



Effects of logging history on invasion of eastern white pine forests by exotic glossy buckthorn (*Frangula alnus* P. Mill.)

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ABSTRACT

The extent to which forests can be invaded by exotic plants and the role of tree harvest in facilitating such invasions are important issues in invasion biology. Our objective was to determine: (a) whether a wide-spread exotic shrub, glossy buckthorn (*Frangula alnus*), can invade a common northeastern US forest type, (b) the extent to which logging facilitates buckthorn invasion, (c) whether buckthorn invades gaps through pre-disturbance ('advance') regeneration, (d) whether or not it forms uneven-aged populations in invaded stands. We selected nine eastern white pine – hardwoods stands in Durham, NH. Three were undisturbed, three were clear-cut, and three were partially cut. Cutting occurred ≥ 6 years prior to sampling. Glossy buckthorn (≥ 0.5 m tall) was present in all stands at the time of sampling and most buckthorn populations were all-aged, suggesting that recruitment by seed continued after initial establishment and that long-term persistence in these stands is possible. Buckthorn was present in four of six cut stands prior to cutting, indicating some advance regeneration. The results support the view that forests are not inherently resistant to invasion by exotic plants. Compared to uncut stands, however, buckthorn had higher densities in clear-cut and partially cut stands. In partially cut stands, buckthorn density was greater in canopy gaps than in adjacent uncut areas. Thus, logging facilitated buckthorn invasion. Given this result and the known negative effects of buckthorn on tree regeneration, control measures should be considered when logging stands where buckthorn invasion is likely.

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1. Introduction

Invasive, non-native plant species are widely distributed in forests of temperate North America (Stapanian et al., 1998; Webster et al., 2006). Some of these exotic species alter forest structure and compete with native plants, including economically important tree species (Woods, 1993; Webb et al., 2000; Gorchoff and Trisell, 2003; Frappier et al., 2003a, 2004; Fagan and Peart, 2004; Miller and Gorchoff, 2004; Dornig and Cipollini, 2006).

Intact forests of native species generally harbor fewer exotic invasive plants than comparable non-forested communities (Crawley, 1987; Rejmánek, 1989), a pattern likely due to 'biotic resistance' (high levels of competition and predation; Levine et al., 2004), coupled with the simple fact that most invasive species are not shade tolerant (Martin et al., 2009). Both theory and empirical evidence suggest that disturbance to the forest canopy increases the frequency and magnitude of exotic invasion (Rejmánek, 1989; Hobbs and Huenneke, 1992; Knapp and Canham, 2000; Webb et al., 2001; Vankat and Roy, 2002; Watkins et al., 2003; Lundgren et al., 2004; Mandryk and Wein, 2006; Searcy

et al., 2006; Belote et al., 2008; Eschtruth and Battles, 2009). While both natural and anthropogenic disturbances occur in forests, they differ in that the timing, size, frequency, and intensity of the latter can be controlled, perhaps in such a way as to minimize invasion (Burnham and Lee, 2010). Thus, investigation of the effects of anthropogenic forest disturbance, especially logging, is warranted.

The effects of logging on exotic plant invasion in temperate forests have been studied with varied results. Some studies show increased abundance and/or species richness of exotic plants after logging (e.g., West and Chilcote, 1968; Selmants and Knight, 2003; Gray, 2005; McDonald et al., 2008a) while others have reported a small or undetectable response (Stapanian et al., 1998; Griffis et al., 2001; Roberts and Zhu, 2002; Kern et al., 2006; DeGasperi and Motzkin, 2007; Dodson et al., 2008; McDonald et al., 2008b). Several studies (Griffis et al., 2001; Haeussler et al., 2002; Dodson et al., 2008) have shown that logging promotes invasion only when accompanied by burning or scarification. Species richness and abundance of invasive plants sometimes increase with the size of the canopy opening (e.g., Belote et al., 2008; Burnham and Lee, 2010). Most of these studies, while informative, were conducted at a broad spatial scale, including many invasive species at many sites (e.g., Stapanian et al., 1998; Gray, 2005; McDonald et al., 2008a), or have examined all invasive taxa as one component of

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the entire plant community (e.g., Griffis et al., 2001; Roberts and Zhu, 2002; Kern et al., 2006). Relatively little information is available on the responses, dynamics, and population structure of individual species.

The target of our research was glossy buckthorn (*Frangula alnus* P. Mill., hereafter *buckthorn*), a shrub or small tree native to Europe. Now widespread in eastern North America (Catling and Porebski, 1994; Johnson et al., 2006; Webster et al., 2006), buckthorn is increasingly reported from forests dominated by eastern white pine (*Pinus strobus* L.) (Catling and Porebski, 1994; Frappier et al., 2003a,b; Fagan and Peart, 2004; Cunard and Lee, 2009; Burnham and Lee, 2010). As buckthorn has been described as shade tolerant (Sanford et al., 2003), or moderately so (Cunard and Lee, 2009), it has the potential to establish and persist in white pine-dominated stands, as well as other forests. This possibility concerns forest managers, as buckthorn is known to inhibit the seedling growth of economically important native trees (Frappier et al., 2003a, 2004; Fagan and Peart, 2004).

Forests dominated by eastern white pine are common in the northeastern USA (Leak et al., 1995). Most existing stands originated by natural regeneration (some by planting) on agricultural land abandoned since 1870. Mature stands are harvested by diverse cutting methods, including clear-cutting, group selection, and single tree selection, although white pine regeneration is best in the large openings created by clear-cutting or shelterwood cuts (Lancaster and Leak, 1978). Buckthorn is a new problem for white pine forests (e.g., Frappier et al., 2003a) and little is known about the ability of buckthorn to invade intact white pine-dominated stands, the role that canopy disturbance might play in promoting buckthorn success, and the most effective means of control.

Our specific objectives were to determine: (1) whether or not buckthorn can invade relatively undisturbed eastern white pine stands; (2) whether or not logging increases the probability of buckthorn invasion or its density (or both) in these stands; (3) whether buckthorn colonizes gaps by pre-disturbance regeneration or by newly dispersed seed; and (4) whether buckthorn populations are even-aged, resulting from a pulse of recruitment after logging, or uneven-aged, the result of continuous recruitment. The ideal way to meet these objectives would be to conduct a manipulative experiment. Different logging treatments, all replicated, would be randomly applied and the success of the invasive shrub would be assessed some time after logging. This approach, however, is not only logistically challenging, but it may take many years to obtain definitive results about the invasion success of a long-lived shrub.

An alternative approach – correlative rather than experimental – is to select an area where the invasive shrub has been long established and where different logging treatments have been applied in the recent past. Sites representing different treatments are then sampled, and shrub abundances and age structure are described and related to the time and intensity of logging. The key tool in this approach is dendroecology, the analysis of annual wood rings. While not commonly encountered in research on invasive plants, dendroecology has been used to describe the spatial spread of invasive woody plants (Frappier et al., 2003b; Wangen and Webster, 2006) and two studies have used tree ring counts to relate establishment of woody invasive plants to past logging events (Silveri et al., 2001; Burnham and Lee, 2010). We believe that this ‘historical-correlative’ approach can be informative in evaluating the effects of forest management on woody plant invasion. We used the historical-correlative approach to examine the invasion of glossy buckthorn into eastern white pine-dominated stands that had been exposed to three treatments: (1) clear-cutting, (2) partial cutting, and (3) no cutting.

2. Materials and methods

2.1. Study sites

We searched for forest stands of old-field origin and dominated by eastern white pine (presently or prior to cutting). We limited the search to Durham, NH, where glossy buckthorn is well established and widely distributed (Frappier et al., 2003a; Cunard and Lee, 2009; Burnham and Lee, 2010). We located four uncut stands, three clear-cut stands, and two partially cut stands on land owned by the University of New Hampshire, and one partially cut stand on land owned by the Town of Durham (see Table 1 for site names and characteristics). We randomly excluded one of the four uncut stands, leaving a total of nine stands. Most sites had reforested naturally after agricultural abandonment, but some planting of eastern white pine and other conifers had occurred at Arsenault, Doe Farm, and Weld. Each stand was ca. 2 ha in area and surrounded on at least three sides by forest, successional thicket, or managed hayfields. In the clear-cut and partially cut stands, logging occurred between six and 30 years prior to sampling. Stands were sampled between May 2005 and July 2008.

2.2. Vegetation sampling

Each stand was sampled at 20 randomly selected points, with the constraint that each point had to be at least 20 m from any other point. Each point was the center of two plots, one of 3 m radius, the other of 7 m radius. In the partially cut stands, the random points fell either into canopy gaps created by tree removal or into uncut (‘canopy’) areas. If the 7 m radius plot resulting from a gap point included both gap and canopy conditions, the plot was shifted until the entire plot was in the gap or the plot center was at the center of the gap, whichever condition was met first. Plots resulting from canopy points were shifted so that no part of the plot was in a gap.

In each 3 m radius plot we tallied all ‘large’ buckthorn individuals, i.e., those ≥ 0.5 m tall, and for each of these we measured height of tallest terminal bud, number of live stems at the base, and diameter (using calipers) of the stem base (or base of the largest stem in multiple-stemmed plants). With lopping shears or saw, we then removed a basal cross-section of the stem (we used the stem with greatest diameter in multiple-stemmed individuals) for later analysis of wood rings. We also tallied all ‘small’ (<0.5 m tall) buckthorn individuals in 3 m radius plots. At all six clear-cut and partially cut sites we excavated the roots of each large buckthorn (≥ 0.5 m tall) to see if there were vegetative connections between what we were calling ‘individuals’.

Woody vegetation was sampled at each site. In each 3 m radius plot we tallied, by species, small trees and shrubs ≥ 2 m in height and up to a diameter at breast height (DBH) of 10 cm. In each 7 m radius plot we identified each tree ≥ 10 cm DBH and measured its diameter.

At partially cut sites we estimated the proportion of canopy trees removed during cutting. Prior to cutting, all three stands were pine-dominated and more-or-less even-aged. Consequently members of the original canopy cohort, living or cut (i.e., stumps), were easily distinguished from other trees, which were mainly smaller hardwoods. In each 7 m radius plot, we tallied the live canopy trees and cut stumps. At the Route 4 site there had been two cuts (Table 1) and thus two age classes of stumps were discerned.

2.3. Dendroecology

Glossy buckthorn produces annual, semi-ring porous wood rings (Schweingruber and Landolt, 2005). As per Burnham and Lee (2010), we prepared the basal end of the stem cross-sections

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