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Lack of exotic hydrilla infestation effects on plant, fish and aquatic bird community measures

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Abstract

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The invasion of hydrilla into North American lakes has provoked concern over loss of native flora and fauna and resulted in costly suppression efforts. We used two data sets to determine if common measures of ecosystem health; abundance, species richness, diversity and evenness, were affected by the presence of hydrilla. Data Set 1 consisted of 27 Florida lakes, 11 of which had hydrilla present for approximately 4–8 years in varying abundances, and 16 did not have hydrilla. Given the number of lakes, each was sampled only once in the summers of 1986-1990 for community measures of aquatic plants, birds, and fish. Data Set 2 consisted of 12 lakes, six with abundant hydrilla for over 23 years and six without hydrilla. These lakes were sampled every year in the summer (with a few exceptions due to weather conditions) between 1999 and 2006 for community measures of aquatic plants and fish. The results for both data sets show that presence of hydrilla had no statistically significant effect ($P > 0.05$) on all community measures tested (*i.e.*, richness, diversity, abundance). Our conclusions support the hypothesis that hydrilla in these Florida lakes has occupied a mostly vacant ecological niche and has not affected the occurrence or relative composition of native species of aquatic plants, birds, and fish. Because pond experiments have found negative effects of hydrilla, further focused research is needed to assist management decisions when considerable resources are to be spent each year on hydrilla suppression.

Key words: biodiversity, non-native species, species richness, subtropical lakes

Non-native species are becoming more prevalent in terrestrial and aquatic systems (Sax and Gaines 2003), creating a need to identify potential impacts of non-native species on native flora and fauna, whether they are negative, positive, or absent. The potential for negative impacts resulting from non-native plant species has caused conservation agencies to spend considerable effort on control and maintenance. However, once non-native species are well established in an ecosystem, they may not have the expected impacts on the other species present. While non-natives can out-compete native species, alter habitats, and change conditions for native flora and fauna (Vitousek *et al.* 1987, Vitousek and Walker 1989, Toft *et al.* 2003, Schlaepfer *et al.* 2005), studies have also found evidence that not all exotic species diminish native plant and animal community composition and/or abundance (Dukes 2001, Stohlgren *et al.* 2006). Exotic species can add to the local biota and increase species richness with no loss or decline of native species (Sax and Gaines 2003, Houlahan and Findlay 2004, Starck *et al.* 2006). Resource managers should identify potential pros, cons, or lack of impacts that non-natives bring to the invaded systems. We tested whether non-native aquatic plant Hydrilla *Hydrilla verticallata* (L.f. Royle) had a negative impact on other aquatic flora and fauna on subtropical lakes in Florida.

Hydrilla is a non-native aquatic plant that became established in Florida's freshwater systems in the 1960s and has since spread across much of the United States. Hydrilla can grow with as little as 1% sunlight, inhabit the entire water column, and grow as deep as 15 m (Langeland 1996). In Florida, it is a particularly prolific species because the warm climate and shallow lakes facilitate its rapid growth. Hydrilla hinders navigation by fouling boat propellers,and can almost completely cover shallow Florida lakes and harm commercial and recreational lake uses (Colle *et al.* 1987). Conversely, many anglers enjoy fishing in hydrilla structures, and hydrilla has been identified as a preferred habitat and major food source for waterfowl (Montalbano *et al.* 1979, Johnson and Montalbano 1984, Esler 1990).

Millions of dollars are spent every year to control this species for navigation and habitat enhancement across the United States and in other countries. The state of Florida has debated use of funds to control hydrilla for over 25 years, and in that time, approximately $174 million was spent by the Florida Department of Environmental Protection to control this plant alone (Jeff Schardt, pers. comm., Feb 2007, Florida Department of Environmental Protection [FDEP]).

Effects of hydrilla on individual plant (Haller and Sutton 1975, Van *et al.* 1999, Colon-Gaud *et al.* 2004), bird (Johnson and Montalbano 1984, Esler 1990), and fish species (Shireman *et al.* 1984, Colle *et al.* 1987, Bettoli *et al.* 1993, Bonvechio and Bonvechio 2006) have been evaluated to some degree. However, the collective impacts of hydrilla on communities of native plants and animals have seldom been evaluated, and few peer-reviewed studies have tested the effects of hydrilla on community composition of biota in lakes. Our approach to evaluating habitat alteration was to sample subtropical lakes of variable trophic states in Florida, USA, during a relatively short time period (approximately 4–8 years) and a longer period (>23 yr) to assess whether the presence of hydrilla was associated with differences in aquatic plant, aquatic bird, and fish community measures.

Materials and methods

Two data sets were used to examine the impact of hydrilla on community measures of flora and fauna in Florida lakes. In the first data set, chlorophyll concentrations, aquatic macrophyte, fish and aquatic bird assemblages, and physical lake attributes from 27 lakes in north central Florida were measured once between June 1986 and June 1990 (Canfield and Hoyer 1992, Hoyer and Canfield 1994). These lakes represented a range of trophic categories (oligotrophic, mesotrophic, eutrophic, and hypereutrophic) based on chlorophyll concentrations and the classification guidelines of Forsberg and Ryding (1980). Within each trophic category was also a wide range of aquatic macrophyte abundances. Each lake served as a replicate for hydrilla presence or absence treatments. The sampling event for each of the 27 lakes took place during May and November for water, aquatic macrophytes, and fish and occurred on the same day at each lake. Bird observations occurred on three separate dates over a one-year period.

Water samples were collected during one sampling event per lake between 1986 and 1990. Six samples (three littoral and three in open water zones) were collected during one day for each lake between May and November. Water samples were collected 0.5 m below the surface in acid-cleaned Nalgene bottles, placed on ice, and returned to the laboratory for analysis. Water was filtered through Gelman type A-E glass fiber filters for chlorophyll *a* determination. Chlorophyll *a* concentrations were measured by using the method

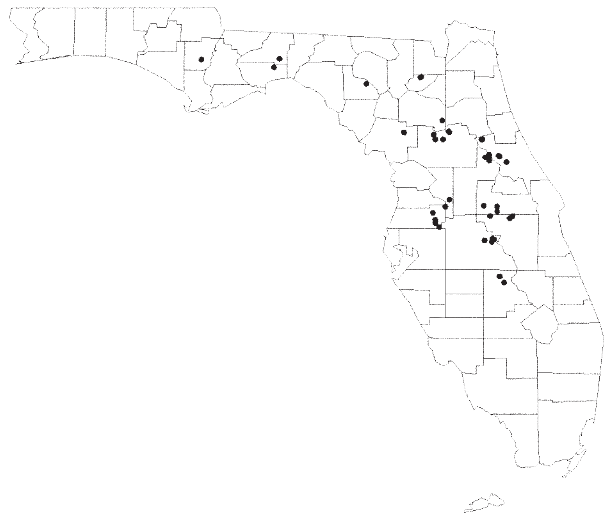

Figure 1.-Location of lakes sampled in Florida.

of Yentsch and Menzel (1963) and the equations of Parson and Strickland (1963).

Aquatic macrophyte density was recorded during one sampling event per lake between May and November. Following procedures described by Maceina and Shireman (1980) and Standard Methods (APAH 1992, Method 10400 C), four transects were established completely across each lake with a boat-mounted Raytheon DE-719 recording fathometer for calculating percentage of lake area covered with macrophytes (PAC), percent volume inhabited with aquatic vegetation (PVI), and lake mean depth. Lake surface area was obtained from the Gazetteer of Florida Lakes (Shafer *et al.* 1986).

Aquatic plant community assemblage and structure was described by species richness in the emergent/littoral zone and was determined using 10 uniformly distributed transects in each lake. Littoral zone was defined by the area of a lake with aquatic vegetation and included area from the shoreline to the edge of vegetation moving toward the center of the lake. Transects for plant richness were evaluated during one sampling event per lake between May and November. Plant species found along transects were recorded. Plant richness, defined as total number of species observed, and richness of submersed plants were also recorded by lake. Hydrilla presence in lakes was identified using these transects. For lakes with hydrilla, plant richness and submersed plant richness for analysis was defined as number of species observed minus 1 (*i.e.*, hydrilla) to remove any treatment effects.

Eleven lakes with hydrilla from the first data set were monitored for hydrilla occurrence by the State of Florida's Bureau of Invasive Plant Management (Jeff Schardt, pers. comm., Feb 2007, FDEP), which began monitoring hydrilla infestations in 1982. The majority of our hydrilla lakes had hydrilla detected by 1983, two were detected by 1985, and one by 1990. Two of the 11 lakes were not inventoried by FDEP, and thus we have no record of the duration of hydrilla occurrence prior to our sampling for those systems. The FDEP treated 6 of the 11 hydrilla lakes in our study with herbicides before our study (2–4 years prior to our study). In addition to already being treated with herbicides, three had grass carp *Ctenopharyngodon idella* added to them in the year of our study. In addition to the Bureau's data, Florida's Department of Natural Resources (FDEP 1982–1999) collected estimates of surface area of hydrilla in some of our lakes. These estimates were collected in the same year as our plant data.

Fish abundance and community composition in the littoral zone was collected during one sampling event per lake between the summer months of May and October. Fish were collected using rotenone within two to six 0.08-ha blocknets at each lake. The number of nets used was determined by lake size. Littoral habitat was defined by the area of a lake with aquatic vegetation and included area from the shoreline to edge of vegetation. If there was minimal aquatic vegetation, the littoral habitat was considered the width of the 0.08 ha blocknet (28 m) along the entire lake. Blocknet areas were treated with 2.0 mg/L rotenone (5% active ingredient, Noxfish). Sampling procedures for blocknetting followed Shireman *et al.* (1983). Fish killed inside the nets were collected for three days, separated by species and counted to determine fish density (Fish/ha).

Aquatic bird abundance (aquatic birds were defined as those using some form of aquatic habitat for life functions) and cumulative species counts at 23 of the lakes were conducted between 1989 and 1990 (Hoyer and Canfield 1994). One daytime bird observation was made for winter, spring, and summer at each lake for a total of three observations per lake. Birds were counted by observers who progressed once around the perimeter of each lake in a small boat. Care was taken to not count birds in duplicate that flushed ahead of the boat. Most birds were identified to species, with the exception of gulls, terns, and crows, which were identified to genus. All three counts were averaged by species and divided by the lake surface area to yield a bird density estimate (birds/km^2). Species richness for individual lakes was defined as total number of species observed during all three counts.

The second data set used in this study included long-term chlorophyll concentrations and fish assemblage data collected on 12 lakes every year between 1999 and 2006 (Florida LAKEWATCH 2007), except for three lakes that were sampled for only five years because of drought and lake access issues. Aquatic macrophyte data were also collected on these lakes every other year. Hydrilla was established in six of the 12 lakes prior to 1983, and the percent area covered with hydrilla in these six lakes averaged 12%, ranging 1–45% (FDEP 1982–1999). The maximum hydrilla coverage for these six lakes during this same time period averaged 27%, with a range of 3–86%. Hydrilla was not detected in the remaining six lakes of the second data set; thus, this set gives a comparison of fish and aquatic plant community assemblages between lakes with abundant hydrilla established for at least 23 years relative to lakes without hydrilla.

Chlorophyll concentrations were measured monthly by volunteers for the second data set according to the methods described in Canfield *et al.* (2002). Aquatic macrophyte data was measured every other year using the same methods listed above (again three lakes missed one yearly sample due to drought and access issues). Six transects were established uniformly around each of the 12 lakes for sampling fish populations. Each year fish were collected in spring (Jan to Apr) using electrofishing for 10 minutes at each transect. At each transect, all fish were separated into species and counted to estimate catch per unit effort (CPUE: fish/hr of electrofishing).

To determine if lakes were sufficiently sampled for fish species, a species exhaustion index (Bachmann *et al.* 1996) was used for both data sets. For the first data set, the cumulative species count in the second to last blocknet was divided by the cumulative number of species counted in the last blocknet. For the long-term data set, the cumulative species count in the second to last year of sampling was divided by the cumulative species counted in the last year. For both data sets, a value $\geq 90\%$ was used as the criterion to indicate that the majority of species had been collected.

Fish and bird community assemblages and structure were described by density estimates, species richness, species diversity, and species evenness. Species diversity was determined using the reciprocal of Simpson's Index (D) (Krebs 1998):

$$1/D = \sum p_i^2$$

where 1/D = Simpson's reciprocal diversity index, and p_i = the proportion of individual species in the community. In this form, Simpson's diversity can most easily be interpreted as the number of equally common species required to generate the observed heterogeneity of the sample (Krebs 1998). Simpson's measure of evenness was calculated as:

$$E_{1/d} = 1/D/s$$

where $E_{1/d}$ = Simpson's measure of evenness; 1/D = Simpson's reciprocal diversity index; and s = Number of species in the sample.

In both data sets, lakes were separated into two groups based upon the presence or absence of hydrilla. Multiple years of data (5–8 years) in the second data set were averaged by lake, so for analyses each lake had only one observation. Two sample t-tests were used to test for differences in lake

Table 1.-Statistics for lake surface area, chlorophyll concentrations, and four measures of aquatic plant, aquatic bird and fish community measures estimated in lakes with (n=11, for approximately 4 to 8 years) and without hydrilla (n=16). All means were compared with a two-sample t-test after appropriately normalizing the data for parametric analyses. No significant differences were found ($p \leq 0.05$).

	Data Set 1					
	Hydrilla Lakes (n=11)			No Hydrilla Lakes (n=16)		
Parameter	Mean	Standard Deviation	Range	Mean	Standard Deviation	Range
Surface Area (ha)	133	73	39-271	94	61	28-254
Chlorophyll (µg/L)	16	29	2-102	48	65	1-206
Percent Area Covered	60	37	0-100	29	38	0-100
Percent Volume Inhabited	41	37	0-98	21	35	0-100
Total Plant Species Richness	14.4	3.7	11-23	11.8	3.4	5-17
Submersed Plant Species Richness	2.5	2.8	0-9	2.1	2.0	0-6
Bird Density (birds/km^2)	245	225	29-803	215	216	9-747
Bird Species Richness	23.7	5.3	15-30	18.9	7.7	8-27
Bird Diversity	5.3	3.0	2.0-11.3	6.1	2.0	1.8-9.1
Bird Evenness	0.2	0.1	0.1-0.5	0.4	0.2	0.2-0.9
Fish Density (fish/ha)	28,090	17,141	4,950-55,793	32,099	42,347	1,056-126,462
Fish Species Richness	17.4	4.0	11-24	14.8	4.8	3-24
Fish Diversity	4.9	1.9	1.7-7.9	3.7	1.4	2.1-7.4
Fish Evenness	0.3	0.1	0.2-0.6	0.3	0.3	0.1-1.4

surface area, lake trophic status (estimated with chlorophyll), and community composition for plants, fish, and aquatic birds between lake types (hydrilla vs. no hydrilla). Significance was determined at $p = 0.05$ for all tests. Percent area covered (PAC) and percent volume inhabited (PVI) with aquatic plants were transformed with an arcsine transformation, and where necessary other variables were $\log_{10}$-transformed to accommodate heteroscedasticity in the data (Sokal and Rolf 1981).

Results

Species richness of plants, fish, and birds increases with lake surface area (Hoyer and Canfield 1994, Bachmann *et al.* 1996, Bachmann *et al.* 2002, Lawton 1999). Thus, it was important that lakes in the categories of hydrilla and non-hydrilla were of similar-sized surface areas when looking for differences in flora and fauna community assemblages. In the first data set, 11 lakes were analyzed with hydrilla and 16 without hydrilla. Lakes ranged in surface area from 28 to 271 ha, and mean lake area did not differ between the lake types (Table 1). In the second data set, six lakes with and six lakes without hydrilla were analyzed. Lake surface area averaged larger in the second data set, ranging in size from 47 to 5541 ha. However, mean lake size again did not differ between the lakes with and without hydrilla (Table 2). Additionally, the range of lake surface areas within each data set were small, showing no significant correlations ($p \leq 0.05$) between lake area and measures of species richness (plants, birds, and fish in Data Set 1; fish and plants in Data Set 2).

Lake trophic status of lakes in both data sets ranged from oligotrophic to hypereutrophic based on measures of chlorophyll. Chlorophyll concentrations ranged from 1 to 206 µg/L and 3 to 132 µg/L in Data Set 1 and Data Set 2, respectively (Tables 1 and 2). Further, no significant differences in the mean chlorophyll concentrations were found between lakes with and without hydrilla in either data set (Tables 1 and 2).

The abundance of aquatic macrophytes in both data sets also ranged considerably. Percent area covered with aquatic macrophytes ranged from 0 to 100% and 2 to 88% in Data Set 1 and Data Set 2, respectively (Tables 1 and 2). Percent volume inhabited with aquatic macrophytes ranged from 0 to 100% and 1 to 44% in Data Set 1 and Data Set 2, respectively. There were no significant differences in PAC or PVI between lakes with and without hydrilla in either data set (Table 1 and 2).

Total aquatic plant species richness (minus 1 if hydrilla was present) in Data Set 1 averaged 14.4 (range 11–23) and 11.8 (range 5–17) in lakes with and without hydrilla, respectively (Table 1). Submersed plant species richness (minus 1 if hydrilla was present) for these lakes averaged 2.5 (range 0–9) and 2.1 (range 0–6) in lakes with and without hydrilla, respectively. These averages for total and submersed plant species richness were not significantly different between lakes with and without hydrilla (Table 1).

Bird density averaged 245 birds/km^2 (range 29–803 birds/km^2) and 215 birds/km^2 (range 9–747 birds/km^2) in lakes with

Table 2.-Statistics for lake surface area, chlorophyll concentrations, and four measures of aquatic plant and fish community measures estimated in lakes with (n=6, for greater than 23 years) and without hydrilla (n=6). All means were compared with a two-sample t-test after appropriately normalizing the data for parametric analyses. No significant differences were found ($p \leq 0.05$).

	Data Set 2					
	Hydrilla Lakes (n=6)			No Hydrilla Lakes (n=6)		
	Mean	Standard Deviation	Range	Mean	Standard Deviation	Range
Surface Area (ha)	2,132	1,892	70-5,541	1,295	1,085	47-3,006
Chlorophyll (µg/L)	22	35	3-93	30	51	3-132
Percent Area Covered	42	32	9-88	16	19	2-53
Percent Volume Inhabited	16	20	1-44	3	2	1-6
Total Plant Species Richness	37.8	11.7	24-53	29.5	9.9	20-45
Submersed Plant Species Richness	11.0	4.0	7-17	8.0	5.4	1-16
Fish CPUE (fish/hour)	343	190	142-605	468	283	156-931
Fish Species Richness	24.2	5.1	19-34	23.0	3.5	19-27
Fish Diversity	4.5	0.9	3.4-5.8	3.7	0.9	2.8-5.1
Fish Evenness	0.3	0.1	0.3-0.4	0.3	0.1	0.2-0.4

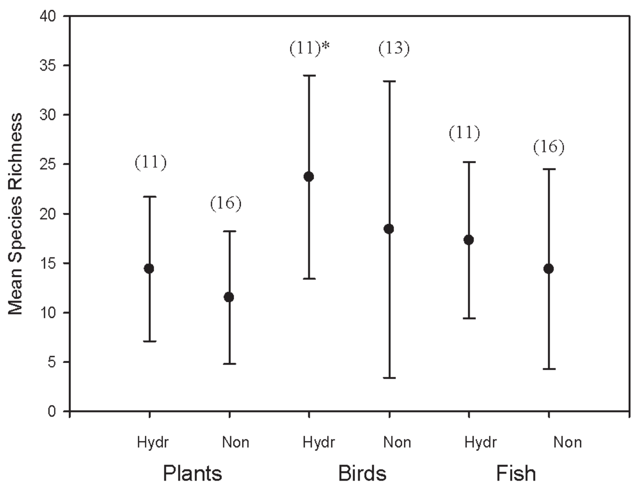

Figure 2.-Mean species richness of plants, aquatic birds, and fish in the subset of lakes between the surface areas of 25 and 300 hectares with hydrilla present (Hydr) and absent (Non). Error bars represent 95% confidence interval. Numbers in parentheses represent N lakes sampled. The star indicates lakes in which the parameter being tested was significantly greater.

and without hydrilla, respectively (Table 1). These averages for bird density were not significantly different between lakes with and without hydrilla. All bird community measures (bird species richness, bird diversity, and bird evenness) in Data Set 1 also showed no significant difference between lakes with and without hydrilla (Table 1).

Fish density in Data Set 1 averaged 28,090 fish/ha (range 4,950–55,793 fish/ha) and 42,347 fish/ha (range 1,056–126,462 fish/ha) in lakes with and without hydrilla respectively (Table 1). These averages for fish density were not significantly different between lakes with and without hydrilla. Fish species richness in Data Set 1 also did not differ significantly between lakes with and without hydrilla (Table 1). The exhaustion index for fish species richness in Data Set 1 showed that 9 of 26 lakes had a value <90% and thus may not have been sufficiently sampled for fish richness. When these samples were removed, we found no significant difference in mean fish richness. Mean fish richness was 17.5 (range 11–24) in lakes with hydrilla (n = 9) and 13.9 (range 3–24) in lakes without hydrilla (n = 10). All other fish community measures (fish diversity and fish evenness) in Data Set 1 also showed no significant difference between lakes with and without hydrilla (Table 1).

Fish CPUE in Data Set 2 averaged 343 fish/hr (range 142–605 fish/hr) and 468 fish/hr (range 156–931 fish/hr) in lakes with and without hydrilla, respectively (Table 2). These averages for fish CPUE were not significantly different between lakes with and without hydrilla. Fish species richness in Data Set 2 also did not differ significantly between lakes with and without hydrilla (Table 2). The exhaustion index for fish species richness in Data Set 2 showed that all lakes exceeded 95%, indicating they had been sufficiently sampled for fish richness. All other fish community measures (fish diversity and fish evenness) in Data Set 2 also showed no significant difference between lakes with and without hydrilla (Table 2).

Discussion

We detected no significant differences in plant community measures between lakes with and without hydrilla in either data set. This is contrary to past suggestions that hydrilla displaces native plants and decreases plant diversity (Haller and Sutton 1975, Colon-Gaud *et al.* 2004). Haller and Sutton (1975) grew hydrilla and eel grass (*Vallisneria americana*) in shallow earthen ponds for two years to test community structure and competitive interactions. They concluded that hydrilla formed a dense canopy, decreased light penetration, and reduced growth rate and abundance of the native eel grass. Although hydrilla has been observed to out-compete the submersed plant *V. americana* in high nutrient soils, under nutrient limiting conditions growth and competition of hydrilla was depressed and *V. americana* became dominant (Van *et al.* 1999). This is evidence that native species may be more competitive than invasives in certain habitats.

Our analysis also showed no significant differences in community measures for bird communities in lakes with and without hydrilla (Data Set 1). However, a study in a Texas reservoir, found increased species richness of birds within areas of the reservoir containing hydrilla compared to plots where hydrilla was removed (Esler 1990). This increase was mostly due to the presence of more duck species (Esler 1990). In Lake Okeechobee, Florida, waterfowl did not use habitat types in proportion to availability, and hydrilla was among the most preferred habitat (Johnson and Montalbano 1984). Hydrilla also supported a greater diversity of duck species than other plant communities at Lake Okeechobee. The reported increase in bird diversity and richness was probably due to the plant's value as a food resource. Hydrilla has been identified as an important food item for ducks and coots wintering in Florida, comprising 32% of the stomach aggregation and having an occurrence rate >25% (Montalbano *et al.* 1979). The lack of differences found in this study was probably because we assessed whole-lake aquatic bird communities and not waterfowl communities using individual habitat types. Similarly, Hoyer and Canfield (1994) examined bird communities using Florida lakes and found no relation between aquatic plant abundance or either bird abundance or species richness after accounting for lake trophic status and lake area.

Fish community measures in both data sets showed no significant differences in lakes with and without hydrilla. Our study tested effects among multiple lakes and did not focus on a particular species but rather the fish community as a whole, and results indicated no differences in fish metrics with hydrilla occurrence.

Most past studies describing fish communities in hydrilla versus no-hydrilla infested waters have been tested through time in a single lake and have focused on sport fish. Shireman *et al.* (1983) assessed fish communities in Lake Baldwin, Florida, before and after hydrilla removal by grass carp. Comparing no macrophytes to the former hydrilla infested state, total number of fish and standing crop of fish fluctuated greatly and did not appear to be related to macrophyte abundance. Species richness also stayed the same, with species composition changing to high percentages of open water species such as threadfin shad (*Dorosoma petenense*) and gizzard shad (*D. cepedianum*). Similarly, Bettoli *et al.* (1993) measured fish community response to complete hydrilla removal via grass carp at Lake Conroe, Texas. They found that fish biomass declined for 8 of 17 species following hydrilla removal. Some planktivorous fishes, including threadfin shad and brook silverside (*Labidesthes sicculus*), increased several-fold following hydrilla removal. The community responses from Bettoli *et al.* (1993) resulted from complete removal of hydrilla and nearly all other aquatic plants, and thus, the changes from a vegetation-based to a plankton-based food web.

Bonvechio and Bonvechio (2006) evaluated relationships between habitat and sport fish populations over a 20-year period at Lake Tohopekaliga, Florida. During this period hydrilla coverage was highly variable, and they found no relation between electrofishing catch rates of largemouth bass (*Micropterus salmoides*) and hydrilla coverage. However, as hydrilla coverage increased, fishing effort and angler catch rates of largemouth bass increased. Colle *et al.* (1987) found that harvestable largemouth bass abundance was not significantly correlated with hydrilla coverage across years at Orange Lake, Florida. Although anglers at Orange Lake were similarly successful during the period of high hydrilla coverage, overall fishing effort declined when hydrilla coverage was high (Colle *et al.* 1987). Thus, although evidence is mixed about the impacts of hydrilla on sport fish abundance, very high hydrilla coverage can cause economic, recreational, and navigational harm to lakes.

Researching the question of invasive species impacts on a single system through time has an advantage over our empirical approach and should be continued in future research efforts. However, our empirical study among multiple lakes with hydrilla over a relatively short time period (Data Set 1, approximately 4–8 years) provides evidence that hydrilla occurrence was not associated with reductions in other species of aquatic flora and fauna. For Data Set 1, we did not measure hydrilla coverage at lakes sampled, but we were able to relate hydrilla measures from other sampling efforts at some of these lakes during the same sample years. For example, FDEP data indicated that two of our study lakes, Okahumpka and Miona, had >70% hydrilla coverage. These two lakes did not have significantly different richness for flora (richness in both lakes was 15) or fauna (richness for birds was 30 and 21, respectively; richness for fish was 20 at both lakes) and were not different in fish diversity relative to other lakes in our study with no hydrilla. However, temporal persistence of hydrilla was not documented at our lakes, and high hydrilla content combined with a long-term presence could eventually create an impact on community richness.

Using Data Set 2 we were able to compare the flora and fauna of lakes with abundant hydrilla for a much longer period (>23 years) and lakes of equal size and trophic status without hydrilla. The analyses again showed no significant differences between the community measures. Individual and/or multiple forms of aquatic plant management probably occurred on all of the lakes in both data sets, with the majority occurring on lakes with hydrilla. Therefore, if aquatic plant management were impacting the flora and fauna of the lakes, we would expect our community measures to be less in the hydrilla lakes (especially for aquatic plant communities), which was not the case for either data set. No significant differences in aquatic plant, fish, and aquatic bird communities measures in lakes with and without hydrilla were found, which indicates that establishment of this invasive species probably occurred in lakes where open niches made resource availability ample for its growth (Brown and Sax 2004). Similarly, recent studies of plants have shown terrestrial cases in which invasives become established and can increase the diversity of the environment without negatively impacting native species (Sax and Gaines 2003). As in terrestrial systems, hydrilla did not negatively influence lake biota for our sample of Florida lakes.

Though we found no significant differences between tested parameters, our results do not state definitively that the presence of hydrilla has no or will not in the future have negative impacts to aquatic systems. We publish these data, however, because there is much confusion, misstatements, and heated debate on managing hydrilla in aquatic systems (Hoyer *et al.* 2005). There are also reservations about publishing data such as these because many scientists wish to err conservatively. For example, one anonymous reviewer of an earlier version of this manuscript wrote the following sentence about our conclusions: " While these conclusions may be technically correct it implies something that may not be true." Obviously, such an argument stands opposed to an objective evaluation of the impacts of non-native species. This type of sentiment is not new, as Dequine (1952) describes management of freshwater commercial fishing nets in Florida over 55 years ago:

> *The fisheries biologist must learn from past example that his proposed management practices must be based upon facts and conclusions that have been thoroughly analyzed and critically examined. If he examines carefully the reasons for enactment of restrictive commercial fishing regulations, he will find that frequently there was little valid evidence of depletion due to overfishing available at the time of enactment, but that the "logical" deduction, sentiment and opinions, and an absence of facts about the effect of commercial fishing were largely responsible.*

We present the data examined in this manuscript to continue critical examinations of the impact of hydrilla on the flora and fauna of lakes. Objective evaluations of these and other data could improve lake management efforts in the future.

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