

# The rise and fall of the black locust (*Robinia pseudoacacia* L.) in the “Siro Negri” Forest Reserve (Lombardy, Italy): lessons learned and future uncertainties

Renzo MOTTA<sup>1\*</sup>, Paola NOLA<sup>2</sup>, Roberta BERRETTI<sup>1</sup>

<sup>1</sup> Dep. AGROSELVITER, University of Turin, Via Leonardo Da Vinci 44, 10095 Grugliasco (TO), Italy

<sup>2</sup> Dep. ECOTER, University of Pavia, Via S. Epifanio, 27100 Pavia, Italy

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## Abstract

- The alluvial forests of the Ticino valley have been greatly reduced in size and are now represented by only a small number of fragmented remnants. To study the natural development of the black locust, an invasive species, on relatively undisturbed lowland forests, two permanent plots were established in 2005 in the “Siro Negri” Forest Reserve.
- The black locust became established almost exclusively between 1940 and 1960. The observed dynamic of the black locust in the Reserve was very similar to what has been observed in its native North American range: following the initial colonization, the black locust firmly established itself in the dominant and intermediate layers but did not regenerate. In addition to the absence of a significant younger population, the decline of the black locust is evident in an elevated mortality rate and higher proportion of black locust biomass in the total coarse woody debris (CWD).
- Our results support the hypothesis that the best strategy to control the spread of black locust is to avoid disturbances that favour black locust colonization, and to wait for natural suppression of the species by other trees. Due to the lack of past reference conditions and the future uncertainties, ongoing monitoring will be needed to fully understand the dynamics of forest ecosystem change in the Reserve.

## Résumé – L’essor et le déclin du robinier (*Robinia pseudoacacia* L.) dans la Réserve Forestière « Siro Negri » (Lombardie, Italie) : leçons et incertitudes futures.

- Les forêts alluviales de la vallée du Tessin ont été fortement réduites en taille et ne sont maintenant représentées que par un petit nombre de vestiges fragmentaires. Pour étudier le développement naturel du robinier, une espèce envahissante, dans des forêts de plaine relativement intactes, deux placettes permanentes ont été établies en 2005 dans la Réserve Forestière « Siro Negri ».
- Le robinier s’est établi presque exclusivement entre 1940 et 1960. La dynamique d’installation observée de *Robinia pseudoacacia* L. dans la réserve est très similaire à ce qui a été observé dans son habitat naturel en Amérique du Nord : après la première colonisation, le robinier s’est fermement établi dans les strates dominantes et intermédiaires, mais il ne s’est pas régénéré. En plus de l’absence d’une population plus jeune, l’évidence du déclin du robinier est révélée par une élévation du taux de mortalité et une plus grande proportion de la biomasse du robinier dans le total des débris ligneux grossiers (CWD).
- Nos résultats appuient l’hypothèse que la meilleure stratégie pour contrôler la propagation du robinier est d’éviter les perturbations qui favorisent sa propagation, et d’attendre la suppression physique de l’espèce par d’autres arbres. En raison de l’absence de références sur les conditions passées, et les incertitudes futures, la surveillance continue sera nécessaire pour comprendre la dynamique de l’écosystème forestier dans la réserve.

\* Corresponding author: [renzo.motta@unito.it](mailto:renzo.motta@unito.it)

## 1. INTRODUCTION

The introduction of exotic species represents one of the greatest biological threats to biodiversity, second only to habitat destruction (Wilcove et al., 1998). In ecosystems devoid of evolutionarily-adapted biological controls, exotic organisms can cause significant and, in some cases, irreparable damage (Driesche and Driesche, 2004). The negative impacts on ecosystem structure and processes are particularly relevant in designated parks, preserves or other natural reserves that protect small patches or relicts of native vegetation (Brothers and Spingarn, 1992), like the Ticino Regional Park. The Ticino “valley” is a traditional rural landscape with semi-natural ecosystems including paddy rice, cornfields, permanent grasslands and water-meadows, riparian forests (Nola, 1991; 1996) and wetlands with high species diversity (Sartori, 1984; Tomaselli and Gentile, 1971). The alluvial forest ecosystems are one of the most threatened natural environments of southern Europe because they have been greatly reduced in size and are now represented exclusively by fragmented remnants (Schnitzler, 1994; Schnitzler et al., 2007). The Park is an extremely important ecological corridor in the highly urbanized Po plain, linking the Alps and the Apennines.

Beginning at the end of the 19th century, exotic, invasive species, such as the black cherry (*Prunus serotina* Ehrh.), the black locust (*Robinia pseudacacia* L.), the tree of heaven (*Ailanthus altissima* (Mill.) Swingle) and the red oak (*Quercus rubra* L.) were introduced into the Park area. These species are now widespread and have contributed to significant ecological problems because they impact biodiversity and forest ecosystem functioning (Motta Fré and Motta, 2000). The southern part of the Park, near the confluence of the Ticino and Po rivers, contains the best-preserved forest patches found in the Park (Sartori, 1984). Even this area, however, has experienced the adverse effects of invasive species, of which black locust is the most widespread and aggressive. The invasion is also favoured by the small size of the alluvial remnants: in fact exotic species are often increased in abundance near forest edges. This increase in abundance could be either because of the increase in resource availability near edges, or because of increased dispersal into forest edges (McDonald and Urban, 2006).

The black locust is native to the south-eastern United States and favours conditions found on the lower slopes of the Appalachian Mountains, with additional populations located along the slopes and forest edges of southern Illinois, Indiana and Missouri (Boring and Swank, 1984; Fowells, 1965).

Its introduction to Europe dates to the early 17th century. The black locust was initially considered an ornamental tree but was also used for reforestation projects in poor soils because of its rapid growth and ability to fix atmospheric nitrogen (Dzwonko and Loster, 1997). The dispersion of the black locust into Italian native forests was fostered by the chestnut blight in the early 20th century. The black locust is now solidly established in northern Italy, ranging from the alluvial forests to the lower mountain forests (Ferraris et al., 2000) and poses a serious threat to native forest vegetation, especially in dry and nutrient-poor sites, as well as lowland forests, even

through these are not typical habitat for this species (Mondino and Scotta, 1987).

In its native range, the black locust is primarily an early seral stage species, found particularly in large clearings such as clear-felled areas and abandoned pastures (Boring and Swank, 1984). The black locust is shade intolerant and is not found in dense woods except as a dominant tree. With sufficient room, its rapid growth rate enables it to compete successfully with more shade tolerant trees. Natural reproduction of the black locust is primarily vegetative through root suckering and stump sprouting (Fowells, 1965); sprout vigour usually increases following top removal by cutting. This characteristic has been widely used in northern Italy for the short-rotation coppice managed for fire-wood production (Pividori and Grieco, 2003). This silviculture management protocol, based on a very frequent disturbance regime, favours the formation of pure black locust stands where the native species have no opportunity to re-establish. Man influences the diffusion of the black locust both directly, by intentional planting, and indirectly because of the species’ aggressive expansion into sites that have been disturbed by human activities.

In the high forests, successful colonization by the black locust is related to large disturbances (e.g. clear cuts or windthrows) that leave areas without substantial woody growth and thus vulnerable to aggressive species like the black locust. Most of the lowland high forests in northern Italy are now managed with a close-to-nature silviculture, with a goal of regenerating natural stands of native species. This is a change from the recent past when silviculture goals were primarily related to wood production. Because of the recognized problems associated with the invasive black locust in native forests, management proposals have been intensively reviewed and discussed. Two management options have been applied, the first of which is to restore native stands by the selective eradication of the exotic species (e.g. EU Life project “Restoration of alluvial woods in the Ticino Park”, LIFE97 NAT/IT/4134). The second option, based on observations of the role of the black locust in its native range, emphasizes the avoidance of disturbances that would encourage colonization, and waiting until native, late seral species gain dominance of a particular forest stand, naturally suppressing the spread of the black locust. The success of selective eradication programs for the black locust and other invasive species is low and implementation of management practices is expensive because of continuous stump sprouting. The second management option may be more ecologically and economically sustainable. Unfortunately, because there are very few stands that have been allowed to develop undisturbed for a long period (at least a few decades), obtaining quantitative data to validate this option is problematic.

An opportunity to study the natural development of the black locust inside silviculturally undisturbed lowland forests is presented in the “Siro Negri” Forest Reserve (Tomaselli and Gentile, 1971).

The goal of this research was to study the spatio-temporal development, and disturbance history, of the current stands in order to: (1) reconstruct the history of the establishment and spread of the black locust; (2) determine the present growth

trends of the black locust (expansion, stability or regression); (3) discuss the potential future management scenario of the Reserve itself; and (4) provide functional management and conservation data that could be applied to the Ticino lowland forests where the black locust is, or may be, a serious ecological problem.

## 2. MATERIALS AND METHODS

### 2.1. Study area

The study site is located on the southern side of the Ticino Regional Park, about 10 km north-east of Pavia, in the municipality of Zerbolò (1504522N – 5006395E) at an elevation of about 63 m a.s.l. The study site is inside the “Siro Negri” Reserve, which was established in 1970 and belongs to the University of Pavia. The Reserve covers 11 ha, historically it has always been forested but formerly was managed for wood production and used as a hunting reserve; nevertheless currently represents one of the best conserved relicts of the original alluvial forest along the Ticino river, and has remained unmanaged for over two decades. The Reserve contains living relicts of an alluvial forest phytosociologically described as a *Polygonato multiflori-Quercetum roboris* association (Sartori, 1984). The area is subject to periodic flooding from the Ticino river and its tributaries; in the last decade, floods occurred in 2000, 2002 and 2005.

### 2.2. Structural analysis

After an in-depth survey of the area, we selected two plots, located at the boundaries of the Reserve, where the black locust is common. Each plot covered 0.25 ha (50 m × 50 m). Inside the plot, all the trees with a diameter of ≥ 7.5 cm at 130 cm above the ground were identified, labelled with numbered plastic tags and mapped. For each tree the following data were collected: species, diameter at a height of 1.30 m (dbh), total height, heights of the lowest living branches (north and south), four radii of the vertical crown projection on the ground along the four cardinal directions (to the nearest 0.1 m). Volumes were calculated according to National Forest Inventory yield tables (Castellani et al., 1984).

In order to describe the vertical structure and to assign individual trees to vertical strata depending on the relative position of tree crowns, we applied the TSTRAT algorithm (Latham et al., 1998):

$$CPS = 0.40 CL + HBLC$$

where *CPS* is the cut-off point per stratum, *CL* is the crown length, and *HBLC* is the height-to-base of the live crown.

Before applying the stratification algorithm, the tree records were sorted in descending order by height and crown length, so that, for trees of equivalent height, the tree with the longest crown was considered first. All trees with heights that equalled or exceeded the competition cut-off height were placed in the same stratum. When a tree height was less than the cut-off height, a new cut-off height value was calculated based on the tallest remaining tree with the longest crown for that tree height. Three layers were identified: dominant, intermediate and the remaining layers, which were classified as the dominated layers. Regenerated trees (height ≥ 10 cm and diameter at 130 cm < 7.5 cm) were mapped and measured (height and four crown projections) in a 500 m<sup>2</sup> subplot (10 m × 50 m) inside the primary plot.

### 2.3. Coarse woody debris (CWD)

CWD was grouped into snags (dead standing trees, dbh > 7.5 cm and taller than 1.3 m), downed logs (piece of fallen stem or branch with a diameter of at least 10 cm at the larger end and length > 1 m) and stumps (short, vertical pieces created by cutting or by windthrow, diameter at the top > 7.5 cm and height < 1.3 m). The volume of a non-broken snag was calculated according to National Forest Inventory yield tables (Castellani et al., 1984). The volume of broken snags, logs and stumps was estimated as a frustum of a cone using the diameter at the top, diameter at ground level (avoiding roots or irregular shape) and length (estimating a plane cutting of the apex parallel to the base), in the following equation:

$$V_{ss} = [(S_o + S_t)/2]L$$

where *V<sub>ss</sub>* is the volume of logs, broken snags and stumps in cubic metres (m<sup>3</sup>), *S<sub>t</sub>* is the area of the small end in square metres (m<sup>2</sup>), *S<sub>o</sub>* is the area of the large end in square metres (m<sup>2</sup>) (the area for the broken snags was calculated from a diameter estimated using the Bitterlich's relascope), and *L* is the length of logs, broken snag or stump in metres (m).

### 2.4. Dendroecological analysis

In order to calculate age structure and analyse growth trends, an increment core was taken at a height of 50 cm from each labelled tree. In the laboratory, all cores were fixed to wooden supports and prepared with a razor until an optimal surface resolution was obtained, allowing the annual rings to be measured. Tree ring width was determined to the nearest 0.01 mm, using the LINTAB device and TSAP software package (Rinn, 1996).

For the cores including pith (23.3% of the total analysed cores), age was taken as the number of rings between the pith and the cambium. For cores that fell short of the pith, with strongly curved innermost rings (54.6%), pith location was estimated graphically. Once pith location had been estimated, missing radius was estimated starting from the innermost ring (Norton et al., 1987). Rings taken from the innermost part of the core were then counted until a segment equal in length to the estimated missing radius was reached; this number was added to the number of rings in the core to obtain the estimated age of the tree at the coring height (Motta and Nola, 2001). This method assumes that the estimated missing rings form concentric circles. Individuals with a rotten core and those for which it was impossible to estimate pith location and thus the number of missing rings, were discarded (22.1%).

The estimated ages reported for all trees were calculated using cores collected at 50 cm above the root collar. Since the procedures utilized for age estimation can introduce errors into subsequent analyses, determining a precise age is not possible; instead, the age structure was constructed for 10-year classes (Payette et al., 1990).

### 2.5. Disturbance history

We found disturbances by identifying abrupt increases in radial growth (releases from suppression) in the pedunculate oak (*Quercus robur* L.) increment cores (trees in the dominant layer). Releases are growth increases, occurring synchronously in neighbouring trees and showing a slow decrease in the following years due to ageing or to

closure of the canopy (Kaennel and Schweingruber, 1995). We defined an abrupt increase as a > 166% increase in ring-width over the previous four years (Schweingruber, 1990). A chronology of all the disturbances recorded from the trees in the two plots was constructed and release data were summarized as the percentage of trees in a given decade that showed a growth release (Nagel et al., 2007; Rozas, 2005).

Releases from suppression chronology was compared to the black locust pattern of the establishment in order to identify which disturbances enhanced the black locust tree recruitment (Motta et al., 1999; Motta and Garbarino, 2003).

### 3. RESULTS

#### 3.1. Stand structure

The stands in both of the studied plots were dense, structurally complex and rich in biomass, as would be expected for stands that have been unmanaged for decades (Fig. 1; Tab. I). The volume of living biomass was higher in plot 2 than in plot 1 ( $512.7 \text{ m}^3 \text{ ha}^{-1}$  and  $405.7 \text{ m}^3 \text{ ha}^{-1}$ , respectively), as were basal area ( $35.3 \text{ m}^2 \text{ ha}^{-1}$  and  $28.9 \text{ m}^2 \text{ ha}^{-1}$ , respectively) and the density of all trees ( $589 \text{ individuals ha}^{-1}$  and  $571 \text{ individuals ha}^{-1}$ , respectively). Eleven species were represented in the two plots: pedunculate oak (*Quercus robur* L.), black locust, elm (*Ulmus minor* Mill.), hornbeam (*Carpinus betulus* L.), white poplar (*Populus alba* L.), field maple (*Acer campestre* L.), hawthorn (*Crataegus monogyna* Jacq.), hazelnut (*Corylus avellana* L.), blackthorn (*Prunus spinosa* L.), bird cherry (*Prunus padus* L.) and black poplar (*Populus nigra* L.). While several of these species were well represented, based on volume and basal area, oak and black locust were the dominant woody species. Oak comprised 48.2% of the volume in plot 1 and 42.8% in plot 2; black locust comprised 35.6% of the volume in plot 1 and 44.3% in plot 2 (Tab. I). The dominance of these two species was also reflected in the percent of overall basal area which, when combined for both species, was 77.4% in plot 1 and 86.2% in plot 2.

Tree size, as measured by trunk diameter, showed a J-shape distribution (Fig. 2). Oaks were generally clustered in the larger diameter classes, black locust in the intermediate (more) and larger (fewer) classes and the other broadleaves had smaller diameters. The calculated cut off points of the vertical layers were 22.8 and 27.4 m for the dominant layer (plots 1 and 2, respectively) and 14.5 and 17.7 m for the intermediate layer, also for plots 1 and 2. The remaining vertical layers were included in the dominated layers (below 14.5 and 17.7 m in plots 1 and 2, respectively).

Oak was present in the dominant and, with a low density, in the intermediate layers in both plots. Even though the black locust may have been represented in all three layers, it was represented primarily in the intermediate and dominant layers. The other broadleaves were concentrated in the dominated layers (Tab. II).

The regeneration layer includes seedlings, scrubs and dominated stems. Tree density in this layer was higher in plot 2 than in plot 1, with 8 680 and 4 060 individuals  $\text{ha}^{-1}$ , respectively. The regeneration layer was largely dominated by

other broadleaves; oak and the black locust were absent in plot 1 while their presence was marginal in plot 2, accounting for only 0.2% and 4.8%, respectively. The observed mortality rates (standing dead trees/standing living trees) in plots 1 and 2 were 7.8 and 10.0%, respectively, for the oak and 61.1 and 47.6%, respectively, for the black locust.

#### 3.2. CWD

The total volume of CWD was  $130.0 \text{ m}^3 \text{ ha}^{-1}$  in plot 1 and  $81.5 \text{ m}^3 \text{ ha}^{-1}$  in plot 2 (Tab. III) representing, respectively, 32.0 and 15.9% of the living biomass. The most represented CWD type were snags in plot 1 while in plot 2 snags and logs are equally represented. The presence of only a few stumps is reflective of both the fact that cutting has not occurred in the stand for decades and to a rapid decay rate. In many cases (> 95% of the total for snags and logs) it was possible to identify the species that comprised the CWD elements. In plots 1 and 2, black locust represented 72.2 and 82.3% of the total, oak represented 26.6 and 15.4% and the other broadleaves represented 1.2 and 2.3%, respectively. Because the wood of the black locust is more resistant to decay than either oak or the other broadleaves (European Committee for Standardisation, 1995) the proportion of black locust in CWD may be overestimated.

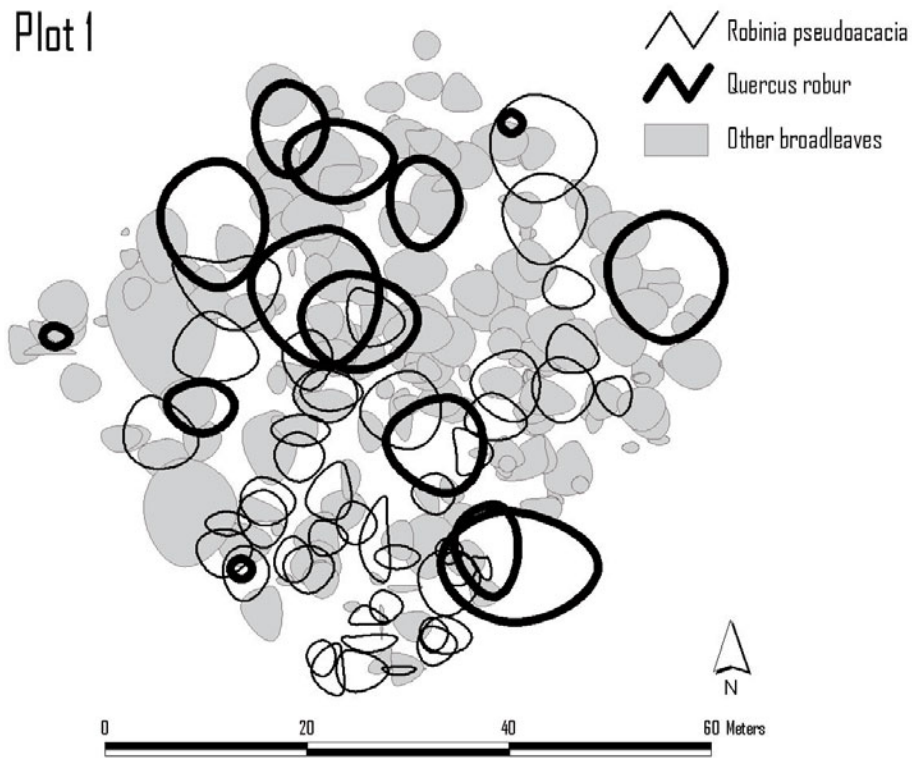
#### 3.3. Age structure

A total of 451 cores were collected in the two plots. The oldest trees were oaks and were determined to be 110 (1896–2005) years old in plot 1 and 148 (1858–2005) years old in plot 2. The succession pattern of trees was evident through evaluation of tree-age data. Oak was the initial dominant woody species (almost all the individuals older than 80 year are oaks) followed by black locust and the other broadleaves (Fig. 3). Oak and black locust trees demonstrated species-specific clumping into only few age classes. Oak trees were  $\geq 75$  y old in plot 1 and  $\geq 85$  years old in plot 2, while black locust trees were, for the most part, 45–65 years old, although a small number fell into the younger age classes. Ages of the other broadleaf trees had much wider ranges, extending from  $\leq 15$  y (both plots) to > 100 y (plot 1 only). Most of the other broadleaves fell into the  $\leq 75$  years old age classes and were the dominant species in the < 55 years old age classes.

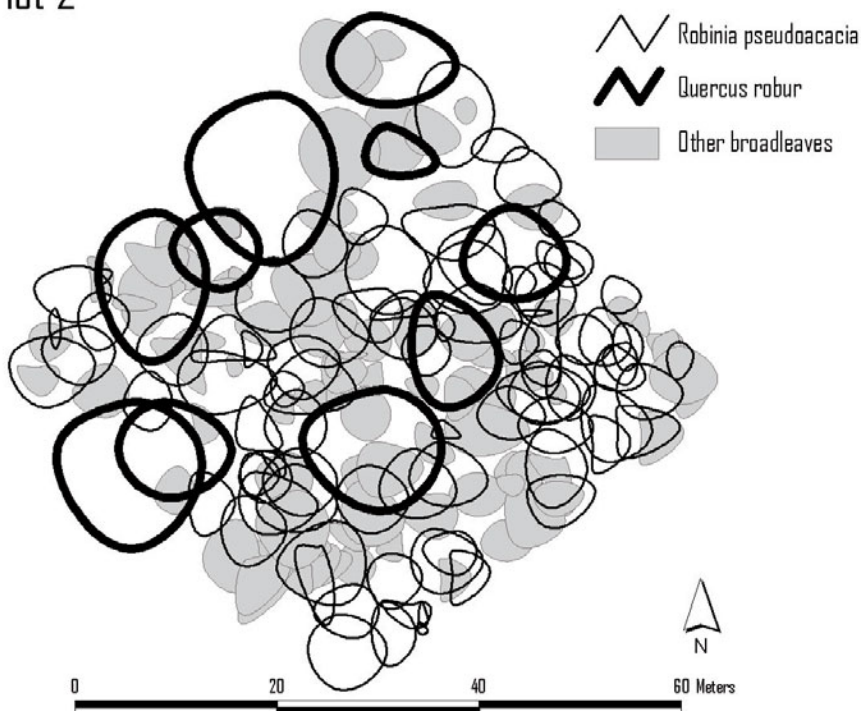
#### 3.4. Disturbance history

Disturbance history was inferred from patterns of release in the annual rings. Sixty-seven (67) releases were recorded. Releases occurred in almost all of the time periods, but three main disturbances (> 30% of individuals showing releases) were observed. The first main disturbance occurred in an extended period of time bracketing 1920, the second one in the 1950s and 1960s and the third one in 1980 (Fig. 4). Taking into account that it might take a few years for a tree to show a release after

Plot 1



Plot 2



**Figure 1.** Stand structure of the two sample plots.

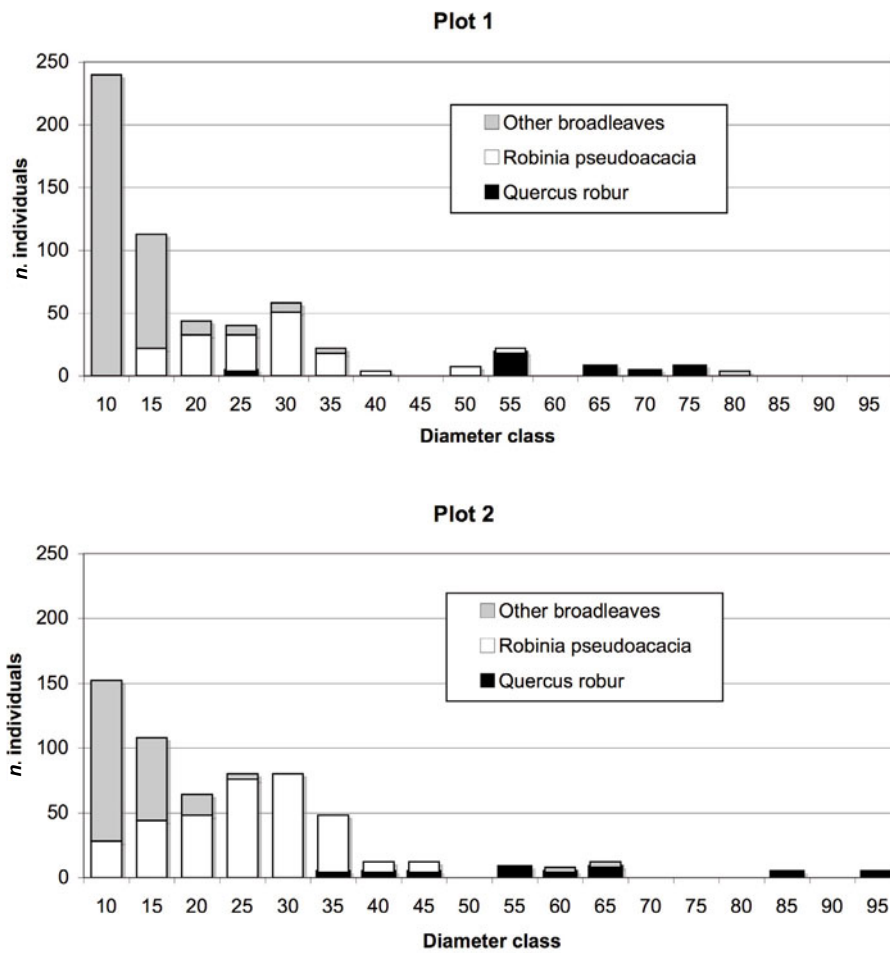
**Table I.** Characteristics of forest stands in the two sample plots.

(a) Plot 1.

Species	Density (ha <sup>-1</sup> )	Basal area (m <sup>2</sup> ha <sup>-1</sup> )	Volume (m <sup>3</sup> ha <sup>-1</sup> )	% of overall density	% of overall basal area	% of overall volume
<i>Quercus robur</i>	40	11.6	195.7	7.0	40.0	48.2
<i>Robinia pseudoacacia</i>	167	10.8	144.5	29.3	37.4	35.6
Other broadleaves	363	6.5	65.5	63.7	22.6	16.2
Total	571	28.9	405.7	100.0	100.0	100.0

(b) Plot 2.

Species	Density (ha <sup>-1</sup> )	Basal area (m <sup>2</sup> ha <sup>-1</sup> )	Volume (m <sup>3</sup> ha <sup>-1</sup> )	% of overall density	% of overall basal area	% of overall volume
<i>Quercus robur</i>	40	12.4	219.5	6.8	35.1	42.8
<i>Robinia pseudoacacia</i>	334	18.0	227.2	56.7	51.1	44.3
Other broadleaves	215	4.9	66.0	36.5	13.8	12.9
Total	589	35.3	512.7	100.0	100.0	100.0



**Figure 2.** Size distribution of woody species in the two sample plots.

**Table II.** Vertical structure of the forest stand in the two sample plots.

(a) - Plot 1.			
	Dominant layer	Intermediate layer	Dominated layers
<i>Quercus robur</i>	28.1	6.9	0.0
<i>Robinia pseudoacacia</i>	65.5	65.5	5.6
Other broadleaves	9.4	27.6	94.4

(b) - Plot 2.			
	Dominant layer	Intermediate layer	Dominated layers
<i>Quercus robur</i>	27.3	10.3	0.0
<i>Robinia pseudoacacia</i>	63.6	87.2	35.6
Other broadleaves	9.1	2.5	64.4

a disturbance because the tree crown must adapt to new light conditions, the first two disturbances coincide with the two world wars and subsequent years, two periods that saw an elevated demand for firewood. The third main disturbance in the 1980s was due to the spread of an elm disease that caused sudden death of the native elms (Sartori pers. comm.) which, at that time, were the second species of the dominant layer in the "Siro Negri" Forest Reserve (Tomaselli and Gentile, 1971).

#### 4. DISCUSSION

Disturbance history and establishment data (Fig. 4) show that the black locust became established almost exclusively during and after the second major disturbance on record (1940–1960). No black locust colonization was documented following the first disturbance in 1920, and only a few individuals were evidently established after the third disturbance in 1980. It is important to remember that the present age structure reflects both establishment and subsequent survival of individuals up to the moment of sample collection. The absence of trees dating from a given period may be due to a lack of establishment and/or high mortality rates of trees regenerating at that time (Johnson et al., 1994). In this case, we can reject the hypothesis that black locust individuals were, in fact, established after the first disturbance in 1920 and then successively cut because black locust usually displays vigorous vegetative reproduction by root suckering and stump sprouting. If there were black locust trees in the 40 s, a cut should have enhanced vegetative reproduction from stumps. In the two plots studied, however, there was no evidence of black locust sprouts resulting from cuts; all black locust stems were individuals and there are no stumps but only single stems.

The situation was completely different in the 1980s when black locust was well established in the two plots. At the beginning of the 1970s elm was an important component of the dominant layer, second only to oak (Tomaselli and Gentile, 1971). Although elm disease eliminated almost all of the dominant elms (Sartori, pers. comm.), mortality was randomly distributed in the stand and did not create large gaps. The residual canopy cover prevented the establishment of a new black locust cohort (only a small number of new black locusts were

**Table III.** Quantity and quality of the coarse woody debris.

	Total volume (m <sup>3</sup> ha <sup>-1</sup> )	Snags (m <sup>3</sup> ha <sup>-1</sup> )	Logs (m <sup>3</sup> ha <sup>-1</sup> )	Stumps (m <sup>3</sup> ha <sup>-1</sup> )
Plot 1	130.0	75.2	54.7	0.1
Plot 2	81.5	41.3	40.1	0.1

recorded) by limiting light penetration. Growth of other native broadleaves that were already present in the dominated layers was enhanced by the lower light levels.

The observed dynamics of the black locust in the Reserve are very similar to those observed in its native, North American range where its dominance is short-lived before it is replaced by longer-lived and more shade-tolerant species (Boring and Swank, 1984). After the black locust became firmly established for the first time, it occupied the dominant and intermediate layers but it did not reproduce in any substantial numbers, resulting in a general reduction in black locust populations over several decades. A high black locust mortality rate and large proportion of black locust biomass in the CWD are additional signs of the decline.

Our results strongly suggest that the best strategy for controlling black locust dispersion is to avoid disturbances that favour black locust colonization and to wait for natural suppression of the species by other tree species. The disturbance event of the 1970s occurred without inducing a new black locust cohort. Removal of a single tree or a small group of trees, that mimics disease-related elm mortality, could be a successful silvicultural strategy in stands where wood and commodities production must be performed but where control of the black locust must also be considered.

Regardless of historical data that provide a fairly clear picture of the dynamic role of the black locust in the Reserve, there remain several uncertainties, the impacts of which cannot be accurately predicted. One of the uncertainties is the supplemental nitrogen made available through the nitrogen-fixing capabilities of the black locust. In addition to enhanced plant growth, the rate of nitrogen cycling may also be increased, with potential effects on successional patterns, disturbance regime and community composition (Wedin and Tilman, 1996). Supplemental nitrogen may impede efforts to restore the native plant community and ecosystem function on sites previously occupied by exotic nitrogen-fixers (Rice et al., 2004). The incremental change of nitrogen levels in the soil is encouraged by nitrogen deposition which is one of the focal processes controlling the carbon balance in temperate and boreal forests (Magnani et al., 2007).

A second source of uncertainty is climate change (Harris et al., 2006) and the rate of increase of atmospheric CO<sub>2</sub>. There are still many unanswered questions about how increased temperatures will impact the growth of black locust and native species. However, experimental data show that the black locust exhibits a strong positive response to incremental increases of CO<sub>2</sub> (Mohan et al., 2007).

The third source of uncertainty is the potential spread of oak disease that has recently been found in many stands in central and southern Europe, but is not yet evident in the Siro

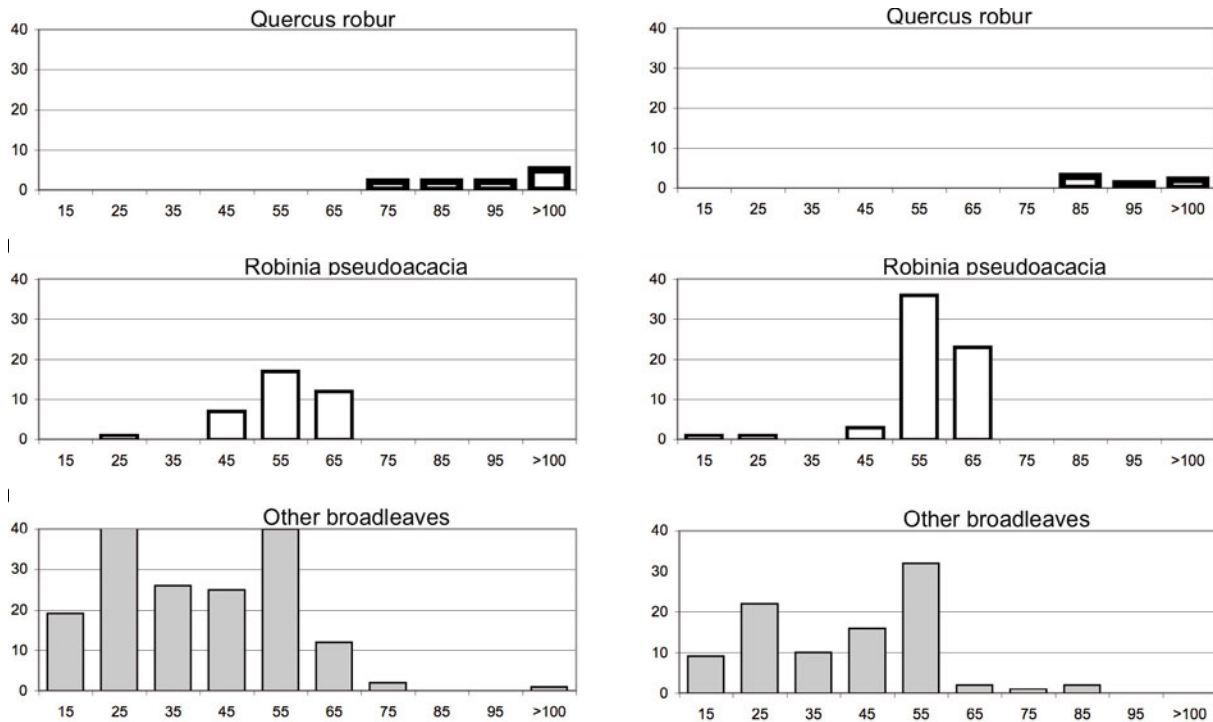


Figure 3. Age structure of woody species in the two sample plots.

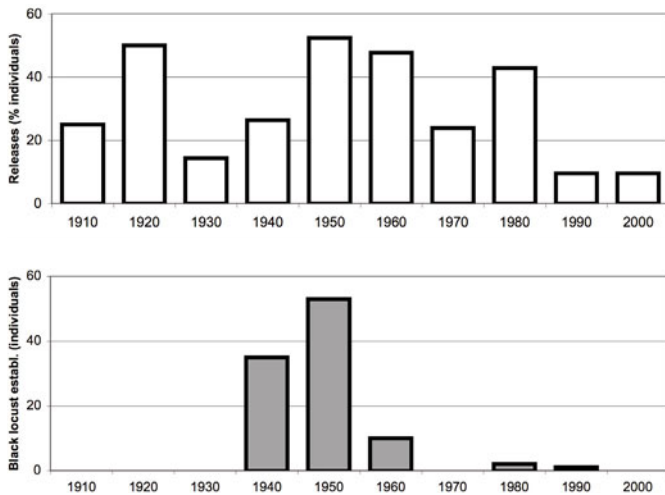


Figure 4. Releases (% of oak trees showing at least one release in the decade) and establishment of black locust individuals.

Negri Forest Reserve (Bergot et al., 2004; Ragazzi et al., 1989; Thomas et al., 2002). The consequences of elm disease that has almost completely removed the elm from these stands are another source of uncertainty. The role that elm tree played in historical system dynamics is not well known. However, the significant reduction in elm populations will undoubtedly alter future dynamic processes, leaving us with no quantitative

reference condition. In fact, the two most common references are historical data from the site to be restored (Swetnam et al., 1999) and contemporary data from reference sites chosen as good analogs of the site to be restored (White and Walker, 1997). In the Ticino alluvial forests it is impossible to obtain such information and a new range of reference conditions must be identified in order to restore at best the ecosystem given the changed biophysical conditions likely to occur in the future. Finally the present alluvial forests are canopy closed systems, in contrast to the past when most of the European alluvial forests, even those considered natural or semi-natural, were more canopy open, thanks to the grazing of wild and the domestic ungulates (Vera, 2000) that are currently absent in the Reserve. Oak regeneration requires sufficient light penetration and large openings (Humphrey et al., 2002). While species such as hornbeam, elm and other broadleaves can successfully regenerate in small forest gaps, light-demanding species like oak and black locust are at a distinct disadvantage. Such was the case in the Reserve in the 1970s. If the gap exceeds a critical size, a new successional series would be initiated involving light-demanding trees, such as oak and black locust. Beginning with the closed forests of today, and without human intervention, if only small-scale gaps arise in the forest infrastructure (e.g. death of a single tree or small group of trees) the resulting forest will consist of mixed broadleaved species with a substantial reduction in the number of oaks and black locusts. Disturbances resulting in large openings will encourage oak regeneration, and are also likely to be favourable to black locust.



All of these uncertainties represent a problem for the management and conservation of the alluvial forests of the Ticino plain. For commercial forestry to be successful and acceptable in this area, it must be conceived and developed in a flexible and adaptable context that considers long-term management decisions under uncertain conditions, resulting in close-to-nature forestry and the conservation of the last remnants of semi-natural forests.

The legacies of past management, and the altered biophysical conditions, have influenced complex ecological dynamics and promoted development along a successional pathway that strongly differs from potential reference conditions (Kirkman et al., 2007). In this context, the close-to-nature silviculture and management approach must be designed to favour native species and natural evolutionary processes through the system's ability to increase in complexity and biodiversity (Nocentini, 2006). Due to the lack of past reference conditions, current conditions represent the only possible reference data for future comparisons. In this perspective, the Reserve and the long-term monitoring plots play a key role by allowing spatially consistent observations of natural processes in the absence of direct human intervention. The data collected from these and future studies may eventually narrow the range of uncertainty and provide useful management information (Buckley, 2008).

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## REFERENCES

- Bergot M., Cloppet E., Pérarnaud V., Déqué M., Marçais B., and Desprez-Loustau M.L., 2004. Simulation of potential range expansion of oak disease caused by *Phytophthora cinnamomi* under climate change. *Global Change Biol.* 10: 1539–1552.
- Boring L.R. and Swank W.T., 1984. The role of black locust (*Robinia pseudoacacia*) in forest succession. *J. Ecol.* 72: 749–766.
- Brothers T.S. and Spingarn A., 1992. Forest fragmentation and alien plant invasion of central Indiana old-growth forests. *Conserv. Biol.* 6: 91–100.
- Buckley Y.M., 2008. The role of research for integrated management of invasive species, invaded landscapes and communities. *J. Appl. Ecol.* 45: 397–402.
- Castellani C., Scrinzi G., Tabacchi G., and Tosi V., 1984. Inventario forestale nazionale italiano. Tavole di cubatura a doppia entrata, Ministero dell'Agricoltura e delle Foreste, Istituto Sperimentale per l'Assessmento Forestale e per l'Alpicoltura, Trento.
- Driesche J.V. and Driesche R.V., 2004. Nature out of place: biological invasions in the global age, Washington, Island Press.
- Dzwonko Z. and Loster S., 1997. Effects of dominant trees and anthropogenic disturbances on species richness and floristic composition of secondary communities in southern Poland. *J. Appl. Ecol.* 34: 861–870.
- European Committee For Standardization, 1995. Durability of wood and woodbased products – natural durability of solid wood – Part 2: Guide to natural durability and treatability of selected wood species of importance in Europe, Brussel, EN 350-2.
- Ferraris P., Terzuolo P., Brenta P.P., and Palenzona M., 2000. La robinia. Indirizzi per la gestione e la valorizzazione. Peveragno (CN), Regione Piemonte, Blu Edizioni.
- Fowells H.A., 1965. Silvics of forest trees of the United States, Washington D.C., USDA.
- Harris J.A., Hobbs R.J., Higgs E., and Aronson J., 2006. Ecological restoration and global climate change. *Restor. Ecol.* 14: 170–176.
- Humphrey J.W., Davey S., Peace A.J., Ferris R., and Harding K., 2002. Lichens and bryophyte communities of planted and semi-natural forests in Britain: the influence of site type, stand structure and deadwood. *Biol. Conserv.* 107: 165–180.
- Johnson E.A., Miyanishi K., and Kleb H., 1994. The hazards of interpretation of static age structures as shown by stand reconstruction in *Pinus contorta* – *Picea engelmannii* forest. *J. Ecol.* 82: 923–931.
- Kaennel M. and Schweingruber F.H., 1995. Multilingual glossary of Dendrochronology, Berne, Wsl/Fnp Birmensdorf, P. Haupt Pub.
- Kirkman L.K., Mitchell R.J., Kaeser M.J., Pecot S.D., and Coffey K.L., 2007. The perpetual forest: using undesirable species to bridge restoration. *J. Appl. Ecol.* 44: 604–614.
- Latham P.A., Zuuring H.R., and Coble D.W. 1998. A method for quantifying vertical forest structure. *For. Ecol. Manage.* 104: 157–170.
- Magnani F., Mencuccini M., Borghetti M., Berbigier P., Berninger F., Delzon S., Grelle A., Hari P., Jarvis P.G., Kolar P., Kowalski A.S., Lankreijer H., Law B.E., Lindroth A., Loustau D., Manca G., Moncrieff J.B., Rayment M., Tedeschi V., Valentini R., and Grace J., 2007. The human footprint in the carbon cycle of temperate and boreal forests. *Nature (London)* 447: 848–850.
- McDonald R.I. and Urban D.L., 2006. Edge effects on species composition and exotic species abundance in the North Carolina Piedmont. *Biol. Invasions* 8: 1049–1060.
- Mohan J.E., Clark J.S., and Schlesinger W.H., 2007. Long-term CO<sub>2</sub> enrichment of a forest ecosystem: implications for forest regeneration and succession. *Ecol. Appl.* 17: 1198–1212.
- Mondino G.P. and Scotta M., 1987. *Robinia pseudoacacia* L. nell'ambiente forestale piemontese. *Inf. Bot. Ital.* 43–48.
- Motta Fré V. and Motta R., 2000. Selvicoltura e ciliegio tardivo (*Prunus serotina* Ehrh.) nella Riserva Naturale Orientata "La Fagiana" (Magenta -MI). *Sherwood* 6: 5–14.
- Motta R. and Garbarino F., 2003. Stand history and its consequences for the present and future dynamic in two silver fir (*Abies alba* Mill.) stands in the high Pesio Valley (Piedmont, Italy). *Ann. For. Sci.* 60: 361–370.
- Motta R. and Nola P., 2001. Growth trends and dynamics in sub-alpine forest stands in the Varaita valley (Piedmont, Italy) and their relationships with human activities and global change. *J. Veg. Sci.* 12: 219–230.
- Motta R., Nola P., and Piussi P., 1999. Structure and stand development in three subalpine Norway spruce (*Picea abies* (L.) Karst.) stands in Paneveggio (Trento, Italy). *Global Ecol. Biogeogr.* 8: 455–471.
- Nagel T.A., Levanic T., and Diaci J., 2007. A dendroecological reconstruction of disturbance in an old-growth *Fagus-Abies* forest in Slovenia. *Ann. For. Sci.* 64: 891–897.
- Nocentini S., 2006. La rinaturalizzazione dei sistemi forestali: è necessario un modello di riferimento? *Forest@* 3: 376–379.
- Nola P., 1991. Primo approccio alla dendroclimatologia della quercia (*Quercus robur* L. e *Quercus petraea* (Mattuschka) Liebl.) in Pianura Padana (Italia Settentrionale). *Dendrochronologia* 9: 71–94.

- Nola P., 1996. Climatic signal in earlywood and latewood of deciduous oaks from northern Italy. In: Dean J.S., Meko D.M., and Swetnam T.W. (Eds.), *Tree rings, environment and humanity*, Radiocarbon, Tucson, pp. 249–258.
- Norton D.A., Palmer J.G., and Ogden J., 1987. Dendroecological studies in New Zealand I. An evaluation of tree age estimates based on increment cores. *N. Z. J. Bot.* 25: 373–383.
- Payette S., Filion L., and Delwaide A., 1990. Disturbance regime of a cold temperate forest as deduced from tree-ring patterns: the Tantaré Ecological Reserve, Québec. *Can. J. For. Res.* 20: 1228–1241.
- Pividori M. and Grieco C. 2003. Evoluzione strutturale di popolamenti cedui di robinia (*Robinia pseudoacacia* L.) nel Canavese (Torino – Italia). *Schweiz. Z. Forstwes.* 154: 1–7.
- Ragazzi A., Fedi I.D., and Mesturino, L. 1989. The oak decline: a new problem in Italy. *Eur. J. For. Pathol.* 19: 105–110.
- Rice S.K., Westerman B., and Federici R., 2004. Impacts of the exotic, nitrogen-fixing black locust (*Robinia pseudoacacia*) on nitrogen-cycling in a pine-oak ecosystem. *Plant Ecol.* 174: 97–107.
- Rinn F., 1996. *TSAP Reference Manual*. Version 3.0. Heidelberg.
- Rozas V., 2005. Dendrochronology of pedunculate oak (*Quercus robur* L.) in an old-growth pollarded woodland in northern Spain: establishment patterns and the management history. *Ann. For. Sci.* 62: 13–22.
- Sartori F., 1984. Les forêts alluviales de la basse vallée du Tessin (Italie du nord). In: Cramer J. (Ed.), *Colloques phytosocologiques, la végétation des forêts alluviales*, Berlin, pp. 201–216.
- Schnitzler A., 1994. European alluvial hardwood forests of large floodplains. *J. Biogeogr.* 21: 605–623.
- Schnitzler A., Hale B.W., and Alsum E.M., 2007. Examining native and exotic species diversity in European riparian forests. *Biol. Conserv.* 138: 146–156.
- Schweingruber F.H., 1990. Dendroecological Information in Pointer Years and Abrupt Growth Changes. In: Cook E.R. and Kariukstis L.A. (Eds.) *Methods of dendrochronology*. Kluwer, Dordrecht, pp. 276–283.
- Swetnam T.W., Allen C.D., and Betancourt J.L., 1999. Applied historical ecology: using the past to manage for the future. *Ecol. Appl.* 9: 1189–1206.
- Thomas F.M., Blank R., and Hartmann G., 2002. Abiotic and biotic factors and their interactions as causes of oak decline in Central Europe. *For. Pathol.* 32: 277–307.
- Tomaselli R. and Gentile S., 1971. La Riserva naturale integrale “Bosco Siro Negri” dell’Università di Pavia. *Atti Ist. Bot. Lab. Critt. Univ. Pavia* 6: 41–70.
- Vera F.W.M., 2000. *Grazing ecology and forest history*. Wallingford, Cabi Publishing.
- Wedin D.A. and Tilman D., 1990. Influence of nitrogen loading and species composition on the carbon balance of grasslands. *Science (Washington)* 274: 1720–1723.
- White P.S. and Walker J.L., 1997. Approximating nature’s variation: selecting and using reference information in Restor. *Ecol. Restor. Ecol.* 5: 338–349.
- Wilcove D.S., Rothstein D., Dubow J., Phillips A., and Losos E. 1998. Assessing the relative importance of habitat destruction, alien species, pollution, over-exploitation, and disease. *BioScience* 48: 607–616.