lyons et al. 1996). Over the past three decades, many states have adopted the use of fish-based IBIs for biological monitoring either in legislation or policy (Beck and Hatch 2009).

To date, most IBIs have been created for lotic systems due to the relative ease of their development and application. These indices are created based on what fish communities would be expected under relatively undisturbed conditions (Plafkin et al. 1989; Whittier 1999). In this case, waterbodies should be as similar as possible in terms of size, habitat, or other classifications when developing an IBI (Plafkin et al. 1989). Most lotic IBIs have been developed by ecoregion for this reason. Lentic systems may differ widely in regard to physical, chemical, and biological characteristics even within the same ecoregion (Whittier 1999). Thus, development of IBIs for a single ecoregion may be problematic or may need to be developed based on other classifications.

Additionally, stocking may override the expected relationships between watershed disturbance and fish assemblage structure. For example, stocking could artificially increase the number and abundance of less tolerant fishes (e.g., top carnivores) or decrease the number of native species through competition and predation (Whittier and Hughes 1998). Previous research has noted that stocking may or may not have an influence on IBI development in lentic systems. For example, Drake and Valley (2005) found that IBI scores were significantly lower for lakes stocked with two or more top predators, but Drake and Pereira (2002) found that lake IBI scores did not differ between lakes with different stocking intensities.

Development of fish-based IBIs in lentic systems may be further inhibited by sampling strategies. Fish community sampling in lentic systems often fails to target all habitats, and, thus, the entire fish assemblage, within the system. Sampling of lake fish communities is often confined to specific locations such as shoreline transects or selected net sites. Finally gears used to sample fish communities may be biased for or against some species (Weaver et al. 1993). Missing species occurrences or under- or overestimating the relative abundance of species may influence metric selection in IBI development.

Despite these limitations, IBIs for lentic waters have been developed, tested, and used over time, including those for Great Lakes littoral zones (Minns et al. 1994), Wisconsin inland lakes (Jennings et al. 1999) and Minnesota inland lakes (Drake and Pereira 2002; Drake and Valley 2005). Lentic IBIs are becoming more common as management agencies recognize that traditional water impairment measures for lentic waters such as trophic state index (TSI; Carlson 1977) and total maximum daily loads (TMDL) fail to capture the effects that anthropogenic perturbations have on biological communities (Beck and Hatch 2009).

To increase the likelihood of successfully developing a lentic fish-based IBI, lakes may need to be classified in some way, though such classifications have not been explored. Surface area is one classification option as lake size is generally positively related to species richness (Matuszek and Beggs 1988). However, greater species richness could artificially indicate better water quality in lakes with greater surface areas

(Smoger and Angermeier 1999, Whittier 1999; Drake and Pereira 2002). To date only Minnesota has developed and tested separate IBIs based on lake size in the same geographic region and found that lake size had little influence on IBI development and use (Drake and Pereira 2002; Drake and Valley 2005).

The development of a lentic fish-based IBI is timely in South Dakota for several reasons. First, recent progress has been made in the development of lotic IBIs across the state (Krause et al. 2013). Secondly, recent watershed land use changes may be influencing water quality in lakes, particularly in the eastern half of the state. Presently, South Dakota has 572 lakes and reservoirs designated for fishery and aquatic life beneficial uses. In 2014, the South Dakota Department of Environment and Natural resources found that at least 85 lakes (14.9%) did not meet their designated beneficial uses based on water quality sampling (SDDENR 2014). Continued watershed degradation, mainly conversion to row crop agriculture, has subsequently impacted fisheries in eastern South Dakota through sedimentation and eutrophication (SDGFP 2014). Row crop agriculture dominates land use in eastern South Dakota and the expansion of row crop agriculture will likely continue. Such land use change could continue to increase the number of lakes that may not meet their designated beneficial use (Sohl et al. 2012; Paul 2016), but information on these impacts to the aquatic biotic community within these watersheds is currently lacking.

Development of a fish-based IBI for lentic systems based on standardized sampling would allow for a rapid assessment of biological integrity over time. Standardized data records date back about two decades and will continue to be collected in future years. Thus, long-term lentic fish community data can be used to develop a historic perspective of biotic community integrity and provide information future impairments or improvements in water quality. Comparisons among years and between lakes of similar sizes will assist managers, scientists, and policy makers in recognizing and quantifying anthropogenic impacts within watersheds.

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CHAPTER 2. DEVELOPMENT OF A FISH-BASED INDEX OF BIOTIC INTEGRITY FOR LAKES IN EASTERN SOUTH DAKOTA

INTRODUCTION

The Clean Water Act of 1972 emphasizes the maintenance and restoration of the chemical, physical, and biological quality of U.S. surface waters. Assessments of water quality most often focus on chemical and physical characteristics of waterbodies, but these characteristics may not fully capture impairments or improvements to water quality (Karr 1981). For example, watersheds that exceeded 80% agricultural land use in Wisconsin were determined to maintain good water quality and habitat over the course of eight years (Wang et al. 1997).

In the past several decades, greater emphasis has been placed on the biological community, namely fish, as an indicator of aquatic ecosystem health. The inclusion of fish community information provides a more holistic measure of water quality when combined with traditional physical and chemical measurements (Karr 1994). The use of fish as bioindicators is specifically beneficial for several reasons. First, fish communities are composed of multiple trophic guilds and may thus reflect changes in food web dynamics in response to water quality changes. Secondly, extensive knowledge of the life history of fishes and their likely responses to degradations in water quality is available. Finally, fish communities can serve as a bridge to public understanding of water quality and watershed disturbance issues (Beck and Hatch 2009). The first efforts to an index of biological integrity (IBI) using fish was developed by Karr (1981). Since that time, other IBIs have been developed across various ecoregions (i.e., Lyons 1992; Bramblett et al. 2005; Whittier et al. 2007; Krause et al. 2013).

To date, most IBIs have been developed for lotic ecosystems. These IBIs are developed based on what fish communities should be present in relatively undisturbed waters and watersheds (Plafkin et al. 1989; Whittier 1999) and how fish communities may change in response to watershed perturbations and subsequent changes in water quality. In order to focus on such fish community changes, IBIs must be developed over systems that are similar in size, habitat, or other characteristics (Plafkin et al. 1989). The development of an IBI by ecoregion has become common practice to minimize variation between these various characteristics.

Development of lentic IBIs has lagged behind those of lotic systems for various reasons. First, lentic systems within the same ecoregion may vary widely in abiotic and biotic characteristics compared to lotic systems within the same ecoregion (Whittier 1999). For example, larger lakes generally tend to provide more habitat which supports larger number of species (Tonn and Magnuson 1982; Tonn et al. 1983). Increased species richness in larger lakes may mask how watershed degradation affects fish communities (Jennings et al. 1999). Secondly, stocking may also alter fish assemblage structure, both directly and indirectly (Brown and Moyle 1991; Fausch 1998). For example, Drake and Valley (2005) found that Minnesota lakes that were stocked with two or more predator species had significantly lower mean IBI scores. Conversely, Drake and Pereira (2002) found that predator stocking had no effect on IBI scores in Minnesota lakes.

Third, lentic sampling methods may not sample all habitats or select for or against certain fishes. Evaluating various IBI metrics usually requires that the entire fish assemblage is represented and that relative abundance is adequately represented (Whittier 1999). While many lentic sampling protocols often use multiple gears to sample fish communities, some habitats and species may still be missing and other species may be over or under represented (Fischer and Quist 2014). Combining relative abundance information across multiple gears is also not possible as different gears select for different species and different levels of effort are used (Weaver et al. 1993).

Despite these limitations, IBIs have been developed previously for lentic communities across North America (e.g., Minns et al. 1994; Jennings et al. 1999; Schultz et al. 1999; Drake and Pereira 2002), and some have been developed based on data collected from standardized annual fisheries assessments. Using extant data collected during such surveys is particularly advantageous as the data provides additional information on the health of the ecosystem with little to no additional sampling. However, most standardized sampling efforts only target recreationally important fishes. Small-bodied fishes may represent important trophic, reproductive, feeding guilds or native fishes (Whittier 1999). For example, other lake IBIs have included metrics associated with abundance or richness of Cyprinids (Minns et al. 1994; Drake and Pereira 2002) or small benthic inhabitants (Jennings et al. 1999).

The development of a lentic IBI is timely for South Dakota. Land use changes from grass and pasturelands to row crop agriculture may result in declining water quality in many areas of the state, particularly east of the Missouri River (Figure 2-1). In 2014, the South Department of Environment and Natural Resources (SDDENR) found that 85 lakes of South Dakota’s 572 lakes (14.9%) did not meet water quality standards for their designated beneficial use based on physical and chemical measurements; in 2012, 58 lakes did not meet these standards (SDDENR 2014). Continued land conversion may impact fisheries in the eastern portion of the state due to subsequent nutrient loading and siltation (SDGFP 2014). The objectives of this study were to develop a fish-based IBI based on extant standardized fish community sampling for small lakes (100 -1,000 ha) in eastern South Dakota and determine whether additional sampling for small-bodied fishes improved IBI development for these systems.

METHODS

Study lakes were chosen from two ecoregions (i.e., the Northwestern Great Plains and the Northwestern Glaciated Plains; Omernik 1987) in eastern South Dakota. Lentic fish communities in these ecoregions are similar, so the two ecoregions were considered as one for the purposes of this study (Table 2-1). Only natural lakes with surface areas between 100-1,000 ha were considered in this study to reduce potential lake size effects on fish communities. Many of the lakes included in the study have been stocked at least once in the past five years. Walleye *Sander vitreus* were the most commonly stocked species, but other species such as Muskellunge *Esox masquinongy*, Yellow Perch *Perca flavescens,* and Smallmouth Bass *Micropterus dolomieu* were also stocked occasionally (SDGFP 2016).

Fish community data were used from South Dakota Department of Game, Fish and Parks (SDGFP) annual standardized sampling surveys conducted between June and August 2011 to 2015. Fish were captured by modified fyke nets or experimental gill nets. Fyke nets were used for littoral species such as Largemouth Bass *Micropterus salmoides*, Smallmouth Bass *Micropterus dolomieu*, and Bluegill *Lepomis macrochirus*. Gill nets targeted open water species, mainly Walleye and White Bass *Morone chrysops*. Both gill nets and fyke nets captured other common species such as Black Bullheads *Ameiurus* *melas.* and Common Carp *Cyprinus carpio*. Modified fyke nets had 0.9 x 1.5-m frames, 0.9-m diameter hoops, a single throat, 0.9 x 15.2-m or 0.9 x 18.3 m lead, and were constructed of 19-mm knotted mesh. Experimental gill nets were 1.8 x 45.8 m with six sequentially ordered 7.6-m monofilament panels of 13-, 19-, 25-, 32-, 38-, and 51-mm bar-mesh. Nets were set overnight and retrieved the next day. Species were identified and counted during each sampling event. Relative abundance was indexed using catch per unit effort (CPUE; number of fish per net for both gears).

Potential fish-based metrics were chosen based on previous studies (Jennings et al. 1999; McDonough and Hickman 1999; Drake and Pereira 2002; Beck and Hatch 2009). Fish were classified by origin, feeding guild, reproductive guild, tolerance to stressors, by family, and habitat preference (Table 2-2; Jennings et al 1999; Drake and Pereira 2002; Whittier et al. 2007).

Selection of specific metrics that best described fish community responses to watershed disturbances followed the procedures of Drake and Pereira (2002) with a few differences (Figure 2-2). The first selection test was used to test whether specific metrics were correlated to any land use variable. If metrics were not correlated to land use, they would not have the ability to distinguish between varying land use in watersheds. Second, each metrics distribution was analyzed. If multiple lakes (>75%) or too few of lakes (<25%) had similar metric values these metrics again would not be able to distinguish between lakes. Metrics were then standardized to metric scores on a continuous scale between 0 and 10. Metrics were standardized using linear interpolation of the 5th and 95th percentiles in relation to their response to watershed disturbance. Principle components analysis (PCA) was then used to minimize watershed land use variables to four watershed categories. Generalized linear models (GLMs) were then used to test the ability of metrics to distinguish between these watershed categories. Metrics that displayed the ability to distinguish between these categories were retain. Finally, metrics were tested against one another using Pearson's correlation. If metrics were highly correlated, a single metric between the pair with the largest F-value from previous GLMs was retained.

First, this study used different land use layers when quantifying land use within watersheds. Lake watersheds were designated from Hydrologic Unit Code (HUC) 12 units in ArcGIS 10.2.2. Land cover types were determined and quantified within each watershed using the 2011 National Land Cover Database (Homer et al. 2015). Land cover categories included open water, developed, barren, forested, shrub/herbaceous, hay or pasture, and row crop, and these categories represented the most common land cover types in eastern South Dakota.

The second deviation of this study from the methods outlined in Drake and Pereira (2002) occurred in the first step of metric selection. I correlated candidate metrics with land cover using Kendall's rank correlation (R Core Team 2016) instead of Spearman's correlation. Kendall's rank correlation handled tied ranks of proportions of land use between lakes within the same watersheds better than Spearman's correlation coefficient.

Finally, the *a priori* value used to evaluate candidate metrics against each other was lower in my study ($\rho > 0.7$) compared to Drake and Pereira (2002; $\rho > 0.8$). I chose this value accordance to other IBI development research both within (Krause et al. 2013) and outside South Dakota (Whittier et al. 2007), but final metric selection would not have differed if I had used the same value as Drake and Pereira (2002; personal observation).

Final IBI scores were then calculated based on remaining metrics. Scores were then summed and scaled to result in final lake scores 0-100. This final scaling was done due to match previous South Dakota IBI development (see Krause et al. 2013).

To evaluate whether the inclusion of smaller-bodied (i.e., non-recreationally important) fishes was necessary to improve lentic IBI development for these systems, a random subset of lakes (N = 17) within the study area were sampled using shoreline seining (Table 2-2). Samples were collected between June and August 2016 using a 7.6-m x 2-m bag seine with 6-mm nylon mesh. Shoreline backpack electrofishing was not employed due to high conductivity (> 1500 μS/cm) of most eastern South Dakota lakes. Site selection was based on standardized fyke net locations established by SDGFP. Five 100-m seine hauls were completed at each lake. All fishes collected were identified and enumerated and relative abundance was indexed as CPUE (number of fish per haul). Relevant fish metrics (Table 2-4) were then added to the list of potential IBI metrics, and metric selection procedures outlined by Drake and Pereira (2002) with the modifications as described above. Resulting metrics were compared to the IBI metrics selected based on extant fish community data only.

RESULTS

Standardized extant data collected between 2011 and 2015 resulted in 28 species from 58 lakes (Table 2-2). Average total species richness among all lakes was seven and varied between two and 14. Additional seining data resulted in the addition of six species among 17 lakes.

Principle components analysis (PCA) of land use within each watershed and population density of the counties overlapping each watershed identified four distinct watershed categories. The first axis (PCA1) explained 35% of the variation with inverse loading between agriculture land use (crop and pasture) and natural land use (forested and shrubland). The second axis (PCA2) explained 24% of the variation with inverse loading between population density and barren land (Table 2-3). Each quadrant was used as a watershed disturbance category (Figure 2-3): 1) high agriculture and high population density; 2) low agriculture and high population density; 3) high agriculture and low population density; and 4) low agriculture and low population density.

Completion of all five metric selections steps could only be completed on the extant dataset. By the fourth step, the seine dataset had only one metric remaining (i.e., proportion of Cyprinids). Due to the lack of ability to distinguish between watersheds within the seine dataset, all further analyses described will only be for the extant standardized dataset.

At the first step of metric selection, 21 metrics failed the Kendall's correlation test with over half of the failed metrics (15) representing richness metrics dealing with various life history traits (Table 2-5; Table 2-6). Eight metrics were eliminated in the second step (distribution) test, including the presence of several families (e.g., Esocidae, Lepososteidae, Moronidae, and Percidae). The third step (generalized linear modeling between watershed disturbance category and metric score with lake size as a main effect) eliminated all but three richness metrics and thirteen relative abundance metrics remained

(Table 2-7). The four final metrics retained after completion of the fourth step (Pearson's correlation) were proportion insectivores, proportion intolerant species, proportion Centrarchids, and proportion Ictalurids.

Final IBI scores ranged from < 1 to 98 (Appendix 1), with the mean IBI score for eastern South Dakota lakes of 28 (95% confidence range; 21-34). Score for eastern South Dakota Lakes were generally skewed towards being in poor quality (Figure 2-4). Final IBI scores for lakes between watershed disturbance categories was significantly different (KW ANOVA; d.f. = 3, $p < 0.01$; Figure 2-5). Watersheds with low agricultural influence tended to have higher IBI scores, yet watersheds with both low agricultural and population density influences were significantly higher than other watershed disturbance categories (Figure 2-5). Final IBI scores were generally higher for SDGFP Region IV (Appendix1).

DISCUSSION

The results of this study add to previous work that demonstrates that the development of lentic IBIs may be possible in spite of inherent limitations involved in these systems compared to lotic systems. Many lakes in eastern South Dakota are heavily stocked and naturally species depauperate. However, neither of these issues prevented a meaningful IBI from being developed for this region. Further, the metrics that were selected in the final analysis may be readily explained.

An effective IBI should respond to large-scale spatiotemporal effects of land use change within watersheds (Jennings et al. 1999). Krause et al. (2013) found that agricultural influence was the most influential factor in the development of a stream IBI for eastern South Dakota. The final four metrics appear to respond to land use changes in relation to agricultural development. Agricultural development due to increased siltation and turbidity negatively affects insectivores. Siltation and turbidity do allow for aquatic macrophytes to establish and can bury macroinvertebrate egg banks (Gleason et al. 2003). Furthermore, harmful herbicides can hinder the abundance of benthic insect production (Dewey 1986), further limiting insectivorous fish production (Wiley et al. 1984). Proportion intolerant species responds to all types of degradation, as these species may be limited by the effects of chemical or physical changes (e.g. temperature, chemical toxins, anoxia). Centrarchids and Ictalurids consist of tolerant species (Drake and Pereira 2002; Whittier et al. 2007), therefore increased levels of watershed degradation result in increased proportions of these two families.

Other lentic IBIs have included similar or a greater number of metrics than those identified in this study. An IBI for lakes in southern New England was composed of seven final metrics (Whittier 1999), while the IBI developed for inland lakes in Wisconsin included just four metrics (Jennings et al. 1999). Karr (1981) originally proposed that IBIs include no fewer than 12 metrics. However, the number of metrics selected within an ecoregion may be dependent upon the fish communities themselves; fewer metrics are expected with more depauperate communities than ones with greater richness (Whittier et al. 2007).

Metrics selected in this study align with other lentic IBIs. Three of the four final metrics (e.g., proportion intolerant species, proportion insectivore, and proportion Centrarchids) have appeared as final metrics in other Midwestern lake IBIs. Intolerant species metrics were used as final metrics for lake IBIs in littoral areas of the Great Lakes

(Minns et al. 1994), Wisconsin inland lakes (Jennings et al. 1999), and Minnesota inland lakes (Drake and Pereira 2002). While proportion of insectivores was the only feeding guild metric retained for eastern South Dakota lakes, the IBI developed for Minnesota's inland lakes revealed that both insectivore richness and composition was important to Minnesota's lake IBI (Drake and Pereira 2002). The final similarity of the this IBI to other lentic IBIs was the use of proportion of Centrarchids. Both littoral areas of the Great Lakes and inland Wisconsin lakes used Centrarchid metrics to develop their respective lake IBIs (Minns et al. 1994; Jennings et al. 1999). Finally, proportion Ictalurids was the only final IBI metric for eastern South Dakota lakes that was not represented by any previous lake IBIs. The similarities and biological relevance of the final IBI metrics for eastern South Dakota lakes emphasizes the legitimacy of the development and utilization of this IBI.

The use of extant standardized sampling data without the inclusion of additional sampling has several advantages. First, data collected in previous years can be analyzed to track how land use may have affected fish communities over time and inform management responses to the continued changes in watershed land use. Second, our results suggest that no additional sampling efforts are required in order to include non-recreationally important fishes in IBI assessments. Other lentic IBI research has supported the idea that small-bodied fishes should be included within metrics (Minns et al. 1994; Jennings et al. 1999; Whittier 1999; Drake and Pereira 2002). However, my results should be considered somewhat cautiously. Sampling of small-bodied fishes was done on a limited subset of lakes, therefore the sample size was small compared to other lake IBI datasets (Jennings et al. 1999; Drake and Pereira 2002). Additionally, the small-bodied fishes collected in eastern South Dakota lakes were either ubiquitous or scarce, making it more difficult within the metric selection process to distinguish between differing levels of watershed degradation. Further study is needed to evaluate whether small-bodied fishes would improve IBI development within these lakes and beyond.

As land use continues to change in South Dakota, it is important to have a rapid assessment tool to evaluate how fish communities respond in kind. Trends in historic and future IBIs calculated on the same lake can be combined with historic and future data on physical and chemical measures to provide a more complete picture of water quality in these systems. Collectively, this information can be used to identify where management interventions (e.g., in-lake habitat restoration, watershed habitat modifications) are needed, prioritize efforts, and monitor fish community changes in response.

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Table 2-1. Mean physiochemical characteristics for eastern South Dakota lakes used in the development of a fish-based index of biotic integrity (95% confidence intervals are given in parentheses).

Variable	All study lakes
N	58
Area (ha)	358 (305-412)
Alkalinity (mg/L $CaCO_3$)	224 (211-237)
Secchi Depth (m)	1.48 (1.26-1.69)
Total Nitrogen (mg/L N)	1.73 (1.38-2.08)
Total P (mg/L P)	0.232 (0.16-0.31)
TSS (mg/L)	14.26 (9.59-18.92)

Table 2-2. Fish species characteristics used to calculate metrics. Species are listed alphabetically by family and then by common name. The following abbreviations are used: I = intolerant, M = moderately tolerant, T = tolerant, In = insectivore, Om = Omnivore, Tc = top carnivore, N = native, NN = non-native, A11 = pelagophil, A12 = lithopelagophil, A13 = lithophil, A14 = phytolithophil, A15 = phytophil, B = nest guarder, B27 = speleophil. Table adapted from Jennings et al. (1999), Drake and Pereira (2002), and Whittier et al. (2007).[g] Species caught in gill nets. [t] Species caught in fyke nets. [s] Species caught in seines

Species	Family	Tolerance	Feeding	Habitat	Origin	Reproductive
Bigmouth Buffalo *Ictiobus cyprinellus* [g,t]	Catstomidae	M	In	WC	N	A12
River Carpsucker *Carpiodes carpio* [g,t]	Catstomidae	T	Om	B	N	A12
Shorthead Redhorse *Moxostoma macrolepidotum* [g,s]	Catstomidae	M	Om	B	N	A13
White Sucker *Catostomus commersonii* [g,t,s]	Catstomidae	T	Om	B	N	A12
Black Crappie *Pomoxis nigromaculatus* [g,t,s]	Centrarchidae	M	Tc	WC	N	B
Bluegill *Lepomis macrochirus* [g,t,s]	Centrarchidae	M	In	WC	N	B
Green Sunfish *Lepomis cyanellus* [t,s]	Centrarchidae	M	In	WC	N	B
Hybrid Sunfish *Lepomis spp.*[g,t]	Centrarchidae	M	In	WC	N	B
Largemouth Bass *Micropterus salmoides* [t,s]	Centrarchidae	M	Tc	WC	NN	B
Orangespotted Sunfish *Lepomis humilis* [g,t,s]	Centrarchidae	T	In	WC	N	B
Pumpkinseed *Lepomis gibbosus* [g,t]	Centrarchidae	M	In	WC	N	B
Rock Bass *Ambloplites rupestris* [g,t]	Centrarchidae	I	Tc	WC	N	B
Smallmouth Bass *Micropterus dolomieu* [g,t,s]	Centrarchidae	I	Tc	WC	NN	B
White Crappie *Pomoxis annularis* [g,t]	Centrarchidae	M	Tc	WC	N	B
Emerald Shiner *Notropis atherenoides* [g]	Cyprinidae	M	In	WC	N	A11
Common Carp *Cyprinus carpio* [g,t,s]	Cyprinidae	T	Om	WC	NN	A14
Fathead Minnow *Pimephales promelas* [s]	Cyprinidae	T	Om	WC	N	B
Golden Shiner *Notemigonus chryoleucas* [g,t,s]	Cyprinidae	M	Om	WC	N	A15
Red Shiner *Cyprinella lutrensis* [s]	Cyprinidae	T	Om	WC	N	B
Sand Shiner *Notropis stramineus* [s]	Cyprinidae	T	Om	WC	N	A14
Spottail Shiner *Notropis hudsonius* [g,s]	Cyprinidae	M	In	WC	N	A12
Northern Pike *Esox lucius* [g,t,s]	Esocidae	M	Tc	WC	N	A15

Table 2-2 Continued. Fish species characteristics used to calculate metrics. Species are listed alphabetically by family and then by common name. The following abbreviations are used: I = intolerant, M = moderately tolerant, T = tolerant, In = insectivore, Om = Omnivore, Tc = top carnivore, N = native, NN = non-native, A11 = pelagophil, A12 = lithopelagophil, A13 = lithophil, A14 = phytolithophil, A15 = phytophil, B = nest guarder, B27 = speleophil. Table adapted from Jennings et al. (1999), Drake and Pereira (2002), and Whittier et al. (2007).[g] Species caught in gill nets. [t] Species caught in fyke nets. [s] Species caught in seines.

Species	Family	Tolerance	Feeding	Habitat	Origin	Reproductive
Black Bullhead *Ameiurus melas* [g,t,s]	Ictaluridae	T	Om	B	N	B27
Channel Catfish *Ictalurus punctatus* [g,t]	Ictaluridae	M	Tc	B	N	B27
Tadpole Madtom *Notorus gyrinus* [t]	Ictaluridae	M	In	B	N	B27
Yellow Bullhead *Ameiurus natalis* [g,t]	Ictaluridae	M	Om	B	N	B27
Shortnose Gar *Lepisosteus platostomus* [g,t]	Lepisosidae	T	Tc	WC	N	A15
White Bass *Morone chryops* [g,t,s]	Moronidae	T	Tc	WC	N	A14
Johnny Darter *Etheostoma nigrum* [s]	Percidae	T	In	B	N	B
Iowa Darter *Etheostoma exile* [s]	Percidae	M	In	B	N	B
Logperch *Percina caproides* [s]	Percidae	M	In	B	N	A13
Walleye *Sander vitreus* [g,t,s]	Percidae	M	Tc	WC	N	A12
Yellow Perch *Perca flavescens* [g,t,s]	Percidae	M	In	WC	N	A14
Freshwater Drum *Aplodintous grunniens* [g,t]	Scianidae	T	Om	B	N	A11

Table 2-3. Results of principle components analysis on watershed land use and population density variables for lakes used in index of biotic integrity development

Principle Component	Eigenvalue	Proportion of variance explained	Cumulative variance explained	Coefficient eigenvectors					
				% Cropland	% Pasture	% Shrub	% Forest	Population density	% Barren
PCA 1	2.77	0.35	0.35	-0.64	-1.91	0.58	0.19	-0.52	0.00
PCA 2	1.96	0.24	0.59	-0.10	-0.18	-0.14	0.02	0.58	-1.75
PCA 3	1.33	0.17	0.76	0.33	0.20	0.49	0.78	-0.05	-0.16

Table 2-4. Metric classes and metric abbreviations for metrics used in the development of a fish-based IBI for lakes in eastern South Dakota.

Metric Class	Metric	Metric Abbreviation
Family	Catostomidae	RA_CAT
Family	Centrarchidae	RA_CENT
Family	Native Centrarchidae	RA_NATCENT
Family	Cyprinidae	RA_CYP
Family	Cyprinidae No Common Carp	RA_NOCOC
Family	Esocidae	RA_ESO
Family	Ictaluridae	RA_ICT
Family	Leposoteidae	RA_LEP
Family	Moronidae	RA_MOR
Family	Percidae	RA_PER
Family	Percidae (Darters)	RA_DART
Family	Scianidae	RA_SCI
Richness	Catostomidae	RC_CAT
Richness	Centrarchidae	RC_CENT
Richness	Native Centrarchidae	RC_NATCENT
Richness	Cyprinidae	RC_CYP
Richness	Cyprinidae No Common Carp	RC_NOCOC
Richness	Esocidae	RC_ESO
Richness	Ictaluridae	RC_ICT
Richness	Leposoteidae	RC_LEP
Richness	Moronidae	RC_MOR
Richness	Percidae	RC_PER
Richness	Percidae	RC_DART
Richness	Scianidae	RC_SCI
Richness	Total	TOT_RC
Richness	Shannon's H	SHAN_H
Richness	Eveness	SHAN_E
Feeding	Insectivore Abundance	PCT_IN
Feeding	Omnivore Abundance	PCT_OM
Feeding	Top Carnivore Abundance	PCT_TC
Feeding	Insectivore Richness	RC_IN
Feeding	Omnivore Richness	RC_OM

Table 2-4 Continued. Metric classes and metric abbreviations for metrics used in the development of a fish-based IBI for lakes in eastern South Dakota.

Metric Class	Metric	Metric Abbreviation
Feeding	Top Carnivore Richness	RC_TC
Habitat	Benthic Abundance	PCT_BN
Habitat	Water Column Abundance	PCT_WC
Habitat	Benthic Richness	RC_BN
Habitat	Water Column Richness	RC_WC
Tolerance	Intolerant Abundance	PCT_INT
Tolerance	Moderate Tolerant Abundance	PCT_MOD
Tolerance	Tolerant Abundance	PCT_TOL
Tolerance	Intolerant Richness	RC_INT
Tolerance	Moderate Richness	RC_MOD
Tolerance	Tolerant Richness	RC_TOL
Reproductive	Pelagophil Abundance	PCT_A11
Reproductive	Lithopelagophil Abundance	PCT_A12
Reproductive	Lithophil Abundance	PCT_A13
Reproductive	Phytolithophil Abundance	PCT_A14
Reproductive	Phytophil Abundance	PCT_A15
Reproductive	Nest Guarder Abundance	PCT_B
Reproductive	Speleophil Abundance	PCT_B27
Reproductive	Pelagophil Richness	RC_A11
Reproductive	Lithopelagophil Richness	RC_A12
Reproductive	Lithophil Richness	RC_A13
Reproductive	Phytolithophil Richness	RC_A14
Reproductive	Phytophil Richness	RC_A15
Reproductive	Nest Guarder Richness	RC_B
Reproductive	Speleophil Richness	RC_B27
Origin	Native Abundance	PCT_NON
Origin	Non-Native Abundance	PCT_NAT
Origin	Native Richness	RC_NON
Origin	Non-Native Richness	RC_NAT

Table 2-5. The number of remaining metrics after each metric evaluation test by metric class for original standardized netting dataset. Richness metrics were only related to richness of families and both Shannon's H and Evenness.

Metric Class	Start	Metric Evaluation Tests			
		Kendall's	Distribution	GLMs	Pearson's
Family	11	8	4	3	2
Richness	14	8	4	0	0
Feeding	6	4	4	4	1
Habitat	4	4	4	2	0
Tolerance	6	4	4	4	1
Reproductive	14	8	8	3	0
Origin	4	1	1	0	0
Total	59	37	29	16	4

Table 2-6. Kendall's rank correlation coefficients between proposed fish assemblage metrics and eastern South Dakota watershed land use values; P ≤ 0.05*, P ≤ 0.005**. Metric abbreviations found in Table 2-4. Land use abbreviations are as follows: OPWT = open water, DEV = developed, BAR = barren, FOR = forested, SHB = shrub land, PAS = pasture, CROP = crop land.

	Land use						
Metric	OPWT	DEV	BAR	FOR	SHB	PAS	CROP
PCT_A11	0.02	-0.05	0.21	-0.06	-0.04	-0.02	0.00
PCT_A12	0.11	0.27**	0.08	0.04	0.04	0.06	0.12
PCT_A13	-0.16	-0.04	0.06	-0.17	-0.12	0.15	0.09
PCT_A14	0.22*	0.10	-0.02	0.05	0.23*	-0.16	-0.04
PCT_A15	0.18*	-0.13	0.06	0.11	0.13	-0.27**	-0.23*
PCT_B	0.26**	-0.13	0.11	0.13	0.14	-0.30**	0.27**
PCT_B27	-0.34**	-0.05	-0.10	0.12	-0.29**	0.20*	0.09
PCT_BN	-0.34**	0.00	-0.09	-0.16	-0.34**	0.25*	0.12
PCT_IN	0.34**	-0.03	0.08	0.18*	0.49**	-0.38**	-0.18*
PCT_INT	0.24*	-0.20	0.09	0.05	0.15	-0.34**	-0.20*
PCT_MOD	0.34**	0.01	0.06	0.16	0.33**	-0.25*	-0.12
PCT_NAT	0.14	-0.03	-0.16	0.12	0.16	-0.01	-0.11
PCT_NON	-0.14	0.03	0.16	-0.12	-0.16	0.01	0.11
PCT_OM	-0.34**	0.01	-0.10	-0.16	-0.32**	0.24*	0.12
PCT_TC	0.22*	0.06	0.15	0.02	0.10	-0.11	-0.04
PCT_TOL	-0.32**	0.01	-0.08	0.15	-0.33**	0.25**	0.13
PCT_WC	0.34**	0.00	0.09	0.16	0.34**	-0.25*	-0.12
RA_CAT	-0.17	-0.01	0.00	-0.09	-0.01	-0.01	-0.03
RA_CENT	0.24*	-0.17	0.09	0.11	0.10	-0.30**	-0.31**
RA_CYP	-0.23*	0.19*	0.07	-0.18	-0.25*	0.27**	0.22*
RA_ESO	0.20*	-0.25*	-0.01	0.10	0.20*	-0.38**	-0.40**
RA_ICT	-0.25*	-0.03	-0.12	-0.06	-0.21*	0.15	0.02
RA_LEP	-0.01	0.18	0.26*	-0.05	-0.04	0.22*	0.21
RA_MOR	0.05	-0.12	0.04	-0.25*	-0.02	-0.21*	-0.18
RA_NATCENT	0.26**	-0.13	0.11	0.12	0.14	-0.31**	-0.28**
RA_NOCOC	-0.05	-0.08	-0.06	-0.18	0.10	-0.18	-0.14
RA_PER	0.24*	-0.02	-0.06	0.13	0.24*	-0.25*	-0.21*
RA_SCI	-0.03	0.02	0.21	-0.18	-0.13	0.10	0.12
RC_A11	0.02	-0.05	0.21	-0.06	-0.03	-0.02	0.00
RC_A12	-0.13	-0.03	0.05	-0.15	0.01	-0.02	-0.09
RC_A13	-0.16	-0.04	0.06	-0.17	-0.11	0.15	0.09
RC_A14	-0.09	0.07	0.12	-0.23*	-0.14	0.04	0.10
RC_A15	-0.02	-0.06	0.10	-0.14	-0.01	-0.12	-0.14
RC_B	0.23*	-0.13	0.04	0.12	0.14	-0.25*	-0.15
RC_B27	-0.14	0.13	-0.17	-0.18	-0.22*	0.16	0.08

Table 2-6. Continued. Kendall's rank correlation coefficients between proposed fish assemblage metrics and eastern South Dakota watershed land use values; P ≤ 0.05*, P ≤ 0.005**. Metric abbreviations found in Table 2-4. Land use abbreviations are as follows: OPWT = open water, DEV = developed, BAR = barren, FOR = forested, SHB = shrub land, PAS = pasture, CROP = crop land.

	Land Use						
Metrics	OPWT	DEV	BAR	FOR	SHB	PAS	CROP
RC_BN	-0.19	0.08	0.00	-0.19	-0.20*	0.17	0.10
RC_CAT	-0.18	-0.03	0.03	-0.01	0.03	0.04	-0.01
RC_CENT	0.23*	-0.13	0.04	0.12	0.14	-0.25*	-0.15
RC_CYP	-0.22*	0.17	0.15	-0.23*	-0.21	0.20	0.18
RC_ESO	0.07	-0.17	0.07	-0.09	-0.03	-0.21	-0.26*
RC_ICT	-0.14	0.13	-0.17	-0.18	-0.22*	0.16	0.08
RC_IN	0.04	-0.07	-0.05	0.08	0.14	-0.08	0.05
RC_INT	0.27*	-0.23*	0.05	0.03	0.16	0.37**	-0.22*
RC_LEP	-0.01	0.18	0.26*	-0.05	-0.04	0.21*	0.21
RC_MOD	0.05	-0.07	-0.04	-0.09	0.01	-0.17	-0.13
RC_MOR	0.05	-0.13	0.05	-0.23*	0.00	-0.23*	-0.20
RC_NAT	0.06	-0.09	0.03	-0.11	0.04	-0.19*	-0.15
RC_NATCENT	0.22*	-0.11	0.05	0.11	0.13	-0.23*	0.13
RC_NOCOC	-0.05	-0.08	-0.06	-0.17	0.10	-0.18	-0.15
RC_NON	-0.01	-0.01	0.13	-0.06	-0.11	-0.02	0.04
RC_OM	-0.17	0.11	0.00	-0.14	-0.12	0.14	0.11
RC_PER	0.15	-0.01	-0.06	-0.03	0.11	-0.18	-0.09
RC_SCI	-0.03	0.02	0.21	-0.18	-0.13	0.10	0.12
RC_TC	0.15	-0.11	0.12	-0.09	0.01	-0.25**	-0.25*
RC_TOL	-0.12	0.03	0.11	-0.13	-0.10	0.07	0.05
RC_WC	0.17	-0.15	0.05	0.00	0.13	-0.28**	-0.16
TOT_RC	0.05	-0.08	0.06	-0.10	0.02	-0.16	-0.11
SHAN_H	0.02	-0.15	-0.15	0.02	-0.06	-0.07	-0.15
SHAN_E	-0.05	-0.18	-0.11	-0.10	-0.03	-0.10	-0.22*

Table 2-7. Mean metric scores by watershed disturbance category estimated by general linear models. Means with the same lowercase letter are not significantly different according to Tukey's multiple-comparisons test ($P < 0.05$). Watershed disturbance category abbreviations are as follows: HAG/HPOP = high agriculture and high population density; LAG/HPOP = low agriculture and high population density; HAG/LPOP = high agriculture and low population density; LAG/LPOP = low agriculture and low population density. See Table 2-4 for metric abbreviations.

		Watershed disturbance category			
Metric	P-value	HAG/HPOP	LAG/HPOP	HAG/LPOP	LAG/LPOP
PCT_B	< 0.01	0.565 [b]	3.082 [ab]	0.193 [b]	5.3781 [a]
PCT_B27	< 0.01	3.088 [b]	5.870 [ab]	3.599 [b]	8.533 [a]
PCT_BN	< 0.01	2.399 [b]	5.427 [ab]	2.676 [b]	8.180 [a]
PCT_IN	< 0.01	1.618 [b]	4.258 [ab]	1.626 [b]	6.196 [a]
PCT_INT	< 0.01	0.292 [b]	0.6172 [b]	0.000 [b]	4.356 [a]
PCT_MOD	< 0.01	2.552 [b]	5.262 [ab]	2.705 [b]	7.341 [a]
PCT_OM	< 0.01	2.409 [b]	5.401 [ab]	2.781 [b]	8.186 [a]
PCT_TC	0.022	1.669 [b]	2.634 [ab]	2.234 [ab]	4.402 [a]
PCT_TOL	< 0.01	2.593 [b]	5.349 [ab]	2.705 [b]	8.119 [a]
PCT_WC	< 0.01	2.399 [b]	5.427 [ab]	2.676 [b]	8.180 [a]
RA_CENT	< 0.01	0.504 [b]	2.752 [ab]	0.172 [b]	5.070 [a]
RA_ICT	< 0.01	8.683 [b]	9.214 [ab]	8.775 [b]	9.721 [a]
RA_NATCENT	< 0.01	0.507 [b]	3.129 [ab]	0.205 [b]	5.316 [a]
RC_B27	< 0.01	3.384 [b]	4.990 [ab]	4.633 [ab]	6.512 [a]
RC_INT	0.02	1.000 [b]	3.462 [ab]	0.000 [b]	5.417 [a]
RC_TC	0.03	2.718 [b]	4.354 [ab]	2.539 [b]	6.036 [a]

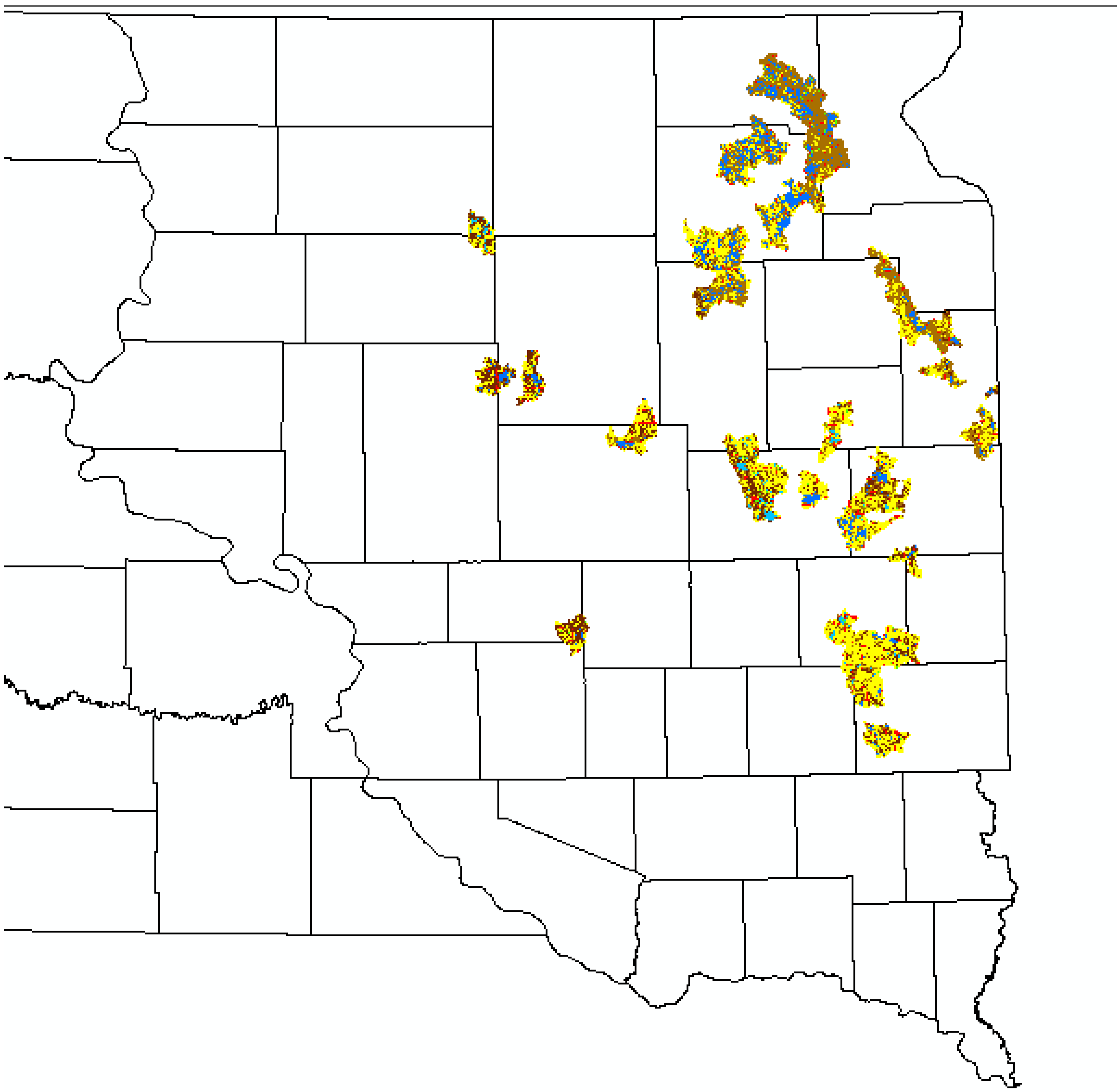

Figure 2-1. Distribution map of watersheds used in developing a fish-based lake IBI for eastern South Dakota. Watershed size was hydrologic unit code (HUC) 12. Some watersheds included multiple lakes used within the assessment.

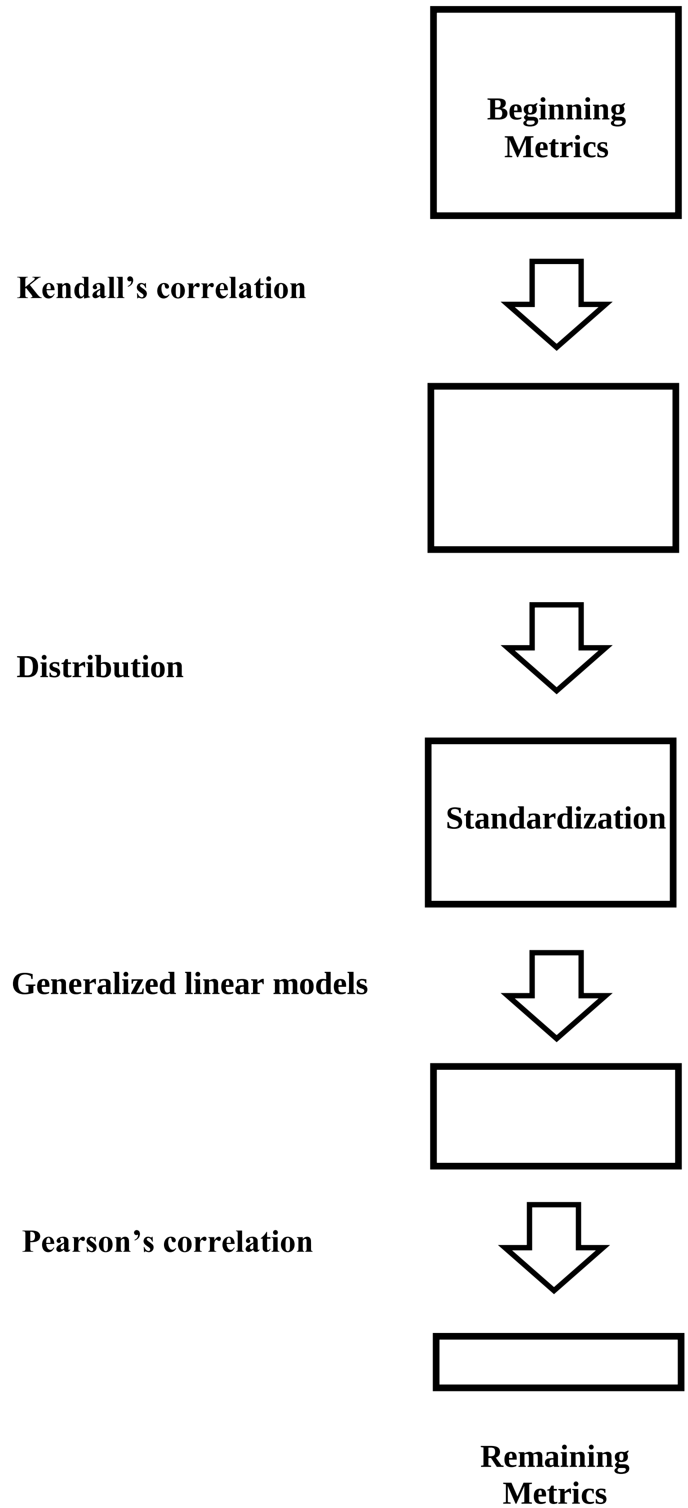

Figure 2-2. Visual model of metric selection process proposed by Drake and Pereira (2002). Proposed metrics begin at the top of the model. Number of metrics remaining is visualized by rectangles, while each statistical test and metric omission is represented by arrows between squares.

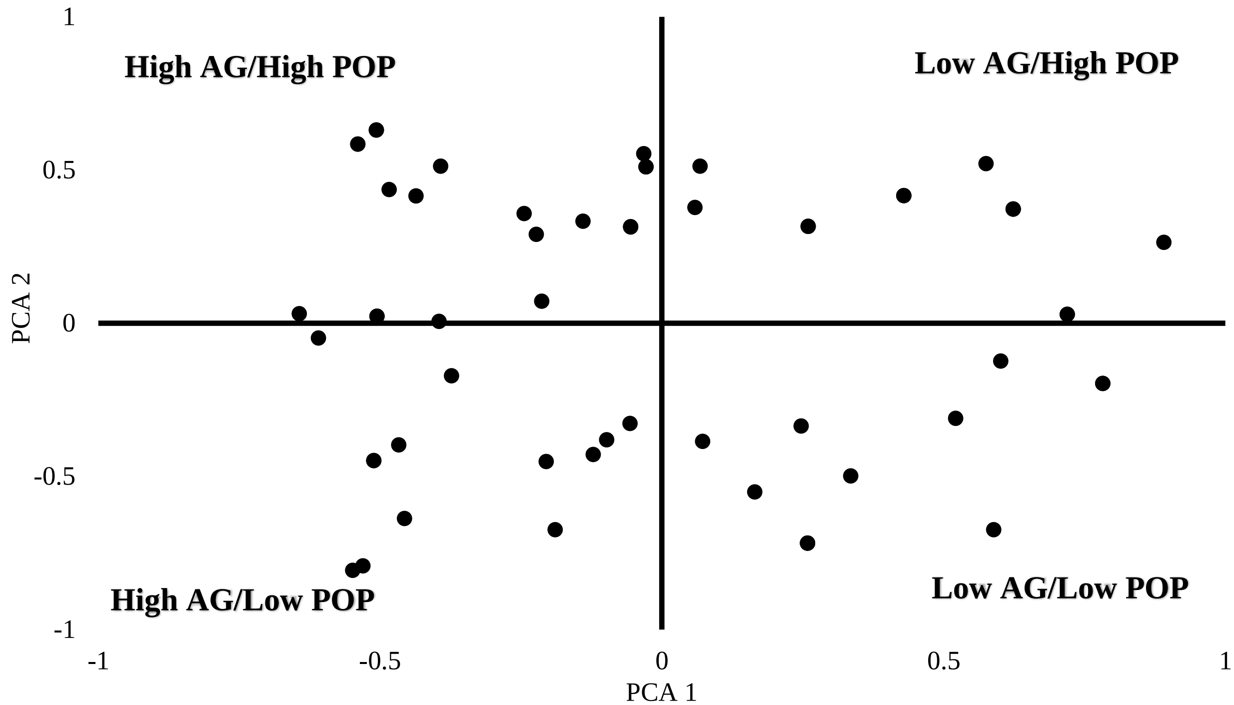

Figure 2-3. Plot of principle component 1 (PCA 1) versus principle component 2 (PCA 2). Each quadrant was used as a watershed disturbance category: high agriculture and high population density, low agriculture and high population density, high agriculture and low population density, and low agriculture and low population density.

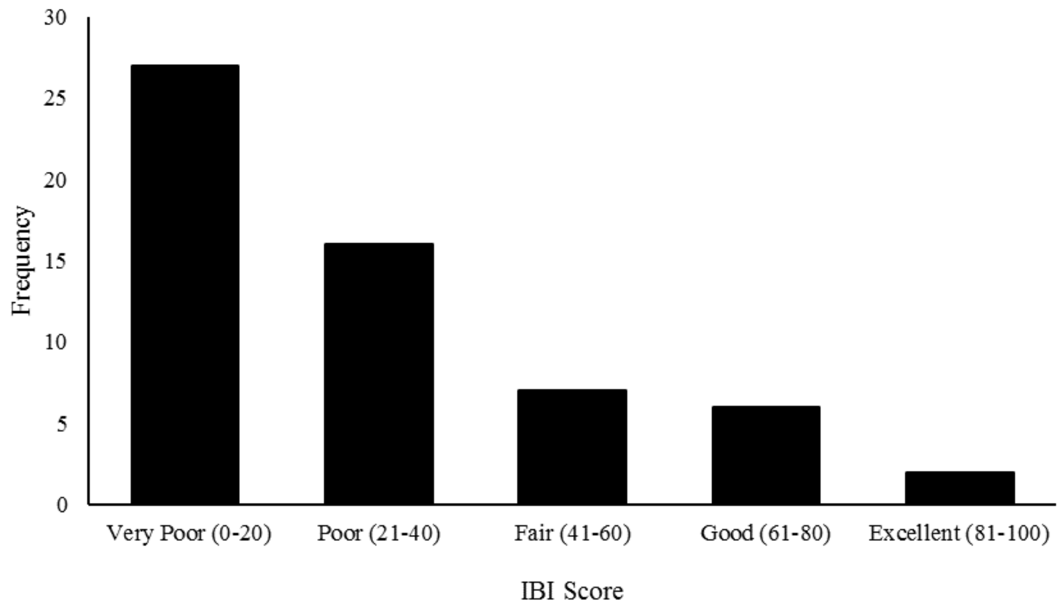

Figure 2-4. Distribution of final IBI scores for all lakes in eastern South Dakota.

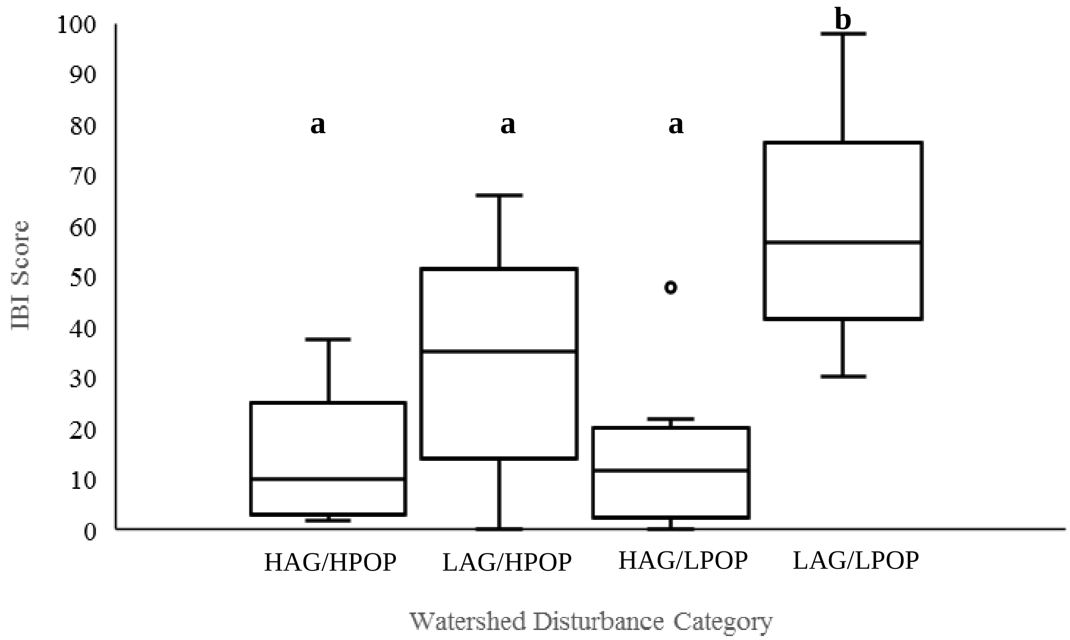

Figure 2-5. Boxplot distribution of final IBI scores by watershed disturbance category. Final IBI scores only differed in the low agriculture/low population density watersheds according to Tukey's multiple comparison test. Watershed disturbance category abbreviations are as follows: HAG/HPOP = high agriculture and high population density (N = 20); LAG/HPOP = low agriculture and high population density (N = 13); HAG/LPOP = high agriculture and low population density (N = 13); LAG/LPOP = low agriculture and low population density (N = 12).

Appendix 2-1. List of final index of biotic integrity scores for lakes in eastern South Dakota. Lakes are broken down by county and South Dakota Department of Game, Fish, and Parks (SDGFP) region. Region 3 is located in southeast South Dakota; Region 4 is located in northeast South Dakota.

Lake Name	County	Score	SDGFP Region	Lake Name	County	Score	SDGFP Region
Twin	Minnehaha	3	3	Lardy	Day	51	4
Clear	Minnehaha	11	3	Lily	Day	30	4
Beaver	Minnehaha	9	3	Opitz	Day	48	4
Diamond	Minnehaha	8	3	Mid-Lynn	Day	48	4
North Island	Minnehaha	2	3	South Rush	Day	35	4
Enemy Swim	Day	95	4	Fish	Deuel	22	4
Byron	Beadle	< 1	3	Clear	Deuel	3	4
Punished Woman	Codington	12	4	Alice	Deuel	3	4
Round	Deuel	22	4	Cochrane	Deuel	66	4
Goldsmith	Brookings	13	3	Bullhead	Deuel	38	4
Oak	Brookings	4	3	North Scatterwood	Edmunds	12	4
Sinai	Brookings	32	3	Norden	Hamlin	31	4
Campbell	Brookings	6	3	Long	Lake	3	3
East 81	Brookings	3	3	Silver	Hutchinson	2	3
East Oakwood	Brookings	26	3	Spirit	Kingsbury	21	3
West Oakwood	Brookings	13	3	Mud	Kingsbury	7	3
Brush	Brookings	2	3	Herman	Lake	19	3
West 81	Kingsbury	16	3	Brant	Lake	16	3
Reid	Clark	48	4	South Red Iron	Marshall	32	4
Cottonwood	Clark	35	4	Cottonwood	Marshall	36	4
Pickerel	Day	62	4	Nine Mile	Marshall	1	4
Minnewasta	Day	24	4	Clear	Marshall	51	4
Blue Dog	Day	71	4	Roy	Marshall	66	4
Antelope	Day	70	4	South Buffalo	Marshall	45	4
Horseshoe	Day	98	4	North Buffalo	Marshall	30	4
Lynn	Day	52	4	North Drywood	Roberts	< 1	4
Campbell Slough	Day	78	4	Twin	Sanborn	18	3
Hazeldon	Day	40	4	Twin	Spink	5	4
Lardy	Day	51	4	Cottonwood	Spink	0	4
Lily	Day	30	4	Mud	Spink	19	4

CHAPTER 3. EVALUATION OF SMALL LAKE INDEX OF BIOTIC INTEGRITY ON LARGER LAKES

INTRODUCTION

The goal of the U.S. Clean Water Act is to protect and restore surface water quality, including physical, chemical, and biological integrity. State and federal regulatory agencies often use physical and chemical properties of surface waters to determine specific thresholds of beneficial uses, but these measures may not fully capture the total impact of human disturbance within watersheds (Karr 1981, 1994; Beck and Hatch 2009). Consequently, greater emphasis has been placed on biological integrity as an indicator of water quality in the last three decades.

One of the most well-known techniques for biological assessment of aquatic ecosystems is the index of biotic integrity (IBI), which uses several metrics to summarize water quality impairments to aquatic communities into a single number (Karr 1981; Barbour et al. 1995). Most fish-based IBIs have been developed for lotic systems across the United States (e.g., Lyons 1992; Mundahl and Simon 1999; Bramblett et al. 2005; Krause et al. 2013). Development of lentic fish-based IBIs have lagged behind that for lotic systems for a number of reasons, including, but not limited to, identifying the appropriate categories of lakes for which to develop the IBIs and a lack of consistency in relationships between watershed disturbance and fish assemblages (Whittier 1999). Variability between lentic classification, even within the same regions, makes it difficult to meet the assumption that the development of an IBI is based on homogenous ecosystems.

Both lotic and lentic IBIs should be developed across similar types or categories that are expected to respond to disturbances in similar ways or have similar fish communities. Lotic IBIs are most often developed by ecoregion for this reason, but no consistently used category exists for lentic IBIs. One category that may be used for lentic IBI development is surface area. Lakes with similar surface area should respond to anthropogenic disturbance in a similar manner (Matuszek and Beggs 1988) and may support similar fish communities. Several studies have shown that larger lakes tend to support a greater number of species (Tonn and Magnuson 1982; Rahel 1986; Matuszek and Beggs 1988; Magnuson et al. 1994; Pierce et al. 1994). For example, lake size explained the greatest variation in species richness between Ontario lakes compared to other physical and chemical properties of those lakes (Matuszek and Beggs 1988). Metrics related to species richness have been shown to be important in distinguishing between varying levels of degradation in other lentic IBIs (Whitter 1999; Drake and Pereira 2002).

To date, only one study has examined the influence of lake size on IBI development (Drake and Valley 2005). An IBI developed for small lakes (48-180 ha; Drake and Pereira 2002) was useful in assessing biotic integrity of large lakes (57-203 ha) in the same geographic region. However, further study is needed to determine whether lentic IBIs in other geographic regions respond in a similar manner. Recent research has developed a lentic IBI for small eastern South Dakota natural lakes (surface area = 100 – 1,000 ha; see Chapter 2 of this thesis), but this IBI has yet to be evaluated for large lakes (surface area > 1,000 ha). Thus, the objective of this study is to determine whether lake size influences IBI development and use in eastern South Dakota natural lakes.

METHODS

Study lakes with surface areas > 1,000 ha were selected from two ecoregions (i.e., the Northwestern Great Plains and the Northwestern Glaciated Plains; Omernik 1987) within South Dakota and data from the two regions were consolidated for this study. This large lake dataset was added to a previous dataset used to develop a small lake (surface area = 100 – 1,000 ha) IBI (see Chapter 2, this thesis; Table 3-1). The data from the small lakes were then considered the “original” dataset, and the data from the large lakes were considered the “test” dataset.

Fish community data from both lake types were collected in the same manner by the South Dakota Department of Game, Fish, and Parks (SDGFP) during annual standardized sampling surveys conducted between June and August 2011 to 2015. Sampling gears included both modified fyke nets and experimental gill nets. Modified fyke nets nets had 0.9 x 1.5-m frames, 0.9-m diameter hoops, a single throat, 0.9 x 15.2-m or 0.9 x 18.3 m lead, and were constructed of 19-mm knotted mesh. Experimental gill nets were 1.8 x 45.8 m with six sequentially ordered 7.6-m panels of 13-, 19-, 25-, 32-, 38-, and 51-mm monofilament bar-mesh. Both nets types were set overnight and retrieved the next day. Species were identified and counted. Fish metrics were assigned and quantified in the same manner as the small lakes dataset (see Chapter 2, this thesis). Lakes from both datasets all have a history of stocking. Walleye *Sander vitreus* are the most commonly stocked species among both lake types, but other species such as

Muskellunge *Esox masquinongy*, Yellow Perch *Perca flavescens*, and Smallmouth Bass *Micropterus dolomieu* are also occasionally stocked.

To test the performance of the small lake IBI on large lakes in this study, I followed the procedures of Drake and Valley (2005) with differences noted in two steps. First, watershed land use was quantified using the 2011 National Land Cover Database (Homer et al. 2015) rather than Minnesota Land Management Information Center reports used by Drake and Valley (2005). Land cover categories included open water, developed, barren, forested, shrub/herbaceous, hay or pasture, and row crop, and represented the most common land cover types in eastern South Dakota.

Second, variables used to test IBI performance on large lakes differed from the Minnesota study. Drake and Valley (2005) tested the relationship between their large lake IBI against trophic state index (TSI) and floristic quality index (FQI). My study only tested the relationship between the large lake IBI and watershed disturbance (as described in the first axis of the principle components analysis outlined below) and did not include physical or chemical measures of water quality. When comparing the metric relationships between small and large lakes. This is because many of these measures, such as TSI, are often not accurate measures for water quality conditions in eastern South Dakota (Kuehl and Troelstrup 2013).

RESULTS

Extant data for 58 small lakes included in this study included 28 total species, but the 11 large lakes included only 18 species (Table 3-2). Average total species was seven for the small lake dataset and 10 for the large lake dataset. River Carpsucker *Carpiodes carpio*, Shorthead Redhorse *Moxostoma macrolepidotum*, Pumpkinseed *Lepomis* *gibbosus*, Emerald Shiner *Notropis atherenoides*, Golden Shiner *Notemigonus crysoleuscas*, Tadpole Madtom *Notorus gyrinus*, and Freshwater Drum *Aplodinotus grunniens* were captured only in the small lakes. Large lakes had no unique fishes compared to small lakes. Water quality parameters did not appear to differ between small and large lakes (Table 3-1).

Principle component analysis (PCA) yielded similar watershed ordination results between the original and test lakes. The first axis (PCA1) explained 33% of the variation and had reverse loading between agricultural land uses (i.e., row crop and pasture) and natural land use (i.e., shrubland and forest; Figure 3-1). The second axis (PCA2) explained 23% of the variation and had reverse loading between county population density and barren land (Figure 3-1; Table 3-3). Quadrants were then used as watershed disturbance categories (i.e., high agriculture and high population density, low agriculture and high population density, high agriculture and low population density, and low agriculture and low population density).

The relationship between IBI scores and watershed disturbance was similar between small and large lakes (Figure 3-2). Not all lakes had water quality data available and some lakes were sampled multiple times, therefore averages of the sampled lakes were used for analysis. Some water quality measures differed between lakes with varying watershed disturbances (Table 3-4). Mean alkalinity was higher in watersheds with higher agricultural influence, and total nitrogen and total phosphorous tended to be lower in watersheds with low agricultural influence and low population densities (Table 3-4).

Individual standardized IBI metric scores were similar between small lakes and large lakes (Table 3-5). Mean IBI score across all lakes was 38 (range = 20 – 98; 95%

confidence interval = 33.5 – 42.5; Appendix 3-1). Lake IBI scores were significantly related to watershed disturbance within watersheds (Linear regression: R^2: 0.25; d.f. = 67,1; F = 21.87; P < 0.001; Figure 3-2). IBI scores for all lakes in eastern South Dakota were skewed towards poor quality (Figure 3-3). Final IBI scores were generally higher watersheds with low agricultural influence and were significantly higher in low agriculture and low population density influence (Table 3-6; Figure 3-4). Pearson's correlation confirmed that all four metrics were correlated to watershed disturbance when all size categories of lakes were combined (Table 3-7).

DISCUSSION

Results from this study indicate that the small lake IBI developed for eastern South Dakota lakes can be applied to larger lakes in the same region. Total IBI scores and all four metrics responded to variation in land use disturbance in a similar manner between the original and test datasets. Land use has been found to be the primary driver of lotic IBI scores in eastern South Dakota as well (Krause et al. 2013). Eastern South Dakota lotic IBI scores were lower in the agriculturally dominated James River valley, while IBI scores for watersheds in the pasture dominated Prairie Coteau were higher (Krause et al. 2013).

Other studies have used other indices of human induced stress, such as trophic state index and floristic quality index to explain patterns in IBIs observed (e.g., Drake and Valley 2005). Eastern South Dakota lakes are commonly shallow and windswept; therefore, lakes are naturally turbid. Much of this turbidity is not related to nutrient input and subsequent planktonic production. This, in turn, makes trophic state indices based on planktonic production (i.e., total phosphorus, Secchi depth, or chlorophyll a) an unreliable source of human induced perturbations (Kuehl and Troelstrup 2013).

Similar to the findings of Drake and Valley (2005), lake size may not be an important category for IBI development in eastern South Dakota for several reasons. First, species richness does not appear to be strongly related to lake size as similar numbers of species were found per lake in both lake datasets. Second, fish communities within the Great Plains aquatic ecosystems are relatively depauperate compared to other ecosystems (Niemela et al. 1999; Bramblett et al. 2005). A greater potential diversity in lake communities could influence relationships between lake size and species richness (Matuszek and Beggs 1988). Third, most lakes (> 85%) in South Dakota are smaller in surface area. Thus, having fewer large lakes may muddle potential lake size-species richness relationships. While my findings corroborate previous findings that lentic IBIs may be developed across a wide array of lake sizes, further study is needed in order to determine the influence of lake size on IBI development in other geographic regions.

Overall, results of this study further support the use of extant data and standardized annual sampling in the development and use of a lentic IBI for eastern South Dakota. The same IBI metrics can be used for both small and large lakes in the same region. Thus, this lentic IBI can be used in tandem with existing data collection to provide a rapid assessment of water quality that can be used in conjunction with traditional physical and chemical measurements to provide a more holistic evaluation. Previously collected standardized annual data can used to calculate past IBIs by year to provide a history of biotic integrity, and future annual data can be readily used to track impairments or improvements to water quality based on the fish community.

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Table 3-1. Mean physiochemical characteristics for eastern South Dakota lakes for lakes within the original small lake dataset used to develop a fish-based IBI (see Chapter 2, this thesis) and test larger lake dataset that included larger than 1,000 ha. Numbers in parentheses represent the 95% confidence intervals.

Variable	Small lakes	Large lakes
N	58	11
Area (ha)	358 (305-412)	7,480 (4,273-10,688)
Alkalinity (mg/L $CaCO_3$)	224 (211-237)	230 (217-244)
Secchi depth (m)	1.48 (1.26-1.69)	2.00 (1.63-2.37)
Total nitrogen (mg/L)	1.73 (1.38-2.08)	1.48 (1.19-1.76)
Total phosphorous (mg/L)	0.23 (0.16-0.31)	0.20 (0.11-0.31)
Total suspended solids (mg/L)	14.26 (9.59-18.92)	9.32 (9.02-9.61)

Table 3-2. Fish species characteristics used to calculate metrics. Species are listed alphabetically by family and then by common name. The following abbreviations are used: I = intolerant, M = moderately tolerant, T = tolerant, In = insectivore, Om = Omnivore, Tc = top carnivore, N = native, NN = non-native, A11 = pelagophil, A12 = lithopelagophil, A13 = lithophil, A14 = phytolithophil, A15 = phytophil, B = nest guarder, B27 = speleophil. Table adapted from Jennings et al. (1999), Drake and Pereira (2002), and Whittier et al. (2007).[1] Species caught in small lakes. [2] Species caught in large lakes.

Species	Family	Tolerance	Feeding	Habitat	Origin	Reproductive
Bigmouth Buffalo *Ictiobus cyprinellus* [1,2]	Catstomidae	M	In	WC	N	A12
River Carpsucker *Carpiodes carpio* [1]	Catstomidae	T	Om	B	N	A12
Shorthead Redhorse *Moxostoma macrolepidotum* [1]	Catstomidae	M	Om	B	N	A13
White Sucker *Catostomus commersonii* [1,2]	Catstomidae	T	Om	B	N	A12
Black Crappie *Pomoxis nigromaculatus* [1,2]	Centrarchidae	M	Tc	WC	N	B
Bluegill *Lepomis macrochirus* [1,2]	Centrarchidae	M	In	WC	N	B
Green Sunfish *Lepomis cyanellus* [1,2]	Centrarchidae	M	In	WC	N	B
Hybrid Sunfish *Lepomis spp.* [1]	Centrarchidae	M	In	WC	N	B
Largemouth Bass *Micropterus salmoides* [1,2]	Centrarchidae	M	Tc	WC	NN	B
Orangespotted Sunfish *Lepomis humilis* [1]	Centrarchidae	T	In	WC	N	B
Pumpkinseed *Lepomis gibbosus* [1]	Centrarchidae	M	In	WC	N	B
Rock Bass *Ambloplites rupestris* [1,2]	Centrarchidae	I	Tc	WC	N	B
Smallmouth Bass *Micropterus dolomiteu* [1,2]	Centrarchidae	I	Tc	WC	NN	B
White Crappie *Pomoxis annularis* [1,2]	Centrarchidae	M	Tc	WC	N	B
Common Carp *Cyprinus carpio* [1,2]	Cyprinidae	T	Om	WC	NN	A14
Emerald Shiner *Notropis atherenoides* [1]	Cyprinidae	M	In	WC	N	A11
Golden Shiner *Notemigonus crysoleucas* [1]	Cyprinidae	M	Om	WC	N	A15
Spottail Shiner *Notropis hudsonius* [1,2]	Cyprinidae	M	In	WC	N	A12
Northern Pike *Esox lucius* [1,2]	Esocidae	M	Tc	WC	N	A15
Black Bullhead *Ameiurus melas* [1,2]	Ictaluridae	T	Om	B	N	B27

Table 3-2 – Continued. Fish species characteristics used to calculate metrics. Species are listed alphabetically by family and then by common name. The following abbreviations are used: I = intolerant, M = moderately tolerant, T = tolerant, In = insectivore, Om = Omnivore, Tc = top carnivore, N = native, NN = non-native, A11 = pelagophil, A12 = lithopelagophil, A13 = lithophil, A14 = phytolithophil, A15 = phytophil, B = nest guarder, B27 = speleophil. Table adapted from Jennings et al. (1999), Drake and Pereira (2002), and Whittier et al. (2007).[1] Species caught in small lakes. [2] Species caught in large lakes.

Species	Family	Tolerance	Feeding	Habitat	Origin	Reproductive
Channel Catfish *Ictalurus punctatus* [1,2]	Ictaluridae	M	Tc	B	N	B27
Tadpole Madton *Notorus gyrinus* [1]	Ictaluridae	M	In	B	N	B27
Yellow Bullhead *Ameiurus natalis* [1,2]	Ictaluridae	M	Om	B	N	B27
Shortnose Gar *Lepisosteus platostomus* [1]	Lepisosidae	T	Tc	WC	N	A15
White Bass *Morone chryops* [1,2]	Moronidae	T	Tc	WC	N	A14
Walleye *Sander vitreus* [1,2]	Percidae	M	Tc	WC	N	A12
Yellow Perch *Perca flavescens* [1,2]	Percidae	M	In	WC	N	A14
Freshwater Drum *Aplodinotus grunniens* [1]	Scianidae	T	Om	B	N	A11

Table 3-3. Results of principle components analysis on watershed land use and population density for pooled dataset of original and test lakes used in the development of a fish-based IBI for eastern South Dakota lakes.

				Coefficient eigenvectors					
Principle Component	Eigenvalue	Proportion of variance explained	Cumulative variance explained	% Cropland	% Pasture	% Shrub	% Forest	Population Density	% Barren
PCA 1	2.68	0.33	0.33	-0.62	-1.93	0.62	0.25	-0.52	-0.07
PCA 2	1.85	0.23	0.56	-0.06	-0.11	-0.16	0.07	0.55	-1.83
PCA 3	1.31	0.16	0.82	0.37	0.23	0.48	0.78	-0.09	-0.14

Table 3-4. Comparison of mean lake characteristics between four watershed disturbance categories. Within rows, means with the same lowercase letter are not significantly different (Tukey's multiple range test: $P \leq 0.05$). Data used was from multiple years and not all lakes were represented. Watershed disturbance category abbreviations are as follows: HAG/HPOP = high agriculture/high population density; LAG/HPOP = low agriculture/high population density; HAG/LPOP = high agriculture/low population density; LAG/LPOP = low agriculture/low population density.

Lake characteristic	N	Watershed disturbance category			
		HAG/HPOP	LAG/HPOP	HAG/LPOP	LAG/LPOP
Alkalinity (mg/L)	119	231 ^{ab}	204 ^{b}	249 ^{a}	204 ^{b}
Secchi depth (m)	123	1.37 ^{a}	1.99 ^{a}	1.66 ^{a}	1.30 ^{a}
Total nitrogen (mg/L)	127	1.80 ^{ab}	1.20 ^{b}	2.24 ^{a}	0.85 ^{b}
Total phosphorous (mg/L)	165	0.27 ^{a}	0.08 ^{b}	0.41 ^{a}	0.04 ^{b}
Total suspended solids (mg/L)	108	13.00 ^{a}	9.20 ^{a}	19.40 ^{a}	8.00 ^{a}

Table 3-5. Mean standardized metric values for eastern South Dakota lakes for lakes within the small lake dataset and large lake dataset. Numbers in parentheses represent the 95% confidence intervals. Metrics abbreviations are as follows: PCT_INT = proportion intolerant species; PCT_IN = proportion insectivores; RA_CENT = proportion Centrarchids; RA_ICT = proportion Ictalurids.

Metric	Small lakes	Large lakes
PCT_IN	0.27 (0.20-0.35)	0.31 (0.23-0.39)
PCT_INT	0.02 (0.01-0.03)	0.03 (0.00-0.08)
RA_CENT	11.0 (5.98-16.0)	9.67 (2.34-17.0)
RA_ICT	48.9 (40.2-57.6)	35.7 (21.0-50.3)

Table 3-6.- Summary of mean values for individual metrics and full IBI scores. Within rows, means with the same lowercase letter are not significantly different (Tukey's multiple range test; *P* < 0.05). Metrics abbreviations are as follows: IBI = full IBI model; PCT_INT = proportion intolerant species; PCT_IN = proportion insectivores; RA_CENT = proportion Centrarchids; RA_ICT = proportion Ictalurids. Watershed disturbance category abbreviations are as follows: HAG/HPOP = high agriculture and high population density; LAG/HPOP = low agriculture and high population density; HAG/LPOP = high agriculture and low population density; LAG/LPOP = low agriculture and low population density.

Metric	Watershed disturbance category			
	HAG/HPOP	LAG/HPOP	HAG/LPOP	LAG/LPOP
PCT_IN	0.15 ^{c}	0.37 ^{ab}	0.18 ^{bc}	0.53 ^{a}
PCT_INT	0.01 ^{b}	0.01 ^{b}	< 0.01 ^{b}	0.07 ^{a}
RA_CENT	0.04 ^{b}	0.13 ^{b}	0.02 ^{b}	0.32 ^{a}
RA_ICT	0.62 ^{a}	0.41 ^{ab}	0.54 ^{a}	0.16 ^{b}
IBI	18 ^{b}	34 ^{b}	20 ^{b}	58 ^{a}

Table 3-7.- Pearson's correlation coefficients for small and large lake pooled data between individual metrics and watershed disturbance (represented by principle component 1; PCA1). Metrics abbreviations are as follows: IBI = full IBI model; PCT_INT = proportion intolerant species; PCT_IN = proportion insectivores; RA_CENT = proportion Centrarchids; RA_ICT = proportion Ictalurids.

Metric	PCA1	P-value
PCT_IN	0.49	< 0.01
PCT_INT	0.31	0.01
RA_CENT	0.56	< 0.01
RA_ICT	0.37	< 0.01
IBI	0.56	< 0.01

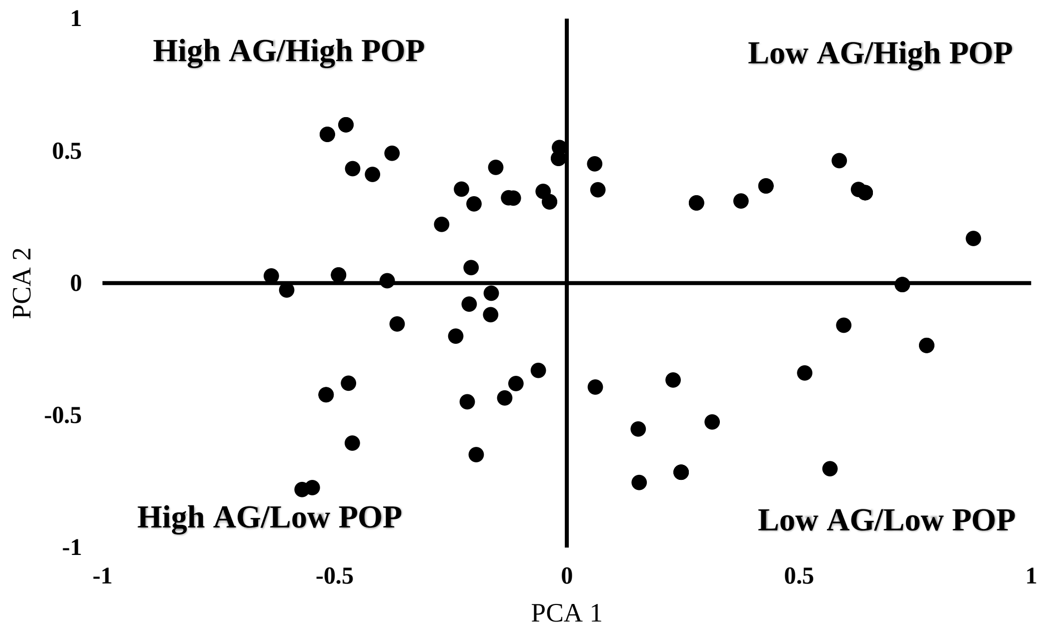

Figure 3-1. Plot of principle component 1 (PCA 1) versus principle component 2 (PCA 2) using both small (< 1,000 ha) lake and large (≥ 1,000 ha) lake datasets. Each quadrant was used as a watershed disturbance category: high agriculture and high population density, low agriculture and high population density, high agriculture and low population density, and low agriculture and low population density.

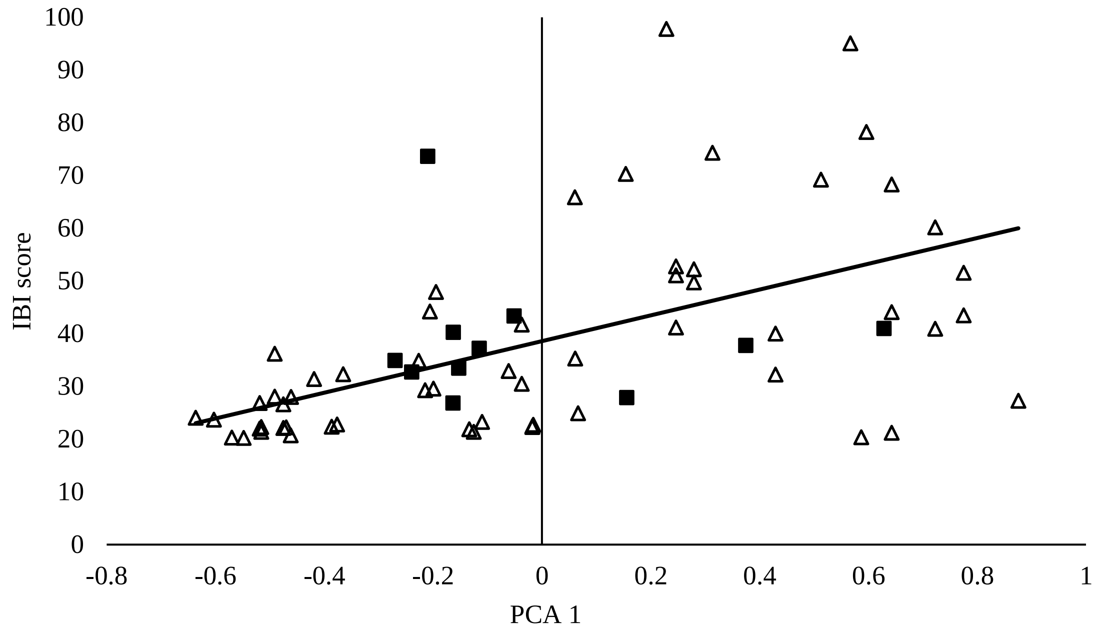

Figure 3-2. Scatterplot of IBI scores for original and test lakes against principle component 1 (PCA1). The solid line is the linear regression of the pooled data. Open triangles represent small (< 1,000 ha) lakes; filled squares represent larger (≥ 1,000 ha) lakes.

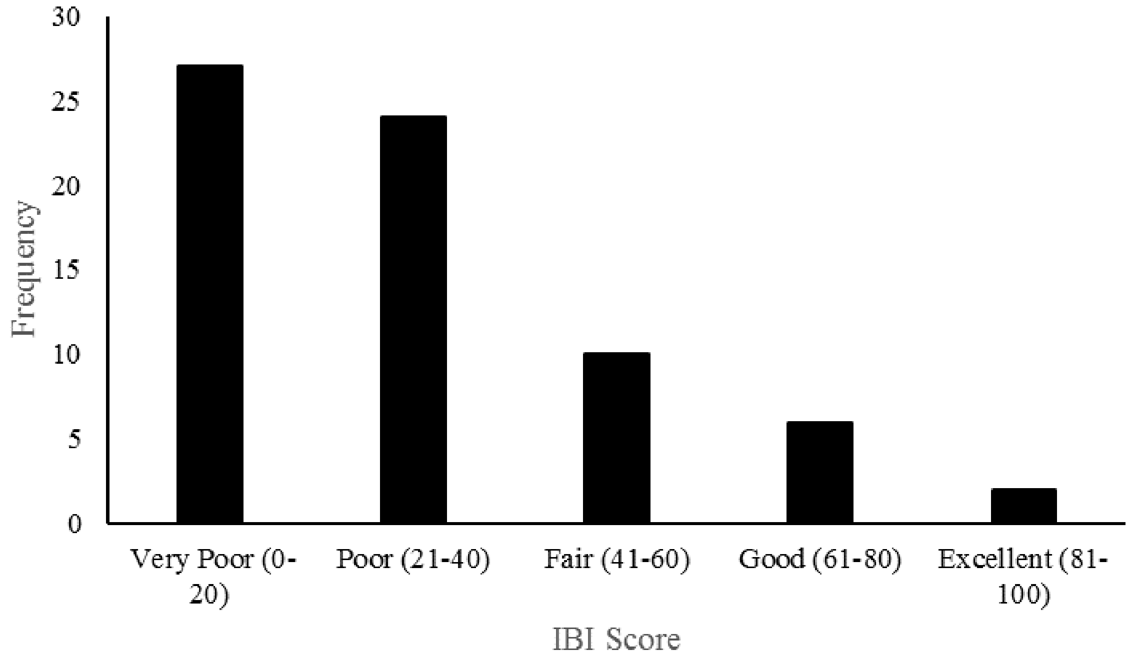

Figure 3-3. Distribution of IBI scores of pooled lakes (< 1,000 ha and ≥ 1,000 ha) in eastern South Dakota.

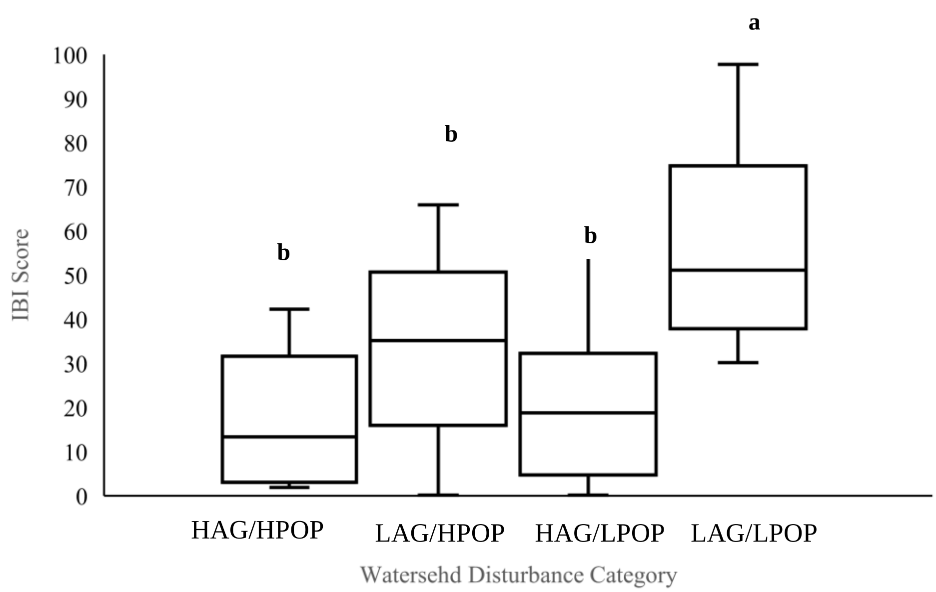

Figure 3-4. Box and whisker plots of lake index of biotic integrity (IBI) scores and watershed disturbance categories Watershed disturbance category abbreviations are as follows: HAG/HPOP = high agriculture and high population density (N = 24); LAG/HPOP = low agriculture and high population density (N = 15); HAG/LPOP = high agriculture and low population density (N = 17); LAG/LPOP = low agriculture and low population density (N = 13). Mean scores denoted with the same lowercase letter are not significantly different (Tukey's multiple range test; $P < 0.05$).

Appendix 3-1. List of final index of biotic integrity scores for lakes in eastern South Dakota. Lakes are broken down by county and South Dakota Department of Game, Fish, and Parks (SDGFP) region. Region 3 is located in southeast South Dakota; Region 4 is located in northeast South Dakota.

Lake Name	County	Score	SDGFP Region	Lake Name	County	Score	SDGFP Region
Albert	Kingsbury	31	3	Mid-Lynn	Day	48	4
Alice	Deuel	3	4	Minnewasta	Day	24	4
Antelope	Day	70	4	Mud	Kingsbury	7	3
Beaver	Minnehaha	9	3	Mud	Spink	19	4
Bitter	Day	42	4	Nine Mile	Marshall	1	4
Blue Dog	Day	71	4	Norden	Hamlin	31	4
Brant	Lake	16	3	North Buffalo	Marshall	30	4
Brush	Brookings	2	3	North Drywood	Roberts	< 1	4
Bullhead	Deuel	38	4	North Island	Minnehaha	2	3
Byron	Beadle	< 1	3	North Scatterwood	Edmunds	12	4
Campbell	Brookings	6	3	Oak	Brookings	4	3
Campbell Slough	Day	78	4	Opitz	Day	48	4
Cattail/Kettle	Marshall	35	4	Pelican	Codington	38	4
Clear	Minnehaha	11	3	Pickerel	Day	62	4
Clear	Deuel	3	4	Poinsett	Hamlin	38	4
Clear	Marshall	51	4	Preston	Kingsbury	41	3
Cochrane	Deuel	66	4	Punished Woman	Codington	12	4
Cottonwood	Clark	35	4	Reid	Clark	48	4
Cottonwood	Marshall	36	4	Round	Deuel	22	4
Cottonwood	Spink	0	4	Roy	Marshall	66	4
Diamond	Minnehaha	8	3	Silver	Hutchinson	2	3
East 81	Brookings	3	3	Sinai	Brookings	32	3
East Oakwood	Brookings	26	3	South Buffalo	Marshall	45	4
Enemy Swim	Day	95	4	South Red Iron	Marshall	32	4
Fish	Deuel	22	4	South Rush	Day	35	4
Goldsmith	Brookings	13	3	Spirit	Kingsbury	21	3
Hazeldon	Day	40	4	Thompson	Kingsbury	38	3
Herman	Lake	19	3	Twin	Minnehaha	3	3
Horseshoe	Day	98	4	Twin	Sanborn	18	3
Kampeska	Codington	37	4	Twin	Spink	5	4
Lardy	Day	51	4	Waubay	Day	36	4
Lily	Day	30	4	West 81	Kingsbury	16	3
Long	Lake	3	3	West Oakwood	Brookings	13	3
Lynn	Day	52	4	Whitewood	Kingsbury	56	3
Madison	Kingsbury	30	3				

CHAPTER 4. SUMMARY AND RESEARCH NEEDS

The development of a lentic Index of Biotic Integrity (IBI) for lakes in eastern South Dakota supports the continued research into the creation and testing of lake IBIs across all ecoregions. Despite the inherent characteristics of lakes, lentic fish communities, and fisheries management for these systems compared to those of lotic systems, it may be possible to develop IBIs in these systems. The results of my studies add to previous research that support the use of IBIs in lentic systems across the U.S. (Jennings et al. 1999; Whittier 1999; Drake and Pereira 2002).

Further, my research demonstrates that lentic IBIs can be developed using extant fish community data from annual standardized surveys that largely target recreationally important fishes. Only two gear types (gill nets and modified fyke nets) were used to collect fishes in these surveys. The necessity of only two different gears is unlike other lake IBIs that use three or more gears for sampling (Whittier 1999; Jennings et al. 1999; Drake and Pereira 2002). The inclusion of small-bodied fishes collected by seining made no difference in IBI development. However, further research on whether small-bodied fishes should be included in the lentic IBI for eastern South Dakota is needed (see Chapter 2). Regardless, my results support the use of this lentic IBI which can be calculated relatively quickly from annual standardized fisheries surveys. The ease of calculation can allow for IBIs of individual lakes to be calculated over previous decades and into the future to identify trends in biotic integrity and relate those to changes in land use or water quality.

My study is only the second to examine whether lake size influenced IBI development. Lake size has been shown in previous studies to relate to species richness

(Matuszek and Beggs 1988; Magnuson et al. 1994). Many IBIs include metrics related to species richness (e.g., Jennings et al. 1999; Whittier 1999; Drake and Pereira 2002), and differences in richness between lakes could influence the ability of the IBI to respond to differences in watershed disturbance or water quality (Drake and Valley 2005). Drake and Valley (2005) also found that an IBI developed for small Minnesosta lakes could also be used to assess large lakes. Fish community variability between lakes may be explained by many other abiotic factors associated with lentic systems, including (but not limited to): lake depth (Matuszek and Beggs 1988; Magnuson et al. 1994), elevations and latitude (Matuszek and Beggs 1988), and alkalinity (Rahel 1986). Further research should examine whether lentic IBIs should be developed based on these other abiotic factors in order to increase the chance of successfully developing lentic IBIs across other ecoregions of the U.S.

The IBI that I developed in my research was restricted to natural lakes in the eastern half of South Dakota. To date, no work has been done to develop a reservoir IBI in this state, but reservoirs are important resources, especially west of the Missouri River. Previous work has developed IBIs including reservoirs in Tennessee (Jennings et al. 1995), New Jersey (Blocksom et al. 2002), and Serbia (Lenhardt et al. 2009). Reservoirs are unique in that their fish communities are often composed of both lentic and lotic species and, thus, may respond in different ways to land use or water quality changes compared to just lentic or lotic systems alone (Powers et al. 2013). Future research could repeat my work on reservoir systems to determine whether IBI development is possible in these systems and whether such IBIs should be based on reservoir size or include small-bodied fishes.

Overall, my research adds to the body of work on IBI development in South Dakota (e.g., Krause et al. 2013) and the development of lentic IBIs across the U.S. As land use continues to change in South Dakota, it is important to have a rapid assessment tool to evaluate how fish communities respond in kind. Trends in historic and future IBIs calculated on the same lake can be combined with historic and future data on physical and chemical measures to provide a more complete picture of water quality in these systems. Collectively, this information can be used to identify where management interventions (e.g., in-lake habitat restoration, watershed habitat modifications) are needed, prioritize efforts, and monitor fish community changes in response.

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