

Chapter 14

Effects of Exotic Tree Plantations on Plant Litter Decomposition in Streams



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Abstract The need for tree-derived industrial products is causing an increase in the land surface covered by fast-growing monoculture plantations throughout the world. Species planted are selected mostly prioritizing their rapid growth, with less consideration to minimizing the negative environmental effects they create. Among the various ecosystems that can be negatively affected by plantations, streams are among the most impacted, as they strongly depend on dead organic matter from the surrounding vegetation. Changes in land use in favour of monocultures can have large consequences on stream biodiversity and functioning since they can lead to alterations in the diversity, quantity, quality and timing of litter inputs. Here, we review the literature dealing with the effects of plantations on litter decomposition in streams, with special focus on eucalyptus and conifer plantations, which have been more thoroughly studied than other planted tree species. The effects of plantations on litter decomposition in streams have a degree of regional specificity, and depend on the composition of detritivores, litter characteristics (of native and exotic species) and local environmental conditions. There is, nonetheless, a need for further research

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describing the effects of specific planted exotic species and for more studies out of Europe and North America.

14.1 Introduction

Trees form dense forests in many parts of the world, but their diversity varies widely depending on their degree of human intervention, from primary growth forests (where humans have had a negligible effect) to the most intensively managed plantations (where tree growth is maximized in detriment of diversity; see Box 14.1). The historical human use of wood and the woodland conversion to agriculture have led to a gradual but profound modification of landscapes and the reduction in global forest cover. Thus, today's natural forests are comprised mostly of modified natural forests (74%) as opposed to primary (i.e., pristine) forests (26%) (FAO, 2015). Also, with the growing global population and demand for forest resources, planted forests are becoming an increasingly important part of the human-modified landscape, alongside urban, agricultural, and natural forest land cover types (FAO, 2018). Globally, natural forests cover 3695 million ha (Fig. 14.1) and are decreasing at an annual rate of 0.24% (FAO, 2015). On the other hand, planted forests cover around 291 million ha, accounting for 7% of the global forest area (Fig. 14.1). The largest area of planted forests is found in the temperate domain, accounting for 150 million ha, followed by the tropical and boreal domains with almost 60 million ha each. Asia has 44.4% of the world's planted forests, followed by Europe (28.6%), Central and North America (15%), Africa (5.5%), South America (2.4%) and Oceania (1.5%) (FAO, 2015; Fig. 14.1). Moreover, planted land surface area is increasing globally at an average annual rate of 1.84%, ranging from 1.1% in Europe to 2.5% in North and Central America (FAO, 2015; Fig. 14.1). Land cover by planted exotic tree species constitutes 25% of planted forests worldwide (FAO, 2015), with dissimilar representation around the globe. Plantations of exotic species dominate in the southern hemisphere, where they represent 88% of the total planted surface in South America, 75% in Oceania, 31% in Africa, and 42% in the Caribbean (Payn et al., 2015). In contrast, continents with more planted forest cover have a lower proportion of exotic plantations, from 25% in eastern Asia to 1% in North America (Payn et al., 2015).

Box 14.1 Glossary box for forest definitions modified from FAO (2006b)

Forest: Land spanning more than 0.5 ha with trees higher than 5 m and a canopy cover of more than 10%, or trees able to reach these thresholds in situ. It does not include land that is predominantly under agricultural or urban land use. Includes native and introduced tree species growing naturally or human-assisted.

Natural forest: A forest composed of indigenous/native trees.

Primary forest: Forest of native species, where there are no clear visible indications of human activities and the ecological processes are not significantly disturbed.

Modified natural forest: Forest of naturally regenerated native species where there are clearly visible indications of human activities. Today most natural forest cover belongs globally to this category.

Planted forest: Forest in which trees, both native and introduced species, have been established through planting or seeding. Includes all stands established through planting or seeding. Includes the subcategories semi-natural forests and plantations.

Semi-natural forest: Forest of native species, established through planting, seeding or assisted natural regeneration, such as thinning or fertilization.

Plantation: Forest of usually exotic, but also native, species established through planting or seeding mainly for production of wood or non-wood goods.

The majority of planted forest area (three quarters) is grown for productive purposes (i.e., production of wood, fibre, fuel or non-wood forest products), while just one quarter is intended for protective purposes (e.g., rehabilitation of degraded lands, combating desertification or protection of soil and water) (FAO, 2006a). Meanwhile, afforestation, the process where new forests are planted across tree-less land (i.e., grasslands and abandoned agricultural lands), has become one of the most technologically simple methods of removing carbon dioxide from the atmosphere and storing it as biomass, while controlling water erosion and dust storms, reducing

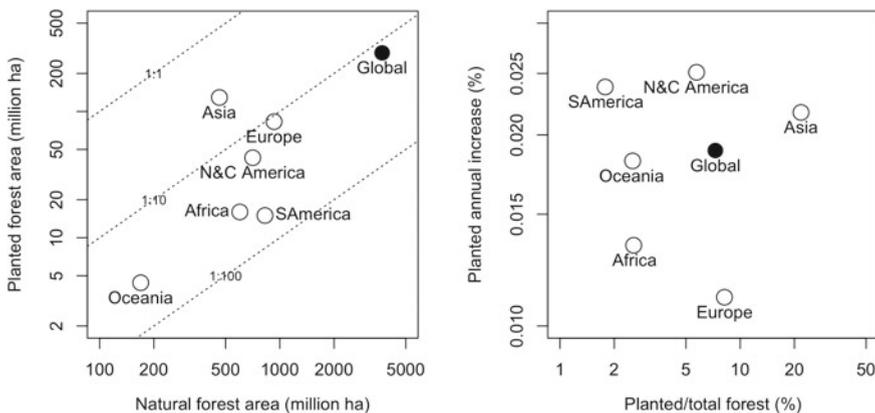


Fig. 14.1 Left: Natural forest area against the planted forest area per continent. Planted:Natural ratio isolines are included as a reference. Right: Annual increase rate (%) of planted forest against the percentage of planted forest in relation to the total forest area. In both cases the global position (black circle) is added. All the axes are in log-scale (Source [FAO, 2015])

river sedimentation, and mitigating small floods. Still, plantations (including both afforestation and reforestation practices) raise concerns about their potential negative effects on ecosystems. For example, plantations can alter nutrient cycling and catchment hydrology, with negative consequences on soil and water quality and quantity (Farley et al., 2008; Jackson et al., 2005; Mátyás & Sun, 2014), or even become a source for invasion when individuals are dispersed from plantations to other areas (Richardson & Rejmánek, 2011).

Exotic tree species have a long history of being used in forestry, mostly because selected species have improved productivity compared with that of native species (e.g., Elfving et al., 2001; Morris et al., 2011). The higher growth rates of selected exotic species, compared with those of native species, can indeed result in shorter rotations (Salmón Rivera et al., 2016). The species used in plantations differ among regions, with conifers dominating colder regions and broadleaves dominating warmer regions; overall, conifers account for 52% of total plantations, with broadleaves representing 37% and the remaining 11% being unspecified (FAO, 2006a). In order of importance, the main coniferous genera by cover area are *Pinus*, *Cunninghamia*, *Larix*, *Picea* and *Cryptomeria* while the main broadleaf genera are *Eucalyptus*, *Populus*, *Acacia* and *Tectona* (Fig. 14.2). Exotic planted woody species may become invasive if they expand naturally beyond plantations (Hayson & Murphy, 2003). Effectively, exotic woody species invading native forests were found to cover 79 million ha worldwide in 2010, with an estimated annual increase rate of 11.3% considering the period 1990–2010 (FAO, 2015). In a study on forestry tree invasiveness, Hayson and Murphy (2003) found that 282 out of 458 species exclusively used in forestry had become naturalised and invasive, with most species belonging to the genera *Acacia*, *Eucalyptus*, *Larix*, *Picea*, *Pinus*, *Populus* and *Tectona*. More recently, Richardson and Rejmánek (2011) pointed to the genus *Acacia* (32 spp.) and *Pinus* (22 spp.) among the most widespread invasive exotic tree species in 15 regions around the globe. They highlighted that four tree species ranked at the top of the invasive list: *Acacia mearnsii* (in 12 regions) and *Pinus pinaster*, *Pinus radiata* and *Pinus elliottii* (in five or more regions each); all are used in forestry for multiple purposes. Moreover, they showed that forestry ranked second (after horticulture) as a main cause of invasive species introduction and dissemination.

Plantations mainly modify terrestrial ecosystems (by replacing native species or by modifying the attributes and reducing the complexity of natural forests), but fresh waters can also be critically altered, with forest streams potentially being the most affected given that they are highly dependent on litter inputs from the surrounding terrestrial ecosystems, which in turn vary with forest composition, structure and production (Fausch et al., 2010; Whiles & Wallace, 1997). In this chapter we review the current knowledge on the effects of tree plantations on leaf litter decomposition capacity of streams. We focus on two case studies, which are the most prolific in scientific research, namely on eucalyptus and conifer plantations. We summarize the main findings of existing research on these species and then briefly review the effects of less-known species. We discuss the implications of changes in litter quality and timing, the consequences of plantation harvesting methods, and the procedures that have been proven to mitigate the effects of plantations on litter decomposition in streams.

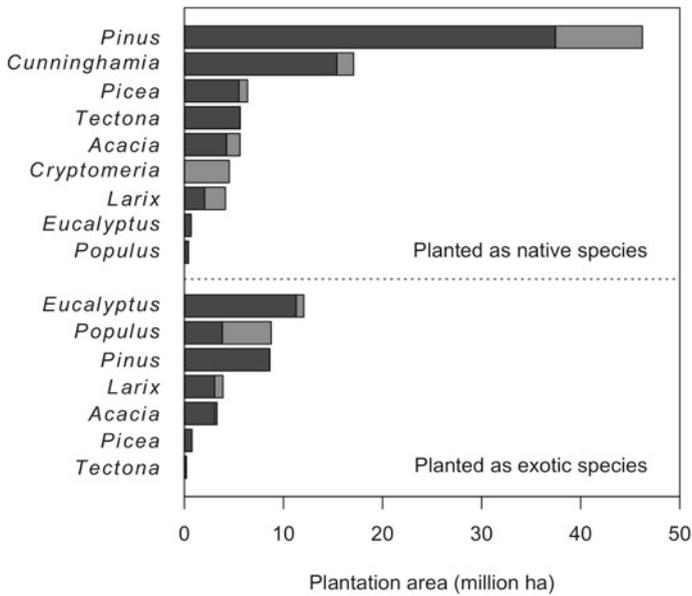


Fig. 14.2 Plantation area of the most common genera in selected countries of the world with the largest areas of planted forest (Algeria, Argentina, Australia, Belgium, Brazil, Bulgaria, Chile, China, Croatia, Czech Republic, Finland, France, India, Indonesia, Iran, Italy, Japan, Latvia, Lithuania, Malaysia, Myanmar, Netherlands, New Zealand, Norway, Philippines, Poland, Slovakia, Slovenia, South Africa, Sudan, Sweden, Turkey, Ukraine, United Kingdom, United States, Vietnam). Plantation areas for production (dark grey) and protection (light grey) goals are shown in the bars (Source [FAO, 2006a])

14.2 Case Studies

14.2.1 *Eucalyptus* Plantations (Fig. 14.3)

Eucalyptus is the most planted tree genus out of its natural range worldwide. In 2009, eucalyptus plantations covered already >20 million ha worldwide, mostly between 35°S and 35°N (Iglesias-Trabado et al., 2009). Three countries alone (Brazil, India and China) contribute with >50% of total eucalyptus plantation area, but they are present in >95 countries (Iglesias-Trabado et al., 2009), often also in stream banks. Still, most studies addressing the effects of eucalyptus plantations on litter decomposition in streams have been performed in central Portugal and northern Spain. In these two locations, the climate is warm temperate (drier in central Portugal and more humid in northern Spain), native forests are characterized by mixed deciduous broadleaf species dominated by *Quercus* spp. and *Castanea sativa*, and the species used in plantations is *Eucalyptus globulus* (Ferreira et al., 2016). Therefore, the current knowledge of this topic is geographically biased and so must be our revision. Information from other regions will be included whenever possible, but in its

absence, extrapolations from temperate regions to other climatic realms should be made carefully as plantation effects may be context dependent (Ferreira et al., 2019).

Eucalyptus plantations can affect stream environmental conditions through multiple pathways (Fig. 14.3). Eucalyptus trees have fast growth rates, with rapid increases in transpiration rates (Forrester et al., 2010). Plantations are generally dense and young (rotation: ~10 years in temperate regions, ~7 years in tropical regions; Ferraz et al., 2013; Gabrielle et al., 2013), they have high water demands (Jackson et al., 2005), and rainfall interception is high, especially compared with deciduous tree species (Ferraz et al., 2013). These features altogether lead to decreases in runoff and in aquifer level (Ferraz et al., 2013; Jackson et al., 2005; Lara et al., 2009). Additionally, soils in eucalyptus plantations are highly hydrophobic (Abelho & Graça, 1996; Walden et al., 2015), likely as a result of the accumulation of oils released from the leaves during decomposition, which hampers the penetration of rain water into the soil and the replenishment of groundwater reservoirs. Higher water consumption by trees and loss of soil permeability lead to changes in stream hydrology, with reduced water flow year round, increased propensity for spates during storms and reduced water availability during the dry season, especially in arid regions (Lara et al., 2009; Scott & Lesch, 1997) (Fig. 14.3). Reduced water availability to feed streams in summer may lead to temporary habitat loss, resulting in streams characterised by isolated pools or completely dry (Canhoto & Laranjeira, 2007). Under these conditions, there is reduced solute dilution capacity, which results in toxic conditions for

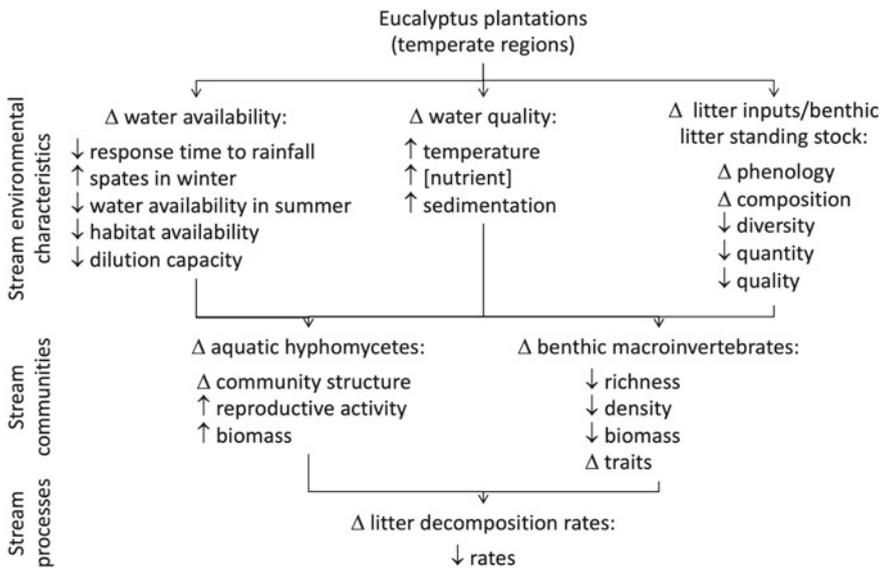


Fig. 14.3 Conceptual diagram of the effects of eucalyptus plantations on stream environmental characteristics, communities and processes in temperate regions, which are characterized by native forests composed of mixed broadleaf deciduous tree species and seasonality in litter fall. Δ indicates change, ↑ indicates increase and ↓ indicates decrease

stream biota due to the accumulation of polyphenolic compounds leached from eucalyptus leaves, reduced pH and reduced dissolved oxygen concentration (Canhoto & Laranjeira, 2007; Canhoto et al., 2013). Water quality may be reduced in streams flowing through eucalyptus plantations also as a result of increases in water temperature if single-stemmed eucalyptus trees with a small crown on the top (the typical tree habit in plantations) replace wide-canopy native tree species (e.g., *Alnus glutinosa*, *Salix* spp.) (Cordero-Rivera et al., 2017). There may also be an increase in dissolved nutrient concentrations due to the use of fertilizers in plantations (Hopmans & Bren, 2007) and soil erosion, especially during and after plantation and harvesting, resulting in increased fine sediment inputs to streams (Siegloch et al., 2014).

Eucalyptus plantations can also alter considerably litter inputs to streams (Fig. 14.3). These changes can be in terms of timing of litter inputs to streams, with inputs peaking during autumn in streams flowing through deciduous forests, while litter inputs are more homogeneously distributed year round or peaking in summer due to water stress under eucalyptus plantations (Abelho & Graça, 1996; Canhoto et al., 2002; Molinero & Pozo, 2003, 2004, 2006; Pozo et al., 1997). The annual quantity of litter inputs to eucalyptus streams may be similar to (Abelho & Graça, 1996) or lower than (Molinero & Pozo, 2003, 2004, 2006; Pozo et al., 1997) litter inputs to streams under native forests. However, the relative composition of litter inputs to streams is strongly altered under eucalyptus plantations. Thus, litter accumulated in eucalyptus streams has a higher fraction of twigs and bark (Molinero & Pozo, 2004; Pozo et al., 1997). Plant species richness and identity in riparian ecotones under plantations is shifted, resulting in litter inputs dominated by the recalcitrant eucalyptus litter (Graça et al., 2002; Molinero & Pozo, 2003, 2006; Pozo et al., 1997). As a result, the quality of litter inputs to eucalyptus streams is decreased (Molinero & Pozo, 2004, 2006; Pozo et al., 1997). Since litter inputs occur mostly during low flow conditions and litter is more recalcitrant (i.e., woody material and eucalyptus leaves), benthic litter standing stock is larger than in streams flowing through native forests (Molinero & Pozo, 2003, 2004; Pozo et al., 1997).

The changes in stream environmental characteristics described above can affect stream communities (Fig. 14.3). The community structure of aquatic hyphomycetes differs between stream types (Bärlocher & Graça, 2002; Ferreira et al., 2006). Species richness is lower in streams flowing through eucalyptus plantations than in those under native forests in central Portugal (Bärlocher & Graça, 2002; Ferreira et al., 2006), while it shows the opposite pattern (Ferreira et al., 2006) or it does not differ between stream types in northern Spain (Chauvet et al., 1997). In general, conidia concentration in the water (Bärlocher & Graça, 2002) and sporulation rates by aquatic hyphomycetes on *A. glutinosa* and *Quercus robur* litter incubated in both stream types does not differ (Ferreira et al., 2006), but maximum sporulation rates can be higher in eucalyptus streams. Fungal biomass is also higher under eucalyptus plantations in Portugal, likely caused by higher water temperature (Ferreira et al., 2006), but not in Spain (Diez et al., 2002; Molinero et al., 1996; Pozo et al., 1998). The stronger effects in Portugal compared to Spain are likely due to the more arid climate in the former region that leads to stronger decreases in water availability and changes in litter inputs in eucalyptus streams (Ferreira et al., 2006).

Regarding benthic macroinvertebrates, there is generally lower taxa richness, density and biomass of total macroinvertebrates and shredders in eucalyptus streams than in streams flowing through deciduous forests (Abelho & Graça, 1996; Cordero-Rivera et al., 2017; Ferreira et al., 2015; Larrañaga et al., 2006, 2009a, 2009b; Monroy et al., 2017) (Fig. 14.3). Negative effects are especially strong for large shredder taxa (e.g., amphipods of the genus *Echinogammarus* and caddisflies of the family Limnephilidae), which are more negatively affected by the poor quality of eucalyptus litter (Larrañaga et al., 2009a; Monroy et al., 2017). Assemblages of detritivore macroinvertebrates in streams flowing through eucalyptus plantations are dominated by taxa that feed on fine particulate organic matter and algae (since litter is recalcitrant in these streams), and are mostly small bodied, with short life cycles, reduced voltinism and active dispersal (likely an adaptation to summer streamflow reduction and drought risk) (Larrañaga et al., 2009a, 2009b; Monroy et al., 2017).

The above-mentioned changes in abiotic and biotic factors may alter litter decomposition in streams (Fig. 14.3). However, it seems that the magnitude and the direction of the effect depend on specific local factors (Chauvet et al., 2016; Ferreira et al., 2016; Graça et al., 2002). A recent meta-analysis based on 92 comparisons between streams flowing through eucalyptus plantations and streams flowing through native deciduous forests reported in 10 published studies found an overall significant inhibition of 22% in litter decomposition rate in streams flowing through plantations when compared with streams flowing through deciduous forests (Ferreira et al., 2016). The magnitude of the effect varies, however, with plantation extent, resource type, litter identity and type of community involved (microbes only or microbes plus macroinvertebrates). The effects of eucalyptus plantations on litter decomposition are significant when eucalyptus plantation extends into the riparian area (inhibition of 27%) but not when plantations keep a native riparian buffer, highlighting the relevance of maintaining riparian ecotones intact to mitigating plantation effects (Ferreira et al., 2016). When considering resource type, significant effects are observed only for leaf litter decomposition (inhibition of 26%) but not for substrates that included woody components, likely due to the larger role played by invertebrates in the decomposition of leaves compared to that of woody substrates (Ferreira et al., 2016). When considering just leaf litter decomposition, significant effects were found when macroinvertebrates had access to the leaves (inhibition by 36%) but not in microbial-driven leaf decomposition. This result suggests that there is functional redundancy among microbial communities and that plantation effects on litter decomposition are mediated through changes in macroinvertebrate communities in temperate regions (Ferreira et al., 2016). Finally, litter quality resulting from contrasting leaf traits also moderates the effects of eucalyptus plantations on leaf decomposition with stronger inhibition being found on more palatable leaf species (inhibition of 51% for alder, *A. glutinosa*, 27% for oak, *Q. robur*, and no effect for eucalyptus, *Eucalyptus globulus*). This again suggests a stronger role of macroinvertebrates in mediating plantation effects on leaf litter decomposition (Ferreira et al., 2016). The re-analysis of a data subset considering only studies addressing the effects of eucalyptus plantation on alder and oak leaf litter decomposition in coarse and fine mesh bags fully illustrates these last results (Fig. 14.4).

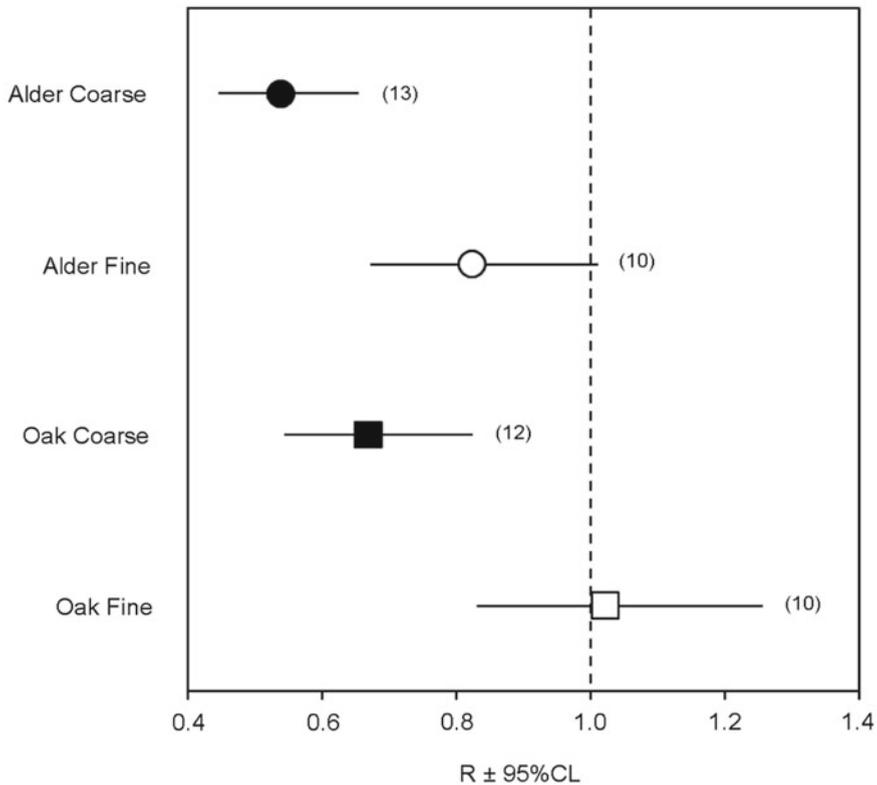


Fig. 14.4 Effect of eucalyptus (*Eucalyptus globulus*) plantations on alder (*Alnus glutinosa*) and oak (*Quercus robur*) leaf litter decomposition (k, dd) in streams in the Iberian Peninsula, when enclosed in coarse and fine mesh bags. $R = k_{\text{Eucalyptus stream}}/k_{\text{Reference stream}}$ (total $n = 45$ from 7 published studies; data subset from Ferreira et al., 2016). $R = 1$ (dashed line) indicates no effect of eucalyptus plantations while $R < 1$ indicates inhibition of k in eucalyptus streams. Effects are significant when the 95%CI (confidence interval) does not include 1 and treatments significantly differ when their 95% CI do not overlap. Values in parenthesis indicate sample size (i.e., number of comparisons between eucalyptus and reference streams). Global R : 0.73 (95% CI: 0.63–0.84). Missing effects sizes were detected to the left of the global R , with the new estimate being lower (Global R : 0.63; 95% CI: 0.54–0.73) indicating that the dataset used is conservative. Eucalyptus plantations inhibit k by an average of 27% (16–37%), but the effect is stronger for coarse than for fine mesh bags (Q_M ($df = 1$) = 11.68, $p < 0.001$). The effect also tends to be stronger for alder than for oak, especially in fine mesh bags, but no significant effect of litter species was found (Q_M ($df = 1$) = 2.92, $p = 0.087$)

The reported effects of eucalyptus plantations on litter decomposition in the Iberian Peninsula may differ from those in other regions because of differences in climate, type of native vegetation, eucalyptus species used in plantations and relative contribution of macroinvertebrates and microbes to litter decomposition, including their identity. A recent coordinated experiment evaluated the effects of eucalyptus plantations on *A. glutinosa* litter decomposition following a paired native forest vs.

eucalyptus plantation design in seven regions differing in environmental conditions: central Portugal, northern Spain, Kenya, Chile, Uruguay, central Brazil and southern Brazil (Ferreira et al., 2019). In general, total litter decomposition (i.e., driven by both microbes and macroinvertebrates) was significantly inhibited by 23% in this study, while microbially-driven litter decomposition was not significantly affected (Ferreira et al., 2019), which agrees with the meta-analysis addressed above. However, the magnitude and direction of plantation effects on litter decomposition varied among regions, suggesting that local conditions are important drivers of the effects of eucalyptus plantations on stream ecosystem functioning. When considering total litter decomposition, the inhibition ranged from 31 to 50% for temperate regions (Spain, Portugal, South Brazil and Uruguay), where macroinvertebrates are known to play a central role on decomposition and are most affected by plantations (see above). Contrastingly, in Kenya, central Brazil and Chile there was high variation in the response of total litter decomposition to plantations among streams within regions resulting in an overall non-significant regional effect. This was attributed to various factors which varied within and among regions: the high variation in the presence of shredders across streams in Kenya and in their contribution to litter decomposition (Boyero et al., 2015; Dobson et al., 2004); the high diversity in benthic organic matter standing stock in eucalyptus streams in central Brazil, allowing a high resource diversity that may sustain diverse decomposer communities, and eucalyptus leaves being of higher nutritional quality than more recalcitrant native leaves (Gomes et al., 2018); and eucalyptus litter being more palatable than recalcitrant native litter in Chile (Ferreira et al., 2019). When considering microbially-driven litter decomposition, no significant effect was found in most cases suggesting that microbial communities are functionally redundant and can adapt to environmental changes created by plantations more easily. However, stimulation of microbial litter decomposition by 32–110% was found in Uruguay and central Brazil and inhibition by 48% in Kenya, suggesting that microbial functional redundancy cannot be generalized across regions (Ferreira et al., 2019). The high solar irradiation and primary production in unshaded eucalyptus streams in Uruguay may allow a priming effect by which microbial decomposers benefit from labile carbon originating from primary producers (Danger et al., 2013; Kuehn et al., 2014) while the high diversity in benthic organic matter standing stock in eucalyptus streams in central Brazil may allow a higher diversity of decomposers and microbial inoculum (Ferreira et al., 2019).

To sum up, the effects of eucalyptus plantations on litter decomposition depend on the region (with stronger effects in regions where macroinvertebrates play an important role on decomposition), extent of plantations (with stronger effects if eucalyptus are planted in riparian areas), resource type (decomposition of leaves being more affected than that of woody substrates), litter species (decomposition of high quality leaves being more affected than that of low quality leaves), and the type of decomposer community involved (decomposition mediated by both microbes and macroinvertebrates being more affected than microbial-driven decomposition). Thus, stronger negative effects of eucalyptus plantations can then be expected for streams naturally receiving high-quality litter and where shredders are abundant.

14.2.2 Conifer Plantations (Fig. 14.5)

Conifer plantations are widespread in temperate and cold regions at both hemispheres. Nevertheless, the majority of studies tackling the effects of conifer plantations, mostly evergreen species (though *Larix* is deciduous), in streams have been carried out in temperate climates, where deciduous broadleaf species grow naturally. Consequently, as above, this review will inevitably offer a biased view of the effects of these plantations on litter dynamics in streams. Contrary to eucalyptus studies, which are based on a single genus (*Eucalyptus*), with most focusing on *E. globulus*, studies on conifers have focused on different species from the genera *Abies*, *Cryptomeria*, *Larix*, *Picea* and *Pinus*. Thus, even if many traits are shared among conifer species, the variability of effects reviewed here is larger than that of eucalyptus effects. Moreover, conifer species are in some cases planted within their natural distribution range or biome, but they are included here because they are planted as monospecific dense stands and/or are intensively managed, thus causing similar harmful effects to those of their exotic counterparts.

Similar to eucalyptus plantations, conifer plantations have been associated with an increase in soil hydrophobicity compared to deciduous forests, which has been linked to increases in surface runoff during rainfall (Butzen et al., 2015). A larger evapotranspiration of conifers compared to deciduous forests has been proven in many studies after the seminal work of Swank and Douglass (1974) and has been shown to reduce annual stream flow, especially in the dry season (Jackson et al., 2005; Jobbágy et al., 2013). Nevertheless, other studies have shown that this effect is context-dependent across the world (Komatsu et al., 2007, 2011). Conifer plantations have been related to water acidification in the UK (Ormerod et al., 1993). However, the majority of studies in other regions show that this forestry activity does not alter stream water physicochemical properties (Martínez et al., 2013, 2016; Riipinen et al., 2010; Swank & Crossley, 1988; Thompson & Townsend, 2004; Woodall & Wallace, 1972).

Most conifer species planted are evergreen, and thus create important differences in the timing of litter inputs to streams compared with streams flowing through deciduous forests where trees shed leaves in autumn (Inoue et al., 2012; Martínez et al., 2016). Consequently, while in streams flowing through deciduous forests there is a peak of benthic storage of coarse particulate organic matter (CPOM) in autumn, litter storage in conifer streams shows less variation throughout the year (Inoue et al., 2012; Martínez et al., 2016). Nevertheless, CPOM annual budgets are often similar between both stream types (Martínez et al., 2013, 2016; Miserendino & Masi, 2010; Riipinen et al., 2009). However, given the lower concentration of nutrients (both nitrogen and phosphorus) in needles than in leaves of many deciduous tree species (Casas et al., 2013; Martínez et al., 2013), streams flowing through conifer plantations store lower amounts of nutrients in benthic CPOM compared to deciduous forest streams (Martínez et al., 2016) (Fig. 14.5).

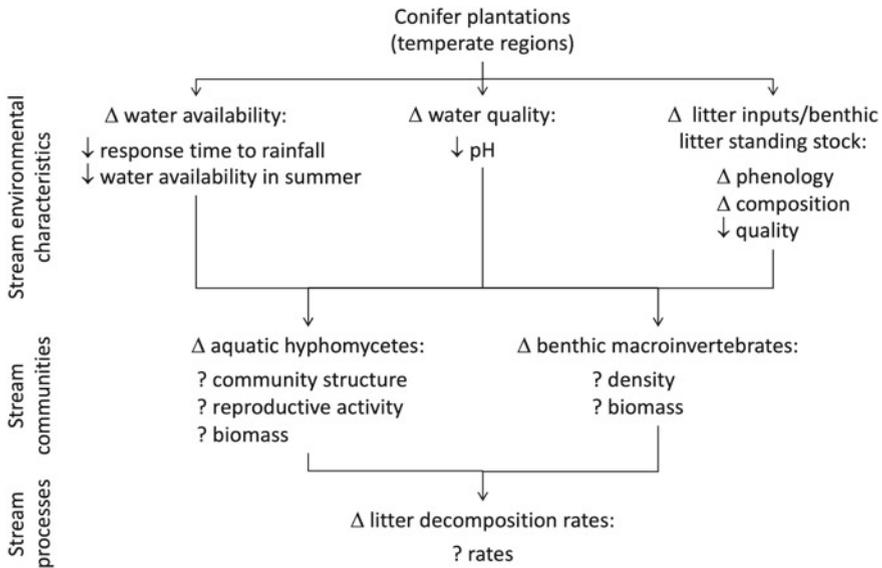


Fig. 14.5 Conceptual diagram of the effects of conifer plantations on stream environmental characteristics, communities and processes in temperate regions, which are characterized by native forests composed of mixed broadleaf deciduous tree species and seasonality in litter fall. Δ indicates change, \uparrow indicates increase, \downarrow indicates decrease and ? indicates contradictory results

Conifer tree species produce litter that is poor in nutrients and tough (Casas et al., 2013; Martínez et al., 2013), so aquatic communities can be energetically challenged in places where conifers are dominant (Fig. 14.5). Efforts to elucidate the effects of conifer plantations on aquatic communities have focused mainly on invertebrates, with information about microbial decomposers being scarce. Two studies addressed the effects of conifer plantations on aquatic hyphomycetes associated with decomposing litter. In northern Spain, Martínez et al. (2013) did not find a clear alteration in aquatic hyphomycete sporulation rates and community structure in streams flowing through *Pinus radiata* plantations compared to those flowing through deciduous forests. In contrast, Ferreira et al. (2017) found distinct aquatic hyphomycete community structure between streams flowing through *Cryptomeria japonica* plantations and streams flowing through native laurel forests in Azores (Fig. 14.5). Regarding the effects on macroinvertebrates, density and biomass of total invertebrates have been found to be lower (Friberg, 1997; Whiles & Wallace, 1997) or similar (Martínez et al., 2013, 2016; Monroy et al., 2017) in streams flowing through pine plantations compared to those under deciduous forests. Similarly, effects on shredders vary, with various metrics such as abundance and biomass being reported as higher (Friberg, 1997; Riipinen et al., 2009, 2010) or lower (Martínez et al., 2013, 2016; Miserendino & Masi, 2010; Riipinen et al., 2010; Whiles & Wallace, 1997) in streams flowing through conifer plantations. Moreover, specific taxa have been shown to respond differently to conifer plantations; while Friberg (1997) and Whiles

and Wallace (1997) found higher abundance of small shredders such as *Leuctra* in streams under conifer plantations, Martínez et al. (2016) and Monroy et al. (2017) reported a size-specific response to pine plantations, with lower density of small detritivores under this type of streams.

A highly consistent result is that conifer needles are processed slower than most deciduous leaves (Albariño & Balseiro, 2002; Hisabae et al., 2011; Martínez et al., 2013; Whiles & Wallace, 1997), due to their low quality (i.e., high toughness and low nutrient concentration) (Casas et al., 2013; Martínez et al., 2013). However, the effect of conifer plantations on litter decomposition capacity of streams is unclear (Fig. 14.6). Martínez et al. (2013) found lower alder litter decomposition rates in streams under conifer plantations. However, other studies showed faster decomposition rates for broadleaves (Riipinen et al., 2009; Whiles & Wallace, 1997) and needles (Whiles & Wallace, 1997) in conifer streams compared to native forest streams. Moreover, Riipinen et al. (2010) did not find differences in litter decomposition rates between streams flowing through broadleaf deciduous forests and conifer plantations when pH was fitted as covariate. Finally, Ferreira et al. (2017) did not find differences in decomposition rates of broadleaves and needles when comparing streams under native evergreen laurel forests and conifer plantations.

A special case is when natural grasslands are converted to plantations with no maintenance of intact riparian bands, as it occurs in mountain grasslands of central Argentina afforested extensively with the conifer *Pinus elliottii*. Grassland afforestation has profound effects in stream ecosystems, as trees growing in riparian ecotones reduce sunlight incidence and water temperature regimes (Cibils-Martina et al., 2017). Planted trees also feed streams with large amounts of CPOM, both woody and needle litter, thus shifting availability of basal food web resources from primary producers to allochthonous organic matter (Principe et al., 2015). As a result, afforested streams have strongly changed community structure and lower total invertebrate abundance and richness (Márquez et al., 2015). Interestingly, needle litter decomposes at similar rates in plantation and grassland streams, indicating that large storage of needle litter in afforested streams is not easily transformed and transferred along food webs (Márquez et al., 2017; Principe et al., 2015).

14.3 Other Planted Species and Management of Plantations

The scientific literature about the effects of eucalyptus and conifer plantations on litter decomposition in streams is steadily growing. Although these studies do not cover all the biomes and the entire latitudinal gradient where eucalyptus and conifer plantations have been established, the current knowledge can already be used with high confidence for predicting large-scale effects of eucalyptus and conifer plantations on streams worldwide. For other planted species, the number of studies comparing litter decomposition in plantation and native forest streams is scarce. A few studies show that litter decomposition rates in streams are reduced under intensive monospecific beech (*Fagus sylvatica*) plantations compared to traditionally managed forests with

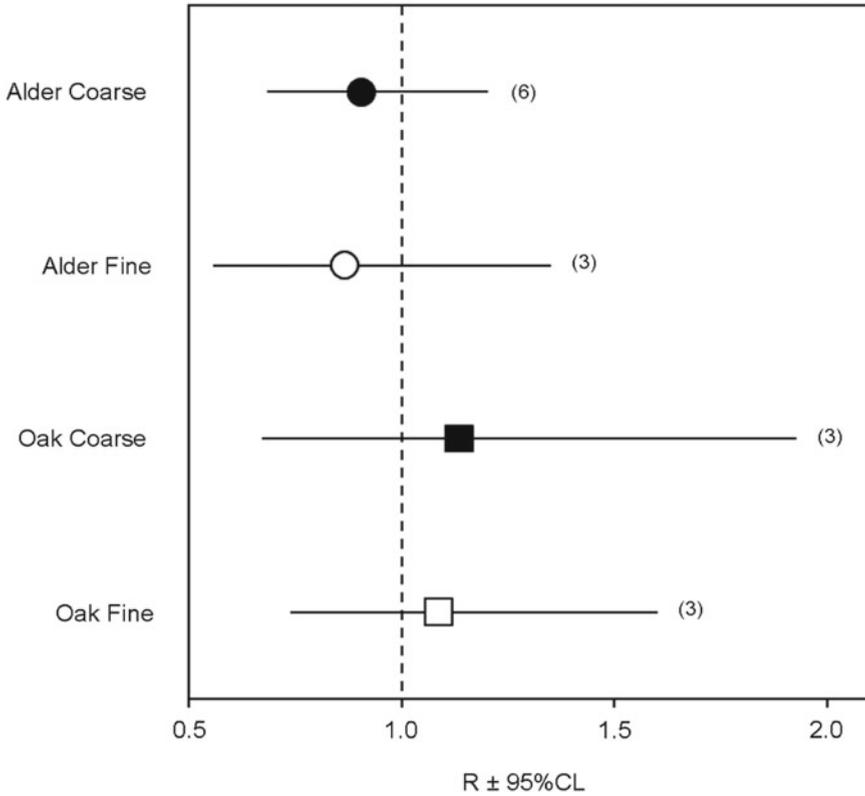


Fig. 14.6 Effect of conifer plantations on alder (*Alnus glutinosa*) and oak (*Quercus robur*) leaf litter decomposition (k ,/dd) in streams, when enclosed in coarse and fine mesh bags. $R = k_{\text{Conifer stream}}/k_{\text{Reference stream}}$ (total $n = 15$ from 2 published studies; data subset from Ferreira et al., 2016). $R = 1$ (dashed line) indicates no effect of conifer plantations, $R < 1$ indicates inhibition and $R > 1$ indicates stimulation of k in conifer streams. Effects are significant when the 95%CI does not include 1 and treatments significantly differ when their 95%CI do not overlap. Values in parenthesis indicate sample size (i.e., number of comparisons between conifer and reference streams). Global R : 0.96 (95% CI: 0.77–1.21). Missing effects sizes were detected to the left of the global R , with the new estimate being lower (Global R : 0.87; 95% CI: 0.70–1.08) but it does not change the result that was not-significant

high tree diversity (Ferreira et al., 2016; Hladyz et al., 2011; Lecerf & Chauvet, 2008; Lecerf et al., 2005). In contrast, litter decomposition rates are stimulated in streams flowing through plane (*Platanus hybrida*) riparian plantations than in streams flowing through native vegetation (Menéndez et al., 2013). Finally, a recent study in Malaysia found faster litter decomposition of both the native *Macaranga* sp. and the exotic oil palm (*Elaeis guineensis*) in streams flowing through oil palm plantations than in streams surrounded by native vegetation; accelerated litter decomposition was mainly driven by microbes, which were boosted by nutrient inputs from plantation fertilizers (Chellaiah & Yule, 2018a).

Without specific studies about the effects of other plantations, we can only infer their effects from the biology, ecology and traits of the planted species. Secondary compounds produced by plants force evolutionary adaptations in consumers in order to cope with that toxicity and gain an advantage over consumers who lack the same adaptation (Futuyma & Agrawal, 2009). This suggests that exotic species used in plantations may have a larger effect than planted native species on stream detritivore feeding and growth, which is more likely if exotic species differ deeply from native species in leaf litter traits. As studies accumulate, it is more evident that leaf traits override the effect of the origin of plant species on decomposition (Davis et al., 2011; Kennedy & El-Sabaawi, 2017). Litter traits such as the concentration of nutrients, structural compounds and toxic chemicals, and toughness are key for understanding the effects of plantations on litter decomposition. However, as the different traits might have synergistic or antagonistic effects on decomposition, predicting the consequences of plantations on stream nutrient cycling only by analysing litter characteristics is unattainable (also, plantations lead to other environmental changes unrelated to litter characteristics). Although some detritivores are able to compensate for the low quality of some resources by increasing their consumption (Flores et al., 2014) and resource preferences do not always correlate with growth and survival of consumers (Larrañaga et al., 2014; Lewis et al., 2017), consumption of litter, or, by extension, litter decomposition rates, might be a good approach to partially predict the effect of plantations on ecosystem level litter decomposition. Litter decomposition rates of worldwide planted exotic species span a wide range (Fig. 14.7), with the slowest (plane and *Cryptomeria japonica*, for total and microbial decomposition, respectively) and the fastest (poplar *Populus × canadensis* and eucalyptus) values falling within the range found for native species (Fig. 14.7). Moreover, decomposition rates of dominant deciduous forest species such as oak (*Quercus* spp.) or beech (*Fagus* sp.), which form old and well-preserved forests across Europe, are similar to those of species used for intensive plantations such as *E. globulus* or *Pinus* spp. (Fig. 14.7). Nevertheless, high tree diversity in native forests provides streams with a large range of litter traits, which make communities in these streams more productive and diverse than those under monoculture plantations. Similarly, the presence of a native species riparian buffer in streams flowing through plantations usually helps mitigating the direct effects of plantations on aquatic communities and on ecosystem processes, such as litter decomposition. For instance, riparian trees, which are adapted to floods, may offer resources of higher quality, such as alder (*Alnus* spp.) litter, which can help sustaining aquatic food webs in streams flowing through plantations with otherwise low quality litter available. Not only riparian buffers strips, but patches of well-preserved native forests in other places of the catchment can also create a similar effect. For example, macroinvertebrate communities in stream reaches surrounded by eucalyptus plantations, but with a large cover of native deciduous vegetation in the catchment were more similar to stream reaches totally surrounded by deciduous native vegetation than to those completely surrounded by eucalyptus plantations (Larrañaga et al., 2009a). Similarly, riparian buffers have been observed to preserve the structure of aquatic communities in streams in intensively managed oil palm plantations (Chellaiah & Yule, 2018b). As expected, decomposition rates are higher

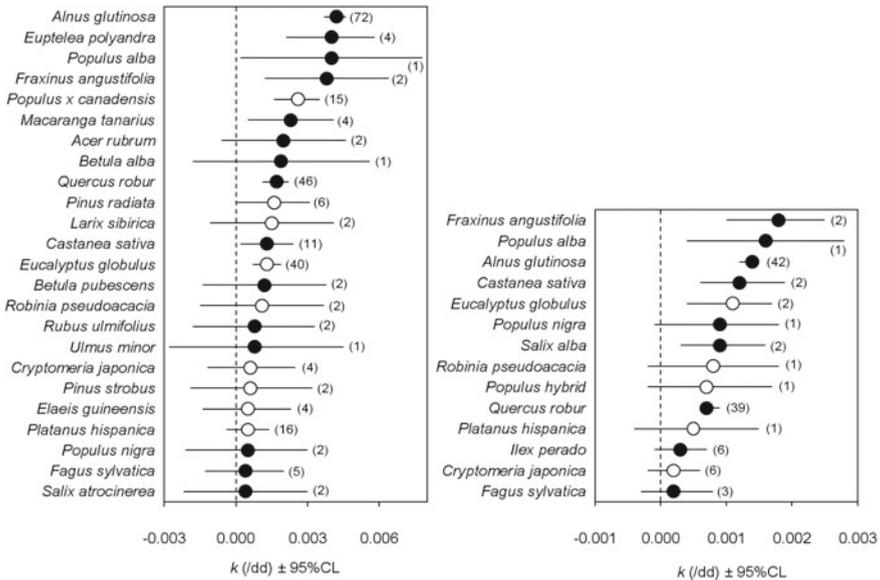


Fig. 14.7 Leaf litter decomposition rates (k , dd) of native (black circles) and exotic (white circles) tree species in streams in coarse (left) and fine mesh bags (right) (total $n = 357$ trials from 30 published studies) addressing effects of exotic plantations on leaf litter decomposition in streams by comparing rates of different species at the same location and/or by comparing rates of the same species in different locations (only studies that report rates for each location separately are considered; species considered native or exotic as defined by the authors). k values were calculated including streams flowing through native forests and through plantations together, but litter decomposition in coarse and fine mesh bags were kept separated. k values are non-significant when 95% CI includes 0 (dashed line). Values in parenthesis indicate sample size (i.e., number of k values). Large 95%CI are generally due to small sample size and interpretation needs to be made with caution. Studies used to build this plot marked with an asterisk (*) in the reference section

in locations where riparian buffers are preserved compared to locations with riparian zones altered by thinning or invaded by exotic species (Casotti et al., 2015). Likewise, litter decomposition rates were similar between streams with native vegetation and streams under eucalyptus plantations maintaining its riparian native vegetation (Ferreira et al., 2016). Beyond providing high quality litter to streams, riparian buffers help reducing nutrient and sediment inputs (de Souza et al., 2013; Jones et al., 2001) and increasing shading and mitigating temperature fluctuations (Broadmeadow et al., 2011; read Feld et al., 2018 for a recent review on riparian buffers). Nonetheless, the characteristics of the riparian buffers needed to preserve decomposition rates remain poorly known. In this line, Lecerf and Richardson (2010) observed that litter decomposition in streams with 10–30 metres wide riparian buffers in native conifer forest catchments that had been harvested 8 years before still differed in decomposition from streams flowing through preserved forest.

In addition to the negative effects attributed to most plantations as trees grow, harvesting is by far the single event with the largest impact on the structure and

functioning of streams. Flow regime is disrupted (Martin et al., 2000), inputs of fine sediments are incremented (Kreutzweiser & Capell, 2001), inputs of organic matter are suddenly augmented during the forestry activities and then reduced (Santiago et al., 2011) and biota is consequently altered (Banks et al., 2007). Road density built for harvesting increases sediment inputs and deposition and slows down in situ litter decomposition (Erdozain et al., 2018). Contrastingly, litter decomposition can be stimulated by the release of soil nutrients from clearcut practices (McKie & Malmqvist, 2009) and by changing plant composition in the riparian zones (Kominoski et al., 2011). Best management practices (BMPs) can reduce the effect of forest harvesting on streams (McBroom et al., 2008; Smolders et al., 2018), but even following BMPs logging is able to reduce decomposition (Kreutzweiser et al., 2008), as BMPs do not totally mitigate the changes originated. As expected, the partial felling of trees (i.e., thinning) instead of large scale clearcuts reduces the impact of plantations on stream communities (Quinn et al., 2004) and litter decomposition (Lecerf & Richardson, 2010). Regardless of changes induced by clearcutting, they are long lasting; more than a decade is usually needed for a full recovery back to reference conditions. For instance, Stone and Wallace (1998) reported stream macroinvertebrate communities had recovered to reference conditions only 16 years after catchment clearcutting. Similarly, effects of harvesting on litter decomposition rates can last more than a decade (Griffith & Perry, 1991; Guevara et al., 2015; Webster et al., 2014; Yeung et al., 2017). In plantations where the harvesting cycle is shorter than the time needed for the recovery of stream communities and decomposition rates, we might anticipate an accumulation of effects of multiple harvesting cycles, although this has not been proven yet.

When to harvest a plantation is a paramount decision in order to maximize the production of resources. As trees develop the biomass produced per year decreases, thus, short harvesting cycles are optimal from an economic point of view, albeit they can become the main culprit of the impact that some planted species have on aquatic systems. Not all planted species are managed equally and there is a link between their productivity and the tree species cultivated. Hengeveld et al. (2012) summarized the applicability of different plantation tree species to different forest management approaches in Europe (Fig. 14.8). These authors consider five categories of forestry, from the most intensive, based on short rotations, to the most natural, i.e., natural reserves. The genera most suited for intensive forestry, and thus least suitable for conservation purposes, were *Eucalyptus*, *Robinia* and *Pseudotsuga*, exotic species in Europe (Fig. 14.8). These three genera are considered totally inadequate for nature reserves in Europe, with *Eucalyptus* not even adequate for close-to-nature forests. On the other end *Carpinus*, *Betula*, *Fraxinus*, *Alnus* or *Fagus* are considered of high value for nature reserves, but inadequate for short rotation forestry (Fig. 14.8). This analysis emphasizes that traits of particular tree species can constrain their use more to production or to conservation. Nevertheless, most species show a relatively high applicability (>20%) to at least three management approaches, which shows that beside the selection of species, the intensity of plantation exploitation can be adapted either to maximize production or to minimize environmental impacts. In this sense, the previously described negative effects of native beech plantations on freshwater

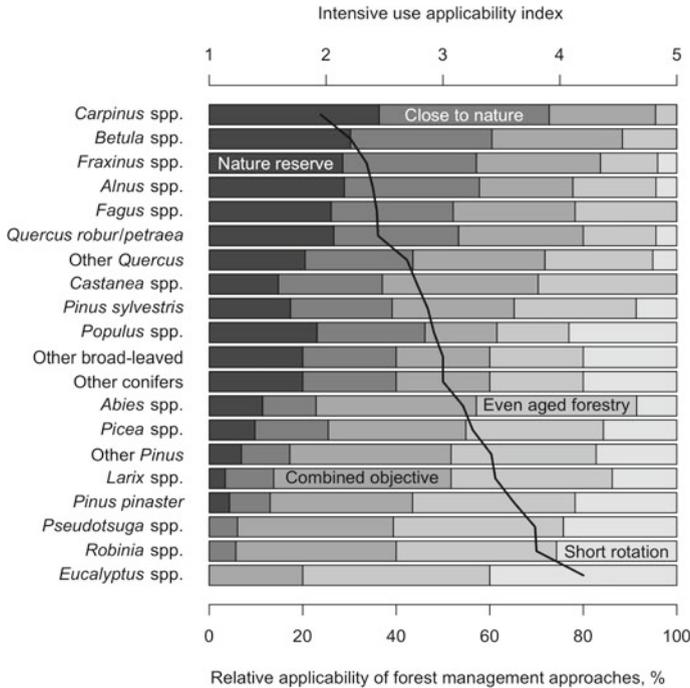


Fig. 14.8 Relative applicability of different tree genus/species for different forest management approaches in Europe, from nature reserves to short rotation forestry. The transversal line compiles the relative applicability into a single index (i.e., Intensive use applicability index; scale in top X axis) considering the five forest management approaches as numbers, from Nature reserve (1) to Short rotation forestry (5) and weighing them by the relative applicability percentages (e.g., *Eucalyptus* spp.: $(3 * 20\% + 4 * 40\% + 5 * 40\%) / 100 = 4.2$). Species are ordered following the applicability to intensive use (less to more, from top to bottom) (Source [Hengeveld et al., 2012])

litter decomposition (Ferreira et al., 2016; Hladyz et al., 2011a, 2011b; Lecerf et al., 2005), illustrates the relevance of *how* forestry is implemented beside *which* is the species selected for plantations.

14.4 Concluding Remarks

From this review we can distil the following ideas:

1. Plantations can alter litter decomposition in stream ecosystems by means of changes in hydrology, water quality, leaf litter inputs (including quantity, quality, timing) and biota.
2. Some planted tree species have received special scientific attention (eucalyptus and conifers), with most of the studies carried out in Europe and North America.

Studies on other tree species and regions are needed to allow drawing more general conclusions.

3. The decomposability of the leaf litter produced by planted species is similar to that of many species dominating native forests. Deficiency or absence of appropriate riparian buffers strongly contribute to the deleterious effect of plantations on stream decomposition capacity.
4. Although planted tree species can display intrinsic economic and environmental values, they can always be managed in a range of different ways to find a trade-off between productivity and conservation.
5. There are proven measures that help mitigate the impacts of plantations on litter decomposition in freshwaters (conserving riparian buffers and following Best Management Practices, for instance). Unfortunately, they are yet to be implemented in many places of the world.

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The ones used to create Fig. 14.7 have an asterisk at the end, “*”.

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