Long-term periodic management of Phragmites australis maintains native brackish wetland plant communities

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Abstract

Complete eradication of invasive plants is often infeasible while in some cases 'functional eradication', the reduction of an invader to low levels with reduced ongoing management costs, is a sustainable option. Non-native *Phragmites australis* has challenged land managers across North America but functional eradication may yet be possible in some scenarios. Here we present data from Chesapeake Bay brackish tidal wetlands where two approaches to *Phragmites* management (long-term, continuous management and short-term, non-continuous management) were used. We demonstrate that the application of herbicides will lead to the establishment of native species, but long-term, continuous management is required to facilitate functional eradication of *Phragmites* by keeping it at low levels of occurrence and restoring native plant communities. Ultimately, historical data from sites that were sampled as part of this study indicate that if management is successful and results in functional eradication of *Phragmites*, then the recovering vegetation will include native species that were present before the sites were invaded by *Phragmites*.

Introduction

Considerations of whether to manage a particular invasive species are complex and often involve factors that are both ecological and social (e.g., costs). One important decision managers face is whether to remove an invasive species completely or partially. Green and Grosholz (2021) suggested that complete eradication may not be necessary if the reduction of invasive species to low populations levels reduces management costs, while also mitigating the ecological effects of the invader, an approach they referred to as 'functional eradication'. Functional eradication may be especially important in situations where complete eradication of an invasive species is not possible or would require resources over long periods of time that would be difficult or impossible to sustain. A non-native lineage of *Phragmites australis* (hereafter referred to as *Phragmites*) is an example of a widespread invasive species in North America where functional eradication should be considered during the process of making management decisions.

Several authors have addressed the complexities associated with the management of *Phragmites*, a widespread invasive species in Chesapeake Bay tidal wetlands (Chambers et al. 1999). *Phragmites* invasion, however, is not limited to the Chesapeake Bay (e.g., Whyte et al. 2008; Meyerson et al. 2010; Kettenring et al. 2012; Lambert et al. 2016), and studies have been conducted to develop best management practices for controlling or eradicating it. Reviews of management (Martin and Blossey 2013; Hazelton et al. 2014) have demonstrated that a one-time attempt to eradicate *Phragmites* is insufficient, and multi-year management are required but few studies have been evaluated the long-term consequences of either short or continuous, long-term *Phragmites* management.

There are numerous negative ecological consequences of *Phragmites* invasion of wetlands (Uddin et al. 2017). Invertebrate communities change (e.g., Gratton and Denno 2005) and the dynamics of Mummichog (*Fundulus heteroclitus*, an important estuarine fish) have been shown to be negatively impacted (e.g., Able and Hagan 2003). Another major consequence of *Phragmites* colonization is that native plant species diversity and abundance decrease (Meyerson et al. 2000; Whyte et al. 2008; Uddin and Robinson 2017) as light becomes limiting in tall and dense stands of *Phragmites*, and allelopathy and the development of a dense litter layer also have negative impacts on native plant species (e.g., Holdredge and Bertness 2011; Uddin et al. 2017; Uddin and Robinson 2017; Rohal et al. 2019b).

What happens when *Phragmites* is successfully eliminated or reached the point of functional eradication, as suggested by Green and Grosholz (2021)? Does a native plant community develop and, if it does, is the composition of the plant community the same as it was prior to the invasion of *Phragmites*? The question of what happens following *Phragmites* removal has been addressed following non-chemical control (e.g., Warren et al. 2002) and herbicide or mechanical management (Faison et al. 2020), and two patterns emerge. First, if management ceases and *Phragmites* has not been completely removed, sites can become dominated by *Phragmites* again (Farnsworth and Meyerson 1999; Martin and Blossey 2013; Hazelton et al. 2014; Rohal et al. 2019b, 2021, 2023). Second, if management is long-term and continuous or sufficiently removes *Phragmites*, vegetation recovery occurs, even if small amounts of *Phragmites* remains (Lombard et al. 2012), a potential example of functional eradication. The question of whether the restored plant community is the same as the community that was present prior to *Phragmites* invasion has not been addressed, as most studies have focused on the plant community that is present following management. In a few instances, comparisons have been made between the plant community that recovered following *Phragmites* removal and a nearby community dominated by native wetland species and no *Phragmites* (Zimmerman et al. 2018; Rohal et al. 2019b, 2023).

Zimmerman et al. (2018) chemically treated *Phragmites* stands for three years in a Hudson River freshwater tidal wetland. They found that vegetation recovery in the treated areas was floristically similar to nearby control areas, but the size of the area treated was important, as larger areas were still missing ‘species characteristic of uninvaded habitat’. Rohal et al. (2019a) evaluated six types of *Phragmites* treatments in Great Salt Lake wetlands (Utah) and found that the recovering plant communities had the same species as reference sites, but recovery was slow, especially for annuals, most likely due to the presence of a dense litter layer following management. In a Chesapeake Bay study, Rohal et al. (2023) found similar results in a study of eight subestuaries that included three years of chemical treatment of *Phragmites*, followed by an additional two years of monitoring plots in treated areas and nearby areas with no *Phragmite*s (i.e., reference sites with no *Phragmites*). Like the Utah study, the recovering plant community had species that were in the reference sites, but the species composition of the recovering vegetation differed from reference sites. Collectively, these studies indicate that native vegetation recovery may take a long time. However, even if *Phragmites* does not reinvade, it is unclear if the species composition will eventually resemble the plant community that was present prior to *Phragmites* colonization.

We conducted a study in which we sampled plant communities and compiled plant community data from other studies in tidal brackish wetlands dominated by herbaceous plants (i.e., tidal brackish marshes) following *Phragmites* management and compared them to plant communities that were present before *Phragmites* invasion. We refer to this as a “retrospective study” because we were able to look backward in time to determine if the plant communities that are currently present had the same species that were also present many years ago. A retrospective approach was possible because the State of Maryland mapped plant communities in tidal wetlands in the Chesapeake Bay more than 50 years ago (McCormick and Somes 1982). The Maryland project, based on analyses of aerial photographs in combination with field sampling, resulted in the McCormick and Somes (1982) publication that characterized the dominant species composition of tidal wetland plant communities and documented the aerial extent of each community in Maryland counties that bordered the Chesapeake Bay. The project also resulted in a set of high-quality black and white maps, now available digitally, at a scale of 1:2,400 that showed the distribution of the plant communities in Maryland's tidal wetlands in 1972.

McCormick and Somes (1982) estimated that *Phragmites* covered an area of 690 Ha or 0.75% of the total wetland area mapped. Since that time, *Phragmites* coverage has expanded dramatically. In the Rhode River subestuary, for example, McCormick et al. (2010) found that 5 *Phragmites* patches mapped in 1972 had increased to 212 identifiable patches by 2007, and the area that the patches covered had increased 25×. Our goal in this project was to sample vegetation in wetlands that had been invaded by *Phragmites* and subsequently had been treated with herbicides for different periods of time (some had long-term, continuous management, and some only had short-term management) and to compare the plant communities that were currently present with plant communities mapped by McCormick and Somes (1982). Our hypothesis was that, given the relatively small number of species that dominate brackish tidal wetlands in Maryland tidal wetlands, plant communities that are present today, following *Phragmites* functional eradication, would be like the plant communities that were present more than 50 years ago. We also hypothesized that sites that had not been managed continuously would be characterized by a reinvasion of *Phragmites* and lower plant diversity.

Methods

We developed three separate data sets for analysis. The first was obtained by sampling 17 brackish tidal wetland sites where *Phragmites* has been sprayed with herbicides (hereafter referred to as *Continuous Management*) by Chesapeake Wildlife Heritage (CWH) for the past 3–22 years (Phillips Boyd, personal communication), and one additional site that was sprayed in 2105, 2019 and 2022 for homeowners by a private consultant (Cathy Oliver, personal communication). Wetlands managed by CWH are on the Eastern Shore of the Chesapeake Bay in Kent and Queen Anne's Counties. The other site is on the Western Shore of Chesapeake Bay in Anne Arundel County. The second data set was based on field sampling of 15 sites where *Phragmites* had been chemically sprayed from 2016 to 2018 (hereafter referred to as *Short-term Management*) by the Anne Arundel County Department of Recreation and Parks (Chris Carroll, personal communication); all sites were in Anne Arundel County. At the 32 sites sampled in the *Continuous* and *Short-term Management* sites, we randomly sampled three locations at each site by estimating the cover of species in 1 X 1 m plots (methods described below).

The third data set was developed from data collected at sites where vegetation has been monitored in permanent 1 × 1 m plots from 2011–2022 (Hereafter referred to as *Monitored*). Wetland sites in the

Monitored data set were in St Mary's, Calvert, and Anne Arundel Counties on the Western Shore and Talbot County on the Eastern Shore of Chesapeake Bay. The project that was the source of the *Monitored* data from 2011–2015 is described in Rohal et al. (2023). Following completion of the project in 2015, 6 sites were integrated into a long-term wetland monitoring program (MarineGeo: https://marinegeo.si.edu/network/upper-chesapeake-bay) maintained by the Smithsonian Environmental Research Center. Two authors (DW, HB) were part of the teams that sampled the sites during the original project and MarineGeo program. As described in Rohal et al. (2023) and continued in the MarineGeo program, vegetation (percent cover of all species) was monitored in randomly selected permanent 1 × 1 m plots at three sites: one with native vegetation (hereafter referred to as '*Native*'), one dominated by *Phragmites* and treated with herbicides in 2011, 2012, and 2013 (hereafter referred to as '*Spray*'), and one dominated by *Phragmites* that served as a control (*Phragmites* present and not managed - hereafter referred to as '*No-Spray*').

Field sampling of the Continuous and Short-term Management sites

In September and October 2022, we sampled plant communities in 32 individual tidal wetland sites where *Phragmites* had been treated using herbicides. As described above, 15 sites were part of the *Continuous Management* dataset and 17 were part of the *Short-term Management* dataset. At each site, three 1 X 1m sampling plots were randomly established and were GPS located using the US Topo Maps phone application (ATLOGIS Geoinformatics Gmbh & Co. KG, 2021) on a Google Pixel 4 and Google Pixel 7 Pro phone. Percent cover of species in each plot was visually estimated using a six-point scale based on the Braun-Blanquet method: 1 = trace, 2 = 1–5%, 3 = 5–25%, 4 = 25–50%, 5 = 50–75%, 6 = 75–100% (Furman et al. 2018). The mid-point of each cover class was used for data analysis.

Historical data linked to the three datasets

Historical vegetation types at each site were identified by linking GPS data that we had collected at the sites with the 1972 Maryland Wetland Maps (https://geodata.md.gov/imap/rest/services/Hydrology/MD_WetlandMaps1972/MapServer). The maps provided codes for the wetland plant communities present at the sites based on interpretation of natural-color stereoscopic aerial photographs. The dominant species associated with the codes for each plant community are described in McCormick and Somes (1982). McCormick and Somes reported that the plant communities were mapped from aerial photographs and verification was based on field-sampling.

Data analyses

We used Nonmetric Multidimensional Scaling (NMDS) ordination to visually compare the differences between modern plant communities in the 32 sites and 93 individual sampling plots within the *Continuous* and *Short-term Management* dataset. The ordination was conducted using the metaMDS function in the *vegan* package (Oksanen 2022) of R version 4.2.1 (2022-06-23 – "Funny-Looking Kid"). Distances were calculated using the Bray-Curtis dissimilarity calculation. The ordination of the percent cover of modern vegetation at sites in the two *Management* data sets yielded 2 dimensions and had a stress of 0.170. A second NMDS ordination compared vegetation present in the 1970s with current vegetation at sites in the two *Management* data sets (the method used to compile the historical and modern data are described below). Ordination of the historical and modern vegetation in the two *Management* data sets was conducted as described above and yielded 2 dimensions with a stress of 0.0912.

The comparison of historical and modern plant communities in the *Management* sites was based on a comparison of the dominant (1972) and most abundant (2022) species present. As noted above, McCormick and Somes (1982) interpreted natural-color stereoscopic aerial photographs to identify wetland plant communities and give them a community code, but there were no actual data on the species present in 1972 at each of the random sites sampled in 2022. By collecting GPS data for each plot that was sampled, we were able to link the location of the sampling sites to the Maryland digital tidal wetland maps. The community codes were marked on the maps (see Supplemental Table 1 for a list of the plant communities that were present in 1972). From the codes, we compiled a list of the dominant species (Supplemental Table 1 – Part A) that were most likely preset in the 1970s based on descriptions in McCormick and Somes (1982).

The brackish high marsh and brackish low marsh plant communities described by McCormick and Somes (1982) contained 12 plant species (Supplemental Table 1). The *Continuous* and *Short-term Management* sites had 33 plant species when sampled in 2022. To ensure that comparisons between historical and modern plant communities in the *Short-term Management* and *Continuous Management* datasets were not biased by the additional plant species identified in modern surveys, we created a simplified dataset which only included species described by McCormick and Somes (1982) for each community type and the four species that had the highest percent cover in 2022. We removed a single plot (Barnstable, Site 27, Plot 1) from the historical and modern *Management* dataset ordination because historically it was the only plot of those in the study that contained *Juncus roemerianus*. When the plot containing *J. roemerianus* was included in the ordination it was plotted as disjunct from the majority of NMDS coordinates for the other plots, obscuring the main community differences between *Continuous* and *Short-term Management.* In 2022, *J. roemerianus* was present at Barnstable in a sizeable monoculture, but our randomly selected modern sampling locations did not include plots in the monoculture.

To numerically compare the differences between historical and modern plant communities in the *Continuous* and *Short-term Management* dataset, we conducted a PERMANOVA analysis using the adonis2 function (Bray-Curtis distances, 999 permutations) in the vegan package (Oksanen 2022) of R version 4.2.1 (2022-06-23 – “Funny-Looking Kid"). We also compared total species richness in the *Management* datasets using one-way ANOVAs calculated in base R. Assumptions of normality were checked in base R by using Shapiro and Wilk’s (1965) *W* statistic modified for computer programs (Royston 1982) and sample sizes up to n = 200, and assumptions of homogeneity of variance were tested using Levene’s (1960) Test. Species richness (W = 0.78864, p-value = 2.026e-08) did not have normal distribution, but Levene's Test indicated homogeneity of variance (F-value = 0.0506, p-value = 0.8226). Per Blanca et al. (2017), one-way ANOVAs are robust against non-normal data, if variances are homogeneous; thus, we feel that our use of ANOVAs are not contraindicated. *Phragmites* cover (W = 0.78938, p-value = 3.273e-10) did not have normal distribution, and Levene's Test indicated a lack of homogeneity of variance (F-value = 26.572, p-value = 1.469e-06). In instances where variances are not homogeneous, like with our *Phragmites* cover data, Welch's one-way ANOVAs are robust against both non-normal data and non-homogeneous variances (Delacre et al. 2017). Thus, we compared *Phragmites* cover in the *Management* datasets using Welch's one-way ANOVA.

To clarify how plant communities differ during three years of *Phragmites* management, we conducted ANOVAs to compare total species richness in the *Monitored* dataset for the 2011 (before spraying *Phragmites*), 2014 (immediately after three years of spraying *Phragmites*), and 2020 (6 years after cessation of spraying *Phragmites*) samplings of all of the sites and plots described by Rohal et al. (2023) for 2011 and 2014 and 6 of the sites and 83 of the plots described by Rohal et al. (2023). For the 2011 and 2014 *Monitored* dataset species richness did not meet assumptions of normality but did meet assumptions of homogeneity of variance; whereas, *Phragmites* cover did not meet assumptions of normality and had heterogeneous variances. Thus, species richness was analyzed using ANOVAs, and *Phragmites* cover was analyzed using Welch's one-way ANOVAs. The 2020 dataset did not have a normal distribution of species richness and homogeneous variances for both species richness and *Phragmites* cover; therefore, we analyzed both species richness and *Phragmites* cover using Welch's one-way ANOVA.

Finally, we identified whether modern plant communities contained a similar or different suite of species as found in the historical imagery from McCormick and Somes (1982). For the comparison of historical versus modern plant communities, we used a Chi-square analysis conducted in base R to determine whether the vegetation sampled in 2022, based on the most abundant species (Supplemental Table 1), would be classified as similar or different from what was present in the 1970s and how sites differed based upon whether they received *Continuous* or *Short-term Management.*

Results

Ordination of cover data collected at the *Continuous* and *Short-term Management* sites in 2022 (Fig. 1) indicate that the plant communities were similar (i.e., the confidence intervals

around the two centroids overlapped), but further evaluation of the two data sets provide insight into differences between the datasets that may be functionally important. The *Continuous Management* sites had significantly ($F = 7.705$, $p < 0.01$) higher species richness and significantly lower ($F = 20.0$ $p < 0.01$) *Phragmites* cover than the *Short-term Management* sites (Fig. 2). Differences in the species richness at the *Continuous Management* versus *Short-term Management* sites were relatively small (3.1 ± 0.2 species m^{-2} versus 2.3 ± 0.2 species m^{-2}, respectively), compared to an almost 3× differences in *Phragmites* cover (15.8 ± 0.2% compared to 45.6 ± 0.29%, respectively).

Changes in the percent cover of *Phragmites* and species richness in the *Monitored* sites provided further insight into what can be expected when management is not long-term and continuous (Fig. 3). There were statistically significant differences in species richness among the two *Phragmites* (*Spray*, *No-Spray*) sites and the nearby sites with native vegetation in 2011 (F = 9.231, p < 0.01), 2014 (F = 5.946, p < 0.01), and 2020 (F = 10.5, p < 0.01). There were statistically significant differences in *Phragmites* cover among the two *Phragmites* (*Spray*, *No-Spray*) sites and the nearby sites with native vegetation in 2011 (F = 136, p < 0.01), 2014 (F = 44.8, p < 0.01), and 2020 (F = 36.9, p < 0.01). The *Spray* and *No-Spray* sites with *Phragmites* in

the pre-treatment year (2011) had lower species richness and a higher percent cover of *Phragmites* compared to the sites with native vegetation (Fig. 3). By 2020, the percent cover of *Phragmites* at the *Spray* sites had increased to 59.6 ± 5.1%, which was slightly higher than the mean value for the *No-Spray* sites, and almost an order of magnitude higher than the plots in *Native* vegetation (Fig. 2). Figure 2 also shows that species richness of the *Spray* plots increased between 2011 and 2014, but by 2020 the pattern had reversed.

Three approaches were used to compare the modern and historical plant communities. First, an ordination (Fig. 4) showed differences between the *Short-term* and *Continuous Management* datasets. There was much more overlap between the plots in the Modern (green dots) and Historical (red dots) *Continuous Management* dataset compared to the *Short-term* datasets (Modern = purple does; Historical = blue does). Second, data from the PERMANOVA of the simplified *Management* dataset (Table 1) showed that that the differences, while small, were significant between the *Continuous Management* and *Short-term Management* (F = 7.5971, p < 0.001) datasets, and plant communities were significantly different between the 1970s and in

2022 (F = 6 5.393, p < 0.001). Third, when we compared the *Continuous* and *Short-term Management* datasets using a Chi-square analysis, the differences between modern and historical communities were not significantly different (χ^2 = 2.3267, df = 1, p = 0.1272). Interestingly, however, there were observable differences in the number and percentage of plots in the two

Table 1
Results of PERMANOVA comparisons of historical (1970s) vegetation described by McCormick and Somes (1982) and modern (2022) vegetation for the *Short-term* and *Continuous Management* datasets using the simplified dataset described in the Methods, comparing the effects of treatment, time, and the time X treatment interaction.

	df	Sum of Squares	R2	F-Statistic	P-Value
Treatment	1	1.688	0.02954	7.5971	0.001
Time	1	14.53	0.25424	65.393	0.001
Treatment:Time	1	0.714	0.0125	3.2149	0.03
Residual	181	40.218	0.70372		

datasets that were classified as being different in 2022 and 1972. For the *Continuous Management* dataset, 24 of the 45 plots (53%) were classified as being different in 2022. For the *Short-term Management* dataset 33 of the 48 plots (69%) differed.

Further insight into temporal changes that occurred following the invasion and management of *Phragmites* were evaluated by comparing species present in 2022 and 1972. Except for three sites, Shady Side Park, Salmon, and Barnette (Supplemental Table 1 – Part B), the species present in 2022 were also present in the 1972. This finding indicates that the native species survived as extant plants during the time that the areas were dominated by *Phragmites*, or they recolonized after management, most likely from seeds in the seed bank (Hazelton et al. 2018). The main difference between the two management categories (*Continuous Management* and *Short-term Management*), however, was not the presence or absence of *Phragmites*, it was present at all but one of the wetlands, but the higher percent cover of *Phragmites* and lower species diversity in the *Short-term Management* wetlands (Fig. 2).

Discussion

Management can reduce *Phragmites*, but complete removal or removing enough of it to enable native plant communities to return—functional eradication—requires time and effort and most often fails (Martin and Blossey 2013; Hazelton et al. 2014). A few studies have documented the recovery of native plant communities for longer periods of time (e.g., Lombard et al. 2012; Rohal et al. 2019b, 2023) or monitored recovery following different management techniques (Farnsworth and Meyerson 1999; Hallinger and Sishler 2009). In general, native species recover at management sites, but differences remain in the composition of the vegetation between managed sites and site where *Phragmites* had not invaded. This project has shown that long-term periodic management can result in the restoration of native plant communities that are similar to the communities that were present before *Phragmites* invasion, but that continuous management is important. Without continuous management, as shown by Rohal et al. (2023) and further demonstrated in this study (Fig. 4), *Phragmites* dominance will return. The economic consequence of short-term management is clear as Martin and Blossey (2013) found that, over a 4-year period, managers in 40 states had spent over $4.6 million on *Phragmites* management with relatively little success. They found that only 20.6% of the respondents strongly agreed that management had resulted in an increase in native plant species, and only 14.2% strongly agreed that *Phragmites* control had been long-term. There have been a few instances where management was successful based on monitoring. In almost all instances, *Phragmites* was not eradicated, even though the diversity of native species increased (Bonello and Judd 2019). What happens in response to *Phragmites* management is context-dependent (Rohal et al. 2019b), and the abundance and cover of *Phragmites* often increases following cessation of management (e.g., Zimmerman et al. 2018; Rohal et al. 2019b, 2023).

The *Monitoring* study reported here and in Rohal et al. (2023) and the comparison of *Short-term* versus *Continuous Management* provide further insight into the difficulties of managing *Phragmites* and clearly provide guidance on the need for regular monitoring of sites and additional management as needed. As an example, one of the managed areas that we sampled as part of the *Continuous Management* effort was owned by three families who contracted with a private provider to manage *Phragmites* in 2015, 2019, and 2022. When we visited the site in 2022, there were scattered *Phragmites* shoots, but more than 90% of the site had native species and the plots that we sampled had a range of native species (*Symphiotricum subulatum*, *Typha angustifolia*, *Hibiscus moscheutos*, *Schoenoplectus americanus*, *Mikania scandens*, *Cyperus strigosus*, *Spartina patens*, *Iva frutescens*, and *Distichlis spicata*). This site is an example of the functional eradication recommended by Green and Grosholz (2021). The same approach has been used by the Chesapeake Wildlife Heritage (CWH) who manages *Phragmites* on the private lands that were the basis of the *Continuous Management* database. CWH applies herbicides when treatment is required based on their assessment of sites or when the private landowners request their assistance. All CWH sites that we sampled had some *Phragmites*, and they have not made any attempts to eliminate it completely from any site. The benefits of *Continuous Management* are clear both financially, as they are less costly in the long-run because the frequency of management decreases over time, and ecologically, as native species recolonize managed sites—providing a competitive environment that minimizes the reinvasion of a site by *Phragmites* seeds (Kettenring et al. 2015). The CWH managed sites had all the species listed in McCormick and Somes (1982) that were known to occur in those plant communities in the 1970s.

Historical Comparisons

Historical data of pre-invasion plant communities provide a unique opportunity to assess the impacts of invasion and the extent of community recovery after invader management. Here we capitalize on a unique opportunity to use the historical data from McCormick and Somes (1982) to assess decadal vegetation change following *Phragmites* management. There are few examples where vegetation change has been monitored for periods of time longer than 1–3 years (e.g., Rohal et al. 2019b; Bonello and Judd 2020), and we are not aware of any in which the vegetation that was present at a site prior to the invasion of *Phragmites* was compared to the vegetation that was present after it had been managed.

Figure 4, which compares sites sampled in 2022 with data from 1972s shows that while there was overlap in the distribution of data points in the ordination, the two sets of sites differed from each other, especially the *Short-term Management* sites where almost 70% of the plots differed between the two time periods. We interpret the differences to be due primarily to the invasion and continued presence of *Phragmites* after the vegetation had been mapped by the Maryland Department of Natural Resources and the shift in species in response to sea level rise (e.g., Lu et al. 2019; Drake 2014). In 1972, only three of sites from the *Management* dataset were classified as a *Phragmites* community and based on McCormick and Somes (1982) the native species that were present at the other sites then was very similar to the list of species that were present in the plant communities when we sampled them in 2022 (Supplemental Table 1B). This comparison of vegetation change over almost 50 years clearly demonstrates that management of *Phragmites* can lead to the recovery of native species that were present prior to the invasion. This result is informative because the species composition of tidal wetland communities has not changed dramatically over the same time period, even though there is some evidence that communities are gradually shifting over time in response to sea level change and other aspects of climate change (e.g., increasing levels of atmospheric CO_2). Drake (2014) summarized 28 years of a long-term project of monitoring vegetation in a brackish tidal wetland plant community that had three dominant species (*Schoenoplectus americanus*, *Spartina patens* and *Distichlis spicata*). While the community type did not change, the abundance of *Schoenoplectus americanus* increased, while the abundance and biomass of the other species declined. Following the removal or reduction in the abundance of *Phragmites*, it seems likely that the plant community type will change very little if the native species are present in or near the site and recolonization can occur from seeds dispersed to the site or regrowth of species that were still present during the period that the site was occupied by *Phragmites*. In a study related to the one reported here, we found that *Schoenoplectus americanus* was present but in low abundance at a site where we removed *Phragmites*. Within two years, it spread rapidly by clonal propagation and became a dominant species (Jacobson et al 2023).

Comparison of Monitored sites that were part of the five-year NOAA project with 1972 data

As described in the Methods, the *Monitoring* data set that we used was originally part of a five-year NOAA project that focused on the responses of *Phragmites*-dominated sites to herbicide application (Hazelton et al. 2014; Rohal et al. 2023). Like results from the analysis of the *Short-term Management* data set, this aspect of the study (Fig. 4) also demonstrated that once management of *Phragmites* ceases, it is likely to re-establish dominance, either excluding native species or resulting in the decline of their populations.

Conclusion

Results of this study demonstrate that in Chesapeake Bay brackish tidal wetlands *Phragmites* management can result in the restoration of native plant communities that were present prior to *Phragmites* invasion of the sites. The dataset provided by Chesapeake Wildlife Heritage and the dataset made available by the private landowners in Anne Arundel County demonstrated that if it is not possible to completely eradicate *Phragmites*, long-term management can assure the restoration and persistence of native plant communities. The effort required to completely eradicate *Phragmites* from a wetland is largely infeasible in many areas of the Chesapeake Bay where the species is now widespread and dominant. Restoration is especially unlikely if the area occupied by *Phragmites* is large, if there are limits to financial resources, or where a given wetland is owned by more than one landowner and one of them is not supportive of efforts to remove *Phragmites* (Epanchin-Niell et al. 2010; Quirion et al. 2018; Rohal et al. 2019a).

In a companion study that was part of this effort (Jacobson et al. 2923), there was one site where *Phragmites* was completely removed through efforts of our research group and citizen scientists. At a low salinity site in Jacobson et al.'s study, following the removal of *Phragmites*, there were more than 50 native species (D. Whigham, personal observation). Other sites at Parkers Creek (Jacobson et al., 2023) were managed for *Phragmites* removal by the American Chestnut Land Trust. At three of those sites, *Phragmites* has been almost completely removed and ongoing management are likely to result in complete eradication. At all three sites, native species have returned and are spreading rapidly (Jacobson et al. 2023). These examples demonstrate that, as shown from this study, continued or periodic management of *Phragmites* has a high probability of restoring a species-rich native plant community.

Declarations

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Author contributions All authors were invovled in the design and conduct of the study. HB and SJ analyzed the data and prepared the figures and tables. DW and HB drafted the manuscript and all authors participated in the editing and review process.

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Competing interests The authors declare no competing interests.

References

1. Able KW, Hagan SM (2003) Impact of Common Reed, Phragmites australis, on essential fish habitat: influence on reproduction, embryological development, and larval abundance of mummichog (Fundulus heteroclitus). Estuaries 26(1):40–50. http://www.jstor.org/stable/1353190
2. Blanca MJ, Alarcón R, Arnau J, Bono R et al (2017) Non-normal data: Is ANOVA still a valid option? Psicothema 29(4):552–557. https://doi.org/10.7334/psicothema2016.383
3. Bonello J, Judd K (2019) Plant community recovery after herbicide management to remove Phragmites australis in Great Lakes coastal wetlands. Restoration Ecology 28:215–221. https://doi.org/10.1111/rec.13062
4. Chambers RM, Meyerson LA, Saltonstall K (1999) Expansion of Phragmites australis into tidal wetlands of North America. Aquatic Botany 64(3):261–273. https://doi.org/https://doi.org/10.1016/S0304-3770(99)00055-8

5. Drake BG (2014) Rising sea level, temperature, and precipitation impact plant and ecosystem responses to elevated CO2 on a Chesapeake Bay wetland: review of a 28-year study. Global Change Biology 20(11):3329–3343. https://doi.org/https://doi.org/10.1111/gcb.12631
6. Delacre M, Lakens D, Leys C (2017) Why psychologists should by default use Welch's t-test instead of Student's t-test. International Review of Social Psychology. https://doi.org/10.5334/irsp.82
7. Epanchin-Niell RS,Hufford MB, Aslan CE et al. (2010) Controlling invasive species in complex social landscapes. Frontiers in Ecology and the Environment 8(4):210–216. https://doi.org/10.1890/090029
8. Faison EK, Elkins G, Kitka K et al (2020) Response of a wetland plant community to management of Phragmites australis (Poaceae) in southwestern Connecticut. Rhodora 122(989):48–52, 45. https://doi.org/10.3119/19-05
9. Farnsworth EJ, Meyerson LA (1999) Species composition and inter-annual dynamics of a freshwater tidal plant community following removal of the invasive grass, Phragmites australis. Biological Invasions 1(2):115–127. https://doi.org/10.1023/A:1010068607630
10. Furman BT, Leone EH, Bell S et al (2018) Braun-Blanquet data in ANOVA designs: comparisons with percent cover and transformations using simulated data. Marine Ecology Progress Series 597. https://doi.org/10.3354/meps12604
11. Gratton C, Denno RF (2005) Restoration of arthropod assemblages in a Spartina salt marsh following removal of the invasive plant Phragmites australis. Restoration Ecology 13(2):358–372. https://doi.org/https://doi.org/10.1111/j.1526-100X.2005.00045.x
12. Green SJ, Grosholz ED (2021) Functional eradication as a framework for invasive species control. Frontiers in Ecology and the Environment 19(2):98–107. https://doi.org/https://doi.org/10.1002/fee.2277
13. Hallinger KD, Shisler JK (2009) Seed bank colonization in tidal wetlands following Phragmites Control (New Jersey). Ecological Restoration 27(1):16. https://doi.org/10.3368/er.27.1.16
14. Hazelton ELG, Mozdzer TJ, Burdick DM et al (2014) Phragmites australis management in the United States: 40 years of methods and outcomes. AoB Plants 6. https://doi.org/10.1093/aobpla/plu001
15. Hazelton ELG, Downard R, Kettenring KM et al (2018) Spatial and temporal variation in brackish wetland seedbanks: implications for wetland restoration following Phragmites Control. Estuaries and Coasts 41(1):68–84. https://doi.org/10.1007/s12237-017-0289-z
16. Holdredge C, Bertness, MD (2011) Litter legacy increases the competitive advantage of invasive Phragmites australis in New England wetlands. Biological Invasions 13(2): 423–433. https://doi.org/10.1007/s10530-010-9836-2
17. Jacobson S, Brooks H, Baldwin A, et al (2023) Transplanting native species to expedite succession following Phragmites australis removal. Final Report. Maryland Sea Grant Program, Annapolis, MD.
18. Kettenring KM, Blois S, Hauber D (2012) Moving from a regional to a continental perspective of Phragmites australis invasion in North America. AoB Plants pls040. https://doi.org/10.1093/aobpla/pls040

19. Kettenring KM, Whigham DF, Hazelton ELG et al (2015) Biotic resistance, disturbance, and mode of colonization impact the invasion of a widespread, introduced wetland grass. Ecological Applications 25(2): 66–480. http://www.jstor.org/stable/24432316
20. Lambert AM, Saltonstall K, Long R et al (2016) Biogeography of Phragmites australis lineages in the southwestern United States. Biological Invasions18(9): 2597–2617. https://doi.org/10.1007/s10530-016-1164-8
21. Levene H (1960) Robust tests for equality of variances. In: Olkin I (Ed.) Contributions to Probabiulity and Statistics. Stanford University Press, Redwood City, California, pp 278–292
22. Lombard KB, Tomassi, D, Ebersole J (2012) Long-term management of an invasive plant: lessons from seven years of Phragmites australis control. Northeastern Naturalist 19:181–193. http://www.jstor.org/stable/41495846
23. Lu M, Herbert ER, Langley JA et al (2019) Nitrogen status regulates morphological adaptation of marsh plants to elevated CO_2. Nature Climate Change 9(10):764–768. https://doi.org/10.1038/s41558-019-0582-x
24. Martin LJ, Blossey B (2013) The runaway weed: costs and failures of Phragmites australis management in the USA. Estuaries and Coasts 36(3):626–632. https://doi.org/10.1007/s12237-013-9593-4
25. McCormick J, Somes HH Jr (1982) The Coastal Wetlands of Maryland. Maryland Department of Natural Resource, Coastal Zone Management Program. Chevy Chase, Maryland.
26. McCormick MK, Kettenring KM, Baron HM et al (2010) Extent and reproductive mechanisms of Phragmites australis spread in brackish wetlands in Chesapeake Bay, Maryland (USA). Wetlands 30(1):67–74. https://doi.org/10.1007/s13157-009-0007-0
27. Meyerson LA, Lambert AM, Saltonstall K (2010) A tale of three lineages: expansion of common reed (Phragmites australis) in the U.S. Southwest and Gulf Coast. Invasive Plant Science and Management 3(4):515–520. https://doi.org/10.1614/IPSM-D-09-00052.1
28. Meyerson LA, Saltonstall K, Windham L et al (2000) A comparison of Phragmites australis in freshwater and brackish marsh environments in North America. Wetlands Ecology and Management 8(2):89–103. https://doi.org/10.1023/A:1008432200133
29. Oksanen J (2022) Package 'vegan'. http://vegan.r-forge.r-project.org/. https://cran.r-project.org/web/packages/vegan/vegan.pdf
30. Quirion B, Simek Z, Dávalos et al. (2018). Management of invasive Phragmites australis in the Adirondacks: a cautionary tale about prospects of eradication. Biological Invasions 20(1): 59–73. https://doi.org/10.1007/s10530-017-1513-2
31. Rohal CB, Cranney C, Hazelton E et al (2019a) Invasive Phragmites australis management outcomes and native plant recovery are context dependent. Ecology and Evolution 9. https://doi.org/10.1002/ece3.5820
32. Rohal CB, Cranney C, Kettenring KM (2019b) Abiotic and landscape factors constrain restoration outcomes across spatial scales of a widespread invasive plant. Frontiers in Plant Science10:481.

https://doi.org/10.3389/fpls.2019.00481

33. Rohal CB, Reinhardt Adams C, Reynolds LK et al (2021) Do common assumptions about the wetland seed bank following invasive plant removal hold true? Divergent outcomes following multi-year Phragmites australis management. Applied Vegetation Science 24(4):e12626. https://doi.org/https://doi.org/10.1111/avsc.12626
34. Rohal CB, Hazelton ELG, McFarland EK et al KM (2023) Landscape and site factors drive invasive Phragmites management and native plant recovery across Chesapeake Bay wetlands. Ecosphere 14(1):e4392. https://doi.org/https://doi.org/10.1002/ecs2.4392
35. Royston JP (1982) An extension of Shapiro and Wilk's W Test for normality to large samples. Journal of the Royal Statistical Society. Series C (Applied Statistics) 31(2):115–124. https://doi.org/10.2307/2347973
36. Shapiro SS, Wilk MB (1965) An analysis of variance test for normality (complete samples). Biometrika 52(3/4):591–611. https://doi.org/10.2307/2333709
37. Uddin MDN, Robinson RW (2017) Responses of plant species diversity and soil physical-chemical-microbial properties to Phragmites australis invasion along a density gradient. Scientific Reports 7(1):11007. https://doi.org/10.1038/s41598-017-11205-0
38. Uddin MN, Robinson RW, Buultjens A et al (2017) Role of allelopathy of Phragmites australis in its invasion processes. Journal of Experimental Marine Biology and Ecology 486:237–244. https://doi.org/https://doi.org/10.1016/j.jembe.2016.10.016
39. Warren RS, Fell PE, Rozsa R et al (2002) Salt marsh restoration in Connecticut: 20 years of science and management. Restoration Ecology 10(3):497–513. https://doi.org/https://doi.org/10.1046/j.1526-100X.2002.01031.x
40. Whyte R, Trexel-Kroll D, Klarer D et al (2008) The invasion and spread of Phragmites australis during a period of low water in a Lake Erie coastal wetland. Journal of Coastal Research 55:111–120. https://doi.org/10.2112/SI55-19.1
41. Zimmerman CL, Shirer RR, Corbin JD (2018) Native plant recovery following three years of common reed (Phragmites australis) control. Invasive Plant Science and Management 11(4):175–180. https://doi.org/10.1017/inp.2018.24

Figures

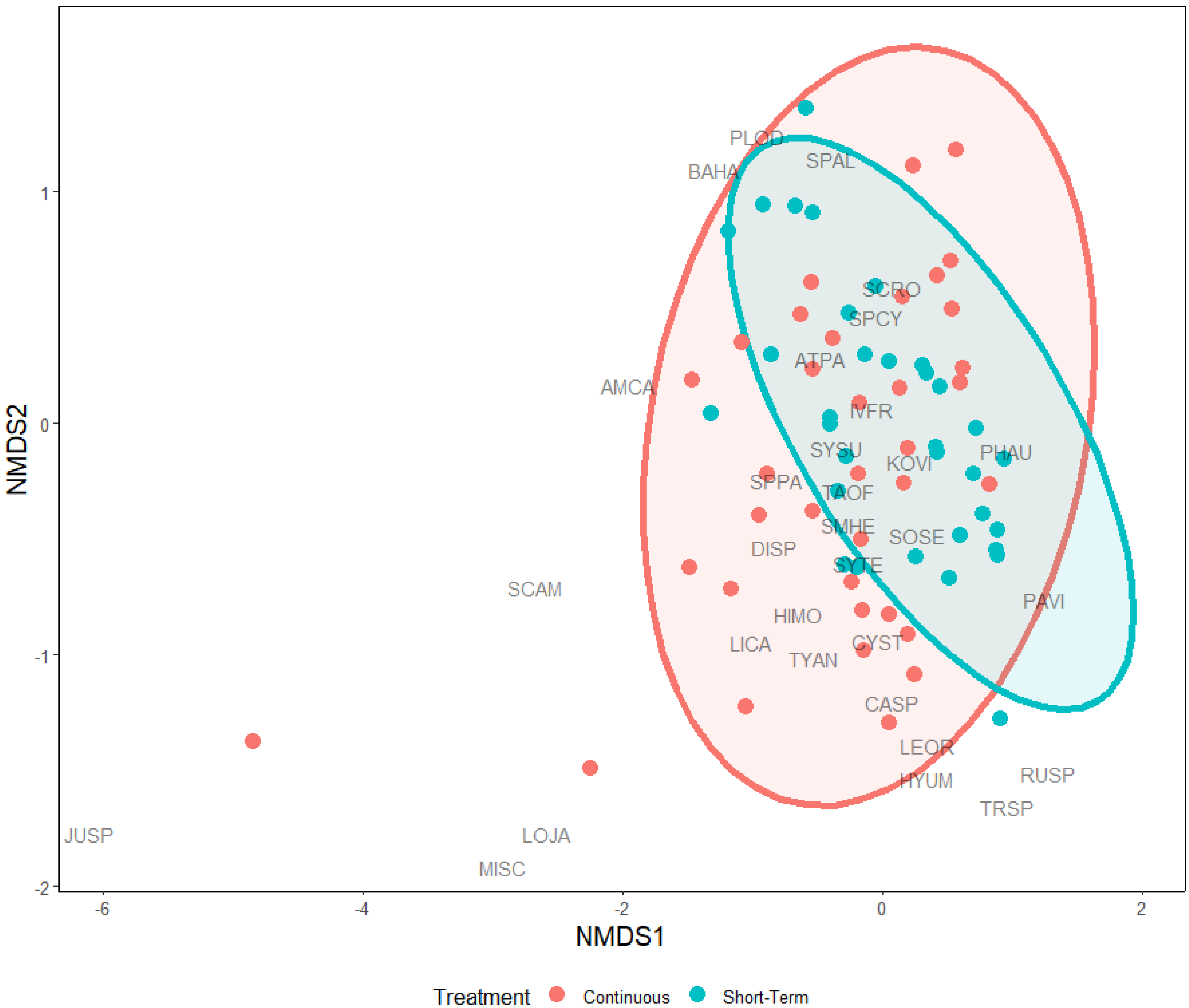

Figure 1

NMDS of modern species cover collected from the 32 sites and 92 individual plots using the Bray-Curtis dissimilarity calculation. Red points indicate the plant communities in the *Continuous Management* dataset (*Phragmites*sprayed continuously for 3-22 years), and blue points indicate plant communities in the *Short-term Management* (sprayed from 2016-2018). Grey four letter codes indicate species scores which are vectors of numbers that represent the position of a given species in the two-dimensional ordinal space. Species codes are as follows: AMCA = *Amaranthus cannabinus,* ATPA = *Atriplex parviflora,* BAHA = *Baccharis halimifolia*, CAST = *Carex* spp., CYST = *Cyperus strigosus,* DISP = *Distichlis spicata,* HIMO = *Hibiscus moscheutos,* HYUM = *Hydrocotyle umbellata,* IVFR = *Iva frutescens,*JUSP = *Juncus* spp., KOVI = *Kosteletzkya virginica,* LEOR = *Leersia oryzoides,* LICA = *Limonium carolinianum,* LOJA = *Lonicera japonica,* MISC = *Mikania scandens*, PHAU = *Phragmites australis,* PLOD = *Pleuchia odorata,* RUSP =

Rubus spp., SCAM = *Schoenoplectus americanus,* SCRO = *Schoenoplectus robustus*, SMHE = *Smilax hedera,*SPAL = *Spartina alterniflora*, SPPA = *Spartina patens,* SOSE = *Solidago sempervirens,* SPCY = *Spartina cynosuroides,* SYSU = *Symphiotricum subulatum*, SYTE = *Symphiotricum tenuifolium*, TAOF = *Taxicum officinale*, TRSP = *Trifolium* spp., TYAN = *Typha angustifolia*. The species codes enclosed in the 95% confidence intervals are the species characteristic of each Treatment group, and the individual points are the individual sites in ordinal space.

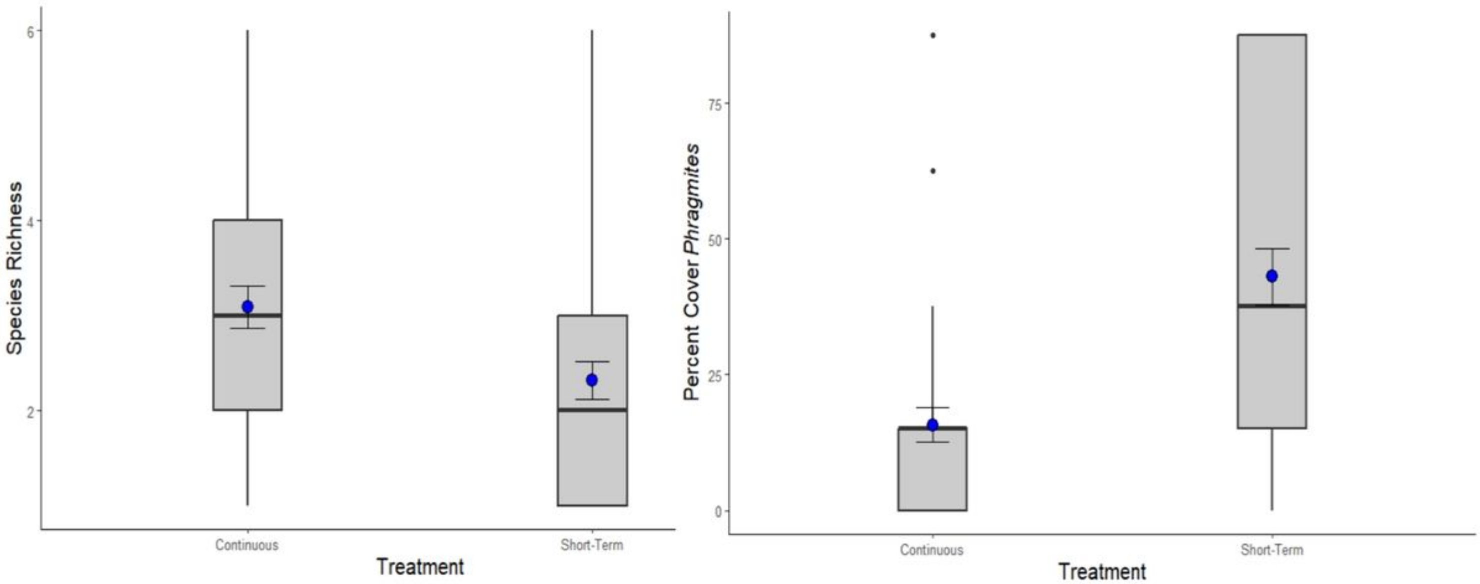

Figure 2

The left diagram shows species richness (F = 7.705, p = 0.006) and the right diagram shows the percent cover of *Phragmites* (F = 23.15, p = 5.7e-06) from the *Continuous Management* sites (17 sites and 51 plots) and *Short-term Management* sites (15 sites and 45 plots). These two plots are violin plots which are made up of three portions: 1) Boxplots, where the box represents the first and third quartile of the data (Q1 and Q3), the horizontal line in the middle represents the median value, and the whiskers extend to the maximum and minimum values. 2) Kernel density plots, where the width of the 'violin' shows the probability density of the points. 3) Blue circles are mean values, and the error bars bounding the mean values are standard error values.

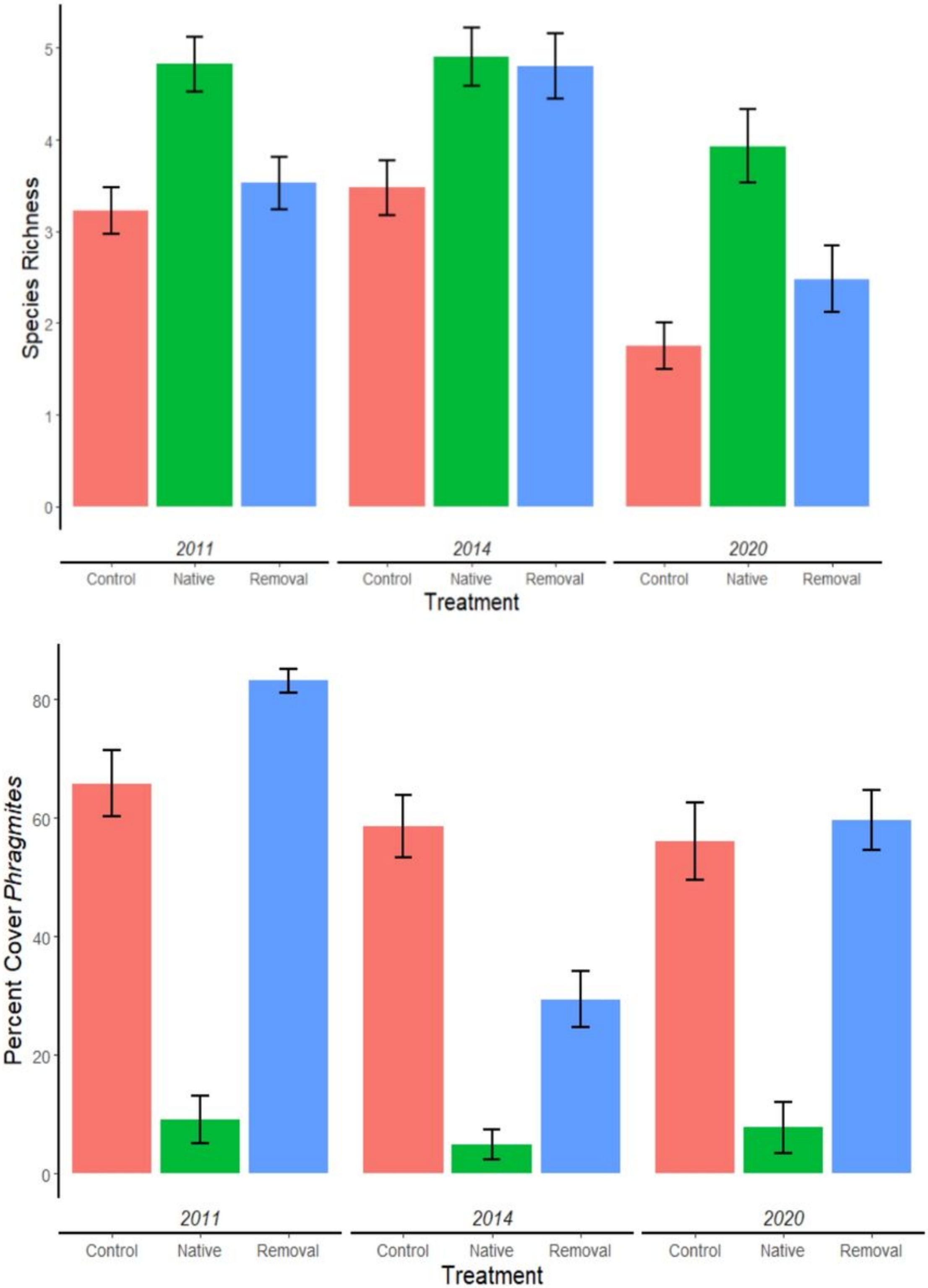

Figure 3

Species Richness (top) and *Phragmites*percent cover (bottom) for control, native, and removal plots from 2011, 2014, and 2020 in 6 sites that were part of the *Monitored* dataset. 2011 was before spraying treatments were initiated, 2014 immediately after spraying treatments concluded, and 2020 was the last year when all the plots included in this analysis were monitored.

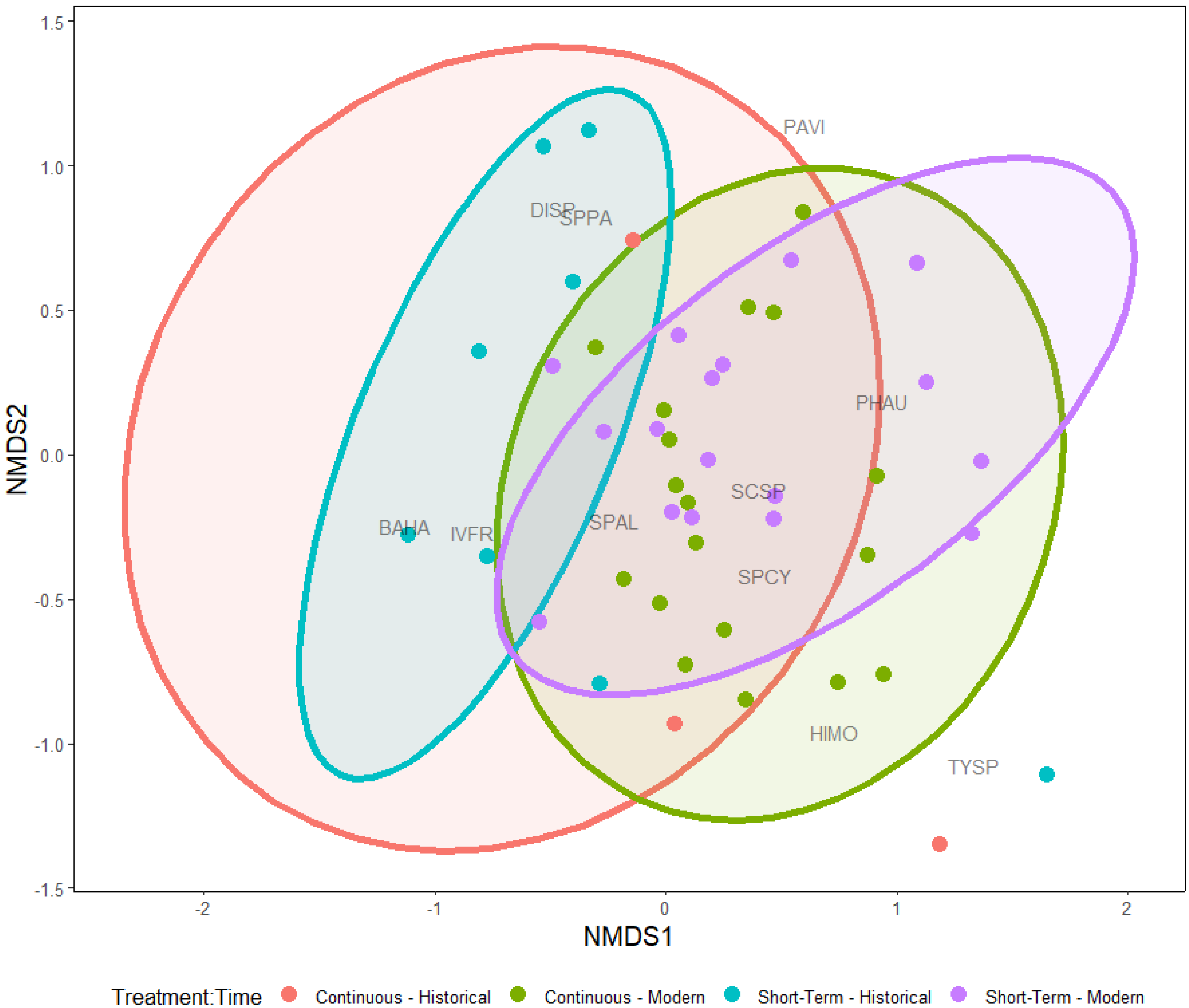

Figure 4

NMDS ordination of historical (1970s) vegetation described by McCormick and Somes (1982) and modern (2022) vegetation for the *Short-term* (n = 15) and *Continuous Management* (n = 17) datasets using the simplified dataset described in the Methods. Red points indicate the historical plant communities in the *Continuous Management* dataset, and green points indicate modern plant communities in the same dataset. Blue points indicate the historical plant communities in the *Short-term Management* dataset, and purple points indicate modern plant communities in the same dataset. The circles around the points are 95% confidence intervals. Grey four letter codes indicate species scores which are vectors of numbers that represent the position of a given species in the two-dimensional ordinal space. Species codes are as follows: BAHA = *Baccharis halmifolia*, DISP = *Distichlis spicata*, IVFR = *Iva frutescens*, HIMO = *Hibiscus moscheutos*, PAVI = *Panicum virgatum*, PHAU = *Phragmites australis*, SCSP = *Schoenoplectus* spp., SPAL = *Spartina alterniflora*, SPCY = *Spartina cynosuroides*, SPPA = *Spartina patens*, TYSP = *Typha* spp. The species enclosed in the 95% confidence intervals are the species characteristic of each Treatment:Time group, and the individual points are the individual sites in ordinal space.

Supplementary Files

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