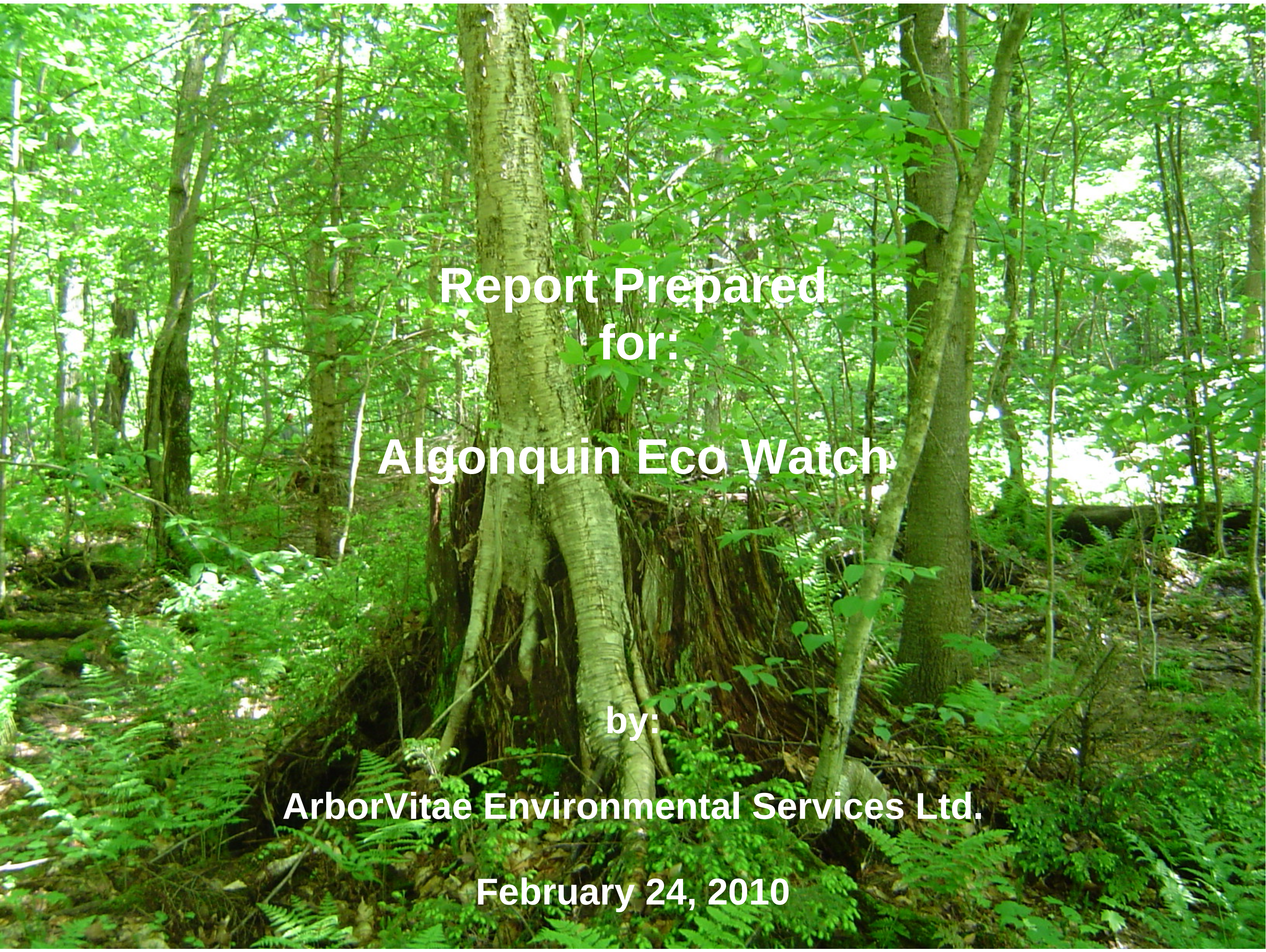
Management of Algonquin Park's West Side Forests and Provision of Associated Habitat



Report Prepared for:

Algonquin Eco Watch

by:

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1 Executive Summary

In 2007, Algonquin Eco Watch requested that ArborVitae Environmental Services Ltd prepare a report examining the management of white and red pine in Algonquin Park. These forests dominate the east side of the Park. Earlier in 2008, Algonquin Eco Watch asked ArborVitae Environmental Services Ltd. (AVES) to examine the management of the forests of the west side of the Park, which are dominated by tolerant hardwoods and eastern hemlock, but also contain a wide range of other coniferous and deciduous species. The intent of the review was to assess the management approach against the most recent science, the major directions of forest management, and to consider how management has affected wildlife.

After reviewing the 2005 Forest Management Plan for the Park, which is the document that is driving management of the forests in the Park's recreation/utilization zone, we considered how the current forest differs from the pre-industrial forest. We found a number of differences regarding the abundance of both tree and wildlife species. Based on evidence from surveyors' notes, other historical records, dendrochronology, and pollen deposition records in lake sediments, it is clear that the west side forests retain many of the same tree species but their relative abundance has shifted. Perhaps the most dramatic change is the loss of most of the white pine, which was logged and did not regenerate on many sites on the west side (it still dominates the east side). There are also documented reductions in the relative abundance of red spruce, larch, yellow birch, eastern hemlock, American elm, and eastern white cedar. The 2005 FMP indicates that red oak and jack pine have also declined and we suspect that, in addition to yellow birch, other mid-tolerant hardwood species such as basswood, black cherry, and white ash (which are at the northern limit of their range in the Park) are less common in today's forest than they were previously. While there is little quantitative evidence for this, such an outcome has been documented in other northern hardwood forests that have been subject to the application of the single-tree selection system.

While the declines in larch and elm are due to disease, the loss of white pine and decline in red spruce are due to lack of renewal following harvesting. Yellow birch and hemlock declines are in part due to heavy browsing by deer and moose, but also due in part to heavy harvesting in the 1920's to the 1960's.

Part of the difficulty in capturing the relative abundance of the various species is that the forest inventory does a poor job of capturing the presence of species that rarely dominate a stand but instead may be present as scattered individuals or small groves. For these species, there is little basis for estimating either a historical baseline or the current levels, and they tend to be overlooked in management.

We also found extensive research documenting the effects of forest management in simplifying the forest, especially when single-tree selection is applied, as is generally the case in western Algonquin Park. The standard application of single-tree selection harvesting favours shade tolerant sugar maple, because it can reproduce best in the conditions that are created and because beech is less favoured for timber and so tends to be discriminated against when timber production is the main goal. The Algonquin Forestry Authority now uses marking criteria that place a priority on retaining more diverse species, including hemlock, red spruce, beech, oak, basswood, black cherry and yellow birch. Unfortunately, the lack of goals in the FMP regarding comprehensive species abundance, and the inability of the inventory to enable an accurate assessment of abundance mean that there are significant unknowns regarding the current forest and how it may be changing.

Finally, it is clear that forest management simplifies the structure of the forest by reducing the presence of large old trees and old growth patches, reducing the incidence of standing snags and downed logs, and creating a more uniform stand. Fire suppression has been effective in reducing the incidence of fires in the Park, and this is especially so on the west side where fire was never as prevalent as on the east. While wind, insects and disease remain active, there is some question as to whether medium and large sized patches are as common as they would have been in the pre-industrial forest, and the lack of ground fires alters the competitive balance between species.

All of these changes have contributed to a simplification and homogenization of the forest. The 2005 FMP has in place some measures to redress these tendencies but they appear to us to be insufficient and could be better integrated.

The changes in forest cover have led to changes in wildlife composition as well, and other factors have also been active. We have documented the impacts of the passenger pigeon on forests and Algonquin, which was within the range of the passenger pigeon, would have been subject to them. Elk and caribou were formerly present in the Park as well, but have been extirpated for reasons that have little to do with forestry. Deer and moose populations have fluctuated while logging as well as disease relationships and perhaps hunting have affected the competitive balance of these two species. Logging is widely credited with attracting deer to Algonquin Park, but errors in the 1960’s managing the winter yards in the Park mean that deer tend to be largely absent during winter.

The most pronounced impacts of forest management seem to fall on beavers and birds. The lack of harvesting around water bodies, the use of selection harvesting, and the suppression of fire mean that there are few openings created near water that will support the poplar and white birch that the beaver prefer, hence populations have declined. Beaver can be a significant ecological force on the landscape and their reduction can lead to secondary impacts. The simplification of the forest has generally led to a reduction in habitat quality, which does not eliminate species but reduces their abundance. In addition, reductions in species such as eastern hemlock and red spruce have negatively affected birds that prefer those tree species, including red shouldered hawks, barred and saw whet owls, wood thrush, oven birds and blackburnian, black throated green and parula warblers.

We have provided a number of recommendations to improve the ability of forest managers to create a more diverse forest and to accelerate the restoration of elements that have been lost or reduced. We also observe that the Park is threatened by several invasive exotic insect pests: emerald ash borer, beech bark disease, and more remotely but most seriously, the Asian long-horned beetle. The ash borer has probably reached the Park while beech bark disease is likely to arrive within several years at most, and both will have major negative impacts. Since there are no real protection measures against either pest, we offer some very limited suggestions to try to mitigate the impacts.

Lastly, we discuss the potential impacts of on-going soil acidification on forest productivity and discuss some measures that might be instituted to help the forest community begin to understand the extent and severity of the issue.

2 Introduction

Algonquin Provincial Park is Ontario's oldest and best-known park. To many people, the Park is an iconic natural environment. Forest management has been an integral part of the Algonquin area dating back to the mid 1800's, and it continues today within the Park. Since its inception in 1893, the Park has provided economic benefits through forest management while at the same time providing abundant recreational opportunities. "An Act to Establish the Algonquin National Park of Ontario", which provided the legal basis for establishing Algonquin as Ontario's first provincial park, was passed by the government of Oliver Mowat in 1893. The Act stated that Algonquin was to serve "as a public park and forest reservation, fish and game preserve, health resort and pleasure ground for the benefit, advantage and enjoyment of the people of Ontario". The Algonquin Park management plan (MNR 1998a) states that the goal of Park management is "to provide protection of natural and cultural features, continuing opportunities for a diversity of low intensity recreational, wilderness and natural environmental experiences, and, within this provision, continue and enhance the Park's contribution to the economic, social and cultural life of the region."

With the Park addressing so many facets of society's needs, land use zoning is a key aspect of the Park's management. The Algonquin Park Master Plan designates seven land use zones in the Park: access, development, historical, nature reserve, wilderness, natural environment, and recreation/utilization. Of these, the recreation/utilization is the only zone in which forest management activities are permitted. It is also the largest zone, comprising approximately 78% of the Park's area.

The forested ecosystems on the east and west sides of the Park are distinctly different, reflecting the two different topographic complexes in Algonquin Park. The Precambrian uplands (i.e., the Algonquin dome) are located on the west side of the Park, and the Ottawa Lowlands that slope down to the Ottawa River are on the east side (Cumming 2005). The soils and moisture regime also differ from one side of the Park to the other; the western side receives more precipitation and the soils are generally deep and fresh. The eastern part of the Park is drier, and contains mixtures of fresh, silty, and sandy soils. Sites there tend to be drier and have a much higher pine component than in the western part of the Park. The two regions are so different that they are classified as separate ecodistricts (or Forest Regions). Ecodistrict 5E-9 is essentially synonymous with the west side tolerant hardwood forests, while Ecodistrict 5E-10 is associated with the white pine dominated forests on the Park's east side (Cumming 2005).

Algonquin Eco Watch is not opposed to logging in the Park, but takes a keen interest in the management of the Park and the conservation of its resources. In 2006, Algonquin Eco Watch commissioned ArborVitae Environmental Services to conduct a review of the pine management practices in the Park (ArborVitae Environmental Services Ltd 2007). With this study, Eco Watch has requested that we examine the management of the forests on the west side, which are primarily tolerant hardwoods with some hemlock.

Algonquin Eco Watch's major concerns can be expressed as a series of questions:

1. How does the forest on the west side of the Park, as it is managed under current practices, compare with the pre-settlement forest?

2. What are the impacts of current management under the single-tree selection system on the forest and the habitat that it provides?
3. How can management practices be modified to create a more biodiverse forest?

The first of these questions is essentially a question of how closely the current forest cover approximates what would have been found in the Park forest had it been unaffected by humans. Almost all of the Park's forests have been harvested at one time or another during the past 150 years, and most areas have been harvested at least several times. The earliest harvesting was focused on removing mature white pine, while subsequent felling removed a wider range of species, especially yellow birch, sugar maple, red pine, white and red spruce, and hemlock, as well as any residual white pine.

Since the formation of the Algonquin Forest Authority, and the re-setting of management unit boundaries, both of which occurred in 1975, the Park is now managed on a more integrated basis as a single management unit. During the past couple of decades, awareness of the value of biodiversity and ecosystem restoration have influenced Park management, however it will take considerable time for the forest to recover from the effects of past harvesting. And of course, in the recreation /utilization zone, harvesting continues and so the question there centres on the characteristics of the forest that will be created.

The tolerant hardwood forests, as well as hemlock, are primarily managed using the single-tree selection system, and recent research from the United States has identified a number of ways in which common applications of selection harvesting reduce the diversity of tree species. This concern motivates the second and third questions.

The intent of this report is to review the implications of the current management direction on the Park's tolerant hardwood communities, which are primarily located on the west side of the forest. Throughout this report, the use of the term "west side" will be taken to be synonymous with the tolerant hardwood dominated forests in Algonquin, even though there are other forest types present on the west side and the tolerant hardwood stands themselves frequently contain some other types of species.

3 Current Tolerant Hardwood Management

Algonquin Park is managed by the Ontario Ministry of Natural Resources (MNR), Parks Ontario and the Algonquin Forest Authority (AFA). The AFA is a Crown corporation, licensed to harvest timber in the recreation/utilization zone. The terms of its licence require the AFA to prepare Forest Management Plans for the Park and administer their implementation. The current plan came into effect on April 1, 2005; a new, ten-year FMP is being developed and is expected to come into effect on April 1, 2010.

For management purposes, the Park's forest is divided into forest units; a forest unit combines the dominant species or species types with the management technique commonly used in their management. The 2005 FMP recognizes 12 main forest units:

- PjCC – jack pine clearcut;
- LCUS – lowland conifer uniform shelterwood;
- PRCC – red pine clearcut;
- SbCC – black spruce clearcut;
- ORUS – red oak uniform shelterwood;
- HeSEL – hemlock selection;
- SFUS – spruce-fir uniform shelterwood;
- IntCC – Intolerant hardwood clearcut;
- MWUS – mixedwood uniform shelterwood;
- HDUS – tolerant hardwood uniform shelterwood;
- PWUS – white pine uniform shelterwood; and
- HDSEL; tolerant hardwood selection.

Table FMP-1 of the 2005 FMP shows that there are 488,619 ha of productive forest in the managed part of the forest (i.e. the recreation/utilization zone), which constitutes 79% of the 618,520 ha of total productive forest within the Park.

3.1 The Relative Abundance of Tolerant Hardwood Stands

The total amount of area of each of the Park's forest units is shown in Figure 1, and two of the three largest forest units contain tolerant hardwood: HDSEL and HDUS.

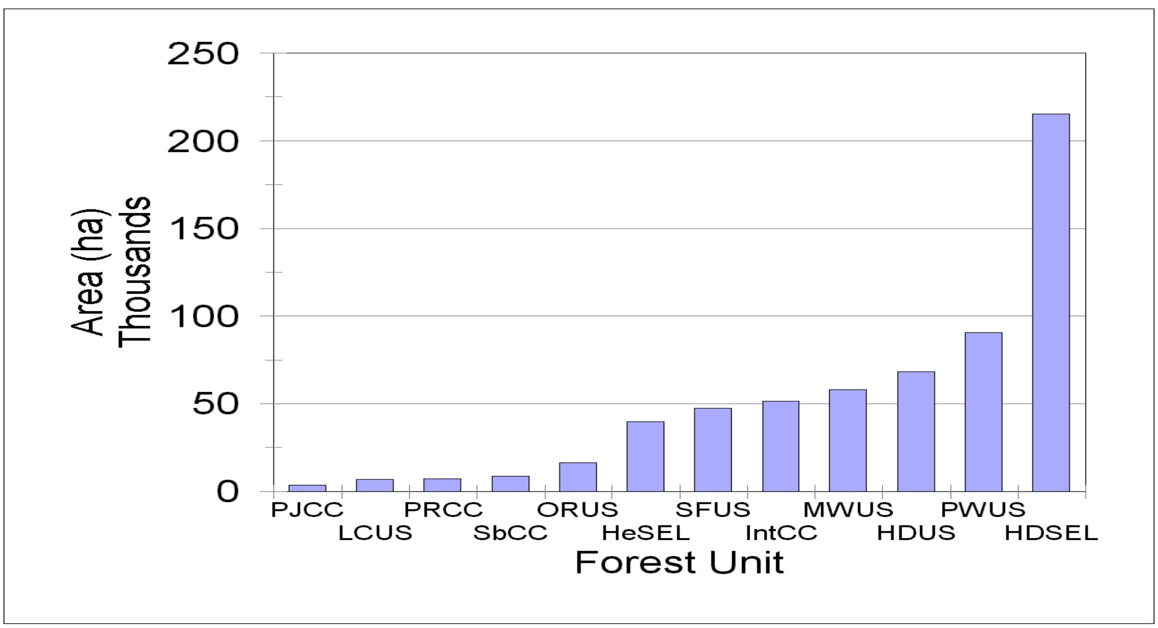

Figure 1. Forest Unit areas (all land use zones) of Algonquin Park (Source: SFMM Input files for 2005 FMP).

The stands in the HDSEL forest unit are managed under the single-tree selection system, while those in the HDUS forest unit are managed using the uniform shelterwood system. The stands in the hemlock forest unit (HeSEL) often contain high proportions of hardwoods, and are important components of the west side forest.

Figure 1 shows the entire Park forest. Within the available managed productive portion of the forest, which is most of the production forest area within the recreation/utilization (R/U) zone, the HDSEL and HDUS forest units cover 175,500 ha and 52,000 ha, respectively. Together, the two forest units account for 47% of the area[1].

3.2 The Species Composition of the Tolerant Hardwood Forest

In both of the tolerant hardwood forest units, sugar maple, beech and hemlock are the most shade-tolerant species. Yellow birch, red maple, and black cherry have a moderate degree of shade tolerance, and red oak is even less shade tolerant; these species are often described as mid-tolerant species. Other mid-tolerants include basswood, American elm and white ash. However, because mid-tolerant species are rarely more than minor components of any stand, the forest inventory does a poor job of distinguishing them. The forest inventory identifies 556 ha as being in the ash working group, 21,287 ha in yellow birch, 16,674 ha of red oak, 9,391 ha as red maple, and 1,700 has other hardwoods, which will include beech, cherry, and other minor species. These areas are based on the entire Park productive forest, which is 618,518 ha, and so the yellow birch occupies 3.4%, the red oak and red maple 2.7% and 1.5%, respectively, and the other two just fractions of one percent. Red oak tends to prefer drier sites and is often mixed with pine, as both of these species are favoured by fire (Guyette and Day 1995). Black ash and red maple are more common on wet sites.

As every forest manager does, the AFA has re-classified the inventory on the basis of management approach, and the hardwood species mentioned above are included within the tolerant hardwood forest unit, except for oak, which has its own forest unit (ORUS). Figure 1 shows that ORUS has a small but meaningful amount of area.

The majority of the stands in the managed tolerant hardwood forest (i.e., 85% by area) are dominated by sugar (hard) maple, with yellow birch representing the largest portion of the remaining area. The relative abundance of the key species within the entire Park tolerant hardwood forest is virtually identical to that in the managed portion.

3.3 Eastern Hemlock

Eastern hemlock is the key conifer species in Ecodistrict 5E-9. It is present in the forest as scattered individuals, in groves within a matrix of hardwood, and as pure or almost pure stands (Martin 1959; Vasiliauskas 1995). It tends to be found on north-facing ridges and upper slopes on coarser soils, and prefers moist, cool conditions (Cumming 2005). Martin (1959) observed that many mature hemlock stands had an understory of eastern white cedar, however he remarked that in many stands the cedar had been cut for telephone poles (e.g. see Bier et al. 1930). Hemlock is also frequently associated with yellow birch. Stands which contain at least 40% hemlock are classed in the HeSEL

[1] Table FMP-9 in the 2005 FMP.

forest unit (See Figure 1), which occupies 6.5% of the total productive forested land in the Park, or about 40,000 hectares (Cumming 2005).

Hemlock forests also contain white and red pine, white and red spruce, and balsam fir on upland sites and black spruce, tamarack, and eastern white cedar on lowland sites. The prevalence of balsam fir and spruce in riparian areas has been observed by numerous authors (Bier et al. 1930; Martin 1959; Rothfels 2002), and Mihell (pers. comm.) observed hemlock as a frequent component of riparian forests.

Hemlock's importance is ecological, rather than commercial. It was heavily cut in the early decades of the twentieth century to support the tanning industry, and is also favoured by the Mennonites in south-western Ontario for barn construction. Campbell Soup Co. also used it as a medium for growing mushrooms, since hemlock is moisture resistant and imparts no flavour to the mushrooms (Dowdell Murray pers. comm.). Hemlock was also heavily cut in the late 1950's and early 1960's to provide props for the construction of Toronto's subway system and then in the 1970's during the building of the Hydro Quebec James Bay power facilities. However, at present, there is little commercial interest in it and little is cut (Wilton 1987, Anderson and Gordon 1994); when it is harvested, it is cut under the group selection system. The 2005 FMP also notes that patches of hemlock in other forest units as small as 0.04 ha will be managed using the group selection method.

Hemlock trees are also long-lived, with mature trees frequently older than 250 years and sometimes living as long as 500 – 600 years (Vasiliauskas 1995). Eastern hemlock is also very shade tolerant, probably more so than maple in the Park (Vasiliauskas 1995), and is well-known for its ability to respond when canopy openings are created after surviving for more than a century suppressed in the understory (Martin 1959; Vasiliauskas 1995). As a result, there is often little correlation between the size and age of a hemlock tree.

Hemlock's shade tolerance means that it often has a deep crown. This feature, combined with its generally stiff branching habit, means that it catches and holds a lot of snow, so that snow levels are generally quite low underneath hemlock stands compared to the surrounding forest. It is also a favourite browse species of deer and moose. These characteristics make it the most important winter shelter species for deer. Theberge (1990) also contended that it was the single most important winter species for moose and the 2005 Forest Management Plan for the Park (Cumming 2005) identified Blackburnian warbler, northern flying squirrel, red-shouldered hawk and barred owl as species with affinity for hemlock.

One management challenge in the Park is that about 85% of the hemlock stands are mature or very mature, and there are very few hemlock trees younger than 100 years of age (Raven 1996). While hemlock seed has no difficulty germinating on the appropriate seedbed, it is a favourite browse species and a long history of heavy browsing by deer and moose has prevented trees from growing into maturity (Vasiliauskas 1995). Vasiliauskas (1995) found that the age gap in hemlock in deer wintering areas was consistent with high deer populations from roughly 1890 to 1960. As discussed in [the wildlife chapter], after that time the deer population began to fall in the Park, and moose began to replace them. The rise in moose populations shifted the most intensive browse pressure to upland areas where moose prefer to winter. Rotted logs also provide very good seedbeds for hemlock (Marx and Walters 2006), and forests with active logging operations tend to have few large logs to act as seedbeds. This factor may also contribute to the renewal problems.

In the absence of browsing, some experts contend that hemlock would gradually dominate the tolerant hardwoods on most sites, by virtue of its higher tolerance for shade and its longer lifespan (Vasiliauskas 1995). Other authors suggest that hemlock and tolerant hardwoods will most often remain co-dominants (Anderson and Gordon 1994). In any event, in the presence of high levels of browsing, and perhaps also logging operations that reduce the quantity of large rotten logs in the forest, sugar maple readily outcompetes hemlock since it is less favoured by deer and moose, and sprouts prolifically after browsing. Thus, it is probable that the current conditions within the Park are leading to a long-term reduction in hemlock abundance and blocking the last leg of the successional process. Forest managers have had little success regenerating hemlock and there has been a call to stop harvesting it altogether in the Park (Theberge 1990). In response to the dearth of hemlock recruitment to the canopy, the AFA has developed a hemlock management strategy which focuses on hemlock retention in tolerant hardwood stands and low levels of harvesting within hemlock stands to optimize regeneration conditions. This issue has been studied numerous times over the years and is well-known, if also difficult to solve.

3.4 Tolerant Hardwood Management

Stands with a superior level of stem quality and a high proportion of sugar maple, beech or other tolerant or mid-tolerant hardwood species are managed under the selection system. In Algonquin Park, harvest entries under the selection system are made at intervals of approximately 25 years, depending on site quality. Prior to each harvest, a tree marker walks the stand and uses spray paint to mark individual trees for removal, selecting the removal trees. Top candidates for removal include poor quality trees, trees that are expected to die during the next harvest interval, mature crop trees, and a selection of trees such that the residual stand will move closer to a target size class structure and species composition.

Where there are patches within a tolerant hardwood stand that have moderate to high proportions of mid-tolerant species in them, the tree marker will mark a higher percentage of trees for removal, which will allow more light to penetrate the stand and can lead to greater proportions of mid-tolerant species in the new regeneration. However, mid-tolerants require more than just additional light – exposure of mineral soil or the presence of downed rotten logs, a high proportion of suitable seed trees retained in the canopy, and coincidence of the harvest with a good seed year are all factors that will increase the chance of securing abundant mid-tolerant renewal. The 2005 FMP notes that scarification may be necessary if the post-harvest site does not have sufficient mineral soil exposure.

The hemlock forest unit is managed under a variant of the selection system known as group selection. Under this approach, small patches of trees are harvested, as was described above in circumstances where there is a significant proportion of mid-tolerant species in a patch. Patch cutting in hemlock is anticipated to create the necessary conditions to provide good renewal conditions for hemlock. For many years, the management challenge with hemlock has been to have the young hemlock trees grow past the stage where they are vulnerable to moose and deer browsing – this issue is discussed in greater detail below.

The HDUS forest unit contains tolerant hardwood stands that are not well suited for the use of the selection system, and so are managed instead under the uniform shelterwood system. In hardwoods, the uniform shelterwood method involves two passes. In the first pass, the low quality and undesirable stems are removed, and the canopy is thinned to allow sufficient light to penetrate to the forest floor to support regeneration. About 60% of the stand volume is removed in the first pass. The residual stand should have a high proportion of high-quality stems (which likely indicate superior genotypes) of desirable species, well-spaced and of a moderate density. A twenty-year period is allowed to elapse, during which time natural regeneration establishes itself. When the remaining canopy is removed in the second pass, the new stand is approximately 20 years old.

Tolerant hardwood stands that are managed under the shelterwood system are those that:

- have too high a percentage of low-quality stems to be suited to the selection system;
- have a stocking level of less than 40%;
- are hard maple or other hardwood stands younger than 80 years of age
- are on site class 3; and
- are in the ash, yellow birch, red (soft) maple forest units.

Stands that have a lot of low-quality stems, a low stocking level, or are younger than 80 years of age are managed under the shelterwood system essentially because it is a way of starting over and producing a new stand that can eventually be managed under the selection system to produce a high proportion of high quality trees in it. Site class three lands are not rich enough to support a high-quality forest, and stands in the ash, yellow birch and red maple working groups have a high proportion of species that are mid-tolerant and cannot be regenerated in any great numbers under the single-tree selection system. Thus, shelterwood is the appropriate system for renewing these stands to their dominant species type.

In addition, the two-cut shelterwood system is applied to the red oak and mixedwood forest units since the approach will maintain red oak, in the former case, and the diversity within the mixedwood stands, in the latter case.

In addition to the two harvest systems, there are several stand tending and improvement cuts that may be undertaken. The 2005 FMP states that polewood stands, which are generally between 80 and 100 years of age, may be commercially thinned, to manage the density and spacing, adjust the species composition, and remove defective trees. The harvested material is generally pulpwood or fuelwood. In older stands, an improvement cutting may be undertaken to remove defective trees and undesirable species – the difference between an improvement harvest and a regular harvest is that the value of the products harvested during the improvement cutting does not cover the cost of harvest – usually some financial incentive is required to make improvement cutting happen. Lastly, a release or pre-commercial thinning treatment may be applied to young stands to improve density and species composition – since these operations generate few if any forest products, they must be paid for from stand renewal and tending budgets.

3.5 The Disturbance Regime in the Tolerant Hardwood Forest

The dominance of the tolerant hardwoods on the Park's west side is due to their fondness for the types of site and soil conditions found there, their resistance to stand-replacing disturbances, and the ability of the main species to reproduce in shade, which is prevalent in these little-disturbed forests. The resistance of tolerant hardwoods to fire is striking, and reflects the moist site conditions (Quinby 2004) as well as the poor flammability of the trees themselves. Destructive, canopy-killing fires have been very rare in tolerant hardwoods (Chandler et al. 1983; Vasiliauskas 1995) and in fact, catastrophic stand destruction may occur only every few thousand years on a site [see review in Quinn (2004) and especially Frelich and Lorimer (1991)]. Rothfels (2002) summarized the estimated pre-settlement fire return time as being between 800 – 1400 years in the hemlock-white pine-northern hardwood forests. Similarly, the 2005 FMP used a natural disturbance cycle length of 857 years for the HwdUS (and presumably HwdSel) forest unit, 202 years for the MWUS, and 250 years for the OrUS forest unit.[2] These forests are so resistant to fire that they have been described as the "asbestos forest" (Rothfels 2002), and the presettlement landscape of east-central North America was essentially a vast mantle of largely mature hardwood-dominated forest.

So what does drive change in these forests? There are two primary natural disturbance patterns; the first, and predominant, is "gap phase" replacement; the death of individual trees from ageing, wind or insects. Tree mortality occurs continuously and researchers estimate that perhaps 8% of the presettlement forest canopy was in small openings resulting from individual fallen trees at any given time (for a more detailed discussion of this see Quinn 2004). Treefall gaps are small, however, and allow little light penetration, giving young tolerant hardwoods the advantage. Creation of these gaps has relatively little ecological impact on species composition.

The second agent of change is much more dramatic; violent storms of two types; 1) severe thunderstorms or "downbursts" that blow down trees, creating openings that are typically 20 -30 ha, and 2) tornados, which can create openings of several thousand hectares. These events are surprisingly common. Cumming (2005) reported that severe wind storms flattened parts of the Park in 1964, 1972, 1973, 1983 and 1999. The most extensive damage was caused by the 1973 storm, which blew down 20 square kilometres of forest between Wilkes and Biggar Lakes. Most recently, there was in 2006 a very widespread wind storm which heavily damaged tens of thousands of hectares of pine in the region.

Because mid-tolerant species generally require openings about the size of a basketball court or larger to regenerate, the larger openings created by storms provide important opportunities for these species to regenerate. Thus, these larger blowdowns have great ecological importance in maintaining species diversity. On very disturbed sites, the mid-tolerants may compete with white birch and poplar, which are pioneer species adapted to regenerating in bright sunny openings. They are often the initial hardwood species to grow in large recently cleared or disturbed areas.

[2] In contrast, in the 2005 management plan, a fire return time of 9,901 years was used, based on disturbance data from 1976 to 1988.

The effects of individual treefall and storms has created a forest that, while largely intact, is pock-marked by frequent openings. The relationship between the number of openings and their size is described by the negative exponential frequency distribution, that is, there are very many small gaps and fewer larger gaps. There is no evident connection between human activity and the frequency or intensity of wind events, and so it is likely that wind events continue to have much the same impact on the forest today as they did prior to European settlement.

4 The Historic Forest

One of the recent major themes of forest management, not just in Algonquin Park, but throughout Canada, is to shift the character of the forest back towards what it had been prior to European settlement. A key approach towards realizing this goal has been to adjust forest management approaches and practices so that they emulate natural disturbance regimes. The premise underlying this approach is that the ecosystem was well-adapted to the processes that were active in it, and that if these processes, or reasonable facsimiles, can be maintained by a combination of management and natural factors, then this will produce a healthy, resilient forest that is likely to be as biodiverse as the pre-settlement forest (Drever et al. 2006; Bergeron et al. 2004). The idea seems, in theory, to be sound, however putting it into practice is more complicated than one might think.[3] For example, one of the challenges is that the pre-settlement forest was not static – it was subject to perturbations due to sudden changes caused by disturbance (e.g. fire, windstorms), as well as more gradual shifts due to succession. However, one would expect to find that over large areas of the same forest type, the overall average condition would be less variable than the extremes that would occur at the level of an individual stand. In other words, prior to industrial influence, on the landscape scale, the key attributes of ecosystems characterized by relatively frequent stand-replacing disturbance could be expected to be stable, exhibiting variation only within a given range.

To successfully implement this approach, the manager needs to have a reasonable notion of what the pre-industrial forest (PIF) was like. This requirement has given impetus to research on the character of historical forests and historical disturbance regimes. As it happens, there has been considerable research undertaken within Algonquin Park on prior forest condition.

Assessments of the PIF use a variety of methods, including physical sampling, record assessment, and modeling. The most common physical sampling techniques include sampling the pollen and charcoal depositions in lake sediments and tree ring analysis (dendrochronology). The records of early surveyors provide the most systematic early records of forests, and the records of exploration parties and other contemporaries are also useful. Modelling approaches generally start with a forest at a given time (such as a forest as inventoried in the 1950's, for example) and simulate how natural processes would change the forest over time. Each of these approaches have their strengths and weaknesses, and they are applicable to differing spatial scales. For example, Rothfels (2002) points out that insect-pollinated tree species, such as maples, fir and elm, tend to produce less windborne pollen and so are under-represented in archeological samples,

[3] A fundamental objection to the entire premise is that climate change is creating conditions that are different from those of the pre-settlement era, and so there is limited value in moving in the direction of recreating a past that is incompatible with the future. But more on this later.

whereas pine, hemlock, and oak produce large amounts of windborne pollen and are prone to over-estimation. However, taken together, these methodologies can be complementary.

A major caveat about the utility of the management approach is its implicit assumption that none of the broader environmental factors have changed over the last centuries. Several reasons why this assumption is not valid readily come to mind: changes in climate, the impact of acid precipitation on soil characteristics, and changes in the role of pests and significant reductions in species such as the American elm and larch. There are few obvious means for the forest manager to counter these shifts – the question then becomes how they will impact the implementation of the broad management strategy.

The first section reviews PIF analyses that cover the entire Park. These analyses are not terribly helpful for this project, since the disturbance regime on the east side of the Park is much more turbulent than that on the west side. Averaging them together will lead to an overestimate of the level of disturbance on the west side of the Park. For this reason, we will focus on analyses centred on the tolerant hardwood forest.

4.1 Whole Park Analyses

Pinto et al. (2006) used the original land surveyor records from 42% of the township boundary lines in the Park to estimate the abundance of the main tree species and compare it with the current abundance. The early surveyors who laid out the townships were instructed to describe the tree cover encountered, identify the distance traversed through each stand type on each township boundary line, and list tree species present in the stand in order of their abundance.

The surveys were conducted between 1858 and 1893, and the surveyor notes for some boundary lines indicated that logging had already occurred. Pinto et al. (2006) reported that records mentioning logging applied to only 0.6% of the forest cover surveyed. It is striking that a very high percentage of burned land was encountered – seven percent was described as recent, clean burn, almost four percent as recent partial burn, and 19% as old burn. While this may seem to be at odds with the apparent lack of logging encountered, we note that 1864 and 1875 were drought years that seem to have been associated with widespread fires (Cywnar 1977). Another three percent of the area was windthrow.

Pinto et al. (2006) found that, compared to their historic abundance, maple, beech, oak, hemlock, spruce, and poplar showed statistically significant increases in the current forest, while pine showed a moderate decrease and larch and cedar have undergone significant decreases. Maple showed the greatest absolute increase, rising from 10.42 in the survey data to 27.12 in the 2005 inventory (the numerical values equal the percentage of the average boundary line along which the species would be encountered). In other words, the abundance of maple almost tripled. An even greater proportionate increase was recorded for oak, which rose from 0.65 to 3.02. Going the other way, larch declined from 4.00 to 0.23, which is mostly attributable to the introduction of the larch sawfly in the early twentieth century.

Rothfels (2002) reviewed a number of analyses of surveyors' notes and considered it likely that logging had some influence on the results. One of the studies (Leadbitter

2002) reported that six of ten randomly selected township surveys explicitly referred to logging. Rothfels also pointed out that logging had reached the Park by 1836, if not earlier, and a reconstructed map of cutover sites shows logging in the interior of the Park by 1866, well before the majority of the Park was surveyed. Thus it seems likely that logging would have affected more of the boundaries than the survey notes indicate.

Leadbitter's review of Park surveyor records (2002) reached some conclusions that were consistent with Pinto et al. (2006). Leadbitter found that the maple working group has increased significantly since presettlement, as have spruce and yellow birch. In contrast to Pinto et al. (2006), Leadbitter found that poplar, white birch and "other hardwoods" working groups have decreased significantly since presettlement; while the pines show no significant change. Rothfels (2002) comments that Leadbitter's findings can be explained by assuming that the fire regime had already been altered by logging at the time of the surveys, so fire-vulnerable species like maple, spruce and yellow birch would be at a low level, while intolerant, disturbance-dependent species like poplar and white birch would be at a population high when compared with the relatively older, less-disturbed forest of today. Pine, already depleted by logging at the time of surveying, might not reveal a population change in these studies, and yet still be under-represented in our present forest.

4.2 Analyses of the Historic Tolerant Hardwood Forest

Quinn (2004) published one of the most recent descriptions of the pre-settlement forest of the west side of Algonquin Park, and compared it with the current forest. He used as his sources analyses of surveyors' notes, forest history, and information from remnants of the original forest. The tolerant hardwood forest has long been dominated by sugar maple, with beech and hemlock also common. A few accounts suggested that hemlock might have been dominant, but the evidence cited by Quinn (2004), as well as the work by Pinto et al. (2006), suggest that, at the landscape level, maple was more abundant than hemlock. White pine and yellow birch were also identified as common species.

As described above, wind tends to be the principle disturbance agent in this forest, creating small gaps as individual trees, or small clumps of trees, are blown down, larger disturbances are caused by downbursts (or microbursts), and tornados create very large blowdowns. Fire was also present in these stands, most often as low-intensity burns that removed the litter and duff, and killed the younger, thin barked woody vegetation. These fires stimulated hemlock and oak renewal. More intense, destructive fires were uncommon, a finding supported by Weeks (cited in Rothfels 2002), who could not find clear evidence of past fire in the sediments of a lake surrounded by a hardwood forest.

In general, the rate of disturbance was lower on the west side of the Park than on the east side, simply because destructive wind events are less common on the west than fires were on the east. As a result, there was a more continuous canopy of mature trees. Quinn (2004) reported that the annual loss of canopy trees has been estimated at between 0.5% and 1.0% by many authors. For example, Frelich and Lorimer (1991) estimated that the average decadal rate of canopy mortality was 5.7 – 6.9% in a northern hardwood-hemlock forest in Michigan; the average residence time of canopy trees was between 145 and 175 years. This provides an indication of the prevalence of single tree gap dynamics.

It is probable (although this is speculative) that about 2% of the natural forest was in large openings. Researchers in the northeastern U. S. have created a model that suggests that about half of the presettlement hardwood forest was in old growth, 13% in early succession (< 24 years old), and the rest in transitional stages but mostly mature (Frelich and Lorimer 1991). These figures are probably approximately accurate for the west side of Algonquin Park, since the forest type is similar to that in NE United States.

While the stands on the west side of the Park have always tended to be strongly dominated by tolerant hardwoods, many of these stands also scattered very large pine. Thompson et al. (2006) counted old stumps of logged pine in current tolerant hardwood forests in the Park, and found densities as high as 10.9 white pine/ha. Thompson et al. (2006) also sampled the white pine in the unlogged Big Crow Wilderness Reserve, located with Algonquin Park, and found an average density of 8.4 stems/ha in what is primarily a maple-beech-hemlock forest. Based on these survey results, Thompson et al. (2006) felt that typical hardwood-dominated stands on the west side of the Park would have had 3 – 8 large white pine/ha. This compares with the current average density of white pine of 0.2 pines/ha in tolerant hardwood stands that have been logged. Quinn (2004) also observed that tolerant hardwood stands today had fewer white pine in them than they formerly did.

Moreover, the pine in today's stands is small in comparison with historical norms. Thompson et al. (2006) reported that the 35 living and recently dead white pine in the Big Crow Reserve had an average diameter of 97 cm, with the largest living tree being 92 cm. The largest dead tree was a stump that was 117 cm, in dbh. There were no red or white pine smaller than 25 cm dbh within the Reserve. In contrast, the current average diameter of white pine in Algonquin is 44 cm, versus an average of 73 cm measured on historical trees (Thompson et al. 2006).

Thompson et al. (2006) reached a similar finding for current boreal mixedwood stands. Historically, these stands had white pine densities of 5 – 7 trees/ha, based on stump counts, versus current levels of 1 pine/ha in stands that have been logged. Reductions in white pine abundance of similar magnitude have been reported for other eastern North American forests (Thompson et al. 2006).

How did white pine enter and persist in these forests? The key factors are the longevity of pine and its semi-tolerant nature, which enables it to compete well with other species in moderate and large sized gaps. Once even a few pine became established in a tolerant hardwood forest, they would live long enough to propagate in the moderate sized openings that periodically opened up due to wind events or the occasional fire.

4.3 Comparing the Present West Side Forest with Pre-settlement Conditions

Most people probably imagine that the present west side Algonquin forest looks much the pre-settlement forest, and indeed the present forest retains many characteristics of the early forest. Most notably, today's forests are still dominated by tolerant hardwood and eastern hemlock, and have little medium or large-scale disturbance in them. Moreover, the selection and shelterwood management systems are designed to maintain these primary stand types. The forests also have a range of tree sizes within them, and contain some large trees.

However, despite the broad similarities, there are a number of significant differences between the current forest and the pre-settlement forest. Perhaps most notably, there have been many changes in the relative abundance of individual species. Compared to the pre-settlement forest, there have been significant reductions in:

- Red spruce (poorly adapted to post-logging fire and heavily cut with no renewal);
- Larch (reduced throughout much of North America following the introduction of the larch sawfly);
- American elm (reduced following the introduction of the Dutch elm disease);
- Yellow birch (with the exception of Leadbitter et al.'s (2002) contrary conclusion, there is widespread agreement that yellow birch has declined due to a combination of logging pressure in the 1930's – 1950's, die-back, fire suppression, and intensive browsing by deer of regeneration);
- Eastern hemlock (reduced due to a combination of logging pressure around the turn of the century and again in the 1960's, as well as intensive browsing by moose and deer of regeneration);
- Cedar (from Cywnar (1978), Leadbitter et al. (2002) and Pinto et al. (2006); perhaps due to harvesting and browsing by deer); and
- White pine (early logging eliminated the seed source, as documented in Thompson et al 2006).

We note that the 2005 FMP reviewed analyses of early surveyor records and it echoes many of these conclusions. The 2005 FMP also indicates that red oak and jack pine have declined, noting that jack pine is at the southern limit of its range in Algonquin Park (Cumming 2005).

More controversially, the FMP also states that the poplar and white birch ecosite now tends to have red and white pine on it, due to management policies that favour the pines and a general lack of natural stand-removing disturbances. Pinto et al (2006) calculated that the percentage length of surveyor lines through poplar stands declined from roughly 19% when the surveys were conducted to an estimated 16% in 2005. This decline was judged to be insignificant. The surveyors tended to label both yellow and white birch as just "birch" in their notes, and so it is difficult to draw any conclusions regarding changes in the relative abundance of white birch using this methodology.

We also suspect that, in addition to yellow birch, other mid-tolerant hardwood species such as basswood, black cherry, and white ash (which reach the northern limit of their range in the Park) are less common in today's forest than they were previously. There is little quantitative evidence for this, since there is little information on these species in general. The forest inventory does not do a good job at capturing the abundance of mid-tolerant species, since they rarely form a major stand component. In fact, by virtue of their low densities in many stands, the inventory may greatly underestimate their presence.

Leadbitter et al. (2002) was the only study we found that examined the mid-tolerants; they found a lower occurrence of basswood, black cherry, elm, and oak in 1990 compared with the forest at the time of the township line surveys. On the other hand, Leadbitter et al. (2002) found that white ash and yellow birch increased. The yellow birch finding is inconsistent with results from Pinto et al. (2006), Vasiliauskas (1995), Smith (1997, cited in Rothfels 2002), and the 2005 FMP. Rothfels (2002) and

Thompson et al. (2006) both suggest that logging had already taken place prior to the surveys in many of the townships Leadbitter reviewed, hence the survey records were not representative of the pre-industrial condition.

The main reason we feel that many mid-tolerant species have declined in abundance is that the widespread application of single tree selection harvesting, as well as the suppression of fire, discriminate against these species by creating few mid-sized or larger openings. The openings created by single tree selection are too small to provide enough light for these species to regenerate, strongly favouring the tolerant species (Webster and Jensen 2007). This was also the conclusion reached by Leadbitter et al. (2002). Yellow birch faces additional pressure, since it is a preferred browse species for deer and moose. Wilton (pers. comm.) also pointed out that there was a philosophy among foresters in the 1960's and 1970's that sought to remove miscellaneous species in order to simplify the species composition of stands. This was well illustrated in the Lake Traverse deer yard, where the forester of the day removed the remaining white pine (cover) to "start over".

It is clear that sugar maple has increased significantly in abundance, and beech may have as well. Early forest studies were divided on the comparative abundance of beech; Cwynar (1978) and Leadbitter et al. (2002) found a decrease while Pinto et al. (2006) reported an increase. There is no readily apparent reason why beech should decrease, and sugar maple increase, since the two species generally prefer similar conditions. The one possible reason for a decrease may be that beech is a less desirable wood, commercially, than maple, and perhaps it was somewhat discriminated against for a period. However, it is recognized also as a valuable mast producer, and the 2005 FMP provides direction to maintain it.

On the other hand, red spruce, which is at the western edge of its range in Algonquin Park, was not even authenticated as being present in Ontario until the 1950's (Gordon 1957). Part of the reason for this is that it is hard to distinguish from white spruce. Most importantly, Anderson and Gordon (1990) cited a number of examples where very good red spruce stands in Ontario were cut in the 1950's and regenerated to a mix of other species (there was scant regard given to spending money on silviculture then). In other words, this seems to be one example where poor forest practices have been the major contributor to the current scarcity of this species.

The contention that white and red pine have declined in Algonquin Park since pre-settlement times needs some further explanation. As described above, Thompson et al. (2006) reported that white pine had declined in both the tolerant hardwood and mixedwood type forests, which make up more than 80% of the Park's forest area. Thompson et al. (2006) also analysed the forest types currently growing on a range of soil types and found that many of the soils that typically support pine dominated forests had other forest types present. As a result, they estimated that white pine dominated forests may have declined by as much as 40%. We think that the decline in red pine has may have been greater, since pine management in the Park is skewed towards the use of the shelterwood system which encourages white pine rather than red pine. In addition, red pine is less tolerant than white pine and so requires larger and more intensive disturbances to regenerate. Fire suppression has, in our view, only added to the reduction in red pine.

The comparison of the historic west side forest with the present day forest focused on relative species abundance. It is probably also the case that there was more area lost to fire in the west portion of the Park in the past than there has been in recent decades, due primarily to improve fire detection and suppression. This may mean that there was a greater proportion of the forest in younger age classes historically, but this is mainly conjecture since little evidence was discovered while researching this paper.

One other difference appears to be the impairment of the soil chemistry due to acid precipitation and nitrogen deposition. Although the acidity of precipitation in the region containing the Park has declined since the 1970's and 1980's, the very slow rate of weathering of the bedrock and timber harvesting have combined to prevent a rebound in the concentration of base cations, such as calcium, in the soils. As a result, the soils remain more acidic than they were previously (Watmough et al 2008). It is our understanding that forest growth rates in the Muskoka – Haliburton region (which includes Algonquin Park) have been below projections, and while there are likely numerous factors contributing to this, it is quite possible that impaired soil productivity is one of them.

5 The Historic Wildlife Community and Recent Changes

Based on the above descriptions, and building on the analyses of Quinn (2004), it is possible to infer some qualities of the historic wildlife community. As with the description of the historic forest, the most tractable way to describe the historic wildlife community is to use today's conditions as a basis for comparison. Because wildlife is very strongly affected by habitat, it is worth briefly summarizing the key differences in the historic forest compared to the present forest. The historic forest:

- had a greater diversity of vegetation communities; prior to human influence, the forests of the Park likely had greater abundances of several conifer species, including red spruce, larch, eastern hemlock, white cedar, and white pine;
- likely had greater representation of mid-tolerants such as yellow birch, basswood, black cherry, American elm, and white ash;
- had less sugar maple and beech;
- had a larger area in old-forest conditions;
- had more old trees;
- had more mid to large-sized gaps in the canopy; and
- had more standing and fallen woody debris (due to the greater area in old forest).

In sum, therefore, it is likely that the historic forest was more diverse, in terms of the relative contribution of tree species to the overall forest community, the distribution of forest and tree ages, and the presence of mid and large sized gaps in the forest.

There have been a host of changes in Algonquin Park not associated with the impact of management on the Park's forest communities. These changes must be taken into account in contrasting the present and historic wildlife communities to put the role of vegetation changes in an appropriate context. These changes include:

- **Roads -** The Park is extensively roaded (there are approximately 6,000 km of road in the Park). Human access has affected wildlife in a number of ways, including: road mortality of some species, creation of barriers to movement, and facilitating legal (and illegal) hunting.

- **Development outside the Park –** The abundance of some of the Park's species (most obviously migratory birds) has been strongly affected by changes which occurred far outside the Park. Loss of overwintering habitat in Central and South America has contributed to striking declines in many species of birds which breed in the Park (Terborgh 1989, Rich et al. 2004).
- **Climate –** Although many impacts of climate change have yet to become fully apparent, some are already occurring. For example, Waite and Strickland (2006) suggested that Park populations of gray jays are declining because warmer winter weather is causing their cached food to rot. In addition, there are several reports of changes in species distributions due to ongoing climate changes (Flannery 2005).
- **Pollution -** Acid rain falling on the Park from sources in Canada and the United States has acidified many of the Park's lakes (Scott 1989). Although there have been significant reductions in sulphur dioxide (SO_2) emissions (which lead to acid rain), there has been little recovery in soil pH (Watmough et al 2008) as even the reduced rates of acid deposition are thought to exceed the critical load capacities of most areas in eastern North America. The combination of acid precipitation and logging are considered to lead to a continued loss of soil calcium. In addition, there is an elevated rate of nitrogen deposition that increases the rate of base cation leaching and may have countered the benefits associated with reduced acid precipitation (Watmough and Dillion 2003).

Quinn (2004, 2005) discussed changes in the Park's wildlife community over time, focusing on ungulates, wolves and beaver. He and others (e.g. Wilton 1987, Runge and Theberge 1974) noted that deer were very likely not found in the Algonquin area prior to the advent of industrial development. The opening up of the forest caused by the widespread logging, which took place through much of the 1800's, created good habitat conditions for deer (primarily an abundance of browse in regenerating forests) and led to expansion of deer into the Park. By the 1920's, the deer population in the Park was estimated in the "tens of thousands" (Robinson 1933, in Quinn 2005). White-tailed deer were an important component of the Park through subsequent decades, although populations fluctuated, largely because of winter conditions. Following declines in the 1960's and 70's, the deer population has remained low, at least partly because deer yards were poorly managed; Wilton (1987) notes that between 1952 and 1971 logging affected between 10 and 79% of the available winter habitat within the six major deer yards in the Park. Deer are now somewhat common in the summer, but leave the Park to overwinter. Quinn (2005) notes that deer are "essentially absent from the present day Algonquin landscape in winter."

Although there are few historic records prior to the late 1800's, moose were likely present in the Park much earlier as the natural range of moose extended through the area. Quinn (2005) cites several historic (late 1800's) records discussing the presence of moose in the Park. Moose were apparently scarce in the 1920's when deer populations were at a peak. Throughout much of the 20th century moose populations have fluctuated in opposition to deer numbers. There is considerable evidence that moose populations are adversely affected by prevalence of meningal worm, (frequently referred to as *P. tenuis* as short for its much longer scientific name *Parelaphostrongylus tenuis*, a parasite carried imperviously by deer, but which is fatal to moose. It is thought that *P. tenuis* – deer-moose dynamics limited moose populations throughout the period when deer were abundant in the Park (Wilton 1987, Quinn 2002).

Beaver were likely more abundant historically in the Park than they are at present. This is so for several reasons: The younger-than-“natural” forests which are common in the managed portion of the Park are less prone to gap disturbances (Quinn 2005) of the sort that open the canopy and create good sources of beaver food. In addition, successful suppression has effectively eliminated fire as a source of disturbance in the Park, also reducing the abundance of early-successional habitat. Thirdly, in the Park, as in most managed forests in Ontario, there is reluctance to harvest to the shore of waterbodies. As a result, through succession, these areas are reverting to conifer-dominated vegetation communities, which have little forage to offer beavers. Fryxell (2001) notes that beaver have declined sharply in the Park since the 1970’s.

Wolves are an iconic symbol of Algonquin Park. Their abundance on the landscape is inextricably linked with that of their prey. Quinn (2002, 2004) has described the interrelation of wolves and their main prey (deer, beaver, and moose) in the context of changes in the Park’s forest. Algonquin’s wolves are small and more adept at killing deer than moose (Pimlott et al. 1969). Moose, when consumed, is most likely scavenged (Forbes and Theberge 1992). Wolves in the Park undoubtedly benefited from the forest harvesting of the 1800’s which resulted in high deer populations. In the logic of the times, wolves were “controlled” in attempts to limit their impacts on deer populations. The Crown Lands Commissioner noted in 1895 that wolves in the Park “are too numerous for the good of the deer. We are making every effort this year to kill them and shall continue it during the winter.” Wolf control continued in one form or another (poisoning, snaring, hunting) through to the 1960’s. According to Quinn (2004), as the changes in habitat affected the relative abundances of deer, moose, and beaver, what was a wolf-deer system during much of the 1900’s has become a wolf-moose-beaver system, with wolves preying on beaver, scavenging moose, and opportunistically taking deer. However, Quinn concluded that the net changes in prey conditions are not so great as to have had a tremendous impact on wolves. Deer, absent from the historic forest, are nearly absent in winter today and beaver and moose, although likely more numerous historically, were not so much more abundant as to leave the Park with significantly fewer wolves.

As mentioned earlier, not all of the changes in the Park’s wildlife communities are simply attributable to forest management. Perhaps nothing exemplifies this more than the passenger pigeon, discussed in detail below. Landscape fragmentation, along with rampant slaughter and overharvest are considered to have caused its demise; it is unlikely that harvesting of hardwoods in the Algonquin area on the scale at which occurred in the mid-1800’s would have played a role in the species’ extinction.

There is tantalizing evidence, summarized by Strickland (2007), that Algonquin was historically inhabited by caribou. Anecdotal evidence exists attesting to their presence in the area into the early 1900’s. We note that caribou are at least as susceptible as moose to *P. tenuis* (Pitt and Jordan 2004, Chowns 2003), so one wonders whether the chain of events which led to deer population explosions in the area following logging may in some way be related to caribou’s decline. This is speculation on our part, however. Other theories related to caribou decline across their present range include factors such as overhunting, landscape fragmentation, increase in predation (indirectly related to forest harvesting), and climate change (Racey et al. 1991, Schaefer 2003).

There is also evidence that elk occurred in the Park area prior to European settlement. Quinn (2004) thinks they were “almost certainly present”, but likely were uncommon because of their preference for open, grassy ranges. Strickland (1976) described evidence of their presence in the Park area, and noted that their disappearance coincided with logging and forest clearing for farms. However, given that elk have a preference for clearings and feed on a wide variety of vegetation, it is unclear why these land use changes should have resulted in their extirpation. Strickland (1976) hypothesized that brainworm may have also been responsible for their disappearance.

Two other changes of note are the increase in raccoons and apparent range expansion of southern flying squirrels. We know of no information on historical population levels of raccoons (although we note that they were at most, uncommon, and perhaps did not exist in the Park). Sanderson (1987) noted that raccoons increased by approximately 15 to 20-fold since the early 1900's owing mostly to changes in land use in Ontario. Steinberg (pers. comm.) noted that there is "likely" a healthy raccoon population throughout the interior of the Park, and probably higher populations close to campgrounds and access points. Given that raccoons feed voraciously on turtle eggs, young birds and bird eggs, amphibians, crayfish, etc, it is possible that the presence of raccoon's in the Park has had impacts on the population levels of other species. There is also evidence, based on MNR research, of immigration of southern flying squirrels into the Park (Steinberg pers. comm.) but it is not clear if they are permanently established.

One striking change in the Park's fauna relates to songbird populations. There is overwhelming evidence of continental declines in migratory songbird populations (Terborgh 1989, Böhning-Gaese et al. 1993, Sauer et al. 2008), with habitat loss on overwintering grounds in Central and South America, fragmentation of breeding areas in southern Canada and the continental United States, and migration mortality, identified as key causal agents. At the same time, there is also considerable evidence attesting to the role of forest management in Canada contributing to habitat loss for some species (Schmiegelow and Mönkkönen 2002, Wedeles and Donnelly 2004).

In section 4.3 we note reduced present relative abundance of several tree species compared to the presettlement forest. Although there has been little specific work relating these changes to changes in songbird populations or habitat use, we note that many bird species have strong affinities for the tree species which have suffered declines. For example:

- Cape May, blackburnian and black-throated green warblers, and saw whet owls all have affinities for red spruce either as foraging or nesting habitats in other parts of their ranges where the tree is more commonly found (Morse 2004, Baltz et al. 1998, Rasmussen et al. 2008, Morse et al 2005);
- Larch is an important food source for white-winged crossbills and other birds which eat conifer seed; it is also important foraging and breeding habitat for palm warblers, Nashville warblers and Connecticut warblers (Benkman 1992, Herbert 1996, Pittochelli 1997; Williams 1996);
- Because of its high sugar content, yellow birch is an important early-season food source for yellow-bellied sapsuckers and ruby-throated hummingbirds, which feed at the sap wells drilled by sapsuckers (Walters et al. 2002, Robinson et al. 1996).

In addition to these changes, it is worth reviewing the work of Martin (1960) who surveyed bird communities in the Park over two field seasons in the early 1950's. He was attempting to discern whether or not distinct bird communities were associated with different forest types. Although the sophisticated analytical tools commonly used in such studies nowadays were not available at the time, he was able to determine that distinct bird communities were present in forest types he classified as bog, boreal forest, deciduous forest, and hemlock forest. Of primary interest here is the bird community associated with the hemlock forest, given its past and likely continuing decline, as discussed earlier. The hemlock bird community had the highest density of territorial males and the greatest number of “principal” species. Most striking was the density of some species found in the hemlock community compared to those found in other communities. He recorded 102 blackburnian warbler males per 100 acres in the hemlock forest, compared with a maximum of 15 in other forest types; also recorded most frequently in hemlock forests were black-throated green warbler (28 in hemlock, compared to a maximum of 6 in other communities), slate-coloured junco (13 in hemlock, max. 4 elsewhere), red-breasted nuthatch (10 hemlock, max. 4 elsewhere). In addition, two species (blue-headed vireo and parula warbler) were found only in the hemlock community. The ongoing decline of hemlock in the Park may be accompanied by declines in these bird species with affinities for it.

While none of these bird species has disappeared from Algonquin due to the declines of the tree species which are disproportionately important to them, we note that the tree species provided superior quality habitat, which, as discussed below, is important for long-term population persistence.

Passenger Pigeons as an Ecological Force

Passenger pigeons once inhabited eastern North America in almost unimaginable numbers. It has been estimated that they comprised 40% of the total bird population in North America (Schorger 1955). The total population is estimated to have been 3 – 5 billion. By comparison, the most abundant bird in North America today, as estimated by Rich et al. (2004), is the American Robin, with a continental population of 320 million. Anecdotes of flocks of pigeons continuing to pass for days and blacking out the sky like an eclipse are common. In his book on the birds of eastern Canada, Tavener (1919) writes: “The immense flocks of passenger pigeons that once darkened the air were one of the wonders of America. The descriptions of their number, if they were not circumstantial and well vouched for by men of undoubted veracity, would sound like wild stretches of the imagination.” Almost inconceivably, passenger pigeons were exterminated within decades of documentation of their incredible numbers. The last pigeons in the wild were recorded around the turn of the 19th century.

Passenger pigeons relied primarily on mast for food and consumed vast quantities of oak, beech, and American chestnut, and other plant foods (Schorger 1955). Algonquin Park was well within the normal range of passenger pigeons and on the border of their principal nesting area. Given their immense numbers and habit of nesting and roosting in huge colonies, they likely had significant impacts on the forests they visited. Pigeons formed large flocks during the late winter/early spring migration. Birds in the flocks roosted and nested in proximity to the end of the breeding season. The selection of nesting and roosting areas was variable because of changing mast crops, so within any one year the impact of pigeons was likely intense in some landscapes, but diffuse in others. Pigeons roosted at night and during storms. Some roosts were used only for a few hours, where others were used for several months (some even for years) (Schorger 1955). Roosts were variable in size, ranging from 0.02 to 10 km^2. Ellsworth and McComb (2003) reported that every centimetre of medium to large tree branches was occupied by roosting pigeons, with some roosting on the backs of others. Ellsworth and McComb (2003) estimated that pigeon biomass in roosts may have ranged between 27,000 and 36,000 kg/ha. Nesting areas tended to be much larger but generally less densely populated than roosts. The typical size of a nesting area was about 80 km^2 although much smaller nesting areas occurred where densities of mass were not great and much greater areas occurred where large supplies of mast were located.

Ellsworth and McComb (2003) examined historical accounts of pigeon roosting and nesting habitat and estimated their potential impact on the forests they inhabited. Pigeon densities in roosts were so great that small trees would be toppled, and the branches of large trees would break (Audubon 1831 and Kalm 1911 cited in Ellsworth and McComb 2003). Although nesting areas tended to have less dense assemblages of birds than roosts, Schorger (1937 in Ellsworth and McComb 2003) noted that some trees had so many nests that they were toppled.

One effect of pigeon nesting congregations would have been to increase the light at the forest floor following physical damage to the canopy. The breakage of tree limbs and toppling of trees would have led to a patchy increase in light under thinned crowns and small gaps. They note that this is an effect similar to other low intensity, frequent disturbances common to eastern forests, such as ice storms and moderate windstorms that topple scattered trees and thin surviving trees through limb breakage. Ellsworth and McComb (2003) estimate that, given the amount of land used for nesting and roosting, about 8% of the pigeon's 1.2 million km^2 breeding area would have been susceptible to these disturbances annually.

Long-term roosts had more intense effects as trees were damaged over a long period. In addition, the massive quantities of excrement produced by the flocks was reported to have caused the death of many understory and overstory plants. Ellsworth and McComb (2003) calculated that a typical roosting assemblage may have deposited 45 kg of nitrogen, 6.9 kg of phosphorous and 6.6 kg of potassium / ha nightly – rates high enough to have detrimental impacts on plants. In addition, the depth of excrement at roosts was recorded as greater than 50 cm in many areas. Wilson (1814 in Ellsworth and McComb) reported seeing dung-covered roosts comprising thousands of ha in which there was no living vegetation. These disturbances were likely stand-replacing over time. Ellsworth and McComb (2003) note that typical return intervals for catastrophic events in northern hardwood forests of 1,000 years (similar to the 800-1,400 years estimated by Rothfels (2002) for Algonquin) may need to be revisited in light of estimates of the impact and extent of passenger pigeon roosts. It seems likely therefore that passenger pigeons added to the forest disturbances caused by abiotic sources (i.e. wind and fire).

Ellsworth and McComb (2003) also noted that pigeon feeding was so intense that they likely had impacts on the prevalence of their preferred mast species in the forest. Bucher (1992) and Ellsworth and McComb theorized that red oak and beech may have been held back in their prevalence in the forest because of the tremendous consumption of mast by pigeons. We note that Pinto (2006) found significant increases in the prevalence of both these species when comparing the 2005 FRI to the township boundary surveys from the 1800's. While neither species is tremendously abundant in the forest (beech increased from 3.61-3.73% in the township boundary surveys of the 1800's to 4.80 -5.29% in the 2005 FRI and red oak increased from 0.09 – 0.12% to 2.81 to 3.02%), the increases in their abundance are statistically significant. Ellsworth and McComb (2003) particularly noted that red oak seemed to be held in check by pigeon feeding. In Algonquin, its prevalence in the forest may have increased 20- 30 times since the 1800s. One wonders if the demise of passenger pigeons may have something to do with this increase.

5.1 Assessment of Changes in Wildlife Habitat and Populations

Quinn (2004) discussed changes in the wildlife community in the hardwood forests of the Park from the presettlement condition. His assessment was, in large part, an attempt to gauge the impact of selection harvesting in the hardwood forests. Much of the discussion focused on the wolf-prey system described above. In assessing other changes, he noted that "The balance of evidence … suggests that the presettlement forest supported a more diverse wildlife community. A higher component of conifer, greater structural diversity, more large trees and snags and more fire and beaver activity would all have added to diversity of habitat and, thus of wildlife." The implication therefore, is of a more depauperate forest, at least from a wildlife perspective, however, the overall conclusion is not really sombre: "The probable historic reduction in wildlife diversity has not, however, been in any sense catastrophic. It is unlikely that any species has been extirpated as a result of changes to the forest and some have benefited."

Although the assessment by Quinn provides valuable insights, there are several aspects of the conclusions and supporting assessment that bear further discussion. These considerations lead to broader discussions of the impact of selection management on wildlife species. The relevant points include:

1. Much of the assessment is based on MNR's wildlife matrix;
2. The absence of extirpation is not equivalent to an absence of significant impacts;
3. Changes in the quality of habitat are not examined; and
4. Responses of wildlife to selection harvesting have been documented, but do not figure prominently in Quinn's conclusions.

These considerations are discussed in detail below.

Assessment based on MNR's wildlife matrix

Given the breadth of the assessment, it was expedient to use a tool which could accommodate considering many species broadly. MNR's wildlife matrix (Bellhouse and Naylor 1997) provided such a tool. Not all of Quinn's assessment is based on the matrix, but it is used to provide a context for the assessment and to consider broad groups of wildlife. The matrix is a coarse tool and should be recognized as such. Holloway et al. (2004) updated the matrix of Bellhouse and Naylor (1997) and noted that there are several limitations to the matrix, including:

a) it is non-spatial and so does not take account of habitat interspersion, habitat heterogeneity and special habitat components;
b) the models do not predict populations, just potential habitat supply; and
c) data available to build the models vary considerably among species. Models for some are based on local data while others are largely based on expert opinion and literature review.

In addition, we note that a recent review of the matrix as a planning tool for forest birds identified concerns regarding its accuracy. Holmes et al (2007) concluded that many of the species models (of 22 assessed) "performed relatively poorly in discriminating between occupied and unoccupied sites", and that "model parameters for at least nine [41%] of the species tested should be reviewed to improve the predictive capability of the models". Our point in raising these concerns is not to be critical of the matrix – *per se*, but to note that conclusions regarding the health of wildlife habitat and populations based on its use need to be tempered with acknowledgement of its limitations.

The absence of extirpation is not equivalent with absence of impacts

The fact that it is unlikely that any species has been extirpated as a result of changes to the forest, while most definitely a good thing, does not mean that important changes may not have occurred. Extirpation is the most dire consequence; goals for impacts on wildlife of forest management are invariably set well above the bar of avoiding extirpation. As described elsewhere in this report, there are notable changes in the Park's wildlife communities compared with the presettlement condition and notable changes which have likely occurred as a result of forest management. The changes described in Section 4.3 indicate that the forest has become more homogenous, as a result of management of efforts. By extension therefore one could conclude that the forest's biodiversity has likely become impoverished as a result of management actions.

Changes in the quality of habitat

The assertion that effects of forest management have been mostly benign is absent of any consideration of changes in the quality of habitat. While animals use many habitat types, relatively few are preferred, and fewer yet are superior (Van Horne 1983, Thompson 2004). As habitats change, so too do the ways animals react and use new habitats. Thompson (2004) notes that it *appears* (his emphasis) that most forest species in Canada are able to adapt to most natural changes. However, he also notes that some species have not adapted well to industrial forest management and the subtle habitat changes which are occurring. He noted that declines in many species of forest songbirds may be at least partly due to changes in the quality of available breeding habitat. Thompson and Welsh (1993) and Voigt et al. (2000) noted that managed forests may support much the same biodiversity, but at reduced population levels for many species in what are actually relatively low-quality habitats. Thompson (2004) drew examples of several high profile species (barred owls, pileated woodpeckers and marten) to illustrate how they can persist in low quality habitats, providing at least some high quality habitat is available proximally. While species may exist in low-quality habitats for a long time, downward trends are inevitable unless local populations are supported by immigration from source populations. For species which prefer undisturbed, old forests, such as those above, and several species of songbirds and small mammals, marten, fisher, lynx, and even many species of insects etc. (McLaren et al. 1998, Euler and Wedeles 2005, Ontario Ministry of Natural Resources 2003), the impacts of forest management which shift ecosystem characteristics away from old growth conditions (and therefore away from superior habitat) may not be manifested in the immediate disappearance of species, but in their gradual decline, making changes hard to detect in the short-term over which most experiments occur and hard even to statistically attribute to loss of old growth.

Responses to Selection Harvesting

Of the studies examining impacts of selection harvesting on wildlife, most have examined songbirds, so that is the focus of this discussion. Portions of this discussion are taken from Wedeles and Donnelly (2004) who discussed impacts of selection harvesting on birds as part of a larger assessment of the impacts of forest management in Canada on songbirds.

Jobes et al. (2004) studied the impact of selection on songbirds in Algonquin Park and found long-term reductions of three species associated with “reference“ (i.e. unharvested) stands. Ovenbird abundance was about 50% lower in recently logged stands and in stands logged 15-20 years earlier. Black-throated blue warbler and yellow-bellied sapsucker abundances were similar to reference stands in recently harvested blocks, but were significantly lower in stands harvested 15-20 years previously. All three of these bird species are commonly considered “interior“ species. Furthermore, we note that although they detected 52 bird species in their surveys (conducted over three field seasons) they only had sufficient data to analyze impacts on 22 species. Therefore, assessments on what are obviously less common species were not included in their analyses. In addition, we note that several species, commonly associated with more open or scrubby habitats (chestnut-sided and mourning warblers and white-throated sparrows) were more common in recent treatment areas than in the reference forests.

Flaspohler et al. (2002) compared bird communities in selectively logged stands (similar to those in Algonquin' hardwood forests) of various ages (up to 29 years post-harvest) in Michigan's Upper Peninsula. They found that twelve species associated with more open and shrubby conditions were not present in stands harvested longer ago. Two species, the black-throated green warbler and ovenbird, were found more frequently in older harvest areas.

Webb (1977) conducted a long-term (ten-year) study of impacts of various levels of diameter limit harvesting on songbirds in New York. One of the harvest treatments compared to unharvested natural areas was a 25% canopy removal. Although the cut was a diameter limit rather than a true selection harvest, the results are likely relevant here. They found that all of the species in the reference forest for which they had sufficient data to analyze were also found in the 25% removal area. Two species characteristic of "undisturbed forest" – the wood thrush and blackburnian warbler - were significantly less common in the harvested area than in the natural area.

Naylor et al. (2004) examined the effects of selection harvesting on productivity of red-shouldered hawks in central Ontario. (Although not stated explicitly in their paper, a map showing the locations of nests they studied seemed to include at least several in Algonquin Park.) They found that heavy cuts (selection or shelterwood cuts with a residual basal area of 14 – 16 m^2/ha) had a significant negative impact on nesting activity, although light harvests (residual basal area of > 20 m^2 and residual canopy closure of ≥ 70%) did not.

We know of relatively few other studies that distinguish between relative levels of harvesting within the selection system as Naylor et al. (2004) did. Wedeles and Van Damme (1995) pointed out that the continuum of tree removal from single-tree selection to group selection could be expected to produce a continuum of effects. In group selection harvests, the removal of a group of neighbouring trees lessens the continuity of vertical habitat diversity, but increases horizontal diversity. The larger openings produce more understory vegetation than do single-tree openings, and in the short-term this would create more habitat for birds that depend on stand openings, but decrease habitat for canopy-using species.

In comparison with silvicultural systems which remove more of the standing timber (i.e. shelterwood and clear-cut systems), the effects of selection harvesting on forest bird species are much less. However, it is apparent that the treatment is not completely benign; a consistent finding of a number of studies (albeit a rather small number) is that some species associated with mature forest have been negatively affected.

6 Invasive Exotic Species

Exotic, or introduced species, have long been a bane of conservation biology (Coblentz 1990, Ricciardi 2007). Fortunately, most exotic organisms cannot manage in their new surroundings and disappear, but some thrive and wreak havoc in their new surroundings. Some exotic species are able to outcompete indigenous ones, as the native species have not evolved to cope with the pressures brought by the exotics. Exotic species may outcompete native ones for resources, introduce diseases, bring strong predatory pressures, or alter the environments in a manner which native species cannot adjust to. By now, most people are familiar with the impacts of Dutch Elm disease, chestnut blight, zebra mussels and gypsy moth – all species which have wrought huge changes to Ontario's ecosystems at the expense of native species.

Unfortunately it seems as if more damaging impacts are in store for the Park area. One of the exotic pests of greatest concern is the emerald ash borer - a shiny green beetle native to Asia. Since its accidental introduction into the United States in the 1990's it has spread to several states and into Ontario. All species of ash are susceptible to it. In the last couple of years it has spread from Michigan into southwestern Ontario. Within the last year it has been detected in the Toronto area (Ontario Forestry Association 2008), in Ottawa (July 25) (CFIAa 2008), and in Sault Ste. Marie (Sept 22) (CFIAb 2008). Its spread though the eastern United States has been rapid, and there seems little chance of arresting its march, or resisting the ecological havoc which it is causing. Although the MNR has stated "It is hoped that over time natural ecosystem controls, such as parasites or disease will help regulate the beetle's population and reduce its impact" (Rose et al. 2006), to date no means of regulating its spread or damage are apparent. Beetles affect mature trees by damaging both the phloem and xylem as they feed under the bark. Infected trees may produce shoots and so may not be killed outright, but the trees stop functioning as mature elements of forest. Black and white ash are not abundant in Algonquin's forests – the ash working group comprises 556 ha of the Park's total area; white ash is described as a "minor component of the Hardwood Uniform Shelterwood forest unit" (Cumming 2005). Nonetheless the virtual loss of these species as functional elements in the Park's hardwood forests would be significant.

Beech bark disease is another exotic infestation of considerable concern. The disease is facilitated by infestation of an insect, wooly beech scale which predisposes trees to attack by a fungus. Both the fungus and the insect were accidentally introduced into North America around 1890 on imported ornamental beech trees brought from Europe to Nova Scotia (Canadian Forest Service 2001). Since then the disease has been moving west slowly. As the disease spreads into new areas, it is preceded by heavy infestation of the scale insect. The subsequent mortality of beech trees is known as the "killing front". In 2007, the Canadian Forest Service reported that the killing front was north of Toronto (Loo 2007), however scale insects have been reported from considerably farther north – along the Ottawa River and just south of Lake Nipissing (Hopkin and Scarr 2004). The disease is common in the northeastern United States in areas at more northerly latitudes than Algonquin Park and has recently been reported in the upper peninsula of Michigan (O'Brien et al. 2001), and so the Park is certainly within the likely range of the disease. Infestation during the killing front leads to mortality of about 85% of trees (Loo 2007), although total mortality after the disease has settled into an area is often considerably greater than 90% (P. Duinker pers. comm.).

The 2005 FMP gives no specific indication of the abundance of beech in the Park. It is grouped with several other species (maple, ironwood, hemlock and yellow birch) as one of the five main species comprising the tolerant hardwoods of the Park. The plan notes that tolerant hardwoods comprise more than 300,000 ha, or almost 49% of the commercially productive forest of the Park. Beech trees are very important providers of wildlife food. Black bears feed extensively on beech nuts, as do deer, red fox, ruffed grouse, rabbits, porcupines, and a host of insect species. Should the killing front extend into the Park, the impact on the Park's ecology could be enormous.

The final and most potentially devastating exotic forest pest is the Asian long-horned beetle. Native to Asia, the beetle was discovered in North America in the New York area in 1996. Infestations in the United States have since been reported in New Jersey, Chicago, and most recently in Massachusetts. In 2003, an area of infestation was discovered in the Woodbridge area, just north of Toronto. While the thought of beech bark disease and the emerald ash borer spreading into the Park (or for that matter anywhere in the hardwood forests of central Ontario) is unnerving, the potential damage that the Asian long-horned beetle may cause makes the notion of its spread into the Park devastating. This is because the beetle does not just kill one species or genus of trees; rather it attacks and kills many hardwood species. Tree species that were found to be attacked in the Woodbridge area included all species of maple, elm, poplars, oaks, birch, ash, and several ornamental species. The beetle does not attack conifers. The majority of Canadian broadleaf trees are at risk from this beetle. Management of infested urban and suburban areas has been very aggressive. In the Woodbridge infestation, all potential host trees within a buffer were cut down, and monitoring of potential host trees within a broader buffer zone continues. As with most serious forest insect pests, the beetle affects trees by boring through and eating the living layer of wood under the bark. To our knowledge, no infestations of the beetle north of the Toronto area have been found, however, the MNR notes that Canada’s temperate climate is well-suited for establishment of the insect as the larva winters deep within trees, protected from harsh winter conditions (Ontario Ministry of Natural Resources 2008). The beetle has no known natural enemies within Canada’s forests and so far, no effective insecticides have been found. The potential impact of this species in natural forest areas is horrendous and ecosystem-altering on a tremendous scale.

Although there are indications of herbaceous exotic plants spreading into the Park, few if any seem to have the potential to spread into forested areas and cause significant ecological damage; most are restricted to roadsides and disturbed areas so far. The exception to this may be garlic mustard. Park staff have found small patches of garlic mustard along Highway 60 (all of which have been treated with herbicide and killed). Garlic mustard is a very invasive species, originally from Europe, which is well established in southern Ontario and throughout much of the northeastern United States. Recent research in southern Ontario, Stinson et al. (2006) found that garlic mustard suppresses the growth of native plants in forests by disrupting the mutualistic associations between vascular plants, including trees, and mycorrhizal fungi. This mechanism likely explains how the species can invade forested areas and the local extirpations of native plants and impoverishment of forests in which garlic mustard becomes established.

The final exotic taxon worthy of some discussion is earthworms. Many people think that earthworms are a natural component of Ontario’s ecosystems, but in fact this is not the case. In most of Canada and the northern United States, native earthworms were extirpated during the Pleistocene glaciations (Reynolds 1994), so most of Canada’s forests have evolved in the absence of this common invertebrate. When people think of earthworms, the night crawler or dew worm most frequently comes to mind, however many other species are also common. There have been 36 species of non-native earthworms reported in North America (Reynolds 1995). Earthworms were likely first introduced with the importation of plants from other countries (Gates 1982).

Earthworms are commonly spread into forested areas by fishermen who dump or lose their live bait near lakes and along portages and trails (Cameron and Bayne 2007, Keller et al. 2007) and by vehicles. Gundale et al. (2005) found that forest sites in Michigan close to roads with a history of timber harvest were more likely to have earthworms than similar sites without roads or a history of timber harvest.

Worms and their cocoons can be spread as “hitchhikers on vehicles” and they may also be present in the soils and gravels that are used in road construction (Gundale et al. 2005, Holdsworth 2007). With the reliance of logging on a road infrastructure and vehicles of various sorts, the risk of spread of earthworms while conducting forest management is high.

Earthworms can seriously alter the ecology of forests in which they become established. Bohlen et al. (2004) explained that “earthworms shift the soil from a slower cycling fungal-dominated system to a faster cycling, bacterial-dominated system … This is accomplished through the redistribution and transformation of soil organic matter as earthworms consume organic-rich forest floor material and incorporate it into underlying mineral soil.” Forests with established earthworm populations are noticeably depauperate in fallen leaves and fine litter, particularly from late summer on. The mat of fallen leaves is simply absent, and in its place one sees the unnatural sight of bare soil on the forest floor. Bohlen et al (2004) draw upon the work of many researchers to identify the known and possible impacts of earthworms in forests:

- Possible net loss of carbon from the forest as the pool of carbon tied up in surface organic matter is exposed to greater mineralization rates through earthworm activity;
- Increased leaching and altered cycling of nitrogen, phosphorus and other nutrients;
- Reduction in seedbed and substrate for seeds and seedlings and exposure of seedlings to desiccation and predation;
- Possible decreased abundance of mycorrhizal fungi, upon which vascular plants rely for assistance in the uptake of water and nutrients from soil;
- Loss in the diversity and abundance of mycorrhizal plants and increases in non-mycorrhizal species;
- Decrease in vigour of existing trees by removal of the forest floor and exposure of fine roots;
- Loss of habitat for soil and surface-dwelling invertebrates; and
- Decreased abundance of vertebrates which prey upon soil invertebrates.

Earthworm-infested forests can become drastically altered in terms of soil chemistry and stability and the diversity and relative abundance of plants and animals. As efforts to understand the impacts of earthworms in forests become more extensive, the related web of effects comes more into focus. For example, a recent study (Marez et al. 2005) suggests that introduced earthworms may play an important role in the ecology of red-backed salamanders in some forests. Earthworms are consumed by adult salamanders and increase their fecundity. However, as Bohlen et al. (2004) point out, in spite of producing more offspring, the overall impact of earthworm establishment is likely detrimental, because the survival rates of young salamanders are negatively affected by reduced populations of smaller invertebrate food sources which become less abundant due to the changes in the forest floor caused by earthworms. In addition, we note that the absence of leaf litter on the forest floor may create a situation in which there is an abundance of food for adult salamanders (i.e. earthworms), but reduced habitat for them. Complex relationships such as this likely also exist for birds, small mammals and predators of soil invertebrates.

There may be a temptation to moderate concerns regarding earthworms because their rate of spread is generally taken to be rather slow – roughly 4 – 10 m/yr (Marinissen et al. 1992, James 1998). However these estimates are probably conservative - Mather and Christensen (1988) found that dew worms are capable of travelling almost 20 m on the soil surface in one evening. Furthermore, when one considers a scenario such as exists in Algonquin, where logging trucks and equipment are frequently deployed in many parts of the forest (due in part to the selection harvesting), and where canoeists/fishermen travel throughout the Park, the potential for establishment at many places seems evident. From multiple nodes of establishment, even a slow rate of spread can have serious implications as the advancement could occur on many fronts.

6.1 Are there any preventative measures?

The threats posed by exotic invasive species create a very bleak picture, since there is no apparent treatment for the emerald ash borer, beech bark disease, and especially against the Asian long-horned beetle.

We suggest that at a minimum, monitoring would be worthwhile. We note that white ash is at the northern extremity of its range in Algonquin Park, and suggest that it may be possible to successfully plant small patches of white ash in suitable microsites in the northern part of the Park, where it may be isolated enough to escape infestation. This would be similar to American elm, which is scattered in the Park and the isolation of some individuals in some areas has prevented the Dutch elm disease from attacking even large mature trees. While this would be a far from ideal outcome, it is preferable to doing nothing. A similar approach might be possible with beech as well.

The nature of these threats, and the lack of apparent remedies, suggest that these are areas where the AFA could be actively encouraging research and experimental treatments, as would be consistent with the Park charter. The existence of some beech that appear to have genetic resistance to the beech bark disease also indicates a potential avenue for trying to maintain a reasonable abundance of beech in the Park.

7 Potential Management Responses

The discussion above has demonstrated that there have been significant changes to the forest composition in Algonquin Park, as a result of the introduction of industrial logging. These changes have shifted the relative abundance of many of the tree species, the age class distribution of the forest, and the structure of stands. There is also strong evidence that these changes have impacted many wildlife species by reducing the abundance of their habitat. This is particularly true for many songbirds, beaver, and deer, and there have been attendant indirect impacts on other wildlife species. Forest management is not the sole culprit, and in fact many of the most significant changes in overall songbird population levels, deer and moose populations, as well as the extirpations of elk and caribou, are more strongly due to cumulative impacts, including habitat changes outside the Park, and outside the country, in the case of songbirds, and due to other biological factors in the case of the ungulates.

The main thrust of this section of the report is to provide recommendations for addressing issues outlined above through the modification of existing forest management approaches. There are likely other measures that could be taken, especially on the unmanaged portion of the Park landbase and perhaps also in the lands surrounding the Park, but these are not discussed here. We have also identified that there is a potential loss in productivity due to soil acidification and nitrogen deposition, and that this issue has tended to be debated outside forestry circles. We also provide some recommendations aimed at developing a better understanding of the issue.

With the exception of the soil chemistry issue, we observe that the AFA is aware of most, if not all, of the issues outlined above and has taken some measures to address some of them. The 2005 FMP contains objectives related to the maintenance of species diversity, but there are few objectives related to species restoration. The plan includes measures to increase the area of the white and red pine forest units, with a target to not reduce the total available area of red and white pine forest units below the 2000 level (i.e. 76,829 ha). This objective is directed to pine-dominated stands, which were the focus of a previous report for Algonquin Eco-Watch (ArborVitae Environmental Services Ltd 2007).

The AFA also follows the “Hemlock Strategy for Algonquin Park”. In the 2005 FMP, the AFA increased the area of the hemlock forest unit by lowering the minimum hemlock percentage abundance requirement from 50% to 40%. In addition, the management system was changed from shelterwood to selection in order to provide more effective management of hemlock regeneration. The hemlock decline issue has been known for some decades, and studied extensively.

There are two other specific goals related to species retention in the 2005 FMP:

- Maintain a minimum of 1,750 ha of jack pine forest unit in the available forest over the next 100 years (currently 2,843 ha, as reported in Table FMP-9); and

- Maintain a minimum of 12,000 ha of the intolerant forest unit (INTCC) area in the available forest over the next 100 years (currently 38,740 ha in FMP-9).

Under the selected forest management direction in the 2005 FMP, both of these forest units are forecast to experience declines in area. In particular, there is a steady decline of the area in the INTCC forest unit from 38,740 ha in 2005 (about 8% of the available forest area) to 13,063 ha in 2105 (3%). This is attributed to the considerable area of “old forest” in this forest unit undergoing natural succession, and post-harvest shift of some area to the mixedwood and pine forest units. The jack pine forest unit declines from 2,841 ha in 2005 to 1,853 ha in 2025, at which level it more or less remains throughout the remainder of the simulation period. Succession is also the main reason for the loss of jack pine area.

In summary, the plan pays considerable attention to maintaining pine, and actually increases the area of available red and white pine over time. The plan also supports an increase in the abundance of hemlock, although there are no quantitative targets. There are no objectives to restore other species.

In addition, some of the strategies to meet forest diversity objectives will increase the abundance of some of the species which have declined relative to their historical levels of abundance. These include:

- Tree species that occur in small numbers in forest units other than their own will be retained to maintain species diversity on the landscape (i.e. scattered pine in a poplar stand, oak in a tolerant hardwood stand);

- Encourage the natural development of advanced conifer regeneration (white pine and hemlock) in hardwood stands in order to more closely resemble pre-settlement forest diversity;

- The prioritization for retaining directly competing trees places red spruce and hemlock as top priority, followed by yellow birch, and then as third priority the mid-tolerant red oak, black cherry, white ash and basswood. Maple is fourth priority and beech fifth. and

- Tree marking prescriptions maintain at least 8 mast-producing trees/ha (oak, beech, black cherry and basswood), and at least 6 cavity nesting trees/ha > 25 cm dbh, if they are available.

Other strategies that will promote biodiversity and the retention of pre-industrial forest characteristics include:

- Woodsworkers will be encouraged to leave in the forest cull ends of logs and cull trees as downed woody debris. Particular efforts will be made to leave large material; and

- Protect, where safety permits, “super size” (DBH > 80 cm.) decadent or declining trees in tolerant hardwood stands to maintain this feature of old growth forests and provide for large snags and downed woody debris.

While these measures are positive, in our view they do not go far enough.

One of the major insights introduced through conservation ecology is the need to develop an integrated approach to management. In our view, given the documented shifts in species composition, we suggest that it would be more appropriate for the forest management plan to contain a more general restoration goal, with explicit targets related for each of the major species groups. In addition to increasing the biodiversity of the forest, restoration would increase the resilience of the forest[4]. As climate change manifests itself, enhancing the resilience of the forest is one strategy to mitigate negative impacts and it may also facilitate the shift in species ranges that is anticipated.

Ideally, there would be restoration targets related to the abundance of mid-tolerant species which have declined in abundance, including yellow birch, basswood, and black cherry. It would normally be appropriate to include targets for white ash, but given the threat posed by the emerald ash borer (see discussion in section 7), restoration is unlikely and preservation is the order of the day.

One of the significant obstacles to setting feasible targets is the inability of the forest inventory to provide an accurate depiction of current abundance, and the lack of suitable data to drive forecasting models. Therefore, forest managers are poorly equipped to

[4] Ecological resilience is the ability to withstand and recover from stresses, disturbance, and other perturbations. Generally, diversity and resilience are positively correlated.

quantify the current level of abundance and assess how that changes over time. As discussed above, the forest inventory does not identify a species unless it accounts for 10% or more of the forest cover. Yellow birch often forms an identifiable component of a stand and so it often does show up in the inventory. It would probably be feasible to develop some form of abundance target for this species.

However a requirement to reach 10% of a stand automatically eliminates smaller trees such as ironwood from showing up in the inventory except under unusual circumstances. Species such as basswood, black cherry and white ash rarely reach the 10% threshold, and so are more apt to be included as tolerant hardwoods or just to be ignored.

There have been many discussions over the years on how one might improve the forest inventory to, among other things, more completely reflect the species composition of the forest. Any progress has usually foundered on the lack of a cost-effective way of accurately recording the species composition of a stand. We observe that tree markers visit each hardwood stand to mark it, and they could be asked to record more data from each stand, or for some of the stands that they mark. The area of group selection cuts that are implemented is also not recorded accurately, and hence another opportunity to try to build up a better forest information database is by-passed.

We recognize that improvements to the inventory will be expensive but we feel that if they are undertaken gradually, there would be room to build up better information for the stands that are marked. Information about the area that that is cut using group tree selection would also be beneficial if it could be tied in with the expected species composition of the renewal. We suggest that it should not be up to industry to pay the full costs associated with inventory improvement – given the heavy recreational use of the Park, it is reasonable to argue that an improved inventory is in the general public interest.

In the absence of the ability to set functional species abundance targets, we suggest that the FMP could consider targets such as creating some number of group shelterwood openings per year where the expected renewal would have a high composition of a specific target species. As an example, such a target could be to create a minimum of twenty openings per year of at least 2 hectares in size where black cherry is likely to be a dominant species in the renewal. If there were not enough areas allocated to meet this target, the requirement could be to plant some openings with black cherry to reach the target level.

A major gap in the restorative aspects of the plan direction, in our view, is the lack of any initiative to restore white pine to the tolerant hardwood forest complex. As discussed above, white pine was frequently present at low densities in the west side forest. Although the lack of disturbance did not favour it, occasional large disturbances coupled with a long lifespan and partial shade tolerance enabled it to maintain a presence. Now that the seed source has largely disappeared from the west side, planting would be required to restore white pine. We feel that this is feasible and would restore an important ecological component to forests on the Park's west side. Again, we do not feel that it should be solely up to the AFA to fund this – it is in the public interest to restore white pine and the government could add funds to the forest renewal trust to pay for some amount of low-density white pine planting in the tolerant hardwood forests.

The scientific literature is quite consistent in identifying that the prolonged application of single-tree selection simplifies and homogenizes the forest, especially when the prescription is heavily weighted towards timber production. The marking guidelines used in Algonquin Park do consider a wider range of values, and so it would not be appropriate to categorize the guidelines as being solely oriented towards timber production. However, one theme running through the ecological research on the impacts of various harvesting regimes is that the continued application of a standard harvest template homogenizes forest over time, and we feel that the AFA would be well-served by experimenting with developing a variety of timber marking approaches. Perhaps it would be feasible to design, say, three sets of marking prescriptions. One set could be oriented towards timber production, a second could have a strong bias toward mast tree retention, and a third to be developed to "maximize diversity". Each marking guideline could be applied on one-third of the commercial forest area, so producing a greater diversity of forest stand types.

A second aspect of homogenization concerns the structure of the forest. Again, the application of a standard harvesting and management approach creates a more uniform pattern. Forest diversity would be enhanced if variation was introduced through the retention of different structural elements, including greater patchiness, variation in retention of culls and large old trees. We also observe that the suppression of fire has removed one source of canopy gaps, and the forest may contain fewer medium and large-sized gaps than in the past. On the other hand, wind events are still creating gaps, and so it is not clear where the present forest stands in relation to estimates of gap prevalence derived for pre-industrial forests. This would be a useful topic of research. If there was a notable reduction in the extent of medium and large gaps, then it might be appropriate to consider introducing some harvesting that would create patches that might range from 5 – 25 ha in size. We recognize the high level of sensitivity to a proposal such as this, and it is why we suggest a preceding study of wind impacts. However, we also point out that gap formation is a critical process that enables the forest to retain diversity, and if it has been seriously curtailed, it would be worthwhile examining whether it could be re-introduced in some manner.

Finally, we suggest that it would be beneficial for the forest community to participate in efforts to assess and rectify the soil chemistry issues that have been outlined. At present, the Ministry of the Environment and a number of non-forestry centred researchers are most active on these issues, and there is a strong case to be made that the forest community should become aware of and begin to work with them. There appear to be at least two measures that could be undertaken that do not require substantial additional resources, since they primarily involve on-going initiatives. The first of these is to coordinate the monitoring of permanent sample plots within the Park and surrounding area, and to expand the characteristics that are measured to include some soil chemistry indicators. These could include not only indicators related to the acidification issue but also soil carbon, thus contributing to the development of a ground work of information on carbon cycling in the Park.

We have recently become aware that there are permanent sample plots associated with the Swan Lake Research Centre and other locations in the Park, and that there are permanent sample plots associated with the Ontario Forest Research Institute (OFRI). The National Forest Inventory also maintains a grid of permanent plots across the country, and we expect that there may be between 6 – 10 of such plots in Algonquin Park, as well as others outside the Park but in similar forest. Integrating these plot networks and associated databases would provide a more robust set of information for assessing forest characteristics that might be affected by the acidification of soil, including growth rates and mortality rates of trees of all sizes, including seedlings. As mentioned, adding soil measurements would yield data that could lead to a better understanding of such factors as the role of harvesting in soil chemistry.

8 Conclusions

We have reviewed the character of the tolerant hardwood forests of the west side of Algonquin Park, and found that there have been significant shifts in the relative abundance and age class distribution of most tree species. In particular, compared to the pre-settlement forest, there have been significant documented reductions in:

- Red spruce;
- Larch;
- American elm;
- Yellow birch;
- Eastern hemlock;
- Cedar; and
- White pine.

The 2005 FMP indicates that red oak and jack pine have also declined and observes that the poplar and white birch ecosite now tends to have red and white pine on them. We also suspect that, in addition to yellow birch, other mid-tolerant hardwood species such as basswood, black cherry, and white ash (which are at the northern limit of their range in the Park) are less common in today's forest than they were previously. While there is little quantitative evidence for this, such an outcome has been documented in northern hardwood forests that have been subject to the application of the single-tree selection system, as Algonquin Park has. In addition, we note that the forest inventory does a poor job of capturing the presence of these species, and they are generally ignored under management.

There is an emerging body of literature that describes the homogenizing effects of widespread implementation of the selection system, and the management approach is also described as contributing to the simplification of the ecosystem. This refers to the effects of reducing variability and structural complexity within individual stands and within the forest at the landscape scale.

We have documented considerable evidence that a number of wildlife species have declined in abundance in the Park, due to habitat loss inside the Park as well as outside the Park, especially so in the case of many migratory birds. Examples include beaver, white-winged crossbill, saw whet owl, and palm, Nashville, Connecticut, Cape May, blackburnian and black-throated green warblers.

There have been fluctuations in the relative abundance of moose and deer, which is a complex relationship within which forest management has some influence, although it is often the case that the influence cannot be categorized as positive or negative. The one instance where forest management has clearly been deleterious concerns the management of the Park's deer winter yards, which are now no longer used.

We also have observed that there were likely elk and caribou in the Park, but they have been extirpated for decades. Similarly, the demise of the passenger pigeon removed a keystone species, but this had little to do with forest management in Ontario. However, these extirpations indicate that the pre-industrial forest was quite different in some significant ways from the present forest.

The AFA is moving on some of these issues, however we feel that the AFA should be moving more forcefully, and that the MNR should be assisting with some of the mitigation measures that we are recommending. The Park is too large, complex and heavily used for a forest manager to be able to manage comprehensively relying only on revenue derived from timber dues.

We offer the following specific recommendations to improve management in Algonquin Park:

- the FMP should contain a general restoration goal, with explicit abundance targets for each of the major species groups;

- the FMP should contain restoration targets related to the mid-tolerant species which have declined in abundance, including yellow birch, basswood, and black cherry;

- the AFA and MNR should jointly develop and implement an initiative to restore white pine to the tolerant hardwood forest complex;

- the AFA and MNR should jointly develop an approach to improve the quality of the forest inventory by enabling it to better track the species presence and abundance in stands. This should be co-funded by MNR and the AFA;

- AFA should increase the diversity of the managed forest by developing a variety of timber marking approaches that will increase the variety of prescriptions and the variety of forest types created under management;

- the AFA should introduce variation in terms of retention of different structural elements, including greater patchiness, variation in retention of culls and large old trees, as well as the retention of appropriate old-growth stands in the R/U zone. Consideration should be given to site preparation of the openings created by group selection, underplanting species other than sugar maple, and removal of patches of sugar maple in the understory; and

- the AFA and MNR should determine whether wind events create sufficient medium and large gaps to emulate the natural patchiness of the pre-industrial forests.

In Algonquin Park, the first line of defence against exotic species should be monitoring. It is likely that the emerald ash borer will appear in the Park by 2010, if it is not already present. Once it reaches the Park, there is little that one can do except to try to prevent it from reaching the more remote pockets of ash. Restrictions on wood movement, and planting ash in remote sites are potential approaches that may slow its movement.

Because not all beech are killed by the beech bark disease, it would seem that AFA and MNR staff should begin to investigate the feasibility of reproducing the trees that exhibit resistance. In this case, it seems that there is some potential for retaining some presence of beech on the landscape, which is important given its prominence as a mast-producer.

Finally, we have identified soil acidification as an on-going cause of reduced productivity and have provided some recommendations that would bring the forestry community into these research efforts and make use of existing permanent sample plots to investigate whether tree growth and survival are being impacted. We also suggest adding soil chemistry measures to the suite of variables that are assessed at each measurement of the plots.

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