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Evaluation of Mechanical Cutting to Control Littoral Purple Loosestrife Stands

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INTRODUCTION

Lythrum salicaria L. is an aggressive invasive species of North American wetlands (Thompson et al. 1987). Different control techniques have been applied to manage the invasive populations, especially chemical and biological control methods (Malecki et al. 1993). Chemical control has been successful in controlling large populations of *L. salicaria* but carries risks of contamination or affecting nontarget species when used improperly (Tu 2009). Biological control, which involves releasing several insect herbivores of *L. salicaria* from its native Eurasian range, may become less effective in deep water habitats (Hight and Drea 1991).

Lythrum salicaria has annual stems that remain standing after winter mortality. These hollow stems are thought to act as conduits for oxygen transport to perennating belowground structures, thereby aiding in their survival over winter (Sculthorpe 1967). Cutting of annual stems, followed by overwinter flooding of at least 5 cm, is a known method for controlling cattails (*Typha* spp.). Beule (1979) obtained an 82% reduction in the number of emerging *Typha* stems the next spring with cutting followed by overwinter flooding.

The aim of our study was to determine if cutting *L. salicaria* stems, both live and dead, followed by over winter flooding would significantly reduce stem density and biomass in the subsequent growing season. This could be a useful management technique in deep water habitats where other control measures are not as effective.

MATERIALS AND METHODS

The study was conducted in two littoral stands of *L. salicaria* in Long Lake, Indiana Dunes National Lakeshore (INDL: 41°50'N; 87°W). Long Lake is an interdunal water body that has its long axis (east-west) parallel to the shoreline of nearby Lake Michigan. The first stand (site 1), located near the northern shore of Long Lake, followed the bed of an old railroad track. The second stand (site 2) was a diamond-shaped patch of *L. salicaria* near the western edge of the lake. A 20-m long transect was established in each site in August 1993. Twenty 1.0 m^2 plots were randomly selected along each transect and divided into 10 control and 10 experimental plots. The plots were marked by flags at the corners for ease of locating the plots the following growing season.

The numbers and heights of all live stems were recorded in all plots in August 1993. Stem densities of each plot were calculated as the number of stems per area of plot. Stem biomass was estimated using a nondestructive method (Chiariello et al. 1989). Ten stems representing a range of heights within the plots were selected at each site outside of the plots, measured for height, then cut, dried, and weighed. Regression equations relating stem height to stem dry weight were then calculated for each site (SYSTAT version 6; Wilkinson 1990). Exponential equations gave the best fitting line to the data ($R^2 = 0.98$ for both sites). Regression equations were then used to estimate dry weight of stems within plots.

Living and dead stems were cut close to ground level in the experimental plots following measurements of stem heights. To prevent seed dispersion, inflorescences were bagged and cut from the stems prior to stem cutting, then dried and burned. The control and experimental plots in both sites were resampled in late May and August 1994 using similar methods. Biomass was estimated nondestructively but with linear regressions of square-root transformed dry weights ($R^2 = 0.98$ and 0.97 for sites 1 and 2, respectively).

Water level gauges were placed near transects in each site in July 1993 close (within 5 m) to the site transect. The gauges were monitored throughout the duration of the experiment (July 1993–August 1994). Water levels were also measured in each plot at each sampling date but were found to be similar to the gauge readings; thus, only the gauge measures were used further. At time of stem cutting (Aug 1993), water levels were 44 and 51 cm in sites 1 and 2, respectively, and covered cut stems by at least 40 cm of standing water throughout winter and the following spring (Figure 1).

Because of significant differences in pretreatment stem densities between the two areas ($F_{1, 35} = 6.75$; $p = 0.014$), further analyses comparing control and experimental plots were conducted separately for each site. Stem densities and estimated biomass were compared between control and experimental plots in each site at each sampling date (precutting: Aug 1993; postcutting: May 1994) by t-tests. Paired t-tests were then used to analyze before and after cutting impact (BACI) for the control and experimental treatments at each site. All analyses were conducted using SYSTAT version 11 (SYSTAT 2004).

RESULTS

Water levels during the study ranged from 44 to 62 cm at site 1 and 51 to 68 cm at site 2, with deeper water consistently

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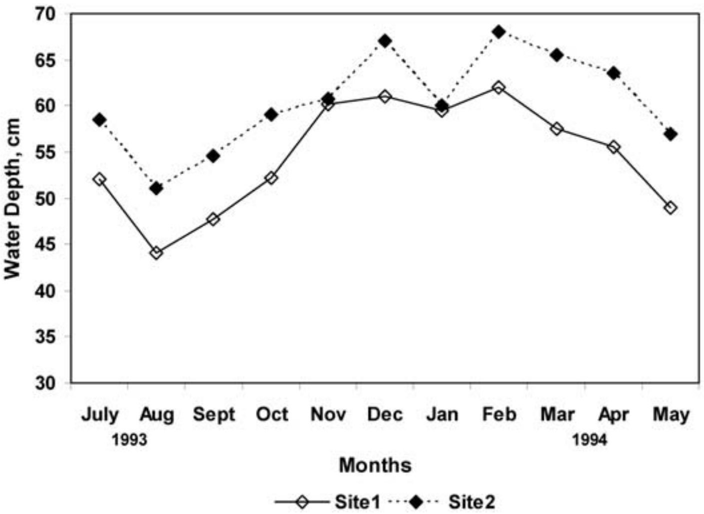

Figure 1. Water levels (cm) in the two experimental sites during the study period.

found in site 2 (Figure 1). The lowest water levels occurred in August 1993 (44 and 51 cm for sites 1 and 2, respectively), coinciding with implementation of the cutting treatment in experimental plots. Water levels were deepest during winter months (Nov 1993 to Feb 1994), with maximum depths measured in February for both sites and then steadily declining to 48 and 57 cm at the two sites, respectively, in May 1994.

Both sites contained dense stands of *L. salicaria*, ranging from about 19 to 37 stems m^{-2} (Figure 2a). Pretreatment stem densities and aboveground biomass (Figure 2b) were similar in control and experimental plots at site 2. There were differences between control and experimental plots at site 1 in August 1993 (precut), with experimental plots having more stems ($t = 2.41$; $p = 0.03$) and greater biomass ($t = 1.98$; $p = 0.06$) than control plots. However, mean stem size (total aboveground biomass per plot/stem number per plot) were similar in August 1993 between designated control and experimental plots in both sites before the cutting treatment ($p = 0.27$ and 0.42 for sites 1 and 2, respectively; Figure 2c).

Stem densities and biomass were significantly lower in experimental versus control plots at each site in May 1994 following cutting and overwinter flooding (all $p < 0.005$; Figures 2a and 2b). There was also a significant decline in average stem size for plants in experimental plots compared to control plots of both sites the following May ($t = 2.45$; $p = 0.04$ and $t = 5.68$; $p < 0.001$ for sites 1 and 2, respectively; Figure 2c).

The BACI analysis showed similar results as t-test comparisons between control and experimental plots. Stem densities in control plots did not change postcutting. Average stem densities were 18.8 and 19.3 stems m^{-2} at site 1 for the August 1993 and May 1994 sampling dates respectively ($t = 0.17$; $p = 0.868$), while there were 31.9 and 33.3 stems m^{-2} for the same dates at site 2 ($t = 0.36$; $p = 0.726$). Conversely, by May 1994 stem densities in the experimental plots had decreased by 93 and 94% at sites 1 and 2 respectively ($p < 0.001$ for both sites; Figure 2a).

Cutting *L. salicaria* stems, followed by overwinter flooding of at least 40 cm, resulted in significant reductions in both

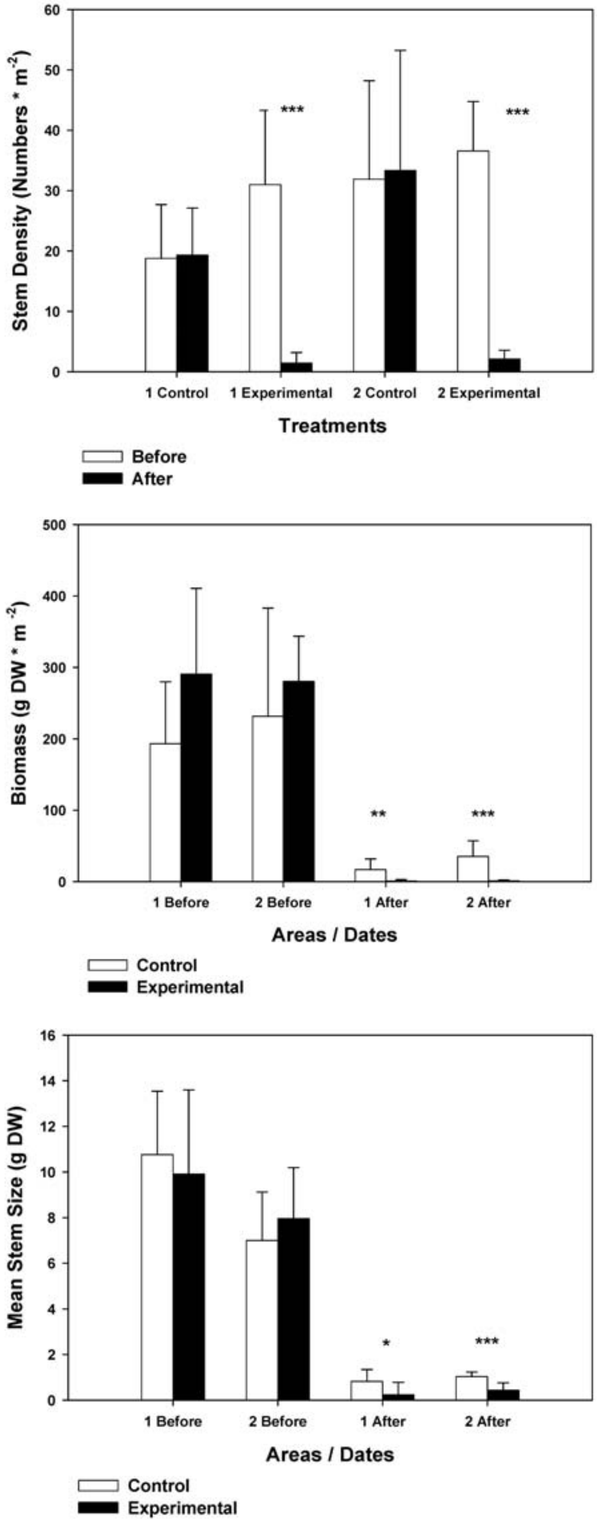

Figure 2. A. Before (Aug 1993) and after (May 1994) control impact (BACI) assessment showing mean (± 1 SD) stem densities of *Lythrum salicaria* in control and experimental plots for the two study sites. B. Mean total aboveground biomass (g DW * m^{-2} ± 1 SD) in control and experimental plots before (Aug 1993) and after (May 1994) cutting treatments in the two study sites. C. Mean individual stem size (g DW ± 1 SD) in control and experimental plots before (Aug 1993) and after (May 1994) cutting treatments in the two study sites. P values of t-tests: * < 0.05; ** < 0.01; *** < 0.001.

stem density and aboveground biomass, with the few re-emerging stems in cut plots being smaller than new stems in control plots. These results indicate that this technique, which has been used to control *Typha* stands (Beule 1979), may be a useful method for controlling *L. salicaria* in deep water habitats.

Depth of overwinter flooding may be of great importance in effectively reducing *L. salicaria* stem emergence the following spring (Haworth-Brockman et al. 1991). For example, summer clipping *L. salicaria* stems 10 cm below the water surface with flooding through fall did not greatly reduce stem densities (Haworth-Brockman et al. 1991), although the length of stems remaining on plants in this study was unclear. Long stem sections remaining on plants experiencing a short duration of flooding could allow plants to resume growth from stems and quickly recover stem densities within the same season. In our study, overwinter water levels were >40 cm above stems, which were cut near ground level. To our knowledge, no study has been conducted to determine the minimum water level or duration of flooding needed to significantly reduce stem re-emergence.

Invasive North American populations of *L. salicaria* can establish and thrive in a greater range of habitats, including deeper water areas, when compared with populations growing in their native Eurasian range (Bastlová-Hanzélyová 2001). Deep water areas present a challenge for current control methods. Chemical control is perhaps the most used method in deeper water and has successfully controlled large stands of *L. salicaria* when used correctly (Smith 2009). However, chemical control may not be an option in sensitive areas and can carry associated risks. For example, chemical agents may be applied to nontarget plant species with negative consequences (Tu 2009). Issues with biological control include particularly insufficient reduction of *L. salicaria* populations by insects in deep water areas (Hight and Drea 1991; comments in Smith 2009). The expansion in range of habitats occupied by *L. salicaria* in North America may represent greater phenotypic plasticity that evolved after the species had successfully established in its secondary range (Richards et al. 2006). If so, biocontrol insects may simply be lagging behind *L. salicaria* in adapting to different habitats in the introduced range. Alternatively, physical impediments might prevent biocontrol insects from being effective in areas like deep water habitats. Further research is needed to answer these questions.

No single control method is best under all circumstances (Tu 2009). There are important limitations to using cutting followed by flooding as a control method (e.g., Heidorn 1990, Smith 2009). First, this method is best used in conjunction with a natural flooding regime. As noted elsewhere (Heidorn 1990), artificially flooding an area invaded by *L. salicaria* can result in greater damage to native plants, with the end result being greater infestation of *L. salicaria*; therefore, knowledge of the hydrologic regime of the system is a prerequisite to the appropriate use of this method. Second, more research is needed to establish the minimum water level and duration of flooding needed to achieve control. We can only be confident that this method will be successful if a site can maintain 40 cm water depth over cut stems through winter. Third, care must be taken to remove all cut stems from the target area followed by proper disposal, preferably by burning. New stems can emerge from small stem fragments left in the field (Stevens et al. 1997) leading to re-infestation. Fourth, cutting should occur before seeds have matured, as was done in our study. If cutting occurs late in the season, such as in early autumn, inflorescences should be bagged and removed from stems prior to stem cutting and disposed of properly (inflorescences from our sites were burned). Fifth, as with any mechanical control method, this technique is labor and time intensive and is therefore most suitable for eradicating small (<100 plants) to medium-sized (100-300 plants) stands. Last, this method will likely require long-term monitoring of the target area and continued cutting of re-emerging stems. Fluctuating water levels are an inherent characteristic of wetland ecosystems (Mitsch and Gosselink 2000); thus, occasional drawdowns will be expected. Soil exposure will lead to germination from the seed bank. Seed production is quite high in *L. salicaria* (Thompson et al. 1987), resulting in an extensive seed bank. It is not known for how long *L. salicaria* seeds remain viable, but 4-year-old seeds had 80% germination (Rawinski 1982). Monitoring is therefore key to removing new plants that emerge due to a drawdown and before natural reflooding occurs.

Even with these caveats, based on the results of this study, cutting followed by overwinter flooding can control small- to medium-sized invasive *L. salicaria* populations when used in the proper context. The many concerns listed above can be alleviated by the use of well-trained volunteers (see comments by Summers in Smith 2009). This technique can be a useful addition to the suite of techniques available to managers, especially in deep water habitats, for customizing management on a case by case basis.

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