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A multiyear year study of three plant communities with purple loosestrife and biological control agents in Virginia

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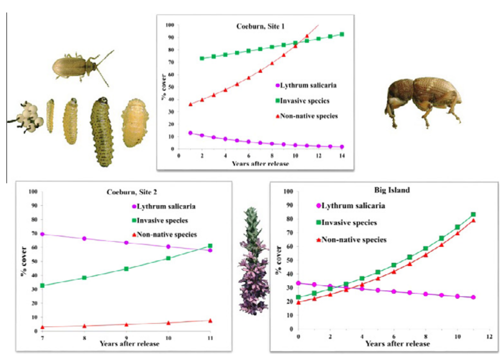
HIGHLIGHTS

- Fourteen years post-release *L. salicaria* declined but invasive species increased.
- A 2nd site after 11 years *L. salicaria* remained high and invasive species low.
- A 3rd site after 11 years deer feeding depressed biological control agent density.
- Agents successfully reduced *L. salicaria* when its density was low and invasives high.

GRAPHICAL ABSTRACT



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ABSTRACT

Purple loosestrife (*Lythrum salicaria* L.) native to Eurasia has become an invasive weed in North America since its introduction in the late eighteenth century. To control *L. salicaria*; *Galerucella calmariensis* L., *Galerucella pusilla* Duftschmidt, and *Hylobius transversovittatus* Goeze were released at Coeburn (Sites 1 and 2) and Big Island, Virginia in 1992 and 1999, respectively. The biocontrol agents and plant community parameters were studied for 11 and 14 years at Big Island and Coeburn, respectively. The *Galerucella* spp. and *H. transversovittatus* became established at both sites. After 8 years *Galerucella* spp. at Coeburn dispersed 1768 m with a mean of 221 m/year from the original release site. *H. transversovittatus* was found 400 m from the original release site at Coeburn after 14 years. Both *Galerucella* spp. were found in similar ratios at Coeburn. At Big Island the biocontrol agents did not spread due to the low *L. salicaria* density beyond the release site. At Coeburn Site 1 *L. salicaria* percent cover, May and fall stem density, seed capsules and total inflorescence length per m^2 significantly declined by 88%, 98%, 97%, 94%, and 88%, respectively. While *L. salicaria* declined non-native and invasive species percent cover increased at Coeburn Site 1. Non-invasive species cover also increased but was lower than the invasive species cover. At Coeburn Site 2 *L. salicaria* was much denser and did not decline after 11 years. Invasive species were also lower in density than at Coeburn Site 1, possibly allowing *L. salicaria* to maintain its competitive advantage over other species at Site 2. No significant reductions occurred in percent cover, stem height, seed capsules or inflorescence length per m^2 in any of the *L. salicaria* metrics at Big Island except the number of stems per m^2 which declined. Deer herbivory was high at Big Island and may have reduced *Galerucella* spp. density and its impact on *L. salicaria* metrics. Percent cover of native, non-native, and invasive species significantly increased at Big Island with non-native species increasing at a faster rate. The biological control agents had a significant impact on *L. salicaria* at Coeburn Site 1 but not at Coeburn Site 2 or Big Island. While *L. salicaria*

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was reduced by the biocontrol agents at Coeburn Site 1, invasive species became more abundant. Non-native and invasive species increased and the quality of this plant community remained low. Future weed biological control programs should include habit restoration.

1. Introduction

Purple loosestrife (*Lythrum salicaria* L.) is native to Eurasia and occurs in the eastern hemisphere in Eurasia, northern Africa, and Australia (Hultén, 1971). It has a perennial rootstock that annually sends up 4–10 stems per root and grows to a height of 2 m (Thompson et al., 1987). Dispersal is primarily through seed with estimates of well over 10,000 seeds per plant (Shamsi and Whitehead, 1974; Thompson et al., 1987; Ture et al., 2004). Since the early nineteenth century *L. salicaria* has become a major weed of wetlands in North America (Stuckey, 1980) infesting over 400,000 acres and reportedly displacing native wetland plant species (Blossey et al., 2001; Thompson et al., 1987). Its present distribution in North America extends from southern Canada and is found in all states except: Arizona, New Mexico, Louisiana, South Carolina, and Florida (EDDMapS, 2015). In Virginia it has been found present in 40 counties (Capel, 1993; Virginia Botanical Associates, 2014). McAvoy et al. (2005) found 51 occurrences of infestations in western Virginia and northeastern Tennessee. Initial establishment is often associated with recently disturbed areas such as industrial, construction, waste, and dump areas (Stuckey, 1980; Wilcox, 1989). Factors that have contributed to invasiveness in North America are its efficient use of nutrients and energy (Nagel and Griffin, 2001) plus the absence of host specific herbivores (Blossey and Notzold, 1995; Galatowitsch et al., 1999; Hight, 1990).

Before the release of the *L. salicaria* biocontrol agents much antidotal evidence existed for the impact of *L. salicaria* on wetlands and native plant species (Lavoie, 2010). Several studies have questioned the general consensus that *L. salicaria* has a negative impact on native plant species and ecosystems in North America (Anderson, 1995; Hager and McCoy, 1998; Hager and Vinebrooke, 2004; Houlahan and Findlay, 2004; Kiviat, 1989; Mahaney et al., 2006; McGlynn, 2009; Swift et al., 1988; Whitt et al., 1999). Studies by Treberg and Husband (1999) and Morrison (2002) found no support for the hypothesis that species richness is decreased in wetlands infested with *L. salicaria*. Farnsworth and Ellis (2001) found that several metrics did not show that *L. salicaria* had a negative impact but when using measures of biomass *L. salicaria* did have a negative impact. Blossey et al. (2001), Grabas and Laverty (1999), Hovick et al. (2011), Nagel and Griffin (2001), Rawinski and Malecki (1984) and Thompson et al. (1987) documented evidence of the negative impact of *L. salicaria*. The conflicting conclusions of these studies are likely due to the very wide range of metrics used to measure impacts and particularly the short duration of some studies. More standardized and longer term studies would greatly help in the ability to confidently determine both impacts of the target weed for pre and post-release studies.

While antidotal observations were likely the main impetus for implementing the *L. salicaria* biological control program, results of research done after release of biocontrol agents in 1992 on the impact of *L. salicaria* on wetlands justify this program (Blossey et al., 2001). Recognizing the limitations of mechanical and chemical control procedures, classical biological control was initiated (Blossey et al., 2001; Mullin, 1998). The goal of releasing the *L. salicaria* biocontrol agents as stated by Malecki et al. (1993) was broadly stated to 'reverse the massive degradation of wetland habitat currently attributed by this species'. This is generally the measure of success in restoration ecology (Ruiz-Jaen and Aide, 2005); reducing the density of the target weed species and allowing the native species that were suppressed to increase in density and diversity.

At the initiation of many biological control programs this is often the implied measure of success for the project. However goals are not often well articulated and direct and indirect effects are often not anticipated or incorporated into an overarching proactive management system (Denslow and D'Antonio (2005). As stated by Seastedt et al. (2008) and van Wilgen et al. (2013) the return of an ecosystem after the release of a biological control agent followed by a reduction of the target species to its former historical community of biotic and abiotic structure is unrealistic and rarely achievable. Van Klinken (2006) described a more realistic ecological criteria for the biological control program for *Parkinsonia aculeata* in Australia that defined success at the population level when: a reduction in density, patch size, and rate of spread and in-fill occurred. These are goals that future projects could use although no mention was made of the negative changes in the plant community that may occur after the decrease in the target species. Carson et al. (2008) proposed six protocols to evaluate the success or failure of biological control programs. Few quantitative post-release assessments have included vegetative responses to the reduction of the target species and would greatly aid in the evaluation of outcomes (Denslow and D'Antonio, 2005). Other studies have investigated the economic benefits of using biological control agents to manage alien species. De Lange and van Wilgen (2010) reported a high benefit:cost ratio (50:1 and 3726:1) for managing invasive Australian trees with biological control agents. Page and Lacey (2006) found an overall benefit cost ratio of 23:1 in weed biological control in Australia. Headrick and Goeden (2001) reported that the economic benefits of biological control were the best tool for ecosystem management in the U.S. Based on these reports the use of biological control to manage alien plant species is very economical but the resulting pant community ecosystem must also be included in determining the success or failure and few studies have done that.

Galerucella calmariensis L., *Galerucella pusilla* Duftschmidt (Coleoptera: Chrysomelidae), and *Hylobius transversovittatus* Goeze (Coleoptera: Curculionidae), were extensively tested and approved for release (Blossey, 1993; Blossey et al., 1994a,b; Hight et al., 1995; Kok et al., 1992a,b; Malecki et al., 1993). Releases began in 1992 with the establishment of the *Galerucella* spp. in many regions of North America (Albright et al., 2004; Dech and Nosko, 2002; Denoth and Myers, 2005; Katovich et al., 2001; Kok et al., 2000; Landis et al., 2003; Lindgren, 2003; McAvoy et al., 1997; McAvoy and Kok, 2002; Piper, 1996; Schooler and McEvoy, 2006; Wiebe et al., 2001; Wiedenmann, 2005). *H. transversovittatus* establishment has occurred in Virginia (McAvoy and Kok, 2002), New York (Hunt-Joshi et al., 2004) and colonization in Washington (Piper, 1996). There are likely unreported releases and establishment of *H. transversovittatus*, and this species may be more prevalent than reported.

Several studies have documented the density of *Galerucella* spp. and the vegetative and reproductive parameters of *L. salicaria* from 2 to 10 years after release. Studies have found reductions in *L. salicaria* stem density (Boag and Eckert, 2013; Denoth and Myers,