



Plantation of coniferous trees modifies risk and size of *Padus serotina* (Ehrh.) Borkh. invasion – Evidence from a Rogów Arboretum case study



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ABSTRACT

Density of natural regeneration of black cherry (*Padus serotina*) depends on distance from the propagule source. Ecological success of this species is higher in coniferous than deciduous forests. The main aim of this study was to assess the interaction between the distance from propagule source and tree stand type (coniferous, deciduous and mixed) on occurrence and density of natural regeneration of black cherry. The study was conducted on 202 experimental plots in Rogów Arboretum (Central Poland), consisting of plantations of alien and native tree species, growing on potential habitats of fertile deciduous forest. The density of natural regeneration was measured in four height classes: 0–0.5 m, 0.5–2 m, 2–5 m and over 5 m. Natural regeneration of black cherry occurred on 79 of the 202 plots, and its density varied from 0 to 25,660 ind. ha⁻¹. The mean density of black cherry was statistically significantly higher ($p < 0.001$) in coniferous (1275.8 ± 374.2 ind. ha⁻¹) than in deciduous (138.3 ± 48.3 ind. ha⁻¹) and mixed (29.3 ± 12.3 ind. ha⁻¹) stands. There was also a negative relationship between distance from propagule source (stand of *P. serotina* established in 1932 in the central part of the Arboretum) and density of natural black cherry regeneration ($R^2 = 0.19$, $p < 0.001$). Tree stand type modified the relationship between distance from propagule source and probability of natural black cherry regeneration occurrence ($p < 0.001$), as well as density of natural regeneration ($p < 0.001$). The study proved that effects of propagule pressure were strongly modified by habitat features, which confirms that plantation of conifers on sites suited for deciduous forests increases the risk of *P. serotina* invasion.

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1. Introduction

In the whole world, 751 species of trees and shrubs are known to be invasive (Rejmánek and Richardson, 2013). Due to their longevity and long duration of the invasion lag time, the invasion ecology of trees and shrubs is insufficiently recognised (Richardson and Rejmánek, 2011). Their impact can include competitive displacement of native species, as well as transforming the features of habitats (Rejmánek, 2014), so that many invasive woody species are called 'transformers' (Vitousek, 1990; Richardson et al., 2000; Crooks, 2002; Ehrenfeld, 2003; Corenblit et al., 2014). Habitat transformations vary depending on species and include the amount of organic matter and nutrients accumulated, rate at which they are recycled, litter pH, and light

availability and microclimate within the forest. The relatively high biomass increment of invasive trees allows these species to displace or limit the increment of native competitors, by limiting soil nutrients (Binkley and Valentine, 1991; Binkley and Giardina, 1998; Finzi et al., 1998a,b) or light availability (Augusto et al., 2003; Knight et al., 2008). Significant changes caused by the introduction of different species of trees are quantifiable after several decades (Reich et al., 2005; Dickie et al., 2006; Hobbie et al., 2006, 2007, 2010; Withington et al., 2006; Dauer et al., 2007; Mueller et al., 2012). These changes of soil properties cause changes in species composition of the understory plant community, confirmed by Augusto et al. (2002, 2003), Wulf and Naaf (2009) and Barbier et al. (2008). Overstory tree species also affect species richness and diversity of the understory. This variability may be explained by pH, nitrogen availability and C:N ratio. Moreover, diversity of the understory is correlated with features which are variable within tree species – stand density and canopy cover (Jagodziński and Oleksyn, 2009).

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In many areas foresters have replaced natural forest plant communities with plantations of tree species, which are ecologically or geographically alien. Alien coniferous tree species have transformed environments created by native deciduous forests (Binkley and Valentine, 1991; Binkley and Giardina, 1998), resulting in homogenisation of the structure of the community and significant decreases of plant species diversity. In plantations there may be no specialised species, as well as a higher share of alien species, in comparison with surrounding areas (Paritsis and Aizen, 2008). Although as a general rule, decreasing spatial microhabitat heterogeneity causes a corresponding decrease in species diversity, alien species do not always decrease heterogeneity of habitat (Crooks, 2002). Species which are nitrogen-fixing and have high water requirements cause the largest habitat changes (Rejmánek, 2014). As a result, many species of invasive trees and shrubs are transformer species (Richardson et al., 2000; Crooks, 2002; Starfinger et al., 2003; Corenblit et al., 2014).

Invasive species very often use fertile habitats with high levels of disturbance, which can be colonised by both native and alien species (Stohlgren et al., 1998, 1999, 2006; Lonsdale, 1999; Davis et al., 2000; Knight et al., 2008; Dyderski et al., 2015; but see also Parker et al., 2010). This tendency is especially prominent in river valleys (Pyšek and Prach, 1994; Stohlgren et al., 1998; Terwei et al., 2013; Corenblit et al., 2014; Dyderski and Jagodziński, 2014; Dyderski et al., 2015), where disturbances are crucial habitat-shaping factors (Hood and Naiman, 2000; Brown and Peet, 2003; Richardson et al., 2007; Schnitzler et al., 2007).

Black cherry – *Padus serotina* (Ehrh.) Borkh. (= *Prunus serotina* Ehrh.) – is one of the most frequent woody neophytes in temperate Europe. It is a shrub from the Rosaceae family, with natural range in eastern North America. It was introduced in Europe in the 17th century (Muys et al., 1992; Starfinger et al., 2003; Godefroid et al., 2005). At the beginning, black cherry was planted as an ornamental shrub, but from the end of the 19th century was widely introduced into forests in Europe, first as a timber tree, and after the failure of this concept, as a soil improver (Muys et al., 1992; Starfinger et al., 2003; Godefroid et al., 2005). In Poland black cherry was introduced in 1813 (Hereźniak, 1992). Starting in the late 19th century, as in most of Europe, black cherry was planted in forests, especially in plantations of Scots pine during the 1950–1970s. Due to its massive introduction, black cherry started to spread and became a forest weed, which has caused problems for forest management. During the last 50 years black cherry colonised almost all of Poland, except for the Carpathians (Tokarska-Guzik, 2005).

Black cherry occurs most frequently in Scots pine stands (Zerbe and Wirth, 2006; Halarewicz, 2011a,b, 2012; Halarewicz and Żońnier, 2014). This species prefers habitats of mixed-coniferous forests (Halarewicz, 2011a), but also occurs in more fertile habitats, for example oak-hornbeam forest and ash-alder alluvial forests (Stypiński, 1977, 1979; Danielewicz, 1994). Through its abilities to compete and transform light conditions in the understory, black cherry is a serious threat to species diversity of forest plant communities (Halarewicz, 2012, 2011a). Godefroid et al. (2005), as well as Halarewicz and Żońnier (2014), found negative relationships between the proportion of black cherry in the shrub layer and understory species richness and diversity.

Most frequently black cherry starts fruit production at ages of 7–10 years in open fields and 20 years in dense tree stands, while the highest fruit production occurs at ages from 30 to 100 years (Burns and Honkala, 1990; Starfinger et al., 2003). Although fruits are eaten by mammals (fox, fallow deer, marten, wild boar) and birds (Bartkowiak, 1970; Starfinger et al., 2003; Deckers et al., 2008), the density of seedlings is highest near propagule sources, because animals disperse only about 20% of all seeds which reach the soil (Pairon et al., 2006).

There is strong evidence from field studies showing that distance from propagule source affects density of natural regeneration (Deckers et al., 2005, 2008; Pairon et al., 2006), as well as positive interaction between planting coniferous species on habitats of deciduous forests and success of black cherry regeneration (Starfinger et al., 2003; Godefroid et al., 2005; Zerbe and Wirth, 2006; Knight et al., 2008; Halarewicz, 2011a, 2012; Halarewicz and Żońnier, 2014). However, at this time there is lack of knowledge of the interaction between invasive *P. serotina* dispersal and environmental filtering by different types of tree stands. Therefore, the aim of this paper was to answer how the plantation of coniferous species modifies the probability of occurrence and density of black cherry natural regeneration. Rogów Arboretum is an excellent place to answer these questions due to high diversity of species cultivated in experimental stands, full historical documentation of plots, homogenous soil conditions and known age of the propagule source (i.e. when black cherry was first introduced in nearby areas). Thus, Rogów Arboretum seems like an ideal site to search for the pattern of *P. serotina* expansion from this kind of natural experiment.

2. Materials and methods

2.1. Area of the study

The study was conducted in the Rogów Arboretum (51°49'N, 19°53'E; 189 m a.s.l.), which is a Forest Experimental Station of Warsaw University of Life Sciences (SGGW). In the Arboretum collections of ornamental woody plants and experimental stands of 75 species of trees, with different ecological requirements, have been established (Table A.1). All plots were located in potential habitat of fertile deciduous forest, and potential natural vegetation is subcontinental oak-lime-hornbeam forest. Long-term meteorological observations (55 years) from the nearest meteorological station (Strzelno), showed a mean annual temperature of 7.2 °C (in January –3.2 °C, in July 17.3 °C), mean annual precipitation of 596 mm, and mean growing season length (calculated as the number of days with mean temperature ≥ 5 °C) of 212 days (Jagodziński and Banaszczak, 2010).

Rogów Arboretum has been a site for numerous studies on alien species of trees and shrubs. For example, experimental plots with *Abies cephalonica*, *Abies grandis*, *Abies procera*, *Acer rubrum* and *Pinus peuce* were examined to check the species richness of plants, fungi and invertebrates (Jagodziński et al., 2011; Kasprowicz et al., 2011; Skorupski et al., 2011; Wojterska et al., 2012; Kałucka et al., 2013). Kasprowicz and Wojterska (2013) studied understory vegetation of 11 stands and reference plots in oak-lime-hornbeam forest. The common garden-grown trees in Rogów Arboretum, representing broadleaf deciduous angiosperm and evergreen conifer species, were studied by Wyka et al. (2012) to find responses of leaf structure and photosynthetic properties to intra-canopy light gradients, and Skorupski et al. (2012) studied chemical composition of leaves and litter on the same plots.

2.2. Methods

The study was conducted in September 2012 on 202 plots, with a total area of 25.7 ha (Table A.1). The plot size ranged from 0.01 to 2.1 ha (only 7 stands had areas greater than 0.5 ha; plots 3.D, 5.O2, 6.11, 6.60, 8.B, 6.A and 8.A), with an average size of 0.13 ± 0.02 ha. All plots were experimental stands of tree species. Stands where the understory was mowed, single trees and shrubs and ornamental collections were excluded from the study. The 202 plots included over 110 tree species; 99 were tree stands with coniferous species, 57 had deciduous trees and 46 were mixed. Most of the stands (172) were dominated by alien species of trees, and

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