Forestry Advance Access published December 3, 2013

Forestry An International Journal of Forest Research

Institute of Chartered Foresters

*Forestry* 2013; **0**, 1–10, doi:10.1093/forestry/cpt050

# Multi-scale assessments highlight silvicultural opportunities to increase species diversity and spatial variability in forests

**Klaus J. Puettmann[1]* and John C. Tappeiner[2]**

[1]*Department of Forest Ecosystems and Society, Oregon State University, Corvallis, OR 97331, USA*
[2]*Department of Forest Engineering, Resources, and Management, Oregon State University, Corvallis, OR 97331, USA*

*Corresponding author. Tel.: +1 5417378974; Fax: +1 5417371393; E-mail: klaus.puettmann@oregonstate.edu

*Received 28 March 2013*

Foresters influence short- and long-term development of plant neighbourhoods, stands and landscapes by their management practices. In the past, most of these practices have aimed to homogenize the composition and structure of stands for efficient wood production. This publication provides an overview of and guidance on how to recognize and efficiently utilize opportunities to modify silvicultural practices, with the goal of increasing species diversity and spatial variability within stands and landscapes at minimum cost. We suggest a procedure for selecting candidate treatments that might be used in this way. We further suggest that multi-scale evaluations (e.g. different objectives, such as economic *vs* ecological goals; different time frames from short-term to rotation lengths; and longer, different spatial scales from plant neighbourhoods to landscapes) can highlight opportunities to increase species diversity and spatial variability during implementations of 'standard' forest management practices. For example, opportunities may derive from situations where management practices did not achieve their intended goals or where natural disturbances can be viewed as stimuli that initiate opportunities to increase heterogeneity. With modifications, silvicultural responses to such conditions can provide efficient, low-cost (or even cost saving) means to increase species diversity and spatial variability. Using replanting, treatment of 'minority' species, variable spacing guidelines, and other examples, we show how varying the spatial and time scales of evaluations for such modified treatments can influence conclusions about costs and ecological impacts. Consequently, the choice of evaluation scales can be a deciding factor in whether treatment modifications are considered as economically justifiable ways to achieve a suite of diverse objectives.

## Introduction

In many parts of the world, recent developments in forestry include efforts to find alternatives to traditional forest management, such as ecological forestry (Franklin *et al.*, 2007), close-to-nature or continuous-cover forestry (Schütz *et al.*, 2012) and variable retention (Gustafsson *et al.*, 2012). These trends developed in response to increased public scrutiny, higher emphasis on a variety of ecosystem goods and services in addition to timber production, and increased ecological understanding about disturbance regimes and historic conditions in stand structures. Consequently, a common theme in these alternative management approaches is a higher emphasis on species diversity and spatial and temporal variability. For example, Muir *et al.* (2002) summarized how variability in young forest stands favoured a diversity of organisms ranging from birds to lichens and bryophytes. This emphasis on increased species diversity and spatial variability is based on the notion that stands with such characteristics would be better able to provide desired ecosystem goods and services, especially in the context of global change (Schütz *et al.*, 2012; O'Hara and Ramage, 2013). However, much research is needed to develop a clear understanding of the value of specific amounts and scales of species diversity and spatial variability, especially as related to the ability of forest ecosystems to adapt to changing conditions (Puettmann, 2011; Messier *et al.*, 2013).

Scaling is an important concept in ecology (Wiens, 1989) that is relevant to silviculture. Traditionally, silviculture has emphasized a single spatial scale, defined by stands (which typically range from 5 to 50 ha), and a single time scale, defined by rotation periods (which typically range from 30 to 80 years) (Nyland, 2002; Tappeiner *et al.*, 2007; Puettmann *et al.*, 2009). Silvicultural practices typically act on small-scale ecological processes, such as seedling establishment, plant competition and individual tree growth (Poage and Tappeiner, 2002; D'Amato and Puettmann, 2004; Dodson *et al.*, in press). As the suite of management objectives has expanded beyond timber production, however, silvicultural practices typically are also evaluated in terms of larger spatial scales, such as conservation goals across landscapes or regions (Pojar *et al.*, 1994; Fall *et al.*, 2004). At the same time, management prescriptions are affected by and affect a suite of ecosystem variables and processes (Tappeiner *et al.*, 2007). These processes act at a wide range of spatial and time scales, ranging from leaf or plant levels (e.g. herbivory; Weisberg and Bugmann, 2003) to landscapes (e.g. fires; Bergeron *et al.*, 2001; Sensenig *et al.*, 2013 or wind;

Hanson and Lorimer, 2007) or even global scales (e.g. climate change; Chmura *et al.*, 2011). In light of trends towards multiple management goals and a better understanding of the importance of cross-scale interactions in ecosystems (Gunderson and Holling, 2002; Peters *et al.*, 2007; Messier *et al.*, 2013), there may not be a single proper or optimal spatial and time scale for the evaluation of the success or efficiency of silvicultural treatments (e.g. the stand level for traditional forest management and the tree level for close-to-nature forestry; Puettmann *et al.*, 2009; Bauhus *et al.*, 2013). This suggests that evaluation procedures benefit from utilizing multiple scales (Messier *et al.*, 2013).

In the following, we provide suggestions for assessing opportunities and implementing silvicultural practices that increase species diversity and spatial variability in forest ecosystems within the current policy, administrative and economic settings and that are typically applied within conventional forest management operations (i.e. reforestation, thinning and reproduction cutting practices). We describe multi-scale evaluation procedures (e.g. logistical, ecological and treatment related) that can be used to assess which practices have good potential to efficiently achieve such goals, and we give examples of how these practices could be modified to increase species diversity and spatial variability, possibly at no or very low cost. We limit our focus to silvicultural practices, i.e. treatments that are applied at stand scales or smaller. However, as their implications play out over various time scales and in landscape settings, we highlight how the interpretation of the practices changes over a gradient of time (short-term to rotation length and longer) and spatial (plant neighbourhoods to landscape) scales.

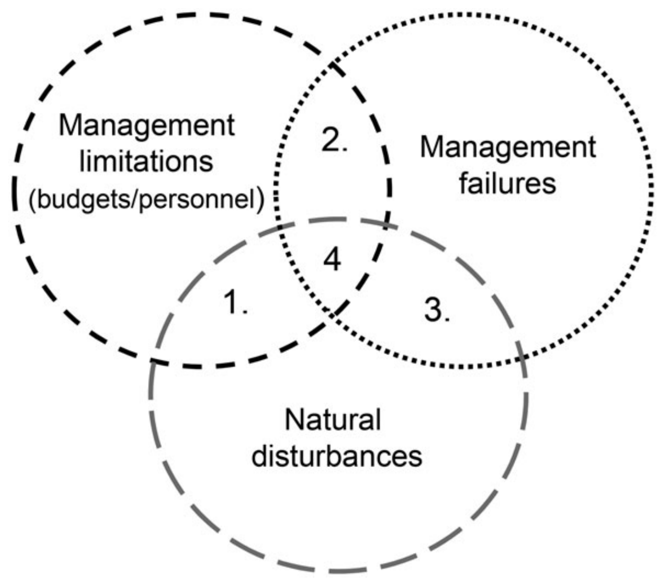


**Figure 1** Conceptual diagram highlighting three issues that can guide silviculturists when looking for opportunities to increase species diversity and spatial and temporal variability in forests. Treatments or options that satisfy multiple criteria and represented in the intersection of two or three circles (#s 1, 2, and 3 and #4, respectively) are of special interest. For example, repeated treatments are more likely to be dropped or modified in years with limited budgets or staffing (#2). Alternatively, stands that not only had regeneration failures, but were also damaged by fires (#3) have a higher likelihood of providing opportunities for deviation from standard practices.

## Identifying opportunities

Those interested in finding opportunities within current management operations to increase species diversity and spatial variability in forests with minimal or no additional costs would benefit from developing a more formal assessment approach. For example, multiple hierarchical scales may be used in making silvicultural decisions (Puettmann *et al.*, 2009). We suggest that silviculturists seeking opportunities to increase species diversity and spatial and temporal variability use criteria from three categories – management limitations, management failures and natural disturbances – to assess their treatment options (Figure 1).

Management limitations may result from logistical constraints that determine the implementation of silvicultural practices (Figure 1). In most organizations, silvicultural treatments, especially those that are considered as an investment, are often constrained by budgets or labour force considerations, including limited staff or personnel turnover. Such constraints can limit applications of silvicultural practices, even when these treatments are known to produce an acceptable rate of return (Huuskonen and Hynynen, 2006). For example, public land management agencies in the Pacific Northwest of North America are forgoing opportunities for pre-commercial thinning (PCT) in many cases, even though these operations are useful in achieving management objectives (e.g. fire and disease risk reduction, creation of wildlife habitat and maximizing timber values).

Management failures include silvicultural treatments that did not result in desired outcomes (Figure 1), necessitating re-treatments at least on portions of the land. Re-treatment operations are often more costly than initial treatments because of the additional expenses involved in surveying for re-treatment needs, the smaller size of operations (more setup costs per unit treated), and additional efforts required to monitor contracts and implementations (e.g. see the section about replanting below). These treatments are also likely to be implemented at a sub-optimal time; for example, seedlings planted during re-treatment may face more competition, as the effects of the initial weed control treatments fade over time (Wagner, 2000).

The third category is based on the notion that any silvicultural regime must be viewed within the context of natural disturbances (Figure 1), which could potentially impact management success at one level or another. Such disturbances are often viewed as 'accidents' or as challenges to be overcome. An alternative viewpoint, however, is that natural disturbances provide an opportunity and set the stage for increasing heterogeneity in otherwise or previously homogenous stands and landscapes (Bergeron *et al.*, 2001).

The three categories shown in Figure 1 are not mutually exclusive, but overlap. Thus, while any category by itself may highlight treatments as candidates for increasing species diversity and spatial variability in forests, treatments that fall in multiple categories provide even more likely opportunities (Figure 1). For example, treatments in response to undesirable developments after natural disturbances and treatments designed to overcome management failures are likely the first to be suspended when budget or workforce availability becomes limiting (represented by Areas 1 and 2 in Figure 1, respectively), especially when management failures and natural disturbances affect the same stand (represented by Area 4 in Figure 1).

Going through an assessment process, e.g. assess management procedures in the context of the issues outlined in Figure 1, will highlight opportunities for modifying silvicultural practices that may not only help achieve ecological goals, but may also be cheaper in the short and/or long-term. Such assessments will not provide suggestions on how practices could be modified. Obviously, one response is to address the concerns that flagged the condition in the first place, for example, by alleviating the administrative or logistics issues. In this paper, however, we limit our discussion to the modification of silvicultural practices that may be highlighted through such an assessment, with a special focus how such modifications can provide efficient means to increase species diversity and spatial variability within and among stands.

Reflecting the notion that cross-scale interactions are crucial in determining ecosystem development (Gunderson and Holling, 2002; Messier *et al*., 2013), treatment modifications should be critically reviewed using the concept of scaling (Wiens, 1989; Hill and Hamer, 2004). Switching scales or assessing treatments along a wide gradient of spatial, time and hierarchical scales (i.e. short- *vs* long-term impacts; small-scale, neighbourhood *vs* large-scale, landscape-level impacts; different sets of ownership objectives) can provide insights that are not evident, when management is focused mostly on stand and rotation age scales. The following examples highlight how such insights may clarify choices and tradeoffs, as well as how the choice of spatial and time scales used for assessing cost and ecological impacts of silvicultural treatments can influence assessment results.

## Developing prescriptions

This section describes several common management situations and practices typically implemented on private and public ownerships: replanting after seedling mortality, leaving hardwoods or minority species during reforestation, variable spacing, advanced regeneration and reproduction cutting. Ways in which these practices could be modified to increase heterogeneity within and among stands at little or no extra cost are also discussed. We also specifically highlight how both prescriptions for and evaluations of silvicultural treatments are influenced by the choice of spatial and time scale used for analyses and assessments. Our aim is to show how changing scales can alter the reputation of treatments, in some cases making modifications of current practices more acceptable or even desirable. Although there is no single proper scale, understanding the bias and implications introduced by the choice of scales provides especially useful insights for choosing management actions aimed at increasing variability. Most of the examples are from the Pacific Northwest of North America, but the implications of selecting time and spatial scales for assessment of management practices are not limited to individual regions and ownerships.

### *Replanting after seedling mortality*

Most planting operations in managed forests result in fairly high seedling survival rates, but rarely reach 100 per cent (Duryea and Dougherty, 1991). Patchy seedling mortality after planting may lead to small-scale openings in developing stands. This mortality then results in patches of shrubs, herbs and grasses in otherwise densely stocked stands (Puettmann and Berger, 2006). These patches provide unique habitats for a variety of insects and wildlife species and have the potential to act as diversity and inoculation centers in future stand development (Paquette and Messier, 2009). Depending on the mortality agent, these patches may be concentrated in one area, for example, drought-related mortality on south-facing slopes or rocky outcrops (Koepke *et al*., 2010) or beaver (*Castor canadensis)* damage in riparian areas (Raffel *et al*., 2009). Alternatively, patches of root disease or mountain beaver (*Aplodantia rufa*) damage are likely to be more randomly distributed throughout a stand (Neal and Borrecco, 1981). Typical stand-level objectives, such as timber production, will emphasize that the whole stand should be stocked. Consequently, landowners have seedling stocking criteria that will trigger replanting. These criteria typically include overall, stand-level stocking standards and the maximum acceptable size of mortality patches. The determination of such thresholds for actions, i.e. replanting, typically emphasizes economic criteria, with less emphasis given to ecological impacts of mortality patches (in part because such impacts are tough to quantify). These economic criteria are based on a number of assumptions, such as future growth and timber values, as well as the timing of harvest operations. It is important to note that these assumptions are based on predicted conditions at rotation age, rather than over a variety of time scales. Figure 2a,b shows how changing the time scales can influence the

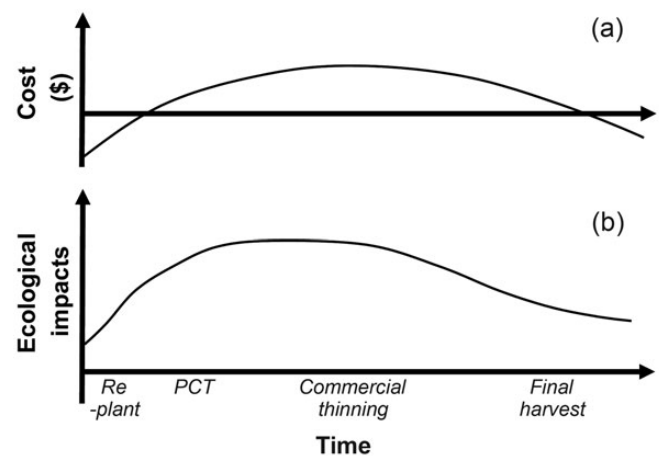


**Figure 2** Relative costs (a) and relative ecological benefits (b) of not replanting as an example changing the time scale used to assess replanting efforts can influence results. The time scale on the *x*-axis is presented in terms of phases in stand development when different management activities typically occur (*PCT*). If assessed early (e.g. during the ‘Replanting’ phase), not replanting is reflected in cost savings (negative costs). If assessed later (e.g. during the ‘Commercial thinning’ phase), lower volume growth due to mortality patches results in lower income from thinnings or early final harvest operations, i.e. not replanting is a net cost for the landowner. As stands mature and neighbouring trees fill in these gaps (e.g. during the ‘Final harvest’ phase), the loss in harvest volume due to the lack of replanting can be minor. Thus, when assessed at such long timescales, not replanting leads to cost savings. In contrast, ecological impacts (b) of the increased heterogeneity due to not replantings will be generally positive and highest, if assessed during the stem-exclusion phase (i.e. during the ‘PCT’ and ‘Commercial thinning’ phase).

outcomes of such evaluations in terms of relative costs and relative ecological impact, respectively.

The following aspects must be considered in these evaluations. First, in many instances, the reasons for the initial seedling mortality in a given stand, such as herbivory, frost pockets, shallow, rocky soils, severe shrub or grass competition, are still in place after replanting. Thus, the likelihood of planting success for replanted seedlings is generally lower than for the initial planting operations. Second, replanting in order to reach standard density targets, such as 1075 trees per hectare (tpha) in Douglas-fir (*Pseudotsuga menziesii*) plantations in the Pacific Northwest of North America or 3000–4000 tpha in Norway spruce (*Picea abies*) plantations in Austria, can be costly. Such efforts include travel and survey time, as well as planting operations, leading to high planting costs per seedlings, especially if the mortality is distributed in small patches throughout stands or in areas with limited access, such as riparian areas (planting costs per seedlings are estimated to be twice as high as initial plantings; D. Anders, Oregon Department of Forestry and M. Gourley, Starker Forest, personal communication). Third, additional expenses may occur, such as repeating weed control treatments when the initial treatment has lost its effectiveness (Wagner, 2000). Fourth, since replanted seedlings are likely younger and thus smaller than those established in the initial planting, the size differential may express itself in slower growth of replanted seedlings, making them likely candidates for removal during early thinnings. Thus, replanted seedlings may not contribute as much to final harvest volumes as initially expected. In addition, neighbouring trees benefit from gap openings (Dodson *et al.*, 2012) left by seedling mortality, thus further reducing the harvest volume losses associated with that initial mortality. Therefore, economic evaluations of the replanted seedlings (in terms of investment return per seedlings) reflect higher costs and lower harvesting volumes than typically applied in rotation-based financial assessments (note that not all landowners apply such rigorous financial assessments and this issue may not apply for many small woodland owners). In many cases, these evaluations will likely suggest overall stocking standards and mortality patch sizes, below which economic losses due to patchy seedling mortality are minor. They may also highlight thresholds under which replanting efforts are more expensive than leaving mortality patches, even in the long-term; thus suggesting win–win situations in terms of ecological and economic benefits.

Figure 2a highlights how an economic assessment of replanting efforts may change as function of time scale, i.e. at which time an assessment takes place. Initially, forgoing planting leads to cost savings (during 'Replanting' and early 'PCT' phases in Figure 2). As stands mature, unplanted areas are not stocked and thus are not producing income (during the 'Commercial thinning' and early 'Final harvest' phases in Figure 2). This is reflected as lost income, i.e. not replanting is a cost to the landowner. However, as stands mature and neighbouring trees overtake and fill in gaps (the later 'Final Harvest' phase in Figure 2), the overall difference in harvested volume due to lack of replanting will shrink. Eventually stands may reach a point where seedling mortality patches are not reflected in lost harvest volume and income; instead, the lack of replanting actually resulted in cost savings when assessed at this longer time scale.

Figure 2b shows that ecological impacts also vary over time. Initially impacts are limited to the small area of the mortality patch (the 'Replanting' phase in Figure 2). As the remaining stand develops through the stem-exclusion phase ('PCT' and 'Commercial thinning' in Figure 2), the ecological values of open conditions and the ensuing variability of vegetation are reflected in higher ecological impacts and higher value of gaps, especially in contrast to conditions within the stand interior (Fahey and Puettmann, 2007, 2008). Later, as neighbouring trees continue to expand (the 'Final harvest' phase in Figure 2), these ecological impacts may be reduced or lost, as open patches close and open conditions and associated variability in understorey vegetation disappear. Depending on the spatial distribution of gaps, their ecological impacts – such as acting as diversity islands or inoculation centers to nearby forest interiors – may be spread throughout the stand or concentrated, e.g. in riparian areas or rock outcrops. Depending on their size, gaps will be occupied more or less quickly by neighbouring tree crowns and roots (Puettmann and Berger, 2006), thus reducing the ecological impacts that gaps provide in otherwise dense plantations. If these impacts are of critical importance, thinning operations can be designed to maintain openings by removing trees adjacent to mortality patches to enlarge gaps. In the case of PCT, these removals will lead to added costs that may never be fully recovered, until the end of the rotation (Figure 3a). On the other hand, positive ecological impacts of larger gaps (Fahey and Puettmann, 2007, 2008) may become especially valuable during the later life of stands, if the gaps act as biodiversity inoculation centers (Figure 3b). This detailed description of the economic and ecological evaluation of re-planting is presented as an example

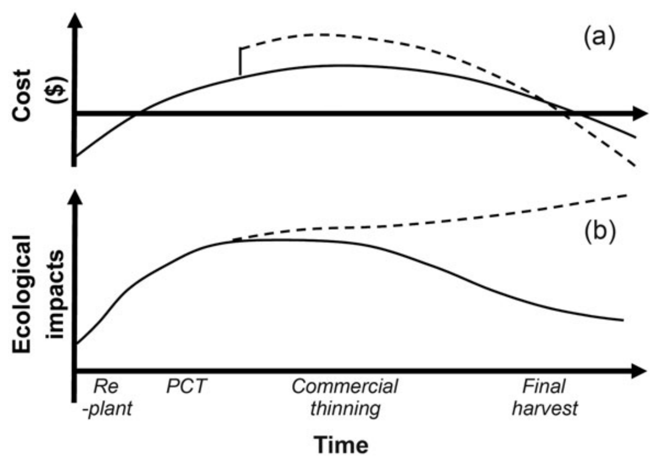


**Figure 3** Relative costs (a) and relative ecological benefits (b) of not replanting as influenced by PCT. The time scale on the *x*-axis is presented in terms of phases in stand development when different management activities typically occur (in italics). The solid line represents a scenario of not replanting and without follow-up treatments to maintain gap openings. (See Figure 2 for a more detailed explanation). The dashed line represents impacts of a PCT entry, which includes the cutting of border trees to enlarge and thus maintain gap openings. The expense of the PCT treatment and lower stand volume due to larger open patches are reflected in higher costs during the 'Commercial thinning' and early 'Final harvest' stage. Similar to untreated stands, the impact of not replanting is partially or fully offset by the trees benefiting from adjacent open patches and more space after the PCT operations, i.e. not PCT-ing will be a cost saving, if assessed at this time scale. On the other hand, ecological impacts (b) of the PCT treatments can increase over time, as the wider gap openings provide more variable vegetation and habitat and can act as inoculation centers for adjacent areas.

of the type of analytical procedure that can be applied to other treatments, including the silvicultural practices described below. However, in all these cases the availability of funds, personnel, material and access to planting sites may also influence decisions whether a treatment, e.g. replanting, can be implemented (see Figure 1).

### *Leaving hardwoods or 'minority' species during reforestation*

Typical site-preparation practices in industrial operations aim to minimize competition from unwanted vegetation to improve growth and survival of crop trees (Wagner *et al.*, 2006); however, leaving individual hardwoods or small patches of hardwoods in conifer plantations provides another opportunity for diversification (Humphrey, 2005). This could mean either deliberately avoiding the removal of selected hardwoods in an initial entry or omitting the removal of hardwoods that were skipped inadvertently during previous treatments. In regenerating conifer stands in the Pacific Northwest, hardwoods such as red alder (*Alnus rubra*), big-leaf maple (*Acer macrophyllum*), Pacific madrone (*Arbutus menziesii*) and tanoak (*Notholithocarpus densiflorus*) sprout vigourously (Hughes *et al.*, 1987; Harrington and Tappeiner, 1997) and often can initially overtop conifers (Harrington and Tappeiner, 2009). These overtopped seedlings rarely grow into the canopy and typically stagnate at shrub height. The analytical procedure applied to the first example (replanting) and pictured in Figure 2 can be applied here over both time and spatial scales, as will be demonstrated below.

Landowners may commonly evaluate the success of stand management at early time stages in terms of gains or losses in crop-tree growth and thus may support control practices to eliminate competition from sprouting hardwoods (Hobbs *et al.*, 1992). For example, controlling hardwood density in young stands in southwestern Oregon will likely increase conifer density and growth in early stages of stand development (Harrington and Tappeiner, 2009). However, at later stages stand timber volume may not be significantly reduced in stands with moderate to low hardwood densities, which also supported more herbs and shrubs (Harrington and Tappeiner, 2009). Therefore, at later stages, when financial assessments are driven by final harvests, the value of some early differences may be diluted or lost. At this stage, the early removal of hardwood clumps may be viewed as an unnecessary expense (Figure 4a). Another example of such a response pattern was found when herbicide applications were successful at initially releasing conifers in mixed conifer/hardwood stands in the northeastern USA. After a few decades, the impact of PCT had overwhelmed the effects of the initial herbicide applications, resulting in stands for which the effects of early herbicide treatments were 'compositionally and structurally indistinguishable' (Olson *et al.*, 2011). Total merchantable volumes and merchantable conifer volume were not statistically significantly different at age 40, while subtle differences remained. In these situations, results of economic assessments of herbicide treatments would also vary with cost of herbicide applications, cost-savings when PCT-ing herbicide treated plots, and value differences between conifer and hardwood species.

Similarly, the specific spatial scale used to assess the costs and ecological impacts of hardwood removal can have a big impact on the results. Figure 4a,b provides examples of how economic and ecological evaluations, respectively, of these treatments can vary, depending on the choice of spatial scales of assessment. For example, bigleaf maple is a common hardwood tree species on riparian and upland sites in interior and coastal mountains from Washington to California. The ecological impacts of bigleaf maple in conifer-dominated landscapes include providing a stem and branch substrate for lichens and bryophytes in lower strata (Cobb *et al.*, 2001), and food for birds and other animals (i.e. ecological impacts at stand and larger scales, depending on the mobility of seed eaters). In contrast, ferns often grow in the maple canopy, and birds use large cavities in maple trunk and branches as nesting sites (i.e. ecological impacts at small spatial scales) (Bunnell, 2013). Litterfall beneath maple is greater and turnover rates are higher than beneath adjoining Douglas-fir, influencing soil organic matter and nutrient content at small spatial scales (Fried *et al.*, 1989). In addition, large maple trees with their yellow fall foliage and their green, aerial 'winter garden' are aesthetically pleasing (at stand scales and larger). Whether any of these characteristics by themselves or in combination justify leaving bigleaf maple in managed conifer stands is – among other things – influenced by the spatial scale of assessment. Specifically, conditions of the larger spatial context are important for such any assessment, e.g. how many maple or similar species are present in the nearby landscape (Figure 4b). In landscapes

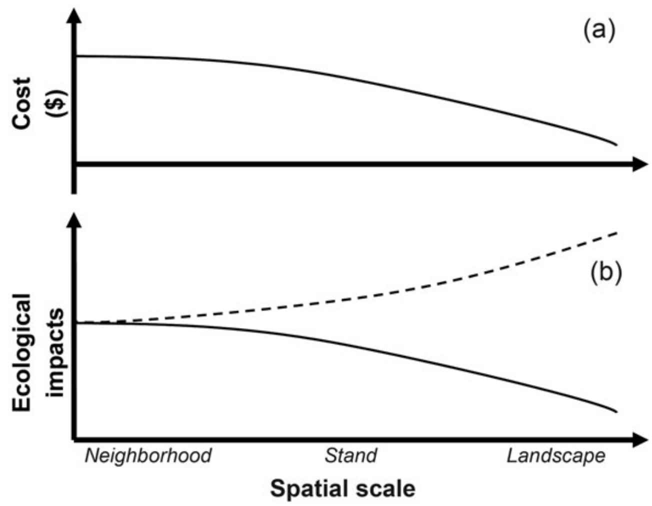


**Figure 4** Relative costs (a) and relative ecological benefits (b) of not replanting as an example of how changing the spatial scale influences the results of leaving residual hardwoods in conifer dominated stands. The spatial scale on the *x*-axis ranges from neighbourhood (e.g. several square meters) to stand (5–50 ha) to landscape (multiple square kilometers) scales. This example assumes assessment at mid rotation (i.e. during the 'Commercial thinning' phase, as used in Figures 2 and 3). The costs of leaving individual hardwoods, as reflected in lower volume in thinning operations are highest, when they are assessed for the hardwood neighbourhood, which may contain no commercial trees. Assessing the costs across a larger and larger areas is basically diluting such effects, e.g. loss of commercial volume due to an individual hardwood patch may be negligible when assessed across a landscape. A similar pattern, including dilution effects is evident for ecological impacts (b), in a landscape with hardwoods present (solid line). In contrast, in a landscape without many hardwoods (dashed line), the ecological impacts of leaving individual hardwoods are higher if these hardwoods are the only ones in a stand or in a landscape, reflecting increased rarity.

where maples or similar species are common, both the costs and the ecological impacts of leaving individual maples are highest when assessed at small scales and will be 'diluted' as assessments are done at larger and larger spatial scales, i.e. few selected hardwoods or their removal have very little impact at landscape scales. On the other hand, in landscapes where maple or hardwoods have been removed in the past, leaving these rare vegetation components not only could lead to cost savings (Figure 4a) but also could have high ecological and social/aesthetic values (Brown, 2012). In ecological terms, the importance of rarity increases with the size of the area with limited hardwoods (Humphries *et al.*, 1995; Arponen *et al.*, 2005), i.e. the ecological impacts of leaving maples will increase with spatial scales (Figure 4b).

### Variable spacing

Density management treatments, including aspects of spatial arrangement of trees could also be assessed in terms of the variety of spatial scales. This approach will highlight discrepancies between the small scales of plant interactions, stand scales used for management prescriptions, and larger scales used for evaluations of, for example, conservation goals. Standard planting and thinning prescriptions are based on the assumptions that competition is the main driver of tree growth pattern and that spatial distributions become more regular over time, overlaid by inherent site variability. Consequently, more-or-less regular spacing should provide optimal stand growth (Getzin *et al.*, 2006, but see DeBell and Harrington, 2002). However, a closer assessment of the variability inherent in sites and forests suggests that variability in tree spacing not only can lead to variability in stand structures, but can also have beneficial effects on stand growth. For example, Marshall and Curtis (2002) report that using tree size and vigor as overriding criteria for crop-tree selection in thinning operations would have resulted in 20 per cent greater stand volume growth, compared with prescriptions that leave trees on a strict spacing. Varying spacing will also likely aid the development of diverse understorey vegetation (Ares *et al.*, 2009), especially if prescriptions include larger gaps (Fahey and Puettmann, 2008; Davis and Puettmann, 2009). Similarly, natural regeneration of tree species, including tree architecture and growth, will reflect harvesting disturbance and resource levels in stands of variable density (Canham, 1988; Beckage *et al.*, 2000; Shatford *et al.*, 2009). Furthermore, variable density thinning can also promote trees of varying sizes, including larger open-grown trees or trees at edges of gaps that develop deeply furrowed bark (Sheridan *et al.*, 2013), long crowns and large branches (Roberts and Harrington, 2008; Dodson *et al.*, 2012). These crown and branch characteristics have been associated positively with wildlife habitat (e.g. Hayward and Escano, 1989), but also lead to concerns about wood quality and values (Johansson, 1992; Macdonald and Hubert, 2002). If the intention for such trees is to accelerate the development of high heterogeneity in stand structures in the long-term, selecting and releasing trees with 'damage', such as forks or top breakage, may be most efficient (Ishii and McDowell, 2002). In addition to their lower economic value, leaving these trees in spots that are difficult to access, such as riparian areas, deep draws or rocky ridgetops, will further reduce the economic costs of such choices.

Harvesting larger openings (e.g. ~0.2–0.5+ ha) as part of variable density thinning prescriptions have ecological and economic implications. For example, larger openings may provide growing conditions for early seral vegetation not found in understorey conditions or in smaller gaps (Fahey and Puettmann, 2008), which also have a higher likelihood of flowering and fruit production (Wender *et al.*, 2004). Creating gaps will make thinnings more financially attractive, especially if logging efficiency is taken into account during gap layout. The additional cost of gap layout is typically less than the economic gain (Kellogg, 1996; Kellogg *et al.*, 1998), as gap creation also implies harvesting dominant and co-dominant trees, which are typically left behind in low-thinning operations. Especially in early entries, including gaps as part of a variable density thinnings can make an otherwise pre-commercial entry profitable and help achieve ecological goals at the same time.

### Advanced regeneration

Naturally established advance regeneration often occurs following partial canopy openings, as found after thinning, windthrow or ice damage (Bailey and Tappeiner, 1998; Shatford *et al.*, 2009; Dodson *et al.*, in press). In these instances, regeneration establishment has no costs and likely is spatially variable (Dodson *et al.*, in press). Thus, advanced regeneration can provide a low-cost way to contribute to the species diversity (Kuehne and Puettmann, 2008) and spatial variability of the current stand (Dodson *et al.*, in press). It may also provide future options for managing the current stand as a mixed-species stand. Finally, it may facilitate the development of multiple canopy layers typically found in late successional conditions (Dodson *et al.*, 2012), or it can be 'banked' for reforestation after final harvesting (Smith, 1992).

The added cost of evaluating advanced regeneration and possibly altering harvesting operations to protect it (Newton and Cole, 2006) can be weighed against cost savings in site preparation and planting across a variety of time and spatial scales (similar to the procedure described above for replanting). Naturally established conifer seedlings are often capable of growing as well as planted seedlings after they are released by clear cutting (Smith, 1992; Tesch and Korpela, 1993; Tappeiner *et al.*, 2007; Shatford *et al.*, 2009). In intensively managed forests, operations typically remove even viable advanced regeneration during site preparation in order to achieve a single-species plantation with uniform spacing and tree size. Again, on a stand scale, this may be justified; however, saving advanced regeneration in portions of stands with viable regeneration of crop-tree species could lower regeneration costs, especially if the planted seedlings are younger and smaller than the advanced regeneration (see the section about the economics of replanting). An additional economic advantage of relying on advanced regeneration may be that these seedlings have grown above competing vegetation and no longer need to be protected from browsing (Saunders and Puettmann, 1999). At the same time, avoiding site-preparation treatments minimizes damage to shrubs and other understorey vegetation that could enhance habitat quality (Ellis and Betts, 2011) and visual attractiveness (Ribe, 1989) of recently harvested sites.

### Reproduction cutting

Reforestation of even-aged stands may also provide opportunities for increasing species diversity and spatial variability in stand structures. This can be accomplished by strategic decisions about which trees to leave or cut in reproduction cutting operations. The choice of the remaining density may range from the extremes of zero (or the minimal number as determined by forest practice rules) to areas with no harvesting (Franklin *et al.*, 1997). More important for increasing species diversity and spatial variability is that the choice of residual tree density is not limited to a single number or proportion, but varies within and among stands to accommodate local ecological and economic conditions (Franklin *et al.*, 2007; Puettmann *et al.*, 2009; Gustafsson *et al.*, 2012). Retaining trees at harvest can enhance aesthetics in the short-term, provide opportunities for the regeneration of multiple tree species in stands, and provide habitat or 'life-boating from current stand to future forest conditions' for a variety of species (Rosenvald and Lohmus, 2008). Besides selecting characteristics directly related to achieving visual quality (Ribe, 1989) or wildlife habitat objectives (e.g. the presence of lichens and mosses, deeply furrowed bark and epicormic or 'platform' branches) (Root *et al.*, 2010), the selection of residual trees can also incorporate economic decision criteria. Specifically, the characteristics of retained trees that indicate lower economic values should be of interest, including broken and multiple top trees; long, wide crowns with large branches; and stem cavities and indicators of rot. Similarly, trees damaged during harvesting operations may provide a low-cost opportunity for the creation of snags (Lewis, 1998). Trees with such characteristics or damage not only have low (if any) economic wood value, they can be a safety risk; accordingly, the costs of felling them can be fairly high. The cost of leaving such trees may further be reduced or even turned into savings, if the trees are located in areas that are difficult and costly to log (e.g. on steep slopes), or where they do not interfere with efficient logging and reforestation operations (e.g. in groups along riparian zones or fens). Integrating such factors into the management of young stands allows selected trees to develop desired habitat characteristics, while minimizing economic losses or even save management expenses; and these trees may become increasingly important habitat features in the current and succeeding stands.

### *Choices of tree species spacing or densities*

Here, we provide an example of how other dimensions (which can be viewed as organizational scales), such as the certainty of assumptions and predictions, may influence the results of treatment assessments, and thus highlight opportunities to increase variability in stands and landscapes. These opportunities are driven by the fact that silviculturists cannot ignore the influences of social aspects, policies and regulations. For example, choices of planting density are often based on assumptions about follow-up treatment regimes. Currently, plantations are often started at relatively high densities, with the intention of PCT after about a decade. However, because of the lack of funds or personnel, high costs or other priorities, PCT operations may not always be implemented, leading to concerns about tree health (Larsson *et al.*, 1993) and fire hazards (Graham *et al.*, 1999) in untreated, dense stands. These aspects are especially noteworthy because less dense plantations are cheaper to establish, grow larger trees more quickly, and retain a variety of shrubs and early successional species longer during stand development (Puettmann and Berger, 2006). On the other side, some of the cost savings when using wide initial spacing may be offset (or more than offset) by the larger branches and thus lower quality wood at time of harvest (Johansson, 1992). However, starting with lower density plantings will allow for future management flexibility (Wilson and Baker, 2001) and better allow foresters to take advantage of the variation in species, size and spatial patterns found in advanced regeneration (Smith, 1992). Switching assessment scales in this context means acknowledging the uncertainty of future conditions and using caution accordingly in management decisions. New tools, such as scenario analysis, explicitly acknowledge uncertainty and bring different, alternative future conditions into the planning process (Palma *et al.*, 2010).

### *Natural disturbances as diversifying agent*

Many natural disturbances, such as windstorms and fires introduce variability into forests at various spatial scales. Specific impacts of such disturbances will vary from small scales, such as tree morphology, micro-topography and soil conditions, to larger scales, such as distance to oceans, exposure to the main wind directions and elevation. The interactions of these scales will influence the frequency and intensity of natural disturbances such as windstorms (Mitchell, 2012) and fires (Kitchen, 2012). The variability within larger fires especially has received recent attention (e.g. Bergeron *et al.*, 2001). Within perimeters of large fires, areas with low-intensity burns may include patches of relatively undamaged trees of various sizes and ages. If left intact, these trees may make up the future stand or provide seed sources to help reforest the burned area and thus will contribute to the structural heterogeneity of future forests. Our observations suggest that often this variability has been ignored and that management decisions, such as salvage or replanting operations, have been based on stand-level assessments of overall mortality patterns and associated harvesting opportunities. An assessment at smaller spatial scale would highlight opportunities to benefit from the variability created by the disturbance to increase spatial variability in future stands.

Alternatively, natural disturbances, such as fires, can be viewed as setting the stage for heterogeneity and, in conjunction with post-fire rehabilitation, can act as a stimulator to provide opportunities for ensuring heterogeneity in the long-term (Bergeron *et al.*, 2003). On sites with dense shrub or hardwood cover that originated post-fire from buried seed banks or from sprouting, planting may result in stands of mixed species (Conard and Radosevich, 1982). For example, without planting, a single-species white fir (*Abies concolor*) stand would likely develop very slowly under a manzanita (*Arctostaphylos patula*) shrub cover in the California Sierra Nevada and would result in dense stands with little diversity. Small-scale variability may be further enhanced by leaving dead and damaged trees strategically in areas where they would not provide sources of ignition or help the spread of future fires. Larger scale opportunities to increase heterogeneity at relatively low costs include sites with difficult, expensive access. Such sites could be targeted to be left untreated in large-scale salvage or replanting operations.

## Conclusion

Our critical review of the silviculture and ecology literature led us to hypothesize that the use of gradients of spatial and time frames can have a great influence on the results when evaluating costs and ecological impacts of silvicultural practices. Based on this hypothesis we discussed options and highlighted examples of modifying prescriptions highlight methodologies (such as utilizing

Figure 1) that can help target opportunities to increase the species diversity and spatial and temporal variability in forested landscapes, while minimizing costs of such modifications or under selected set of conditions may even lead to cost savings. Our hypothesis also suggests that the past emphasis on homogenous vegetation conditions, i.e. on stand scales and rotation lengths, has limited or biased our choices of silvicultural treatments (see also O'Hara and Nagel, 2013). In contrast, evaluation procedures that address multiple spatial and time scales make opportunities to increase species diversity and spatial variability in forests more obvious. At the same time, these procedures allow assessing economic impacts of such opportunities, thus facilitating the search for lower cost or even win–win situations. Formal testing of our hypothesis is necessary to quantify actual costs and ecological benefits (i.e. to put units and specific axis labels on Figures 2–4) in a variety of ecosystems and ownership settings. In the meantime, a general appreciation of the value of different spatial and time scales for assessment of management practices will help forest managers to address a wider suite of ownership goals. Although these procedures require a change in both viewpoint and work procedures, new technologies such as GPS (Ribe, 1989; Wing and Eklund, 2008; Wing *et al*., 2008), GIS (Koehl *et al*., 2006) and LIDAR (Andersen *et al*., 2005) facilitate planning at multiple spatial and time scales. These tools will help ease the additional workload required to inventory and manage variability at multiple time and spatial scales.

## References


Andersen, H.-E., McGaughey, R.J. and Reutebuch, S.E. 2005 Estimating forest canopy fuel parameters using LIDAR data. *Remote Sens. Environ.* **94**, 441–449.

Ares, A., Berryman, S.D. and Puettmann, K.J. 2009 Understory vegetation response to thinning disturbance of varying complexity in coniferous stands. *Appl. Veg. Sci.* **12**, 472–487.

Arponen, A., Heikkinen, R.K., Thomas, C.D. and Moilanen, A. 2005 The value of biodiversity in reserve selection: representation, species weighting, and benefit functions. *Conserv. Biol.* **19**, 2009–2014.

Bailey, J.D. and Tappeiner, J.C. 1998 Effects of thinning on structural development in 40- to 100-year-old Douglas-fir stands in western Oregon. *For. Ecol. Manage.* **108**, 99–113.

Bauhus, J., Puettmann, K.J. and Kuehne, C. 2013 Is Close-to-Nature Forest Management in Europe compatible with managing forests as complex adaptive forest ecosystems? In *Managing Forests as Complex Adaptive Systems: Building Resilience to the Challenge of Global Change*. Messier, C., Puettmann, K.J. and Coates, K.D. (eds). Earthscan.

Beckage, B., Clark, J.S., Clinton, B.D. and Haines, B.L. 2000 A long-term study of tree seedling recruitment in southern Appalachian forests: the effects of canopy gaps and shrub understories. *Can. J. For. Res.* **30**, 1617–1631.

Bergeron, S., Gauthier, V., Kafka, P. and Lefort, D. 2001 Natural fire frequency for the eastern Canadian boreal forest: consequences for sustainable forestry. *Can. J. For. Res.* **31**, 384–391.

Bergeron, Y., Gauthier, S., Nguyen, T., Leduc, A., Drapeau, P. and Grondin, P. 2003 Developing forest management strategies based on fire regimes in northwestern Quebec, Canada. Project Reports 2003/2004, Sustainable Forest Management Network.

Brown, A. 2012 Biodiversity: rarity value. *Nat. Clim. Change* **2**, 230–230.

Bunnell, F. 2013 Sustaining cavity-using species: patterns of cavity use and implications to forest management. *ISRN Forestry*, vol. 2013, Article ID 457698, 33 p. doi:10.1155/2013/457698.

Canham, C.D. 1988 Growth and canopy architecture of shade-tolerant trees – response to canopy gaps. *Ecology* **69**, 786–795.

Chmura, D.J., Anderson, P.D., Howe, G.T., Harrington, C.A., Halofsky, J.E., Peterson, D.L., Shaw, D.C. and St. Clair, B.J. 2011 Forest responses to climate change in the northwestern United States: Ecophysiological foundations for adaptive management. *For. Ecol. Manage.* **261**, 1121–1142.

Cobb, A.R., Nadkarni, N.M., Ramsey, G.A. and Svoboda, A.J. 2001 Recolonization of bigleaf maple branches by epiphytic bryophytes following experimental disturbance. *Can. J. Bot.* **79**, 1–8.

Conard, S.G. and Radosevich, S.A. 1982 Post-fire succession in white fir *Abies concolor* vegetation in the northern Sierra. *Madrono* **29**, 42–56.

D'Amato, A.W. and Puettmann, K.J. 2004 The relative dominance hypothesis explains interaction dynamics in mixed species *Alnus rubra/Pseudotsuga menziesii* stands. *J. Ecol.* **92**, 450–463.

Davis, L.R. and Puettmann, K.J. 2009 Initial response of understory vegetation to three alternative thinning treatments. *J. Sustainable For.* **28**, 904–934.

DeBell, D.S. and Harrington, C.A. 2002 Density and rectangularity of planting influence 20-year growth and development of red alder. *Can. J. For. Res.* **32**, 1244–1253.

Dodson, E.K., Ares, A. and Puettmann, K.J. 2012 Early responses to thinning treatments designed to accelerate late successional forest structure in young coniferous stands of western Oregon, USA. *Can. J. For. Res.* **42**, 345–355.

Dodson, E., Burton, J. and Puettmann, K.J.. Local over- and understory conditions affect natural regeneration dynamics after partial harvests in Douglas-fir forests, Oregon, USA. *For. Sci.* (in press).

Duryea, M.L. and Dougherty, P.M. (eds). 1991 *Forest Regeneration Manual*. Kluwer Academic Publishers.

Ellis, T.M. and Betts, M.G. 2011 Bird abundance and diversity across a hardwood gradient within early seral plantation forest. *For. Ecol. Manage.* **261**, 1372–1381.

Fahey, R.T. and Puettmann, K.J. 2007 Ground-layer disturbance and initial conditions influence gap partitioning of understory vegetation. *J. Ecol.* **95**, 1098–1109.

Fahey, R.T. and Puettmann, K.J. 2008 Patterns in spatial extent of gap influence on understory plant communities. *For. Ecol. Manage.* **255**, 2801–2810.

Fall, A., Fortin, M.-J., Kneeshaw, D.D., Yamasaki, S.H., Messier, C., Bouthillier, L. and Smyth, C. 2004 Consequences of various landscape-scale ecosystem management strategies and fire cycles on age-class structure and harvest in boreal forests. *Can. J. For. Res.* **34**, 310–322.

Franklin, J.F., Berg, D.R., Thornburgh, D.A. and Tappeiner, J.C. II. 1997 Alternative silvicultural approaches to timber harvesting: variable retention harvest systems. In *Creating a Forestry for the 21st Century*. Franklin, J.F. and Kohm, K. (eds). Island Press, 111–139 pp.

Franklin, J.F., Mitchell, R.J. and Palik, B.J. 2007 Natural Disturbance and Stand Development Principles for Ecological Forestry. USDA Forest Service. Northern Research Station. General Technical Report NRS-19.

Fried, J.S., Boyle, J.R., Tappeiner, J.C. II and Cromack, K. Jr. 1989 Effects of bigleaf maple on soils in Douglas-fir forests. *Can. J. For. Res.* **20**, 259–266.

Getzin, S., Dean, C., He, F., Trofymow, J.A., Wiegand, K. and Wiegand, T. 2006 Spatial patterns and competition of tree species in a Douglas-fir chronosequence on Vancouver Island. *Ecography* **29**, 671–682.

Graham, R.T., Harvey, A.E., Jain, T.B. and Tonn, J.R. 1999 *The Effects of Thinning and Similar Stand Treatments on Fire Behavior in Western Forests*. USDA Forest Service, Pacific Northwest Research Station. PNW-GTR-463.

Gunderson, L. and Holling, C.S. (eds). 2002 *Panarchy: Understanding Transformations in Human and Natural Systems*. Island Press.

Gustafsson, L., Baker, S.C., Bauhus, J., Beese, W.J., Brodie, A., Kouki, J., Lindenmayer, D.B., Lõhmus, A., Pastur, G.M., Messier, C., Neyland, M., Palik, B., Sverdrup-Thygeson, A., Volney, W.J.A., Wayne, A. and Franklin, J.F. 2012 Retention forestry to maintain multifunctional forests: a world perspective. *BioScience* **62**, 633–645.

Hanson, J.J. and Lorimer, C.G. 2007 Forest structure and light regimes following moderate wind storms: implications for multi-cohort management. *Ecol. Appl.* **17**, 1325–1340.

Harrington, T.B. and Tappeiner, J.C.I. 1997 Growth responses of young Douglas-fir and tanoak 11 years after various levels of hardwood removal and understory suppression in southwestern Oregon, U.S.A. *For. Ecol. Manage.* **96**, 1–11.

Harrington, T.B. and Tappeiner, J.C. 2009 Long-term effects of tanoak competition on Douglas-fir stand development. *Can. J. For. Res.* **39**, 765–776.

Hayward, G.D. and Escano, R.E. 1989 Goshawk nest-site characteristics in western Montana and northern Idaho. *Condor* **91**, 476–479.

Hill, J.K. and Hamer, K.C. 2004 Determining impacts of habitat modification on diversity of tropical forest fauna: the importance of spatial scale. *J. Appl. Ecol.* **41**, 744–754.

Hobbs, S.D., Tesch, S.D., Owston, P.W., Stewart, R.E., Tappeiner, J.C. II and Wells, G.E. (eds). 1992 *Reforestation practices in southwestern Oregon and northern California*. Oregon State University Press.

Hughes, T.F., Latt, C.R., Tappeiner, J.C. II and Newton, M. 1987 Biomass and leaf-area estimates for varnishleaf *Ceanothus*, deerbrush, and whiteleaf manzanita. *West. J. Appl. For.* **2**, 124–128.

Humphrey, J. 2005 Benefits to biodiversity from developing old-growth conditions in British upland spruce plantations: a review and recommendations. *Forestry* **78**, 33–53.

Humphries, C.J., Williams, P.H. and Vane-Wright, R.I. 1995 Measuring biodiversity value for conservation. *Ann. Rev. Ecol. Syst.* **26**, 93–111.

Huuskonen, S. and Hynynen, J. 2006 Timing and intensity of precommercial thinning and their effects on the first commercial thinning in Scotch pine stands. *Silva Fenn.* **40**, 645–662.

Ishii, H. and McDowell, N. 2002 Age-related development of crown structure in coastal Douglas-fir trees. *For. Ecol. Manage.* **169**, 257–270.

Johansson, K. 1992 Effects of initial spacing on the stem and branch properties and graded quality of *Picea abies* (L.) Karst. *Scand. J. For. Res.* **7**, 503–514.

Kellogg, L.D. 1996 A comparison of logging planning, felling, and skyline yarding costs between clearcutting and five group-selection harvesting methods. *West. J. Appl. For.* **11**, 90–96.

Kellogg, L.D., Milota, G.V. and Stringham, B. 1998 *Logging planning and layout costs for thinning: Experience from the Willamette Young Stand Project*. Oregon State University Forest Research Laboratory Research Contribution 20 p.

Kitchen, S.G. 2012 Historical fire regime and forest variability on two eastern Great Basin fire-sheds (USA). *For. Ecol. Manage.* **285**, 53–66.

Koehl, M., Magnussen, S.S. and Marchetti, M. 2006 Sampling Methods, Remote Sensing and GIS Multiresource Forest Inventory. Springer.

Koepke, D.F., Kolb, T.E. and Adams, H.D. 2010 Variation in woody plant mortality and dieback from severe drought among soils, plant groups, and species within a northern Arizona ecotone. *Oecologia* **163**, 1079–1090.

Kuehne, C. and Puettmann, K.J. 2008 Natural regeneration in thinned Douglas-fir stands in western Oregon. *J. Sustainable For.* **27**, 246–274.

Larsson, S., Oren, R., Waring, R.H. and Barrett, J.W. 1993 Attacks of mountain pine beetle as related to tree vigor of ponderosa pine. *For. Sci.* **29**, 395–402.

Lewis, J.C. 1998 Creating snags and wildlife trees in commercial forest landscapes. *West. J. Appl. For.* **13**, 97–101.

Macdonald, E. and Hubert, J. 2002 A review of the effects of silviculture on timber quality of Sitka spruce. *Forestry* **75**, 107–138.

Marshall, D.D. and Curtis, R.O. 2002 Levels-of-growing-stock cooperative study in Douglas-fir: Report No. 15-Hoskins: 1963–1998. USDA Forest Service. Pacific Northwest Research Station. Research Paper PNW-RP-537.

Messier, C., Puettmann, K. and Coates, D. 2013 *Managing Forests as Complex Adaptive Systems: Building Resilience to the Challenge of Global Change*. Routledge, The Earthscan Forest Library, 353 p.

Mitchell, S.J. 2012 Wind as a natural disturbance agent in forests: a synthesis. *Forestry* **86**, 147–157.

Muir, P.S., Mattingly, R., Tappeiner, J., Bailey, J., Elliot, W., Hagar, J., Miller, J., Peterson, E. and Starkey, E. 2002 Managing for Biodiversity in young Douglas-fir Forests in Western Oregon. Biological Science Report USGS/BRD/BSR-2002-0006.

Neal, F. and Borrecco, J. 1981 Distribution and relationship of mountain beaver to openings in sapling stands. *Northwest Sci.* **55**, 79–86.

Newton, M. and Cole, E.C. 2006 Harvesting impacts on understory regeneration in two-storied Douglas-fir stands. *West. J. Appl. For.* **21**, 14–18.

Nyland, R.D. 2002 *Silviculture: Concepts and Applications*. McGraw-Hill.

O'Hara, K.L. and Nagel, L.M. 2013 The stand: revisiting a central concept in forestry. *J. For.* **111**, 335–340.

O'Hara, K.L. and Ramage, B.S. 2013 Silviculture in an uncertain world: utilizing multi-aged management systems to integrate disturbance. *Forestry* **86**, 401–410.

Olson, M.G., Wagner, R.G. and Brissette, J.C. 2011 Forty years of spruce–fir stand development following herbicide application and precommercial thinning in central Maine, USA. *Can. J. For. Res.* **42**, 1–11.

Palma, C., Bull, G., Goodison, A. and Northway, S. 2010 Scenario Analysis: The traditional and emerging Canadian forest industry. Forest Resource Management Working Paper. 32 p.

Paquette, A. and Messier, C. 2009 The role of plantations in managing the world's forests in the Anthropocene. *Front. Ecol. Environ.* **8**, 27–34.

Peters, D.P.C., Bestelmeyer, B.T. and Turner, M.G. 2007 Cross-scale interactions and changing pattern-process relationships: consequences for system dynamics. *Ecosystems* **10**, 790–796.

Poage, N.J. and Tappeiner, J.C. 2002 Long-term patterns of diameter and basal area growth in old-growth Douglas-fir trees in western Oregon. *Can. J. For. Res.* **39**, 1518–1530.

Pojar, J., Diaz, N., Steventon, D., Apostol, D. and Mellen, K. 1994 Biodiversity planning and forest management at the landscape scale. In *Expanding Horizons of Forest Ecosystem Management, Proceedings of the Third Habitat Futures Workshop*. Gen. Tech. Rep. PNW-GTR-336, Department of Agriculture, Forest Service, Pacific Northwest Research Station, pp. 55–70.

Puettmann, K.J. 2011 Silvicultural challenges and options in the context of global change: simple fixes and opportunities for new management approaches. *J. For.* **109**, 321–331.

Puettmann, K.J. and Berger, C.A. 2006 Development of tree and understory vegetation in young Douglas-fir plantations in western Oregon. *West. J. Appl. For.* **21**, 94–101.

Puettmann, K.J., Coates, K.D. and Messier, C. 2009 *A Critique of Silviculture: Managing for Complexity*. Island Press.

Raffel, T.R., Smith, N., Cortright, C. and Gatz, A.J. 2009 Central place foraging by beavers (*Castor canadensis*) in a complex lake habitat. *The Am. Midl. Nat.* **162**, 62–73.

Ribe, R.G. 1989 The aesthetics of forestry: what has empirical preference research taught us? *Environ Manage.* **13**, 55–74.

Roberts, S.D. and Harrington, C.A. 2008 Individual tree growth response to variable-density thinning in coastal Pacific Northwest forests. *For. Ecol. Manage.* **255**, 2771–1781.

Root, H.T., McCune, B. and Neitlich, P. 2010 Lichen habitat may be enhanced by thinning treatments in young *Tsuga heterophylla-Pseudotsuga menziesii* forests. *Bryologist* **113**, 292–307.

Rosenvald, R. and Lohmus, A. 2008 For what, when, and where is green-tree retention better than clear-cutting? A review of the biodiversity aspects. *For. Ecol. Manage.* **255**, 1–15.

Saunders, M.R. and Puettmann, K.J. 1999 Use of vegetational characteristics and browsing patterns to predict deer damage in eastern white pine (*Pinus strobus*) plantations. *North. J. Appl. For.* **16**, 96–102.

Schütz, J.P., Pukkala, T., Donoso, P.J. and von Gadow, K. 2012 Historical emergence and current application of CCF. In *Continuous Cover Forestry*. Pukkala, T. and von Gadow, K. (eds). Springer, 1–28 pp.

Sensenig, T., Bailey, J.D. and Tappeiner, J.C. 2013 Stand development, fire and growth of old-growth and young forests in southwestern Oregon. *For. Ecol. Manage.* **291**, 96–109.

Shatford, J.P.A., Bailey, J.D. and Tappeiner, J.C. 2009 Understory tree development with repeated stand density treatments in coastal Douglas-fir forests of Oregon. *West. J. Appl. For.* **24**, 11–16.

Sheridan, C.D., Puettmann, K.J., Huso, M.M.P., Hagar, J.C. and Falk, K.R. 2013 Management, Morphological, and Environmental Factors Influencing Douglas-Fir Bark Furrows in the Oregon Coast Range. *West J. Appl. For.* **28**, 97–106.

Smith, D. 1992 Ideas about mixed stands. In *The Ecology and Silviculture of Mixed-Species Forests*. Kelty, M.J., Larson, B.C. and Oliver, C.D. (eds). Kluwer Academic Publishers , 281–287 pp.

Tappeiner, J.C. II, Maguire, D.A. and Harrington, T.B. 2007 *Silviculture and Ecology of Western U.S. Forests*. Oregon State University Press, 440 p.

Tesch, S.D. and Korpela, E.J. 1993 Douglas-fir and white fir advance regeneration for renewal of mixed-conifer forests. *Can. J. For. Res.* **23**, 1427–1437.

Wagner, R.G. 2000 Competition and critical-period thresholds for vegetation management decisions in young conifer stands. *For. Chron.* **76**, 961–968.

Wagner, R.G., Little, K.M., Richardson, B. and McNabb, K. 2006 The role of vegetation management for enhancing productivity of the world's forests. *Forestry* **79**, 57–79.

Weisberg, P.J. and Bugmann, H. 2003 Forest dynamics and ungulate herbivory: from leaf to landscape. *For. Ecol. Manage.* **181**, 1–12.

Wender, B.W., Harrington, C.A. and Tappeiner, J.C. 2004 Flower and fruit production of understory shrubs in western Washington and Oregon. *Northwest Sci.* **78**, 17.

Wiens, J.A. 1989 Spatial scaling in ecology. *Funct. Ecol.* **3**, 385–397.

Wilson, J.S. and Baker, P.J. 2001 Flexibility in forest management: managing uncertainty in Douglas-fir forests of the Pacific Northwest. *For. Ecol. Manage.* **145**, 219–227.

Wing, M.G. and Eklund, A. 2008 Vertical measurement accuracy of mapping-grade global positioning systems receivers in three forest settings. *West. J. Appl. For.* **23**, 83–88.

Wing, M.G., Eklund, A., John, S. and Richard, K. 2008 Horizontal measurement performance of five mapping-grade Global Positioning System receiver configurations in several forested settings. *West. J. Appl. For.* **23**, 166–171.