Chapter 23
Utilization of Non-native Wood by Saproxylic Insects

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Abstract Whether intentionally or accidentally introduced, non-native woody plants now feature prominently in many ecosystems throughout the world. The dying and deadwood produced by these plants represent novel resources for saproxylic insects, but their suitability to these organisms remains poorly understood. We herein review existing knowledge about the utilization of non-native wood species by saproxylic insect communities and also provide several previously unpublished case studies from the USA, Germany, Portugal/Spain, and New Zealand. The first case study suggests that the relative number of beetle species utilizing non-native vs. native wood varies greatly among wood species, with some non-native species (e.g., *Albizia julibrissin*) supporting a high beetle diversity. A decomposition experiment found that termites did not readily attack three non-native wood species and did not contribute significantly to their decomposition in contrast to what has been shown for a native pine species. The second case study found two species of non-native wood to support a lower richness of beetles compared to two native wood species in Germany, with *Pseudotsuga menziesii* supporting particularly few species which formed just a small subset of the community collected from native *Picea abies*. The third case study, from Iberia, found *Eucalyptus* to support a relatively small number of insect species with generalist host preferences. The fourth

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case study provides a list of insects reported from non-native pine and *Eucalyptus* in New Zealand. Based on our literature review and these new case studies, we conclude that non-native wood species can support diverse insect assemblages but that their suitability varies greatly depending on host species as well as the host specificity of the insect(s) under consideration. Although many generalist species appear capable of using non-native woody resources, more research is needed to determine whether non-native wood species have any value in promoting the conservation of the most threatened taxa.

23.1 Introduction

Non-native woody plants are becoming increasingly common in forested landscapes around the world (Richardson and Rejmánek 2011). Many of these species have been and continue to be introduced intentionally, with motivations ranging from an inherent appreciation for novelty among horticulturalists and landowners (Buchler et al. 1981; Reichard and White 2001; Spongberg 1990); a commercial interest in specific non-timber forest products (Robb and Travis 2013); a desire to stabilize hill slopes, soils, and river banks (Wilkinson 1999) or reduce flooding (Dray et al. 2006; Kon et al. 1993); and demand for fast-growing species for use in timber production (Richardson and Rejmánek 2011). While the global forest area is declining, the area of planted forests is increasing and now covers 278 M ha or ~7% of the total global forest area. Most planted forests are of native species; however, 18–19% are plantations of primarily even-aged stands of non-native tree monocultures (Payn et al. 2015). At the same time, many non-native tree and woody shrub species, including some plantation species, have escaped cultivation and have developed self-sustaining (i.e., “naturalized”) populations across large areas (Essl et al. 2010; Rejmánek 2014; Ledgard 2001). There is currently great interest in knowing how the expansion of non-native plantations and the spread of non-native woody plants may affect biodiversity and ecosystem processes (Brockerhoff et al. 2008; Gibson et al. 2011; Felton et al. 2013; Peterken 2001; Krumm and Vítková 2016).

A number of factors are thought to influence the diversity of insects utilizing non-native plants. Southwood (1961) showed a positive correlation between the diversity of insect herbivores and the cumulative abundance of a tree species over time, with the rarest and most recently introduced tree species supporting the fewest insect species. Support for this hypothesis was reported by Brändle et al. (2008) in Germany who showed the species richness of herbivorous insects feeding on non-native plant species increased with time since the host plant was introduced. It has also been shown that the diversity of insects associated with a particular host plant is generally higher for widely distributed species than for species confined to smaller geographic areas (Branco et al. 2015a). Phylogenetic relatedness between non-native and native plant species has also been found to be important in determining the diversity of insects utilizing a non-native species. In Europe, for example, Branco et al. (2015a) showed that the number of native forest insects on non-native tree species, and also the damage they caused, was higher when native congeneric trees were present. Gossner et al. (2009) conclude from their study on phytophagous insect communities on introduced and native tree species in Europe that phylogenetic conservatism is an important mechanism in explaining communities on introduced trees, but whether it outweighs other mechanisms such as geographic contingency and mass effects, i.e., immigration from locally abundant plant species, depends on the interplay of phylogenetic scale, local abundance of native tree species, and the biology and evolutionary history of the phytophage taxon.

While the hypotheses summarized above were primarily developed for green food webs (i.e., based on living plant material), it remains poorly understood how the production of dead plant material by non-native woody plants may affect brown food webs. Herbivorous insects feeding on living plant tissues exhibit a high degree of host specificity, and this holds true for insects that feed on dying and recently deadwood (Stokland et al. 2012). However, as decomposition proceeds, woody material becomes increasingly dominated by fungi, and saproxylic (i.e., dependent on dying and deadwood) insect communities are known to become less host specific (Stokland et al. 2012). This pattern suggests non-native woody plants may have a stronger influence on green food webs than brown food webs. Indeed, a recent meta-analysis found invasive plants affect the trophic structure of green and brown food webs differently with the nature of the difference varying among ecosystems (McCary et al. 2016). However, none of the studies used in that analysis focused on wood-dependent species, underscoring a shortage of studies on the utilization of non-native wood species by saproxylic insects.

A number of studies have employed passive trapping techniques (e.g., flight intercept traps) to compare saproxylic insect diversity between non-native and native tree species (Lachat et al. 2007; Gossner and Ammer 2006; Gossner 2004). While these efforts typically indicate lower diversity associated with non-native tree species (Lachat et al. 2007; Puker et al. 2014; Buse et al. 2010), there are indications that such differences can be largely determined by microclimatic conditions which vary with stand composition, stratum, and year (Gossner and Ammer 2006). Furthermore, it is difficult to disentangle the effects of wood quality from those of wood quantity when the availability of woody debris is typically lower in younger forests dominated by non-native species. In western Africa, for example, Lachat et al. (2006) found the volume of wood in natural forests to be 7- and 25-fold greater than that in two non-native plantations. The variety of woody debris also varied greatly in that study, with a particular shortage of highly decomposed wood and standing dead trees in the non-native plantations. Another complicating factor within managed forests is the effects of silvicultural practices, e.g., pruning and thinning and the final harvesting that create large pulses of deadwood within the forest systems with little natural mortality in between. To isolate the effects of wood quality from those of wood quantity and variety, this paper focuses on studies in which insects were collected directly from woody substrates either by hand or by using emergence devices. We provide an overview of the published literature (Table 23.1) as well as several unpublished case studies from our own research. This chapter is divided into two parts to distinguish between naturalized non-native

Table 23.1 Chronology of studies comparing saproxylic insect communities between locally native and non-native tree species

Country	Non-native species	Native species	Decay class	Main findings	Citation
Benin	*Tectona grandis* and *Senna siamea*	*Afzelia africana*, *Anogeissus leiocarpus*, *Ceiba pentandra*, *Dialium guineense*, *Diospyros mespiliformis*, *Mimusops andongensis*, and *Zanthoxylum zanthoxyloides*	I–IV	Higher rarefied species richness in wood from natural forests compared to wood from non-native plantations. Community composition also varied considerably between native and non-native wood species.	Lachat et al. (2006)
Brazil	*Eucalyptus urophylla*, *Melia azedarach*, *Pinus elliottii*	*Lophantera lactescens* Ducke, *Inga marginata* Willd	I–II	*E. urophylla* and *M. azedarach* were, respectively, the most (100%) and least (17%) readily attacked by termites.	Trevisan et al. (2008)
France	*Picea sitchensis*, *Pinus strobus*, *Abies grandis*, *Pseudotsuga menziesii*, *Thuja plicata*	*Picea abies*, *Pinus sylvestris*, *Abies alba*	I	No significant difference in bark beetle species richness between native and non-native wood species but community composition varied considerably.	Bertheau et al. (2009)
New Zealand	*Pinus radiata*	*Schefflera digitata*, *Melicytus ramiflorus*, *Aristotelia serrata*	II	Greater beetle abundance and richness associated with *P. radiata* than with native wood species	Sky (2011)
Germany	*Pseudotsuga menziesii*	*Acer* sp., *Betula pendula*, *Carpinus betulus*, *Fagus sylvatica*, *Fraxinus excelsior*, *Larix decidua*, *Picea abies*, *Pinus sylvestris*, *Populus* sp., *Prunus avium*, *Quercus* sp., and *Tilia* sp.	I–II	Few beetle species associated with *P. menziesii*	Müller et al. (2015), Gossner et al. (2016)

Italy	*Robinia pseudoacacia*	*Populus alba* and *Quercus robur*	II	No difference in beetle richness or composition among tree species	Della Rocca et al. (2016)
Poland	*Quercus rubra*	*Quercus robur* and *Quercus petraea*	Mature trees	No evidence of *Cerambyx cerdo* associated with *Q. rubra*	Oleksa and Klejdysz (2017)
Chile	*Pinus radiata*	*Nothofagus* spp.	I–IV	Beetle richness and abundance lower in *Pinus* vs. *Nothofagus*	Fierro et al. (2017)
USA	*Ligustrum sinense*, *Melia azedarach*, *Albizia julibrissin*	*Fraxinus pennsylvanica*, *Quercus phellos*, *Liquidambar styraciflua*	I–II	Beetle richness and composition differed among the six wood species. The non-native species *A. julibrissin* and *L. sinense* yielded the second highest and lowest beetle richness, respectively.	Current chapter, case study 1
Germany	*Pseudotsuga menziesii*, *Quercus rubra*	*Picea abies*, *Quercus robur*	I	Beetle abundance, richness, and diversity lower in *P. menziesii* than all other species	Current chapter, case study 2

Only studies that collected insects directly from the wood are included here woody plants (e.g., invasive species) (Part I) and those that have been planted intentionally (e.g., non-native plantations) (Part II).

23.2 Part I: Invasive Non-native Species

Richardson and Rejmánek (2011) compiled a global list of 622 species of invasive non-native trees and shrubs. They found Australia to have the highest number of species (183) followed by Southern Africa (170), North America (163), the Pacific Islands (147), and New Zealand (107). The top reasons they cited for these introduction were horticulture (62%) followed by forestry (13%), food production (10%), and agroforestry (7%). While many naturalized plant species occur at low densities and go largely unnoticed, others form thick monocultures, with the potential to completely displace native forest ecosystems (Fig. 23.1b). Although generally unwanted and perceived to have mostly negative effects (Peltzer et al. 2015), invasive non-native species can sometimes provide important services, e.g., by allowing forests to recolonize abandoned agricultural areas (Lugo 2004) and contributing to forest ecosystem services such as erosion control, flood mitigation, and pollination (Branco et al. 2015b). The degree to which the deadwood produced by invasive non-native woody plant species is utilized by saproxylic insects remains mostly unknown, but several recent studies suggest this may depend on the species of tree as well as the insect(s) under consideration. Della Rocca et al. (2016) reported no difference in the species richness or composition of beetle communities associated with wood from an invasive non-native tree species (*Robinia pseudoacacia* L.) and that from two native tree species (*Populus alba* L. and *Quercus robur* L.) in Italy. By contrast, Oleksa and Klejdysz (2017) found that *Cerambyx cerdo* L., a threatened cerambycid beetle strictly associated with mature oaks (especially *Q. robur*) in Europe, did not utilize the non-native *Q. rubra* in Poland. Although the absence of *C. cerdo* from *Q. rubra* may have been due in part to the smaller size of *Q. rubra* compared to the native oaks examined in that study, these findings suggest non-native oaks may provide little benefit to this threatened beetle in Europe. Because host specificity is one characteristic common to many threatened insect taxa, species of greatest conservation concern may be less accepting of non-native wood species than less vulnerable species. Taken together, these findings suggest non-native woody plants can provide suitable resources for many saproxylic insects although this may be less true for species specific to one or several host species. In the following case study, we present the results from an effort undertaken in the southeastern USA to compare saproxylic insect activity among several non-native and native woody plant species.

Fig. 23.1 Examples of forests dominated by non-native woody plants: (**a**) Douglas-fir plantation in Germany (M. Gossner); (**b**) invasive Chinese privet in Georgia, USA (M. Ulyshen); (**c**) *Eucalyptus* plantation in southern Brazil (G. Overbeck); (**d**) *Pinus radiata* plantation in New Zealand (M. Ulyshen); (**e**) invasive camphor tree in Mississippi, USA (M. Ulyshen)

23.2.1 Case Study I: Utilization of Non-native Wood Species by Beetles and Termites in the Southeastern USA—Implications for Conservation and Insect-Mediated Decomposition

The trade of tree species between North America, Europe, and Asia has a long history (Spongberg 1990), and the lasting effects of these exchanges are evident today throughout the USA. Of the 163 species of invasive non-native trees and shrubs reported from North America by Richardson and Rejmánek (2011), at least 20 are rapidly invading the forests of the southeastern USA (Miller 2003; Miller et al. 2010). We selected three non-native woody plant species for use in this study: *Ligustrum sinense* Lour. (Chinese privet), *Albizia julibrissin* Durazz. (mimosa), and *Melia azedarach* L. (chinaberry). We also selected three native species: *Fraxinus pennsylvanica* Marsh. (green ash), *Quercus phellos* L. (willow oak), and *Liquidambar styraciflua* L. (sweetgum). These species were selected primarily on the basis of their abundance and availability in the area although green ash was chosen because it belongs to the same family (Oleaceae) as Chinese privet. All three of the non-native species originated in Asia and were introduced into the USA for ornamental purposes in the eighteenth or nineteenth centuries. Chinese privet is an evergreen shrub that forms dense thickets that prevent native plant regeneration (Fig. 23.1b). Mimosa (a legume) and chinaberry both grow commonly in disturbed or open conditions throughout the southeastern USA but, unlike Chinese privet, rarely form dense monocultures. This two-part study sought to (1) compare saproxylic beetle communities among the three non-native and three native wood species mentioned above and (2) quantify termite feeding activity in the three non-native wood species and the contributions of these insects to decomposition.

23.2.1.1 Methods

Beetle Community Analysis

Forty sections measuring 0.5 m in length and 14.0 ± 0.3 (range 7.6–27.2) cm in diameter were cut from the three non-native and three native woody plant species listed above (i.e., 240 sections in total) from trees felled for this purpose in forests near Athens, Georgia, USA, in May 2011. Ten sections from each tree species were placed at each of four hardwood-dominated floodplain forests within the upper Oconee river watershed in northeastern Georgia [i.e., Scull Shoals Experimental Forest (33°46′17.04″N 83°16′59.46″W), Watson Springs (33°41′59.26″N 83°17′43.18″W), Sandy Creek Nature Center (33°58′53.17″N 83°22′56.40″W), and the State Botanical Gardens of Georgia (33°54′2.93″N 83°23′16.46″W)]. Half of these sections were collected after 3 months in August 2011, whereas the rest were collected after 1 year in early June 2012. The collected sections were placed in rearing bags (Ulyshen and Hanula 2009) to collect all emerging insects over a period of 6 months. For each collection period (i.e., 3 months and 1 year), beetles from the five logs belonging to each species were pooled for each of the four sites. All captured beetles were either identified to species or assigned to morphospecies except for several excluded groups for which such designations were not possible given limits of time and expertise. Excluded groups were the family Ciidae, some Ptiliidae, the staphylinid subfamilies Aleocharinae and Pselaphinae, and male ambrosia beetles belonging to the genus *Xyleborus*. Altogether these excluded groups accounted for about 8% of all beetles captured.

We used sample-based rarefaction in EstimateS (Colwell 2013) to compare the number of species collected among the different wood species. Because there were large differences in abundance among wood species, we replotted the curves against an x-axis of individual abundance, as recommended by Gotelli and Colwell (2011). To compare the beetle communities collected from the six wood species, we performed nonmetric multidimensional scaling, PERMANOVA, two-way cluster analysis, and indicator species analysis using PC-ORD (McCune and Mefford 2011). For each sampling period and beetle species captured, we standardized abundance by wood volume. This was done by dividing the total wood volume (calculated from the length and diameter of each wood section) sampled for each tree species at each location by the largest volume of wood sampled for each sampling period. Beetle abundances were then divided by these values. We pooled standardized abundance across the two sampling periods to construct our main data matrix. Species present in less than three samples (i.e., the 24 combinations of tree species and site) were excluded, resulting in a final matrix consisting of 59 species. Abundance values were relativized by species maxima, and the Bray-Curtis distance measure was used in all analysis. For the two-way cluster analysis, we used the group average linkage method.

Termite Activity and Contributions to Wood Decomposition

Twenty logs 0.5 m in length were cut from forests growing near Athens (Georgia, USA) from each of the three non-native tree species mentioned above (i.e., Chinese privet, chinaberry, and mimosa) in June 2012. The trees were felled for this purpose in forests growing near Athens, Georgia. The logs were transported back to the laboratory in Mississippi. Image pro plus (Media Cybernetics, Inc., Rockville, MD) was used to calculate the original area of one end of each log (not including bark) using photographs taken of the cut ends. Half of the logs (ten of each species) were individually sealed within stainless steel mesh (0.38 mm openings) bags (“protected”), whereas the other logs were left “unprotected.” The same mesh bag design was used to successfully exclude termites and most beetle species in a previous study (Ulyshen 2014). At one site on the Noxubee Wildlife Refuge (northern Mississippi), a 5 m × 6 m grid was established (five line transects (rows), each containing six plots). Each of these 30 plots received a caged and an uncaged log from the same species (separated by about 0.5 m). Logs were collected in January 2015 after 31 months in the field. Following the methods described in Ulyshen (2014), two 3–4 cm-thick disks were cut from the end of the log that had been photographed at the beginning of the study. The thickness of the interior disk was measured at four equidistant points, and the mean thickness and the original surface area of the log, as measured from the initial photographs, were used to calculate the initial wood volume of the disk. Cut disks were photographed before oven drying them at 102 °C for 24 h. Following Ulyshen et al. (2016), a 20 × 20 grid was superimposed over the images and the percentage of cells with visible damage from termites was determined. After measuring the final dry weights of the disks, those with visible termite activity were burned to extract and subsequently quantify the mass of termite-imported soil. The soil weight was subtracted from the final dry disk weights following Ulyshen and Wagner (2013). Specific gravity was measured by dividing final dry wood weight (without soil) by initial wood volume (as defined above). Effect sizes (Hedges' d) and their 95% confidence intervals were calculated from these specific gravity measurements (unprotected-protected). A negative effect size suggests insects sped up wood loss (resulting in lower specific gravity) relative to the protected treatment, and the effect is considered significant when confidence intervals do not include zero. Finally, we calculated the decay rate constants for the three tree species based on their initial and final specific gravities using the single exponential model. Initial specific gravity was based on disks collected from the trees when they were initially felled (0.44, 0.57, and 0.50 for mimosa, privet, and chinaberry, respectively). Only unprotected logs were included in these calculations.

23.2.1.2 Results

Beetle Community Analysis

Overall, 2810 beetles representing 127 species or morphospecies were collected in this study. The highest total numbers of species/individuals came from sweetgum (69/1001), followed by mimosa (55/712), willow oak (55/665), chinaberry (43/242), green ash (32/98), and Chinese privet (27/92) (Fig. 23.2). Nonmetric

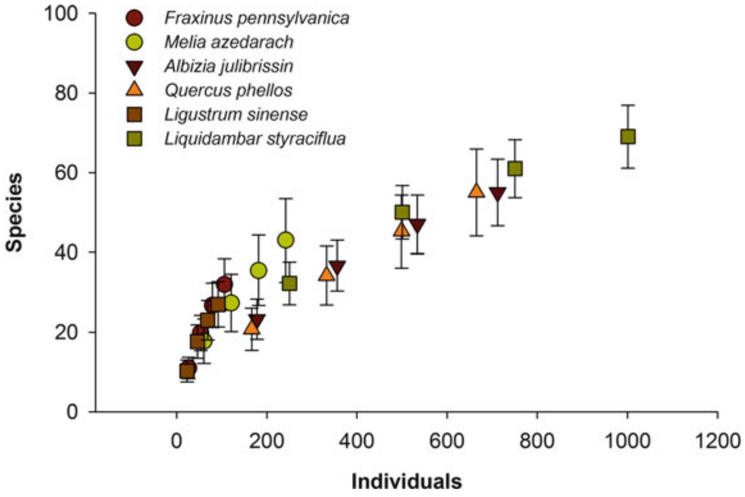

Fig. 23.2 Sample-based rarefaction from case study 1

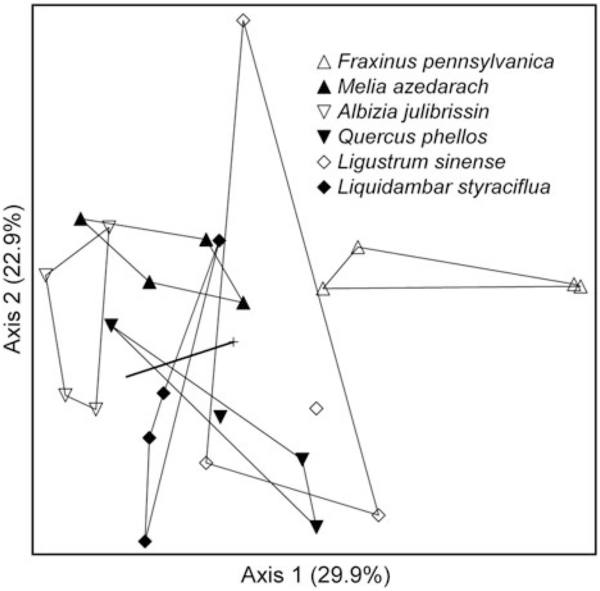

Fig. 23.3 NMS ordination of beetle assemblages among the six tree species for case study 1. Each symbol represents a separate set of logs from which insects were collected and the different symbols represent different species.

multidimensional scaling yielded a three-dimensional solution with a stress of 16.6. The ordination is depicted in two dimensions in Fig. 23.3, using the axes with the highest R^2 values (0.30 and 0.23 for axes 1 and 2, respectively). The ordination shows considerable separation in community composition among many of the tree species, and the superimposed joint plot shows the strength and direction of correlation with total beetle richness. All but 4 of the 15 pairwise comparisons between tree species were statistically significant based on PERMANOVA. These were green ash vs. Chinese privet ($p = 0.06$), chinaberry vs. willow oak ($p = 0.05$), mimosa vs. willow oak ($p = 0.05$), and willow oak vs. Chinese privet ($p = 0.06$). Two-way cluster analysis yielded two clusters for wood species, one consisting of green ash, chinaberry, and Chinese privet and the other consisting of the other three species (Fig. 23.4). Finally, based on indicator species analysis, one beetle species was significantly associated with green ash [*Tricorynus* sp. (Indicator Value (IV) = 75; $p = 0.01$)], one with chinaberry [*Dendroides canadensis* LeConte (Pyrochroidae) (IV = 75; $p = 0.01$)], five with mimosa [*Placonotus zimmermanni* (LeConte) (Laemophloeidae) (IV = 57.2; $p < 0.01$); *Platysoma leconti* Marseul (Histeridae) (IV = 45.8; $p = 0.04$); *Silvanus muticus* Sharp (Silvanidae) (IV = 52.9; $p = 0.03$); *Cossonus corticola* Say (Curculionidae) (IV = 80.3; $p < 0.01$); *Aegomorphus quadrigibbus* (Say) (Cerambycidae) (IV = 94.9; $p < 0.001$)], two with willow oak [*Platydema ruficorne* (Stürm) (Tenebrionidae) (IV = 75.5; $p < 0.01$); *Platydema subcostata* Laporte and Brulle (Tenebrionidae) (IV = 68.6; $p = 0.01$)], and five with sweetgum [*Leptostylus asperatus* (Haldeman) (Cerambycidae) (IV = 96.7; $p < 0.001$); *Xylosandrus crassiusculus* (Motschulsky) (Curculionidae) (IV = 70.5; $p = 0.01$); *Urographus fasciatus* (DeGeer) (Cerambycidae) (IV = 63.3; $p = 0.04$);

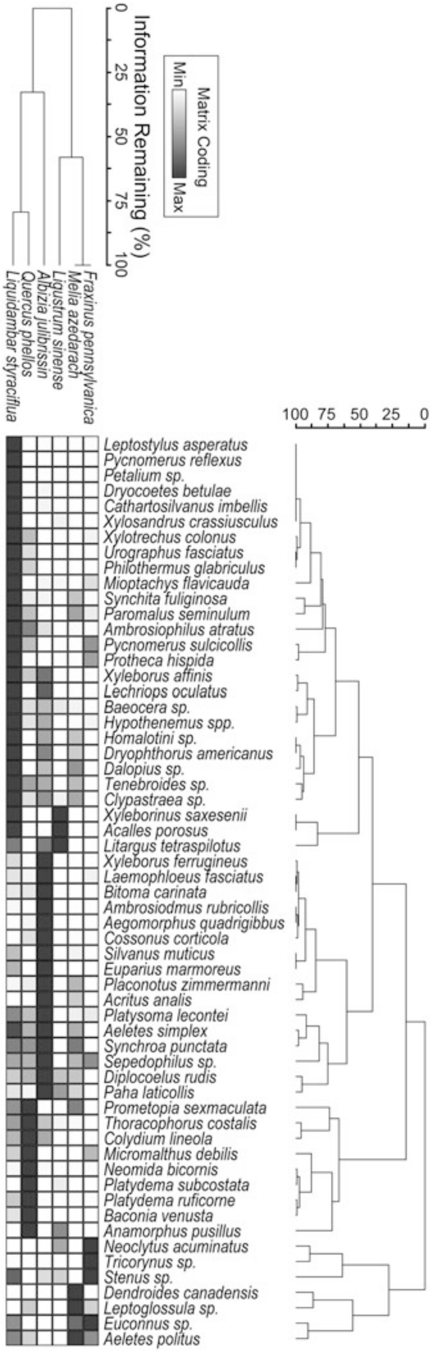

Fig. 23.4 Two-way cluster analysis of the six tree species and beetle species captured from at least three of the samples in case study 1

Pycnomerus reflexus (Say) (Zopheridae) (IV = 75; $p = 0.01$); *Petalium* sp. (Ptinidae) (IV = 100; $p < 0.001$)].

Termite Activity and Contributions to Wood Decomposition

Based on the percentage of grid cells with visible damage, Chinese privet experienced the highest level of termite activity (6.13% ± 1.27), followed by mimosa (3.48% ± 2.22) and chinaberry (2.17% ± 1.63). Effect size (unprotected-protected) was not statistically significant for Chinese privet [0.20 (−0.68 to 1.08)], mimosa [−0.23 (−1.11 to 0.65)], or chinaberry [0.03 (−0.84 to 0.91)], indicating that termites and other insects did not significantly accelerate decomposition of these species. The decay rate constant (k) for unprotected logs was 0.27, 0.10, and 0.12 for Chinese privet, mimosa, and chinaberry, respectively, corresponding to half-lives of 2.6, 6.9, and 5.8 years.

23.2.1.3 Discussion

Beetle abundance, richness, and composition varied greatly among the six wood species, with no consistent differences between non-native and native wood species. For example, the non-native mimosa yielded the second highest number of individuals and species, whereas Chinese privet, another non-native wood species, yielded the fewest. These findings are consistent with previous research from Europe where some non-native tree species (e.g., *R. pseudoacacia* in Italy) support diverse beetle assemblages (Della Rocca et al. 2016), whereas others (e.g., *Pseudotsuga menziesii* (Mirbel) Franco in Germany) are utilized by few species (Gossner et al. 2016) (Table 23.1). There is clearly a need for further research to better understand the extent to which various non-native wood species are utilized by saproxylic insect assemblages and how this might be influenced by the composition of the native plant community. As all 14 significant indicator beetle species are generalists known from a wide range of host species in our area, these findings are not remarkable. The fact that nearly half of them were more strongly associated with non-native wood species than with native wood species, however, reinforces the message that non-native trees can provide highly suitable resources for some species.

In our decomposition experiment, all three non-native wood species experienced low levels of termite activity after 31 months. This contrasts greatly with the much higher levels of termite activity observed after just 24 months in a previous study involving native loblolly pine (*Pinus taeda*). In that study, which used the same methods and took place in the same general area, an average of 42% and 25% termite damage was reported from *P. taeda* logs placed in unflooded and seasonally flooded forests, respectively (Ulyshen et al. 2016). The relatively low levels of termite activity observed in the three non-native wood species likely explain why we detected no significant effect of termites on wood decomposition in the current study, whereas Ulyshen (2014) found termites (among other insects) to significantly accelerate loblolly pine decomposition [Hedges *d* and 95% CIs: −0.74 (−1.39 to −0.10)] after 31 months in the same study area. Termites are known to strongly prefer certain types of wood over others, feeding less readily on the densest woods or on those containing high concentrations of extractives (Bultman and Southwell 1976). Results from other studies comparing how readily termites attack native and non-native wood species are mixed, suggesting that attack rate depends more on the properties of wood species than wood origin. In a comparison of three non-native and two native wood species in Brazil, for example, Trevisan et al. (2008) found non-native *Eucalyptus* and non-native chinaberry to be the most and least readily attacked, respectively (100 vs. 17%). This latter finding is consistent with our finding that chinaberry is highly resistant to termites.

While our results suggest insects may contribute less to the decomposition of non-native wood species than native wood species, this needs to be tested on a larger number of tree species before any broad conclusions can be reached. It also remains uncertain whether rates of fungal-driven decomposition vary between non-native and native wood species, but our results suggest the responses of fungi may differ from those of termites. Despite experiencing less termite activity than loblolly pine, for example, Chinese privet decomposed 25% faster (i.e., the decay rate for loblolly logs unprotected from insects for 31 months was 0.21). A recent meta-analysis found no difference in how quickly leaves and roots of non-native vs. native plant species decompose (Jo et al. 2016), suggesting that a species' place of origin is less important than interspecific differences in chemistry and perhaps other properties in determining decomposition rates.

23.3 Part II: Non-native Plantations

Planted forests, including even-aged single species stands, make up a growing proportion of the world forest cover, especially in the southern hemisphere, and commonly consist of fast-growing non-native genera like *Pinus*, *Eucalyptus*, and *Acacia* (Wingfield et al. 2015; Payn et al. 2015). Although forest cover is increasing throughout Europe, plantations of non-native trees are driving this pattern in many places (e.g., *Picea sitchensis* (Bong.) Carr. in Ireland and *Eucalyptus* in Spain) (Anonymous 2017). The non-native range of *Eucalyptus* is now particularly widespread and growing, with Asia currently having the largest coverage, followed by South America, Africa, Europe, and North America (Wingfield et al. 2015; Payn et al. 2015). The extent to which individual species have naturalized and subsequently spread beyond stand boundaries varies greatly and is influenced by a number of traits, e.g., seed size (Buckley et al. 2005). Some non-native tree species show relatively little potential for invasion (including many species of *Eucalyptus*; see Richardson (1998)), whereas others have escaped cultivation and pose a serious threat to native ecosystems (Richardson et al. 2011; Ledgard 2001). An analysis by Essl et al. (2010) suggests that conifer species used in commercial forestry have a significantly higher probability of becoming invasive than those planted for other purposes. Non-native monocultures are often planted in areas with a history of forest cover, but this is not always the case. Major afforestation efforts in China, for instance, are motivated in part by a desire to reduce flooding and erosion and sometimes include areas where forests never grew (Kon et al. 1993). Although generally thought to be bad for biodiversity, non-native plantations can provide important habitats for a wide range of native organisms [including threatened species (Pawson et al. 2010)] and can be particularly beneficial when established on degraded lands rather than displacing native ecosystems (Bremer and Farley 2010; Pawson et al. 2008; Lugo 2004). In their review of this topic, Bremer and Farley (2010) concluded that although non-native plantation forests support fewer specialist species than natural ecosystems, they should not be completely dismissed as "green deserts" by conservation biologists.

23.3.1 Case Study 2: Saproxylic Beetles Utilizing Crown Deadwood of Living Native and Introduced Trees in Germany

Central Europe is characterized by a comparatively low diversity of native tree species due to postglacial dispersal limitations (Normand et al. 2011). To increase available tree species in forestry, fast-growing native tree species, in particular Norway spruce *Picea abies* ((L.) H. Karst.), have been extensively planted outside their natural distributional ranges (Schelhaas et al. 2003). Such stands experienced large-scale losses following wind throws and bark beetle outbreaks, which are expected to further increase due to climate change (Pawson et al. 2013). This fostered the discussion on expanding the use of fast-growing non-native tree species, which may better adapt to future climatic conditions, in production forestry to reduce the risk of future stand losses (Bolte et al. 2009).

Douglas-fir (*Pseudotsuga menziesii* (Mirbel) Franco) introduced from western North America and red oak (*Quercus rubra* L.) introduced from the eastern North America are among the economically most important tree species in Central Europe (Nyssen et al. 2016). The ecological consequences of their introduction are, however, still debated (Schmid et al. 2014; Vor et al. 2016; Gossner 2004, 2016). Studies on the consequences for saproxylic insects are still rare and mostly consist of passively sampling adult individuals (Gossner and Ammer 2006; Gossner 2004) although some have involved experimental rearing from wood/log sections (Gossner et al. 2016). Here we test whether crown deadwood from introduced Douglas-fir and red oak can provide habitat for saproxylic beetles that is comparable to native Norway spruce (outside the natural range) and pedunculate oak (*Quercus robur* L.). Crown deadwood is known to contribute substantially to the volume of deadwood in forests, in particular for oak trees (Ammer et al. 2008), and it provides habitat for a specific saproxylic insect community (Bouget et al. 2011; Ulyshen and Hanula 2009).

23.3.1.1 Methods

The study was conducted in two forest stands in southern Bavaria, Germany. All study sites were located in a landscape called "Schotterriedel" which features soils of high nutrient content that promote rapid tree growth. European beech (*Fagus sylvatica* L.) would naturally dominate these landscapes ("Collin and High Montane Beech Forests"; Walentowski et al. 2006), but today Norway spruce (*P. abies*) is the tree species with the highest proportion in the surrounding forests. Annual precipitation reaches 750–800 mm and mean annual temperature is 7–8 °C.

Conifers were studied in a mature mixed Douglas-fir–spruce forest (mean age 94, 84–104) of 6.8 ha near Edelstetten (10°25′15″ E, 48°17′10″ N; 550 m a.s.l.). The forest was dominated by spruce (69%), followed by Douglas-fir (29%). Additionally, single pedunculate oak (*Quercus robur* L.), European beech (*F. sylvatica*), and Scots pine (*Pinus sylvestris* L.) trees occurred. Oaks were studied in a mixed pedunculate oak—European beech stand (mean age 119, 85–148) of 4.8 ha near Ettenbeuren (10°23′52″ E, 48°22′25″ N; 520–535 m a.s.l.). European beech (65%) dominated, followed by pedunculate oak (30%) and hornbeam (*Carpinus betulus* L.) (4%). Groups of red oak (*Q. rubra*) were admixed, and additional single trees of birch (*Betula pendula*), larch (*Larix decidua*), and spruce (*P. abies*) occurred.

Deadwood branches were harvested with a handsaw in crowns of three mature trees of each Douglas-fir (*P. menziesii*), Norway spruce (*P. abies*), pedunculate oak (*Quercus robur* L.), and red oak (*Q. rubra*) in the first week of March 2001 (Fig. 23.5a). The heights of the harvested branches were between 15 and 28 m in oaks and 20 and 35 m in conifers. Branches were carefully lowered to the ground using ropes (Fig. 23.5b) and stored in plastic bags, separated by tree species, individual tree, and diameter class. As the diameters of branches varied between species, being lower in conifers than oaks, we distinguished between two diameter classes in conifers (≤3 cm and >3 cm, with 1–2 replicates of each per tree and a total of 10 samples per tree species) and four in oaks (<5 cm, 5–6 cm, 6–7 cm and >7 cm, with one replicate of each per tree and a total of 12 samples per tree species). The deadwood amount per bag was standardized to 6280 cm^2 surface area (total: 276,320 cm^2). Only branches of comparable decay stage were selected (wood still hard and with complete bark coverage). Samples were transported to Freising and transferred into a transparent plastic (1 mm thick) tube that was covered by a gauze at the top and attached to a funnel with a sampling jar, containing 1.5% copper-sulfate solution in summer and ethylene glycol in winter to avoid freezing, at the bottom (Fig. 23.5c). The gauze facilitated air circulation to prevent mold growth. Tubes were hung on a scaffold in front of the Technical University of Munich in Freising. The incubation under field conditions occurred from March 2001 to February 2002. Afterward the samples were transferred into plastic barrels (Fig. 23.5d) and further incubated at room temperature (20 °C). Sampling jars were emptied in 2-week intervals until October 2001 and then monthly thereafter until October 2002.

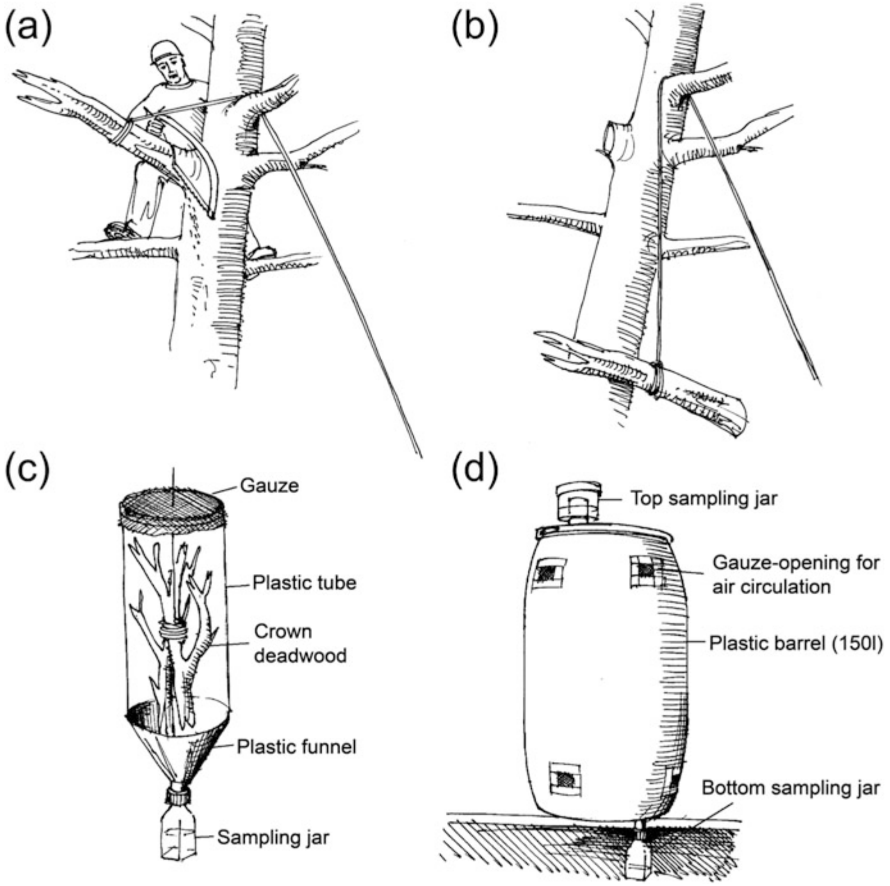

Fig. 23.5 Sampling an incubation of crown deadwood from case study 2. (**a**) Dead branches of early decay stages were harvested in tree crowns of mature trees with a hand saw. (**b**) Branches were carefully lowered and stored in plastic bags, separated by tree species, tree, and diameter class. (**c**) For the first year, deadwood branches were incubated in plastic tubes under field conditions. (**d**) The branches were transferred to plastic barrels and incubated at room temperature after 1 year in spring

23.3.1.2 Data Analyses

All data were analyzed in R version 3.3.1 (R Core Team 2016). To obtain estimates of saproxylic beetle diversity for the four different tree species, we used a framework published recently (Chao et al. 2014). This “diversity accumulation curve” framework extends methods for rarefaction and extrapolation of species richness (species accumulation curve; Colwell et al. 2012). We estimated species diversity curves for Hill numbers based on sample size and sample coverage. Coverage is defined as the proportion of the total number of individuals in an assemblage that belong to species represented in the sample (Chao et al. 2014). The reference sample size for sample size-based estimates was the number of samples taken from a particular tree species. At $q = 0$ (0D), rare and abundant species are weighted equally (species richness); at $q = 1$ (1D), species are weighted in proportion to their frequency in the sampled community; and at $q = 2$ (2D), abundant species receive more weight relative to their frequency. Diversity of beetles emerging from the four different tree species was compared for Chao's BSS (Chao et al. 2014), that is, the higher value of the minimum doubled reference sample size and the maximum reference sample size among tree species, and Chao's BSC (Chao et al. 2014), that is, the higher value of the minimum coverage for doubled reference sample size and the maximum reference coverage among tree species. Significant differences in diversity between tree species were judged by nonoverlapping confidence intervals (Schenker and Gentleman 2001).

To visualize differences in the compositions of saproxylic beetle communities among tree species, we used nonmetric multidimensional scaling (NMDS) plots produced with the metaMDS function in R package vegan (Oksanen et al. 2016) with a maximum of 20 random starts and two dimensions. PERMANOVAs on Bray-Curtis matrices with 9999 permutations (function adonis in vegan) were run to test the effect of tree species on community composition.

To identify indicator species, the approach by Dufrêne and Legendre (1997) was applied. We used the enhancement of the method described by De Caceres and Legendre (2009) and De Caceres et al. (2010) which is provided by the R package "indicspecies." We used the multilevel pattern analysis (function multipatt) that allows tests of association between species patterns and combinations of site groupings. We tested the null hypothesis that the preference of a particular beetle species for one of the tree species or a combination of tree species is due to chance only, using 9999 permutations to calculate p values for each combination. Indicator values (IndVal.g), ranging from 0 (no association) to 1 (complete association) are identical to the values returned by the original function of Dufrêne and Legendre (1997).

23.3.1.3 Results

In total, 557 individuals of 56 saproxylic beetles emerged from the harvested branches. Most specimens and species emerged from native pedunculate oak (197/24), followed by native spruce (184/22), introduced red oak (119/18), and introduced Douglas-fir (57/10). Of these species, 32% were singletons and 30% doubletons. *Ennearthron cornutum* (Gyllenhal, 1827) (Ciidae: 176, all tree species), *Pityophthorus pityographus* (Ratzeburg, 1837) (Curculionidae, Scolytinae: 130, only Norway spruce), and *Dasytes caeruleus* (De Geer, 1774) (Dasytidae: 98, all tree species) reached highest abundance.

The estimated mean sample coverage at Chao's BSS of 20 samples (double minimum reference sample size) was highest in red oak (0.868), followed by Douglas-fir (0.809), Norway spruce (0.755), and pedunculate oak (0.732). Saproxylic beetle diversity was lower on Douglas-fir than on all other tree species for $q = 1$ and $q = 2$ (nonoverlapping confidence intervals in Fig. 23.6). When

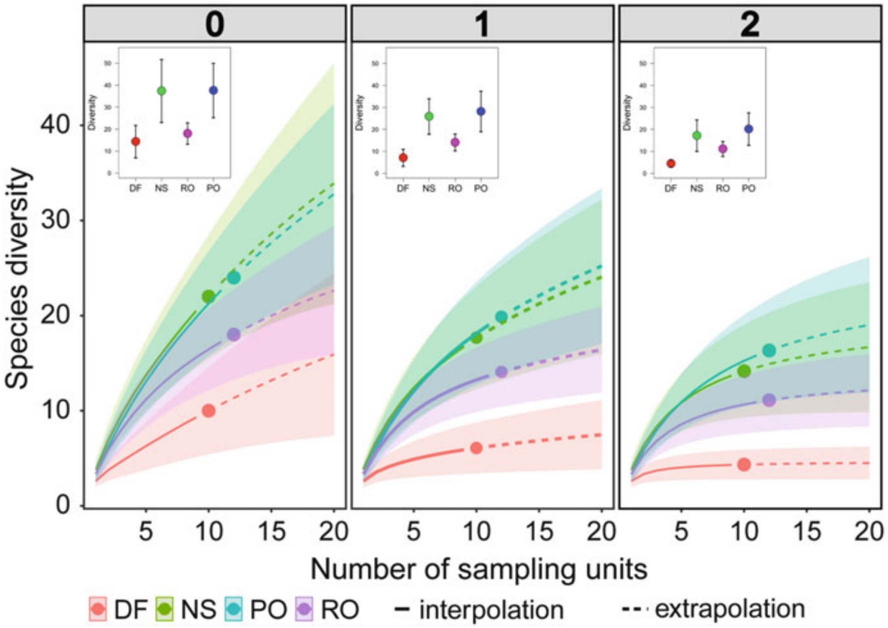

Fig. 23.6 Rarefaction and extrapolation of saproxylic beetle γ-diversity emerging from dead canopy branches harvested from tree crowns of mature Douglas-fir (DF, $n = 10$ branches from three trees), Norway spruce (NS, $n = 10/3$), pedunculate oak (PO, $N = 12/3$), and red oak (RO, $n = 12/3$), case study 2. The deadwood amount sampled per branch was standardized to 6280 cm^2 surface area. Panels show diversity quantified for Hill numbers 0, 1, and 2 (rarefaction = solid line, extrapolation = dashed line) based on frequencies of species occurrences. All curves include 95% confidence intervals obtained by bootstrapping based on 200 replications. The insets show estimated diversity at equal Chao's Base Sample Coverage of 0.788

correcting for differences in sample coverage by using Chao's BSC (0.788), the diversity on both introduced tree species was significantly lower at $q = 0$. Douglas-fir showed lower diversity also for $q = 1$ and $q = 2$, being even lower than that of introduced red oak at $q = 2$ (see insets of Fig. 23.6).

Conifers and oaks showed distinct communities (Fig. 23.7). While the communities of native and introduced oaks differed, the saproxylic community present on Douglas-fir was only a subset of the community observed on Norway spruce. A PERMANOVA showed significant differences between tree species ($F_{3,18} = 2.626$, $p < 0.001$, $R^2 = 0.304$), but this could have been influenced by the considerable difference in multivariate spread among tree species.

Six of the 56 sampled beetle species were significant indicators of particular tree species. Four species were indicators of a single tree species (two for pedunculate oak, two for red oak), one species was indicative of both Douglas-fir and Norway spruce, and one species was a significant indicator taxa for Douglas-fir, Norway spruce, and red oak. Except for *Conopalpus testaceus* (Olivier, 1790) (Melandryidae) on pedunculate oak, all indicator species seem to be host generalists

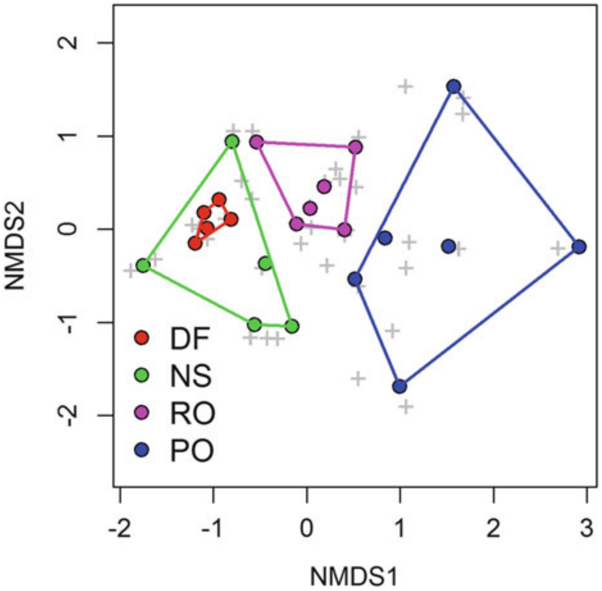

Fig. 23.7 Nonmetric multidimensional scaling (NMDS) plot (stress value = 0.118) showing the composition of saproxylic beetles emerging from deadwood branches harvested in the tree canopy of Douglas-fir (DF), Norway spruce (NS), pedunculate oak (PO), and red oak (RO), case study 2. In each tree species two branches per diameter class and tree were pooled due to low sample size. Gray crosses show the beetle species position in the two-dimensional ordination space

(Table 23.2). Overall, on non-native trees, only host generalists were observed, except two individuals of *Anobium costatum* Aragona, 1830 (Ptinidae), for which *Fagus* is described as preferential host, on red oak, and one individual of the ash specialist *Hylesinus fraxini* (Panzer, 1799) (Curculionidae, Scolytinae) on Douglas-fir (also one individual on spruce).

23.3.1.4 Discussion

This case study clearly shows that crown deadwood of introduced Douglas-fir and red oak is less suitable for native saproxylic beetle species than native spruce and pedunculate oak. The lower diversity on these two introduced tree species is partly supported by flight interception trap studies in the canopy (Gossner and Ammer 2006; Gossner 2004). Significant differences based on these passive samplings were, however, only observed in particular years or in particular stand types (pure stands in red oak/pedunculate oak and beech-dominated stands in Douglas-fir/Norway spruce). Results of the present study come from mixed native/introduced conifer and oak stands. Whether stand- and year-specific effects observed by passive sampling reflect a sampling bias needs to be clarified in future studies. For Douglas-fir, a recent large-scale study in which saproxylic beetles were reared from experimentally exposed deadwood shows that the significantly lower diversity of beetles developing in the introduced tree is independent of region and forest management (Gossner et al. 2016). This supports the idea that Douglas-fir provides a less suitable habitat for native saproxylic species.

Table 23.2 Beetle species that were found to be indicator species for one tree species or a combination of them in case study 2

Tree species	Beetle family	Beetle species	N	IndVal	P.value	Feeding guild	Habitat	Body size	Tree species
PO	Scraptiidae	*Anaspis thoracica* (L.)	4(4)	0.707	0.0409	xz	Deadwood with bark	2.75	*Betula*, *Fagus*, *Carpinus*, *Quercus*, *Ulmus*, *Tilia*, *Corylus*
PO	Melandryidae	*Conopalpus testaceus* (Olivier)	5(5)	0.707	0.0375	xm	Deadwood with bark	6.00	*Fagus*, *Quercus*
RO	Cerambycidae	*Mesosa nebulosi* (Fabricius)	8(8)	0.816	0.0044	x	Deadwood with bark	12.0	*Carpinus*, *Acer*, *Quercus*, *Fagus*, *Tilia*, *Ulmus*, *Alnus*, *Castanea*, *Salix*, *Populus*
RO	Staphylinidae	*Anomognathus cuspidatus* (Erichson)	4(4)	0.707	0.0383	z	Deadwood with bark	1.65	*Quercus*, *Fagus*, *Acer*, *Aesculus*, *Populus*, *Salix*, *Betula*, rarely *Picea*, *Pinus*
DF +NS	Melyridae	*Dasytes plumbeus* (Muller)	16 (16)	0.949	<0.001	z	Deadwood with bark	4.05	Broadleaved trees
DF +NS +RO	Melyridae	*Dasytes caeruleus* (De Geer)	98 (91)	0.905	0.0282	z	Deadwood with bark	5.50	*Fagus*

The number of individuals (N) sampled in total and on the indicator tree species (in brackets) is shown. Additional information on feeding guild (x = xylophagous, z = zoophagous, m = myetophagous), mean body size, and tree host genera based on Koch (1989–1992) and Palm (1959) is given

While communities on red oak were located between pedunculate oak and conifers in an ordination diagram, communities on Douglas-fir seems to be similar among samples and located within the ordination space spanned by Norway spruce samples (Fig. 23.7). Hence, Douglas-fir seems to provide habitat for only a subset of mainly generalist species that are not restricted to native conifers. In contrast, the saproxylic community associated with the introduced red oak seems to be a mixture of species that colonize native oak and native conifers. This suggests that with the exception of a few individuals of more specialized species, polyphagous species just expanded their host range, but specialists showed no major host shifts, which would imply a major change in ecology.

Our results provide evidence that non-native tree species alter saproxylic beetle communities with unknown consequences for wood decomposition. A recent study from Germany, however, indicates that wood decomposition in non-native Douglas-fir is lower compared to native tree species (Kahl et al. 2017). Future studies are needed in evaluating the degree to which current plantations of non-native tree species affect rare and threatened saproxylic species as well as potential pest species and their antagonists. Moreover, comprehensive studies on the consequences of observed saproxylic community alterations for ecosystem processes such as wood decomposition and pest control are of great interest. This will be a great step forward toward an evidence-based process for evaluating the establishment of non-native tree species in Europe in light of nature conservation and forest management.

23.3.2 Case Study 3: Saproxylic Insects Utilizing Eucalyptus in Western Iberian Peninsula

23.3.2.1 Background

Plantations with *Eucalyptus* trees, mostly *Eucalyptus globulus* Labill. introduced from Australia, currently cover large areas in the western Iberian Peninsula, including Portugal and northwestern Spain. *Eucalyptus* plantations in Iberia first appeared in Portugal in the 1850s and began to be widely planted and economically important since the 1940s. From 1995 to 2010, the total land area covered in *Eucalyptus* plantations in Portugal increased by 13% (ICNF 2013). At present there are over 810,000 ha of *Eucalyptus* plantation forests in Portugal, accounting for about 33% of the forest surface (ICNF 2013). Similarly, there are 760,000 ha of *Eucalyptus* in Spain, the vast majority in Galicia, causing controversy and social concern as *Eucalyptus* plantations are perceived to have negative ecological effects particularly on biodiversity (Veiras and Soto 2011) and are also thought to increase wildfire risk (Anonymous 2017). Nevertheless, the high productivity of the *Eucalyptus* forests and the high industrial return, mainly for pulp production, renders this forest use highly compensatory when compared to native pines or hardwoods.

For more than 150 years, *Eucalyptus* forests in the Iberia region were characterized by their extremely high health status. This situation changed with the increased arrival of *Eucalyptus* pests from their region of origin, including two Australian longhorn beetles of the genus *Phoracantha* (*semipunctata* and *recurva*) (Hurley et al. 2016). On the other hand, native insect pests are not usually a menace for *Eucalyptus* plantations in the region, which is in great part explained by the absence of congeneric tree species in the European flora that could harbor potential common pests (Branco et al. 2015a). Nevertheless, in a study conducted by Lombardero and Fernández (1997), in Galicia, the authors found occasional attacks to living trees from native xylomycetophagous insects (Table 23.3). Three species of ambrosia beetles were reported: *Xyleborinus saxesenii* (Ratzeburg), *Xyleborus dispar* Fabr. (Col: Scolytinae), and *Platypus cylindrus* Fabr. (Col: Platypodinae).

The *Eucalyptus* trees colonized by the three ambrosia beetles were always found to be under stress from forest fires, pathogenic fungi, or application of herbicides (Lombardero and Fernández 1997). The most frequent species, *X. saxesenii*, was recorded from 11 different host species (mostly broadleaf) native to the study area, Galicia in Spain (Table 23.3). *Xyleborus dispar* was rarer on *Eucalyptus* and only found in highly decaying trees or logs, so this species was not considered as representing a menace to living trees. Finally, *P. cylindrus* was found in one unique site in decaying trees affected by pathogenic fungi (Table 23.3). In the Iberian Peninsula, *P. cylindrus* has been mostly associated with cork oak, *Quercus suber* L. (Sousa and Inácio 2005).

Coppice forestry is the main form of silviculture used in Iberian *Eucalyptus* plantations. Cuttings allow repeated harvest of poles at about 10-year intervals, usually until a third or fourth rotation. Stems are regenerated from shoots formed at the stumps of the living tree. In a study conducted in Portugal, Cabral (1983) surveyed the saproxylic insects associated with the decay of stumps, in the years after cutting. The author studied five regions with stands presenting similar age structure to list the presence/absence of saproxylic insects (Table 23.4). Additionally, to obtain the succession of saproxylic insects following stump age, five stands over 40 years old, with plots covering all rotations after the first cutting, i.e., second, third, and fourth rotation, were sampled. Trees were selected from cuts completed at

Table 23.3 List of xylomycetophagous associated with *Eucalyptus* trees under biotic or abiotic stress in Galicia (adapted from Lombardero and Fernández 1997) from case study 3

Species	Host stress factor	Number of sites	Native known local host plants[a]
Xyleborinus saxesenii	Fire, pathogenic fungi, herbicide	6	*Alnus glutinosa*, *Castanea sativa*, *Juglans regia*, *Prunus avium*, *Prunus persica*, *Quercus robur*, *Quercus pyrenaica*, *Salix cinerea*, *Sambucus nigra*, *Ulmus glabra*
Xyleborus dispar	Fire, pathogenic fungi	3	*Alnus glutinosa*, *Castanea sativa*, *Fagus sylvatica*, *Populus nigra*, *Prunus persica*, *Quercus robur*
Platypus cylindrus	Pathogenic fungi	1	*Quercus suber*

[a] Data on local native host species retrieved from Lombardero (1995)

Table 23.4 List of saproxylic insects sampled from *Eucalyptus* at five sites in Portugal, Serra do Caramulo, Mata do Escaroupim, Região do Oeste, Odemira, and Tapada da Ajuda in Lisbon from case study 3

Species	Abundance	Number of sites (out of 5)	Native known host species
Reticulitermes lucifugus Rossi (Blattodea: Kalotermitidae)	+++	5	*Castanea sativa*, *Celtis australis*, *Cercis siliquastrum*, *Fraxinus angustifolia*, *Olea europaea*, *Pinus pinaster*, *Pinus pinea*, *Populus* sp., *Quercus suber*, *Quercus* sp.
Kalotermes flavicollis Fabr. (Blattodea: Kalotermitidae)	+	2	*Amygdalus communis*, *Castanea sativa*, *Celtis australis*, *Ceratonia siliqua*, *Cercis siliquastrum*, *Fraxinus angustifolia*, *Olea europaea*, *Pinus pinaster*, *Pinus pinea*, *Populus* sp., *Quercus* spp., *Prunus* sp., *Pittosporum undulatum*, *Rhamnus alaternus*, *Tilia* spp., *Ulmus* spp.
Esperia sulphurella Fabr. (Lep., Oecophoridae)	+++	5	_
Esperia oliviella Fabr. (Lep., Oecophoridae)	++	3	_
Dorcus parallelipedus L. (Col., Lucanidae)	+	2	*Quercus* spp., *Pinus pinaster*, Broadleaves
Ampedus sanguineus L. (Col., Elateridae)	+++	4	_
Misolampus gibbulus Herbst. (Col., Tenebrionidae)	+++	4	_
Nalassus tenebrioides Germ. (Col., Tenebrionidae)	++	3	_
Coelometopus clypeatus Germ. (Col., Tenebrionidae)	+	1	_
Corymbia fontenayi (Muls.) (Col., Cerambycidae)	+++	4	Broadleaves and conifers
Trichius fasciatus L. (Col., Cetonidae)	+	1	_
Valgus hemipterus L. (Col., Tenebrionidae)	+	2	_

different times. Average volume per stump within this study varied from about 0.07 m^3 on second rotation to more than 0.5 m^3 on fourth rotation.

The main saproxylic insect species found on the sampled sites were a termite *Reticulitermes lucifugus* Rossi (Blattodea: Kalotermitidae); two moth species of the genus *Esperia*, family Oecophoridae; a click beetle *Ampedus sanguineus* L. (Col.,

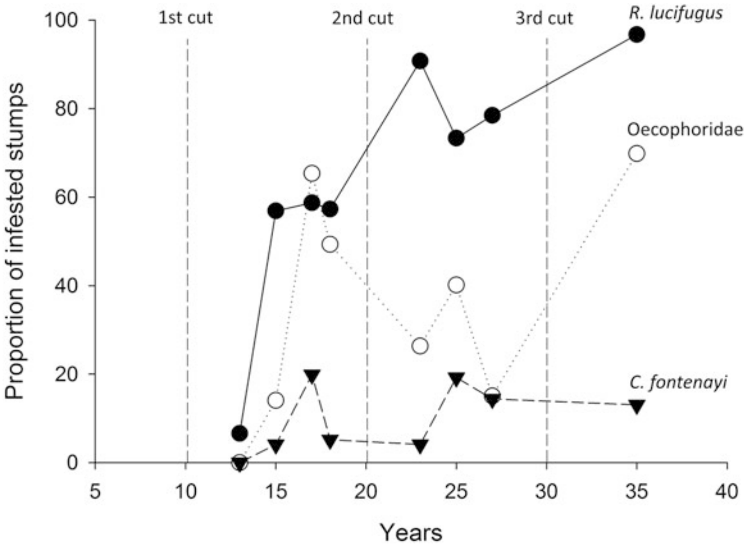

Fig. 23.8 Proportion of infested stumps by three main saproxylic insects, the termite *Reticulitermes lucifugus*, two moth species of the genus *Esperia*, family Oecophoridae, and the longhorn beetle *Corymbia fontenayi* Muls. Oeste, Portugal. Adapted from Cabral (1983), from case study 3

Elateridae); a longhorn beetle *Corymbia fontenayi* Muls. (Col., Cerambycidae); and *Misolampus gibbulus* Herbst. (Col., Tenebrionidae). The termite was the most frequent insect; it appeared after the first cut and reached almost 100% by the fourth rotation (Fig. 23.8). Its occurrence is quite conspicuous and easy to identify due to the presence of the colonies with nymphs and the stratified appearance of the consumed wood. The two species of *Esperia* share the same habitats and frequently coexist on the same stand. The larvae were found to first feed under the bark of the stumps and later penetrate the xylem. Both species have a 1-year life cycle, but with separate emergence times: February–March for *E. sulphurella* Fab. and April until July for *E. oliviella* Fabr. (Cabral 1983). Early instars of *A. sanguineus* apparently feed on wood, producing small round galleries in the wood, but later become predatory. The longhorn beetle *C. fontenayi* was quite common; it was present in all sampled sites (Table 23.4) and found on *Eucalyptus* stumps from the second to fourth rotation (Fig. 23.8). Its frequency was nevertheless low, usually between 10 and 20%. The beetle larvae consume wood of different sizes. The tenebrionids occurred only in highly decomposed wood.

23.3.2.2 Discussion

Although *Eucalyptus* is known to support diverse assemblages of saproxylic insects within the native range of the genus (Grove and Forster 2011a, b; Lawson and

DeBuse 2016), very little is known about the value of dying and dead *Eucalyptus* wood to saproxylic insects outside of Australia. A number of scolytine, platypodine, and cerambycid beetle species are viewed as potential pests of *Eucalyptus* in South America (Monteiro and Garlet 2016; Dorval et al. 2007), however, and at least one effort has been made to understand the value of *Eucalyptus* woody debris to other saproxylic insects, including termites (Trevisan et al. 2008). The studies summarized in this case study indicate that *Eucalyptus* can be utilized by some insect species, but many of them are known to have broad host ranges or are associated with wood at advanced stages of decomposition. There is a strong need for more research aimed at describing the diversity and succession of insects associated with *Eucalyptus* wood given the widespread and growing non-native range of the genus throughout many parts of the world. A recent study from Chile suggests that *Eucalyptus* plantations may create conditions unfavorable for saproxylic insects beyond the influence of wood characteristics. Fierro et al. (2017) found that non-native pine wood present in *Eucalyptus* plantations supported a lower density and richness of saproxylic beetles than similar woody debris in pine plantations nearby. The researchers suggested the toxic properties of *Eucalyptus* leaf litter and the relatively dry conditions of *Eucalyptus* stands may have negatively impacted saproxylic insect diversity.

23.3.3 Case Study 4: Native Saproxylic Species Colonizing Non-native Tree Species in New Zealand

New Zealand is an unusual country in that there is an almost complete separation of production forestry from the public conservation estate of native forests. Small areas of private native forest are managed for sustainable timber production (MPI 2013); however, the vast majority of wood products in New Zealand are produced from 1.7 million ha of even-aged, single species, plantation forests, predominantly *Pinus radiata* D. Don (90%) (FOA and MPI 2016). *Eucalyptus* spp. is also widely planted in New Zealand, covering approximately 23,300 ha (FOA and MPI 2016). These managed forest stands also provide habitat for diverse communities of plants (Brockerhoff et al. 2003), birds (Seaton et al. 2010), and invertebrates (including many saproxylic taxa) (Pawson et al. 2008, 2011), including threatened species (Brockerhoff et al. 2005; Pawson et al. 2010).

To protect plantations, New Zealand has strict quarantine regulations; however, some species continue to cross the border and establish in the native and productive ecosystems. New Zealand has operated a forest health surveillance scheme since the 1950s (Bulman 2008). Although the focus of the program has changed over the years, it has and continues to maintain a strong emphasis on identifying new incursions of non-native species. Observations of forest insects and pathogens from trees throughout New Zealand have been collated into the New Zealand Forest Health Database (FHDB). The FHDB comprises 213,563 records of both native and non-native tree pests and diseases and their hosts. Unfortunately survey methodologies have changed significantly over the decades, and it is not possible to quantify survey effort from the data available. As such a quantitative analysis is not possible. However, the FHDB provides a unique opportunity to assess the use of non-native tree species by both native and non-native saproxylic species. Here we compare the numbers of non-native and native saproxylic species that have colonized *P. radiata* or a *Eucalyptus* sp., and we also summarize the number of hosts colonized by each species.

23.3.3.1 Methods

To evaluate the use of non-native *P. radiata* and *Eucalyptus* hosts by saproxylic species, we summarize the records of insects from the termite families Kalotermitidae, Rhinotermitidae, and Termopsidae, the hymenopteran family Siricidae, the hemipteran family Aradidae, and the beetle families Anthribidae, Belidae, Bostrichidae, Bothrideridae, Brentidae, Buprestidae, Carabidae (Rhysodinae), Cerambycidae, Ciidae, Colydiidae, Corylophidae, Cryptophagidae, Cucujidae, Curculionidae, Latridiidae, Lucanidae, Lymexylidae, Mycetophagidae, Prostomidae, Ptinidae, Salpingidae, Silvanidae, Tenebrionidae, Ulodidae, and Zopheridae. Data was filtered to ensure that only observations where the species was associated with the host tree were included. This meant excluding those observations tagged with "agent not associated with disorder and/or host."

23.3.3.2 Results

A total of 55 saproxylic species are recorded in the FHDB as being associated with *P. radiata* (Table 23.5). Of these, 14 species were non-native and 41 were native. The average number of host trees (native and non-native) colonized by an individual non-native saproxylic beetle species that colonized *P. radiata* was 6.3 (±3.2 95% CI) vs. 9.8 (±6.8 95% CI) for the average individual native beetle species. The most common families of saproxylic beetles that colonized *P. radiata* were Curculionidae (19 species) and Cerambycidae (18 species). The highest diversity of non-native beetles was from the family Curculionidae, whereas Cerambycidae was the most diverse native family.

A total of 14 species of saproxylic beetles were recorded in the FHDB from *Eucalyptus* spp. (Table 23.6). Although many more species of beetles utilize *Eucalyptus* in New Zealand, most are foliar feeders. Of those saproxylic species recorded, four were non-native with the remaining ten native. The average number of hosts (all species, including *Eucalyptus* spp.) for non-native species was 5.4 (4.1 ± 95% CI), and for native species was 31.4 (27.0 ± 95% CI). The dominant group of saproxylic species recorded on *Eucalyptus* were wood borers from the family Cerambycidae.

Table 23.5 Observations of saproxylic beetle species recorded on *P. radiata* as part of forest health surveillance monitoring from case study 4

Beetle species	Family	Biostatus	Number of records from *P. radiata*	Total number of records all hosts	Year of first record	Number of hosts
Araecerus palmaris (Pascoe)	Anthribidae	Non-native	1	6	2001	4
Euciodes suturalis Pascoe	Anthribidae	Non-native	2	2	2016	1
Arhopalus ferus (Mulsant)	Cerambycidae	Non-native	110	162	1983	20
Bethelium signiferum (Newman)	Cerambycidae	Non-native	1	28	1979	10
Hylastes ater (Paykull)	Curculionidae	Non-native	49	54	1979	4
Hylurgus ligniperda (Fabricius)	Curculionidae	Non-native	30	35	2003	4
Rhinocyllus conicus (Frolich)	Curculionidae	Non-native	1	7	2001	5
Stenoscelis hylastoides Wollaston	Curculionidae	Non-native	1	6	2016	5
Kalotermes banksiae Hill	Kalotermitidae	Non-native	1	1	2001	1
Sirex noctilio Fabricius	Siricidae	Non-native	36	39	1983	3
Amarygmus tristis Blackburn	Tenebrionidae	Non-native	6	7	1979	2
Phymatus hetaera (Sharp)	Anthribidae	Native	9	9	1979	1
Phymatus phymatodes (Redtenbacher)	Anthribidae	Native	6	6	2003	1
Lasiorhynchus barbicornis (Fabricius)	Brentidae	Native	1	2	2001	2
Agapanthida pulchella White	Cerambycidae	Native	1	2	2001	2

(continued)

Table 23.5 (continued)

Beetle species	Family	Biostatus	Number of records from *P. radiata*	Total number of records all hosts	Year of first record	Number of hosts
Blosyropus spinosus Redtenbacher	Cerambycidae	Native	1	2	Unknown	2
Coptomma variegatum (Fabricius)	Cerambycidae	Native	7	12	1979	6
Eburilla sericea (White)	Cerambycidae	Native	1	14	1990	7
Hexatricha pulverulenta (Westwood)	Cerambycidae	Native	60	88	2001	15
Hybolasius vegetus Broun	Cerambycidae	Native	2	6	2003	5
Leptachrous strigipennis (Westwood)	Cerambycidae	Native	5	10	2001	4
Ochrocydus huttoni Pascoe	Cerambycidae	Native	1	11	1984	6
Oemona hirta (Fabricius)	Cerambycidae	Native	5	215	1983	132
Prionoplus reticularis White	Cerambycidae	Native	77	134	2016	20
Ptinosoma ptinoides (Bates)	Cerambycidae	Native	1	1	2008	1
Somatidia antarctica (White)	Cerambycidae	Native	6	6	2016	1
Somatidia grandis Broun	Cerambycidae	Native	1	1	2012	1
Xylotoles griseus (Fabricius)	Cerambycidae	Native	4	40	1979	19
Xylotoles laetus White	Cerambycidae	Native	9	25	1983	14
Zorion minutum (Fabricius)	Cerambycidae	Native	3	38	1979	22
Euophryum confine (Broun)	Curculionidae	Native	2	3	2006	2
Hoplocneme hookeri White	Curculionidae	Native	2	2	2001	1

(continued)

Table 23.5 (continued)

Beetle species	Family	Biostatus	Number of records from *P. radiata*	Total number of records all hosts	Year of first record	Number of hosts
Mitrastethus baridioides Redtenbacher	Curculionidae	Native	44	73	2000	11
Pachycotes peregrinus (Chapuis)	Curculionidae	Native	13	25	2013	5
Pentarthrum zealandicum Wollaston	Curculionidae	Native	2	6	2003	4
Phrynixus terreus Pascoe	Curculionidae	Native	3	4	1963	2
Platypus apicalis White	Curculionidae	Native	13	24	1983	4
Psepholax macleayi (Schonherr)	Curculionidae	Native	5	8	2006	4
Psepholax sulcatus White	Curculionidae	Native	6	14	2005	6
Rhopalomerus tenuirostris Blanchard	Curculionidae	Native	1	2	1972	2
Scolopterus aequus Broun	Curculionidae	Native	1	3	2006	3
Torostoma apicale Broun	Curculionidae	Native	24	42	1963	7
Xenocnema spinipes Wollaston	Curculionidae	Native	19	25	2001	3
Kalotermes brouni Froggatt	Kalotermitidae	Native	11	109	1979	61
Salpingus bilunatus Pascoe	Salpingidae	Native	2	2	2006	1
Brontopriscus pleuralis (Sharp)	Silvanidae	Native	4	5	2010	2
Artystona rugiceps Bates	Tenebrionidae	Native	4	5	2002	2
Tanychilus metallicus White	Tenebrionidae	Native	1	1	2002	1

(continued)

Table 23.5 (continued)

Beetle species	Family	Biostatus	Number of records from *P. radiata*	Total number of records all hosts	Year of first record	Number of hosts
Uloma tenebrionoides (White)	Tenebrionidae	Native	15	17	1997	3
Stolotermes inopinus (Gay)	Termopsidae	Native	4	5	2009	2
Stolotermes ruficeps Brauer	Termopsidae	Native	70	103	1993	19
Syrphetodes marginatus Pascoe	Ulodidae	Native	1	1	2013	1
Pristoderus antarcticus (White)	Zopheridae	Native	4	4	2014	1

23.3.3.3 Discussion

Based on the available records, more native saproxylic species are recorded as utilizing *P. radiata* than *Eucalyptus* spp. in New Zealand. One possible explanation for this is the fact that *P. radiata* covers a much larger land area than *Eucalyptus*. Or alternatively, it could reflect greater relatedness between *P. radiata* and conifers native to New Zealand, i.e., Podocarpaceae. Irrespective of host type (*P. radiata* and *Eucalyptus* spp.), individual non-native beetle species were recorded from fewer species of host trees than native beetle species (Tables 23.5 and 23.6). This suggests that colonizing non-native species could be more host specific than the native insects which utilize a wider range of both non-native and native tree species. This suggests that native species are less host specific than the exotic species that have colonized although this is tentative given that the FHDB does not provide exhaustive information on associations. Interestingly this effect was much stronger for *Eucalyptus* spp. where seven of the ten native species were known from more than ten hosts. It was a feature of *Eucalyptus* records that highly polyphagous native species were represented by few observations that indicate that *Eucalyptus* is likely to be a marginal host for such species. Few species of termites have been recorded in association with *P. radiata* or *Eucalyptus* spp. in New Zealand. This reflects the low diversity of native termites in New Zealand (Bain and Jenkin 1983) and the strict import controls on commodities that have the potential to be infested by termites.

Table 23.6 Observations of saproxylic beetle species recorded on *Eucalyptus* spp. as part of forest health surveillance monitoring from case study 4

Beetle species	Family	Biostatus	Number of records from *Eucalyptus* spp.	Total number of records all hosts	Date of first record	Number of hosts
Callidiopis scutellaris (Fabricius)	Cerambycidae	Non-native	28	34	1985	13
Phoracantha semipunctata (Fabricius)	Cerambycidae	Non-native	1	1	2004	1
Tessaromma undatum Newman	Cerambycidae	Non-native	18	20	2000	6
Porotermes adamsoni (Froggatt)	Termopsidae	Non-native	1	3	1993	2
Ctenoneurus hochstetteri (Mayr)	Aradidae	Native	1	1	1985	1
Coptomma lineatum (Fabricius)	Cerambycidae	Native	3	43	1983	15
Eburida picta (Bates)	Cerambycidae	Native	1	1	2014	1
Hexatricha pulverulenta (Westwood)	Cerambycidae	Native	1	88	1984	15
Oemona hirta (Fabricius)	Cerambycidae	Native	6	215	1983	132
Prionoplus reticularis White	Cerambycidae	Native	1	134	1979	20
Xylotoles laetus White	Cerambycidae	Native	1	25	2000	14
Psepholax sulcatus White	Curculionidae	Native	1	14	1983	6
Kalotermes brouni Froggatt	Kalotermitidae	Native	4	109	1979	61
Stolotermes ruficeps Brauer	Termopsidae	Native	3	103	1993	19

23.4 Conclusions and Future Directions

Based on the information currently available, the diversity of saproxylic insects associated with decomposing wood varies considerably among tree species, and this is true for both native and non-native taxa. Many non-native wood species appear to provide highly suitable material for saproxylic insect communities and may offer a way to increase the availability of deadwood in some forests. In Italy, for example, Della Roca et al. (2016) found no difference in saproxylic beetle species richness and composition among the non-native invasive *R. pseudoacacia* and two native wood species. Based on these results, the researchers suggested felling *R. pseudoacacia* trees for the dual purpose of controlling this invasive species and increasing the amount of deadwood available to saproxylic organisms. However, while non-native woody material may provide suitable habitat for many generalist species, it remains almost entirely unknown whether these novel resources will be of any value to specialist species, including threatened taxa of greatest conservation concern. In one of the only published studies to explore this question, Oleksa and Klejdysz (2017) found no evidence that *Cerambyx cerdo*, a threatened specialist of old oaks in Europe, can utilize *Q. rubra*, a non-native species from North America. Similarly, Della Rocca et al. (2017) found that *R. pseudoacacia* did not impact the occurrence of *Lucanus cervus* (L.) in Europe as long as it covered less than 70% of the landscape and stressed the importance of preserving native trees in invaded landscapes. Although the four case studies presented herein report many examples of native saproxylic insects utilizing non-native wood species, it is important to note that all of these insect species are known generalists with healthy populations within the regions studied. There is currently little evidence that non-native wood species will provide much benefit to the most threatened members of the saproxylic insect fauna.

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