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Development of IBI metrics for lakes in southern New England

Thomas R. Whittier
Dynamac International, Inc.
200 SW 35th St.
Corvallis, OR 97333
whittier@heart.cor.epa.gov
541-754-4455

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INTRODUCTION

Among the principles implicit in the field of ecology are assumptions that: (1) living organisms react to changes in their ecosystems, (2) we should be able to detect and quantify changes in biological assemblages due to adverse stress caused by human activity, and (3) these changes should be predictable and generalizable to broad classes of ecosystems and assemblages. In the early 1980s Karr and his colleagues successfully applied these principles to fish assemblages in warmwater streams in the Midwest (Karr 1981, 1991; Karr et al. 1986; Lyons 1992; OEPA 1988). The resulting Index of Biotic Integrity (IBI) has been repeatedly modified for other assemblages, regions and ecosystems (e.g., Leonard and Orth 1986; Miller et al 1988; Steedman 1988), with varying degrees of success.

To date there has been only limited effort toward developing IBIs for lakes. Minns et al. (1994) developed an eight metric IBI for littoral areas in the Great Lakes. Dionne and Karr (1992) and Jennings et al. (1995) presented preliminary results for a Reservoir Fish Assmeblage Index for the Tennessee Valley, but concluded that additional work was needed. The only other published research for inland lakes is in Wisconsin by Jennings et al. (this book). They evaluated metric variability related to sampling methods, and how the metrics performed as indicators of human induced stress. They were satisfied with the performance of only four metrics.

In this chapter, I review some of issues that may account for the apparent difficulty in developing inland lake IBIs. I then present an evaluation of several candidate metrics for lakes in southern New England, and discuss some the implications of the results and needs for further research.

IMPEDIMENTS TO DEVELOPING LAKE IBIS

The reasons that development of lake IBIs lags behind that for warmwater streams may be divided into four groups: (1) scope of assessment, (2) sampling issues, (3) perceived values of lakes and the management practices that support those values and (4) ecological factors. The issues discussed below are generalizations meant to emphasize differences, exceptions to each point can be found. Clearly, these concerns are not the exclusive domain of lake assessments.

SCOPE OF ASSESSMENT

For streams the conceptual and actual units for biological assessments are not entire streams, but rather separated points or reaches along the stream length. In reporting results, the points or reaches are plotted or tabled as individual samples and the assessment of biotic condition usually integrates the individual samples and changes along the stream length. For lakes, the conceptual units are usually whole lakes, although sampling is usually at stations or portions of shoreline; and the data are generally combined to represent the entire lake. Only when lakes are sufficiently large are sampling, analysis and assessment done on subunits (e.g., coves, bays, arms).

Another facet of this issue is one of "apples and oranges". Biological assessments need to be developed for classes of reasonably comparable ecosystems (Plafkin et al. 1989). An IBI is based on expected-characteristics of a particular assemblage type, in a particular size and type of waterbody, in a particular ecoregion or basin. For streams, assemblage expectations are usually defined for ecoregion (or basin), temperature type (cold or warm), and stream size. The classes of ecoregions and temperature types usually can encompass a very large number of streams in the same assessment framework.

For lakes, classing waterbodies into reasonably comparable groups is more complex. Stream size describes habitat volume. Habitat volume in lakes is more multi-dimensional. Lake volume defines only the maximum possible habitat volume. The useable volume may be limited by oxygen depletion (either natural or amplified by anthropogenic eutrophication) and temperature extremes in summer or oxygen depletion in winter and thus may vary greatly within and among years. Natural differences in stratification regimes affect expectations for the fish assemblages, as does basin shape which determines the proportion of the habitat that is littoral versus pelagic. Watershed position and relative connectedness are also important; a large headwater lake or a large isolated lake could be expected to have a fish assemblage which differs from a large well connected lake closer to a mainstem river.

SAMPLING ISSUES

Biological assessments require data collected in a consistent manner, that represent or index the resident assemblages (Plafkin et al. 1989). For wadeable streams, electrofishing provides appropriate data for IBI assessments (e.g., OEPA 1988). For lakes no single gear is sufficient to sample in all habitat types nor for all species. Numerous studies have examined the issue of gear selectivity, sufficiency of effort and sampling variability in northern lakes (e.g., Weaver et al. 1993). Whenever more than one gear or method is used, questions arise of how and whether to combine data, and what constitutes a unit of sampling effort. Jennings et al. (this book) propose evaluating and using data from different methods for different metrics, as well as limiting the data collection to littoral assemblages.

Problems with using multiple gear are compounded further when certain effective methods cannot be applied in all lakes. Night beach seining is often the only effective method for collecting cyprinids, darters and sculpins, but many lakes lack beaches clear of obstacles, suitable for seining. Electrofishing is a good single method for sampling the littoral zone, but losses its effectiveness in low conductivity lakes common in northern regions, and in lakes with very narrow littoral zones. Likewise, gill nets are the only reliable method for collecting salmonids and other pelagic species, but resource managers of low productivity northern lakes often restrict or forbid the use of gill nets.

PUBLIC PERCEPTIONS AND MANAGEMENT OF LAKES

The public, and therefore management agencies, value different things about lakes than they value for streams. The public often values a park-like aesthetic for lakes, as a place for homes, vacation cabins, and parks for camping, picnics, and swimming. People like the water; they like to see it, be near it, on it, or in it, and often they want to use the water for drinking. Except for drinking water, lakes are more amenable to these things than are streams. Activities in, on, or near the water lead people to want water clear of obstacles (vegetation, snags, etc), for boating, fishing and swimming. And of course, people want good fishing, regardless of whether the desired species is native or even suited for "their" lake.

Lakes are often managed to enhance the values listed above, usually to the detriment of biotic integrity. Snags and weeds are cleared for boating, swimming, docks, and front yard aesthetics. Retaining walls or riprap are added for erosion control (Jennings et al. 1996; Christensen et al. 1996). Lakes have been subjected to stocking and other manipulations of fish assemblages for well over a century. In southern New England much of the public is not aware that most of the currently widespread game fish are not native (e.g., largemouth and smallmouth basses, bluegill, northern pike, black crappie, brown, rainbow and lake trouts).

Management goals are closely linked with who does the managing. Streams tend to be managed by, or under the authority of state departments of environmental quality, who are concerned with issues defined by the Clean Water Act and subject to oversight by the U.S. EPA. This is in part due to historic problems from point-source pollution on streams and rivers. As the obvious water quality problems in streams are corrected, many of these agencies are shifting their regulatory emphasis to the biological condition of streams, for which the IBI is well suited.

Lakes, on the other hand, are managed by various legal entities. In some lake-rich regions where lakes receive more recreational use than streams, lakes in the public domain are often managed by state fish and game departments. Their mandates emphasize management for fishability, not biotic integrity. Unfortunately in some instances, interagency politics has led to lakes and streams becoming the exclusive territory of different agencies wherein environmental quality agencies often only become involved with lakes over eutrophication and fish tissue contaminants issues. Finally, unlike streams and rivers, lakes often have what amounts to owners: private individuals and families, lake and home owners associations, sports clubs, organization camps, resorts, and water districts. Technically many of these lakes are public, but in reality the owners of the land surrounding lakes limit access to and activities on "their" lake, and the public agencies often choose not to press their legal mandate. It is reasonable to assume that most lake owners do not manage for biotic integrity.

ECOLOGICAL FACTORS

In general, lakes tend to be new geographic features compared with rivers and streams, and fishes have had longer to speciate in, and adapt to flowing water. Thus, there tends to be fewer lake-dwelling species. For example, in southern New England the regional species pool of lake dwellers is considerably smaller than the already depauperate species pool for lotic systems (Miller et al. 1988; Halliwell et al. This book). There are only three native sunfish; pumpkinseed is nearly ubiquitous, and the other two are rather uncommon. Depending on the location, there are one or two each of suckers, darters, and native top carnivores, and three to five possible lake-dwelling minnows (Whittier et al. 1997a). In addition, for more than a century the fish fauna in New England has been liberally augmented with non-natives. In the Northeast an estimated 64% of all lakes have at least one non-native species, with non-native individuals outnumbering the natives in 25% of lakes (T. Whittier unpub. data). Introductions are especially common in southern New England where nearly all lakes have non-natives.

Fish species' tolerances to stress (whether natural or anthropogenic) in lake ecosystems often differ from their tolerances in stream ecosystems. Whittier and Hughes (in review) evaluated 45 species' tolerances to five anthropogenic stressors in Northeast lakes. They found that eight species usually classified as tolerant or moderately tolerant of disturbance in streams appear to be intolerant or moderately intolerant of degraded conditions in lakes. Five species usually classified as intermediately tolerant in streams were very tolerant in Northeast lakes.

METHODS

SAMPLE DESIGN & FIELD METHODS

The data used here were from the U.S. Environmental Protection Agency's Environmental Monitoring and Assessment Program (EMAP) Northeast Lakes Pilot (Larsen et al. 1991, 1994; Stevens 1994). EMAP sampled fish assemblages, as well as water chemistry, zooplankton, physical habitat and riparian birds in 179 lakes and reservoirs during the summers of 1992-94 in the Northeast USA (New England, New York, New Jersey). The lakes were selected, using a systematic random design, from all lakes > 1 ha. For this study, I used the 50 lakes in southern New England (MA, CT, RI and the southern 1/3 of NH) and five additional lakes, purposively selected and sampled in 1991 for methods evaluation (Larsen and Christie 1993).

Fish assemblages were sampled with overnight sets of gillnets, trapnets and minnow traps, and by night seining (Whittier et al. 1997b). The level of effort was determined by lake size. The sampling objective was to collect a representative sample of the fish assemblage at each lake. Sites were selected using a stratified random design. Littoral fish sampling was done at random stations within each macrohabitat class. Pelagic sample sites were chosen in random directions from the deepest location. The collected fish were identified to species and counted. As part of the EMAP Quality Assurance procedures (Chaloud and Peck 1994) specimens of all species were vouchered with the Museum of Comparative Zoology at Harvard University to confirm identifications and for permanent archival. To develop the metrics evaluated in this study, I combined data from all gear. Water samples were collected at 1.5 m at the deepest part of the lake. Field methods are detailed in Baker et al. (1997), which is available from the author. Field and laboratory data from the Northeast Lakes Pilot may be found on the EMAP's website (http://www.epa.gov/docs/emfjulte/html/dataI/surfwatr/index.html).

QUANTITATIVE METHODS

I chose two measures of anthropogenic stress to evaluate metric performance: total phosphorus (TP) as a measure of eutrophication stress, and the extent of human activity in the watershed as a measure of generalized human induced stress. The latter is an indirect measure of stress, but is based on the premise that increased human activity in the watershed increases the frequency and strength of a multitude of anthropogenic perturbations to the lake ecosystem. Although acidification is also a stressor in some areas of the Northeast, only four of the sampled lakes in southern New England had pH<6.0. Preliminary examination of the fish assemblage data did not reveal any distinct qualitative acidification effects, thus I used pH stress only to aid interpretation of metric behavior.

Watershed-scale measures of human disturbance were developed from digitized coverages of human population and road density (Census Bureau 1991; USEPA 1992), and land use/land cover (USGS 1990) from which proportions of the watershed in urban, industrial/commercial, residential, forested, wetlands and agricultural categories were calculated (C. Burch Johnson, unpub. data). To estimate human disturbance in the watershed, I used the first axis scores of a principal components analysis (PCA) (Gauch 1982, PROC PRINCOMP; SAS 1985) of the land cover (% forest, % urban, % agricultural), human population density and road density, as described in Whittier et al. (1997a). The first principal component accounted for 62% of the variability in these five variables, with % forests loading negatively, and all human activity variables loading positively (Table 1).

To determine species' native ranges I examined species maps and descriptions in a variety of fisheries texts (Kendall 1914; Hubbs and Cooper 1936; Hubbs and Lagler 1964; Scott and Crossman 1973; Lee et al. 1980; Trautman 1981; Becker 1983; Smith 1985; Schmidt 1986; Underhill 1986; Page and Burr 1991), and state biological survey reports from New York (NYSDC 1927-39) and New Hampshire (NHFGC 1937-39). Trophic guild and habitat preferences were based on these sources along with the summary tables in Halliwell et al (this book), Karr et al. (1986), Ohio EPA (1989), Lyons (1992), and Minns et al. (1994). Most tolerance classifications were from Whittier and Hughes (in review); for the remaining uncommon species I used information contained in all of the above sources to make tolerance classifications.

Due to the relatively low native species richness in New England, I anticipated that several commonly used metrics would not be effective. Thus, I examined a large number of potential metrics (e.g., Miller et al. 1988; Simon and Lyons 1995). To evaluate candidate metrics I considered the following: How many species contributed to the metric? What was the statistical distribution of raw scores among lakes? Was there a lake size effect? How do raw metric scores relate to the two measures of stress? In particular, do the raw metric values distinguish between the most degraded and the least degraded lakes? What ecological characteristics do the metrics represent and how do we expect these to change over a range of natural conditions and human-induced stress? I primarily used a graphical-based approach (Fore et al. 1996), examining scatterplots of raw metric data and residuals from regressions .

RESULTS

In the summers of 1991-94, EMAP sampled 55 lakes and reservoirs in southern New England (SNE; Figure 1). All, except three, lakes were <300 ha (Figure 2). Eighteen of the lakes are in the Northeastern Highlands ecoregion (NHE), with the remainder in the Northeastern Coastal Zone (NCZ) ecoregion (Omernik 1987). Ten lakes were eutrophic or hypereutrophic, with the remainder split between mesotrophic and oligotrophic (Figure 2a-b). Phosphorus values tended to be higher in NHE and in lakes <100ha. There was no association with lake size for watershed disturbance (Figure 2c-d). Watersheds were less disturbed in NHE. Water level control structures are common on Northeast lakes (pers. obs.); 10 of SNE sample lakes have "Reservoir" in their names. Named SNE reservoirs tend to be the larger lakes and have relatively low stressor values.

Forty fish species were collected, 18 of which are native to the region, 20 are introduced and two are native to some of the lakes (Table 2). Of the 40 species collected, 18 were found in three or fewer lakes. Assemblages were characterized by centrarchids, yellow perch, chain pickerel, golden shiner and bullheads. Bluegill tended to dominate in the south, being replaced by yellow perch to the north.

SPECIES RICHNESS METRICS

Nearly all of the original (Karr 1981) and variants (Miller et al. 1988; Simon & Lyon 1995) of the species richness metrics were problematic in SNE. The total number of species EMAP collected was similar to the total (44) used by Minns et al. (1994) in the Great Lakes. But slightly more than half of the SNE species are not native to New England, compared to 25% non-native species collected by Minns et al. Many of the New

England non-natives were introduced 100+ years ago and are firmly established (NHFGC 1937-39). Some authors suggest that these species are now "naturalized" and should be considered as part of the resident species pool (Halliwell et al. this book). Under that proposal only the two non-North American species (common carp and brown trout) and populations maintained by stocking (nearly always salmonids in SNE) could be considered non-native. There are currently no state run warm-water species stocking programs in SNE (D. Halliwell pers. comm.) I followed a strict interpretation of native status for this assessment.

The number of native species in SNE lakes increased with lake size (r^2=0.41, p=0.0001; Figure 3a), ranging from 2 to 9 species (11 in one lake). Two features of these data differed from those in Midwestern streams. First, there were apparently no fishless lakes in SNE, while some fairly large streams (watersheds up to 2000 km^2) in Ohio were fishless (Whittier and Rankin 1992). Second, the data formed a band rather than a wedge shape (i.e., variance about the regression line was fairly constant rather than increasing with lake size). To be a useful metric native species richness residual scores, from the regression with lake size, should generally decrease with increasing stressor scores (i.e., fewer than "expected" natives in more stressed lakes and more native species in less stressed lakes). However, there was no apparent pattern in these data (Figure 3b-c). Jennings et al. (this book) also found no relationship between native species richness and human impact as measured by a Trophic State Index. Total species richness showed similar patterns and lack of relationship to stressor measures. Treating "naturalized" (non-stocked) North American species as natives produced a pattern similar to that for total species richness.

The number of introduced species was a useful metric in Great Lakes littoral areas (Minns et al. 1994). In SNE all, except three, lakes had between 1 and 6 non-native species (Figure 4a), generally increasing with lake size (r^2=0.32 p=0.0001). Residual scores plotted against TP showed a lack of pattern with increased stress, except that several of the relatively low TP lakes with high residuals (more non-native species than expected) were stocked with 1-3 salmonids (Figure 3b). Non-native species residuals showed the expected pattern with watershed disturbance (Figure 3c); a number of lakes with high residual scores and intermediate watershed disturbance were stocked. Thus, the number of non-native species adjusted for lake size should be a useful metric.

Most other species richness and composition metrics used in other IBIs did not appear to be applicable in SNE. There were only two rarely collected darters (and no sculpins) (Table 2). Likewise, there were only two suckers, with the creek chubsucker collected only once. Cyprinid species richness was somewhat higher; however 4 of 5 native minnows were uncommonly collected. Golden shiner was widespread (72% of sampled SNE lakes) and is tolerant of degraded conditions. Only 8 of the 55 sampled lakes had native cyprinids other than golden shiner; six of these had only one other minnow. Native minnow species richness may be a useful metric, but it would be essentially a binary metric. Of the three native sunfish in SNE (Table 2), pumpkinseed was ubiquitous (91% of sampled lakes). The other two sunfish were uncommon and were not collected in the same lakes, preferring very different habitats. Therefore a native sunfish metric contained very little information.

Atlantic salmon and fathead minnow were the only SNE lake fish rated intolerant of anthropogenic stress in Northeast lakes by Whittier and Hughes (in review). Lowering the threshold to tolerance scores of 11 (of 16) added 13 uncommon species (11 of these were collected in only 1 or 2 lakes each; Table 2). Only five lakes had more than one species from the expanded list of intolerants and most had none. There was a very weak relationship between lake size and intolerant species (r^2=0.14, p=0.0045). Maximum intolerant species (uncorrected for lake size) tended to decrease with increased TP, most lakes with intolerant species >0 had moderate to low watershed disturbance. Thus, this metric may be useful, although most lakes would get the minimum metric score.

One of the features of the original sucker species metric was their longevity, providing a "multiyear integrative perspective" (Karr et al. 1986). To address the large-bodied, long-lived component of lake fish assemblages, I selected five relatively large native species to comprise an analogous metric. There was a significant, but weak, relationship between large species richness and lake size (r^2=0.19, p=0.009; Figure 5a). There was little association between increased TP and fewer large native species (Figure 5b). Both the regression residuals (Figure 5c) and the raw species counts tended to decrease with increased watershed disturbance. The four lakes circled in the lower left of Figure 5c are subject to manipulations (e.g., reclamation, draining) which are not detected by either stressor measures. This appears to be a useful metric. A similar analysis using 10 native small species (4 minnows, 2 sunfish, 2 darters, and killifish) showed no associations with any of the stressor measures.

Perhaps the most frequently replaced metric is % green sunfish (Simon and Lyons 1995). In SNE, % bluegill would be the most likely candidate. Bluegill is not native and sometimes dominated the assemblages, being most abundant in CT and RI, becoming less dominant in MA and being replaced by yellow perch as a dominant in NH. It occurred in only two NHE lakes. In the NCZ ecoregion where it was most widespread, % bluegill was not associated with higher stressor measures. A variant metric might be % individuals of the most abundant non-native species (often bluegill). This did not improve the metric performance. Somewhat better was the proportion of individuals of all non-native species (or conversely proportion of native individuals). This metric showed no lake size effect, and no relationship with TP (Figure 6a). However, there was a tendency for the % non-native individuals to increase with watershed stress. Most of the lakes in the upper left of Figure 6b had small watersheds and were probably more stressed than indicated by the watershed analysis. This metric is probably useful.

There were nine tolerant species collected (Table 2). Eleven of the 12 smallest lakes had >90% tolerant individuals (Figure 7a). Pumpkinseed were fairly abundant in many of these ponds. Removing pumpkinseed from this metric had the greatest effect in smaller lakes. There was a clear association between % tolerants and TP. With pumpkinseed included, all eutrophic and hypereutrophic lakes (except one) had >80% tolerant individual (Figure 7b). The pattern was less dramatic with pumpkinseed removed (Figure 7d). For watershed disturbance, the pattern of increasing proportion of tolerant individuals with increasing stress was stronger with pumpkinseed removed (Figure 7c & e).

TROPHIC COMPOSITION METRICS

As with the species richness metrics, the trophic composition metrics were problematic in SNE lakes. For example, chain pickerel and American eel were the only native top carnivores (Table 2). Other natives such as yellow perch, white perch and brook trout are piscivorous as large adults. However, white perch is native in only about a third of the SNE lakes and tolerant of stressed conditions. Brook trout is native to SNE streams, but is nearly always maintained by stocking in lakes. The most widespread top carnivore was the non-native largemouth bass (in 50 of 55 sampled lakes) which was often found in quite degraded lakes, as was black crappie. I selected eight species that are strongly piscivorous and not highly tolerant of stressed conditions (Table 2). I also examined this metric with largemouth bass included. Any species maintained by stocking was not included for that lake, which effectively eliminated Atlantic salmon and lake trout.

For the restricted species list (excluding largemouth bass), there was a slight tendency for % top carnivores to increase with lake size. The highest scores occurred in the lakes with TP < 20 and low to moderate watershed disturbance (Figure 8a-b). The one outlier lake had 19% of individuals as rock bass; removing rock bass would still leave this lake with one of the highest % top carnivores scores. Including largemouth bass tended to decrease the resolution of this metric (Figure 8c-d). That is, the peaks of the scatter plots moved to the right on the stressor plots, and the range of scores approximately doubled. There is concern that removing a highly tolerant species changes the metric from a trophic guild metric into a tolerance metric. I believe that the original intent of the metric is maintained with the reduced list.

A % insectivorous individuals metric also required some adjustments. Only seven uncommonly collected species meet the strict definitions of insectivory of Halliwell et al. (this book). I selected six additional native generalist feeder species that tend toward the insectivorous end of the trophic spectrum, and which are not highly tolerant of degraded conditions (Table 2). With this species list % insectivorous individuals scores related very well to TP and watershed disturbance (Figure 9b-c). The three outlier lakes in the upper right of the watershed disturbance plot all had yellow perch as the most abundant species. Also problematic for this metric was that 11 of 12 smallest lakes had <10% insectivores (Figure 9a). Eight of these small lakes had pumpkinseed as a dominant species. Adding pumpkinseed to the insectivore species list for small lakes removed the size effect, but did not improve the relationship with the stressor measures.

There are few true omnivore species (with a substantial portion of their diet including plant material) in the Northeast (Halliwell et al. this book), but a fairly large number of generalist feeder species (feeding at several trophic levels rather than primarily from one trophic level [Minns et al. 1994]). I selected 10 species for a generalist feeder metric (Table 2). The maximum scores for % generalist feeder individuals decreased with lake sizes greater than about 30 ha (Figure 10a). Most of the small lakes had high scores due generally to dominance by bluegill and pumpkinseed. The lowest scores (< 30%) were all in relatively low productivity lakes (TP < 20). The highest scores tended to be in lakes with moderate levels of stress (Figure 10b-c).

DISCUSSION

Seven of the candidate metrics appear to be successful and should provide a framework for future refinement of lake metrics and development of an IBI for inland lakes in southern New England. Two are species richness metrics, non-native species richness adjusted for lake size as a negative metric, and large species richness (which may not need to be adjusted for lake size) as a positive metric. The % non-native individuals and % tolerant individuals metrics could be considered in the richness and composition category (Karr et al 1986) or in the indicator species category of Simon and Lyons (1995) as negative metrics. Minns et al. (1994) placed the former metric in the abundance and condition category. It may be possible to develop an intolerant species metric, or a small species metric. With judicious selection of species, all three trophic composition metrics appear to be useful.

The results presented here also illustrate some of the conceptual and ecological issues important for assessing biotic integrity of inland lakes, and differences between lotic and lentic fish assemblages. Lakes in New England are naturally species depauperate (Schmidt 1986). In this way they are somewhat analogous to coldwater streams, where increased total species richness is usually an indication of degradation (Lyons et al. 1996). In New England lakes, the source of additional species is introductions, rather than immigration from warmwater streams. The relationship between native species richness and introductions is complex for SNE lakes. Moyle and Light (1996) proposed that most successful invasions occur without loss of native species and that non-native predators should have greater effect on native assemblages than non-predators. A substantial proportion of non-native species in SNE are predators. For the Northeast as a whole, Whittier et al. (1997a) demonstrated an association between increased predator richness (usually from non-natives) and lower minnow richness. However, preliminary analyses could not demonstrate an overall decrease in native species richness with increased numbers of non-natives species (T. Whittier unpub. data). Thus, in SNE there appears to adequate ecological "space" for the additional non-native species in general, but some portions of the native assemblage may have been extirpated. It probably cannot be known whether SNE lakes had more minnow species prior to introduction of additional littoral predators.

A third rule for species invasion (Moyle and Light 1996) was that increased invasion success occurs when native assemblages are depleted or disrupted. The greater introduced species richness, and higher proportion of non-native individuals in more stressed lakes support this idea. However, it appears that these stress levels have not generally been high enough to eliminate native species, because native species richness showed no association with anthropogenic stress.

Fish species tolerance of, or sensitivity to degraded lake conditions appear to differ from their tolerances to impaired lotic systems. When I used species tolerance classifications from stream IBIs the proportion of tolerant individuals was lowest in the most stressed lakes (T. Whittier unpub. data). This led Whittier and Hughes (in review) to evaluate individual species tolerances to five stressor measures in Northeast lakes, and assign new tolerance ratings. Applying these revised classes to the SNE assemblage data generally produced the expected associations with stressor measures, except in the smallest lakes and ponds (≤ 10 ha). These results, coupled with the larger species pool for lotic fish compared with lentic fishes, pose some interesting questions relative to how lake ecosystems stress fish compared stream ecosystems, and whether there are any truly intolerant warmwater lake species.

The trophic composition metrics presented a number of interesting challenges, in addition to the guild membership issues (e.g., lack of objective criteria, dietary plasticity and changes with age) discussed by Minns et al. (1994). In SNE lakes, a number of species with obvious trophic guild membership react to degraded conditions in the opposite manner to what the metric should indicate. The clearest examples are the piscivores. For most of SNE chain pickerel and American eel are the only native top carnivores. Largemouth bass is clearly a top carnivore, but is tolerant of degraded lake conditions. Black crappie and white perch are usually placed in the piscivore guild, but are two of the most tolerant species in SNE lakes (Whittier and Hughes in review), and among the dominant species in many stressed lakes. Finally, 16 of 40 species collected by EMAP in SNE lakes could be considered as piscivores (Halliwell et al. this book), producing very high % piscivorous individuals scores (40 -100% for more than half of the lakes). In selecting species for this metric, I did not try to choose the most sensitive piscivores, but limited the list to the most carnivorous species that were also not highly tolerant, regardless of the native status. This produced metric values in the 0 - 10% range and the expected association between the metric and stressor scores. Similar adjustments were needed for the other two trophic composition metrics.

Additional work is needed in a number of areas. First, biomass data may have the potential to improve all of the trophic guild metrics, and provide additional abundance and condition metrics (Minns et al. 1994). However EMAP, which was not primarily a fisheries survey, did not collect those data. Second, some standard unit of sampling effort needs to be established. Multiple gear sampling makes that a complex problem. A sampling scheme combining electrofishing and gill nets may provide abundance data with lower variability. Third, additional work on lake classification schemes is needed. Even within a relatively homogenous area like southern New England there are questions about whether lakes in the Northeast Coastal Zone ecoregion should be held to be same standard as those in the Northeast Highlands ecoregion. Finally, work is needed to develop metric scoring and an overall IBI, and to validate both the metrics and the index with additional data.

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REFERENCES

Baker, J. R., Peck, D. V., and Sutton, D. W. (Eds). 1997. *Environmental Monitoring and Assessment Program--Surface Waters: Field Operations Manual for Lakes.* EPA/620/R-97/001. U.S. Environmental Protection Agency, Washington, DC.

Becker, G. C. 1983. *Fishes of Wisconsin.* University of Wisconsin Press. Madison, WI.

Census Bureau. 1991. *Census of population and housing, 1990: Summary tape file 1 on CD-ROM (name of State).* Washington, DC.

Chaloud, D. J., and Peck, D. V. (eds.). 1994. *Environmental Monitoring and Assessment Program: Integrated Quality Assurance Project Plan for the Surface Waters Resource Group, 1994 Activities.* EPA 600/X-91/080 (Revision 2.00). U.S. Environmental Protection Agency, Las Vegas, NV.

Christensen, D. L., Herwig, B. R., Schindler, D. E., and Carpenter, S. R. 1996. Impacts of lakeshore residential development on coarse woody debris in north temperate lakes. *Ecological Applications*. 6:1143-1149.

Dionne, M., and Karr, J. R. 1992. Ecological monitoring of fish assemblages in Tennessee River reservoirs. in *Ecological Indicators*. McKenzie, D. H., Hyatt, D. E., and McDonald, V. J.. Eds., Elsevier Applied Science. London. 259-281.

Fore, L. S., Karr, J. R., and Wisseman, R. W. 1996. Assessing invertebrate responses to human activities: evaluating alternative approaches. *Journal of the North American Benthological Society*. 15:212-231.

Gauch, H. G. Jr. 1982. *Multivariate Analysis in Community Ecology*. Cambridge University Press, New York, NY.

Halliwell, D. B. 1989. A classification of streams in Massachusetts - to be used as a fisheries management tool. Ph.D. Dissertation. University of Massachusetts. Amherst, MA.

Halliwell, D. B., Langdon, R. W., Daniels, R. A., Kurtenbach, J. P., and Jacobson, R. A. (this book). Classification of freshwater fishes of the Northeastern United States for use in the development of indices of biological integrity. in *Assessing the Sustainability and Biological Integrity of Water Resource Quality Using Fish* Assemblages. Simon, T. P. Ed., CRC Press. Boca Raton, FL.

Hubbs, C. L., and Cooper, G. P. 1936. *Minnows of Michigan*. Bulletin of the Cranbrook Institute of Science. No. 8. Bloomfield Hills, MI.

Hubbs, C. L., and Lagler, K. F. 1964. *Fishes of the Great Lakes Region.* Bulletin of the Cranbrook Institute of Science. No. 26. Bloomfield Hills, MI.

Jennings, M. J., Fore, L. S., and Karr, J. R. 1995. Biological monitoring of fish assemblages in Tennessee Valley reservoirs. *Regulated Rivers: Research and Management.* 11:263-274.

Jennings, M. J., Bozek, M., Hatzenbeler, G., and Fago, D. 1996. Fish and habitat. in *Shoreline Protection Study: A Report to the Wisconsin State Legislature.* PUBL-RS-921-96. Wisconsin Department of Natural Resources. Madison, WI. pp 1-48.

Jennings, M. J., & 7 others. (this book). Toward the development of an index of biotic integrity for inland lakes in Wisconsin. in *Assessing the Sustainability and Biological Integrity of Water Resource Quality Using Fish Assemblages*. Simon, T. P. Ed., CRC Press. Boca Raton, FL.

Karr, J. R. 1981. Assessment of biotic integrity using fish communities. *Fisheries* 6(6):21-27.

Karr, J. R. 1991. Biological integrity: a long-neglected aspect of water resource management. *Ecological Applications*. 1:66-84.

Karr, J. R., Fausch, K. D., Angermeier, P. L., Yant, P. R., and Schlosser, I. J. 1986. Assessing Biological Integrity in Running Waters A Method and Its Rationale. *Special Publication No. 5. Illinois Natural History Survey*. Champaign, IL.

Larsen, D. P., and Christie, S. J. Eds. 1993. *EMAP-Surface Waters 1991 Pilot Report.* EPA/620/R-93/0 U.S. Environmental Protection Agency. Corvallis, OR.

Larsen, D. P., Stevens, D. L., Selle, A. R., and Paulsen, S. G. 1991. Environmental Monitoring and Assessment Program, EMAP-Surface Waters: A northeast lakes pilot. *Lake and Reservoir Management.* 7:1-11.

Larsen, D. P., Thornton, K. W., Urquhart, N. S., and Paulsen, S. G. 1994. The Role of sample surveys for monitoring the condition of the Nation's lakes. *Environmental Monitoring and Assessment.* 32:101-134.

Lee, D. S., Gilbert, C. R., Hocutt, C. H., Jenkins, R. E., McAllister, D. E., and Stauffer, J. R., Jr. 1980. *Atlas of North American Fishes.* North Carolina Biological Survey. Raleigh, NC.

Leonard, P. M., and Orth, D. J. 1986. Application and testing of an index of biotic integrity in small, coolwater streams. *Transactions of the American Fisheries Society.* 115:401-415.

Lyons, J. 1992. *Using the Index of Biotic Integrity (IBI) to Measure Environmental Quality in Warmwater Streams of Wisconsin.* General Technical Report NC-149. U.S. Department of Agriculture, Forest Service, North Central Experimental Station.

Lyons, J., Wang, L., and Simonson, T. D. 1996. Development and validation of an index of biotic integrity for coldwater streams in Wisconsin. *North American Journal of Fisheries Management.* 16:241-256.

Miller, D. L., & 13 others. 1988. Regional application of an index of biotic integrity for management. *Fisheries*. 13(5):12-20.

Minns, C. K., Cairns, V. W., Randall, R. G., and Moore, J. E. 1994. An index of biotic ir assemblages in littoral zone of Great Lakes' areas of concern. *Canadian Journal of Fish Sciences*. 51:1804-1822.

Moyle, P. B., and Light, T. 1996. Biological invasions of fresh water: empirical rules and *Biological Conservation*. 78:149-161.

New Hampshire Fish & Game Commission (NHFGC). 1937-1938. *Biological Surveys* -- : Concord, NH.

New York State Department of Conservation (NYSDC). 1927-1939. *Biological Surveys* -- Albany, NY.

Ohio Environmental Protection Agency (OEPA). 1988. *Biological Criteria for the protectic life: Volume I. The role of biological data in water quality assessment*. OEPA Division of \ Monitoring and Assessment. Columbus, OH.

Ohio Environment Protection Agency (OEPA). 1989. *Biological Criteria for the Protection c Volume III. Standardized Biological Field Sampling and Laboratory Methods for Assessing Macroinvertebrate Communities*. OEPA Division of Water Quality Planning and Assessment. OH.

Omernik, J. M. 1987. Ecoregions of the conterminous United States. *Annals of the Association of Geographers*. 77: 118-125.

Page, B. M. and Burr, L. M. 1991. *A Field Guide to Freshwater Fishes*. Houghton Mifflin. Boston

Plafkin, J. L., Barbour, M. T., Porter, K. D., Gross, S. K., and Hughes, R. M. 1989. *Rapid bioasses protocols for use in streams and rivers: benthic macroinvertebrates and fish.* U.S. Environmental Protection Agency. EPA/444/4-89-001. Washington, DC.

SAS Institute. 1985. *SAS User's Guide: Statistics, Version 6*. Cary, NC.

Schmidt, R. E. 1986. Zoogeography of the Northern Appalachians. In. *The Zoogeography of North American Freshwater Fishes*. Hocutt, C. H., and Wiley, E. O. Eds. John Wiley & Sons. New York. p 159.,

Scott, W. B., and Crossman, E. J. 1973. *Freshwater Fishes of Canada*. Bulletin of the Fisheries Rese Board of Canada. 184.

Simon, T. P., and Lyons, J. 1995. Application of the Index of Biotic Integrity to Evaluate Water-Resour Integrity in Freshwater Ecosystems. in *Biological Assessment and Criteria: Tools for Water Resource Planning and Decision Making*. Davis, W.S. and Simon, T. P. Eds., Lewis Publishers. Boca Raton, FL. 245-262.

Smith, L. C. 1985. *The Inland Fishes of New York State*. Department of Environ
Albany, NY.

Steedman, R. J. 1988. Modification and assessment of an index of biotic integrity t
in southern Ontario. *Canadian Journal of Fisheries and Aquatic Science* 45:492-

Stevens, D. L., Jr. 1994. Implementation of a national monitoring program. *Journal
Management*. 42:1-29.

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

Underhill, J. C. 1986. The Fish Fauna of the Laurentian Great Lakes, the St. Lawrence
Newfoundland and Labrador. in *The Zoogeography of North American Freshwater F*
and Wiley, E. O. Eds., John Wiley & Sons. New York. pp. 105-136.

U.S. Environmental Protection Agency (USEPA), Office of Information Resources Ma
Block/Precinct Level Coverage User Documentation. Washington, DC.

U.S. Geographic Survey (USGS) 1990. *Land Use and Land Cover Digital Data from 1
1:100,000-scale Maps. Data Users Guide 4*, Reston, VA.

Weaver, M. J., Magnuson, J. J., and Clayton, M. D. 1993. Analyses for differentiating lit
assemblages with catch data from multiple sampling gears. *Transactions of the America*
122:1111-1119.

Whittier, T. R., and Hughes, R. M. (In review). Evaluation of fish species tolerances to environmental stressors in Northeast USA lakes.

Whittier, T. R., and Rankin, E. T. 1992. Regional patterns in three biological indicators of stream condition in Ohio. in *Ecological Indicators*. McKenzie, D. H., Hyatt, D. E., and McDonald, V. J. Eds., Elsevier Applied Science. London. pp 975-995

Whittier, T. R., Halliwell, D. B., and Paulsen, S. G . 1997a. Cyprinid distributions in Northeast USA lakes: evidence of regional-scale minnow biodiversity losses. *Canadian Journal of Fisheries and Aquatic Sciences* 54:1593-1607.

Whittier, T. R., Vaux, P., Merritt, G. D., and Yeardley, R. B., Jr. 1997b. Fish Sampling. in *Environmental Monitoring and Assessment Program--Surface Waters: Field Operations Manual for Lakes*. Baker, J. R., Peck, D. V., and Sutton, D. W. Eds., EPA/620/R-97/001. U.S. Environmental Protection Agency, Washington, DC.

Table 1. Principal components analysis of watershed-level human influence measures: three lan variables (% forest, % urban, % agricultural), human population density(#/km^2), and road densi variables except % forest were $\log_{10}(x+1)$ transformed. The first Principal Component (PCA-1 generalized human influence.

Principal Component	Eigenvalue	Proportion of Variance	Cı
PC-1	3.09	0.62	
PC-2	0.88	0.17	

Eigenvectors

	PCA-1	PCA-2
% Forests	-0.42	0.34
% Urban	0.49	0.31
% Agricultural	0.32	-0.81
Population Density	0.52	0.21
Road density	0.46	0.30

Table 2. Fish species native status, tolerance ranks and trophic guild as used in this study, collected at 55 southern New England lakes by EMAP during summers 1991-94. Tolerance rating scores from Whittier & Hughes (in review) range 4-16, I=Intolerant, MI=Moderately Intolerant, M=Moderate, MT=Moderately Tolerant, T=Tolerant. For Trophic Guild: TC=Top Carnivore, GF=Generalist Feeder, IN=Insectivore.

Species	# of Lakes as Natives	# of Lakes as Introduced or Stocked	Tolerance Rating	Trophic Guild
American eel *Anguilla rostrata*	8		9/M	TC
alewife *Alosa pseudoharengus*	1	1	10/M	
gizzard shad *Dorosoma cepedianum*		1	--/M	
common carp *Cyprinus carpio*		3	5/T	GF
common shiner *Luxilus cornutus*	1		12/M	IN
golden shiner *Notemigonus crysoleucas*	37	2	5/T	GF
bridle shiner *Notropis bifrenatus*	2		13/MI	IN
spottail shiner *N.* hudsonius	1		14/MI	IN
fathead minnow *Pimephales promelas*		1	15/I	GF
creek chub *Semotilus atromaculatus*	2		12/M	IN
fallfish *S. corporalis*	5		11/M	IN

Table 2. (continued)

Species	# of Lakes as Natives	# of Lakes as Introduced or Stocked	Tolerance Rating	Tropl Guil
white sucker *Catostomus commersoni*	23		8/MT	GF
creek chubsucker *Erimyzon oblongus*	1		9/M	IN
white catfish *Ameiurus catus*		2	--/M	
yellow bullhead *A. natalis*		20	6/MT	GF
brown bullhead *A. nebulosus*	42		5/T	GF
channel catfish *Ictalurus punctatus*		2	--/M	
tadpole madtom *Noturus gyrinus*		1	--/MI	IN
northern pike *Esox lucius*		3	10/M	TC
chain pickerel *E. niger*	34		7/M	TC
rainbow smelt *Osmerus mordax*		2	11/M	GF
rainbow trout *Oncorhynchus mykiss*		7	--/MI	
Atlantic salmon *Salmo salar*		1	15/I	TC*
brown trout *S. trutta*		3	--/M	

Table 2. (continued)

Species	# of Lakes as Natives	# of Lakes as Introduced or Stocked	Tolerance Rating	Trophic Guild
brook trout *Salvelinus fontinalis*		6	13/MI	
lake trout *S. namaycush*		2	14/MI	TC*
banded killifish *Fundulus diaphanus*	11		10/M	IN
mummichog *F. heteroclitus*		1	--/T	GF
white perch *Morone americana*	4	8	5/T	
rock bass *Ambloplites rupestris*		2	12/M	TC
banded sunfish *Enneacanthus obesus*	6		9/M	IN
redbreast sunfish *Lepomis auritus*	5		10/M	IN
pumpkinseed *L. gibbosus*	50		4/T	GF
bluegill *L. macrochirus*		35	4/T	GF
smallmouth bass *Micropterus dolomieu*		15	10/M	TC
largemouth bass *M. salmoides*		50	5/T	
black crappie *Pomoxis nigromaculatus*		21	4/T	

Table 2. (continued)

Species	# of Lakes as Natives	# of Lakes as Introduced or Stocked	Tolerance Rating	Trophic Guild
swamp darter *Ethesotoma fusiforme*	3		--/M	IN
tesselated darter *E. olmstedi*	2		11/M	IN
yellow perch *Perca flavescens*	44		6/MT	IN

Figure Captions

Figure 1. Locations of the 55 lakes sampled by the Environmental Monitoring and Assessment Program (EMAP) in southern New England during the summers of 1991-94.

Figure 2. Relationship between sampled southern New England lake surface areas and two stressor measures: total phosphorus (ug/L), and watershed disturbance (1st axis PCA scores for % forest, % urban, % agricultural landuses, road density and human population density), plotted by ecoregions (a & c: solid circles = Northeast Highlands ecoregion, open circles = Northeast Coastal Zone ecoregion), and lake type (b & d: solid circles = named Reservoirs, open circles = named Ponds or Lakes).

Figure 3. Native Species Richness. (a) Number of native species by lake size. (b) Residuals of the native species by lake size regression plotted against total phosphorus. c) Residuals plotted against watershed disturbance PCA axis 1 (higher axis 1 value = increased human activity in the watershed).

Figure 4. Non-Native Species Richness. (a) Number of non-native species by lake size. (b) Residuals of the non-native species by lake size regression plotted against total phosphorus. c) Residuals plotted against watershed disturbance PCA axis 1 (higher axis 1 value = increased human activity in the watershed).

Figure 5. Native Large Species Richness. (a) Number of native large species by lake size. (b) Residuals of the native large species by lake size regression plotted against Total Phosphorus. c) Residuals plotted against watershed disturbance PCA axis 1 (higher axis 1 value = increased human activity in the watershed).

Figure 6. Percent of individuals of non-native species. (a) % non-native individuals b % non-native individuals by watershed disturbance PÇA axis 1 (higher axis 1 value = i activity in the watershed).

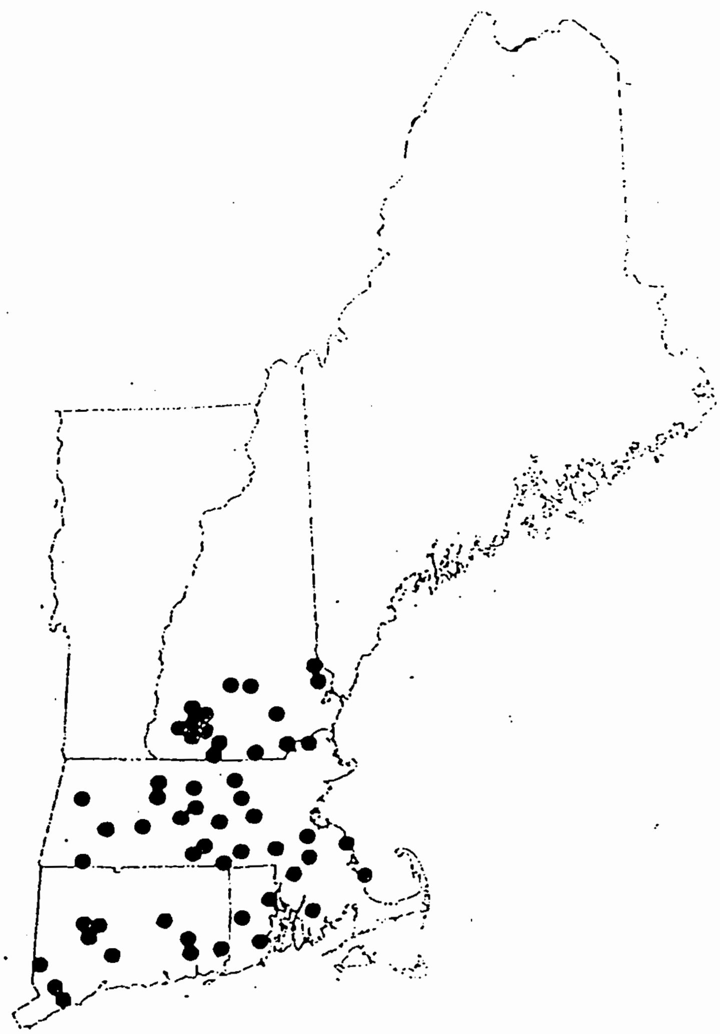
Figure 7. Percent of individuals of tolerant species (a) % tolerant individuals by lake siz pumpkinseed. (b) % tolerant individuals by total phosphorus, including pumpkinseed. c individuals by watershed disturbance including pumpkinseed. d) % tolerant individuals l excluding pumpkinseed. e) % tolerant individuals by watershed disturbance excluding pu

Figure 8. Percent of individuals of top carnivore species. (a) % top carnivore individuals l excluding largemouth bass. (b) % top carnivore individuals by watershed disturbance excl bass. c) % top carnivore individuals by total phosphorus, including largemouth bass. (d) % individuals by watershed disturbance including largemouth bass.

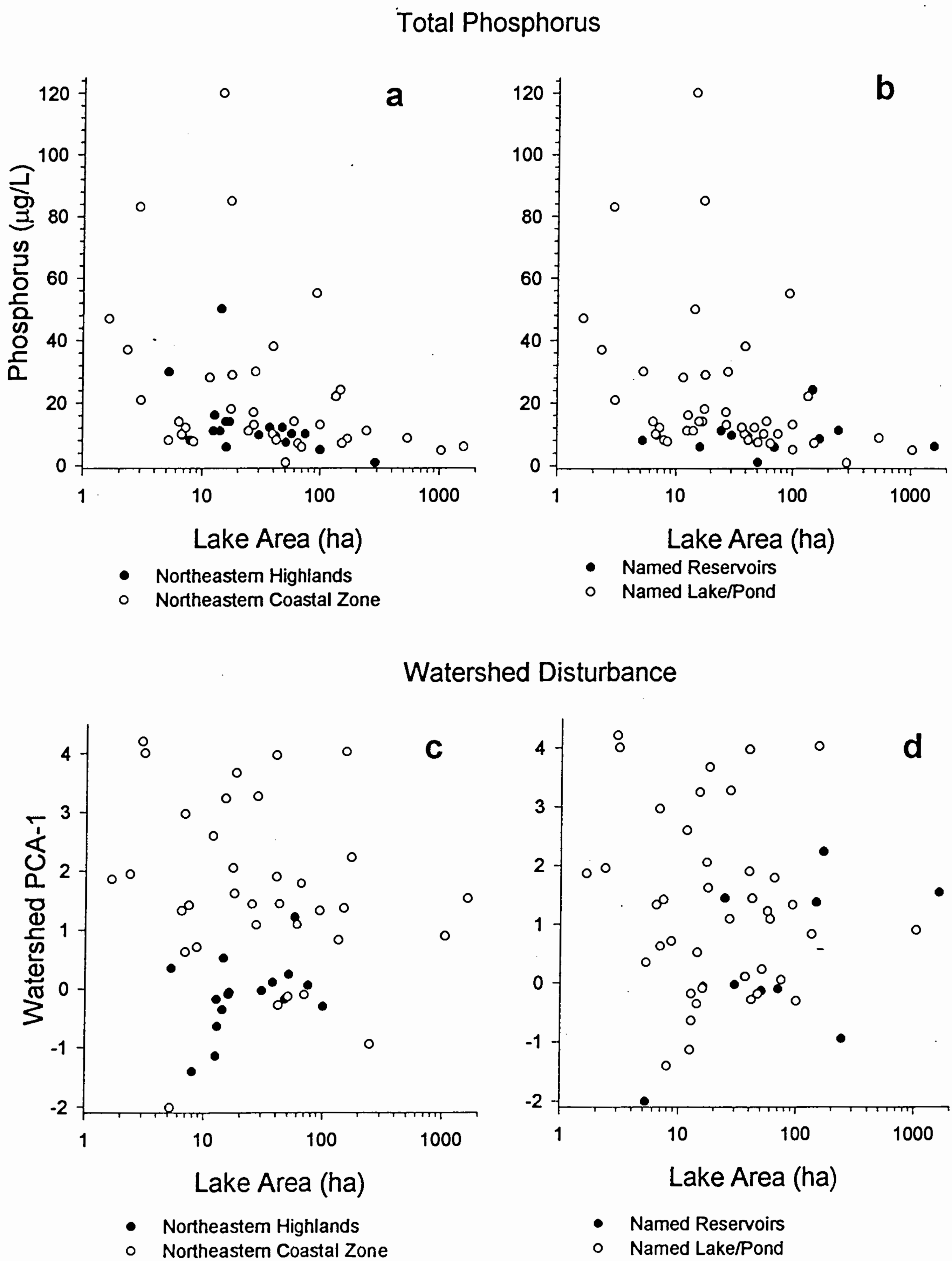
Figure 9. Percent of individuals of insectivore species. (a) % insectivore individuals by lak insectivore individuals by total phosphorus, c) % insectivore individuals by watershed dist 1 (higher axis 1 value = increased human activity in the watershed).

Figure 10. Percent of individuals of generalist feeder species. (a) % generalist feeder individ size, (b) % generalist feeder individuals by total phosphorus, c) % generalist feeder individu disturbance PCA axis 1 (higher axis 1 value = increased human activity in the watershed).

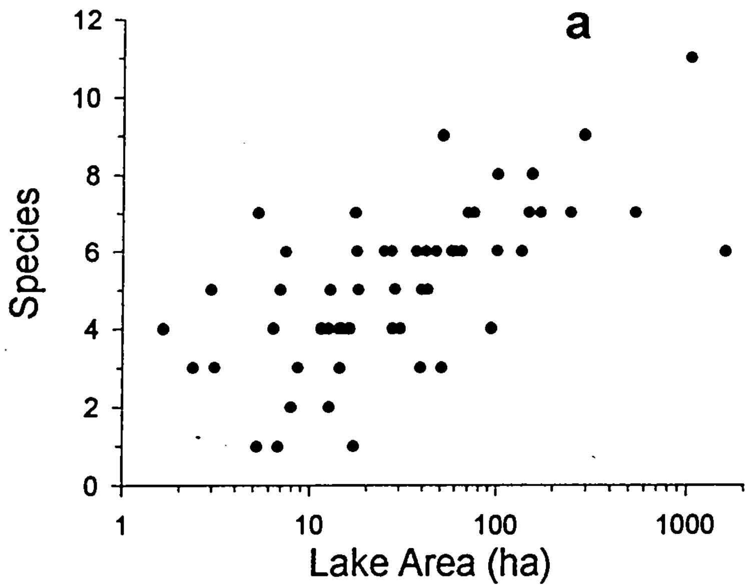
Figure 1



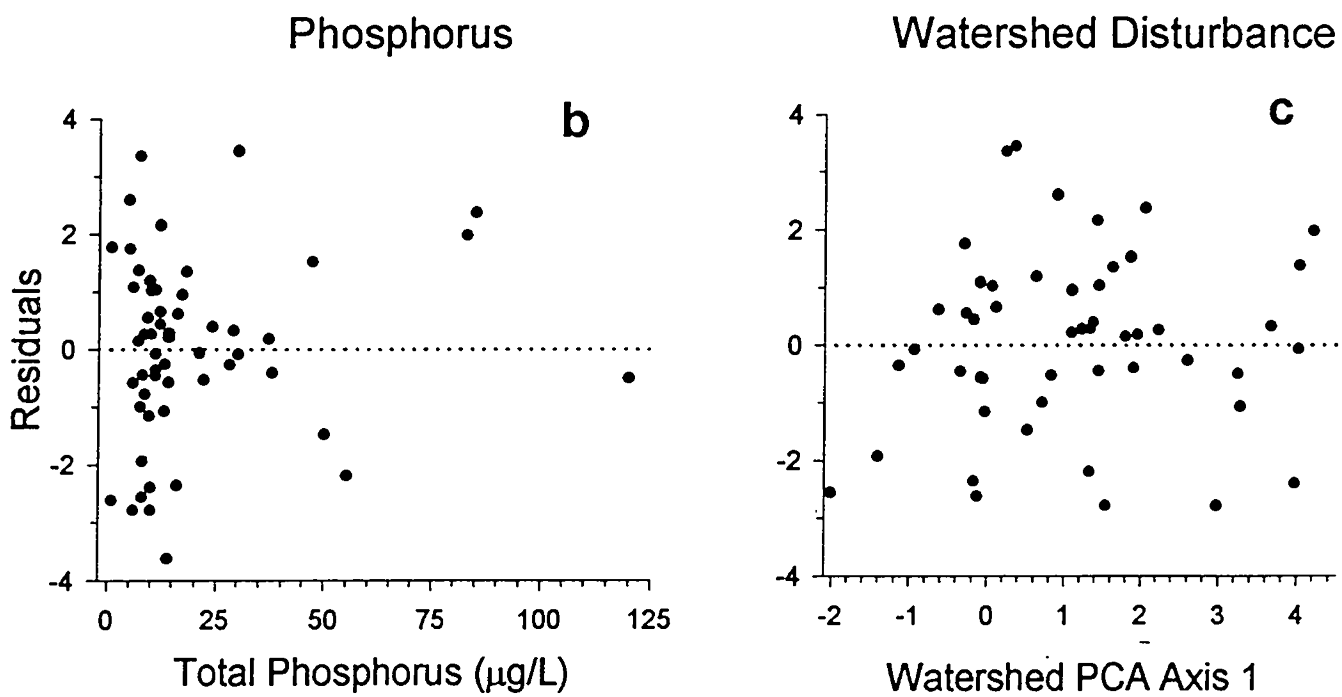
Southern New England Lakes



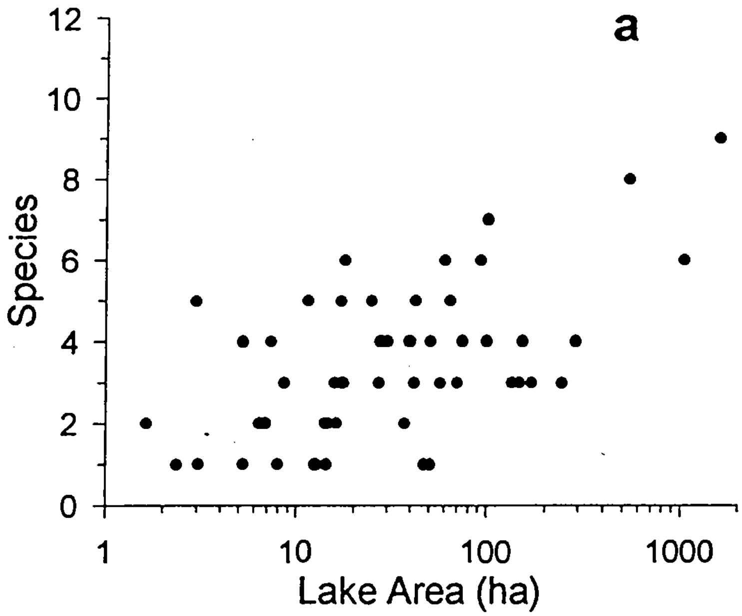
Native Species



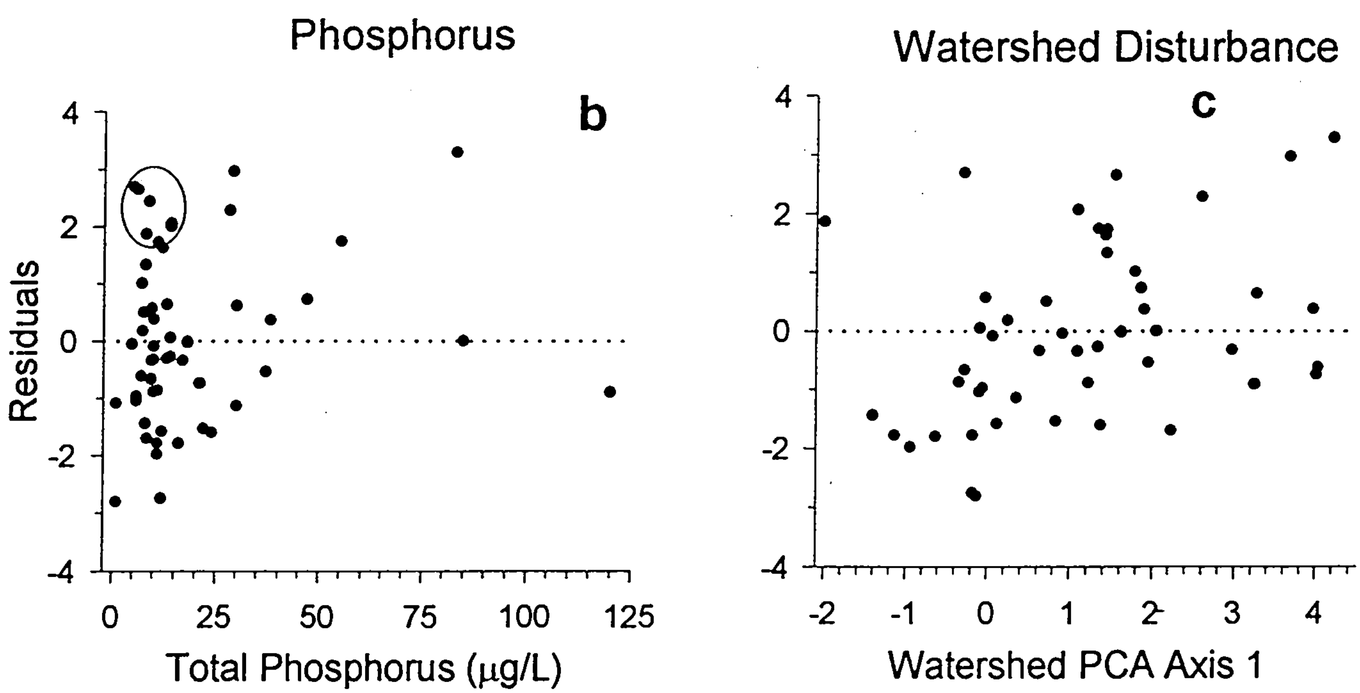
Species	Lakes
Pumpkinseed	50
Yellow Perch	44
Brown Bullhead	42
Golden Shiner	37
Chain Pickerel	34
White Sucker	23
Banded Killifish	11
American Eel	8
Banded Sunfish	6
+ 11 Others	

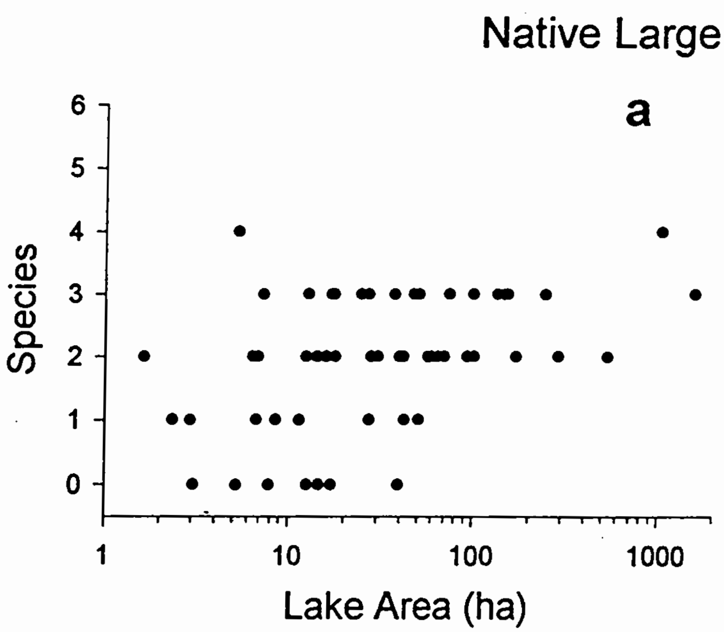


Non-Native Species

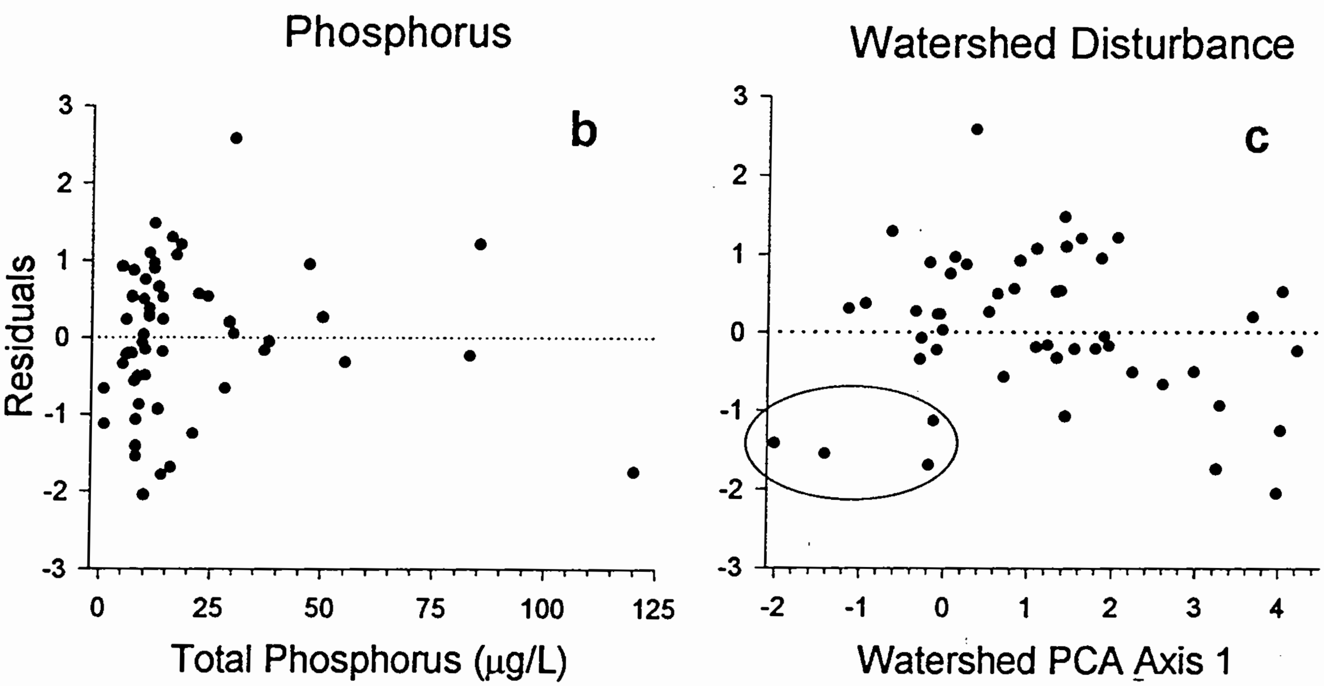


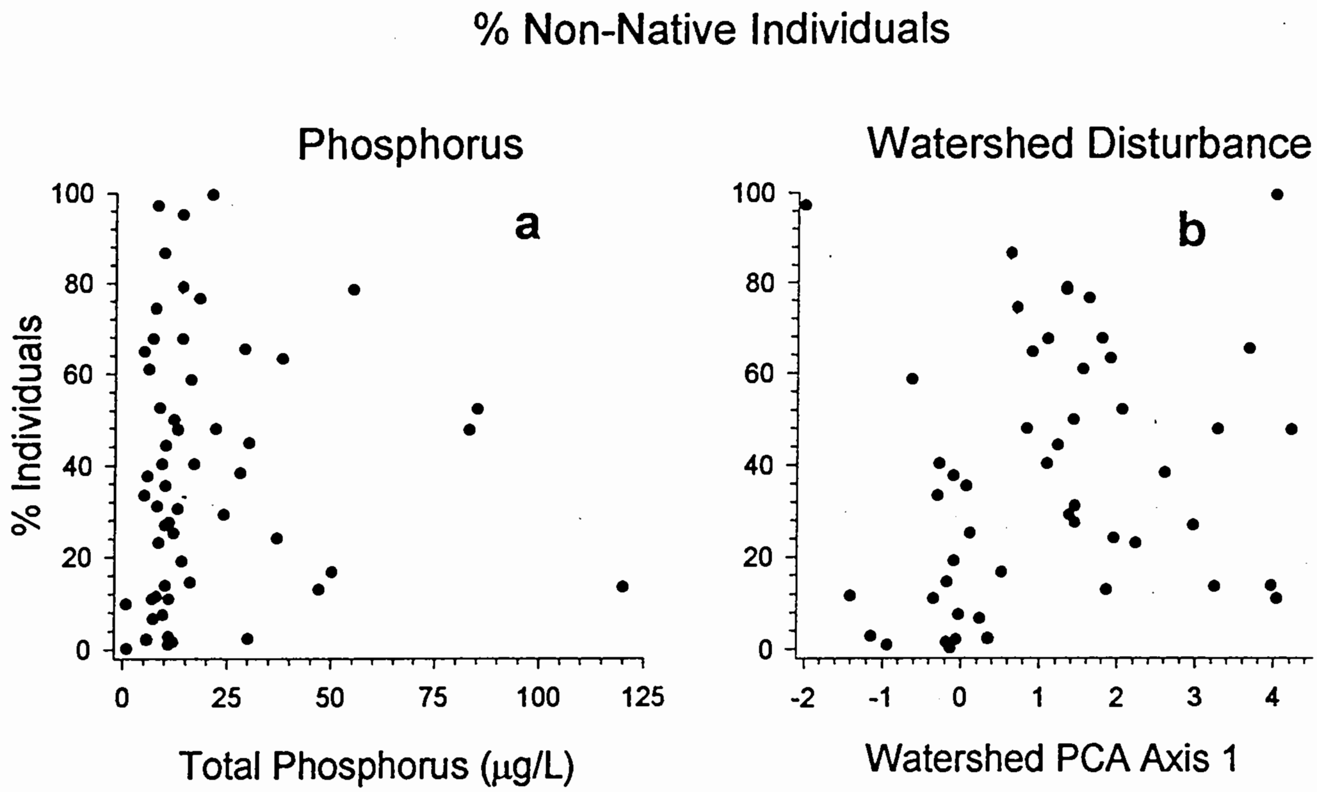
Species	Lakes
Largemouth Bass	50
Bluegill	35
Black Crappie	21
Yellow Bullhead	20
Smallmouth Bass	15
Rainbow Trout	7
Brook Trout*	6
Common Carp	3
Northern Pike	3
Brown Trout	3
+ 10 Others	

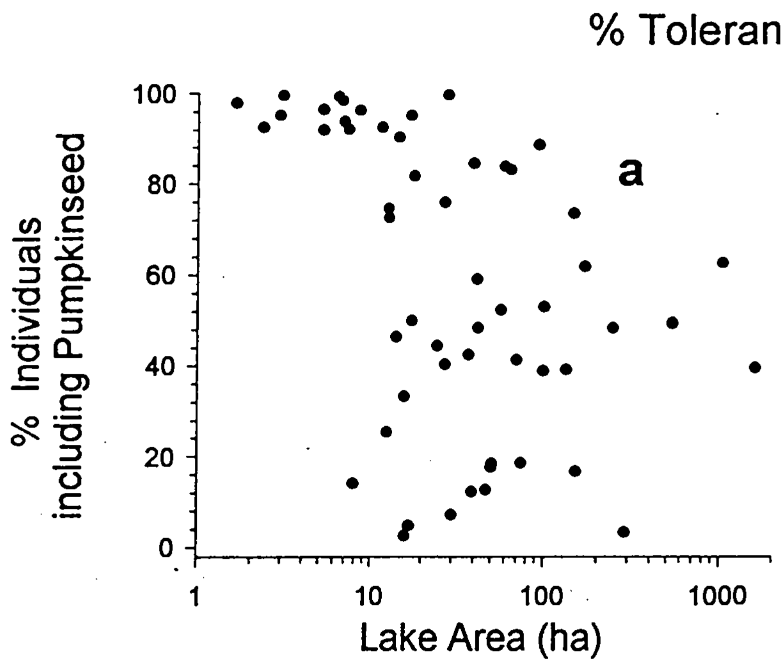




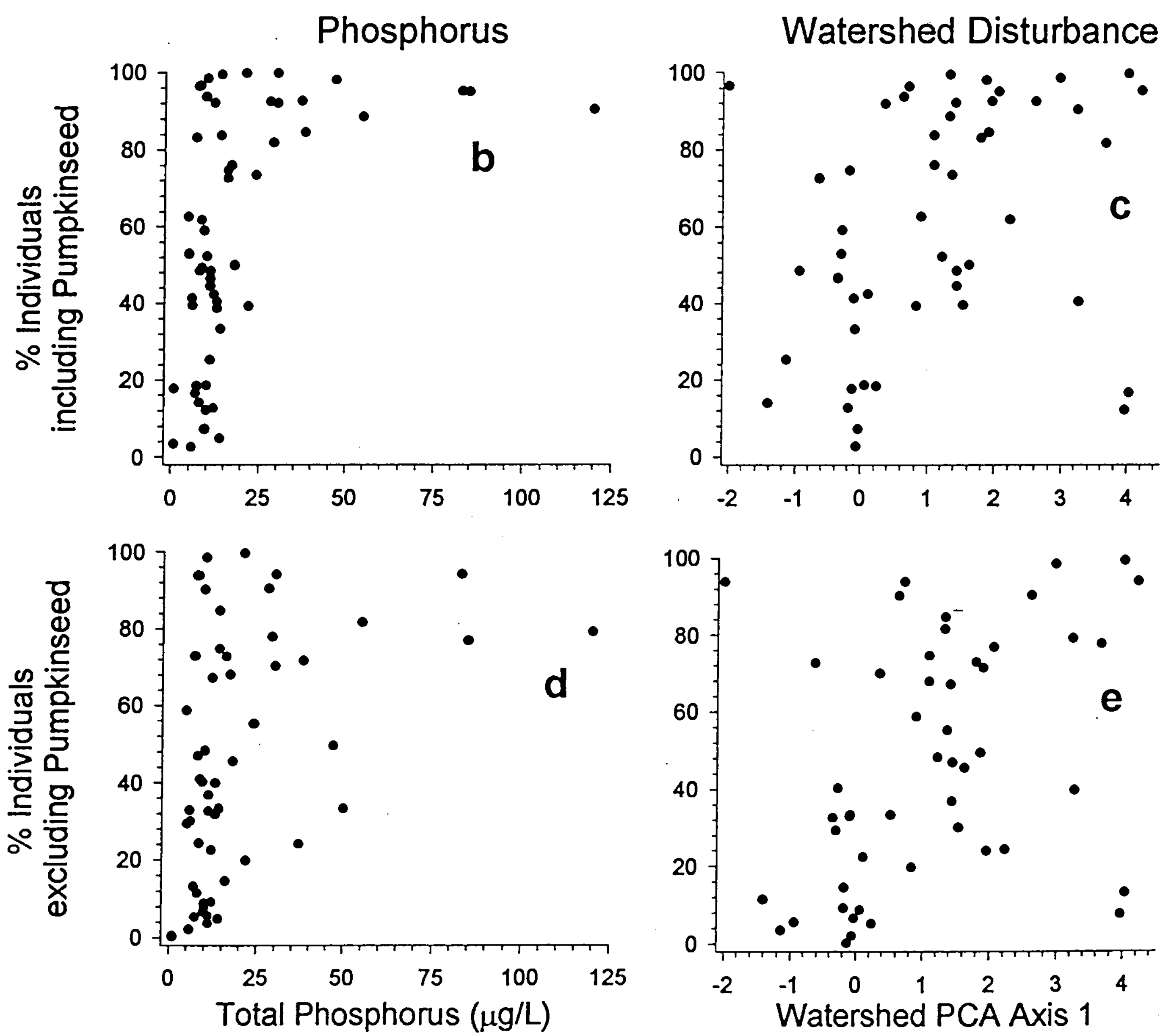
Species	Lakes
Brown Bullhead	42
Chain Pickerel	34
White Sucker	23
American Eel	8
Creek Chubsucker	1



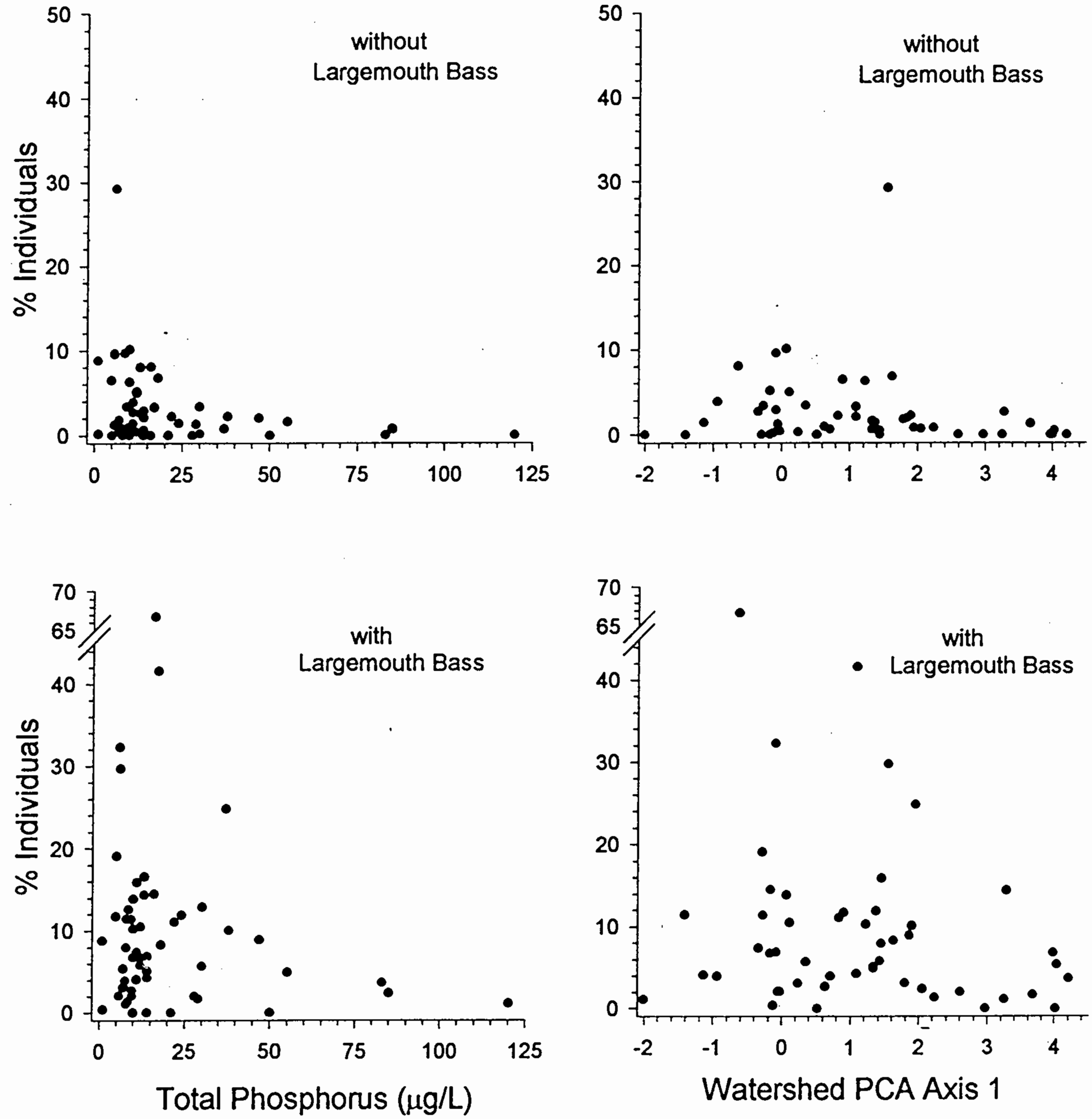

% Non-Native Individuals
Phosphorus
Watershed Disturbance
a
b
% Individuals
100
80
60
40
20
0
0
25
50
75
100
125
Total Phosphorus (μg/L)
-2
-1
0
1
2
3
4
Watershed PCA Axis 1

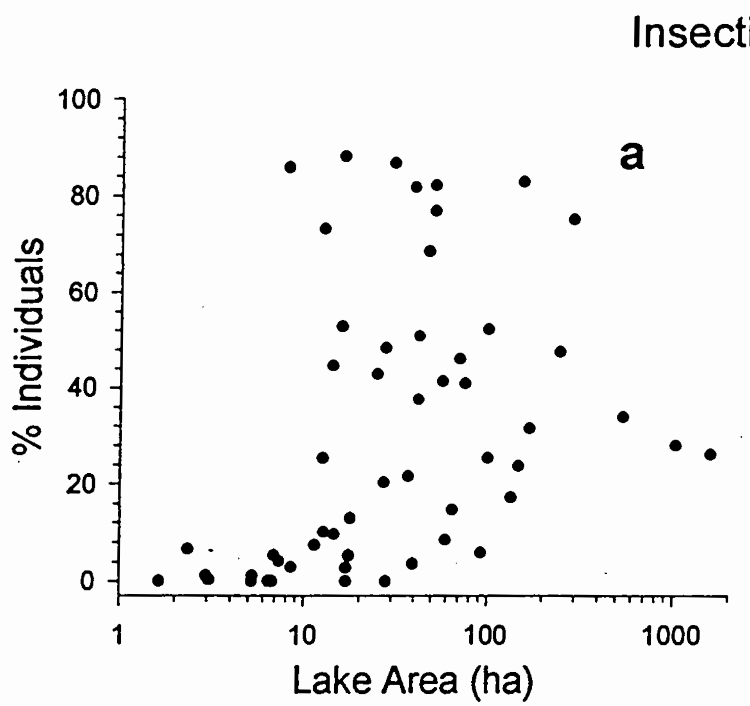


Species	Lakes
Largemouth Bass	50
Pumpkinseed	50
Brown Bullhead	42
Golden Shiner	39
Bluegill	35
Black Crappie	21
White Perch	12
Common Carp	3



Top Carnivores





Species	Lakes
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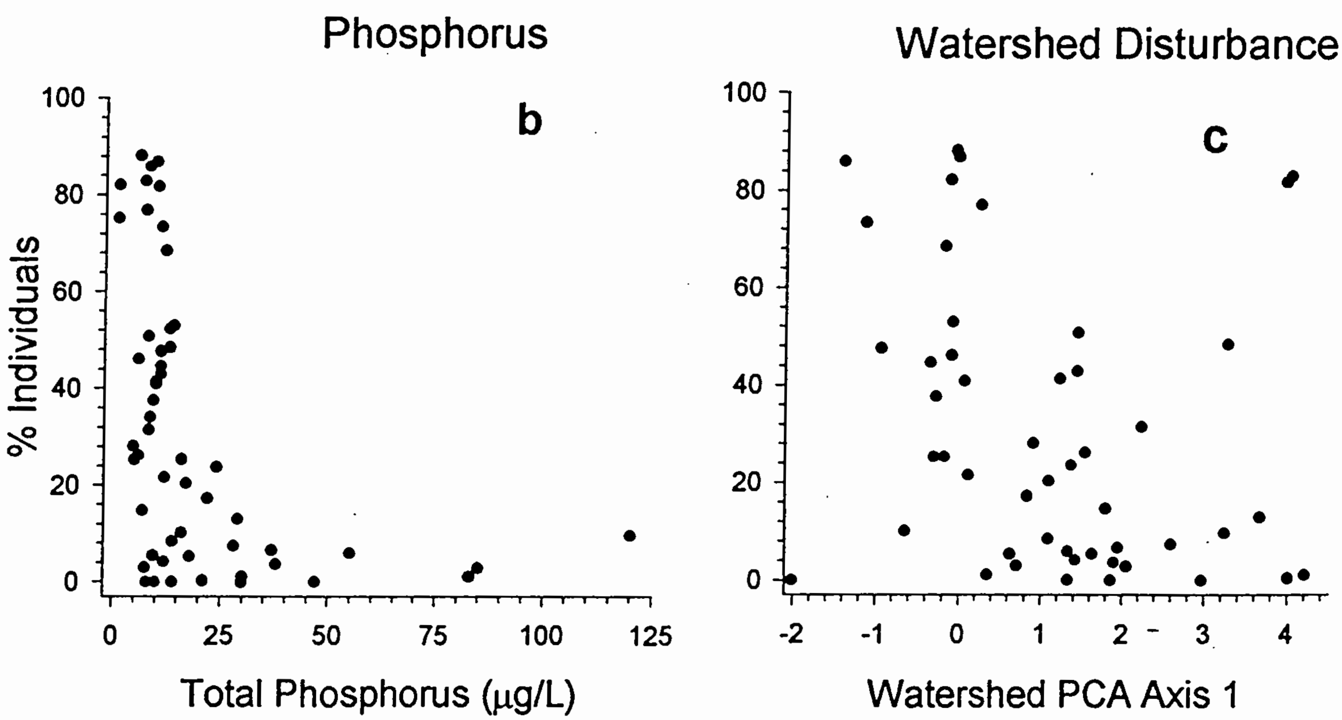
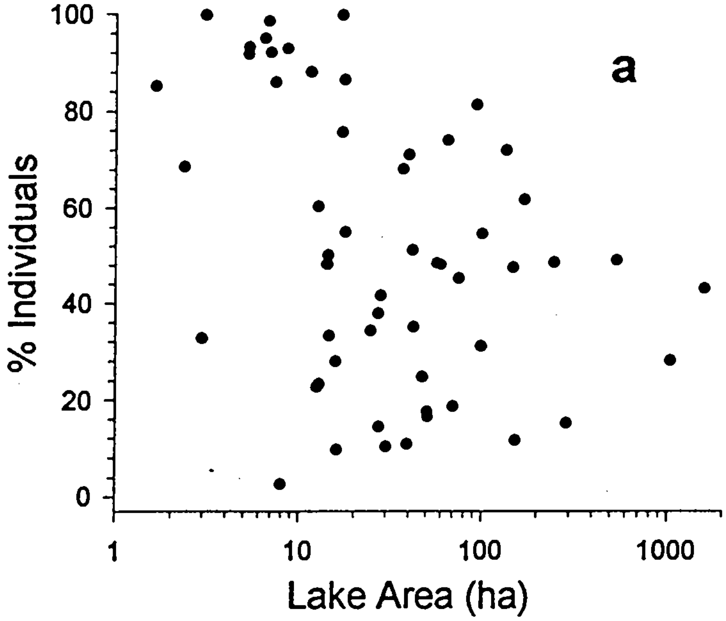
Redbreast Sunfish	5
Fallfish	5
Swamp Darter	3
Tesselated Darter	2
Bridle Shiner	2
Creek Chub	2
+ 4 Others	



Generalist Feeders



Species	Lakes
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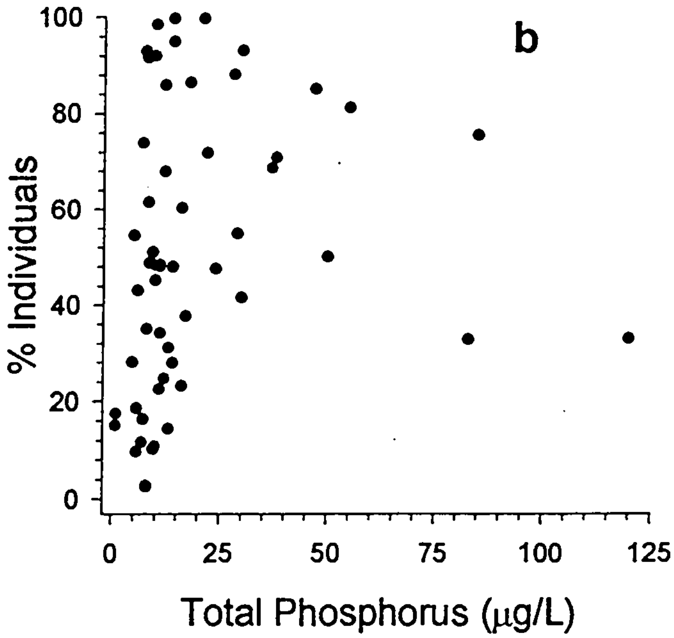
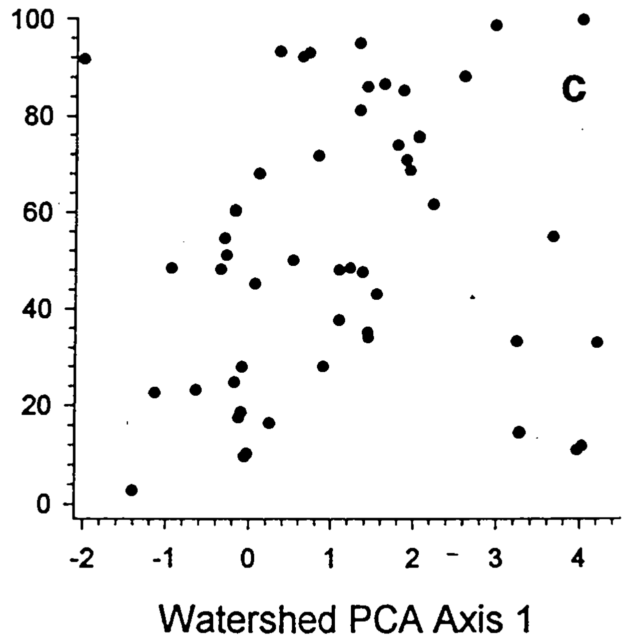
Golden Shiner	39
Bluegill	35
White Sucker	23
Yellow Bullhead	20
Common Carp	3
Rainbow Smelt	2
Mummichog	1
Fathead Minnow	1

Phosphorus



Watershed Disturbance



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TECHNICAL REPORT DATA *(Please read instructions on the reverse before co*		
1. REPORT NO. EPA/600/A-97/089	2.	:
4. TITLE AND SUBTITLE Development of IBI metrics for lakes in southern New England		5. REPORT DATE
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9. PERFORMING ORGANIZATION NAME AND ADDRESS Dynamac International, Inc. US EPA NHEERL 200 SW 35th Street Corvallis, OR 97333		10. PROGRAM ELEMENT NO.
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12. SPONSORING AGENCY NAME AND ADDRESS US EPA ENVIRONMENTAL RESEARCH LABORATORY 200 SW 35th Street Corvallis, OR 97333		13. TYPE OF REPORT AND PERIOD COVERED
		14. SPONSORING AGENCY CODE EPA/600/02
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16. Abstract:

In the early 1980's J.R. Karr and his colleagues developed the Index of Biotic Integrity (IBI) to assess the ecological condition of fish assemblages in warmwater streams in the Midwest, as a biological tool to further the goals of the Clean Water Act. The IBI has been repeatedly modified for other assemblages, regions and ecosystems, with varying degrees of success. However to date, there has been only limited effort toward developing IBIs for inland lakes. The development of lake IBIs lags behind that for warmwater streams due in part to the increased complexity in the scope of assessment and sampling issues in lakes. In addition, streams tend to be managed, or under the authority of state departments of environmental quality which are concerned with issues defined by the Clean Water Act. These agencies have been shifting their regulatory emphasis to the biological condition of streams. Lakes, on the other hand tend to be managed by state fish and game departments whose mandates emphasize management for fishability, not biotic integrity. Finally, a number of ecological factors are less well understood for lake fish assemblages.

This manuscript evaluates the performance of numerous candidate metrics for lake fish assemblages in southern New England, comparing metric behavior with total phosphorus (as a measure of eutrophication stress) and a generalized measure of human activity in the watersheds (% of land in urban and agriculture, road and population density). This manuscript presents solutions to issues of species tolerances to stressors in lakes and to trophic guild membership. These modifications improved the performance of four candidate metrics (% individuals of highly tolerant species, % top carnivore individuals, % insectivorous individuals, % generalist feeder individuals). A total of seven metrics appear to be useful for evaluating biotic integrity of lakes in southern New England.

17. KEY WORDS AND DOCUMENT ANALYSIS		
a. DESCRIPTORS	b. IDENTIFIERS/OPEN ENDED TERMS	c. COSATI Field/Group
New England, lakes, metric development, index of biotic integrity		
18. DISTRIBUTION STATEMENT	19. SECURITY CLASS *(This Report)*	21. NO. OF PAGES: 43
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