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Traditional coppice forest management drives the invasion of *Ailanthus altissima* and *Robinia pseudoacacia* into deciduous forests

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ABSTRACT

The non-native tree species Black Locust (*Robinia pseudoacacia* L.) from N America and Tree of Heaven (*Ailanthus altissima* (Mill.) Swingle) from E Asia are invading both anthropogenic and near-natural habitats throughout Europe. So far, few studies have focused on their invasion into forests and none has addressed the influence of coppice management, which is characterized through regular clear-cuts. To retrace the invasion during the last 50 years, we studied the abundance in the canopy as well as the natural regeneration of these species in the coppice forest of Gargazon (South Tyrol, Northern Italy), which was ideal due to its patchy mosaic of different forest age (time since last coppicing). We analyzed which factors favor the presence and abundance of *A. altissima* and *R. pseudoacacia* regeneration. Tree saplings were sampled on 113 plots each measuring 7 m^2 in a total study area of 25.5 hectares. On each plot, the following environmental variables were recorded: canopy cover per species, total herb layer cover, distance to clearing, stone cover, slope, and aspect. Soil moisture was derived from herbaceous indicator species. Hurdle models were used for data analysis to account for the zero-inflated data structure. The results showed that *R. pseudoacacia* has established earlier in the studied forest than *A. altissima*, but *A. altissima* has been spreading with higher speed during the last 30 years. Our study exemplifies the invasion of both species synchronously to the coppice cycle, that means they colonize fresh clear-cuts, establish, and remain. We conclude that the currently applied coppice management, which consists of repeated clear cuttings each 20–30 years, favors the spread of both invasive species. Thus, an adaptation of the management system is needed to avoid further invasion.

1. Introduction

Coppice forest management as a silvicultural system has a long history and is still practiced on over 21 million hectares in Southern and Southeastern Europe (Puumalainen, 2001). Usually, relatively small forest patches of about 0.5 hectares, depending on country and forest ownership, are clear-cut each 20–30 years (Piussi, 2006; Bartha et al., 2008). In some regions like in our study area South Tyrol, Northern Italy, coppice is favored for its rockfall protection efficiency (high number of stems), while in other regions it is maintained for its heritage and nature conservation value (Kirby and Thomas, 2000; Hedl et al., 2010). Nevertheless, each clear-cut displays a disturbance event (Roberts and Gilliam, 1995) and disturbances are generally regarded to facilitate biological invasions (Catford et al., 2012). Alien species are often good colonizers, so that early stages of secondary succession are very susceptible to invasion (Catford et al., 2012). Growth conditions are additionally facilitated on clearings, because resource availability is high and competition from native tree species is reduced (Alpert et al., 2000). In fact, non-native *Ailanthus altissima* and *Robinia pseudoacacia* occur in European forest ecosystems mainly on clearings, along the fringes of forests and in sparsely stocked woodlands (Celesti-Grapow et al., 2010; Kohler, 1963; Kowarik, 2010). In the insubric region of Switzerland, regeneration of *R. pseudoacacia* and *A. altissima* has been abundant after a forest fire and the authors concluded that few native species will be able to compete with them after stand-replacing forest fires (Maringer et al., 2012). These findings lead to the question whether coppice management facilitates their invasion in a similar way because the clearings display safe sites (*sensu* Harper et al., 1961) for the regeneration and establishment of both light-demanding pioneer species.

While the ecology of Tree of Heaven (*A. altissima* (Mill.) Swingle) and Black Locust (*R. pseudoacacia* L.) is well understood in urban-industrial ecosystems, much less is known about their invasion

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into semi-natural woodlands and how forest management activities can unintentionally favor these species (Kowarik and Böcker, 1984; Gutte et al., 1987; Celesti-Grapow and Blasi, 2004; Kowarik and Säumel, 2007; Basnou, 2009). Currently, there is hardly any information on the invasive potential of *Ailanthus* and *Robinia* in coppice systems (Hegi, 1925; Lohmeyer, 1976; Ließ and Drescher, 2008). This is a serious knowledge gap considering on the one hand the vast abundance of coppice forests especially in the Mediterranean Basin (Puumalainen, 2001) and on the other hand the high biodiversity that is found there (Cowling et al., 1996).

Biological invasions are identified as one of the major causes for biodiversity loss (Hassan et al., 2005; Kowarik, 2010; Millennium Ecosystem Assessment, 2005) and changes of ecosystem structure and function (Vitousek et al., 1996; Higgins et al., 1999; Mooney, 1999; Nentwig, 2007). Changes in plant community composition and negative impacts on the recruitment of native species were reported (Richardson, 1998). Fuentes-Ramirez et al. (2011) showed that some invasive species can form monospecific stands impeding the recolonization of native species. Consequently, managing and combating non-native species is one of the major objectives of the current European biodiversity strategy (European Commission, 2011). To develop such measures it is important to understand the invasion dynamics.

From a theoretical point of view, coppice management should favor the invasion of *A. altissima* and *R. pseudoacacia* for a second reason. Since it has been shown that mechanical control measures, which are often applied to repress invasive tree species, actually stimulate vegetative regeneration (Burch and Zedaker, 2003; Böcker and Dirk, 2008), the frequent clear-cuts in coppice forest likely enhance the vegetative regeneration of *A. altissima* and *R. pseudoacacia*. Furthermore, the vegetative offspring of *R. pseudoacacia* and *A. altissima,* which are still connected to the mother plant by roots, is considered more shade-tolerant than the generative offspring (Kowarik, 1995, 1996; Knapp and Canham, 2000). In addition to disturbance and the subsequent change in resource availability, environmental stress has been described as a major factor influencing habitat invasibility (Alpert et al., 2000). However, *A. altissima* and *R. pseudacacia* are known to be very drought-tolerant even outside their natural range (e.g. Celesti-Grapow and Blasi, 2004).

To better understand the invasion dynamics of the two alien species into coppice forest systems, we addressed the following questions:

(1) Are *Ailanthus* and *Robinia* more abundant in younger forest patches than in older ones? Do abundance in canopy and regeneration show the same pattern?
(2) Which factors favor the regeneration of *Ailanthus* and *Robinia* in coppice forests in general?
(3) Do non-native saplings occur more frequently than native saplings and do they grow faster?

2. Materials and methods

2.1. Study site

The study was carried out in the upper Etsch valley in South Tyrol, Northern Italy, south of the city of Meran (46°34′50″N, 11°12′46″E). The investigated coppice forest is located near the village of Gargazon at the eastern slope of the valley between 260 and 430 m a.s.l. The climate is sub-mediterranean with a mean annual precipitation of about 700 mm with maxima in summer and autumn. The mean annual temperature is 11.5 °C (Autonome Provinz Bozen – Südtirol, 2010). During the summer an absolute maximum temperature of 40 °C can be reached. The geology is volcanic porphyry with nutrient-poor and acidic Syrosem soil (Peer, 1995). The intensive frost shattering during the post-glacial period created stony slopes with different stone or block size (Lüth, 1993; Bosellini, 2007).

Two forest types that are well-adapted to those warm and dry slope habitats are the *Antherico liliaginis-Quercetum pubescentis* (further referred to as very dry oak forest) on the driest areas of the study area and the *Fraxino orni-Ostryetum quercetosum pubescentis* (further referred to as hophornbeam-manna ash forest) on the slightly moister areas (Peer, 1995). Additionally, *Quercus petraea* is often part of the forest as standard tree, i.e. a tree that is kept longer than one rotation period to ensure natural regeneration from seeds and high-quality timber instead of firewood only. In addition to the above mentioned species, *Acer campestre*, *Ailanthus altissima*, *Celtis australis*, *Prunus avium*, *Prunus mahaleb*, *Robinia pseudoacacia*, *Sorbus domestica*, *Sorbus torminalis, Tilia cordata*, *Ulmus glabra*, and *Ulmus minor* occur.

2.2. Sampling procedure

Considering the patchiness of different stand age and small-scale environmental variation in the municipality forest of Gargazon, tree regeneration was systematically sampled in a 45-m grid. This was laid over the total study area of 25.5 ha resulting in 126 sampling points. After exclusion of some points due to inaccessibility or man-made structures, such as trails, 113 points were inventoried. Around each point, two circles were sampled. Firstly, a circle with a radius of 1.5 m (plot size: 7 m^2) for recording natural regeneration and secondly one with a radius of 2 m (plot size: 12.6 m^2) for the inventory of herbaceous indicator species. Slope correction was taken into account. Field sampling was carried out in May and June 2011.

Natural regeneration, i.e. saplings >10 cm in height and <7 cm dbh (diameter at breast height), were counted per tree species and per height class (11–33, 34–66, 67–99, 100–199, >199 cm). Since it would have been too time-consuming to excavate all individuals, we could not differentiate between generative and vegetative saplings. The two oak species *Quercus petraea* and *pubescens*, which occur in the study area, are difficult to differentiate due to hybridisation (Salvini et al., 2009; Viscosi et al., 2009). Hence, oaks were identified as one species (*Qu. petraea et pubescens*).

As suggested by Ellenberg et al. (1991), herbaceous indicator species were used to get an estimate of soil moisture. More precise technical measurements were not possible due to the sample size and stony ground. Using indicator values of herbaceous species to retrieve site information which may influence regeneration, is an approved method (Ammer et al., 2004; Zerbe and Wirth, 2006). Based on previous vegetation data from the study site (Radtke, 2011, unpublished data), 14 relatively abundant species were chosen as indicator species, i.e. 7 indicators for dry sites and 7 for moist sites (Table 1). Plots were classified as “moist” or “dry” when only indicator species for moist or dry sites were found, respectively. Plots with both dry and moist site indicators were classified as ‘‘medium”. Moisture indicator values and nomenclature of the species follow Landolt et al. (2010).

Forest stand age, i.e. time since last coppicing, was derived from a map of the forest service in ten-year classes (0–10, 11–20, 21–30, 31–40, >40 years). Canopy tree composition was visually estimated in 10% classes when standing on the plot center within a circle of app. 20 m radius. In addition we recorded the presence of single mature *Ailanthus* and *Robinia* trees (dbh >7 cm) as potential diaspore sources.

We estimated the distance to the border of the next clearing in meters to consider edge effects. Clearings are characterized by high light and resource availability as well as a specific micro-climate in general. These features clearly distinguish clearings from patches of all other age classes. The distance of sample points on clearings