Negative or positive effects of plantation and intensive forestry on biodiversity: A matter of scale and perspective

by Henrik Hartmann[1], Gaëtan Daoust[2], Brigitte Bigué[3] and Christian Messier[4]

ABSTRACT

Terrestrial biodiversity is closely linked to forest ecosystems but anthropogenic reductions in forest cover and changes in forest structure and composition jeopardize their biodiversity. Several forest species are threatened because of reduced habitat quality and fragmentation or even habitat loss as a result of forest management activities. In response to this threat, integrated forest management (IFM) was developed in the early 1990s and has been applied over large spatial scales ever since. While IFM seeks to satisfy both human resource demands and ecosystem integrity, the whole forest matrix is affected and this may also have negative impacts on biodiversity. The concept of forest zoning (e.g., Triad) avoids these issues by physically separating land uses from each other. The zoning approach has been developed in the same period as IFM, but there are still very few examples of large-scale applications. This may be because its distinctiveness from IFM may not always seem clear and because forest zoning is not easily implemented. Here we explain these differences and show that IFM and the zoning approach are indeed different management paradigms. We advocate the use of high-yield plantations within the zoning paradigm as a means for biodiversity conservation and review the literature (with an emphasis on the northern hemisphere and on plantation forestry within a land-zoning approach) on impacts of forest management activities on biodiversity. Furthermore, we give advice on issues that require consideration when implementing forest zoning at both the stand and the landscape levels. We recommend several small changes in design and management of forest plantations as a means to significantly increase their biodiversity value. We conclude that while forest zoning seems an adequate strategy for the Canadian forestry sector, a shift in paradigm must carry over to policy-makers and legislation if this approach is to succeed.

Key words: biodiversity, landbase zoning, forest management, intensive silviculture, plantation forests

RÉSUMÉ

La biodiversité terrestre est étroitement liée aux écosystèmes forestiers, mais la réduction du couvert forestier par des activités humaines et des changements dans la structure et la composition forestière constituent une menace pour la biodiversité. Plusieurs espèces forestières sont menacées par la dégradation, la fragmentation ou même la perte totale de leur habitat occasionnées par l'aménagement forestier. Pour y remédier, l'aménagement intégré des ressources forestières (AIRF) a été développé et appliqué à grande échelle depuis le début des années 1990. Quoique l'AIRF vise à la fois à satisfaire la demande de produits et services forestiers et à protéger l'intégrité des écosystèmes forestiers, cette approche engendre une pénétration presque totale de la matrice forestière et pose ainsi des risques pour la biodiversité. Par contre, le zonage forestier (ex. Triade) vise à réduire la pénétration du territoire forestier par une séparation spatiale des objectifs d'aménagement. Quoique développé dans la même période que l'AIRF, l'application du zonage à grande échelle est très récente. Ceci pourrait être dû au fait que le zonage est souvent considéré comme un outil de l'AIRF et non comme une stratégie différente d'aménagement. Aussi, toutes les étapes de la planification du zonage sont complexes et sa mise ne œuvre est ainsi difficile. Dans cette revue de littérature, nous expliquons les différences entre l'AIRF et le zonage et nous montrons que les deux approches sont en fait des paradigmes d'aménagement distincts. Nous promouvons l'utilisation de plantations à haut rendement dans le cadre du zonage comme un moyen de conservation de la biodiversité. Nous revoyons la littérature (en mettant l'emphase sur l'hémisphère nord et sur la foresterie de plantation dans l'approche de zonage) sur l'impact des activités d'aménagement forestier sur la biodiversité. De plus, nous donnons des conseils sur la mise en œuvre du zonage forestier à la fois à l'échelle du peuplement et du paysage ainsi que des recommandations pour le design et la gestion de plantations afin d'augmenter leur valeur pour la biodiversité. Nous concluons que, bien que le zonage forestier semble une stratégie adéquate pour le secteur forestier canadien, un changement de paradigme doit s'établir parmi les décideurs politiques et dans la législation pour que cette approche si prometteuse peut réussir.

Mot-clés : biodiversité, zonage forestier, aménagement forestier, sylviculture intensive, foresterie de plantations

[1]Max–Planck Institute for Biogeochemistry, Hans–Knöll Str. 10, 07745 Jena, Germany. Corresponding author. E–mail: hhart@bgc-jena.mpg.de.
[2]Canadian Forest Service, Laurentian Forestry Centre, 1055 Rue du P.E.P.S. Sainte–Foy (Québec) G1V 4C7. E–mail: Gaetan.Daoust@RNCan–NRCan.gc.ca.
[3]Réseau Ligniculture Québec, Pavillon Charles–Eugène–Marchand, Université Laval, Québec (Québec) G1K 7P4. E–mail: Brigitte.Bigue@sbf.ulaval.ca.
[4]Université du Québec à Montréal, succursale Centre–Ville, CP. 8888, Montréal (Québec) H3C 3P8. E–mail: messier.christian@uqam.ca.

Henrik Hartmann

Gaëtan Daoust

Brigitte Bigué

Christian Messier

Introduction: Forest Management as a Threat to Biodiversity

Terrestrial biodiversity is closely linked to forest ecosystems and forests are considered "essential to economic development and the maintenance of all forms of life" (UNO 1992). The world's forests cover almost 30% of the land surface, comprise more than 4 billion hectares and the majority of terrestrial species depend on or dwell in these forests (Gaston 2000, FAO 2007). However, human activities reduce the natural forest cover by 13 million hectares annually and despite afforestation and reforestation efforts, the world's forest cover was reduced by 3% during the period 1990 to 2005 at an annual rate of 0.2%; hence the world is losing 20 000 hectares of forest every day (FAO 2007).

In Canada the forest cover has been reduced by 77 million hectares since European colonization began but the remaining 402 million hectares of forest and forested land include about 10% of the world's forests and 30% of boreal forests and offer habitats to about two-thirds of an estimated 140 000 species in Canada (Mosquin *et al.* 1995). However, this forest cover continues to shrink and to be modified due to human activities including forestry. In 2007 alone, about 47 700 hectares were deforested in Canada (CFS 2009). The major cause of deforestation in Canada is agriculture and forestry partially counteracts the loss of forest cover with annual afforestation efforts (9 400 hectares in 2005, CFS 2009). However, the resulting net loss of forest cover may have an important impact on biodiversity. In Quebec alone, more than 2000 plant, 95 fish, 19 amphibian, 15 reptile, 223 bird and 63 mammal species live in approximately 50 million hectares of forest (MRNFQ 2008a). About 0.92% of Quebec's commercial forest is harvested annually and more than 50 woodland species have been declared endangered or vulnerable (MRNFQ 2008a).

In the southern parts of Quebec, habitat destruction or degradation from urbanization and agricultural activities are responsible for this threat to biodiversity but in more remote and forested regions, impacts from forest harvest (e.g., habitat fragmentation, changes in stand age, density and composition, snag removal) have been identified as causal links to threats to biodiversity (MRNQ 1996).

In past decades an integrated forest management approach (IFM)—the management of the ensemble of all desirable forest resources at the stand scale—has been advocated in response to these threats. More recently, forest zoning at the regional level has been proposed as yet another potential solution to human threats to forest biodiversity. However, there still is much confusion about what the forest zoning approach actually is, how it differs from IFM and what its advantages are.

This literature review aims at elucidating these points. We critically define the concept of biodiversity and show the differences between IFM and the zoning approach. We then review the literature on impacts of forest management on biodiversity at the stand and the landscape levels. Finally, we provide conceptual advice on implementing intensive and plantation forestry within a zoning approach as a means to conserve biodiversity within specific areas, to maintain it on a larger scale or even restore biodiversity where depleted by former land uses.

What Exactly Is Biodiversity?

Biodiversity is "the variation of life at all levels of biological organization" (Gaston and Spicer 2004) and, as such, a diffuse concept that can be defined for organisms (e.g., individuals), groups of individuals (e.g., species) or groups of species (e.g., functional groups) or even for biophysical components of the environment (e.g., habitats) and their characteristics (e.g., complexity of a forest canopy). Furthermore, biodiversity can be defined at different hierarchical (e.g., organismic, spatial) scales and for diverse attributes such as composition, structure and function. Hence, there is no unique and equivocal definition of what biodiversity actually means.

Measures of biodiversity further complexify the issue. Most intuitively, biodiversity is often measured as richness, i.e., the number of different expressions of an observable attribute (e.g., alleles, species identity) over the extent of interest (e.g., loci, area). However, the number of individuals may not be uniformly distributed across species, and measures of abundance can be used to quantify this distribution. Indices of biodiversity, like the Shannon Index, include both richness and abundance and are more informative (Gaston and Spicer 2004).

However, what one wants to measure depends on specific objectives and constraints. In forest ecology, we often quantify the richness, abundance or diversity indices of species or functional groups (e.g., shrubs, herbs, trees). While there are good reasons for the use of particular species because of their ecological significance in key functions of ecosystems (i.e., species that offer habitat to other species or that are determinants of the trophic chain), the choice of these species is often questionable, comprises a host of pitfalls and may not be useful in all circumstances (Bengtsson 1998; Simberloff 1998, 1999; Bifolchi and Lodé 2005). Hence, all of these measures will only give a partial picture of the biodiversity and this even more so because for some species groups (e.g., soil animals like mites) applying the species concept may be problematic altogether (Bengtsson 1998).

Integrated Forest Management and Forest Zoning – What's the Difference?

Integrated forest management has been developed and applied on large scales since about the early 1990s (Behan 1990, Born and Sonzogni 1995). It stipulates that the "protection of diversity must be incorporated into everything we do every day on every acre, whether preserve or commodity land" (Franklin 1989: 8). Integrated forest management seeks to provide a sustained flow of timber and non-timber forest products by setting up a framework of management priority areas. Protected areas with no industrial resource extraction are to provide essential habitats and wildlife refuge areas. The managed landbase can be subdivided into zones depending on management objectives, i.e., commercial use with conservation of unique features (e.g., habitats), commercial use for timber production and other forest resources and enhanced management zones subjected to the protection of environmental quality. However, integrated forest management affects the whole forest matrix (even though some chunks of forest may be excluded from actual harvest) and therefore environmental impacts from forest operations will also affect the whole forest matrix. For example, it has been recognized that forest management contributes indirectly to environmental damage through its road network via erosion, siltation of watercourses and increases in hunting pressure on wildlife populations (Forman and Alexander 1998).

The forest zoning approach, on the other hand, seeks to avoid disturbances of the whole landbase. The concept of zoning has a long history in urban development (e.g., separating industrial and residential developments) and has also been employed in natural settings for the protection of habitats since the late 1800s (Grove 1992). The forest zoning approach builds on designating permitted uses of land, not only land-use priorities as within IFM, which are based on mapped zones that separate one land use from another. The concept was proposed in the early 1990s as a three-zone approach in Maine, USA (Seymour and Hunter 1992) but it is only recently that examples of large-scale application of this concept have emerged (e.g., Messier *et al.* 2009). Although the zoning approach seems at first glance quite similar to IFM, it is a conceptually different paradigm. Because some of the management objectives (e.g., biodiversity conservation) can only be defined on broader scales that span across priority zones, commercial timber extraction in IFM is often subject to harvest constraints (e.g., minimum harvesting age, harvesting rates, size, spatial and temporal distribution of cutblocks) and therefore *residualized* (i.e., effectively being treated as a non-priority objective) even within enhanced management zones (Sahajananthan *et al.* 1998). The zoning approach avoids *residualizing* timber extraction (or any other land use) because land use objectives are geographically separated.

Economic analysis has shown that the zoning approach is, at the broad level, a more viable strategy for sustainable multiple use of forests than the stand-scale approach of integrated forest management (Vincent and Binkley 1993). By geographically separating competing management objectives, land zoning allows for more intensive production and an increase in cumulative returns for specialized practices on their respective landbases instead of mediocre returns of generalized practices across the entire landbase (Boyland *et al.* 2004). A comparison of several simulated management strategies, including integrated management and several zoning scenarios, showed that forest zoning made it possible to produce more timber and maintain more old growth in the landscape than the other strategies examined (Coté *et al.* 2009).

Forest zoning directly contributes to the conservation of biodiversity by reducing the overall length of the road network associated with forest operations (Sahajananthan *et al.* 1998) and is thought to relieve harvesting pressure on natural forests through increased fibre production on an intensively managed commodity landbase of limited geographical extent. As a consequence, extensive proportions of natural forests could be taken out of the forest management scheme and set aside for conservation purposes (Sedjo and Botkin 1997). The remainder of forest stands would be managed under a less intensive management approach by emulating natural disturbance regimes (Hunter 1999, Bergeron *et al.* 2002) and implementing new approaches based on complexity and resilience (Chapin *et al* 2009, Puettmann *et al.* 2009). In this zone, the road network may be made less permanent or decommissioned after harvest, since the objective is to let the forest reorganize itself following harvest, further reducing the negative impact of permanent roads. Expected gains in productivity from the use of high-yield plantations will make it possible to maintain an acceptable level of environmental constraints, such as riparian or roadside vegetation buffers, not only throughout the entire landbase but also within the intensive management zone (Krcmar *et al.* 2003). Furthermore, intensive (plantation) forestry will be an important element in sustaining international economic competitiveness (Park and Wilson 2007) and may be unavoidable when biodiversity conservation is at stake (Wagner *et al.* 2004).

Are Plantations Biological Deserts?

Forest plantations, especially exotic single-species plantations, are thought to offer a less favourable habitat than natural forests (Hunter 1999, Hartley 2002) and have a reputation for being "biological deserts" (Allen *et al.* 1995, Dyck 1997). However, objectively evaluating the value of plantations for conservation purposes is not trivial and requires assessing their contribution to biodiversity at higher spatial scales (Brockerhoff *et al.* 2008). Hence, one should consider (i) whether plantations reduce harvest pressure on natural forests, (ii) what kind of land use or vegetation they replace, (iii) what other potential alternative land uses there are to be compared with, (iv) whether local species had enough time to colonize and adapt to the new habitat and (v) are plantations managed for production purposes only or with conservation goals in mind (Brockerhoff *et al.* 2008).

However, these issues are rarely considered when plantations are assessed for their biodiversity value and therefore studies from around the world show that single-species plantations are, at the stand scale, often less diverse than natural or semi-natural forests with respect to birds (e.g., Baguette *et al.* 1994, Gjerde and Sætersdal 1997, Twedt *et al.* 1999), arthropods (e.g., Samways *et al.* 1996, Fahy and Gormally 1998, Anderson and Death 2000, Davis *et al.* 2000, Magura *et al.* 2000) or plants (e.g., Fahy and Gormally 1998, Humphrey *et al.* 2002, Aubin *et al* 2008). Nevertheless, it has also been shown that forest plantations can contribute to restoring some of the floristic diversity on abandoned agricultural land (Newmaster *et al.* 2006, Aubin *et al.* 2008) and some plantations may have a surprisingly diverse understory (e.g., Allen *et al.* 1995, Keenan *et al.* 1997, Oberhauser 1997). Furthermore, some plantations can have communities as diverse as natural or secondary forests of birds (e.g., Clout and Gaze 1984, Brockie 1992, Kwok and Corlett 2000), fungi or invertebrate species (e.g., Humphrey *et al.* 1999, 2000, 2002; Ohsawa 2004). Moreover, plantations can have, in the absence of management strategies aiming at eliminating naturally occurring woody understory species, a "catalytic" effect by facilitating the colonization of early and even late successional tree species and other floristic elements from the surrounding forest (Brockerhoff *et al.* 2008).

The influence of plantation type (conifer vs. broadleaf) on biodiversity is ambiguous and results from mechanisms driven by a species' ecological footprint on light, water and nutrient availability, as well as by physical effects and the presence of phytotoxic compounds in the litter (Aubin *et al.* 2008, Barbier *et al.* 2008). In a conifer-dominated landscape, understory plant diversity was greater in pure broadleaf stands than in mixed-species or pure coniferous stands (Oaten and Larsen 2008). However, differences in woody species richness between broad-leaved and conifer stands was dependent on patch size and the greater diversity observed in small patches of broad-leaved forests decreased with patch size and may be due to different management regimes (Estevan *et al.* 2007). Furthermore, some conifer plantations may have greater plant species richness than broad-leaved secondary forests (Nagaike 2002).

Keeping in mind that biodiversity in plantations is dependent on the local natural settings (e.g., proximity, type, and age of neighbouring natural forest stands, see Smith *et al.* 2005), it has been observed that pure stands (i.e., deciduous or coniferous) could support, in some cases, a richer understory vegetation than mixed-species stands and that species richness was generally greater in deciduous stands than in coniferous stands (Barbier *et al.* 2008). Similarly, plant and carabid beetle richness was greater in semi-natural deciduous forests than in conifer plantations (Fahy and Gormally 1998). Understory functional groups and environmental conditions of deciduous plantations converged toward those of old naturally regenerated forests although species diversity remained low and understory structure developed poorly compared with unplanted stands (Aubin *et al.* 2008). Diversity also changed throughout the development cycle of forest stands. In general, species diversity was greater in the early stages and declined during later stages of stand development and early stages host mainly generalist species whereas later mature stands favoured specialist species (Smith *et al.* 2005). During the stand development cycle, understory vegetation composition changed from annual to woody species (Eycott *et al.* 2006) and in conifer stands most vascular plant species disappeared rapidly during stand development; however, bryophyte species could persist in these stands (Smith *et al.* 2005).

The negative impact on wildlife species of replacing natural forests with exotic plantations has been well documented (e.g., Clout and Gaze 1984, Estades and Temple 1999, Lindenmayer *et al.* 1999, Magura *et al.* 2000) although biodiversity tends to increase with plantation age (Smith *et al.* 2005). On the other hand, even exotic single-species plantations can contribute to restoring animal diversity on degraded land (Lugo 1997) and, in heavily modified landscapes, may offer the only available habitat for local species (Pawson *et al.* 2008) or even for endangered or threatened species (Barbaro *et al.* 2008). Moreover, exotic single-species plantations may sometimes represent the very last remaining habitat for locally endemic species (Berndt *et al.* 2008).

When plantations replace other productive land uses, such as agriculture, they are usually considered the "lesser evil" even among environmentalists (Brockerhoff *et al.* 2008). In support of this statement, Stephens and Wagner (2007) showed that when prior land use was considered in the comparisons, plantations may be of greater diversity than the land use they replace. The authors conclude that "increasing biodiversity is a desired outcome that can be achieved through appropriate forest management that includes the use of plantations" (Stephens and Wagner 2007: 312). In light of this information, it can be assumed that the use of sensitive and appropriate forest management practices during planning, execution and maintenance of forest plantation can avoid the creation of biological deserts and may even contribute to the conservation, promotion and restoration of biodiversity.

Impact of Intensive Forest Management on Biodiversity

Stand-scale considerations

Forest management practices have the potential to affect biodiversity at many spatial scales (Wigley and Roberts 1997). At the stand level, silvicultural treatments, such as site preparation, planting, mechanical or chemical vegetation management and thinning, alter stand density and age, and tree species composition as well as soil properties and understory vegetation. The changes in stand properties may then affect plant and animal habitat quality and biodiversity.

In general, silvicultural treatments associated with intensive forest management cause a temporal increase in species diversity and shifts in the relative abundance and species composition of overstory tree, understory herb, and shrub communities (Rowland *et al.* 2005). Although some of the impacts from intensive management practices, such as stand type conversion, stand structural changes and especially age-class truncation, may decrease habitat suitability for several vertebrate wildlife species that prefer old-growth forests, these same changes may also increase habitat availability for species favouring young to mature coniferous-dominated forests (Thompson *et al.* 2003).

Site preparation practices such as scarification, ploughing, tilling, crushing, and slash burning may have negative impacts on species such as salamanders and invertebrates associated with or entirely restricted to the forest floor (deMaynadier and Hunter 1995). The greatest impact of these treatments is on the availability of dead organic matter—cavity trees, snags, coarse woody debris, and the organic horizon of the forest floor (Freedman *et al.* 1993, 1996)—and minimizing intensive site preparation will help maintain coarse woody debris and thus biodiversity (Carey and Johnson 1995). Plant species diversity is not at all or somewhat positively affected by ploughing or tilling (Haeussler *et al.* 2002, 2004), but moderate treatments may be necessary to prevent site colonization by ruderal plants (Soo *et al.* 2009) and other early colonizers, many of which could be invaders.

Although exotic tree species are often preferred for reforestation or afforestation because of their superior yield and tolerance to native forest pests, indigenous plant and animal species are usually better adapted to native tree species and can more readily access their resources (Harrison *et al.* 2000). However, even exotic tree species may have beneficial effects on biodiversity if planted on degraded sites or on specific soil types (Chiarucci 1996, Lugo 1997). Furthermore, several studies have shown that many local plant and animal species can make use of the habitats created by exotic tree species (Barlow *et al.* 2007).

Chemical release treatments were applied in Canada on a large scale in the 1970s and 1980s but increased public resistance has amounted to a complete ban on the use of herbicides in Quebec's forests (Fortier and Messier 2006). However, chemical release treatments cause only—if any—short-term effects on wildlife biodiversity (Lautenschlager 1993, Thompson *et al.* 2003) which may in some cases even be positive (Newton *et al.* 1989). Similarly, herbicide applications usually do not cause changes in understory vegetation diversity and reported changes are ephemeral (Rowland *et al.* 2005). Mechanical treatments are even less prone to cause changes to biodiversity (Fredericksen *et al.* 1991) because they do not reinitiate the successional pathway of the ground vegetation, unlike chemical treatments.

Fertilizer treatments may temporarily increase plant species richness (Fehlen and Picard 1994), change vegetation composition (Nohrstedt 1994) or the abundance of specific plant forms (Olsson and Kellner 2006) but usually have no lasting impact on plant diversity or richness (Demchik and Sharpe 2001) or on the diversity of plant functional groups (Bauhus *et al.* 2001).

Thinning increases light penetration in the forest stand and thereby favours pioneer species (Moore and Allen 1999). Increases in light and nutrient availability from thinning facilitates the development of dense herbaceous understory vegetation (Griffis *et al.* 2001) and improved vertical structure following thinning increases bird diversity and habitat use (Bisson and Stutchbury 2000, King and DeGraaf 2000). However, potential changes in forest floor habitat condition from thinning did not negatively influence the use of sensitive indicator species (salamander) (Messere and Ducey 1998) and thinning may increase the abundance of particular wildlife species (Sullivan and Klenner 2000).

Clearcutting sets forest stands back to an earlier successional stage (Hansen *et al.* 1991, Christensen and Emborg 1996) and although there are some similarities between clearcuts and forest fires, clearcutting can hardly fulfill all the ecological functions of this type of natural disturbance (McRae *et al.* 2001, Bergeron *et al.* 2002). Furthermore, the use of heavy machinery can cause soil compaction, decrease water infiltration and soil aeration and destroy soil structure (McClurkin and Duffy 1973, Xu *et al.* 1999) and soil compaction may cause decreases in plant diversity, specifically of grasses and other herbaceous plants in some instances (Mellin 1995). However, by increasing radiation levels on the forest floor, clearcutting also increases biomass production and diversity of the plant community (Perison *et al.* 1997, Yorks and Dabydeen 1999). Progressive removal of the canopy layer (shelterwood cut) may enhance the diversity of the plant community compared with complete, patch, or no canopy removal (Beese and Bryant 1999). Also, there may be a threshold of cutblock size beyond which clearcutting impacts on biodiversity increase disproportionately to their size (Pawson *et al.* 2008). Clearcutting impacts on wildlife are ambiguous and as with any other form of management decision—including taking no action at all—clearcuts favour some species while disadvantaging others (Bunnell *et al.* 1999, Bayne and Hobson 2000). Furthermore, impacts on wildlife depend on the spatial extent of the silvicultural action in relation to the living space of the studied species (Potvin *et al.* 1999) but in general the loss of vertical structure and removal of snags and coarse woody debris are important factors related to habitat loss of most vertebrate and invertebrate species (Humphrey *et al.* 2003).

Landscape-scale considerations

The landscape is a mosaic of different biophysical elements (e.g., mountain ranges, forests, agricultural fields, human infrastructures, streams and lakes) and the spatial distribution of critical resources within these elements determines the availability of habitats and the distribution of species within the landscape (Wiens *et al.* 1987, Debinski *et al.* 2001). Biodiversity itself is a dynamic concept that is defined at different spatial and temporal scales. Within a landscape, successional pathways alter the composition of animal and plant communities (Kimmins 1997, Bunnell *et al.* 1999) as well as their structure and functional role (Westworth and Telfer 1993). Hence, the role of a species or a functional group (e.g., pioneer species) within the processes of an ecosystem may be important in an early successional phase but declines in later phases (McLaren 1996, Kimmins 1999). The assessment and management of biodiversity must take this dynamic into account (Hansen *et al.* 1995, Gaines *et al.* 1999). However, most studies on the impact of forest management on biodiversity are limited to small spatial (i.e., forest stands or plantations) and temporal scales (often not more than several years) and therefore do not include this temporal dynamic (Bunnell and Huggard 1999, Crawley and Harral 2001). While we recognize the difficulty of obtaining data on larger spatial and temporal scales, we must stress the fact that short-term data may possibly have contributed to the overall criticism of plantation and intensive forestry. As has been shown by Smith *et al.* (2005) and Aubin *et al.* (2008), there was a substantial gain in the biodiversity value during the development cycle of plantations. Hence, long-term data may reveal that plantations are generally of greater value as habitats than commonly thought.

While studies over longer time windows may help improve the assessment of long-term impacts of intensive forest management on biodiversity (Rowland *et al.* 2005), spatial constraints may be harder to surmount. Even within a single stand, the assessment of all taxa is extremely difficult and costly so it seems even more impossible to accurately measure the biodiversity of a whole watershed or even a landscape (Humphrey *et al.* 2003). Hence, any assessment of the biodiversity of either a natural forest or a plantation will necessarily always be incomplete.

At the stand scale, many silviculture interventions result in some increases in biodiversity compared to what was present in the same stand before, and its negative public reputation (Wagner *et al.* 1998) may not always be justified. However, the key issue in conserving and promoting biodiversity with forest management lies within the coordination of silvicultural activities at the landscape scale. In the past, forest managers have defined their silvicultural approach at the stand scale and thereby neglected the spatial and temporal variability of forested landscapes (Kimmins 1995). It is the repetition of similar and simplified stand-scale management practices over a larger landscape that has created the problem with traditional forestry practices as discussed in a recent book by Puettmann *et al.* (2009). In the boreal forest of Scandinavia, the large-scale transformation of forest habitats into homogenous and production-intensive systems, with an extensive use of exotic tree species plantations, has posed a threat to biodiversity and has led to major public concerns about forest management practices (Larsson and Danell 2001). These problems can be avoided when objectives are defined over larger spatial scales as is the case with the forest zoning approach.

Recommendations

At the landscape scale

Forest zoning seems a particularly well-adapted strategy for the Canadian forestry sector. More than 90% of Canadian forests are publicly owned and therefore fulfill a major requirement for successful application of a zoning approach: ownership control over entire landscapes, cooperation between agencies and flexibility in assigning land allocations to any of the zones (Sarr and Puettmann 2008). Proportions of landbase required for each zone are context-specific and depend on overall management objectives. However, estimates for intensive silviculture range from 10% (Sedjo and Botkin 1997) to 30% (Sahajananthan *et al.* 1998). Messier *et al.* (2003a) envision for Canada proportions of 14% for intensive silviculture (including 4% of fibre farms), 74% for ecosystem management and 12% for conservation, whereas a case study covering 1.2 million hectares of forest land in coastal British Columbia allocated proportions of 65% for timber production, 25% for habitat (e.g., long rotations or modified partial harvesting to retain habitat structures such as large snags, large downed wood) and 10% for old-growth (biodiversity values) (Boyland *et al.* 2004). The first large-scale application of the zoning approach was implemented in central Quebec and divides the landscape into conservation areas (11%), intensive forestry areas that includes fast-growing plantations (20%) and an extensively managed areas (69%) (Messier *et al.* 2009).

The challenge for forest managers will be to merge multiple and competing objectives, such as timber production, recreational values, habitat quality and biodiversity, over the landscape (Lindenmayer *et al.* 1999) while protecting the diversity of the whole suite of ecosystems within conservation zones covering only 10% to 14% of the landbase. This requires identifying critical structures and functions, species, communities and ecosystems of the landscape and the conservation of the most important (Lautenschlager 1995), although the criteria for identifying these species, ecosystems and attributes are not well defined (Sample 2003). However, the inverse approach of allocating the richest sites to intensive silviculture first and then establishing conservation sites may exacerbate any effect, whether positive or negative, of intensive silviculture on biodiversity because of the positive relationship between site productivity and animal and plant diversity (Thompson *et al.* 2003). A more appropriate approach would establish a core of conservation sites that are representative of the ecosystem diversity within the landscape and are large enough to sustain landscape-scale processes first before deciding where to put the intensive forestry areas (as has been done in the TRIAD project of central Quebec: Messier *et al.* 2009). Also, areas crucial for the visual quality of the landscape, exceptional wildlife habitat, water resources and other desired values must be identified (Sahajananthan *et al.* 1998). Critical ecosystems such as riparian zones should be protected (Sedjo and Botkin 1997) and conservation zones should comprise high-value conservation forests (HVCF) (Jennings and Jarvie 2003). Within the zone of active forest management, age-class truncating may threaten species adapted to old-growth habitat (Thompson *et al.* 2003) and a diverse configuration of silvicultural units at the landscape level should be maintained to promote biodiversity (Sargent 1992, Wigley and Roberts 1997, Smith *et al.* 2005, Eycott *et al.* 2006).

Considering the potential impact of management practices on the composition and structure of forests, the impact of forest management on biodiversity at the landscape level may be assessed using the spatial distribution and proportion of forest stands under different management regimes (Hansen *et al.* 1991, Freedman *et al.* 1996) because the resulting stand characteristics can serve as indicators of habitat quality at the landscape level (Watts *et al.* 2005). For example, changes in forest edge length, core habitat proportion or isolation of forest patches resulting from silvicultural treatments may cause habitat fragmentation for several species (e.g., Ripple *et al.* 1991, Mladenoff *et al.* 1993, Spies *et al.* 1994, Jules 1998). On the other hand, complex stand structure, the presence of big and old trees, snags and coarse woody debris generally increase the habitat quality of a stand (Humphrey *et al.* 2003, Humphrey 2005). The temporal variability of such indicators can be modeled with long-term simulations of vegetation succession (Perry and Millington 2008) and simulation models can give estimates of habitat availability and quality over entire landscapes (Messier *et al.* 2003b, Marcot 2006). Simulation models can also model direct effects of different forest management strategies on vegetation characteristics and habitat quality (Klenner *et al.* 2000, Shifley *et al.* 2008) and their sustainability can be evaluated with dynamic-landscape metapopulation models (Wintle *et al.* 2005). Furthermore, prioritization tools can provide estimates of threats to biodiversity resulting from alternate management strategies and therefore can contribute to increasing transparency, accountability and efficiency of public investment in biodiversity (Wintle 2008). Algorithms using biodiversity indicators have been developed to facilitate landbase allocation (Boyland *et al.* 2004) and conservation site selection (Walker and Faith 1998). Such tools can be linked to optimization procedures (Haight *et al.* 2000) and assist in finding trade-off solutions between different sets of forest values (e.g., biodiversity vs. fibre production; Faith *et al.* 1996) and in analyzing the feasibility of alternate zoning scenarios (Krcmar *et al.* 2003).

Within the zone of intensive forest management, the use of herbicides could be permitted because increases in fibre production are necessary for allocating a large proportion of production forests to conservation and ecosystem management (Fortier and Messier 2006). With an expected increase in the global demand for wood, the use of herbicides may actually become a crucial part of biodiversity conservation (Wagner *et al.* 2004) although this is only true if forest conservation objectives in exchange for increased fibre production are guaranteed by rigorous laws (Paquette and Messier 2009). Governmental commitments towards forest zoning (e.g., Government of Alberta 1997, MRNFQ 2008b) are a first and important step in initiating the zoning approach but more concrete measures for critically and objectively assessing and verifying management achievements as well as promoting cooperation between different agencies will be necessary to ensure a successful application. The implementation of a zoning approach within a 0.86 million hectare public forest management unit in central Quebec has been proven economically viable and socially acceptable and forest zoning could be a good strategy for general application in the Canadian forest management sector (Messier *et al.* 2009).

At the stand scale

Within stands at the landscape level and in areas where forest zoning does not apply—mostly the southern belt of Canada where urban development, agriculture and private forests dominate—a stand-scale approach must be used for conserving and promoting biodiversity. Because plantations can hardly fulfill all the ecological functions of the natural forest (Christian *et al.* 1998), replacement of natural forests, where they still exist, with plantations should be avoided. Similarly, conversion of secondary or semi-natural forests to plantations should also be avoided (Dyck 1997, Brockerhoff *et al.* 2008) although this may be justified in specific circumstances, i.e., partial conversion of selected sites to high-yield plantations within large forest tracts at close proximity to processing industries. However, afforestation of abandoned agricultural fields could contribute to maintaining or even increasing the biodiversity value of the landscape by adding forest habitat to the agricultural matrix and by mitigating edge effects and increasing connectivity between patches of remnant natural forest (Brockerhoff *et al.* 2008).

Minor adjustments in design and management are often enough to improve the biodiversity value of plantations at the stand level without compromising their productivity (Hartley 2002). However, the mechanisms that cause various species to accept an environment as their habitat are complex and predicting the impact of changes in plantation management on biodiversity is difficult. However, plantation design and management can add structural components and complexity to plantation forests which, in turn, increase their potential to host a larger community of plant and animal species (Allen *et al.* 1996, Moore and Allen 1999). To do so, foresters can apply several of the following measures (e.g., Kimmins 1995, Christensen and Emborg 1996, Bunnell *et al.* 1999, Thompson *et al.* 2003): (i) maintain snags and old trees during partial or clearcuts; (ii) create or maintain coarse woody debris by leaving large bole sections during harvesting or by girdling standing trees; (iii) maintain patches of natural forest in plantations; (iv) create edges of natural, nonlinear shape and maintain a balanced edge–area ratio in plantations; (v) preserve wetland patches and open patches within plantations; and (vi) vary spacing between rows of planted trees. Similarly, Norton (1998) identifies four types of management action that may improve the biodiversity value of plantations: (i) retain indigenous plant and animal communities; (ii) establish a greater diversity of planted species; (iii) plant a diversity of tree species along streams and roads; and (iv) modify silvicultural practices within plantations by increasing, for example, rotation length between thinnings or stand age at harvest.

Increasing ecosystem diversity at the landscape level can be achieved by planting on a rotational basis, i.e., a spatial and temporal mix of species and age classes (Norton 1998). Similarly, early and heavy thinning of some plantations combined with late or no thinning of other plantations will create a mosaic of relatively open areas and dense thickets (Hartley 2002). Indigenous tree species often have characteristics quite similar to exotic species and should be favoured during plantation establishment (Sedjo and Botkin 1997) because several animal species may depend entirely on indigenous species (Hartley 2002). Mixed-species plantations are often more productive than monocultures (Kelty 1992, Potvin and Gotelli 2008) and more resistant to pests (Stiell and Berry 1985) and have ecological, operational and economic benefits even if one species comprises 90% of a site (Hartley 2002). Further benefits anchored within the functional significance of tree species diversity (e.g., facilitation, niche differentiation, redundancy) are being investigated on long-term research sites (Scherer-Lorenzen *et al.* 2007). Either way, tree improvement programs should maximize the genetic diversity of the selected tree species in order to preserve the species' capacity to adapt to changing environmental conditions and to natural disturbances (Lambeth and McCullough 1997, Rajora 1999). During site preparation, severe treatments favouring nutrient leaching and erosion should be avoided (Hartley 2002) as should the scarification, piling and burning of slash and coarse woody debris (Carey and Johnson 1995). Retention of native vegetation within plantations creates "life boats" for animal species and facilitates the recolonization of plant species and can be achieved by conserving patches of native vegetation during harvest operations as well as by leaving untreated "skips" during herbicide applications (Hartley 2002).

Some Last Words

This literature review has shown that forest management disturbs, through its impact on the abiotic and biotic components of forest ecosystems, the established equilibrium between native species and thereby modifies biodiversity. Even though there is an ever-increasing body of literature on biodiversity, there still is a need for further studies on non-vascular plant, fungi, and soil organism responses as well as for long-term studies that widen our understanding of the long-term effects of forest management on biodiversity (Humphrey *et al.* 2003, Rowland *et al.* 2005). However, as Humphrey *et al.* (2003: 111) stated "The need for rapid dissemination of data, interpreted in the form of readily usable management advice, means that researchers are unlikely to be given such opportunities for long-term studies. Instead, there will be an increasing requirement for interim advice, based on available data coupled with ecological wisdom." This literature review aims at providing the former. However, because of the complexity of biodiversity in terms of its attributes, scales and measures, we cannot go beyond general advice. Specific advice requires the definition of biodiversity objectives at a local and regional scale. Here, forest managers can make a substantial contribution by providing the necessary ecological wisdom.

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