

Assessing forest management scenarios on an Aboriginal territory through simulation modeling

by Hugo Asselin^{1,*}, Mario Larouche² and Daniel Kneeshaw²

ABSTRACT

The dominant management strategy in boreal forests—aggregated clearcuts (AC)—faces increased criticism by various stakeholders, including Aboriginal people. Two alternative strategies have been proposed: dispersed clearcuts (DC) and ecosystem-based management (EM). We modelled the long-term and landscape-scale effects of AC, DC, and EM on a set of indicators of sustainable forest management relevant to an Aboriginal community's values: (1) forest age structure; (2) spatial configuration of forest stands; (3) road network density; and, (4) forest habitat loss to clearcuts. EM created a forest age structure closer to what would result from a natural disturbance regime, compared to AC and DC. Cut blocks were more evenly distributed with EM and DC. The road network density was lower and increased slower with EM, thus reducing the potential for conflicts between forest users. Under EM, a higher forest cover was maintained (and thus potential wildlife habitat) than in AC or DC. The EM scenario provided the best outcome based on the four measured indicators, partly because the constraints imposed on the modeling exercise led it to harvest less than the other scenarios. Annual allowable cut should thus be a key factor to consider to ensuring better compliance with Aboriginal criteria of sustainable forest management.

Key words: modeling; landscape; Indigenous people; forest management; social acceptability

RÉSUMÉ

La principale stratégie d'aménagement en forêt boréale—les coupes agglomérées (CA)—est de plus en plus critiquée par plusieurs parties prenantes, incluant les peuples autochtones. Deux stratégies alternatives ont été proposées : les coupes dispersées (CD) et l'aménagement écosystémique (AE). Nous avons modélisé les effets à long terme et à l'échelle du paysage de CA, CD et AE sur un ensemble d'indicateurs d'aménagement forestier durable liés aux valeurs d'une communauté autochtone : (1) la structure d'âge de la forêt; (2) la configuration spatiale des peuplements forestiers; (3) la densité du réseau routier; et, (4) la perte d'habitat forestier due aux coupes totales. AE a créé une structure d'âge plus proche de celle qui résulterait d'un régime naturel de perturbations, comparativement à CA et CD. Les blocs de coupe étaient répartis plus uniformément sur le territoire avec AE et CD. Le réseau routier était moins étendu et se développait moins vite avec AE, réduisant ainsi le potentiel de conflits entre usagers de la forêt. AE a aussi maintenu plus de couvert forestier (et donc plus d'habitat faunique potentiel) que CA ou CD. Le scénario AE a obtenu de meilleurs scores pour les quatre indicateurs mesurés, en partie parce que les contraintes imposées à l'exercice de modélisation ont résulté en moins de coupes que dans les autres scénarios. La possibilité forestière annuelle devrait par conséquent être un facteur clé à considérer pour assurer une meilleure conformité aux critères autochtones d'aménagement forestier durable.

Mots clés : modélisation; paysage; Autochtones; aménagement forestier; acceptabilité sociale

Introduction

Forest planners and managers are increasingly faced with the challenge of reconciling the economic, social and ecological values of multiple stakeholders (Côté and Bouthillier 1999, Ananda and Herath 2003, Robson and Hunt 2010) including Aboriginal people (Castro and Nielsen 2001, Wyatt 2008, Dhital *et al.* 2013). Aboriginal cultures are closely linked to the land and are thus particularly affected by forestry activities (Gladu and Watkinson 2004). Indeed, Aboriginal well-being relies on intricate historical, familial, cultural, and political connections to the land (Adelson 2000) and traditional activities are rooted within a knowledge-practice-belief com-

plex (*sensu* Berkes 2012) centred on ethical and sustainable use of the land and resources. Hence, Aboriginal people's knowledge, needs and views should be taken into account in forest management strategies (Cheveau *et al.* 2008, Parrotta and Trospen 2012, Uprety *et al.* 2012).

Until recently, the dominant management strategy in North American boreal forests has been aggregated clearcuts (Franklin *et al.* 2002). This approach faces increased criticism and is not deemed appropriate to meet all social and environmental needs (Pâquet and Bélanger 1997, Bliss 2000, Germain 2012). Extensive clearcuts have driven ecosystems outside their natural range of variability in terms of age-class

¹ Centre d'étude de la forêt, Chaire de recherche du Canada en foresterie autochtone, Université du Québec en Abitibi-Témiscamingue, 445 boulevard de l'Université, Rouyn-Noranda, QC, J9X 5E4, Canada. *Author for correspondence: Hugo.Asselin@uqat.ca

² Centre d'étude de la forêt, Département des sciences biologiques, Université du Québec à Montréal, C.P. 8888, Succursale Centre-ville, Montréal, QC, H3C 3P8, Canada.

representation, with younger age classes being overrepresented compared to preindustrial figures (Cyr *et al.* 2009). Aggregated clearcuts also cause habitat loss for forest-obligate species (e.g., Smith and Schaefer 2002). Two alternative management strategies have been proposed to replace aggregated clearcuts: dispersed clearcuts (Booth *et al.* 1993, Bélanger 2001), and ecosystem-based management (Seymour and Hunter 1992, Bergeron *et al.* 1999). The dispersed clearcut strategy, in which cut blocks are more evenly distributed across the landscape, would affect Aboriginal family hunting grounds more evenly than the agglomerated clearcut strategy, and could thus be more acceptable. Ecosystem-based management, by emulating spatiotemporal patterns created by natural disturbances to which Aboriginal people have adapted over centuries, could also be judged more acceptable. However, these assumptions have not yet been tested, in large part because dispersed clearcuts and ecosystem-based management have been implemented too recently to generate landscape level patterns to be evaluated by entire communities.

The effects of forest management through time and across large spatial scales (thousands of square kilometres) are difficult to envision. Decision support tools can be used to consider the long-term and large-scale effects of different management strategies on forest characteristics. To help take into account the Aboriginal perspective, frameworks of criteria and indicators of sustainable forest management have been designed in collaboration with Aboriginal communities (e.g., Natcher and Hickey 2002, Karjala *et al.* 2004, Saint-Arnaud *et al.* 2009). However, decision making too often focuses on short spatiotemporal scales (stand scale; one lifetime), whereas Aboriginal people generally consider long-term effects and legacies (Menzies and Butler 2006). Effects of forest management at the landscape scale and over several generations are not commonly addressed by planners during consultation processes, although they could help Aboriginal people make more informed choices. Here, simulation modeling was used to compare the long-term and landscape-scale effects of three forest management scenarios (aggregated clearcuts, dispersed clearcuts, and ecosystem-based management) on a set of sustainable forest management indicators relevant to the values of an Aboriginal community.

Methods

Study area

The study area corresponds to the ancestral territory of the Kitcisakik Algonquin community (Fig. 1). The territory covers approximately 6000 km², of which some 80% is forested, and it is subdivided into 29 family hunting grounds. It is located in the western balsam fir (*Abies balsamea* L.) – yellow birch (*Betula alleghaniensis* Britton.) bioclimatic sub-domain (south) and in the western balsam fir – paper birch (*Betula papyrifera* Marsh.) sub-domain (north) (Saucier *et al.* 1998). Spruce budworm (*Choristoneura fumiferana* Clem.) and forest tent caterpillar (*Malacosoma disstria* Hbn.) outbreaks are the main natural disturbances (Ndione 2014). The mean natural forest age in a broad region encompassing the study area is 150 years (Grenier *et al.* 2005, Ndione 2014), which provides a sound estimate of the historical fire cycle (Bergeron *et al.* 2001). Forest harvesting

dates back to at least the 19th century, mostly consisting of selective cuts until the 1970s, when clearcut logging became the main disturbance shaping the landscape (Ndione 2014) and reducing forest age and connectivity.

The approximately 460 members of the Kitcisakik Algonquin community maintained a semi-nomadic lifestyle until the late 20th century, and their livelihood and culture are still largely based on hunting, trapping, fishing and gathering (Saint-Arnaud *et al.* 2009). The Kitcisakik territory is under governmental jurisdiction and straddles five forest management units. More than 60% of the productive forests have been logged over the last 50 years, largely without taking into account Aboriginal knowledge, needs and views, and without compensation for the lost potential to pursue traditional activities. Community members are consulted before annual management plans are implemented, but they are largely dissatisfied with the process and consider it occurs too late in the planning agenda, and is conducted primarily to provide the government and forest managers with a good conscience (Ndione 2014). This has caused widespread resentment in the community, and has led to conflicts with the forest industry (Saint-Arnaud *et al.* 2009).

Forest management scenarios

Three forest management scenarios were modelled for the Kitcisakik territory using SELES (Spatially Explicit Landscape Event Simulator; Fall and Fall 2001): aggregated clearcuts (AC), dispersed clearcuts (DC) and ecosystem-based management (EM). The Vermillion Landscape Model (VLM) developed in SELES (Fall *et al.* 2004, Didion *et al.* 2007), was modified by Larouche (2008) to create the Kitcisakik Landscape Model (KLM). Detailed parameters of the KLM are provided in Appendix 1 and in Larouche (2008). Data on forest stand characteristics (tree species, stand age, road network, and hydrological network) were retrieved from the database of the third decennial forest inventory of Quebec's Ministry of Forests, Wildlife and Parks and transformed into 0.25 ha per pixel raster layers. The oldest age class combines all stands older than 100 years. Consequently, as suggested by Fall *et al.* (2004) and Didion *et al.* (2007), ages of stands currently > 100 years old (8%) were redistributed randomly between 100 and 300 years to produce a more realistic representation of the age class structure of old forests in the landscape.

The time-step used in the model was five years. This is sufficient to observe patterns occurring over a rotation across the landscape, but would be inappropriate for modeling processes with a rapid turnover rate that typically occur at smaller scales. The VLM includes four sub-models: logging, road building, wildfire, and aging. The original wildfire and aging sub-models used were based on disturbance and forest species parameters specific to the Kitcisakik territory, whereas the logging and road building sub-models were modified. The logging sub-model of the VLM, which only includes two types of logging (aggregated clearcuts and ecosystem-based management) was modified to introduce a dispersed clearcuts scenario (Larouche 2008). In the AC scenario, mature forests (≥ 75 years) were harvested in blocks separated by 50-m bands of residual forest, aiming for an annual harvest rate of 1% of the total forested area. Cut blocks could be aggregated, provided that at least a third of the man-

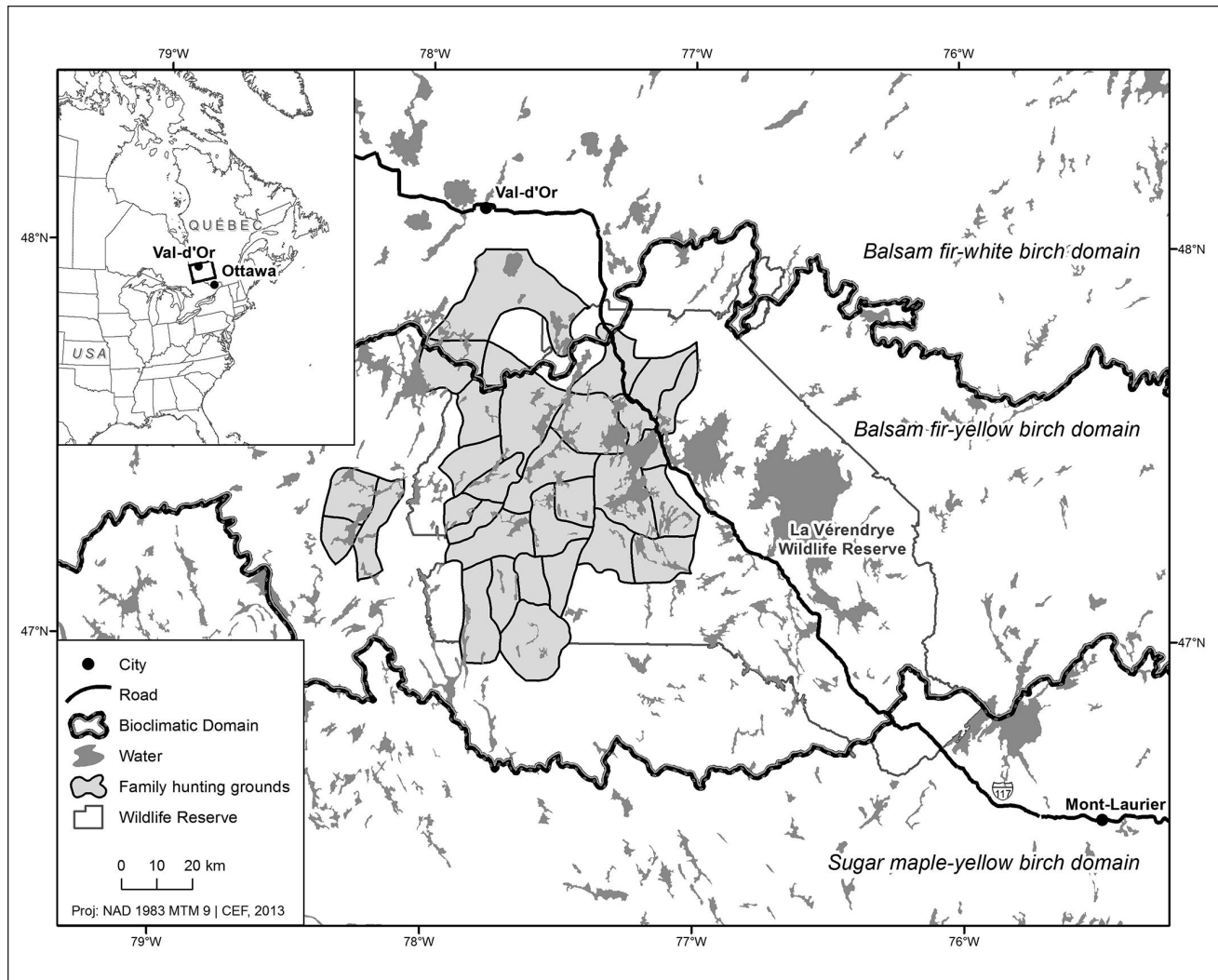


Fig. 1. Location of the study area in western Quebec, corresponding to the ancestral territory of the Kitcisakik Algonquin community, divided into 29 family hunting grounds.

agement sub-units consisted of forests ≥ 7 m high at all times (based on forestry guidelines of the Quebec Ministry of Forests, Wildlife and Parks). The same parameters were used in the DC scenario, but at least half of each management sub-unit had to consist of residual forests (i.e., ≥ 3 m) at all times, in addition to at least a third of each management sub-unit consisting of forests ≥ 7 m high. The ecosystem-based management strategy of the VLM was modified so that instead of lengthening forest rotation to reach a predetermined portion of stands older than 100 years, the KLM ecosystem-based strategy aimed at producing target proportions of three “cohorts” (0–100 years, 100–200 years, and 200–300 years) by varying logging techniques (Bergeron *et al.* 1999). Clearcuts were used to create first cohort stands (i.e., early successional, post-catastrophic disturbance stands). A first type of partial cut (P1) was used to create second cohort stands by harvesting 25% of the basal area of the oldest stems. A second type of partial cut (P2) was used to create third cohort stands by harvesting 25% of the basal area of mature stems, while leaving 25% of the oldest trees to age. From an ecological perspective,

stand-replacing forest fires were simulated by clearcuts, whereas spruce budworm outbreaks which transform the first cohort even-aged stands into second cohort uneven-aged stands were simulated by P1. Gap dynamics characterizing the transition from mature to old-growth stands were simulated by P2.

Cut block area varied from one scenario to another. For AC, 90% of the cut blocks were 70–100 ha in size and 10% were 101–150 ha. For DC, 90% of the cut blocks were 50–70 ha and 10% were 71–150 ha. Cut block area for AC and DC followed the forestry guidelines of the Quebec Ministry of Forests, Wildlife and Parks. For EM, a wider range of cut block areas was used to better emulate the variability of stand size created by natural disturbances (Fall *et al.* 2004): 70% of the cut blocks were 100–150 ha, 25% were 151–300 ha, and 5% were 301–400 ha.

The road building sub-model of the VLM was modified by Larouche (2008) to distinguish primary from secondary roads. Starting from the existing road network, primary roads were added at each simulation step until saturation (approximately

Table 1: Criteria and indicators used to compare the three forest management scenarios.

Criteria*	Indicators	Justification
Sites and zones of aboriginal interest	Age structure, i.e., proportion (%) of stands in each cohort	Keeping the age structure inside the historic range of variability is essential to maintain landscape heterogeneity and to support various traditional activities such as hunting, gathering of medicinal plants, knowledge transmission, or sacred ceremonies (Berkes and Davidson-Hunt 2006, Owen <i>et al.</i> 2009, Saint-Arnaud <i>et al.</i> 2009, Uprety <i>et al.</i> 2013).
Aboriginal land tenure and equity between families	Spatial distribution of cohorts also indicative of fragmentation, number of patches (NP), mean patch size (MPS), and size of the largest patch (LP)	The impacts of forestry activities, both positive and negative, must be spread evenly on the territory, so that all families have equal access to resources (Deutsch and Davidson-Hunt 2010).
Access to ancestral lands and ecosystem integrity	Extent of the road network, i.e., annual construction of primary and secondary roads (km)	Development of the road network reduces ecosystem integrity by causing habitat fragmentation (Trombulak and Frissell 2000). It also gives access to territories that were previously inaccessible to forest users, thus increasing the potential for conflicts between Aboriginal people and other users (Kneeshaw <i>et al.</i> 2010, Adam <i>et al.</i> 2012).
Ecosystem health, biodiversity, animal health, population densities, and possibility to pursue subsistence and other cultural and spiritual activities	Wildlife habitat availability, measured negatively as forest habitat loss, i.e., area clearcut annually (ha)	Habitat loss following a clearcut considerably affects First Nations activities by reducing or displacing local populations of plants and wildlife, especially forest obligate species (Burgess <i>et al.</i> 2005, Larouche <i>et al.</i> 2007, Weir and Corbould 2010, Cheveau <i>et al.</i> 2013).

*Modified from Saint-Arnaud *et al.* (2009)

1 km/km²). At each simulation step, secondary roads were created to reach the cut blocks located < 2 km from an existing primary road. For cut blocks located < 500 m from an existing road, neither primary nor secondary roads were created.

Aboriginal criteria and indicators of sustainable forest management

For each scenario, 25 simulations were carried out in order to follow the spatiotemporal evolution of four indicators of sustainable forest management judged relevant to the Aboriginal perspective over a total simulation period of 300 years: forest age structure, spatial configuration of forest stands, road network density, and habitat loss to clearcuts (Table 1). The indicators were related to four sets of criteria developed for the Kitcisakik territory in consultation with the community (Saint-Arnaud *et al.* 2009). To be used in the modeling exercise, indicators had to be measurable and trackable on forest maps. The selected indicators thus do not fully encompass the complexity of Aboriginal worldviews and knowledge however they cover recurrent preoccupations expressed by Aboriginal people with regards to forestry at the scale being evaluated. As well as being able to be modeled, the indicators were also chosen to cover a range of preoccupations such as changes in forest conditions and forest esthetics, habitat modifications, accessibility and fragmentation. Table 1 describes the relationship between the original criteria developed with the community and the rationale for the measurable indicators that were modeled. These indicators were discussed and validated with the community (Asselin and Basile 2012).

Results

Age structure

The proportions of the three cohorts in the landscape undergo large fluctuations over the first rotation in all three scenarios and do not stabilize until after about 150 years of simulation (Fig. 2). The proportion of the youngest first cohort stands decreased quickly in all scenarios and reached its lowest level after 40–50 years of simulation (62% for AC and DC and 44% for EM; Fig. 2a). The proportion of first cohort stands then increased at different rates depending on the scenario considered, to stabilize at 90% (AC), 70% (DC), and 50% (EM).

The proportion of 2nd cohort stands increased sharply in all scenarios following the beginning of the simulation, reaching peak values after 40–50 years of simulation (Fig. 2b). The peak was higher for EM (52%) than for AC and DC (36%). The proportion of 2nd cohort stands then decreased in all scenarios to stabilize at 8%, 15% and 25% for AC, DC and EM, respectively. The proportion of 3rd cohort stands remained below 5% for all scenarios during the first 100 years of simulation (Fig. 2c). Between 100 and 150 years the proportion of 3rd cohort stands increased abruptly and stabilized at 15% after 240 years in the DC scenario and at 25% after 210 years in the EM scenario. The proportion of 3rd cohort stands in the AC scenario remained below 4% throughout the entire simulation.

Spatial configuration of the cohorts

At simulation equilibrium, patches of 1st and 2nd cohort stands were larger and less numerous for AC than for DC or

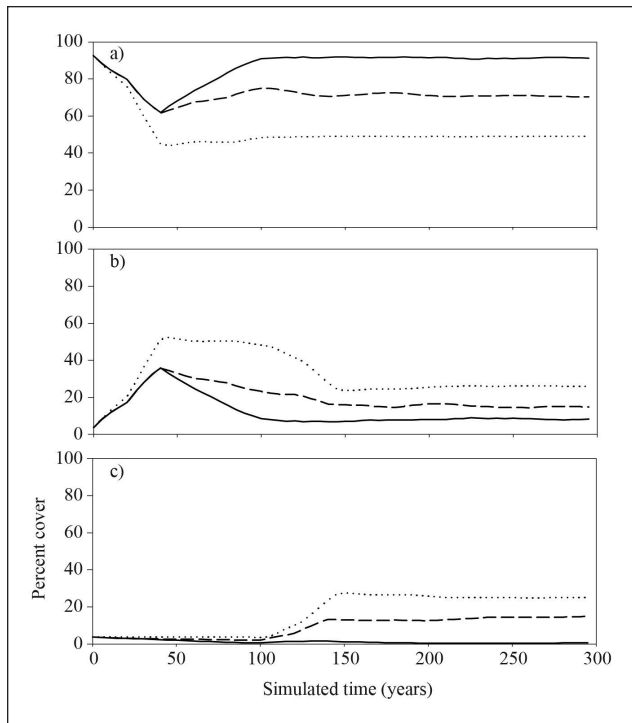


Fig. 2. Average proportion of (a) 1st cohort stands, (b) 2nd cohort stands, and (c) 3rd cohort stands for 25 simulation runs with three forest management scenarios: ecosystem-based management (dotted line), dispersed clearcuts (dashed line) and aggregated clearcuts (solid line).

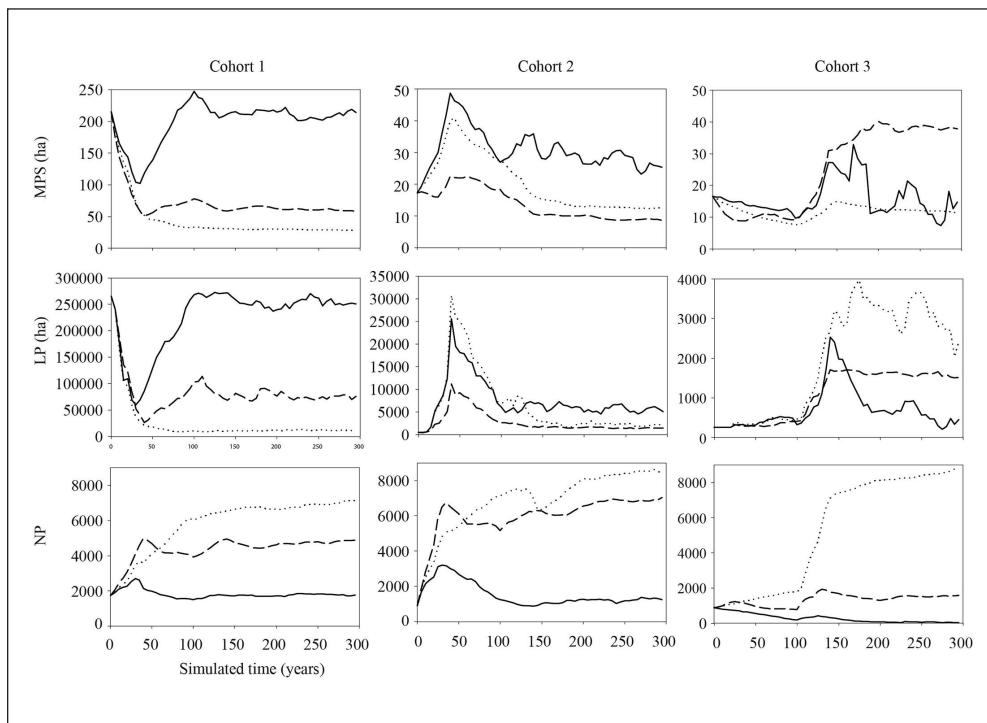


Fig. 3. Mean values for 25 simulation runs of mean patch size (MPS), size of the largest patch (LP) and number of patches (NP), for the three cohorts and for each management scenario: ecosystem-based management (dotted line), dispersed clearcuts (dashed line) and aggregated clearcuts (solid line). Note the varying y-axis scales.

EM (Fig. 3). For 3rd cohort stands, mean patch size was higher for DC, although the size of the largest patch and the number of patches were higher for EM (Fig. 3).

Access to territory and landscape connectivity

The cumulative number of kilometres of primary and secondary roads was compiled for each scenario (Fig. 4). In all scenarios, the landscape ended up being saturated with more than one primary road per km². However, saturation occurred more quickly for the DC scenario (50 years) than for either the EM or AC scenarios (both at 80 years). The DC scenario required approximately 30% more secondary roads at the end of simulation time than the other two scenarios, and this difference occurred rapidly after the simulations began.

Forest habitat availability

Forest habitat availability was evaluated as being inversely proportional to the loss of forest cover, estimated by the area harvested annually by clearcut. Annual loss of forest cover was around 2000 ha for EM, and twice as much (around 4000 ha) for AC (Fig. 5). The DC scenario showed great variability in the area harvested annually by clearcut. After the first 35 years during which 1% of the forest cover was removed annually (around 4000 ha), the simulation went through a boom-bust cycle where every peak phase failed to attain the 1% threshold initially parameterized for area harvested annually by clearcut.

Discussion

Maintaining a diversity of stand ages at the landscape scale, including old-growth forests, provides habitat for a variety of

wildlife and plant species on which traditional activities rely (Berkes and Davidson-Hunt 2006, Owen *et al.* 2009, Saint-Arnaud *et al.* 2009, Uprety *et al.* 2013). Of the three forest management scenarios analyzed in this study, only EM succeeded in preserving, at the landscape scale and for a long period of time (> 300 years), a forest structure comparable to the historic range of variability resulting from the natural fire cycle estimated at 150 years (Grenier *et al.* 2005), i.e., 50% of stands over 100 years (Cyr *et al.* 2009). This analysis, however, supposes that the partial cutting techniques used in the 2nd and 3rd cohort stands effectively emulate the conditions found in natural forests (e.g., Bergeron

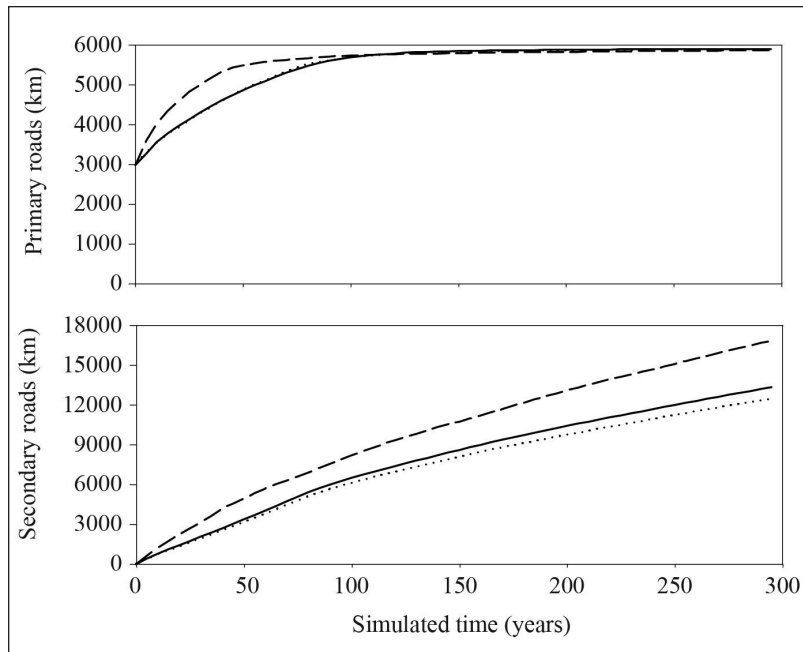


Fig. 4. Cumulative number of kilometres of primary and secondary roads over 300 years of simulation (25 runs) for three management scenarios: ecosystem-based management (dotted line), dispersed clearcuts (dashed line) and aggregated clearcuts (solid line).

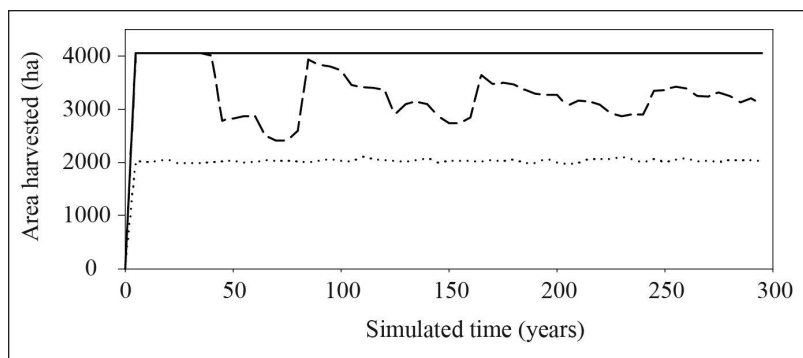


Fig. 5. Area harvested annually by clearcut (ha) over 300 years of simulation (25 runs) for three management scenarios: ecosystem-based management (dotted line), dispersed clearcuts (dashed line) and aggregated clearcuts (solid line).

et al. 1999). Partial harvesting systems that remove the largest trees may maintain continuous cover and minimise colonisation by invasive early successional weed species but may reduce habitat for wildlife species requiring large trees (Nilson *et al.* 2001). In similar forests, partial cuts were found to maintain mature forest characteristics but not to accelerate the development of old-growth characteristics (Haeussler *et al.* 2007). This is thus a reminder that it is critical that appropriate stand-level decisions be made to ensure that potentially positive results at the landscape scale are in fact manifest. In this sense it is worth noting that, across the managed landscape, DC and AC only maintained 30% and < 10% of stands older than 100 years, respectively.

Second and third cohort stands are of considerable importance because of their contribution to biodiversity (Spies and Franklin 1996, Burton *et al.* 1999, Kneeshaw and Gauthier 2003). In addition to their ecological significance, old forests

are inextricably related to the maintenance of Aboriginal cultures (Booth 1994, Krmar *et al.* 2006). Values associated with old forests include wildlife habitat, medicine, heritage, and sacredness (Owen *et al.* 2009). In the scenarios presented in this study, recruitment of 2nd cohort stands started at the beginning of the simulation, with little difference between scenarios for the first 25 years and then peaking after 40–50 years of simulation, the peak being higher for EM and DC than for AC. The more than two and a half decades required for differences to occur could contribute to the negative perception of Aboriginal communities towards all kinds of forest management (Saint-Arnaud *et al.* 2009). The DC scenario generated three times more 2nd and 3rd cohort stands than AC, but only half as much as EM. However, the fact that change could be perceived in one person's lifetime could lead to eventual changes in perception. The even slower change in the proportion of the oldest 3rd cohort stands (> 100 years) would be unsatisfactory for decisions made only for the current generation. The modeled changes may thus be useful in convincing the community that choices for forest management can be made that will leave future generations with a better forest, which is important in the culture as the people see themselves as stewards of the land (Davidson-Hunt and Berkes 2003, Saint-Arnaud *et al.* 2009). Leaving better forests for future generations may also provide economic advantages (e.g., jobs) to some community members.

Patch size was more variable in the EM scenario, which could favour a greater variety of wildlife species with varying interior and edge habitat needs by creating greater landscape heterogeneity. Higher retention of patches of 2nd and 3rd cohort stands in DC and EM can be explained by reduced harvest rates compared to AC. Indeed, DC failed to reach the 1% targeted annual harvest rate, probably because of constraints to cut block dispersal. In the EM scenario, only half of the area was clearcut, whereas the rest of the forest was managed with one of two types of partial cuts. These harvests may in themselves disturb wildlife habitat with multiple entries occurring in stands over a large portion of the landscape (Vanderwel *et al.* 2009, Le Blanc *et al.* 2010, Fenton *et al.* 2013).

Reduced harvest in the DC and EM scenarios will have environmental and sociocultural advantages, but is an obvious drawback from a profitability point of view. EM maintained at least one patch of 3rd cohort stands of greater dimension than the other scenarios for the entire simulation. Old-growth stands must have a sufficient dimension to maintain interior forest species and ecosystem integrity (Angermeier and Karr 1994, Haeussler and Kneeshaw 2003).

Although the simulations started at time 0 with a territory that had already been heavily disturbed (more than 60% of

the Kitcisakik territory has been clearcut over the last 50 years (Saint-Arnaud *et al.* 2009), over the long-term (> 100 years), both EM and DC restored and maintained a certain proportion of old forests in every family hunting ground. A variety of habitats in each family hunting ground could be expected to have greater social acceptability, as equity in amount of disturbance between family hunting grounds would ensure that all families have equal chances of practicing cultural activities (Deutsch and Davidson-Hunt 2010). From a wildlife perspective, AC resulted in an aggregation of old forests which could be important for some species such as caribou (Courtois *et al.* 2007) or marten (Cheveau *et al.* 2013).

EM and DC did not reduce the amount of primary roads, compared to AC. Nevertheless, EM necessitated fewer secondary roads than either AC (8% more) or DC (36% more). Moreover the territory was saturated with primary roads faster for DC than for EM or AC. Although increased access to territory might seem a positive outcome for Aboriginal people (facilitating access to hunting grounds), road building contributes to the deterioration of forest ecosystems (Trombulak and Frissell 2000) and has both positive and detrimental impacts on Aboriginal people (Kneeshaw *et al.* 2010, Adam *et al.* 2012). For example, Adam *et al.* (2012) showed that road networks created inter-community tensions as knowledge and access to the land were no longer dependent on traditional cultural norms. Furthermore, increased access to the territory by non-aboriginal users can cause conflicts with Aboriginal people (Kneeshaw *et al.* 2010). The slower development of the road network in the EM scenario would give more time to Aboriginal people to adopt measures to mitigate impacts. However, EM would lose its advantage with regards to road density if harvest levels were increased to be the same levels as in AC, underlining the importance of harvest levels.

Because EM was parameterized so that the annual harvest would be half by clearcut and the rest by partial cuts, it maintained more forest cover than the other two scenarios. Earlier research has suggested that partial cuts maintain some of the habitat requirements of wildlife species of high Aboriginal importance, like marten (*Martes americana*) or fisher (*Pekania pennanti*), which avoid recent clearcuts (Weir and Corbould 2010, Cheveau *et al.* 2013). Moreover, partial cuts could allow for the restoration or maintenance on the landscape of tree species of high value for Algonquin people, like eastern white-cedar (*Thuja occidentalis*) (Larouche *et al.* 2007) or white pine (*Pinus strobus*) (Burgess *et al.* 2005). Maintaining forest cover after harvesting is also more acceptable from an aesthetic point of view (Yelle *et al.* 2008), thereby increasing social acceptability (Germain 2012). It could thus reduce the resentment with regards to the deterioration of forest ecosystems (Saint-Arnaud *et al.* 2009). The area lost to clearcut only reflects part of the habitat requirements of all wildlife species. Nevertheless, most species of high cultural interest need forest cover for some or most of their activities. Species such as moose (*Alces alces*) and deer (*Odocoileus virginianus*) which benefit from patchy environments require coniferous forest for winter habitat whereas forestry has greatly increased the abundance of early successional shade-intolerant hardwoods (Carleton and MacLellan 1994, Laquerre *et al.* 2009). Furthermore, clearcuts are a major and recurrent issue in studies on the preoccupations of aboriginal people with regard to forest management

(e.g., Sapic *et al.* 2009, Germain 2012). The area lost to clearcut is also an integrative index. Modeling habitat suitability indices (HSI) requires choosing target organisms one at a time due to different habitat needs. It also requires a thorough understanding of the biology of the organism in question and the ability to model all the factors that influence its fitness (Roloff and Kernohan 1999). This highlights the complexity of our exercise and also the need to follow-up this work with studies based on other indicators and other scenarios.

Conclusion

Among the three scenarios modelled in this study, the constraints imposed for ecosystem-based management led it to harvest less and thus it scored best on the four measured indicators, pointing towards the fact that annual allowable cut should be a key factor to put on the table to ensure better compliance with Aboriginal criteria of sustainable forest management. Using standard linear programming, Dhital *et al.* (2013) showed that implementing ecosystem-based management and taking Aboriginal land-use into account resulted in a 7%–21% reduction of the annual allowable cut, compared to a “business as usual” scenario. Even after such a reduction, the resulting annual allowable cut values were still within the range of harvest levels in that area between 2000 and 2010. Using partial cuts to harvest some of the wood biomass while maintaining forest cover could allow for some traditional activities to continue (Owen *et al.* 2009, Saint-Arnaud *et al.* 2009, Uprety *et al.* 2013) while lowering the pressure on the annual allowable cut (Ruel *et al.* 2013).

EM was the scenario that performed the best in the indicators tested, creating a more even distribution of stands from the three cohorts across the landscape, in line with the Aboriginal land tenure system where each family should have equal access to resources. It should however be stressed that this study was conducted at the landscape scale and relied on a limited number of indicators. Consequently, although the results point towards higher acceptability of the ecosystem-based management scenario, further studies are needed to identify and test additional indicators and scenarios at different spatial and temporal scales. Participatory techniques would be particularly appropriate in this regard (Fraser *et al.* 2006). Given the current state of the landscape, many of the advantages of EM would not be expressed for generations whereas the economic disadvantages may occur more rapidly. As sagaciously expressed by a member of the Kitcisakik community when presented with the results of the simulations, “ecosystem-based management is not the best scenario but the least worst”.

Acknowledgements

Funding for this study was provided by the Sustainable Forest Management Network of Centres of Excellence, the Natural Sciences and Engineering Research Council of Canada, the Social Sciences and Humanities Research Council of Canada, and the Canadian Forest Service. Many thanks to Andrew Fall and Annie Belleau for help with the SELES software. This work greatly benefitted from fruitful discussions with Yvan Croteau, Marie Saint-Arnaud, Charlie Papatie, Robert Penosway, and other members of the Kitcisakik Algonquin community, as well as the project's industrial and governmental partners.

References

- Adam, M.-C., D. Kneeshaw and T. M. Beckley. 2012. Forestry and road development: direct and indirect impacts from an aboriginal perspective. *Ecol. Soc.* 17(4): 1.
- Adelson, N. 2000. Being alive and well: Health and the politics of Cree well-being. University of Toronto Press, Toronto, Canada.
- Ananda, J. and G. Herath. 2003. Incorporating stakeholder values into regional forest planning: a value function approach. *Ecol. Econ.* 45(1): 75-90.
- Angermeier, P.L. and J.R. Karr. 1994. Biological integrity versus biological diversity as policy directives. Protecting biotic resources. *BioScience* 44(10): 690-697.
- Asselin, H. and S. Basile. 2012. Éthique de la recherche avec les Peuples autochtones : qu'en pensent les principaux intéressés? *Éthique publique* 14(1): 333-345.
- Bélanger, L. 2001. La forêt mosaïque comme stratégie de conservation de la biodiversité de la sapinière boréale de l'Est. L'expérience de la Forêt Montmorency. *Nat. can.* 125(3): 18-25.
- Bergeron, Y., B. Harvey, A. Leduc and S. Gauthier. 1999. Forest management guidelines based on natural disturbance dynamics: stand- and forest-level considerations. *Forest. Chron.* 75(1): 49-54.
- Bergeron, Y., S. Gauthier, V. Kafka, P. Lefort and D. Lesieur. 2001. Natural fire frequency for the eastern Canadian boreal forest: consequences for sustainable forestry. *Can. J. Forest Res.* 31(3): 384-391.
- Berkes, F. 2012. *Sacred Ecology*, 3rd edition. Routledge, New York, NY, USA.
- Berkes, F. and I.J. Davidson-Hunt. 2006. Biodiversity, traditional management systems, and cultural landscapes: examples from the boreal forest of Canada. *Int. Soc. Sci. J.* 58(187): 35-47.
- Bliss, J.C. 2000. Public perceptions of clearcutting. *J. Forest.* 98(12): 4-9.
- Booth, D. 1994. *Valuing nature: The decline and preservation of old growth forests.* Rowman and Littlefield Publishers Inc., London, UK.
- Booth, D.L., D.W.K. Boulter, D.J. Neave, A.A. Rotherham and A.D. Welsh. 1993. Natural forest landscape management: A strategy for Canada. *Forest. Chron.* 69(2): 141-145.
- Burgess, D., C. Robinson and S. Wetzel. 2005. Eastern white pine response to release 30 years after partial harvesting in pine mixed wood forest. *Forest Ecol. Manage.* 209(1): 117-129.
- Burton, P. J., D.D. Kneeshaw, and K.D. Coates. 1999. Managing forest harvesting to maintain old growth in boreal and sub-boreal forests. *Forest. Chron.* 75(4): 623-631.
- Carleton, T.J. and P. MacLellan. 1994. Woody vegetation responses to fire versus clear-cutting logging: A comparative survey in the central Canadian boreal forest. *Ecoscience* 1(2): 141-152.
- Castro, A.P. and E. Nielson. 2001. Indigenous people and co-management: Implications for conflict management. *Environ. Sci. Policy* 4(4): 229-239.
- Cheveau, M., L. Imbeau, P. Drapeau and L. Bélanger. 2008. Current status and future directions of traditional ecological knowledge in forest management: a review. *Forest. Chron.* 84(2): 231-243.
- Cheveau, M., L. Imbeau, P. Drapeau and L. Bélanger. 2013. Marten space use and habitat selection in managed coniferous boreal forests of eastern Canada. *J. Wildlife Manage.* 77(4): 749-760.
- Côté, M.-A. and L. Bouthillier. 1999. Analysis of the relationship among stakeholders affected by sustainable development and forest certification. *Forest. Chron.* 75(6): 961-965.
- Courtois, R., J.-P. Ouellet, L. Breton, A. Gingras and C. Dussault. 2007. Effects of forest disturbance on density, space use, and mortality of woodland caribou. *Ecoscience* 14(4): 491-498.
- Cyr, D., S. Gauthier, Y. Bergeron and C. Carcaillet. 2009. Forest management is driving the eastern North American boreal forest outside its natural range of variability. *Front. Ecol. Environ.* 7(10): 519-524.
- Davidson-Hunt, I. and F. Berkes. 2003. Learning as you journey: Anishinaabe perception of social-ecological environments and adaptive learning. *Conserv. Ecol.* 8(1): 5.
- Deutsch, N. and I. Davidson-Hunt. 2010. Pikangikum family hunting areas and traplines: Customary lands and aboriginal land use planning in Ontario's Far North. In: M.G. Stevenson and D.C. Natcher (eds.). *Planning Co-existence: Aboriginal considerations and approaches in land use planning.* pp. 149-170. Canadian Circumpolar Institute Press, Edmonton, AB, Canada.
- Dhital, N., F. Raulier, H. Asselin, L. Imbeau, O. Valeria and Y. Bergeron. 2013. Emulating boreal forest disturbance dynamics: Can we maintain timber supply, aboriginal land use, and woodland caribou habitat? *Forest. Chron.* 89(1): 54-65.
- Didion, M., M.-J. Fortin and A. Fall. 2007. Forest age structure as indicator of boreal forest sustainability under alternative management and fire regimes: A landscape level sensitivity analysis. *Ecol. Model.* 200(1): 45-58.
- Fall, A. and J.A. Fall. 2001. A domain-specific language for models of landscape dynamics. *Ecol. Model.* 141(1): 1-18.
- Fall, A., M.-J. Fortin, D. Kneeshaw, S. Yamasaki, C. Messier, L. Bouthillier and C. Smyth. 2004. Consequences of various landscape-scale ecosystem management strategies and fire cycles on age-class structure and harvest in boreal forests. *Can. J. Forest Res.* 34(2): 310-322.
- Fenton, N.J., L. Imbeau, T. Work, J. Jacobs, H. Bescond, P. Drapeau and Y. Bergeron. 2013. Lessons learned from 12 years of ecological research on partial cuts in black spruce forests of northwestern Québec. *Forest. Chron.* 89(3): 350-359.
- Franklin, J.F. et al. 2002. Disturbances and structural development of natural forest ecosystems with silvicultural implications, using Douglas-fir forests as an example. *Forest Ecol. Manage.* 155(1): 399-423.
- Fraser, E.D.G., A.J. Dougill, W.E. Mabee, M. Reed and P. McAlpine. 2006. Bottom up and top down: Analysis of participatory processes for sustainability indicator identification as a pathway to community empowerment and sustainable environmental management. *J. Environ. Manage.* 78(2): 114-127.
- Germain, R. 2012. Évaluation de l'acceptabilité sociale par la communauté algonquienne de Pikogan d'une stratégie d'aménagement écosystémique. M.Sc. thesis, Département des sciences appliquées. Université du Québec en Abitibi-Témiscamingue, Rouyn-Noranda, QC, Canada.
- Gladu, J.P. and C. Watkinson. 2004. Measuring sustainable forest management: a compilation of aboriginal indicators. A report for the Canadian model forest network - aboriginal strategic initiative.
- Grenier, D., Y. Bergeron, D. Kneeshaw and S. Gauthier. 2005. Fire frequency for the transitional mixedwood forest of Timiskaming, Quebec, Canada. *Can. J. Forest Res.* 35(3): 656-666.
- Haussler, S. and D.D. Kneeshaw. 2003. Comparing forest management to natural process. In: P.J. Burton, C. Messier, D.W. Smith and W.L. Adamowicz (eds.). *Towards sustainable management of the boreal forest.* pp. 307-368. NRC Research Press, Ottawa, ON, Canada.
- Haussler, S., Y. Bergeron, S. Brais and B.D. Harvey. 2007. Natural dynamics-based silviculture for maintaining plant biodiversity in *Populus tremuloides*-dominated boreal forests of eastern Canada. *Botany* 85(12): 1158-1170.
- Karjala, M.K., E.E. Sherry and S.M. Dewhurst. 2004. Criteria and indicators for sustainable planning: a framework for recording Aboriginal resource and social values. *Forest Policy Econ.* 6(2): 95-110.
- Kneeshaw, D. and S. Gauthier. 2003. Old growth in the boreal forest: A dynamic perspective at the stand and landscape level. *Environ. Rev.* 11(S1): S99-S114.
- Kneeshaw, D., M. Larouche, H. Asselin, M.-C. Adam, M. Saint-Arnaud and G. Reyes. 2010. Road rash: Ecological and social impacts of road networks on First Nations. In: M.G. Stevenson and D.C. Natcher (eds.). *Planning Co-existence: Aboriginal considerations and approaches in land use planning.* pp. 171-184. Canadian Circumpolar Institute Press, Edmonton, AB, Canada.

- Krcmar, E., G.C. van Kooten, H. Nelson, I. Vertinsky and J. Webb. 2006. The Little Red River Cree Nation's forest management strategies under a changing forest policy. *Forest. Chron.* 82(4): 529-537.
- Laquerre, S., A. Leduc and B.D. Harvey. 2009. Augmentation du couvert en peuplier faux-tremble dans les pessières noires du nord-ouest du Québec après coupe totale. *Ecoscience* 16(4): 483-491.
- Larouche, C., J.-C. Ruel and J.-M. Lussier. 2007. Factors affecting northern white cedar (*Thuja occidentalis*) seedling establishment and early growth in mixedwood stands. *Can. J. Forest Res.* 41(3): 568-582.
- Larouche, M. 2008. La modélisation de scénarios d'aménagement forestier à l'échelle du paysage : un outil d'aide à la décision en foresterie autochtone. M.Sc. thesis, Département des sciences biologiques, Université du Québec à Montréal, Montreal, QC, Canada.
- Le Blanc, M.-L., D. Fortin, M. Darveau and J.-C. Ruel. 2010. Short term response of small mammals and forest birds to silvicultural practices differing in tree retention in irregular boreal forests. *Ecoscience* 17(3): 334-342.
- Menzies, C.R. and C. Butler. 2006. Introduction. Understanding ecological knowledge. In: C.R. Menzies (ed.). *Traditional Ecological Knowledge and Natural Resource Management*. pp. 1-17. University of Nebraska Press, Lincoln, NE, USA.
- Natcher, D.C. and C.G. Hickey. 2002. Putting the community back into community-based resource management: A criteria and indicators approach to sustainability. *Hum. Organ.* 61(4): 350-363.
- Ndione, P.D. 2014. Impacts de la foresterie industrielle sur les activités traditionnelles autochtones en forêt tempérée mixte. Ph.D. thesis, Institut de recherche sur les forêts, Université du Québec en Abitibi-Témiscamingue, Rouyn-Noranda, QC, Canada.
- Nilsson, S.G., J. Hedin and M. Niklasson. 2001. Biodiversity and its assessment in boreal and nemoral forests. *Scand. J. Forest Res.* 16(S3): 10-26.
- Owen, R.J., P.N. Duinker and T.M. Beckley. 2009. Capturing old-growth values for use in forest decision-making. *Environ. Manage.* 43(2): 237-248.
- Pâquet, J. and L. Bélanger. 1997. Public acceptability thresholds of clearcutting to maintain visual quality of boreal balsam fir landscape. *Forest Sci.* 43(1): 46-55.
- Parrotta, J.A. and R.L. Trosper (eds.). 2012. *Traditional forest-related knowledge: Sustaining communities, ecosystems and biocultural diversity*. World Forest Series vol. 12. Springer Dordrecht, Netherlands.
- Robson, M. and L.M. Hunt. 2010. Evaluating local multi-stakeholder platforms in forest management in Ontario. *Forest. Chron.* 86(6): 742-752.
- Roloff, G.T. and B.J. Kernohan. 1999. Evaluating reliability of habitat suitability index models. *Wildlife Soc. B.* 27(4): 973-985.
- Ruel, J.-C., D. Fortin and D. Pothier. 2013. Partial cutting in old-growth boreal stands: An integrated experiment. *Forest. Chron.* 89(3): 360-369.
- Saint-Arnaud, M., H. Asselin, C. Dubé, Y. Croteau and C. Papatie. 2009. Developing criteria and indicators for aboriginal forestry: mutual learning through collaborative research. In: M.G. Stevenson and D.C. Natcher (eds.). *Changing the Culture of Forestry in Canada: Building Effective Institutions for Aboriginal Engagement in Sustainable Forest Management*. pp. 85-105. Canadian Circumpolar Institute Press, Edmonton, AB, Canada.
- Sapic, T., U. Runesson and M.A. Smith. 2009. Views of Aboriginal people in northern Ontario on Ontario's approach to Aboriginal values in forest management planning. *Forest. Chron.* 85(5): 789-801.
- Saucier, J.-P., J.-F. Bergeron, P. Grondin and A. Robitaille. 1998. Les régions écologiques du Québec méridional (troisième version). L'Aubelle 124: S1-S12.
- Seymour, R.S. and M.L. Hunter Jr. 1992. New forestry in eastern spruce-fir forests: principles and applications to Maine. Maine Agricultural and Forest Experiment Station, Miscellaneous Report 716.
- Smith, A.C., and J.A. Schaefer. 2002. Home-range size and habitat selection by American marten (*Martes americana*) in Labrador. *Can. J. Zool.* 80(9): 1602-1609.
- Spies, T.A. and J.F. Franklin. 1996. The diversity and maintenance of old-growth forests. In: R.C. Szaro and D.W. Johnson (eds.). *Biodiversity in Managed Landscapes: Theory and Practice*. pp. 296-314. Oxford, New York, NY, USA.
- Trombulak, S.C. and C.A. Frissell. 2000. Review of ecological effects of roads on terrestrial and aquatic communities. *Conserv. Biol.* 14(1): 18-30.
- Uprety, Y., H. Asselin, Y. Bergeron, F. Doyon and J.-F. Boucher, J.-F. 2012. Contribution of traditional knowledge to ecological restoration: Practices and applications. *Ecoscience* 19(3): 225-237.
- Uprety, Y., H. Asselin and Y. Bergeron. 2013. Cultural importance of white pine (*Pinus strobus* L.) to the Kitcisakik Algonquin community of western Quebec, Canada. *Can. J. Forest Res.* 43(6): 544-551.
- Vanderwel, M.C., S.C. Mills and J.R. Malcolm. 2009. Effects of partial harvesting on vertebrate species associated with late-successional forests in Ontario's boreal region. *Forest. Chron.* 85(1): 91-104.
- Weir, R.D. and F.B. Corbould. 2010. Factors affecting landscape occupancy by fishers in North-Central British Columbia. *J. Wildlife Manage.* 74(3): 405-410.
- Wyatt, S. 2008. First Nations, forest lands, and "aboriginal forestry" in Canada: From exclusion to comanagement and beyond. *Can. J. Forest Res.* 38(2): 171-180.
- Yelle, V., L. Bélanger and J. Pâquet. 2008. Acceptabilité visuelle de coupes forestières pour la pessière noire : comparaison de la coupe à blanc traditionnelle et de différents types de rétention végétale chez divers groupes d'intérêt issus d'une région ressource forestière. *Can. J. Forest Res.* 38(7): 1983-1995.

Appendix 1. Parameters of the simulations used to model the three management scenarios for the Kitcisakik territory (more details in Larouche (2008)).

Parameters	Values			
Target harvest rate (ha/year)	4055 (1% of total forested area)			
Minimum stand age for harvesting (years)	75			
Proportion of each harvesting type (%)		Clearcut	Partial cut 1	Partial cut 2
	AC	100	0	0
	DC	100	0	0
	EM	50	25	25
Time interval between partial cuts (years)	10			
Size classes of harvested areas (ha)		Small	Medium	Large
	AC	70–100	101–150	–
	DC	50–70	71–150	–
	EM	100–150	151–300	301–400
Proportion of each size class (%)		Small	Medium	Large
	AC	90	10	0
	DC	90	10	0
	EM	70	25	5
Mean stand age at 7 m (years)	35			
Minimum proportion of stands ≥ 7 m in a management sub-unit (%)	33			
Minimum proportion of residual forest ≥ 3 m high in a cut block (%)	50 (only for dispersed clearcuts)			
Maximum distance between a cut block and a primary road (km)	2			
Distance between a cut block and a road not necessitating new road construction (m)	500			