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Gene R. Hatzenbeler , Jeffrey M. Kampa , Martin J. Jennings & Edward E. Emmons

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Gene R. Hatzenbeler, Jeffrey M. Kampa, Martin J. Jennings
*Wisconsin Department of Natural Resources*
*Bureau of Integrated Science Services*
*810 W. Maple St.*
*Spooner, WI 54801*

Edward E. Emmons
*Wisconsin Department of Natural Resources*
*Bureau of Integrated Science Services*
*1350 Femrite Dr.*
*Monona, WI 53716*

## ABSTRACT

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Biological monitoring uses assemblage structure to assess condition of ecological systems. Taxa that effectively integrate impacts within the system of interest are useful for biological monitoring, whereas taxa that do not demonstrate predictable responses can provide ambiguous or misleading indicators. We compared the effectiveness of aquatic plant and fish assemblages for biological monitoring in 16 small lakes (< 80 ha). The lakes were limnologically similar but differed in extent of lakeshore development and type of watershed land-cover. Linear regression analysis revealed that the quality of the aquatic plant community declined with increasing lakeshore development (number of dwellings per km of shoreline), which is the primary source of impacts within this group of northern Wisconsin lakes. As lakeshore development increased, we observed a decrease in the Floristic Quality Index (FQI) of a lake, number of plant species per lake, number of highly intolerant plant species per lake, and the species richness and frequency of occurrence of floating vegetation. Conversely, fish species richness, centrarchid species richness, number of small benthic fish species, intolerant fish species richness and the proportion of the total catch of intolerant and vegetative-dwelling fish were not related to lakeshore development. These results indicate that, within the range of conditions observed, aquatic plant communities are more sensitive to lakeshore development than fish communities. Neither aquatic plant species composition nor fish assemblage variables were correlated with watershed land cover types; however all the watersheds were relatively small and undisturbed. In small lakes with few fish species, aquatic plants can be used as biological indicators for monitoring ecological conditions.

Key Words: monitoring, lakes, fish, aquatic plants, development, land-cover, FQI, IBI.

Lakeshores of inland lakes in North America are being rapidly developed for residential and recreational activities. Radomski and Goeman (2001) recently reported a rate of residential development on Minnesota's clear-water centrarchid-walleye lakes that is six times the level seen during the 1950s. Similar trends have been observed elsewhere, including Wisconsin (WDNR 1998). Effective management of inland lake resources requires an understanding of biological responses to these changes, and the ability to assess current condition and trends within lakes and lake districts. A monitoring program based on an appropriate set of indicators (Karr and Chu 1997) can provide this feedback and provide an objective basis for making management decisions.

Successful biological monitoring measures attributes that integrate a wide range of potential impacts, providing an overall summary of ecological condition (Karr and Chu 1997). Rather than focus on individual disturbances, the emphasis is on ecological indicators that can respond to multiple sources of ecological stress. The potential impacts of residential and recreational development on lakes include inputs of nutrients and contaminants through non-point sources (Downing and McCauley 1992, Schindler 2001) and alteration of physical habitat (Christensen et al. 1996, Jennings et al. in press). Activities associated with development, such as motorboating, also impact systems by directly cutting plants and by scouring sediments (Asplund and Cook 1997). These changes separately and in combination might be expected to change biological assemblages. Jennings et al. (1999) found that whole lake basin-scale effects, as indicated by water quality, were associated with differences in fish assemblage composition of Wisconsin lakes, and similar patterns were observed in lakes of the northeastern USA (Whittier and Hughes 1998). Radomski and Goeman (2001) measured abundance of emergent and floating-leaf macrophytes in Minnesota lakes, and found greater abundance adjacent to lightly developed shores than were found adjacent to moderately or heavily developed shores. Thus, either plant or fish assemblages may provide useful information that summarizes the ecological status of lakes where lakeshore development is occurring.

Extensive work has been done to develop and apply biomonitoring approaches for streams and rivers using fish (Karr 1981, Lyons 1992, Lyons et al. 2001), macroinvertebrates (Hilsenhoff 1982), aquatic macrophytes (Carbiener et al. 1990, Robach et al. 1996), and diatoms (Patrick 1973, Lobo et al. 1995). Less has been done to develop biomonitoring approaches for lakes, which differ from streams in several ecologically important attributes. Many of the impacts on streams, such as changes in flow regime, lead to rapid and direct response of the biota, in contrast to lakes, in which the effects of impacts can be more gradual in onset because of lake volume but more persistent, because of high retention time. Natural constraints on species movements vary among lakes depending upon the degree of connectivity to other aquatic systems. In contrast to the longitudinal riffle-pool sequence of streams, lake biota may segregate between littoral and pelagic habitats, which require different sampling approaches. Thus, existing biomonitoring methods developed for streams are not immediately transferable to lakes without additional investigation and validation.

Initial work with fish (Jennings et al. 1998) and aquatic macrophytes (Nichols 1999) as environmental quality indicators have been conducted in Wisconsin lakes. Jennings et al. (1998) evaluated metrics based on the index of biotic integrity (IBI) concept (Karr 1981, Karr and Chu 1997), and found six fish assemblage metrics (numbers of native species, centrarchid species, intolerant species, small benthic species, and proportional abundance of intolerant fish and vegetation-dwelling fish) to demonstrate consistent relationships to status of environmental quality in lakes greater than 80 surface ha. However, these relations have not been validated in small lakes (surface area < 80 ha). The Floristic Quality Index (FQI) is designed to evaluate the resemblance of the flora of an area to that of an undisturbed site (Swink and Wilhelm 1994). The FQI takes into account the species richness (i.e., number of plant species present) and the sensitivity of each individual plant species to environmental conditions such as turbidity, substrate preferences, rooting strength, primary reproduction means, and tolerance to water drawdown (Nichols 1999). Our objective was to compare biological monitoring approaches based on fish and plant assemblages for their effectiveness in integrating impacts related to lakeshore development and watershed land cover in small lakes.

# Methods

## *Lake Selection*

Study lakes were selected to minimize ecoregional and limnological differences that might affect the composition of the fish and aquatic plant assemblages. If lakes are similar in natural characteristics, variance unrelated to human activity is minimized. The 16 lakes were all located in northwestern Wisconsin and belong to the same lake class as defined by objective criteria described in Emmons et al. (1999). All the lakes are characterized by relatively small surface area (surface area range 15.1 to 80 ha), high landscape position (small watershed area), and similar depth (Table 1). The dominant substrates in the lakes were fine sand and organic matter.

## *Aquatic Plant Surveys*

Sixteen aquatic plant surveys were conducted during July and August of 2000 and 2001. Eight lakes were sampled during each year. The surveys were conducted by SCUBA and snorkeling within each lake along 20 randomly selected transects positioned perpendicular to shore. The presence of plant species was recorded within a 30 $cm^2$ quadrat at 3-m intervals along the transect starting at the land-water interface to 30 m offshore.

A FQI score was calculated for each lake by multiplying the mean coefficient of conservatism score of all plant species sampled in the lake by the square root of the number of aquatic macrophyte species per lake. Plants were assigned coefficient of conservatism scores (CS) based on information developed by Nichols (1999). Conservatism is the estimated probability that

Table 1.–Morphometry data and values of independent variables of the 16 northern Wisconsin Lakes used in this study. PAG is agriculture plus grassland.

| Lake | Surface Area (ha) | Maximum Depth (m) | Watershed Area (ha) | Dwelling/ km of shoreline | % Forest cover | % PAG cover |
|---|---|---|---|---|---|---|
| Atkins | 73.3 | 24.4 | 227 | 14.3 | 85.9 | 4.0 |
| Bass | 54.1 | 9.4 | 257 | 12.9 | 88.4 | 0.0 |
| Bass-Patterson | 78.3 | 10.6 | 445 | 8.6 | 72.1 | 10.0 |
| Beartrack | 40.4 | 11.0 | 1931 | 10.9 | 71.9 | 3.0 |
| Cisco | 39.6 | 32.0 | 34 | 6.0 | 97.9 | 0.0 |
| Crystal | 46.3 | 8.8 | 97 | 10.1 | 83.1 | 7.6 |
| Ellison | 45.8 | 5.4 | 324 | 18.2 | 64.5 | 2.0 |
| Island | 24.6 | 15.5 | 302 | 17.1 | 62.0 | 0.0 |
| Kirby | 38.3 | 5.8 | 2297 | 4.2 | 64.4 | 11.3 |
| McLain | 62.5 | 9.1 | 402 | 11.6 | 73.1 | 1.6 |
| Poquette | 40.4 | 7.0 | 261 | 13.5 | 37.2 | 62.1 |
| Tahkodah | 63.3 | 5.5 | 243 | 11.7 | 71.5 | 17.8 |
| Thirty | 30.4 | 8.2 | 2361 | 6.8 | 79.3 | 11.7 |
| Tozer | 15.1 | 14.0 | 973 | 14.6 | 48.9 | 41.2 |
| Ward | 37.9 | 13.1 | 1951 | 14.8 | 51.2 | 34.5 |
| Warner | 73.3 | 22.8 | 516 | 13.6 | 63.6 | 16.2 |

a plant would occur in a relatively undisturbed landscape, and the score is based on the sensitivity of the plant to environmental conditions such as turbidity, substrate preference, rooting strength, primary reproductive means, and tolerance to water drawdown (Nichols 1999). The CS scores range from 1 to 10; from most to least tolerant of changes in their environment (Nichols 1999).

Species richness was calculated with only native taxa. Plant species were assigned to one of the four "tolerance" groups based on their conservatism score: highly intolerant (CS of 9-10), intolerant (CS of 7-8), moderately tolerant (CS of 5-6) and tolerant ((CS of 1-4) Table 2). Plants were also assigned to their structural form of emergent, floating and submersed vegetation type and the frequency of occurrence of each structural form was determined for each lake.

## *Fish Sampling*

Jennings et al. (1998) identified six fish assemblage metrics for potential use in the development of a lake IBI. Four of the 6 fish assemblage metrics were measures of species richness: number of native species, centrarchid species, intolerant species, and small benthic species; the other two fish assemblage metrics were measures of proportional abundance: percentages of total individuals caught represented by intolerant fish and by vegetation-dwelling fish (Table 3). Describing the fish assemblage in lakes requires multiple sampling gear (Weaver et al. 1993). Hence, Jennings et al. (1998) recommended the use of multiple sampling methods to generate metric values. However, the data from the different sampling methods should not be combined to generate a specific metric value, except for the number of native species per lake. Following Jennings et al. (1998) and Jennings et al. (1999) sampling protocols, we set four randomly placed mini-fyke nets (5.6 mm ace mesh) per lake to determine the metric values for the number of centrarchid species per lake and proportion of vegetative dwellers to total catch per lake. Also, 10 randomly placed stations that were 30 linear meters of shoreline were seined (5.6 mm ace bag seine) to depth of 1 meter. Seining was used to determine the metric values for the number of intolerant and small benthic species and proportion of intolerant species to total catch per lake.

## *Lakeshore Development*

Lakeshore development was quantified in 9 lakes by counting the number of dwellings from aerial photographs of the shoreline taken during 1998-1999. For the remaining 7 lakes, the dwellings were counted by an observer cruising the shoreline in a boat. In both cases, dwellings were considered to be part of the

Table 2.–Aquatic plant species represented in tolerance groupings based on their conservatism score (Nichols 1999).

| Highly Intolerant (CS 9-10) | Intolerant (CS 7-8) | Moderately Tolerant (CS 5-6) | Tolerant (CS 1-4) |
|---|---|---|---|
| *Dulichium arundinaceum* | *Acorus calamus* | *Eleocharis acicularis* | *Ceratophyllum demersum* |
| *Elatine minima* | *Bidens beckii* | *E. palustris* | *Elodea canadensis* |
| *Eriocaulon aquaticum* | *Brasenia schreberi* | *Lemnar minor* | *Potamogeton pectinatus* |
| *Gratiola aurea* | *Callitriche palustris* | *Najas flexilis* | *Sagittaria latifolia* |
| *Littorella uniflora* | *Chara spp.* | *Nuphar variegata* | *Scirpus validus* |
| *Lobelia dortmanna* | *Equisetum fluviatile* | *Nymphaea odarata* | *Typha angustifolia* |
| *Myriophyllum farwellii* | *Glyceria borealis* | *Polygonum amphibium* | *T. latifolia* |
| *M. tenellum* | *Isoetes echinospora* | *Potamogeton foliosus* | |
| *Pontederia cordata* | *Juncus pelocarpus* | *P. natans* | |
| *Ranunculus flammula* | *Myriophyllum sibiricum* | *P. zosteriformis* | |
| *Sagittaria graminea* | *Najas gracillima* | *Ranunculus longirostris* | |
| *Scirpus subterminalis* | *Nitella spp.* | *Scirpus acutus* | |
| *Sparganium angustifolium* | *Nuphar advena* | *S. americanus* | |
| *S. fluctuans* | *Potamogeton amplifolius* | *Sparganium eurycarpum* | |
| *Utricularia gibba* | *P. diversifolius* | *Vallisneria americana* | |
| *U. purpurea* | *P. epihydrus* | *Zosterella dubia* | |
| *U. resupinata* | *P. gramineus* | | |
| | *P. nodosus* | | |
| | *P. praelongus* | | |
| | *P. pusillus* | | |
| | *P. robbinsii* | | |
| | *P. spirillus* | | |
| | *Utricularia vulgaris* | | |
| | *Zannichellia palustris* | | |

riparian zone if they were located within 100m of the land/water interface.

## *Watershed Cover-types*

Cover-types within watersheds were quantified with satellite imagery in ARC/INFO. The predominant cover type within 30 $m^2$ blocks were defined as forest, wetland, open water, grassland, agriculture, shrub, and barrens and was expressed as proportion of the watershed. Urban land cover was not concentrated in any of the watersheds. For the final analyses, grass and agriculture were combined because they were not well differentiated in the GIS data set. Based on direct observation on the ground, much of the grass cover type was used for pasture. Though wetland, open water, shrub and barrens cover types were found in some watersheds; they were not very concentrated and hence were dropped from the final analysis. This left only two categories: forested and the combination of agriculture and grassland (PAG).

Table 3.–Fish species represented in the fish assemblage metrics based on taxonomic groups and functional ecological guilds. Intolerant designations are based on Lyons (1992).

| Centrarchid | Intolerant | Benthic species | Vegetation-dwelling |
|---|---|---|---|
| *Lepomis macrochirus* | *Esox masquinongy* | *Noturus gyrinus* | *Umbra limi* |
| *L. gibbosus* | *Ambloplites rupestris* | *Etheostoma flabellare* | *Esox lucius* |
| *L. cyanellus* | *Micropterus dolomieu* | *E. nigrum* | *E. masquinongy* |
| *Ambloplites rupestris* | *Etheostoma exile* | *E. exile* | *Notropis heterodon* |
| *Micropterus salmoides* | *Notropis heterodon* | *Percina caprodes* | *N. heterolepis* |
| *M. dolomieu* | *N. heterolepis* | *P. maculata* | *N. volucellus* |
| *Pomoxis nigromaculatus* | *Cottus bairdi* | *Cottus bairdi* | *Noturus gyrinus* |
| | | | *Etheostoma exile* |

## *Statistical Analyses*

We used linear regression analysis (Proc REG, SAS 1990) to evaluate the relationships between plant metrics (FQI, total plant species richness per lake, the number of plant species per "tolerance grouping" and plant species richness and frequency of occurrence per vegetative structural form) and lakeshore development (number of dwellings per km of shoreline) and watershed land cover types (expressed as a proportion).

We used linear regression analysis to evaluate the relationship between the fish metrics (species richness of native fish species, centrarchid species, intolerant fish, small benthic fish and proportional abundance of intolerant fish and vegetation-dwelling fish of the total catch) and lakeshore development and watershed land cover types. For all regression models alpha was set at 0.05. Lakeshore development or watershed cover type variables were included in the regression models as fixed effects. All proportional data were arcsin-sqaure root transformed to better approximate a normal distribution. All abundance data were log transformed to better approximate a normal distribution.

Lake surface area has been identified as being important in predicting fish species richness in northern Wisconsin lakes (Jennings et al. 1998, Magnuson et al. 1998). Therefore, we evaluated the species richness-area relationships within our 16 lake data set to determine whether this variable should be included as an effect in the regression models.

# Results

## *Lake Area*

Linear regression analysis revealed no significant species richness-lake area relationship for fish ($F=0.78$, $P=0.392$) or aquatic plants ($F= 1.67$, $P=0.217$) within the 16 lakes; therefore, lake area was not included as an effect in the final regression models.

## *Lakeshore Development*

Linear regression models revealed consistent relations between aquatic plant assemblage attributes and lakeshore development. FQI scores were significantly lower for each lake as the number of dwellings per km of shoreline per lake increased (Fig. 1, Table 4). Both, the number of plant species per lake (Fig. 2, Table 4) and the number of highly intolerant plant species (Fig. 3, Table 4) were inversely related to number of dwellings per km of shoreline. The floating vegetation component of the aquatic plant community of the lakes decreased in species richness (Fig. 4, Table 4) and frequency of occurrence (Fig. 5, Table 4) with increasing lakeshore development. Other components of the macrophyte community in the lakes, such as the number of intolerant plant species, the number of moderately tolerant plant species, the number of tolerant plant species, and number of species and abundance of both emergent and submersed vegetation were not correlated with lakeshore development ($p > 0.05$, Table 4).

Linear regression models revealed that there was no significant relationship between any of the six fish assemblage metrics and lakeshore development ($p > 0.05$, Table 4).

## *Watershed Cover-types*

Cover-types in the 16 watersheds averaged 71% forest cover (range 37.2 to 97.9), and 14% grassland/ agriculture land (range 0.0 to 62.1). The linear regression models revealed no significant relations between any of the biological indicator variables and watershed cover-types variables ($p > 0.05$).

# Discussion

Our results indicated a decrease in the overall quality of the aquatic plant assemblage with increasing lakeshore development, as total number of species, especially intolerant species, declined. The results for the floating vegetation are consistent with the trends and patterns observed in other Wisconsin (Jennings et al. 2003) and Minnesota lakes (Radomski and Goeman 2001). Nichols et al. (2000) suggested that low abundance of floating vegetation is often an indication

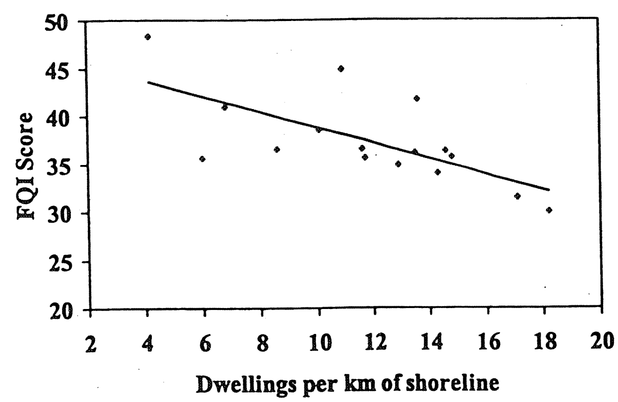


Figure 1.–Relationship between Floristic Quality Index (FQI) and the number of dwellings per kilometer of shoreline.

Table 4.–Sumary of results for linear regression analyses evaluating relationship between dwelling per kilometer of shoreline and the metrics of the plant and fish assemblages.

| | Mean (Range) | F-value | Model P-value | Model R-square |
|---|---|---|---|---|
| **Plant Metrics** | | | | |
| FQI | 37.3 (30.0-48.4) | 12.37 | 0.003 | 0.47 |
| Plant species richness | 28.1 (19-47) | 8.26 | 0.012 | 0.37 |
| Highly intolerant | 6.1 (3-12) | 20.00 | <0.001 | 0.59 |
| Intolerant | 10.1 (6-14) | 0.85 | 0.372 | 0.06 |
| Moderately tolerant | 6.4 (4-10) | 1.46 | 0.247 | 0.09 |
| Tolerant | 1.7 (0-4) | 2.14 | 0.166 | 0.13 |
| Emergent Species richness | 7.7 (3-17) | 1.44 | 0.250 | 0.09 |
| Floating Species richness | 2.1 (0-5) | 12.96 | 0.003 | 0.48 |
| Submersed Species richness | 15.4 (9-26) | 2.42 | 0.142 | 0.15 |
| Emergent Frequency | 108.3 (47-205) | 0.25 | 0.623 | 0.02 |
| Floating Frequency | 21.6 (0-120) | 8.87 | 0.010 | 0.39 |
| Submersed Frequency | 386.8 (132-714) | 2.65 | 0.126 | 0.16 |
| **Fish Metrics** | | | | |
| Native species | 11.7 (7-16) | 1.41 | 0.255 | 0.09 |
| Centrarchids | 4.8 (3-6) | 0.13 | 0.726 | 0.01 |
| Intolerant | 1.7 (0-5) | 0.12 | 0.737 | 0.01 |
| Small benthic | 1.3 (0-3) | 0.76 | 0.398 | 0.05 |
| Percent intolerant | 0.09 (0-0.72) | 0.24 | 0.632 | 0.02 |
| Percent vegetative | 0.02 (0-0.10) | 2.51 | 0.136 | 0.15 |

of high lakeshore development because riparian landowners will physically or chemically removed floating vegetation for a swimmable beach area. Boating can have direct (cutting of plants and uprooting them) and indirect (increase turbidity and increase suspended solids) effects on abundance of floating vegetation (Liddle and Scorgie 1980, Yousef et al. 1980, Asplund and Cook 1997)

The observed negative correlation between aquatic plant assemblage quality and lakeshore development in this study differs from the findings of Nichols (2001), who reported that FQI either increased or remained the same in 33 lakes that were sampled four or more times over a period of five or more years. The lakes studied by Nichols (2001) were greater in surface area and were highly variable in regards to several limno-

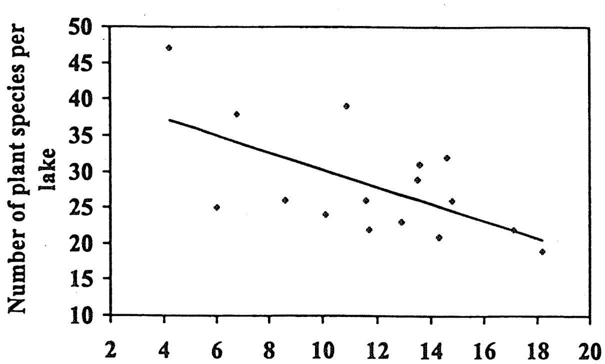


Figure 2.–Relationship between the number of plant species per lake and the number of dwellings per kilometer of shoreline.

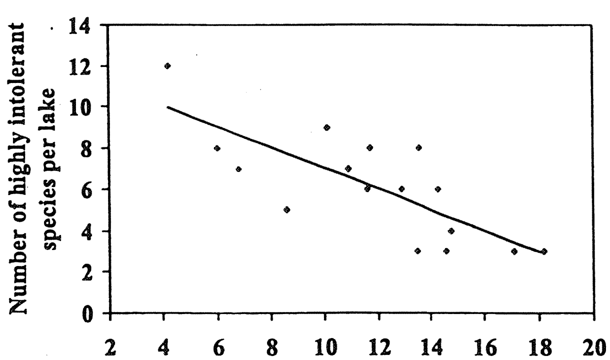


Figure 3.–Relationship between the number of highly intolerant plant species per lake and the number dwellings per kilometer of shoreline.

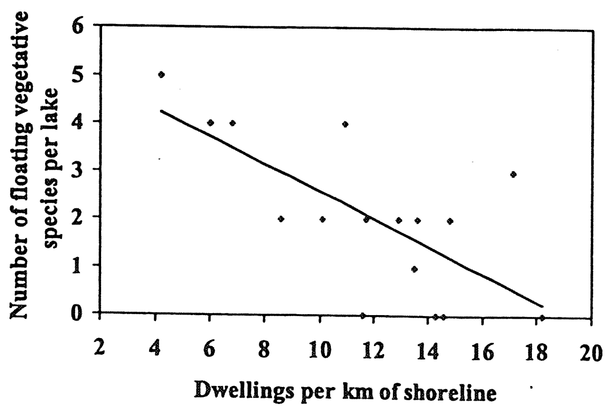


Figure 4.-Relationship between the number of floating vegetative species and the number of dwellings per kilometer of shoreline.

logical characteristics. He assumed that lakeshore development increased over the years of the study and analyzed the data by lake using linear regression analysis. However, Nichols (2001) warned that results from his study should be interpreted with caution because the quality of aquatic plant communities may have declined from human impacts before sampling had occurred in these lakes. Our study design differs from Nichols (2001) in the use of similar lakes with a range of development rather than developed lakes over time. Therefore, our study lacks the problem of interpreting past effects of development.

None of the fish assemblage metrics were significantly related to lakeshore development within this set of lakes, which differs from results observed in a set of Wisconsin lakes with larger surface areas (Jennings et al. 1998). Jennings et al. (1998) found that the number of intolerant fish and the proportion of intolerant fish had the strongest relationship with independent measures of environmental quality, and similar results were observed in a study of northeastern USA lakes (Whittier and Hughes 1998). The differences in the average number of intolerant species captured per lake between the studies appeared to influence the relationships. Our mean catch was 1.75 intolerant fish species per lake in lakes less than 80 ha in surface area, while Jennings et al. (1998) found a mean of 4.5 intolerant fish species per lake in lakes 80 ha in surface area. Mean catch for the number of small benthic species in lakes less than 80 ha was 1.3 species per lake, while Jennings et al. (1998) observed 4 species per lake in lakes 80 ha in surface area. The difference can be attributable to species-area relations, which have been documented in Wisconsin lakes (Jennings et al. 1998, Magnuson et al. 1998). The simple fish communities of most small lakes (< 80 ha surface area) contain insufficient variation to provide an effective set of indicators. In addition, processes at larger scales can obscure relations between environmental quality and fish assemblage structure. Specifically, human-mediated movements of fish (i.e., stocking, bait bucket transfers) among systems are known to effectively homogenize assemblage structure (Radomski and Goeman 1995, Rahel 2000). These movements arguably are an impact, yet they can mask the predictable relations to other impacts such as lakeshore development, water quality degradation or habitat alteration, leading to ambiguity (Jennings et al. 1999, Jennings et al. 2003, Radomski and Goeman 2001). Thus, a fish IBI-type approach appears to have limited value for assessing condition of lakes less than 80 ha surface area in northwestern Wisconsin.

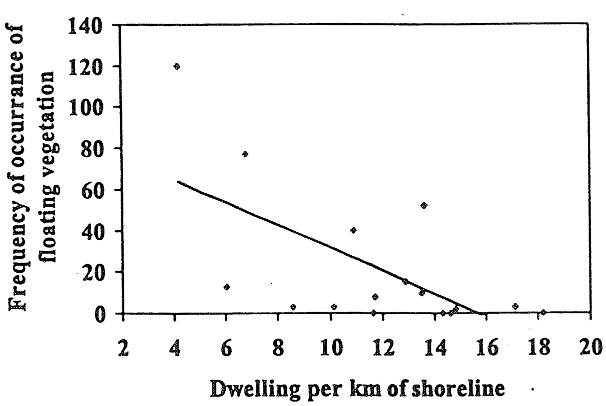


Figure 5. Relationship between the frequency of occurrence of floating vegetation and the number of dwellings per kilometer of shoreline.

Neither aquatic plant or fish assemblages had any relations with land cover types within the watershed of the lakes. Other studies have demonstrated that agriculture and urban development within a watershed can lead to an increase in the input of non-point sources of nutrients and sediments that lead to negative impacts on ecosystem health (Wang et al. 1997, Schindler 2001, Wang et al. 2001). We may not have observed this relationship in the present study because of the relatively narrow range of watershed cover types (Table 2). Hence, the potential range of non-point source inputs may have been small and obscured a relationship.

The results from this study indicate that in lakes less than 80 ha, the aquatic plant assemblage provide a more sensitive set of indicators of the effects of lakeshore development than did the fish assemblage. Inclusion of aquatic plant sampling would be beneficial as part of a monitoring approach to determine the status and trends in the health and condition of a lake ecosystem. Further research would be required to implement the FQI and aquatic plant communities as monitoring tools in other lake classes and ecoregions.

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