

# Is the cure worse than the disease? Comparing the ecological effects of an invasive aquatic plant and the herbicide treatments used to control it

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## Abstract

Invasive species are known to have negative ecological effects. However, few studies have evaluated the impacts of invasive species relative to the effects of invasive species control, thereby limiting our ability to make informed decisions considering the benefits and drawbacks of a given management approach. To address this gap, we compared the ecological effects of the invasive aquatic plant Eurasian watermilfoil (*Myriophyllum spicatum* L.) with the effects of lake-wide herbicide treatments used for *M. spicatum* control using aquatic plant data collected from 173 lakes in Wisconsin, USA. First, a pre-post analysis of aquatic plant communities found significant declines in native plant species in response to lake-wide herbicide treatment. Second, multi-level modeling using a large data set revealed a negative association between lake-wide herbicide treatments and native aquatic plants, but no significant negative effect of invasive *M. spicatum*. Taken together, our results indicate that lake-wide herbicide treatments aimed at controlling *M. spicatum* had larger effects on native aquatic plants than did the target of control—invasive *M. spicatum*. Our comparison reveals an important management tradeoff and encourages careful consideration of how we balance the real and perceived impacts of invasive species and the methods used for their control.

**Key words:** invasive species control, impacts, herbicides, aquatic plants, macrophytes

## Introduction

Humans are transporting species faster, farther, and more frequently than ever before, thus providing opportunities for species to establish populations far outside of their native range. These invasive species can have adverse effects on native species and recipient ecosystems (Ricciardi 2007;

Ehrenfeld 2010). In response, significant effort is directed toward suppressing or eradicating invasive species using a variety of approaches such as pesticide application, mechanical removal, and biological control.

Reducing invasive species populations can mitigate the adverse effects on native species and ecosystems (Simberloff 2009); however, control actions also have the potential for unintended and harmful effects on native species and ecosystems (Zavaleta et al. 2001; Bergstrom et al. 2009; Rinella et al. 2009; Lu et al. 2015). Given that both invasive species and invasive species control can have negative effects on native species and ecosystems, there is a clear need to compare their relative effects. Here, we directly compare the ecological effects of an invasive species relative to those of herbicide treatments often used for control. We ask: is the management “cure” for invasive species worse than the disease it is intended to treat? The ability to make this direct comparison is of great value to natural resource managers who often grapple with management trade-offs and are ultimately interested in minimizing negative impacts on native species and ecosystems.

This study examines the nonnative aquatic plant *Myriophyllum spicatum* (Eurasian watermilfoil), which in many parts of North America is a notorious nuisance. For example, lakefront property values in the U.S. states of Wisconsin and Washington were 13%–19% lower on lakes invaded by *M. spicatum* (Provencher et al. 2012; Olden and Tamayo 2014). Recreational impacts following *M. spicatum* invasion have also been well documented (Horsch and Lewis 2009; Eiswerth et al. 2000). However, results contrast when it comes to *M. spicatum*’s ecological effects. *Myriophyllum spicatum*, like many invaders, is often assumed to have adverse ecological impacts—and this has been verified in a few studies that examine large or rapidly expanding populations (Madsen et al. 1991; Boylen et al. 1999). Other studies stop short of declaring adverse ecological effects, do not link *M. spicatum* invasion to reductions in native species across the landscape, or reveal the abundance distributions are not remarkably different from those of native species (Trebitz and Taylor 2007; Hansen et al. 2013b; Gräfe 2014; Muthukrishnan et al. 2018).

Given the potential for undesirable consequences, herbicide treatments are often used as a management tool to control *M. spicatum* populations. Lake-wide chemical treatments (frequently using one of several different formulations of 2,4-dichlorophenoxyacetic (2,4-D) acid alone or in combination with other herbicides) have been shown to produce short-term reductions in *M. spicatum* populations (Kovalenko et al. 2010; Kujawa et al. 2017). Yet several studies have also found that large-scale herbicide treatments can cause significant declines in native aquatic plants, in addition to the target invasive species (Wagner et al. 2007; Nault et al. 2014, 2018).

While research suggests that both invasive *M. spicatum* and lake-wide herbicide treatment can have negative effects on native plant species, no documentation exists to compare the magnitude of these negative effects. Here, we used a large observational data set for lakes in the state of Wisconsin, USA, and statistically compared the effects of *M. spicatum* and lake-wide herbicide treatments on native aquatic plant communities. First, we evaluated the impacts of lake-wide herbicide treatment on native plant species using a pre–post comparison that assesses native species declines. Second, we used a multi-level modeling framework to statistically compare the effects of herbicide treatment and the effects of invasive *M. spicatum* on native aquatic plants. Taken together, we examine whether the negative ecological effect of lake-wide herbicide treatments used to control *M. spicatum* exceeds the negative ecological effect of *M. spicatum*. Our results underscore the need for developing a better understanding of the relative impacts of invasive species and the methods that are being used to control their populations.

## Methods

### Aquatic plant surveys

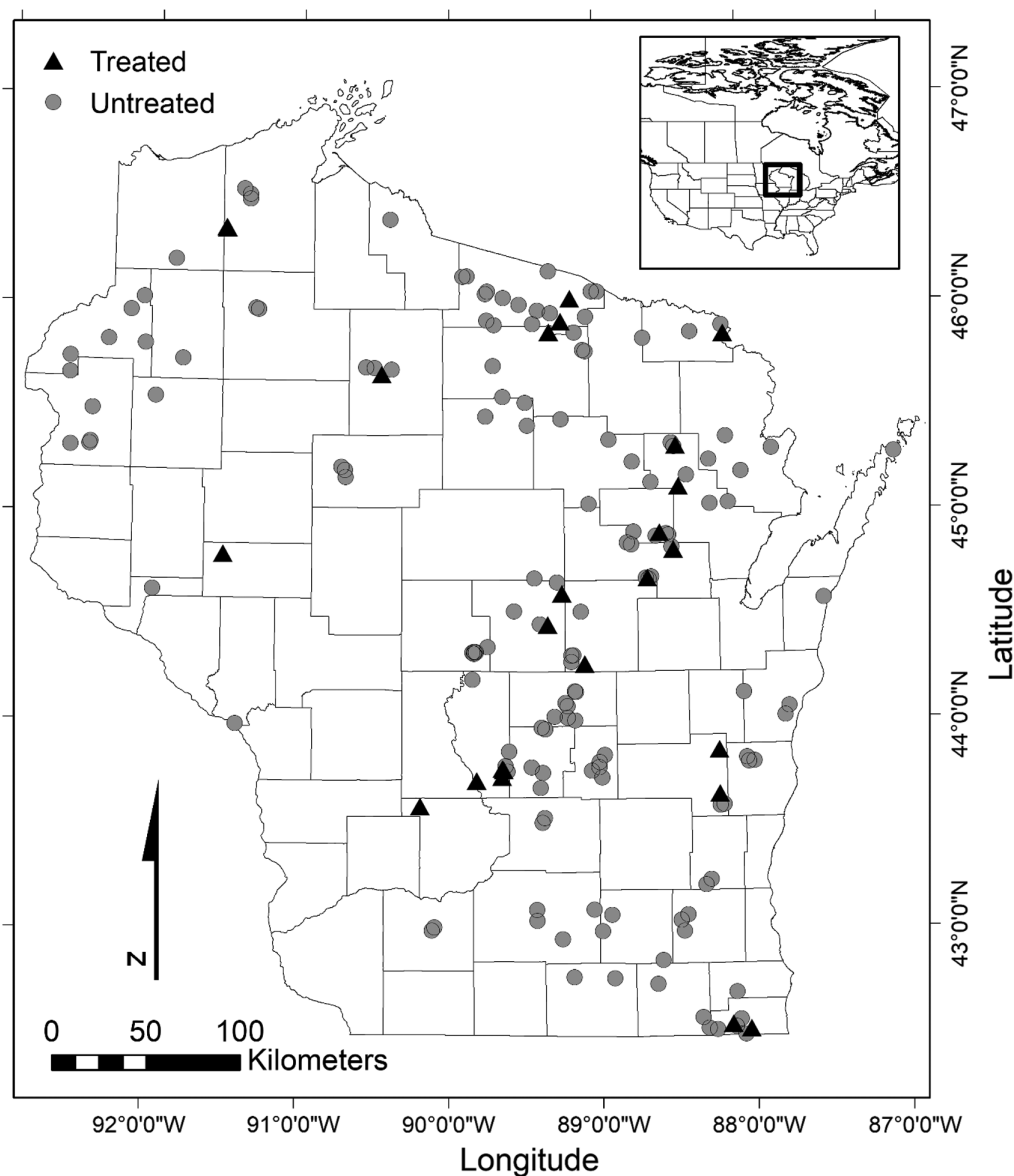
We obtained aquatic plant survey data from 442 Wisconsin lakes sampled between 14 May and 12 October 2005–2012 under the aquatic plant management program administered by the Wisconsin Department of Natural Resources (DNR). Some lakes were surveyed multiple times, resulting in a final set of 634 aquatic plant surveys. A subset of the lakes in this larger data set was used in this analysis ([Fig. 1](#)). Surveys employed a grid-based point-intercept sampling method to sample submersed, emergent, floating leaf, and free-floating species presence from a boat at multiple points per lake ([Hauxwell et al. 2010](#)). The number of points scaled with lake littoral area and shoreline complexity ([Mikulyuk et al. 2010](#)) and, on average, there were 199 sampling points per lake littoral zone, ranging from 10 to 1017. At each littoral sampling point, observers used a double-sided bow rake attached to a 4.5-m pole to remove plants from a 0.3-m<sup>2</sup> area. A similar rake attached to a rope was used to collect plants from sampling points deeper than 4.5 m. All live plants detached by the rake were identified to species ([Crow and Hellquist 2000a, 2000b](#)), except for macroalgae species (i.e., *Chara* and *Nitella*), which were identified to genus. Species present in fewer than 5% of lakes were excluded because we lacked sufficient data to describe their occurrence patterns. For each aquatic plant species, we estimated the frequency of occurrence in the littoral zone as the proportion of sampled points a species was detected. We use this as our metric of aquatic plant abundance (hereafter abundance) as the response variable for all analyses.

We obtained lake environmental data from [Papeş et al. \(2011\)](#). Water clarity and alkalinity are known to be important drivers of plant community composition; therefore, we used estimates of Secchi depth (m) and alkalinity (mg CaCO<sub>3</sub>/L) as environmental variables ([Vestergaard and Sand-Jensen 2000](#); [Mikulyuk et al. 2011](#)). Missing values comprised 4% of the total number of observations and were imputed via predictive mean matching ([Little 1988](#)).

### Pre-post herbicide treatment analysis

To evaluate the effect of lake-wide herbicide treatment on aquatic plant communities, we performed a pre-post treatment comparison, using untreated lakes sampled across time as a background against which to compare observed patterns in species abundance. We used DNR treatment records to identify lakes that experienced lake-wide herbicide treatment for *M. spicatum* that also had pre- and post-treatment plant data. Lakes were only included if a pretreatment survey occurred fewer than three years prior to treatment and the post-treatment survey occurred within a year of treatment ( $N = 25$  lakes; [Table S1](#)). Herbicide treatments took place in early spring (April–May) while plant community surveys took place later in the growing season (late June to early September). Treatments varied with respect to herbicide formulation and application rate, but all were designed to attain lake-wide herbicide concentrations capable of effectively controlling *M. spicatum* and employed liquid or granular formulations of 2,4-D acid ([Table S1](#)).

We asked whether aquatic plant species change more after a herbicide treatment compared with background rates of interannual change. For each species in each of the 25 treated lakes, we used Pearson's  $\chi^2$  test of independence to assess whether there was a significant pre- to post-treatment change in the population. For each lake, we counted the number of species with statistically significant increases or decreases. To describe the background rate of interannual variation, we conducted the same analysis using two randomly selected years for each untreated lake that had multiple plant surveys ( $N = 46$  lakes). We compared the observed patterns across treated and untreated systems as a measure of the effect of lake-wide herbicide treatment on aquatic plant communities.



**Fig. 1.** Map showing treated ( $n = 25$ ) and untreated ( $n = 148$ ) lakes included in the pre-post and comparative analysis. Untreated lakes in the larger dataset that were not included in either analysis are not shown. Map created using ArcMap 10.6.1. Data sources: North American States and Provinces from Tele Atlas North America, Inc. (ESRI). County Boundaries from U.S. Census Bureau's 1990 TIGER/Line files.

### Effect of herbicide treatment and *M. spicatum* effects on aquatic plants

We employed the multilevel modeling framework described by [Jackson et al. \(2012\)](#) to simultaneously evaluate the effects of environmental variables, including invasive *M. spicatum* and herbicide treatment, on native aquatic plant species and community composition. For each predictor included in the model, the fixed-effects coefficient ( $\beta$ ) reflects the overall mean response of all aquatic plant species to that predictor. The model also includes a set of random effects coefficients whereby

$SD(u)$  reflects the variance in species-specific responses to each predictor. A random effects coefficient that is high indicates a high degree of variability in species-specific response coefficients to that predictor and thus implies an effect on community composition.

We specified a multi-level model to describe aquatic plant abundance as a function of a set of predictors, selecting the form of the predictors using exploratory scatterplots. Because the number of treated lakes was small ( $N = 25$ ) relative to untreated lakes ( $N = 363$ ), we balanced the data set using a matched-set approach (Breslow and Day 1987). For each treated lake in the study, we matched five untreated lakes that were most similar to the pretreatment plant community using the Bray–Curtis dissimilarity measure computed on species presence–absence data (Bray and Curtis 1957). This allowed us to increase data coverage for individual species so we could quantify species-specific responses. The resulting set of 125 untreated control lakes matched on community similarity included lakes with ( $N = 78$ ) and without ( $N = 47$ ) populations of *M. spicatum*. Rather than select one untreated lake for each treated lake, we matched multiple untreated cases. We combined this set of 125 matched untreated lakes with the 25 treated lakes to produce a final set of 150 lakes.

Our multi-level model estimated the fixed effect of *M. spicatum* abundance on native plant abundance and community composition for all lakes, using pretreatment data when relevant. We also included a factor for the occurrence of a lake-wide herbicide treatment. Our model also accounted for environmental influences by estimating fixed effects for water clarity (Secchi depth) and its square as well as alkalinity. We estimated species-specific random slopes and their correlations for each variable. We also fit uncorrelated intercepts for species and lakes and included an observation-level random effect to account for overdispersion (extra variability as expected with ecological data) (Browne et al. 2005; Barr et al. 2013; Harrison 2015). Prior to analysis, we scaled all continuous predictors to mean zero and unit variance. We assumed a binomial error distribution for the response variable and employed a logit link function to linearize predictors. All models were fit by maximum likelihood using zero Gauss–Hermite quadrature points with the function “glmer” in lme4 version 1.1-7 and R version 3.1.2 (R Core Team 2014; Bates et al. 2015). We use likelihood ratio tests conducted on nested models with and without the term in question to report the significance of fixed and random effects. Significance tests for the random effects require testing parameters at the edge of their possible range ( $\sigma = 0$ ) which produces inflated  $p$ -values. We adjusted  $p$ -values for random effects test by dividing by 2 (Bolker et al. 2009).

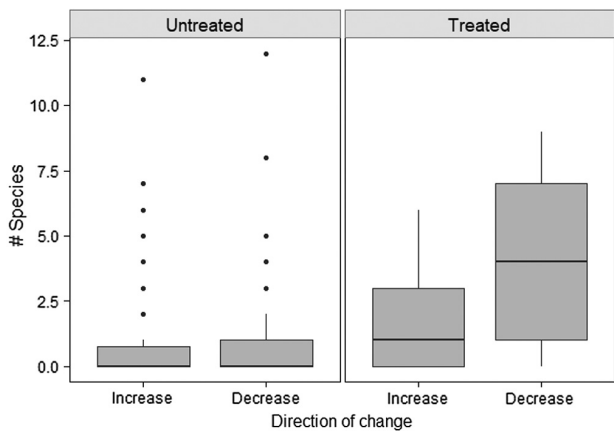
## Results

### Pre-post treatment analysis

Our comparison of aquatic plant communities before and after herbicide treatment revealed herbicide treatments are, indeed, associated with native aquatic plant species declines (Fig. 2). In the 25 lakes with pre- and post-treatment data, the mean number of species that significantly decreased per lake was greater than the number of species that significantly increased ( $\chi^2$  test, decreases:  $\bar{x} = 4.08$ ,  $SD = 3.1$ ; increases:  $\bar{x} = 1.64$ ,  $SD = 1.9$ ). For our reference group of 46 untreated lakes, the mean number of species that significantly increased and decreased across years was similar (increase:  $\bar{x} = 0.96$ ,  $SD = 2.2$ ; decrease:  $\bar{x} = 1.15$ ,  $SD = 2.4$ ). Overall, a generalized linear model fit to species counts using a quasipoisson error structure revealed that herbicide treatment was a significant predictor of the number of species decreases ( $t = -3.7$ ,  $p < 0.001$ ) but not species increases ( $t = -1.2$ ,  $p = 0.23$ ).

### Effect of treatment and *M. spicatum* on aquatic plants

Our multi-level model revealed negative coefficients for Secchi<sup>2</sup>, alkalinity, and herbicide treatment on overall native aquatic plant abundance, as indicated by significant fixed effects  $\beta$  parameters (Table 1). While there was a negative effect of lake-wide herbicide treatment on native aquatic plants



**Fig. 2.** Number of aquatic plant species demonstrating statistically significant increases and decreases between years in lakes that received a lake-wide herbicide treatment and those that did not.

**Table 1.** Coefficients describing the overall (fixed) and species-specific (random) responses to centered and standardized predictors in aquatic plant communities estimated by a multilevel generalized linear model.

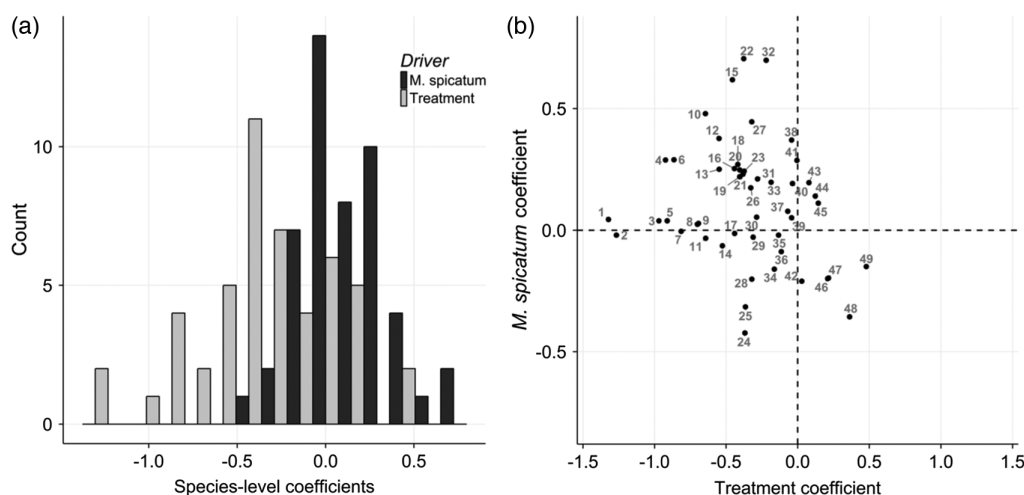
| Predictor           | Fixed effects<br>( $\beta$ , overall coefficient) | Random effects (SD( $u$ ),<br>species-specific variation) |
|---------------------|---|---|
| Intercept           | <b>−5.98</b>                                      | <b>2.18</b>   |
| Secchi              | 0.21  | <b>0.82</b>   |
| Secchi <sup>2</sup> | <b>−0.32</b>                                      | <b>0.27</b>   |
| Alkalinity          | <b>−0.54</b>                                      | <b>1.47</b>   |
| <i>M. spicatum</i>  | 0.11  | <b>0.33</b>   |
| Treatment           | <b>−0.35</b>                                      | <b>0.68</b>   |
| N                   | 7350  | —   |
| Log likelihood      | −13 032   | —   |

**Note:** Data includes aquatic plant surveys on a matched set of 150 lakes with similar communities. Significant ( $p < 0.05$ ) predictors in bold determined by likelihood ratio tests on nested models without the indicated predictor.

( $\beta = -0.35$ ,  $p = 0.047$ ), there was no significant effect of invasive *M. spicatum* on native aquatic plants ( $\beta = 0.11$ ,  $p = 0.22$ ; [Table 1](#)).

Random effects for all predictors were statistically significant, indicating that species-specific responses to predictors were highly variable. The species-specific coefficients for herbicide treatment were more variable than species-specific coefficients for *M. spicatum* (treatment: SD( $u$ ) = 0.68,  $p = 0.007$ ; *M. spicatum*: SD( $u$ ) = 0.34,  $p < 0.001$ ; [Table 1](#)). The distribution of species-level coefficients for herbicide treatment and for *M. spicatum* indicates that species-specific response to herbicide treatment tends to vary widely and is typically negative ([Fig. 3a](#)). About 82% of native aquatic plant species in the study set had a negative association with herbicide treatment (i.e., negative coefficient), whereas only 33% had a negative association with *M. spicatum*. In contrast, 67% of native aquatic plant species had a positive association with *M. spicatum* (i.e., positive coefficient; [Fig. 3b](#)).





**Fig. 3.** (a) Frequency distribution of species-specific (random) coefficients for *M. spicatum* and herbicide treatment estimated by a multilevel generalized linear model that also accounts for the fixed effects of alkalinity and water clarity in 150 lakes. (b) Biplot of species-specific coefficients shows the individual species responses to the two drivers. Species ID labels listed in [Table S2](#).

Consistent with the pre–post analysis, species with negative responses to herbicide treatment included monocotyledons and dicotyledons of a variety of growth forms. Species that were negatively associated with *M. spicatum* were often short in stature, whereas those that were associated positively generally had taller growth forms or were free-floating.

## Discussion

Our study used two complementary approaches to evaluate the ecological effects of the invasive aquatic plant *M. spicatum* and the effects of lake-wide herbicide treatments. First, using an extensive set of data on aquatic plant communities in Wisconsin lakes, a pre–post comparison revealed that native aquatic plant species exhibited more significant declines following lake-wide herbicide treatment relative to untreated lakes ([Fig. 2](#)). Pre–post comparisons are a direct and powerful approach for making inferences about ecological effects. Unfortunately, a similar pre–post comparison was not possible for *M. spicatum* invasions, since invasions are unplanned, and pre-invasion data are exceedingly rare. Thus, to complement the pre–post analysis for herbicide treatment, we conducted a second analysis using comparative multi-level modeling to statistically compare the effects of herbicide treatment and *M. spicatum* on native aquatic plant species and communities ([Jackson et al. 2012, 2014](#)). We found that lake-wide herbicide treatment was negatively associated with native aquatic plant abundance overall with the majority (82%) of individual native aquatic plant species exhibiting a negative coefficient (i.e., negative responses; [Fig. 3](#)). The highly divergent species-specific responses to herbicide treatment suggest that there is an association between lake-wide herbicide treatment and aquatic plant community composition.

*Myriophyllum spicatum* appears to have a relatively minor effect on native plant species abundance and community composition. In fact, for individual aquatic plant species, the association among *M. spicatum* and native species abundance was usually positive: 67% of species-specific *M. spicatum* coefficients in the multi-level model were positive. Our findings do not suggest that competitive displacement of native species by *M. spicatum* is strong or ubiquitous, at least at a lake-wide, cross-system scale. In communities where competition is a major structuring force, covariance among population abundances is on average expected to be negative ([Houlahan et al. 2007](#)). On the contrary,

our findings suggest that factors other than interspecific competition, like facilitation or environmental filtering, may better explain broadscale aquatic plant community patterns. Native species and *M. spicatum* may be responding in concert to environmental conditions, or *M. spicatum* and other native plants may create conditions that are mutually supportive of aquatic plant establishment and expansion.

At first glance, previous work on the effects of *M. spicatum* on native plant communities appears contradictory; evidence exists for negative, neutral, and positive effects. Upon closer examination, negative effects are often reported from local-scale studies on selected lakes or sites within lakes, whereas reports of neutral or positive relationships come from studies conducted on a larger spatial scale (Boylan et al. 1999; Trebitz and Taylor 2007; Gräfe 2014; Muthukrishnan et al. 2018). This latter explanation is consistent with our study, which failed to discern negative effects of *M. spicatum* on native aquatic plants at the lake-wide scale across the landscape.

In a meta-analysis of 199 studies on invasive plant impacts, Vilà et al. (2011) found that 86% of studies used comparative data to quantify impact, but most of those compared uninvaded sites with sites that were highly invaded. Such a comparison may not be realistic. Highly invaded sites are not necessarily representative, as studies have found that aquatic invasive species are most often present at relatively low densities (Hansen et al. 2013b). By exploring the association among the abundance of *M. spicatum* and native plant species at the lake-wide scale and in many lakes, we present a more realistic picture of the actual impact of the species on the landscape.

Quantifying invasive species effects is difficult for several reasons. Pre-invasion data are often lacking, thus making direct pre-post comparisons nearly impossible. Experimental manipulations of invasive species presence or abundance provide a solution, but these tightly controlled experiments are often expensive, impractical, and offer only a limited perspective on the full community dynamics of a lake. Comparative data sets involving multiple sites or waterbodies, such as those from governmental monitoring programs, are more readily available. Statistical approaches like the one used here provide a path toward rigorous evaluation of a comparative data set.

Quantification of recovery following invasive species control or eradication is another common approach to assessing invasive species effects, but that approach can be problematic as well: eradication is difficult to achieve, and the invaded community may never fully recover to pre-invasion or pretreatment conditions (Hansen et al. 2013a). Different approaches to understanding both invasive species effects and the effects of their management can yield conflicting results. This underscores the importance of combining multiple lines of evidence, as we do here, when attempting to evaluate or quantify invasive species impacts.

The ecological response metrics used in our assessment relate to native aquatic plant species and communities. There are many other potential response variables that could be used for comparing ecological effects of invasive species and invasive species control. For example, *M. spicatum* can change the structural geometry and composition of lake littoral habitat, alter light regimes, and influence lake biogeochemistry (Madsen et al. 1991; Barko et al. 1994). While there is little evidence that *M. spicatum* directly affects fish abundance, there is support for a significant effect on macroinvertebrates (Duffy and Baltz 1998; Kovalenko and Dibble 2011). While we failed to find evidence for *M. spicatum* effects on native plant communities, it is important to recognize other potential ecological effects of *M. spicatum*, though more work is needed to clarify magnitude and mechanism.

In contrast to the patterns observed with *M. spicatum*, lake-wide chemical treatments that are used to control this invasive aquatic plant are associated with significant negative effects on native



aquatic plant species abundance and overall aquatic plant community composition. Previous research on the ecological effects of herbicide treatments is variable: some studies report minimal effects on native aquatic plants, whereas other studies observe species declines that can be long-lasting (Kovalenko et al. 2010; Wersal et al. 2010; Nault et al. 2014). Contradictory findings may be explained by the spatial scale of treatment, water chemistry, and differences in herbicide products, rates, and exposure time (Frater et al. 2016; Nault et al. 2012, 2018). Our study uses data from many aquatic plant communities to reveal evidence that lake-wide herbicide treatments may be associated with ecological effects on nontarget species and aquatic plant communities. The paired analyses do suggest a treatment-related effect, but it is important to realize that there may be uncaptured factors that contribute to the patterns we observed. Accounting for environmental variation and matching treated lakes to untreated control lakes that had similar plant communities were two important steps that contribute to the strength of our inferences, but there may yet exist underlying causal factors common to treated lakes that may not be directly related to the lake-wide application of herbicide.

Our study associates lake-wide herbicide treatments with nontarget effects on native aquatic plants, but the timing and longevity of these effects is unknown. We should track species abundance and plant community change after lake-wide herbicide treatment for multiple years to identify whether observed ecological effects last. Unfortunately, treated lakes in this study were typically subjected to follow-up management actions after the initial treatment, which limited our ability to explore this question. We conjecture that if native species fail to recover from lake-wide herbicide treatments as quickly as *M. spicatum*, the invasive species may continue to present a management problem despite ongoing investment in control, leading to synergistic negative effects on native species (Rinella et al. 2009). In light of our findings, we recommend an adaptive, integrated, pest-management approach that utilizes diverse strategies to achieve management goals, especially given that some commonly utilized aquatic herbicides (i.e., 2,4-D, fluridone) have been associated with milfoil hybridization events and increased herbicide resistance (Thum et al. 2012; Larue et al. 2013; Berger et al. 2015; Gill and Goyal 2016).

In conclusion, whether the lake-specific effects of the invasive species are adverse and severe enough to justify the risk posed by herbicide treatment deserves much more careful consideration than has occurred in the past. Lake management decisions must consider diverse stakeholder values and ecological health, and our work provides insights that may be incorporated into aquatic plant management decision-making frameworks (Kumschick et al. 2012). We conclude that unless there is strong evidence of high ecological, social, or economic impact for an invasive aquatic plant, aggressive chemical control at a lake-wide scale might do more harm than good.

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## Author contributions

AM, JH, and MJVZ conceived and designed the study. AM, EK, MEN, SVE, KIW, MB, JH, and MJVZ performed the experiments/collected the data. AM and MJVZ analyzed and interpreted the data. AM, EK, MEN, SVE, KIW, MB, JH, and MJVZ drafted or revised the manuscript.

## Competing interests

The authors have declared that no competing interests exist.

## Data availability statement

All relevant data are within the paper and Supplementary Material.

## Supplementary materials

The following Supplementary Material is available with the article through the journal website at doi:[10.1139/facets-2020-0002](https://doi.org/10.1139/facets-2020-0002).

Supplementary Material 1

Supplementary Material 2

## References

- Barko JW, Smith CS, and Chambers PA. 1994. Perspectives on submersed macrophyte invasions and declines. *Lake and Reservoir Management*, 10(1): 1–3. DOI: [10.1080/07438149409354163](https://doi.org/10.1080/07438149409354163)
- Barr DJ, Levy R, Scheepers C, and Tily HJ. 2013. Random effects structure for confirmatory hypothesis testing: keep it maximal. *Journal of Memory and Language*, 68(3): 255–278. PMID: [24403724](https://pubmed.ncbi.nlm.nih.gov/24403724/) DOI: [10.1016/j.jml.2012.11.001](https://doi.org/10.1016/j.jml.2012.11.001)
- Bates D, Maechler M, Bolker BM, and Walker S. 2015. Fitting linear mixed-effects models using lme4. *Journal of Statistical Software*, 67: 1–48. DOI: [10.18637/jss.v067.i01](https://doi.org/10.18637/jss.v067.i01)
- Berger ST, Netherland MD, and Macdonald GE. 2015. Laboratory documentation of multiple-herbicide tolerance to fluridone, norflurazon, and topiramazone in a hybrid watermilfoil (*Myriophyllum spicatum* × *M. sibiricum*) population. *Weed Science*, 63(1): 235–241. DOI: [10.1614/WS-D-14-00085.1](https://doi.org/10.1614/WS-D-14-00085.1)
- Bergstrom DM, Lucieer A, Kiefer K, Wasley J, Belbin L, Pedersen TK, and Chown SL. 2009. Indirect effects of invasive species removal devastate world heritage island. *Journal of Applied Ecology*, 46(1): 73–81. DOI: [10.1111/j.1365-2664.2008.01601.x](https://doi.org/10.1111/j.1365-2664.2008.01601.x)
- Bolker BM, Brooks ME, Clark CJ, Geange SW, Poulsen JR, Stevens MH, and White JS. 2009. Generalized linear mixed models: a practical guide for ecology and evolution. *Trends in Ecology & Evolution*, 24(3): 127–135. PMID: [19185386](https://pubmed.ncbi.nlm.nih.gov/19185386/) DOI: [10.1016/j.tree.2008.10.008](https://doi.org/10.1016/j.tree.2008.10.008)
- Boylan CW, Eichler LW, and Madsen JD. 1999. Loss of native aquatic plant species in a community dominated by Eurasian watermilfoil. *Hydrobiologia*, 415: 207–211. DOI: [10.1023/A:1003804612998](https://doi.org/10.1023/A:1003804612998)
- Bray JR, and Curtis JT. 1957. An ordination of upland forest communities of southern Wisconsin. *Ecological Monographs*, 27: 325–349. DOI: [10.2307/1942268](https://doi.org/10.2307/1942268)

Breslow NE, and Day NE. 1987. Statistical methods in cancer research. Vol. 2. International Agency for Research on Cancer, Lyon, France.

Browne WJ, Subramanian SV, Jones K, and Goldstein H. 2005. Variance partitioning in multilevel logistic models that exhibit overdispersion. *Journal of the Royal Statistical Society: Series A (Statistics in Society)*, 168(3): 599–613. DOI: [10.1111/j.1467-985X.2004.00365.x](https://doi.org/10.1111/j.1467-985X.2004.00365.x)

Crow GE, and Hellquist CB 2000a. Aquatic and wetland plants of northeastern North America. Vol. 1. Pteridophytes, gymnosperms and angiosperms: dicotyledons. University of Wisconsin Press, Madison, Wisconsin.

Crow GE, and Hellquist CB 2000b. Aquatic and wetland plants of northeastern North America. Vol. 2. *In* Angiosperms: monocotyledon. University of Wisconsin Press, Madison, Wisconsin.

Duffy KC, and Baltz DM. 1998. Comparison of fish assemblages associated with native and exotic submerged macrophytes in the Lake Pontchartrain estuary, USA. *Journal of Experimental Marine Biology and Ecology*, 223(2): 199–221. DOI: [10.1016/S0022-0981\(97\)00166-4](https://doi.org/10.1016/S0022-0981(97)00166-4)

Ehrenfeld JG. 2010. Ecosystem consequences of biological invasions. *Annual Review of Ecology, Evolution, and Systematics*, 41: 59–80. DOI: [10.1146/annurev-ecolsys-102209-144650](https://doi.org/10.1146/annurev-ecolsys-102209-144650)

Eiswerth ME, Donaldson SG, and Johnson WS. 2000. Potential environmental impacts and economic damages of Eurasian watermilfoil (*Myriophyllum spicatum*) in western Nevada and northeastern California. *Weed Technology*, 14(3): 511–518. DOI: [10.1614/0890-037X\(2000\)014\[0511:PEIAED\]2.0.CO;2](https://doi.org/10.1614/0890-037X(2000)014[0511:PEIAED]2.0.CO;2)

Frater P, Mikulyuk A, Barton M, Nault M, Wagner K, Hauxwell J, and Kujawa E. 2016. Relationships between water chemistry and herbicide efficacy of Eurasian watermilfoil management in Wisconsin lakes. *Lake and Reservoir Management*, 33(1): 1–7. DOI: [10.1080/10402381.2016.1235634](https://doi.org/10.1080/10402381.2016.1235634)

Gill HG, and Goyal G. 2016. Integrated pest management (IPM): environmentally sound pest management. INTECH, Rijeka, Croatia.

Gräfe S. 2014. Relationship between the invasive eurasian milfoil (*Myriophyllum spicatum* L.) and macrophyte diversity across spatial scales. M.Sc. thesis, University of Ottawa, Ottawa, Ontario.

Hansen GJA, Ives AR, Vander Zanden MJ, and Carpenter SR 2013a. Are rapid transitions between invasive and native species caused by alternative stable states, and does it matter? *Ecology*, 94(10): 2207–2219. PMID: [24358707](https://pubmed.ncbi.nlm.nih.gov/24358707/) DOI: [10.1890/13-0093.1](https://doi.org/10.1890/13-0093.1)

Hansen GJA, Vander Zanden MJ, Blum MJ, Clayton MK, Hain EF, Hauxwell J, et al. 2013b. Commonly rare and rarely common: comparing population abundance of invasive and native aquatic species. *PLoS ONE*, 8(10): e77415. PMID: [24194883](https://pubmed.ncbi.nlm.nih.gov/24194883/) DOI: [10.1371/journal.pone.0077415](https://doi.org/10.1371/journal.pone.0077415)

Harrison XA. 2015. A comparison of observation-level random effect and beta-binomial models for modelling overdispersion in binomial data in ecology & evolution. *PeerJ*, 3: e1114. PMID: [26244118](https://pubmed.ncbi.nlm.nih.gov/26244118/) DOI: [10.7717/peerj.1114](https://doi.org/10.7717/peerj.1114)

Hauxwell J, Knight S, Mikulyuk A, Nault ME, Porzky M, and Chase S. 2010. Recommended baseline monitoring of aquatic plant in Wisconsin: sampling design, field and laboratory procedures, data entry and analysis, and applications. Wisconsin Department of Natural Resources PUB-SS-1068 2010, Madison, Wisconsin.

- Horsch EJ, and Lewis DJ. 2009. The effects of aquatic invasive species on property values: evidence from a quasi-experiment. *Land Economics*, 85(3): 391–409. DOI: [10.3368/le.85.3.391](https://doi.org/10.3368/le.85.3.391)
- Houlahan JE, Currie DJ, Cottenie K, Cumming GS, Ernest SKM, Findlay CS, et al. 2007. Compensatory dynamics are rare in natural ecological communities. *Proceedings of the National Academy of Sciences of the United States of America*, 104(9): 3273–3277. PMID: [17360637](https://pubmed.ncbi.nlm.nih.gov/17360637/) DOI: [10.1073/pnas.0603798104](https://doi.org/10.1073/pnas.0603798104)
- Jackson MM, Turner MG, Pearson SM, and Ives AR. 2012. Seeing the forest and the trees: multilevel models reveal both species and community patterns. *Ecosphere*, 3(9): 1–16. DOI: [10.1890/ES12-00116.1](https://doi.org/10.1890/ES12-00116.1)
- Jackson MM, Turner MG, and Pearson SM. 2014. Logging legacies affect insect pollinator communities in southern Appalachian forests. *Southeastern Naturalist*, 13(2): 317–336. DOI: [10.1656/058.013.0213](https://doi.org/10.1656/058.013.0213)
- Kovalenko KE, and Dibble ED. 2011. Effects of invasive macrophyte on trophic diversity and position of secondary consumers. *Hydrobiologia*, 663(1): 167–173. DOI: [10.1007/s10750-010-0570-7](https://doi.org/10.1007/s10750-010-0570-7)
- Kovalenko KE, Dibble ED, and Slade JG. 2010. Community effects of invasive macrophyte control: role of invasive plant abundance and habitat complexity. *Journal of Applied Ecology*, 47(2): 318–328. DOI: [10.1111/j.1365-2664.2009.01768.x](https://doi.org/10.1111/j.1365-2664.2009.01768.x)
- Kujawa ER, Frater P, Mikulyuk A, Barton M, Nault ME, Egeren SV, and Hauxwell J. 2017. Lessons from a decade of lake management: effects of herbicides on Eurasian watermilfoil and native plant communities. *Ecosphere*, 8(4): e01718. DOI: [10.1002/ecs2.1718](https://doi.org/10.1002/ecs2.1718)
- Kumschick S, Bacher S, Dawson W, Heikkilä J, Sendek A, Pluess T, et al. 2012. A conceptual framework for prioritization of invasive alien species for management according to their impact. *NeoBiota*, 15: 69–100. DOI: [10.3897/neobiota.15.3323](https://doi.org/10.3897/neobiota.15.3323)
- Larue EA, Zuellig MP, Netherland MD, Heilman MA, and Thum RA. 2013. Hybrid watermilfoil lineages are more invasive and less sensitive to a commonly used herbicide than their exotic parent (*Eurasian watermilfoil*). *Evolutionary Applications*, 6(3): 462–471. PMID: [23745138](https://pubmed.ncbi.nlm.nih.gov/23745138/) DOI: [10.1111/eva.12027](https://doi.org/10.1111/eva.12027)
- Little RJA. 1988. Missing-data adjustments in large surveys. *Journal of Business & Economic Statistics*, 6(3): 287–296. DOI: [10.1080/07350015.1988.10509663](https://doi.org/10.1080/07350015.1988.10509663)
- Lu X, Siemann E, He M, Wei H, Shao X, and Ding J. 2015. Climate warming increases biological control agent impact on a non-target species. *Ecology Letters*, 18(1): 48–56. PMID: [25376303](https://pubmed.ncbi.nlm.nih.gov/25376303/) DOI: [10.1111/ele.12391](https://doi.org/10.1111/ele.12391)
- Madsen JD, Sutherland JW, Bloomfield JA, Eichler LW, and Boylen CW. 1991. The decline of native vegetation under dense Eurasian watermilfoil canopies. *Journal of Aquatic Plant Management*, 29: 94–99.
- Mikulyuk A, Hauxwell J, Rasmussen P, Knight S, Wagner KI, Nault ME, and Ridgely D. 2010. Testing a methodology for assessing plant communities in temperate inland lakes. *Lake and Reservoir Management*, 26(1): 54–62. DOI: [10.1080/07438141003666848](https://doi.org/10.1080/07438141003666848)
- Mikulyuk A, Sharma S, Van Egeren S, Erdmann E, Nault Michelle E, and Hauxwell J. 2011. The relative role of environmental, spatial, and land-use patterns in explaining aquatic macrophyte

community composition. Canadian Journal of Fisheries and Aquatic Sciences, 68(10): 1778–1789. DOI: [10.1139/f2011-095](https://doi.org/10.1139/f2011-095)

Muthukrishnan R, Hansel-Welch N, and Larkin DJ. 2018. Environmental filtering and competitive exclusion drive biodiversity-invasibility relationships in shallow lake plant communities. Journal of Ecology, 106(5): 2058–2070. DOI: [10.1111/1365-2745.12963](https://doi.org/10.1111/1365-2745.12963)

Nault ME, Mikulyuk A, Hauxwell J, Skogerboe JD, Asplund T, Barton M, et al. 2012. Herbicide treatments in Wisconsin lakes. Lakeline, Spring: 21–26.

Nault ME, Netherland MD, Mikulyuk A, Skogerboe JG, Asplund T, Hauxwell J, and Toshner P. 2014. Efficacy, selectivity, and herbicide concentrations following a whole-lake 2,4-D application targeting Eurasian watermilfoil in two adjacent northern Wisconsin lakes. Lake and Reservoir Management, 30(1): 1–10. DOI: [10.1080/10402381.2013.862586](https://doi.org/10.1080/10402381.2013.862586)

Nault ME, Barton M, Hauxwell J, Heath E, Hoyman T, Mikulyuk A, et al. 2018. Evaluation of large-scale low-concentration 2,4-D treatments for Eurasian and hybrid watermilfoil control across multiple Wisconsin lakes. Lake and Reservoir Management, 34(2): 115–129. DOI: [10.1080/10402381.2017.1390019](https://doi.org/10.1080/10402381.2017.1390019)

Olden JD, and Tamayo M. 2014. Incentivizing the public to support invasive species management: eurasian milfoil reduces lakefront property values. PLoS ONE, 9(10): e110458. PMID: [25333619](https://pubmed.ncbi.nlm.nih.gov/25333619/) DOI: [10.1371/journal.pone.0110458](https://doi.org/10.1371/journal.pone.0110458)

Papeş M, Sallstrom M, Asplund TR, and Vander Zanden MJ. 2011. Invasive species research to meet the needs of resource management and planning. Conservation Biology, 25: 867–872. DOI: [10.1111/j.1523-1739.2011.01714.x](https://doi.org/10.1111/j.1523-1739.2011.01714.x)

Provencher B, Lewis DJ, and Anderson K. 2012. Disentangling preferences and expectations in stated preference analysis with respondent uncertainty: the case of invasive species prevention. Journal of Environmental Economics and Management, 64(2): 169–182. DOI: [10.1016/j.jeem.2012.04.002](https://doi.org/10.1016/j.jeem.2012.04.002)

R Core Team. 2014. R: a language and environment for statistical computing. R Foundation for Statistical Computing, Vienna, Austria.

Ricciardi A. 2007. Are modern biological invasions an unprecedented form of global change? Conservation Biology, 21: 329–336. PMID: [17391183](https://pubmed.ncbi.nlm.nih.gov/17391183/) DOI: [10.1111/j.1523-1739.2006.00615.x](https://doi.org/10.1111/j.1523-1739.2006.00615.x)

Rinella MJ, Maxwell BD, Fay PK, Weaver T, and Sheley RL. 2009. Control effort exacerbates invasive-species problem. Ecological Applications, 19(1): 155–162. PMID: [19323180](https://pubmed.ncbi.nlm.nih.gov/19323180/) DOI: [10.1890/07-1482.1](https://doi.org/10.1890/07-1482.1)

Simberloff D. 2009. We can eliminate invasions or live with them. Successful management projects. Biological Invasions, 11(1): 149–157. DOI: [10.1007/s10530-008-9317-z](https://doi.org/10.1007/s10530-008-9317-z)

Thum RA, Heilman M, Hausler PJ, Huberty LE, Tynning P, Wcisel DJ, et al. 2012. Field and laboratory documentation of reduced fluridone sensitivity of a hybrid watermilfoil biotype (*Myriophyllum spicatum* × *Myriophyllum sibiricum*). Journal of Aquatic Plant Management, 50: 141–146.

Trebitz AS, and Taylor DL. 2007. Exotic and invasive aquatic plants in Great Lakes coastal wetlands: distribution and relation to watershed land use and plant richness and cover. Journal of Great Lakes Research, 33(4): 705–721. DOI: [10.3394/0380-1330\(2007\)33\[705:EAIAP\]2.0.CO;2](https://doi.org/10.3394/0380-1330(2007)33[705:EAIAP]2.0.CO;2)

Vestergaard O, and Sand-Jensen K. 2000. Alkalinity and trophic state regulate aquatic plant distribution in Danish lakes. *Aquatic Botany*, 67(2): 85–107. DOI: [10.1016/S0304-3770\(00\)00086-3](https://doi.org/10.1016/S0304-3770(00)00086-3)

Vilà M, Espinar JL, Hejda M, Hulme PE, Jarošík V, Maron JL, et al. 2011. Ecological impacts of invasive alien plants: a meta-analysis of their effects on species, communities and ecosystems. *Ecology Letters*, 14(7): 702–708. PMID: [21592274](https://pubmed.ncbi.nlm.nih.gov/21592274/) DOI: [10.1111/j.1461-0248.2011.01628.x](https://doi.org/10.1111/j.1461-0248.2011.01628.x)

Wagner KI, Hauxwell J, Rasmussen PW, Koshere F, Toshner P, Aron K, et al. 2007. Whole-lake herbicide treatments for eurasian watermilfoil in four Wisconsin lakes: effects on vegetation and water clarity. *Lake and Reservoir Management*, 23(1): 83–94. DOI: [10.1080/07438140709353912](https://doi.org/10.1080/07438140709353912)

Wersal RM, Madsen JD, Woolf TE, and Eckberg N. 2010. Assessment of herbicide efficacy on Eurasian watermilfoil and impacts to the native submersed plant community in Hayden Lake, Idaho, USA. *Journal of Aquatic Plant Management*, 48: 5–11.

Zavaleta ES, Hobbs RJ, and Mooney HA. 2001. Viewing invasive species removal in a whole-ecosystem context. *Trends in Ecology & Evolution*, 16(8): 454–459. DOI: [10.1016/S0169-5347\(01\)02194-2](https://doi.org/10.1016/S0169-5347(01)02194-2)