UNIVERSITÉ DU QUÉBEC À MONTRÉAL

CARACTÉRISATION DE LA DISPONIBILITÉ DES ARBRES POTENTIELS À LA NIDIFICATION DU GARROT D'ISLANDE DANS LA FORÊT BORÉALE DE L'EST DU QUÉBEC

MÉMOIRE

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DE LA MAÎTRISE EN BIOLOGIE

PAR

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AVANT-PROPOS

Ce mémoire comporte deux chapitres rédigés sous forme d'articles. Conformément aux exigences de la maîtrise en biologie, j'ai procédé à la supervision et à la récolte des données, à l'analyse des résultats et à la rédaction de deux articles à titre de première auteure. Le premier article, **Availability of standing trees for large cavity nesting birds in the eastern boreal forest of Québec, Canada** sera soumis à la revue Forest Ecology and Management alors que le second, **Cavity availability for Barrow's Goldeneye (*Bucephala islandica*) in the eastern boreal forest of Québec, Canada**, sera soumis à une revue ornithologique arbitrée.

Mon directeur de recherche, Pierre Drapeau, ma co-directrice, Sylvie Gauthier, ainsi que Michel Robert, biologiste au Service canadien de la faune, en sont les coauteurs.

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RÉSUMÉ

L'aire de nidification de la population de l'Est du Garrot d'Islande (*Bucephala islandica*), désignée préoccupante au Canada, se situe majoritairement dans la forêt boréale de l'Est du Québec. Les sites de nidification utilisés par le Garrot d'Islande décrits pour la population de l'Ouest de l'Amérique du Nord consistent principalement en de grandes cavités excavées par le Grand Pic (*Dryocopus pileatus*) dans des arbres de gros diamètre. Les connaissances quant aux structures de reproduction utilisées par la population de l'Est sont, à ce jour, méconnues. Toutefois, il est fort probable que l'espèce requiert également des arbres de nidification de gros diamètre. Préalablement au présent travail, il n'existait pratiquement aucune information sur la disponibilité de gros arbres et de grandes cavités au sein de l'aire de nidification du Garrot d'Islande dans l'Est de l'Amérique du Nord. Il était donc opportun d'acquérir ces connaissances fondamentales d'autant plus que l'aménagement forestier pratiqué dans cette région fortement dominées par des forêts surranées, les plus susceptibles d'abriter de grande quantité d'arbres de gros diamètre, prend rapidement de l'expansion. Une diminution de la disponibilité et du recrutement d'arbres de gros diamètre pourrait ainsi affecter la disponibilité d'arbres de nidification dans les secteurs aménagés fréquentés par le Garrot d'Islande.

Le premier chapitre de ce mémoire porte sur la disponibilité d'arbres morts et vivants de gros diamètre dans les principaux types de couverts forestiers âgés ainsi que dans les parterres de coupe et les forêts résiduelles (bandes riveraines et séparateurs secs) de la forêt boréale de l'Est. La disponibilité d'arbres de gros diamètre était fortement reliée à la présence de sapins baumiers (*Abies balsamea* (L.) Mill) dans les peuplements. Le patron de distribution des classes de dégradation des arbres morts était sensiblement le même pour les différents types de peuplement, les classes de dégradation intermédiaire étant plus abondantes que les classes récentes et avancées. Toutefois, les peuplements dominés par l'épinette noire (*Picea mariana Mill.*) possédaient des densités réduites pour chacune des classes de dégradation. Ces résultats suggèrent que les sapinières pourraient jouer un rôle clé au sein de la forêt boréale coniférienne en raison du bon recrutement de grosses tiges à divers degrés de détérioration. Quant aux forêts résiduelles, elles ont non seulement des densités en gros arbres morts moins élevées que les peuplements de la forêt naturelle, mais les peuplements de haute qualité en termes de bois mort, identifiés en forêt naturelle, ne sont pas représentées dans ces forêts constituant les principaux legs au sein des paysages aménagés.

Le second chapitre traite de la disponibilité de cavités naturelles et excavées dans les forêts boréales naturelles, résiduelles et récoltées de l'Est québécois. La disponibilité des arbres de nidification pour la population de l'Est du Garrot d'Islande s'avère différente de celle décrite pour la population de l'Ouest puisque la présence du Grand Pic n'a pas été détectée dans la portion de l'aire de nidification couverte par cette étude. L'espèce dépend donc de la disponibilité de cavités naturelles pour les sites de nidification en forêt boréale coniférienne. Bien qu'aucune espèce d'arbre ne se soit avérée plus sujette à supporter des cavités, une forte proportion des arbres porteurs était des chicots fortement dégradés. La densité de cavités adéquates pour le garrot s'est toutefois avérée extrêmement faible. La probabilité des essences boréales à supporter des cavités est plus élevée chez les tiges de gros diamètre. Or,

la disponibilité de ces arbres potentiellement porteurs de cavités est fortement réduite dans les parterres de coupe ainsi que dans les forêts résiduelles comparativement aux peuplements naturels.

La présente étude montre que l'aménagement forestier, tel que pratiqué actuellement, ne favorise pas la rétention de peuplements possédant une forte abondance d'arbres de gros diamètre, structure clé pour le Garrot d'Islande et la grande faune cavicole. De nouvelles approches d'aménagement favorisant le maintien d'arbres à valeur faunique à l'échelle du peuplement et du paysage devraient être appliquées dans les secteurs fréquentés par le Garrot d'Islande en période de nidification afin d'assurer le maintien et le recrutement d'arbres morts de gros diamètre.

Mots clés : arbres morts, arbres d'intérêt pour la faune, Garrot d'Islande, cavités, aménagement forestier, forêts résiduelles

INTRODUCTION

1.1 La forêt boréale de l'Est: dynamique et arbres de gros diamètre

La dynamique forestière boréale a longtemps été décrite comme étant régie par un cycle de feu assez court, où les vieux peuplements ne sont pas dominants dans la mosaïque forestière (Johnson 1992; voir Bergeron et al. 2001). Lorsque le cycle de feu est long, les épidémies d'insectes et les remplacements par trouées sont les principaux facteurs modifiant la structure forestière (Blais 1983; Kuuluvainen 1994; Bergeron et Leduc 1998; Kneeshaw et Bergeron 1998; McCarthy 2001). Dans la forêt boréale de l'Est du Québec, le cycle de feu est d'environ 300 ans (Gauthier et al. 2001; Bergeron et al. 2006), permettant aux perturbations agissant à petite échelle d'être les principaux agents modificateurs de la structure forestière (De Grandpré et al. 2000; Boucher et al. 2003; Pham et al. 2004). Cette région est donc caractérisée par l'omniprésence de vieilles forêts puisque la proportion de forêts surannées dépasse les 70 % pour ce territoire (Bergeron et al. 1999; Boucher et al. 2003). La mosaïque forestière naturelle en forêt boréale de l'Est est donc principalement constituée d'une mosaïque de peuplements à dominance résineuse qui dépassent largement l'âge d'exploitabilité (De Grandpré et al. 2000; Boucher et al. 2003). Boucher et al. (2006) ont montré que les peuplements dominés par le sapin baumier (*Abies balsamea* (L.) Mill) atteignent une structure irrégulière plus rapidement que les peuplements d'épinettes noires (*Picea mariana* (Mill.), ce qui serait lié à la plus grande productivité des sites occupés par le sapin. De plus, les peuplements de sapins baumiers sont plus susceptibles aux épidémies de Tordeuse des bourgeons de l'épinette (TBE) (MacLean 1984), la dernière ayant sévi à la fin des années 1970 dans l'Est du Québec (Morin et Laprise 1990). Conséquemment, ces différences quant à la structure et la dynamique de la mortalité entre les types de peuplements de la forêt boréale de l'Est sont à même de se refléter dans leur disponibilité respective en arbres morts et vivants de gros diamètre et leur potentiel à abriter des arbres à valeur faunique.

L'abondance des arbres morts sur pied et des débris ligneux de gros diamètre constituent des caractéristiques structurales importantes des forêts ayant dépassé l'âge d'exploitabilité (Spies

et al. 1988; Hansen et al. 1991; Tyrell et Crow 1994; Sturtevant et al. 1997; Clark et al. 1998; Siitonen et al. 2000; Kneeshaw et Gauthier 2003; Thompson et al. 2003a; Desponts et al. 2004). La variabilité dans l'abondance d'arbres morts au sein même des vieilles forêts est reliée au taux de mortalité ainsi qu'au taux de décomposition des tiges mortes. La productivité des peuplements ainsi que la prévalence d'agents de perturbation influencent le taux de mortalité des arbres (Linder et al. 1997; Sippola et al. 1998; Rouvinen et al. 2002; Nilsson et al. 2002) alors que le taux de décomposition des tiges varie selon les différentes composantes spécifiques des peuplements (Harmon et al. 1986; Kruys et al. 2002).

1.2 Les arbres morts de gros diamètre et la faune aviaire cavicole

Les arbres morts remplissent plusieurs fonctions écologiques dont celle de fournir un substrat de nidification à plusieurs organismes (Harmon et al. 1986). La faune aviaire comporte beaucoup d'espèces cavicoles, c'est-à-dire des espèces qui dépendent des arbres morts ou sénescents portant des cavités pour y pondre leurs oeufs. Parmi celles-ci, on trouve les excavateurs primaires, principalement représentés par des oiseaux de l'ordre des Piciformes, qui excavent eux-mêmes leurs cavités de nidification (Short 1982; Raphael et White 1984; Winkler et al. 1995). Certaines espèces sont qualifiées d'excavateurs faibles, puisqu'elles peuvent excaver des cavités seulement dans des arbres dont l'état de décomposition est avancé, ne possédant pas les caractéristiques anatomiques leur permettant d'excaver du bois dur (Schepps et al. 1999). Ces espèces, telles que la Mésange à tête noire (*Poecile atricapillus*) et la Sittelle à poitrine rousse (*Sitta canadensis*) sont également reconnues pour utiliser des cavités excavées par d'autres espèces (Martin et Eadie 1999; Aitken et al. 2002; Martin et al. 2004). On trouve enfin les utilisateurs secondaires de cavités, c'est-à-dire des espèces incapables d'excaver elles-mêmes leurs cavités, qui dépendent donc de la disponibilité de cavités creusées par les excavateurs primaires ou de cavités naturelles (Von Haartman 1957; Martin et Eadie 1999). Il existe parmi les utilisateurs secondaires une variété d'espèces de tailles différentes, allant du petit Troglodyte mignon (*Troglodytes troglodytes*; longueur: 10 cm) à la Chouette rayée (*Strix varia*; longueur: 60 cm). Les utilisateurs secondaires de cavités sont souvent intimement associés à un excavateur primaire (Martin et Eadie 1999).

La densité d'utilisateurs de cavités peut être limitée par la disponibilité de sites de nidification dans un écosystème (Brawn et Balda 1988; Newton 1994). Bien que peu d'études aient porté directement sur les utilisateurs de grandes cavités, on peut penser que ces espèces sont plus susceptibles d'être limitées compte tenu de la taille minimale requise pour qu'une cavité de nidification leur convienne. De plus, en l'absence d'espèces excavatrices, ces grands utilisateurs secondaires dépendent de la disponibilité de cavités naturelles dans le milieu. Or, il existe très peu d'informations sur les cavités naturelles dans la littérature (voir Bai et al. 2003; Remm et al. 2006). La plupart des études menées sur les utilisateurs secondaires sont effectuées dans des systèmes où les excavateurs primaires sont présents et, par conséquent, où les cavités excavées sont abondamment utilisées (Martin et Eadie 1999; Bonar 2000; Evans et al. 2002). Toutefois, quelques études portant sur la nidification du Canard branchu (*Aix sponsa*) en forêt riveraine ont répertorié les cavités naturelles utilisées par cette espèce (Gilmer et al. 1978; McComb et Noble 1981; Soulliere 1988; Lowney et Hill 1989; Yetter et al. 1999; Roy Nielsen et al. 2007). Les généralités ressortant de ces études sont que les arbres portant les cavités sont majoritairement des feuillus vivants, de gros diamètre. En forêt boréale conifférienne, la disponibilité de cavités est généralement plus faible qu'en forêt feuillue (Waters et al. 1990; Bai et al. 2003) puisque les espèces conifériennes sont moins propices à la formation de cavités résultant du bris d'une branche.

1.3 Effets de l'aménagement forestier sur les arbres morts et leurs utilisateurs

L'aménagement forestier modifie la mosaïque forestière en augmentant la proportion de jeunes peuplements au détriment de peuplements matures et surannés (Hansen et al. 1991; Bergeron et al. 1999; Drapeau et al. 2000, 2002, 2003; Harper et al. 2003). De plus, la rétention d'éléments structuraux essentiels au maintien de la diversité biologique comme le bois mort n'est pas généralement appliqué dans les pratiques forestières traditionnelles (Hansen et al. 1991; Franklin 1993; Franklin et al. 2000). Le temps de rotation des interventions sylvicoles est souvent plus court que le cycle des perturbations naturelles (Gauthier et al. 1996; Bergeron et al. 2002) et ne permet donc pas le recrutement d'arbres présentant les caractéristiques pouvant éventuellement permettre la nidification d'espèces cavicoles. De plus, le faible nombre de tiges sur pied présent après une coupe laisse très peu

d'arbres rémanents comparativement aux forêts affectées par une perturbation naturelle majeure (Imbeau et al. 1999; Payer et Harrison 2000; Fergusson et Elkie 2003), ce qui entraîne une réduction de la diversité de classes de diamètre et de degrés de décomposition de la nécromasse dans ces peuplements.

Les forêts résiduelles laissées sur les sites aménagés en vertu du *Règlement sur les normes d'intervention dans les forêts du domaine de l'État* (RNI) (Ministère des Ressources naturelles du Québec 1996) peuvent servir de refuges aux espèces forestières pendant la régénération des parterres de coupe. Toutefois, Darveau et al. (2001) ont émis certaines réserves quant à la valeur des bandes riveraines comme refuges pour certaines espèces de petits mammifères. Ces habitats linéaires semblent également ne pas convenir à plusieurs espèces d'oiseaux forestiers (Darveau et al. 1995; Whitaker et Montevecchi 1997). Imbeau et Desrochers (2002), Leboeuf (2004) et Gagné (2006) notent tout de même que les habitats linéaires résiduels dans les agglomérations de coupes totales peuvent être utilisés par les pics boréaux comme le Pic à dos rayé (*Picoides dorsalis*) et le Pic à dos noir (*Picoides articus*) en raison de la mortalité accrue des arbres par chablis à l'interface forêt/parterre de coupe. Mascarúa López et al. (2006) ont quant à eux montré que les forêts linéaires possédaient de plus faibles densités d'arbres vivants et une mortalité accrue comparativement aux forêts naturelles dans les peuplements d'épinettes de la forêt boréale de l'Ouest du Québec. Enfin, la capacité de ces forêts à fournir un habitat adéquat aux espèces qui nécessitent des structures spécifiques telles que les arbres morts de gros diamètre demeure à ce jour peu documentée.

1.4 Le Garrot d'Islande: une espèce cavicole vulnérable en forêt boréale

L'aire de répartition de la population de l'Est du Garrot d'Islande se situe presque entièrement au Québec, au nord de l'estuaire et du golfe du Saint-Laurent, entre Charlevoix et la Basse-Côte-Nord (Robert et al. 2000a, 2002), dans les domaines bioclimaciques de la sapinière à bouleau blanc de l'Est et la pessière noire à mousse de l'Est (Robitaille et Saucier 1998). Depuis la publication d'un rapport de situation en novembre 2000, la population de l'Est du Garrot d'Islande a été désignée préoccupante par le Comité sur la situation des

espèces en péril au Canada (COSEPAQ) (Robert et al. 2000b; COSEPAQ 2006). En vertu de la *Loi sur les espèces en péril* du Canada, un plan de gestion devra être préparé par le gouvernement du Canada (Anonyme 2003). Au Québec, le Garrot d'Islande fait partie de la *Liste d'espèces de la faune vertébrée menacées ou vulnérables susceptibles d'être ainsi désignées* (Gazette officielle du Québec 2003), publiée en vertu de la *Loi sur les espèces menacées ou vulnérables* (L.R.Q., c, E-12.01).

Le Garrot d'Islande est une espèce de canard cavicole qui nécessite des cavités de grande taille pour sa nidification (Evans et al. 2002). Plusieurs caractéristiques comportementales communes au genre *Bucephala* sont propres à la période de nidification. Par exemple, les garrots défendent un territoire où ils excluent les conspécifiques et souvent les autres espèces de garrots du lac de nidification (Savard 1982). De plus, les femelles recherchent leur cavité de nidification en prospectant du haut des airs. À partir de juin, les femelles immatures, qui ne nichent qu'à leur troisième année de calendrier (soit à l'âge de deux ans), ainsi que celles n'ayant pas eu de succès reproductif, feront de la prospection pour dégoter la cavité qu'elles utiliseront l'année suivante (Eadie et Gauthier 1985). Enfin, les femelles sont philopatriques (Savard et Eadie 1989), c'est-à-dire qu'elles ont tendance à retourner, année après année, aux mêmes sites de nidification; les femelles immatures retournent aussi dans les secteurs où elles sont nées pour faire leur prospection de cavités de nidification.

En Amérique du Nord, la seule étude effectuée à ce jour sur les sites de nidification naturels du Garrot d'Islande a été conduite pour la population de l'Ouest, en Colombie-Britannique, par Evans (2003). Cette étude a permis de caractériser 41 cavités naturelles, parmi lesquelles seulement deux n'avaient pas été excavées par des pics. Les arbres utilisés étaient principalement des peupliers faux-trembles (*Populus tremuloides* Michx.) (70 %) et des sapins de Douglas (*Pseudotsuga menziesii* (Mirb.) Franco) (19 %). L'utilisation des essences était proportionnelle à l'occurrence des cavités de Grand Pic, qui semble sélectionner préférablement le peuplier faux-tremble pour creuser ses cavités de nidification. Les arbres utilisés pour la nidification par le Garrot d'Islande étaient beaucoup plus gros ($\geq$ 35cm; moyenne = 38,6 $\pm$ 1,9 cm) que les diamètres moyens des arbres disponibles non utilisés (moyenne = 26,4 $\pm$ 2,2 cm). De plus, les cavités situées les plus hautes dans les arbres étaient

davantage utilisées, probablement pour éviter la prédation et parce qu'elles sont plus facilement accessibles en vol (Evans 2003).

La forêt boréale où se trouve la population de l'Est du Garrot d'Islande est un écosystème différent de celui décrit pour la population de l'Ouest jusqu'à maintenant. Les essences résineuses dominent et les arbres feuillus y sont rarement abondants (Ministère des Ressources naturelles du Québec 2002; Boucher et al. 2003), l'occurrence du Grand Pic est généralement faible (Gauthier et Aubry 1995) et la densité des peuplements est plus élevée que dans les forêts ouvertes où ont été menées les études en Colombie-Britannique. On peut donc supposer que dans les forêts coniférienes, les canards cavicoles doivent davantage dépendre, pour leur substrat de nidification, de la décomposition naturelle des gros arbres morts sur pied que de l'activité des grands excavateurs de cavités, et ce même si les cavités naturelles sont probablement moins nombreuses en forêt résineuse qu'en forêt feuillue. Malheureusement, très peu d'études ont été effectuées sur la densité et la formation des cavités naturelles non-excavées, qui semblent souvent utilisées de façon marginale pour la nidification des utilisateurs de cavités de grande taille.

1.5 Importance de l'étude

La prédominance des vieux peuplements en forêt boréale a récemment été mise en évidence (Bergeron et al. 1999; Boucher et al. 2003). Bien que la forêt boréale ne soit pas généralement reconnue pour offrir des quantités élevées d'arbres de gros diamètre comparativement à d'autres écosystèmes (Nilsson et al. 2002), on peut supposer que les forêts qui échappent aux perturbations pour de longs intervalles de temps peuvent avoir un potentiel de recrutement de gros chicots assez important, dans la mesure où la physiologie des espèces présentes le permet. Toutefois, les connaissances concernant la disponibilité et la répartition de ces arbres dans les forêts boréales matures du Québec sont restreintes (Darveau et Desrochers 2001). De plus, l'extension de l'aménagement forestier en forêt boréale soulève un questionnement quant à l'impact d'un aménagement équienne sur une faune adaptée aux vieilles forêts (Imbeau et al. 2001; Drapeau et al. 2003) et sur la valeur des forêts laissées en place en ce qui concerne la disponibilité d'éléments structuraux tels que les arbres

de gros diamètre et les arbres morts. Afin de pouvoir évaluer la valeur des forêts rémanentes dans le paysage forestier transformé par l'aménagement, il est impératif d'acquérir davantage de connaissances dans les paysages où les opérations forestières ont été jusqu'à maintenant limitées.

La perte d'habitat de nidification est l'une des principales menaces pesant sur la population de l'Est du Garrot d'Islande (Robert 2003). Le statut précaire de cette espèce oblige l'élaboration rapide de plans de conservation afin de freiner ou prévenir l'altération de son habitat en période de reproduction. La majeure partie de l'aire de nidification du Garrot d'Islande se situe en forêt boréale commerciale où des activités de récolte sont en cours ou seront incessamment pratiquées. Les connaissances acquises dans la présente étude permettront d'évaluer la qualité des différents peuplements naturels et résiduels de la forêt boréale en ce qui a trait aux arbres potentiels à la nidification du Garrot d'Islande. Des recommandations d'aménagement forestier découleront de cette étude et serviront, en outre, à préserver l'habitat de bien d'autres espèces qui utilisent les arbres morts en forêt boréale.

1.6 Objectif de l'étude

Le présent mémoire, composé de deux chapitres, a pour objectif général de caractériser les arbres potentiels à la nidification du Garrot d'Islande dans la forêt boréale coniférienne aménagée de l'Est du Québec.

Le premier chapitre vise à documenter les patrons de distribution des arbres vivants et morts dans les peuplements naturels, résiduels et récoltés de la forêt boréale de l'Est. Ses objectifs sont (1) de caractériser la disponibilité des arbres morts et vivants de gros diamètre au sein des peuplements matures et surannés dominés par différentes essences forestières et situés sur différents types de dépôt de surface couvrant la variabilité naturelle des peuplements; (2) d'analyser la distribution des classes de dégradation des arbres morts de gros diamètre et (3) de comparer la disponibilité des arbres de gros diamètre ainsi que la distribution des classes de dégradation entre les peuplements naturels, résiduels et récoltés.

Le second chapitre vise à documenter la disponibilité de cavités naturelles et excavées dans l'aire de nidification de la population de l'Est du Garrot d'Islande. Plus spécifiquement, ses objectifs sont (1) d'évaluer la présence régionale du Grand Pic en tant que principal excavateur de grandes cavités; (2) de caractériser les cavités naturelles et excavées dans les principaux peuplements forestiers naturels résiduels et récoltés; (3) de caractériser les cavités adéquates pour la nidification du Garrot d'Islande et (4) d'évaluer le potentiel du paysage forestier aménagé à produire des arbres porteurs de cavités.

En outre, ces deux chapitres seront suivis d'une conclusion générale synthétisant les principaux résultats découlant de ce mémoire ainsi que leurs implications dans la mise en oeuvre de stratégies d'aménagement forestier durable.

AVAILABILITY OF STANDING TREES FOR LARGE CAVITY NESTING BIRDS IN THE EASTERN BOREAL FOREST OF QUÉBEC, CANADA

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Abstract

Large cavity-nesting birds depend on large diameter trees for suitable nest-sites. The increased spatial extent of commercial timber harvesting is modifying forest structure over large landscapes and may compromise the availability of large trees. In this study, our objectives were to (1) characterize the availability of large living and dead trees in old-growth stands dominated by different tree species and surficial deposits that encompass the range of natural cover types of the Québec's eastern boreal forest; (2) analyze the distribution of trees among decay-classes; and (3) compare the availability of large trees between unharvested, remnant, and harvested stands for the entire range of decay-classes. A total of 116 line transects were distributed across unharvested forests, remnant linear forests, and cutblocks in cutover areas. Forest stands (black spruce (*Picea mariana*), balsam fir (*Abies balsamea*)-black spruce, balsam fir-white spruce (*Picea glauca*) and balsam fir) reflected a gradient of balsam fir dominance. Remnant forests selected were isolated since 5 to 15 years. Analyses were performed at two diameter cut-off values. Trees with DBH $\geq$ 20 cm were considered for availability of total trees whereas trees with DBH $\geq$ 30 cm were considered for availability of large trees. Forest stands comprised high proportions of standing dead trees (33 % of all stems, 8 % were large dead stems). Availability of total and large standing trees increased with the dominance of balsam fir in stands. Forest stands located on thick surficial deposits showed higher densities of large dead trees for every stand types suggesting a higher productivity on those sites. Availability of stems according to decay-classes showed a dome-shape distribution with higher densities of snags in intermediate decay stage. However, for large stems, black spruce stands showed a lower availability that was consistent across all decay-classes. Pure balsam fir stands were absent from sampled remnant forests. All remnant stands showed a dramatic decline in large trees availability compared to unharvested balsam fir stands. Densities of large dead trees increased with time since forest strips were created. Thus, availability of total standing stems (DBH $\geq$ 20 cm) is not clearly affected by forest management. However, remnant and harvested forests (strips and clearcut) are not representative of unharvested forests when considering large trees (DBH $\geq$ 30 cm) availability. Forest-dwelling wildlife relying on large diameter trees for breeding is likely to be affected by large-scale harvesting in coniferous boreal forests.

Keywords: *standing dead trees, old-growth forest, boreal forest, forest management, remnant forest, cavity-nesters*

Introduction

Dead trees are recognized as crucial attributes for biodiversity in forest ecosystems (Harmon et al. 1986; Hansen et al. 1991; Hunter 1999; Kuuluvainen 2002). A large range of vertebrate and invertebrate species use decaying trees during a period of their life cycle. Among them, cavity-nesting birds are well known for their dependence on dead trees (Von Haartman 1957; Raphael and White 1984; Zarnowitz and Manuwal 1985; Martin and Eadie 1999). Woodpeckers use snags for foraging and creating their nesting cavity, whereas secondary cavity-nesters breed in excavated cavities or seek natural cavities. Large secondary cavity users such as ducks and owls are more likely affected by the growing scarcity of large dead trees through the loss of overmature and old-growth forests in timber managed landscapes, especially in boreal forests where large trees are rare (Nilsson et al. 2002). Expansion of commercial timber harvesting across the boreal forest pinpoints the need to gain knowledge on the natural range of variation in large trees availability and quality in this forest ecosystem.

Availability of dead trees varies throughout forest's succession cycle (Spies et al. 1988). Several studies have shown the greater availability of large diameter snags in old-growth forests (Cline et al. 1980; Harmon et al. 1986; Lee et al. 1997; Sturtevant et al. 1997; Clark et al. 1998; Lee 1998; Spetich et al. 1999; Siitonen et al. 2000; Thompson et al. 2003a; Shorohova and Tetioukhin 2004). Site productivity and disturbance regimes influence recruitment rate, and decomposition rate acts on residence time of large dead trees within overmature forests (Harmon et al. 1986; Sippola et al. 1998; Spetich et al. 1999; Shorohova and Tetioukhin 2004). Abiotic factors like local climate variation, topography, and soil conditions may also influence large snag densities although few studies have directly taken them in account (but see Rouvinen et al. 2002).

Québec eastern boreal forest is characterized by long fire cycles (Gauthier et al. 2001; Bergeron et al. 2006) caused by humid regional climate (Grondin 1996), and is primarily regulated by small scale disturbances such as insect outbreaks and gap regimes (De Grandpré et al. 2000; Blais 1983; Boucher et al. 2003; Pham et al. 2004; Boucher et al. 2006). Such

disturbances enable dominance of old-growth forests (Bergeron et al. 1999; Boucher et al. 2003), which may allow wet boreal forest to offer a greater availability of large-diameter trees and snags than do dryer continental boreal forests. However, the notion of large trees varies from one forest ecosystem to another depending on the life history characteristics of tree species that dominate the canopy. Bird species with wide distribution often show different requirements in nesting tree diameter among regions. For example in western Oregon and Washington, Brown (1985) proposed minimum DBH of 30 and 60 cm for Black-backed Woodpecker (*Picoides borealis*) and Barrow's Goldeneye (*Bucephala islandica*) cavity-trees, respectively. Both values correspond to the maximum values of DBH observed for nesting trees found in Québec boreal forest for these two species (see Appendix 1). Along with size requirements, decay stage has been recognized as an important attribute of snag quality for cavity-nesting birds (Mannan et al. 1980; Raphael and White 1984; Flemming et al. 1999; Drapeau et al. 2002; Imbeau and Desrochers 2002a; Nappi et al. 2003). As lightly decayed snags are known to be used for feeding by woodpeckers (Imbeau and Desrochers 2002a; Nappi et al. 2003), moderately decayed snags are often preferred for excavating cavities, especially in coniferous species (see Bunnell et al. 1999). Highly decayed snags are susceptible to be used by weak excavators (Mannan et al. 1980; Raphael and White 1984; Harestad and Keisker 1989) and can develop natural cavities that can be utilized as nesting or roosting sites. Hence, forests with higher availability of trees at different decay stages may support a higher diversity of cavity-nesting birds.

Even-aged management strategies in landscapes dominated by old-growth forests generate major changes in the forest mosaic, increasing the proportion of young forests (regular structure) while old-growth forests (irregular structure) decrease (Hansen et al. 1991; Bergeron et al. 1999; Drapeau et al. 2000, 2002, 2003; Harper et al. 2003). Moreover, timber rotations are often shorter than natural disturbance cycles (Gauthier et al. 1996), which do not allow recruitment of old-growth structural elements in second-growth forests (Desponts et al. 2002, 2004; Roberge and Desrochers 2004). Riparian buffer strips and upland forest strips separating cutblocks are the principal remaining forest stands in even-age managed forest landscapes. Even though they are left for purposes other than retention of old-growth attributes, these strips may become refuges for forest-dwelling wildlife in intensively

managed areas. Although several studies have stated that linear forests provide shelter to numerous organisms after clearcutting operations (Darveau et al. 1995; Darveau et al. 2001; Potvin and Bertrand 2004), forest strips may not provide suitable habitats for species intolerant to fragmentation or requiring specific structures over the long-run.

The context of accelerated loss of old-growth habitats in the boreal forest as area under management progresses urges for knowledge on large tree (live and standing dead) distribution patterns. This knowledge can set baseline targets for permanent retention of legacy trees (*sensu* Franklin 1993) to maintain favourable habitat conditions for snag users, particularly large cavity-nesting birds. Likewise, in harvested landscapes it is important to determine how remnant forests may be representative with regards to their structural attributes if we intend to consider these habitats in conservation planning of forest managed landscapes. In this chapter, we document distribution patterns of live and standing dead trees in unharvested, remnant, and harvested forest stands in the eastern boreal forest of Québec. The objectives of this study were to (1) characterize the availability of large living and dead trees in old-growth stands dominated by different tree species and surficial deposits that encompass the range of natural cover types of eastern Québec boreal forest; (2) analyze the distribution of large trees among decay-classes; and (3) compare the availability of large trees between unhavested, remnant, and harvested stands for the entire range of decay-classes. We discuss how our results can serve as baseline knowledge for ecosystem management plans with regards to key habitat attributes such as large live and dead trees in the eastern boreal forest of Québec.

Study area

The study was carried out 50 km north of Saguenay River, Québec, Canada (Fig. 1), in the balsam fir-white birch biogeoclimatic zone (Robitaille and Saucier 1998). The region is characterized by hills with steep slopes and average altitude is high (640 m). Climate is cold and humid with a mean annual temperature of 0°C and mean annual precipitation varying between 1 100 and 1 300 mm (Robitaille and Saucier 1998). Mature forests in the area are composed of balsam fir (*Abies balsamea* (L.) Mill) and black spruce (*Picea mariana* (Mill.) B.S.P.) stands or mixed species stands. White spruce (*Picea glauca* (Moench) Voss) and, in a

lesser extent, white birch (*Betula papyrifera* Marsh.) are growing as codominant species. Fire cycle has been approximated to 280 years (Bergeron et al. 2006). The last spruce budworm outbreak affected numerous forest stands north of Saguenay River in the late 1970's (Morin and Laprise 1990; MNRFQ 2004). Timber harvesting has extended to the study area by the early 1980's.

Sampling design

Unharvested forest stands

Unharvested forest sites were selected according to age and species composition identified on digitized maps (MRNFQ 2004). We selected stands categorized as ≥ 90 years old because they are considered mature and ready to be harvested by the Québec Ministry of Natural Resources and Wildlife (MRNFQ). Species composition consisted in three cover types: balsam fir (> 75 % cover of balsam fir and white spruce), black spruce (> 75 % cover of black spruce) and mixed balsam fir and black spruce stands (> 75 % cover balsam fir and black spruce). Since MRNFQ's classification does not account for white spruce as a dominant tree species in their stand classification, stand types were reclassified using basal area values of each tree species measured in our sampling plots for every sampled unit because white spruce is a species that grows large diameter trees and was abundant in some forest stands of the study area.

Remnant forests and cutblocks

Québec's legislation on public lands requires forest strips retention in managed areas. Forest strips with a minimal width of 20 m (riparian forest strips) have to be maintained around permanent lakes and along streams. For cutblocks covering 100 ha and 100 to 150 ha, minimal 60 and 100 m wide forest strips (upland forests strips), respectively, have to be retained until surrounding regeneration reaches 3 m high (Québec Government 2003). Partial harvesting is, however, allowed in remnant forests strips. Within cutblocks, no live tree retention is prescribed and dead trees are often cut for worker's security. Mature and old-

growth remnant forests as well as cutblocks resulting from clearcuts with protection of soil and regeneration (CPRS) were selected over a gradient of time since harvest (from 5 to 15 years after harvesting). A total of 17 riparian strips and 18 upland forest strips aged > 90 years old as well as 25 cutblocks were selected for data collection.

Data collection

Sampling was done during the summers of 2004 and 2005. In 2004, linear transects (400 m long x 20 m wide; 0.8 ha) were surveyed in cutblocks and unharvested forest sites. In 2005, line transects in unharvested forests were reduced to 0.2 ha (100 m x 20 m) because a preliminary analysis had indicated that 0.2 ha was sufficient to adequately characterize large live and dead trees in these sites. Cutblocks line transects remained at 400 m x 20 m to account for the low stem density in open areas. Line transects in remnant forests also covered 0.2 ha and were located parallel to cutblock edge. All living and standing dead trees with diameter at breast height (DBH) ≥ 20 cm and height ≥ 1 m were recorded. Recently fallen trees were recorded in remnant stands in order to compare total standing stem density before harvesting. Species, DBH and height were measured and decay-class was assessed (Fig. 2).

Surficial deposits in the study area generally consisted in coarse glacial deposits (till) varying in thickness depending of topography. At the stand level, surficial deposit type (1A: thick undifferentiated till deposit, moderate drainage and R1A: thin undifferentiated till deposit, fast drainage), age and height were assessed using digitized forest cover maps (MRNFQ 2004). Unharvested forest stands were found on both surficial deposits, whereas remnant forests were mainly found on thick deposits. Therefore thin deposits for unharvested forest stands were not considered in the comparisons made between the two forest types.

Each tree species' basal area was calculated for each line transects and was used to reassess stand type. Stands were either classified as balsam fir stand (≥ 75 % of total basal area consisting of balsam fir), balsam fir co-dominated by white spruce (balsam fir basal area < 75 %; white spruce basal area < 75 % but higher than black spruce), balsam fir co-dominated by black spruce (balsam fir basal area < 75 %; black spruce basal area < 75 % but higher than

white spruce), and black spruce stand (≥ 75 % of total basal area consisting of black spruce). Density (number of stems per ha) was calculated for each sampling unit as a measure of availability for different components (tree species, decay-class, diameter-class) describing live and dead tree populations. After forest stands reclassification, we obtained a sample of 11 balsam fir stands, 21 balsam fir-white spruce stands, 15 balsam fir-black spruce stands and 15 black spruce stands. Within remnant forests, it resulted in 3 balsam fir-white spruce stands, 11 balsam fir-black spruce stands, and 15 black spruce stands.

Statistical analysis

Availability of live and dead trees was measured for two diameter cut-off values. After reviewing existing data on cavity-nesters requirements in the boreal forest (Appendix 1), we considered stems with 20 and 30 cm DBH values as a minimum for small and large cavity nesters, respectively. Hence, total tree availability for all cavity-nesting birds was measured on stems with DBH ≥ 20 cm and large tree availability was measured on stems with DBH ≥ 30 cm.

Two-ways ANOVA models were used to determine which stand variables (surficial deposit type and stand type for unharvested stands and stand type and time since harvest for remnant stands) influenced living and dead tree densities for total and large trees in unharvested and remnant forests. One-way ANOVAs were performed to assess differences in mean tree densities for each decay-class among stand types and to compare mean tree densities between remnant, harvested, and unharvested forests for each stand type. For all ANOVA models, variables were log transformed or square-rooted prior to analysis to meet normality and/or homoscedasticity conditions whenever necessary. When a significant difference was found, a Tukey HSD test was conducted to determine which stand types significantly differed from one another. Non-parametric Kruskal-Wallis tests were conducted when both normality and homoscedasticity could not be obtained. To know which treatment differed from each other, pairwise comparisons using Mann-Whitney tests were conducted and the significance level was adjusted with the Bonferroni correction ($\alpha = 0.0034$). Because stand type classification represented a gradient in balsam fir dominance within stand, a linear regression with surficial

deposit type as a categorical covariable was used to measure the relationships between total basal area of living trees and balsam fir basal area. Chi-square tests were used to compare the distribution of categorical variables among tree species in unharvested forests and to compare balsam fir and white birch variables in unharvested and in remnant forests. Values were considered significant at $p \leq 0.05$.

Results

Overall dead tree population

Of all the trees sampled in forest stands (unharvested and remants), 33 % and 8 % were dead trees with DBH ≥ than 20 and 30 cm, respectively. In clearcut stands, proportions of total and large dead trees were 82% and 26 %, respectively. Diameter-class distribution of dead trees differed among tree species ($\chi^2 = 533.64$, DF = 12, $p < 0.0001$) (Fig. 3). White spruce had the largest proportion of very large snags (> 40 cm DBH). Both balsam fir and white birch had higher proportions of stems with DBH ≥ 30 cm than black spruce. Black spruce was absent in the largest diameter-class. When considering all dead trees measured, balsam fir and white birch contributed to 63 % and 21 % of the large dead trees, respectively.

Living and dead tree densities in unharvested forest

The sampled stand types reflected a gradient of increasing balsam fir dominance as its density and basal area reached their maximum in balsam fir stands, their minimum in black spruce stands, and intermediate levels in mixed white spruce-balsam fir and black spruce-balsam fir stands (Table 1). The relationship between basal area of all living trees and balsam fir basal area, when controlling for surficial deposit type, was very strong ($r^2 = 0.78$).

Living and dead tree populations showed similar patterns along this balsam fir gradient, with mean DBH and total stem density increasing from pure black spruce stands to pure balsam fir stands (Table 1-2). White birch basal area also increased along the balsam fir gradient while black spruce basal area decreased. Proportions of large living and dead stems for each tree

species varied similarly in all stand types. Large balsam fir and white birch stems were more represented in the dead tree population than expected, whereas large white spruce and black spruce stems were more prone to be found in the living tree population.

Total living and dead stem density (DBH $\geq$ 20 cm) differed among stand types but not among surficial deposit types (Table 3, Fig. 4 A-C). Black spruce stands had significantly lower mean living and dead stem densities than stands with higher proportion of balsam fir. Balsam fir stands did not show significant differences with the two mixed-species stands for either living or dead stem densities. However, when looking at large trees (DBH $\geq$ 30 cm), mean stem density for both living and dead trees in balsam fir and balsam fir-white spruce stands were significantly higher than those obtained in stands where only black spruce was the dominant tree species (Fig.4 B-D). A significant relationship between stem density and surficial deposit type was found only for large diameter stems (Table 3). Stands dominated by balsam fir had a higher mean living stem density of large stems on thin deposits, whereas stands dominated by black spruce had higher mean density of large stems on thick deposits, although deposit effect was not significant (Fig. 4C). For large dead stems, availability was significantly higher for stands located on thick deposits (Fig. 4D).

Decay-class distribution in unharvested forests

Overall tree species proportions in each decay-classes showed two different distributions ($\chi^2 = 50.174$, DF = 12, $P < 0.0001$) (Fig. 5). White spruce and black spruce had a higher proportion of dead stems in recent decay-classes whereas balsam fir and white birch stem distributions were more represented in advanced decay-classes.

Decay-class density patterns showed a unimodal distribution in each stand type (Fig. 6A). Dead tree availability was higher for intermediate decayed tree (decay-class 6), and both recently dead trees (decay-class 4) and advanced decayed trees (decay-class 8) had lower densities in every stand type. However, black spruce stands tended to differ from stands with higher proportion of balsam fir with lower mean densities of intermediate decayed trees (Fig. 6A).

When looking at large dead trees (DBH $\geq$ 30 cm), the unimodal curve faded as the proportion of balsam fir in the stand decreased (Fig. 6B). Stands with a higher proportion of black spruce showed a decline in the availability of dead stems. Black spruce stands had lower availability in every decay-class except for stage 8, whereas black spruce-balsam fir stands had lower availability of intermediate stages 5 and 7.

Dead tree availability: comparisons between unharvested and remnant forests

There were no balsam fir remnant stands sampled in the study area even though forest maps indicated so (Fig. 7). Therefore, unharvested balsam fir stands could not be compared with equivalent remnant stands. Sampled forest strips were mainly represented by black spruce and balsam fir-black spruce stands and few balsam fir-white spruce stands ($n = 3$) were found in our study area. When displaying all stand types, the highest availability of total dead trees and more particularly of large dead trees, was in balsam fir stands (Fig. 7A-7B). Remnant stands had always lower dead tree densities than their unharvested counterpart. Availability of total dead stems had its lowest density values in cutblocks (Fig. 7A), but this difference was less for large dead trees compared with remnant forests (Fig. 7B). However, for remnant and cutblock stands, dead tree densities remained very low compared with unharvested fir stands.

Comparisons between unharvested and remnant forests for total dead and living tree densities showed no significant differences and these results were consistent over the three stand types (Table 4). When considering large living trees, balsam fir-black spruce stands had a significantly higher density of trees in unharvested than in remnant forests, whereas in black spruce stands there was a marginally ($p < 0.088$) higher availability of large trees in unharvested forests. For large dead trees, the difference between unharvested and remnant forests was significant for black spruce stands, and marginal ($p < 0.074$) for balsam fir-white spruce stands. Despite the large difference between unharvested and remnant stands for the latter cover type, significance in this could not be assessed probably because of low sample size (Table 4).

When comparing unharvested stands, pooled remnant stands and cutblocks with regards to stems decay-classes, every decay-class for both total and large trees showed significant differences (Table 5). Cutblocks had the lowest densities in every decay-class but means were statistically different from unharvested stands for lightly decayed snags only. In remnant forests, mean densities appeared significantly different from balsam fir dominated stands for lightly decayed large dead trees. Both remnant and cutblock stands did not show the unimodal pattern observed for unharvested stands as densities were nearly constant throughout decay-classes.

Balsam fir and white birch in unharvested and remnant forests

Balsam fir and white birch were the two main providers of large diameter snags and accounted for more than 80 % of large dead trees throughout the study area. Both species were less present in remnant forests because stands dominated by black spruce. Diameter-class and decay-class distribution showed no difference for white birch in unharvested and remnant forests ($\chi^2 = 3.052$, DF = 4, $p = 0.5492$). However, balsam fir in remnant forests had a higher proportion of small dead stems (DBH < 25 cm) and a smaller proportion of large stems (DBH $\geq$ 30 cm) than balsam fir in unharvested stands ($\chi^2 = 45.64$, DF = 4, $p < 0.0001$).

Decay-class distribution for dead stems of white birch did not show any difference in unharvested and remnant forests ($\chi^2 = 3.60$, DF = 4, $p = 0.4627$). For balsam fir, proportion of stems in decay-class 5 was higher in remnant forests and proportion of stems in decay-class 7 was lower in remnant forests than in unharvested forests ($\chi^2 = 23.792$, DF = 4, $p < 0.0001$).

Effect of time since harvest on living and dead tree availability in remnant forests

Total and large living tree density in remnant strips was linked to stand type as in unharvested stands (Table 6; see Fig. 7). Total and large trees were more numerous in remnant strips composed partly of balsam fir than in pure black spruce strips. Time since forest strip creation did not influence living tree densities. For dead trees, relationships were

not as consistent. No relation was found for total dead tree densities. However, large dead tree densities were related to both stand type and time since harvest (Table 6). Availability of large dead trees increased from strips created in the 2000's (3.5 ± 2.0 stems/ha) to those created in 1990's (18 ± 6.8 stems/ha). When testing for total standing stem and recently down stem density, stand type ($F = 7.12$, $p = 0.0041$) had a significant effect but year since harvest had no significant influence ($F = 1.76$, $p = 0.196$). However, both stand type ($F = 12.65$, $p = 0.0002$) and year of harvest ($F = 5.93$, $p = 0.0088$) had a significant effect for large stems.

Discussion

Availability of large trees in unharvested forests

Variability in the amount of dead wood in boreal forest ecosystems has been related to mortality and decay rates of trees. Mortality rate is a function of site productivity (Linder et al. 1997; Siippola et al. 1998; Rouvinen et al. 2002; Nilsson et al. 2002) and disturbance agents prevailing in the ecosystem. As for decay rate, it varies between forest types and tree species (Harmon et al. 1986; Kruys et al. 2002). Our study focused primarily on differences in the availability of large trees occurring among different stand types in boreal forests of eastern Québec. Stand types, as defined in our study, reflect a gradient of balsam fir dominance. Overall basal area of living trees was strongly related to balsam fir basal area. Bergeron and Dubuc (1989) have associated balsam fir stands with more productive sites in the boreal forest and more recently, Boucher et al. (2006) have shown that balsam fir stands reached an uneven-aged structure more rapidly than spruce dominated stands, which they also linked to site productivity. Hence, the balsam fir basal area gradient observed in our study likely reflects a gradient in site productivity. Furthermore, the association of white birch and white spruce with balsam fir stands contributes to enhance the availability of large trees since these companion species produce relatively large stems (Burns and Honkala 1990).

In our study area, snag densities in balsam fir stands were similar to those reported by Thompson et al. (2003a) in Newfoundland with 130 stems/ha for trees ≥ 20 cm DBH and by

Desponts et al. (2004) in the Gaspé Peninsula, Québec, who obtained 36.7 stems/ha for snags ≥ 30 cm DBH. However, our snag densities in black spruce stands were higher than those reported by Mascarúa López et al. (2006) and Leboeuf (2004) (18 and 20.5 stems/ha for snags ≥ 20cm, respectively) in the western part of Québec's boreal forest. Moreover, differences obtained for both living and dead tree densities between old-growth coniferous stand types support results reported by Aakala et al. (2007), who found an increasing density and volume from stands dominated by black spruce to pure balsam fir stands of Québec 's North Shore region.

Large diameter tree (DBH ≥ 30 cm) availability was also found to be influenced by surficial deposit types. The role of surficial deposits on forest structure has been recognized in several studies conducted in boreal forests of eastern Canada. In northwestern Québec, Gauthier et al. (2000) and Harper et al. (2003) found important differences in forest composition between organic and coarse deposits of coniferous forest stands. In northeastern Québec, Boucher et al. (2006) showed that environmental factors, including surficial deposits and drainage, influenced stand structure in balsam fir and black spruce stands, although site productivity was the main factor. We did not identify an effect of deposit type on forest composition, although we found structural differences among forest types in tree densities. The effect of surficial deposits we obtained for densities of large dead trees suggest that productivity could be higher on thick and moderately drained soils regardless of stand type. On the contrary, results obtained for large living trees in balsam fir dominated stands located on thin deposits indicate that a lower ratio of large dead trees was found on those sites. Because of their poor retention of water and nutrients, thin surificial deposit sites are likely to take a longer time before reaching stand breakup stage (Robichaud and Methven 1993), thus slowering the rate of dead tree recruitment. However, because this result only appeared for balsam fir stands, higher dead tree density on thick deposits (productive sites) could also be related to a higher mortality rate resulting from the last spruce budworm outbreak. Unfortunately, we cannot confirm this hypothesis since no dendrochronological analysis was conducted in our study.

We obtained an overall proportion of standing dead trees (DBH ≥ 20 cm) of 33 % in forest stands. This proportion is remarkably higher than the 10 % average reported by Nilsson et al.

(2002) for overmature boreal and temperate forests from European and North American studies. In our study we have focussed on wildlife trees, using 20 cm as a cut-off. As mortality probability increases with tree diameter (Runkle 2000; Eid and Tuhus 2001; Busing 2005), this may explain why we obtained a higher percentage of dead trees. However, in their own study experiment, Nilsson et al. (2002) found an increasing proportion of dead trees when DBH cut-off reached 50 cm. The high proportion we measured may also have resulted from the last spruce budworm outbreak that occurred in our study area, in the late 1970's (Blais 1983; Morin and Laprise 1990) suggesting that disturbance regimes also play a role in modifying the dead tree recruitment, even in stands moderately affected by insect outbreaks several years after disturbance event.

Decay-class distribution in unharvested forest

We found a dome-shaped curve distribution of total snags across decay-classes with a maximum density occurring in decay-class 6 for every stand type. Although direct comparisons of decay-class distributions have to be made cautiously because decay-classification varies between studies, several authors have reported a unimodal distribution of decay-class when describing standing dead trees (Cline et al. 1980; Jonnson 2000; Aakala 2005). Decay-class models of Norway spruce snags from Kruys et al. (2002) showed that intermediate decay-classes had a longer residence time than recent and advanced decay-classes. Using the same decay classification as ours in balsam fir and black spruce stands, Aakala (2005) showed that decay-classes associated with recently dead trees had shorter residence time than intermediate decay-classes. However, using dendrochronogical analysis, Aakala (2005) also estimated that balsam fir in decay-classes 6 and 7 were dead since approximately 16 and 21 years, respectively, a time period corresponding to the last spruce budworm outbreak in our study area. Even though our study sites were not severely affected by this spruce budworm outbreak (MNRQ 2004), a differed mortality induced by weaker defoliation may well have taken place. Although mortality has been quantified in areas were severe defoliation has occurred (e.g MacLean 1980; Bouchard et al. 2005), few studies have paid attention to moderately disturbed stands. Unfortunately, our sampling design does not allow to further document this explanation. Yet we can reasonably presume that the dome-

shaped distribution of degradation class we obtained likely resulted from a difference in residence time among decay-class combined with the effect of the last spruce budworm outbreak.

The lower availability of each decay-class for total and large snags in black spruce dominated stands is probably the result of a low density effect. Aakala (2005) obtained a dome-shape distribution with a maximum density for decay-class 5 when sampling trees with DBH $\geq$ 10 cm. Covering a larger diameter range would have allowed us to see a more realistic distribution pattern of decay-classes, given the lower abundance of large trees in those stands. However, black spruce stands do not provide a wide variety of degradation classes for large trees. Although current knowledge of tree degradation and outbreak effects prevent us to predict future variation of decay-class distribution, balsam fir stands are predisposed to offer greater variety of degradation classes because of their higher densities of large trees, and consequently, to offer a better potential for wildlife trees.

Effect of forest management on large dead trees

Impact of forest management on primary and secondary cavity nesting birds have been reported by various authors in many forests ecosystems (Harmon et al. 1986; Hansen et al. 1991; Ohman et al. 1994; Imbeau et al. 2001; Drapeau et al. 2002; Bütler et al. 2004; Everett and Otter 2004; Jonsson et al. 2005). In boreal forests, recent acceleration of forest harvesting has lead to studies addressing concerns about impacts of extensive forest management on specific forest structures such as snag availability in second growth forests (Sturtevant et al. 1997; Desponts et al. 2003, 2004; Drapeau et al. 2002, 2003, 2005) as well as on forest-dwelling wildlife (Doyon et al. 1999; Imbeau et al. 1999; Drapeau et al. 2000; Thompson et al. 2003b; Gagné 2006). In Québec's managed boreal forests, riparian and upland forests strips are often the only forest stands with old-growth characteristics remaining along with forests located on steep slopes, inaccessible to machinery. Unlike riparian forests in hardwood ecosystems, boreal riparian forests are not considered as unique and diversified habitats (MacDonald et al. 2006). Although they may harbour different bird communities (Whitaker and Montevecchi 1997) and a higher bird richness (LaRue et al. 1995; Macdonald

et al. 2006), boreal riparian forests tend to be similar to boreal interior forests in terms of richness of mammals and amphibians, and vegetation structure (Hannon et al. 2002; Macdonald et al. 2006). However several studies have showed that man-induced forest strips, either riparian or non-riparian, showed structural differences when compared to unharvested forests. In western Québec, Mascarúa López et al. (2006) showed that remnant boreal forests had lower density of living trees and lower canopy cover as compared to interior forest. Whitaker and Montevecchi (1997) obtained lower standing dead trees densities for riparian and non riparian strips in wet balsam fir forests in Newfoundland.

We did not find significant differences in total dead tree densities between unharvested stands and forest strips. However, when looking at large trees, black spruce stands showed a significantly lower abundance for both living and dead trees. As for mixed species stands (balsam fir-black spruce and balsam fir-white spruce), the trend was less clear. Although it is generally stated that windthrow is the principal cause for decreasing stem density in remnant strips (Esseen and Renhorn 1998; Ruel et al. 2001), uprooted trees did not seem prevalent in our study sites. Lowest densities in remnant forest types could not be the result of partial cutting, which is allowed for riparian and upland strips, since no harvest was conducted after strips were created in our sites. Lower availability of trees seemed to reflect more a productivity effect than a disturbance effect, as balsam firs were smaller in DBH in remnant stands and less abundant in advanced decay-class than in interior stands, which suggests a lower productivity in remnant forests. Furthermore, large living tree populations did not vary between years since forest strip creation. However, total large stem density (living, standing dead and recently down stems) varied between years since timber harvesting, indicating a difference in the initial stem density of stands. Indeed, local variation in productivity were observed in our study area and, considering that timber harvesting tend to be spatially aggregated, stem density was probably influenced by both time since harvest and local variations in productivity. The increased availability of large dead trees over time since harvest was conducted suggests, however, a recruitment of standing dead trees probably caused by environmental changes such as canopy opening, drought or heavy wind exposure (Esseen 1994; Peltonen 1999).

All remnant stands showed dramatically lower large trees availability compared to unharvested balsam fir stands. The fact that we did not encounter pure balsam fir stands in the 29 remnant strips sampled raises questions about the compositional importance of forests strips regarding the retention of key structural elements such as large diameter standing dead trees. The reasons why balsam fir stands were absent can be twofold. First, for riparian forest strips, ecological conditions near lakes may not favour the establishment of pure balsam fir stands because of environmental factors such as seasonal flooding in lower elevations (Denneler et al. 1999). Whitaker and Montevecchi (1997) also reported a higher black spruce and lower white birch basal areas in riparian forests surrounded by pure balsam fir stands. This stand composition difference probably has an effect on large trees availability, at least in regions where balsam fir is abundant as in our study area. On the other hand, upland forest strips are resulting from decisions made by forest managers, so strip selection may well be made with the objective of maximizing harvested volumes. Stands with lower basal area may thus be retained as long as they comply with harvesting regulations, which do not provide guidelines for the densities of large trees (live or dead) in these remnant forests.

Besides differences in availability of large diameter trees, remnant and harvested stands showed a different distribution of decay-classes. As cutblocks had lower availability for every decay-class compared to balsam fir dominated stands, differences were exacerbated for large and advanced decayed trees in remnant forests. Considering that the main species responsible for the excavation of large cavities, Pileated Woodpeckers (*Dryocopus pileatus*), is nearly absent in our study area (Vaillancourt, Chapter 2), large excavated cavities are very rare. Therefore, natural cavities availability probably constitutes a critical attribute for large secondary cavity nesters. Although natural cavity formation is a rare event in coniferous forests (Bai et al. 2003; Vaillancourt, Chapter 2), trees in later stage of decay are the most susceptible to support them especially for coniferous species (e.g. Bunnell et al. 1999). Therefore, maximizing the retention of large trees in advanced stages of decay in managed landscapes could contribute to ensure availability of this type of structure. However, guidelines for maintaining over, on the long run, an adequate distribution of trees across the decay-class gradient will require more research on the decay dynamics of tree species in the

different landscape contexts generated by timber harvesting (single tree retention within clearcuts, patch retention in clearcuts, remnant blocks of forests).

Management implications

Availability of total dead trees was not so greatly reduced in remnant stands, so one can suppose that cavity nesters, particularly those using small cavities, may not be limited by tree availability in these residual stands, although other factors such as edge effects (Imbeau and Desrochers 2002b; Gagné 2006), fragmentation, or predation (Darveau et al. 1997) may have an impact on their occurrence. However, the decline in large dead trees at the landscape scale due to the extent of harvesting areas and the absence of rich large tree density stands within remnant strips raises major concerns about the capacity of these forest remnants to provide appropriate structures for large cavity nesters and sustain local populations. Moreover, the possibility of future harvesting in remnant strips (since trees located in remnant strips are included in the annual allowable cut estimates), which is provided in Quebec's current forestry regulations, may threaten the overall availability of large diameter trees in these residual stands.

The eastern boreal forest of Québec is naturally dominated, particularly on mesic site conditions, by balsam fir stands that harbour the largest trees in this ecosystem. However, the corresponding high timber volume of these stands has leaded them to be harvested in priority during the last 20 years (Ministère des Ressources naturelles du Québec 2002). One consequence of this shift towards the prevalence of forest management as a main disturbance agent in this ecosystem is the considerable reduction in the availability of large trees. Our study has shown that linear remnant habitats near riparian environments do not harbour the largest balsam fir trees. Accordingly, forest planners should leave larger forest strips to increase availability of large live and dead trees in remnant strips of these harvested areas. Moreover, when deciding upland forest strips location, stands with large live and dead trees, ideally productive balsam fir stands, should be prioritized to ensure the retention of high quality snags, and these forests should not be harvested but rather included in a permanent retention management strategy. Québec's current forestry regulations do not promote the

retention of high quality forest stands in the managed landscape that offer habitat conditions (including large live and dead trees) of older forests for biodiversity maintenance. Along with other studies (Potvin and Bertrand 2004; Darveau et al. 1995, 2001; Mascarúa López et al. 2006; Gagné 2006), we urge managers to consider leaving a greater variability of forests remnants in terms of width, composition and structural attributes, including large trees, in order to maintain suitable habitat for forest-dwelling wildlife.

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Figure 1. Location of our study area 50 km north of Saguenay River in Eastern Québec, Canada (48°33'N; 70°18'W).

Figure 2. Decay-stage classification for living and dead trees (modified from Imbeau and Desrochers 2002a) used to characterize decay stages of sampled trees within our transect lines. Decay-classes 4 to 8 represent dead trees.

Figure 3. Proportion (%) of dead stems by tree species among DBH class in unharvested stands. White spruce provided the highest proportion of large diameter stems, and both balsam fir and white birch had higher proportions of stems in higher DBH classes than black spruce.

Figure 4. Effects of stand type and surficial deposit on living and dead tree availability for total (DBH ≥ 20 cm) and large trees (DBH ≥ 30 cm). Total tree densities (A-C) were only affected by stand type and showed a decreasing density towards black spruce stands, whereas large tree densities (B-D) were also influenced by surficial deposit type (see Table 3). Note the changing scale on y axis.

Figure 5. Proportion (%) of dead stems by tree species among decay-class in unharvested stands. This figure shows that black and white spruce were more represented in recently decayed snags, whereas balsam fir and white birch had higher proportions of stem in advanced decay-classes.

Figure 6. Mean densities of dead trees with regards to decay-class in unharvested stands. (A) This figure shows that total dead trees (DBH ≥ 20 cm) are more abundant in intermediate decay-classes for each stand type and (B) this dome-shape curve fade for black spruce dominated stands when only considering large dead trees (DBH ≥ 30 cm).

Figure 7. Availability of dead trees in unharvested stands, remnant stands and cutblocks. (A) This figure shows that availability of total dead trees (DBH ≥ 20 cm) is similar between unharvested and remnant stands but (B) differences appeared for balsam fir-white spruce and black spruce stands regarding large trees (DBH ≥ 30 cm). Remnant and harvested stands have considerably lower large trees densities than balsam fir stands (see Table 4).

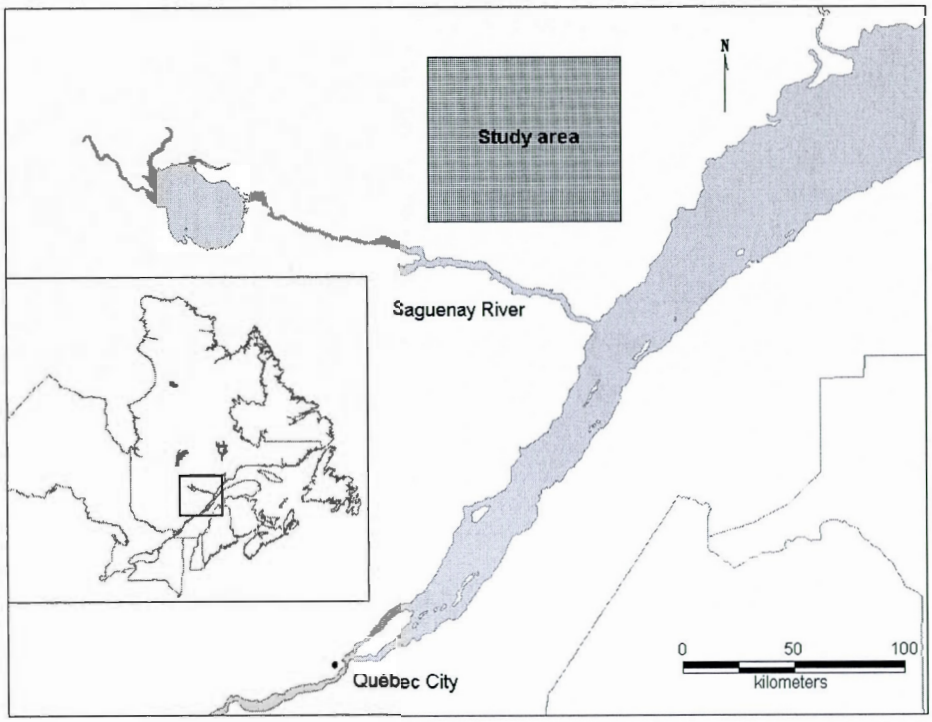

Figure 1. Vaillancourt et al.

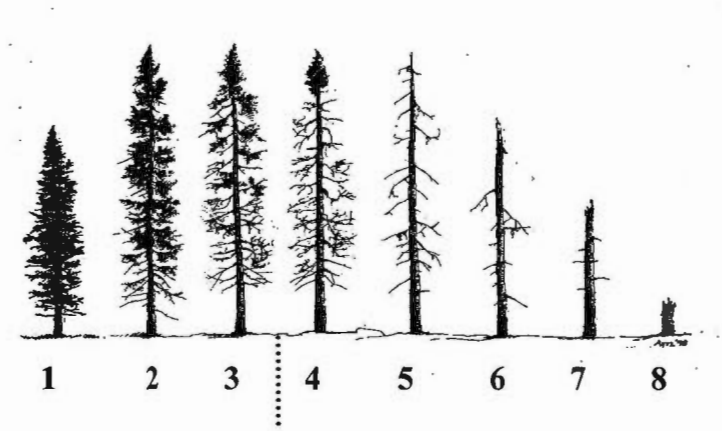

Figure 2. Vaillancourt et al.

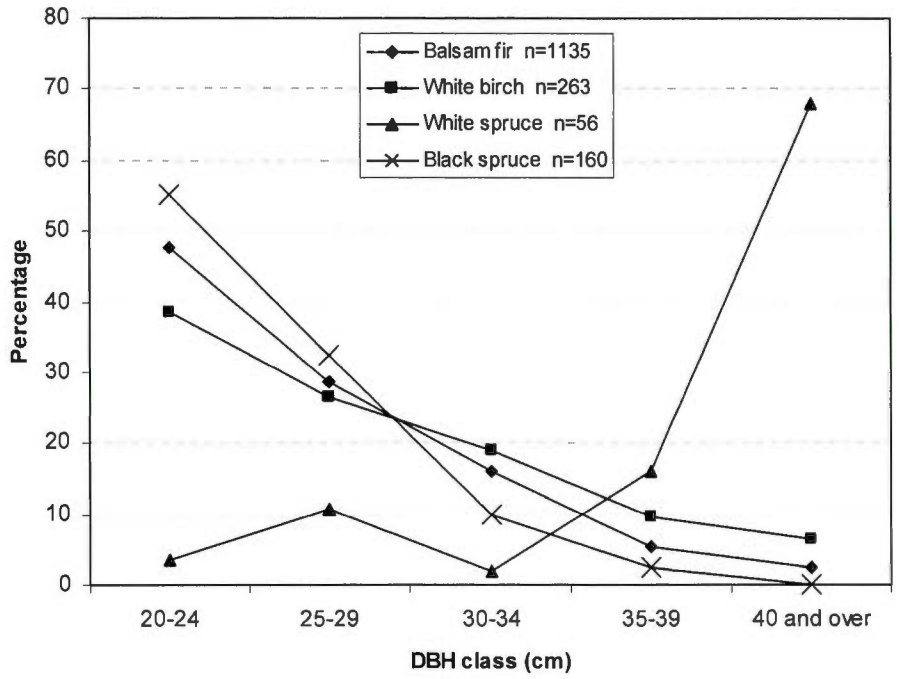

Figure 3. Vaillancourt et al.

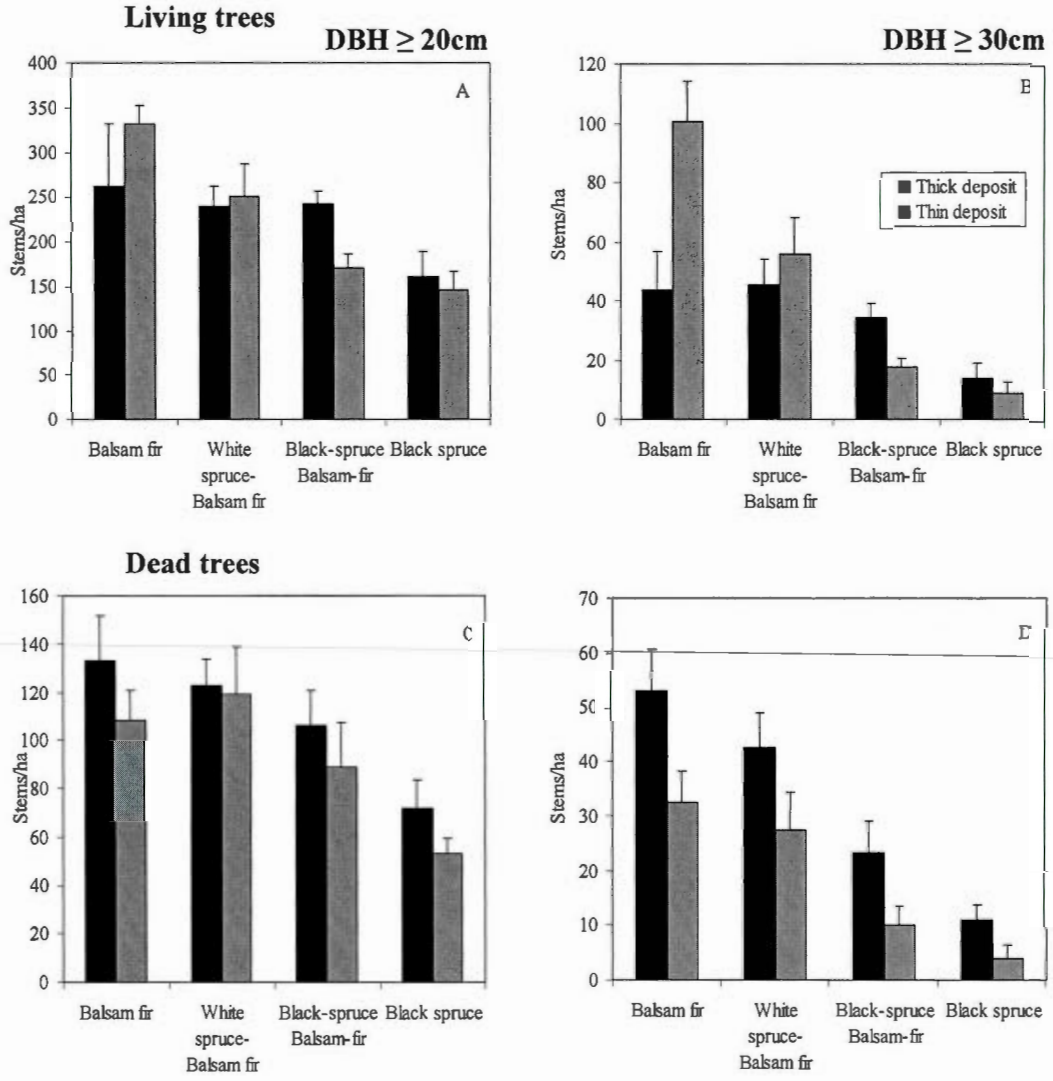

Figure 4. Vaillancourt et al.

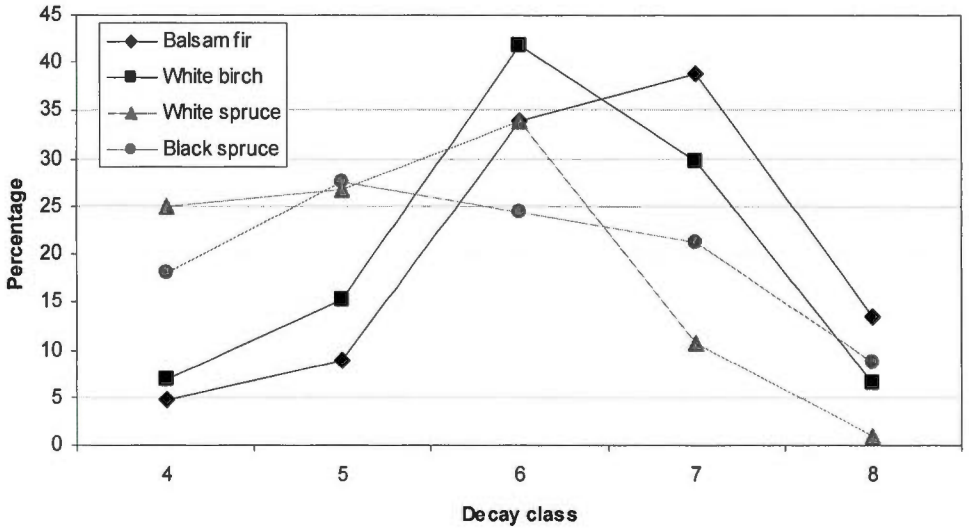

Figure 5. Vaillancourt et al.

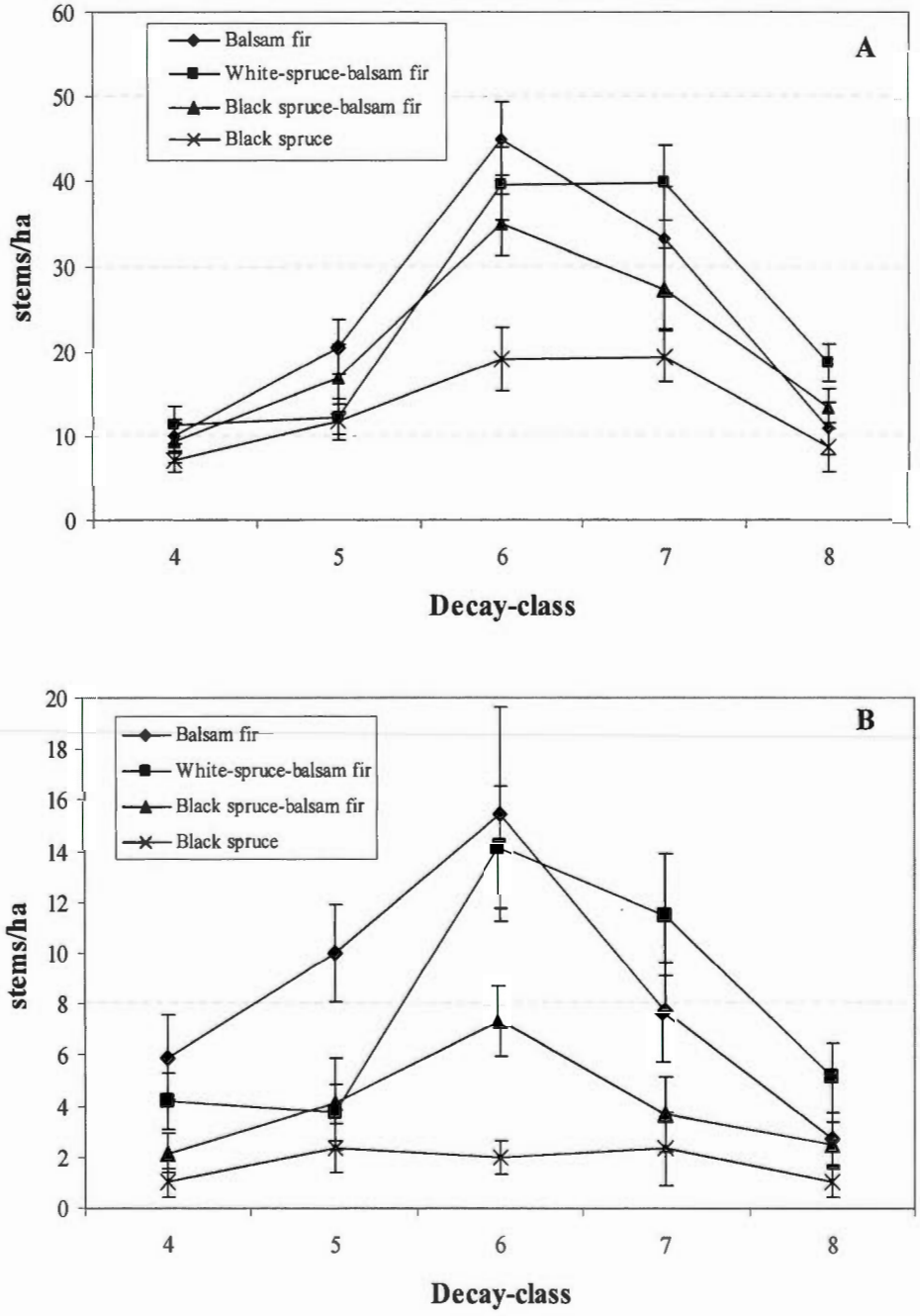

Figure 6. Vaillancourt et al.

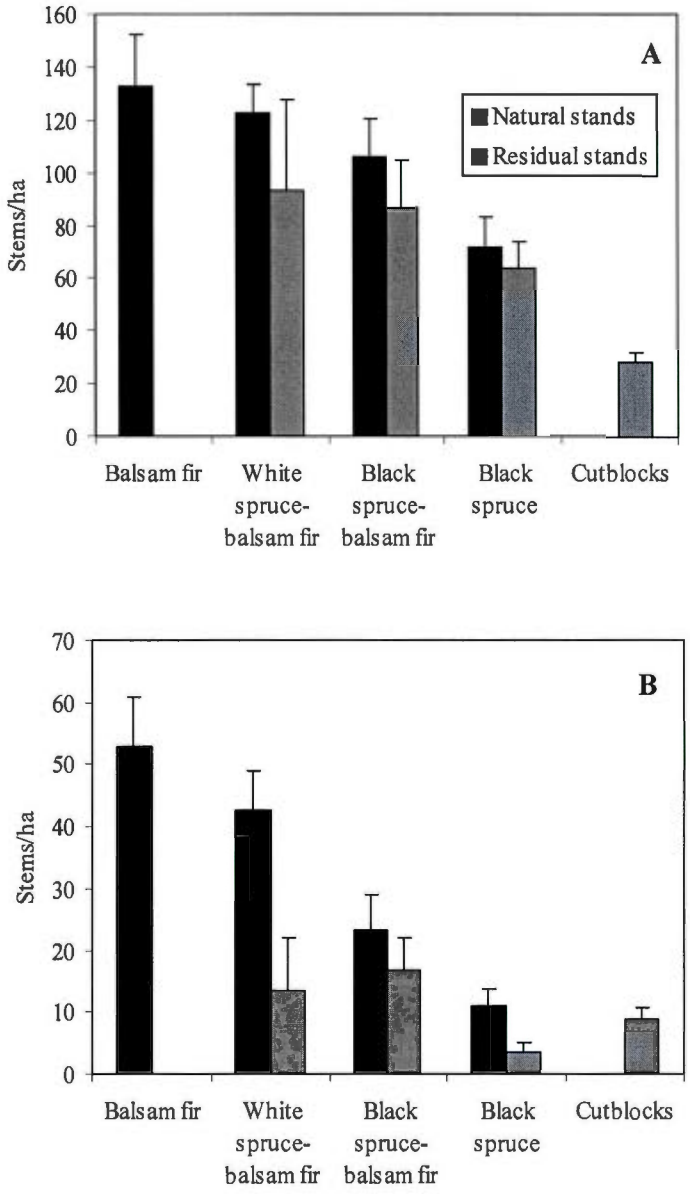

Figure 7. Vaillancourt et al.

Table 1. Characteristics of living trees among four cover types in eastern boreal forest of Québec, Canada, 2004-2005.

Characteristic	*Black spruce*	*Balsam fir-Black spruce*	*Balsam fir-White spruce*	*Balsam fir*
Total no. of trees sampled	466	1408	1334	659
No. of stands	15	15	21	11
Mean DBH (cm)*				
Total	23.9 (0.2)	24.2 (0.1)	25.9 (0.2)	26.4 (0.2)
Thick deposit	23.9 (3.8)	24.2 (4.7)	25.8 (5.7)	25.3 (5.7)
Thin deposit	23.8 (3.4)	23.8 (3.8)	26.4 (6.9)	27.0 (5.4)
Mean basal area (DBH ≥ 20cm) (m^2/ha)*				
Total	7.1 (0.9)	10.8 (0.8)	13.6 (1.2)	17.0 (2.0)
Balsam fir	0.18 (0.08)	4.3 (0.5)	7.0 (0.8)	12.5 (1.5)
White birch	0.069 (0.035)	0.60 (0.15)	1.0 (0.2)	1.7 (0.2)
White spruce	0.28 (0.26)	1.6 (0.3)	4.8 (0.6)	2.7 (0.4)
Black spruce	6.6 (0.8)	4.3 (0.4)	0.80 (0.18)	0.22 (0.09)
Mean density (stems/ha)*				
Trees with DBH ≥ 20cm	155.3 (19.7)	223.3 (13.2)	242.3 (18.9)	299.1 (33.9)
Thick deposit	160.5 (89.0)	242.6 (42.5)	238.9 (88.7)	261 (57.2)
Thin deposit	145 (48.6)	170 (30.8)	250.8 (88.2)	330.8 (53.4)
Trees with DBH ≥ 30cm	12.3 (3.4)	30.0 (4.0)	48.4 (6.6)	75.0 (12.5)
Thick deposit	14 (15.4)	34.5 (15.7)	45.4 (30.5)	44 (28.8)
Thin deposit	9 (8.2)	17.5 (6.5)	55.8 (30.2)	100.8 (32.2)
% of large stems density (DBH ≥ 30cm)†				
Balsam fir	0	24	37	68
White birch	0	3	12	11
White spruce	14	40	47	20
Black spruce	86	34	4	1

*Standard deviations are in parentheses

†Represents the distribution of large stems among species for each stand types

Table 2. Characteristics of dead trees among four cover types in eastern boreal forest of Québec, Canada, 2004-2005.

Characteristics	*Black spruce*	*Balsam fir-Black spruce*	*Balsam fir-White spruce*	*Balsam fir*
Total no. of snags sampled	197	613	771	263
No. of stands	15	15	21	11
Mean DBH (cm)*				
Total	24.6 (0.3)	25.4 (0.2)	28.4(0.2)	28.3 (0.4)
Thick deposit	24.7 (4.3)	25.7 (5.5)	28.7 (6.9)	29.2 (7.7)
Thin deposit	24.4 (3.5)	24.2 (4.0)	26.8 (6.7)	27.4 (6.1)
Mean basal area (m^2/ha)*				
Total	3.2 (0.5)	5.5 (0.8)	7.8 (0.9)	8 (0.8)
Balsam fir	1.2 (0.3)	3.1 (0.6)	4.6 (0.5)	5.0 (0.6)
White birch	0.10 (0.08)	0.64 (0.11)	1.2 (0.2)	1.7 (0.4)
White spruce	0	0.18 (0.09)	1.1 (0.3)	0.73 (0.38)
Black spruce	1.7 (0.2)	0.41 (0.08)	0.005 (0.005)	0.05 (0.05)
Mean density (stems/ha)*				
Snags with DBH ≥ 20cm	65.7 (7.9)	101.6 (11.5)	121.8 (9.2)	199.5 (11.0)
Thick deposit	72 (35.2)	106.3 (47.6)	122.8 (41.0)	133 (42.1)
Thin deposit	53 (14.8)	88.8 (37.1)	119.2 (48.7)	108.3 (29.9)
Snags with DBH ≥ 30cm	8.7 (2.2)	19.8 (4.6)	38.2 (5.2)	41.8 (5.4)
Thick deposit	11 (8.9)	23.3 (19.3)	42.5 (25.2)	53 (17.2)
Thin deposit	4 (5.5)	10 (7.1)	27.5 (16.4)	32.5 (13.7)
% of stems density DBH ≥ 30cm†				
Balsam fir	19	53	53	59
White birch	4	10	20	26
White spruce	0	8	11	9
Black spruce	62	4	0	0

*Standard deviations are in parentheses

†Represents the distribution of large stems among species for each stand types. Percentage does not add up to 100 % because of unidentified species

Table 3. Summary of two-way ANOVAs for the effects of stand type and surficial deposit type on total and large living and dead trees densities (See Fig. 4).

Source of variation	*df*	MS	*F*-Value	*P*
Total trees (DBH ≥ 20 cm)				
Living tree density				
Stand type	3	54.209	7.08	0.0004
Deposit	1	0.017	0.002	0.963
Stand type x deposit type	3	11.869	1.55	0.212
Error	54	7.657		
Dead tree density				
Stand type	3	29.274	7.85	0.0002
Deposit	1	7.792	2.09	0.154
Stand type x deposit type	3	0.678	0.182	0.908
Error	54	3.728		
Large trees (DBH ≥ 30 cm)				
Living tree density				
Stand type	3	58.76	20.03	< 0.0001
Deposit	1	4.03	1.17	0.2836
Stand type x deposit type	3	14.03	4.09	0.011
Error	54	3.4328		
Dead tree density				
Stand type	3	49.04	16.59	< 0.0001
Deposit type	1	26.902	9.10	0.0039
Stand type x deposit type	3	0.206	0.07	0.976
Error	54	2.956		

Table 4. Comparisons of living and dead tree densities between unharvested and remnant stands for each forest type in eastern boreal forest Québec, Canada, 2004-2005 (See Fig. 7). Mean values and standard deviations (in parentheses) are reported.

Stand type	*n*	*Unharvested forest*	*n*	*Remnant forest*	*Analysis*	*Error DF*	*P*
Total living trees (DBH ≥ 20 cm)							
Balsam fir-White spruce	15	238.9 (21.2)	3	236.7 (48.4)	ANOVA	17	0.967
Balsam fir-Black spruce	11	242.6 (22.2)	9	208.3 (24.6)	ANOVA	19	0.315
Black spruce	10	160.5 (23.1)	15	105.3 (18.9)	ANOVA	23	0.078
Total dead trees (DBH ≥ 20 cm)							
Balsam fir-White spruce	15	122.8 (11.3)	3	93.3 (25.3)	ANOVA	17	0.303
Balsam fir-Black spruce	11	106.3 (15.3)	9	86.7 (16.9)	ANOVA	19	0.401
Black spruce	10	72.0 (12.0)	15	63.7 (9.8)	ANOVA	23	0.596
Large living trees (DBH ≥ 20 cm)							
Balsam fir-White spruce	15	45.4 (7.5)	3	43.3 (16.9)	Mann-Whitney	1	0.812
Balsam fir- Black spruce	11	34.5 (4.0)	9	18.3 (4.4)	ANOVA	19	**0.015**
Black spruce	10	14.0 (2.4)	15	5.0 (2.8)	Mann-Whitney	1	0.088
Large dead trees (DBH ≥ 20 cm)							
Balsam fir-White spruce	15	42.5 (6.2)	3	13.3 (13.9)	ANOVA	17	0.074
Balsam fir-Black spruce	11	23.3 (5.4)	9	16.7 (5.9)	Mann-Whitney	1	0.154
Black spruce	10	11.0 (1.2)	15	3.7 (1.8)	Mann-Whitney	1	**0.016**

Table 5. Comparisons of total and large dead trees between unharvested stands, remnant stands and cutblocks broken by decay-class in eastern boreal forest of Québec, Canada, 2004-2005. Mean densities are reported with standard deviations in parenthesis. For each decay-class, values with different letters show significant differences among stand types.

	Balsam fir	*Balsam-fir White spruce*	*Balsam fir-Black spruce*	*Black spruce*	*Remnant stands*	*Cutblocks*	*P*
DBH ≥ 20 cm							
Decay-class 4 [a]	12.0 (± 3.7) a	10.4 (± 2.7) a	10.8 (± 2.9) a	7.3 (± 1.7) a	7.8 (± 1.3) a	1.0 (± 0.3) b	<0.0001
Decay-class 5 [b]	25.0 (± 4.5) a	11.5 (± 2.8) a	18.9 (± 5.2) a	14.0 (± 2.4) a	14.6 (± 2.6) a	2.2 (± 0.6) b	<0.0001
Decay-class 6[a]	48.0 (± 8.3) ab	39.3 (± 5.2) a	38.1 (± 4.1) ab	23.0 (± 3.8) abc	25.5 (± 4.8) b	9.4 (± 1.6) c	<0.0001
Decay-class 7 [a]	33.0 (± 10.8) ab	41.9 (± 5.7) a	26.4 (± 6.4) ab	20.0 (± 4.2) ab	18.9 (± 3.8) b	9.3 (± 1.2) b	<0.0001
Decay-class 8	15.0 (± 3.5) ab	19.8 (± 2.5) a	12.2 (± 2.1) ab	8.5 (± 3.9) b	9.4 (± 1.7) b	6.2 (± 0.9) b	<0.0001
DBH ≥ 30 cm							
Decay-class 4 [c]	7.0 (± 2.0) a	3.8 (± 1.3) a	2.5 (± 1.0) ab	0.9 (± 0.6) b	0.7 (± 0.3) b	0.5 (±0.2) b	0.001
Decay-class 5[c]	15.0 (± 1.6) a	3.6 (± 1.2) b	5.6 (± 2.3) ab	2.7 (± 1.2) b	3.3 (± 1.1) b	1.0 (±0.3) b	0.003
Decay-class 6 [a]	25.0 (± 7.1) a	16.5 (± 3.2) a	8.6 (± 1.7) ab	2.3 (± 0.8) bc	2.4 (± 0.9) bc	3.2 (±0.9) c	<0.0001
Decay-class 7 [a]	3.0 (± 1.2) b	13.8 (± 2.9) a	4.1 (± 2.0) b	3.2 (± 1.9) b	1.7 (± 0.7) b	2.5 (±0.6) b	<0.0001
Decay-class 8 [c]	3.0 (± 1.2) a	5.5 (± 1.6) a	2.5 (± 1.1) a	0.9 (± 0.6) a	0.9 (± 0.4) a	1.8 (±0.3) a	0.044

[*] Data have been square rooted to obtain homoscedasticity

[†] Data have been log transformed to obtain homoscedasticity

[§] Non-parametric Kruskal-Wallis and pairwise Mann-Whitney tests have been conducted for those densities

Table 6. Summary of two-ways ANOVAs for the effects of stand type and year of harvest on total and large living and dead trees densities in remnant forests.

Source of variation	*Df*	MS	*F*-Value	*P*
Total living tree density (Sq root)				
Stand type	2	5.39	7.85	0.0027
Year of harvest	2	0.66	0.95	0.4007
Error	22	0.69		
Total dead tree density				
Stand type	2	1894.36	0.94	0.4046
Year of harvest	2	4173.52	2.08	0.1491
Error	22	2008.77		
Large living tree density				
Stand type	2	1760.21	25.06	< 0.0001
Year of harvest	2	160.84	2.29	0.1249
Error	22	70.23		
Large dead tree density				
Stand type	2	557.62	5.93	0.0087
Year of harvest	2	411.37	4.38	0.0251
Error	22	93.97		

Appendix 1. Data on mean cavity nesting tree's DBH for primary and secondary cavity nesting birds in boreal forest throughout Canada.

Species	*Location*	*Main tree species*	*n*	*min*	*max*	*Mean DBH*	*SE*	*SD*	*Source*
Data from Québec, Canada									
Red-breast Nuthatch	Abitibi	Boreal mixedwood	6	19.1	50.5	30.3	4.9		Nappi et al. unpublished
Northern Flicker	Abitibi		9	28.4	58.9	38.4	3.0		Nappi et al. unpublished
Duck sp.	Abitibi		5	38.4	45.5	44.1	1.9		Nappi et al. unpublished
Saw-whet Owl	Abitibi		3	46.4	58.9	52.1	3.7		Nappi et al. unpublished
Three-toed Woodpecker	Abitibi	Black spruce	3	22.5	32.8	28.8	3.2		Nappi et al. unpublished
Black-backed Woodpecker	Charlevoix	Black spruce	97	13.8	33.0	22.0	0.4		Nappi et al. unpublished
Common Goldeneye	Portneuf	Balsam fir	6	36	69	50	5.1		Maisonneuve et al. 2002
Hooded Merganser	Portneuf		4	30	84	52	11.4		Maisonneuve et al. 2002
Barrow's Goldeneye	Saguenay	Balsam fir	5	34	66	40.8	2.4		M.Robert CWS-Québec, pers.comm.
Data from British Columbia, Canada									
Red-breast Nuthatch	Central B.C.	Boreal mixedwood	24	17.3	36.2	26.4	1.0		Harestad and Keisker 1988
Downy Woodpecker	Central B.C.		5	19.1	31.4	26.3	2.4		Harestad and Keisker 1988
Northern Flicker	Central B.C.		17	19.8	48.7	31.9	2.4		Harestad and Keisker 1988
Red-breast Nuthatch	Interior B.C.	Boreal mixedwood	166			25.7		13.5	Martin et al. 2004
Downy Woodpecker	Interior B.C.		26			25.1		6.2	Martin et al. 2004
Three-toed Woodpecker	Interior B.C.		20			25.8		7.2	Martin et al. 2004
Northern Flicker	Interior B.C.		236			35.4		10.9	Martin et al. 2004
Saw-wet Owl	Interior B.C.		10			38.1		9.3	Martin et al. 2004
American Kestrel	Interior B.C.		26			40.7		8.0	Martin et al. 2004
Bufflehead	Interior B.C.	Boreal mixedwood	89			36.3	0.9		Evans 2003
Barrow's Goldeneye	Interior B.C.		41			47.2	2.0		Evans 2003

CAVITIES AVAILABILITY FOR BARROW'S GOLDENEYE (*BUCEPHALA ISLANDICA*) IN THE EASTERN BOREAL FOREST OF QUÉBEC, CANADA

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Abstract

The eastern North American population of Barrow's Goldeneye (*Bucephala islandica*), which is listed as "Special Concern" in Canada, breeds in the eastern boreal forest of Québec, Canada. In the western North American population, in British Columbia, this species is known to nest primarily in abandoned Pileated Woodpecker (*Dryocopus pileatus*) cavities, whereas nesting requirements for the eastern population remain poorly documented. In this paper, we characterize the availability of natural and excavated cavities in managed coniferous boreal forest landscapes within the breeding area of the eastern population of Barrow's Goldeneyes. More specifically, our objectives were to: (1) determine the regional abundance of the Pileated Woodpecker, a species that directly affects Barrow's Goldeneye cavity availability; (2) measure the availability of natural and excavated cavities in harvested, remnant and unharvested stands; (3) partial out cavities that have suitable characteristics with regards to Barrow's Goldeneye breeding requirements; and (4) assess the potential of managed landscapes to provide cavity trees for Barrow's Goldeneyes. Woodpecker surveys were conducted using a modified version of the Breeding Bird Survey method in our study area and in the adjacent southeastern area to include both the boreal coniferous landscape and the adjacent lowland mixedwood landscape. Cavities were searched in 116 line transects distributed in unharvested and linear remnant balsam fir (*Abies balsamea*) and black spruce (*Picea mariana*) stands and in cutblocks. No Pileated Woodpeckers were detected in the breeding area of Barrow's Goldeneye, its distribution being restricted to lowland area were trembling aspen is abundant. Of the 110 cavities found, 50 were natural and 60 had been excavated by smaller size woodpeckers. All cavities were in standing dead trees and both excavated and natural cavities occurred mainly in balsam fir snags, though proportionately to its availability in the study area. Mean cavity density and cavity tree DBH were highest in balsam fir stands. Only 10 trees (0.2 % of all trees sampled) supported cavities that were considered suitable for the breeding of Barrow's Goldeneyes, in terms of dimensions and canopy openness above cavity. Suitable cavities were mainly (89 %) apical (chimney) in relatively short snags showing advanced decay state. We determined a DBH threshold for each tree species after which probability of cavity occurrence is enhanced (potential cavity trees) using ROC analysis. Remnant linear forest sites had lower potential tree densities than their unharvested equivalent. Large cavities are thus a rare component in this boreal forest landscape and current forest

management practices reduce the potential of this landscape to provide cavity trees. We discuss alternative management methods needed to ensure retention and recruitment of large decaying trees in Barrow's Goldeneye breeding areas.

Keywords: *Barrow's Goldeneye, cavity-nesting birds, boreal forest, remnant forests*

Introduction

The worldwide distribution of Barrow's Goldeneyes (*Bucephala islandica*) is divided into two distinct populations in North America and one in Iceland. The western North American population comprises the majority of individuals with an estimated population of 200,000 goldeneyes spread out mostly west of the Rocky Mountains in Canada and Alaska (Eadie et al. 2000), whereas the eastern North American population is estimated at 6000 individuals and is nearly restricted to Québec province (Robert et al. 2000, 2002; Robert and Savard *in press*). Iceland comprises a small population varying between 1000 and 2000 goldeneyes (Einarsson et al. 2006). The eastern North American population (hereafter eastern population) of Barrow's Goldeneyes is the only North American population of *Bucephala* species considered at risk, the Committee on the Status of Endangered Wildlife in Canada having designated the population as 'of special concern' in 2001 (COSEWIC 2006).

As all *Bucephala* species, Barrow's Goldeneye is a secondary cavity nester which uses cavities located in trees as nesting sites (Peterson and Gauthier 1985; Eadie et al. 1995; Eadie et al. 2000). The Icelandic population breeds, however, in rock cavities (Einarsson 1997). Barrow's Goldeneyes use artificial nest boxes for breeding when those are provided (Savard 1988; Evans et al. 2002; Savard and Robert 2007; Einarsson pers. comm.). In western North America, several studies conducted in mixedwood forests have documented the use of abandoned Pileated Woodpecker (*Dryocopus pileatus*) cavities by Barrow's Goldeneye for nesting (Bonar 2000; Eadie et al. 2000; Martin et al. 2004). In a comparative study between nest boxes and natural cavities in British Columbia, Evans et al. (2002) found 41 natural nesting cavities of which 39 were large excavated cavities. Cavity trees used were predominantly trembling aspen (*Populus tremuloides* Michx.), and in a lesser extent, Douglas fir (*Pseudotsuga menziesii* (Mirb.) Franco), that had larger diameters than available trees (Evans 2003).

In eastern North America, Barrow's Goldeneye range is mainly located across Québec's eastern boreal forest (Robert et al. 2000, 2002) where long fire cycles (Gauthier et al. 2001; Bergeron et al. 2004, 2006) provides on a high proportion of the land base, old-growth

coniferous forest cover that include large diameter trees (Aakala et al. 2007; Vaillancourt, Chapter 1). However, these trees are mainly coniferous species, deciduous species being rare within this ecosystem (MRNQ 2002; Boucher et al. 2003). So far, there is no published information on breeding cavities used by eastern population of Barrow's Goldeneyes (see Robert et al. 2000; Savard and Robert 2007) and the potential of eastern boreal forests to provide large cavities remains poorly known but is presumably lower than in mixedwood boreal forest of western North America for two reasons. First, Pileated Woodpecker's occurrence in coniferous boreal forest is low and this species reaches its northern limit at the southern edge of Barrow's Goldeneye's distribution range in Québec (Gauthier and Aubry 1995). Consequently, large excavated cavities are likely to be rare, exacerbating the critical role of natural cavities for large secondary users such as the Barrow's Goldeneye. Second, although very few studies have documented the occurrence of natural cavities in coniferous tree species (see Bai et al. 2003), these may be less prone than hardwood species to develop hollows because of the absence of branches preventing the formation of large limb holes.

Timber harvesting has become the main disturbance in Québec eastern boreal forest during the past decade (MRNQ 2002). This region is thus experiencing a net loss of older forests (> 100 years) and the availability of large trees (live and dead) may be reduced with regards to its original importance under natural disturbance regimes (Gauthier et al. 2001). Hence, recruitment of large cavity trees has probably been and will certainly be again modified throughout the Barrow's Goldeneye eastern population breeding range. This loss of habitat and the concomitant loss of key structural attributes (e.g. large decaying trees) have been identified as major threats for species associated to mature and overmature forests (Gutierrez and Carey 1985; Hansen et al. 1991; Imbeau et al. 2001).

In this paper, we document the availability of natural and excavated cavities in managed coniferous boreal forest landscapes within the breeding area of the eastern population of Barrow's Goldeneyes. More specifically, our objectives were to (1) determine the regional abundance of cavity excavators, particularly the Pileated Woodpecker, a species that directly affects Barrow's Goldeneye cavity availability; (2) measure the availability of natural and excavated cavities in unharvested, remnant and harvested forests; (3) partial out cavities that

have suitable characteristics with regards to Barrow's Goldeneye breeding requirements; and finally, (4) assess the potential of unharvested, remnant and harvested forests to provide cavity trees. We discuss how this baseline knowledge can be integrated in conservation strategies for Barrow's Goldeneye and other large cavity users in eastern boreal forests.

Methodology

Study area

The study was conducted north of Saguenay River in Zones d'Exploitation Contrôlée (ZEC; controlled wildlife harvesting zones) Chauvin and Martin-Valin (48°33'N; 70°18'W) (Fig. 1), covering 1500 km^2 of which half has been under recent extensive timber harvesting. This area is part of the balsam fir-white birch bioclimatic zone and is characterized by steep hills with an average altitude of 640 m (Robitaille and Saucier 1998). The forest cover is predominantly composed of stands regenerating from the different harvest operations that occurred in the last 15 years. Currently, unharvested mature and older (> 90 years old) forests correspond to 20 % of the study area including blocks of forest left on steep terrain. Mature and older forest stands are dominated by balsam fir (*Abies balsamea* (L.) Mill.) or black spruce (*Picea mariana* (Mill). B.S.P.). White spruce (*Picea glauca* (Moench) Voss) and white birch (*Betula papyrifera* Marsh.) are also present as codominant species and associated to balsam fir stands (Vaillancourt, Chapter 1). The first breeding record of Barrow's Goldeneyes in eastern North America originated from the study area in 1998 (Robert et al. 2000). Annual surveys conducted on 60 lakes of the study area in May and June for the 1999-2003 period yielded between 24 and 43 pairs each spring (Savard and Robert 2007).

Woodpeckers' surveys

To determine the importance of primary excavators, particularly Pileated Woodpeckers, in the breeding area of Barrow's Goldeneye, we conducted a survey using a modified version of the Breeding Bird Survey method (Sauer et al. 2000). This survey was conducted in our study area and in the adjacent southeastern area to include both the boreal coniferous landscape and

the adjacent lowland mixedwood area where Pileated Woodpecker is more likely to occur given the importance of second-growth trembling aspen stands. A total of 75 stations were located along the forest road network (ca. 300 km). Thirty, 20, and 25 stations were located in coniferous forests, regenerating clearcuts, and mature mixedwood stands dominated by trembling aspen, respectively. All stations were at least 3 km apart, and all were visited three times between 28 May and 20 June 2005 to measure the presence of woodpeckers using playback. Calls were played three times successively at each station between 5:00 a.m. and 11:00 a.m. and consisted in territory calls and drumming recordings that lasted 1 minute followed by a listening period of 30 seconds. Calls were played with this procedure for all woodpecker species known to breed at this latitude (i.e., Downy, Hairy, Black-backed, American Three-toed Woodpeckers (*Picoides pubescens, P. villosus, P. articus, P. dorsalis*); Yellow-bellied Sapsucker (*Sphyrapicus varius*); Northern Flicker (*Coleaptes auratus*); and Pileated Woodpecker). Here, we only report Pileated Woodpecker occurrence patterns. This species was considered present at a sampling station if it was seen or heard once in any of all three visits.

To compare the distribution of Barrow's Goldeneyes and Pileated Woodpeckers, we used data originating from an helicopter survey conducted by the Canadian Wildlife Service (Québec Region). This survey was designed specifically for Barrow's Goldeneyes and covered 424 lakes of our study area between 28 May and 1 June 2001 (Robert et al. *in prep.*). Distances between the two distribution ranges and to sites where species occurred were measured using ARCGIS 9.1.

Site selection for cavity survey

The study area is a forest mosaic that comprises a variety of regenerating coniferous and mixedwood stands that has been harvested since the early 1980's combined with widespread natural forest stands. In managed areas, Québec legislation requires the retention of a 20 m wide forest strip around lakes and along streams and rivers (riparian strips) and, since 1996, clearcut blocks must be separated by upland forest strips that vary in width from 60 m (when cutblocks are less than 100 ha) to 100 m (when cutblocks are larger than 100 ha) (Québec

Government 2003). To have a sample of all forest cover types, we selected sites from both the unharvested and harvested portions of our study area. In chapter 1, we found that the composition of forest stands is an important driver in large trees availability within this system, balsam fir stands harbouring more large trees than black spruce stands. Hence, a stratified (black spruce vs. balsam fir dominance) random procedure was applied on digital forest cover maps (MRNFQ 2004) to select accessible stands of unharvested forests and remnant strips (riparian and upland) of 90 years or more (commercial timber harvesting age). Dominance in the composition of the forest cover was based on the criterion that either black spruce or balsam fir covered more than 50 % of the stand total basal area. Overall 35 balsam fir stands and 56 black spruce stands were selected for line transect sampling. Twenty-five clearcut sites were also randomly selected throughout the study area and consisted in clearcuts with protection of regeneration and soil (CPRS) harvested between 1981 and 2000.

Cavity survey

We surveyed cavities along line transects covering 0.2 ha (100 m x 20 m) in forest sites, and 0.8 ha (400 m x 20 m) in cutblocks, because of lower standing tree density. All living and dead trees with diameter at breast height (DBH) $\geq$ 20 cm and height $\geq$ 1m were measured. Species, DBH and height were noted and decay-class was assessed (Fig. 2). Each tree was inspected with binoculars for cavities and we used an infra-red video camera installed at the end of a perch and joined to a Sony Handycam® CCD-TRV65 NTSC camera to verify the presence of suspected cavities. We first determined if a cavity was naturally formed or excavated by a woodpecker. Natural cavities were classified as lateral or apical (chimney cavity). Based on size and shape, woodpecker cavities were classified as small (created by Downy Woodpecker), medium (Hairy, Black-backed or American Three-toed Woodpeckers), large (Northern Flicker) or very large (Pileated Woodpecker). For each cavity, we measured entrance height and noted the orientation of opening for lateral cavities. All cavity locations were recorded with a GPS.

Cavity suitability

To find suitable nesting cavities, goldeneyes do cavity prospecting which consists in searching and visiting cavities during summer, in preparation for the next breeding season (Eadie and Gauthier 1985). Considering that Barrow's Goldeneyes search nesting cavities while flying, we assumed detectability by prospecting females could be influenced by canopy obstruction above cavity. Hence, to estimate the susceptibility of cavities to be detected by prospecting birds, we used a semi-quantitative measure of canopy closure according to the following categories: 1) entirely clear; 2) partially closed; and 3) entirely closed.

To assess cavity suitability for Barrow's Goldeneyes, we measured floor diameter and cavity depth in all cavities. Entrance diameter was also measured for lateral cavities. Critical values were derived from nests recently discovered by the Canadian Wildlife Service on the study area (n = 5; MR., *unpubl. data*) and data obtained from nests found in British Columbia (n = 41, ME., *unpubl. data;* see Evans 2003). In both studies minimal values obtained for cavity floor diameter were 16 and 14 cm; cavity depth, 15 and 25 cm; and entrance dimensions, 40 x 11 cm and 10 x 10 cm, respectively. Based on these data, minimal cavity floor diameter and cavity depth were fixed to 14 cm and 15 cm, respectively. We considered entrance opening diameter larger than 10 cm as adequate. To be considered suitable, cavities had to meet all three criteria and be located under entirely clear or partially closed canopy.

Statistical analyses

Preliminary comparisons between remnant forest strips and forest stands in unmanaged areas did not show significant differences for the above response variables. Hence, a non-parametric Kruskal-Wallis analysis was conducted on pooled sampling sites (strips and stands) by corresponding cover type. Comparisons of densities and mean DBH and height of cavity trees were made among three treatments: clearcuts, balsam fir, and black spruce cover types. Non-parametric Kruskal-Wallis tests were conducted because both normality and homoscedasticity could not be assessed. We conducted pairwise comparisons using Mann-Whitney tests. To avoid type I errors (probability of incorrectly rejecting a true null

hypothesis) the statistical significance of Mann-Whithney test was adjusted with the Bonferroni method for a significance level of $p = 0.017$ (0.05/3 pairwise comparisons). Chi-square tests were used to compare the distribution of categorical variables (canopy closure and decay-class) and proportion of cavity trees among treatments.

Projections on the potential of different forest cover types to provide cavity trees for Barrow's Goldeneye was estimated with regards to tree species and DBH, two factors deemed to have an effect on the presence of cavities at the tree level. First, logistic regression models on presence or absence of a cavity were conducted (one for each tree species), with DBH as the independent variable. Second, for each model, we used the receiver-operating characteristic (ROC) curve to determine the optimum probability of presence of a cavity for which the fit between observed and predicted observations of the regression model is maximized. This optimum probability (hereafter cut-off value) is derived from 2 x 2 confusion matrices that compare observed values of presence or absence with predictions from the regression model and is reported on a graph plotting sensitivity (proportion of true positive correctly predicted) and specificity (proportion of true negative correctly predicted) values across the entire range of detection probabilities of the response variable (in our case, presence of cavities) (Manel et al. 2001). Model performance can be assessed from the area under the curve (AUC) where a value of 1.0 indicates a perfect model and a value of 0.5 indicates no significant differences between presence and absence (DeLeo 1993). We chose the maximum accuracy to determine cut-off values, i.e. the point where both sensitivity and specificity are maximised (DeLeo 1993). Cut-off values for each tree species was thus selected on the basis of an optimum, instead of a default probability value of 0.5 that has often been used in ecological studies dealing with specie's occurrence data (Fielding and Bell 1997; Collingham et al. 2000; Manel et al. 2001; Guénette and Villard 2004, 2005). Third, to determine the corresponding DBH threshold values, we used the equation parameters of the regression model by isolating the x_t parameter in the following regression formula:

$$x_t = \frac{\ln [1/c-1]+\beta_0}{-\beta_1}$$

where β_1 is the coefficient of the logistic regression, β_0 is the intercept and c is the optimum cut-off value obtained from ROC analysis. Finally, even though tree decay stage may be an important feature in Barrow's Goldeneye nesting trees, our projections did not consider this attribute given our lack of knowledge on transition rates between decay-classes of snags (*sensu* Kruys et al. 2002) for the tree species under study.

Comparisons of different forest cover types to provide potential cavity trees was calculated at the stand scale as the total density of potential trees (trees above the DBH threshold values obtained with ROC analyses). Five treatments were considered as our independent variables: unharvested black spruce stands and balsam fir stands, remnant black spruce and balsam fir strips, and clearcuts. In this case, it was important to determine how remnant habitats differed from natural forests (controls) given the increasing importance of managed forests in this part of the boreal ecosystem. We conducted a one-way ANOVA to compare these densities among treatments. Our response variable (potential tree density) was square-rooted prior to the analysis to meet normality and homoscedasticity conditions. A HSD-Tukey test was conducted *a posteriori* to assess differences between treatments.

Results

Occurrence and distribution of Barrow's Goldeneyes and Pileated Woodpeckers

Pileated Woodpeckers were detected at 7 (9 %) of our playback stations. Occurrences were not evenly distributed across the study area, all of them being located in mature mixedwood stations southeast of the area used by Barrow's Goldeneyes (Fig. 3). The shortest distance between woodpecker and goldeneye locations was 8 km. When tracing occurrence ranges for our two species, the distance between the northern limit of Pileated Woodpecker range and the southern edge of Barrow's Goldeneye range was only 1 km. The two distribution ranges were therefore contiguous with a very small overlap.

Characteristics of trees supporting excavated and natural cavities

We found 110 cavities on 86 of the 7746 measured trees in the 116 line transects sampled (Table 1). All cavities, natural and excavated, were found on standing dead trees (hereafter snags). Most excavated cavities were from medium size woodpeckers, with few small and large cavities and no very large excavated cavities (Fig. 4). Of the 50 natural cavities found, 67 % were apical and 33 % were lateral. Cavity densities for both types (natural and excavated) varied among treatments (Table 1). Balsam fir stands harboured three times more cavities than black spruce stands and clearcuts. The proportion of cavity trees differed among treatments ($\chi 2 = 16.474$; DF = 2; $p = 0.0003$), being lower in black spruce stands and higher in balsam fir stands and clearcuts. Mean DBH of cavity trees varied among forest types and was smaller in spruce stands than in balsam fir stands and clearcuts (although marginaly significant), whereas mean cavity tree height did not differ significantly among treatments (Table 1).

Significant differences were observed among forest types in canopy closure surrounding cavity trees ($\chi 2 = 19.942$; DF = 4; $p = 0.005$; Table 1). Black spruce stands had the highest proportion of cavity trees under closed canopy (52.9 %), whereas cavities found in clearcuts were more often located under an entirely open canopy (78.6 %). Balsam fir stands had a higher proportion of cavity trees under partially closed canopy (47.4 %). No significant differences was observed between treatments ($\chi 2 = 11.137$; DF = 6; $p = 0.08$; Table 1) with regards to cavity tree decay-class distribution. All forest types had proportionally less cavity trees in recently dead trees. Both forest stands had a higher proportion of cavity trees in decay-class 7, whereas cutblocks had a higher proportion in decay-class 6.

Cavities were generally found in balsam fir snags, although excavated cavities in clearcuts were mainly found in white birch snags (Table 1). No significant differences were found when comparing distribution of species between cavity trees and non-cavity trees throughout forest types ($\chi 2 = 4.146$; DF = 5; $p = 0.5286$; Fig. 5) indicating that cavities are occurring in tree species according to their prevalence in the area.

Suitability of cavities for Barrow's Goldeneye

Since no very large excavated cavity were found, further field measures of cavities for determining their potential suitability for breeding Barrow's Goldeneyes could only be conducted on natural cavities. Among the 2866 snags measured during this study, we conducted detailed measurements on 45 natural cavities. Among those, 27 did not meet our suitability criteria whereas 10 (22 %) cavities met the criteria and were partly visible (Table 2). Characteristics of all suitable cavities, including those under closed canopy, are shown in Table 2. More than half of visible cavities were located in clearcuts, whereas cavities considered not visible were more evenly distributed among stand types. Only one lateral cavity was suitable, the remaining being apical. The majority of cavities (61 %) were in highly degraded balsam fir trees (decay-class 7-8).

Availability of potential natural cavity bearing trees in unharvested and remnant old-growth forests, and cutblocks

Trees DBH thresholds obtained from ROC analyses were similar among tree species (Table 3). Model accuracy for each species was moderate ($0.6 \leq AUC \leq 0.71$) (given the low sample size of cavities) but showed a trend for cavity occurrences in larger trees. Density of trees corresponding to these thresholds (potential trees) differed among forest types (Table 4). Potential cavity tree density was higher in balsam fir stands than in spruce stands and clearcuts (Fig. 6). Finally, unharvested stands had higher numbers of potential cavity trees than corresponding linear stands for both balsam fir and spruce stands.

Discussion

Availability of cavities

Pileated Woodpeckers were absent from areas used by Barrow's Goldeneyes. All Pileated Woodpecker occurrences were located in mature mixedwood stands, mainly resulting from old clearcuts, where trembling aspen was abundant (MRNFQ 2004). The association of

Pileated Woodpeckers with trembling aspen in mixedwood boreal forest is well established for western Canada, where several studies showed that Pileated Woodpeckers preferred this tree species to excavate nesting cavities (Harestad and Keisker 1989; Martin and Eadie 1999; Martin et al. 2004). Trembling aspen was also recognized as a keystone species for woodpeckers in the eastern boreal mixedwood forest, where this tree species is significantly preferred for cavity excavation (Gasse 2007). Finally, trembling aspen is susceptible to fungi attacks early in its maturation, making it easy to excavate before it dies (Harestad and Keisker 1989).

The absence of Pileated Woodpeckers in Barrow's Goldeneye breeding area is unlikely to be related to lack of large diameter trees (> 20 cm), which are abundant in our study area (Vaillancourt, Chapter 1). However, these large trees are mainly conifers and white birches. The scarcity of large trembling aspens in the Barrow's Goldeneye breeding area and the close association of Pileated Woodpecker with trembling aspen probably explain why no very large excavated cavity was detected during this study. In Québec, trembling aspen is more common in the western portion of the boreal forest (Saucier et al. 1998), where fire cycles are much shorter than in the eastern portion of the boreal forest (Gauthier et al. 2001; Bergeron et al. 2006). Dominance of old-growth coniferous stands in the eastern portion of Québec boreal forest (Bergeron et al. 1999; Boucher et al. 2003), may also explain why the Pileated Woodpecker is a rare breeder overall on the Québec North Shore (Gauthier and Aubry 1995).

Considering that the breeding range of the eastern population of Barrow's Goldeneye is confined to eastern Québec (Robert et al. 2000, 2002), the contribution of Pileated Woodpecker in the supply of large cavities for this secondary user is likely to be unsignificant, if not trivial; which is contrary to what prevails for the western North American population. The largest excavated cavities found in our study area were created by the Northern Flicker. Although a few Barrow's Goldeneye nesting sites have been identified as enlarged flicker cavities (Palmer 1976; Bellrose 1980), such cavities are normally considered too small for Barrow's Goldeneyes and are rather utilized by the smaller size Bufflehead (*B. albeola*) (Peterson and Gauthier 1985; Evans et al. 2002). Thus, all these

results are indicating that the eastern population must strongly rely on natural cavities for nesting.

Differences were observed within our three stand types concerning cavity availability, quality (e.g. diameter and decay-class) and detectability. Although differences in methodology do not allow direct comparisons with other studies, the cavity densities we obtained for balsam fir stands were in the same order of magnitude than densities reported for boreal forests in a review made by Remm et al. (2006) for temperate and boreal forests. Furthermore, balsam fir stands offered a higher intermediate canopy closure, larger cavity trees, and a higher abundance of potential cavity trees as identified by our threshold analysis. Balsam fir stands are recognized to offer higher densities of large diameter trees compared to spruce dominated stands (Aakala et al. 2007; Vaillancourt, Chapter 1), which surely contributes to increase the densities of cavity trees. In clearcuts, the lower abundance of cavities is mainly explained by the reduction in stem density considering that we found a similar proportion of cavity trees among dead trees than in balsam fir stands. Consequently, tree density reduction contributes to enhance degree of canopy openness around cavities, therefore increasing detectability of cavities in those areas.

In boreal mixedwood forests, it has been shown that trembling aspen plays an important role in communities of cavity nesting birds (Peterson and Gauthier 1985; Harestad and Keisker 1989; Martin et al. 1999; Aitken et al. 2002; Gasse 2007). Although we found a high proportion of cavities in balsam fir, we can not say that balsam fir represents a key species because the proportion of cavities on this tree species was proportional to its availability in the study area. All cavities detected during this study were on dead trees, predominantly on highly degraded trees, especially in natural forest stands. For excavated cavities, since all cavities encountered were not recently created we can not determine at which decay stage they were excavated and we can not therefore exclude that some old cavities may have been excavated in live trees. The formation of natural cavities, particularly apical (chimney type), is, however, clearly a long process of decay for both coniferous species and white birch. Thus, we suggest that tree decay stage may be a more reliable predictor of cavity occurrence

than tree species, and that high densities of advanced decaying trees may represent a key element for providing large nesting cavities for Barrow's Goldeneyes.

Suitable cavities for Barrow's Goldeneye

Since very little information is available on the use of natural nesting sites by Barrow's Goldeneye in boreal forests, suitability assessment relied exclusively on physical constraints related to size opening and detectability. From 7746 trees including 2866 snags, only 18 (0.2 %) contained a cavity large enough to theoretically provide a nesting site for Barrow's Goldeneyes, which clearly indicates that natural cavities suitable for this bird are rare in coniferous boreal forests. Furthermore, all suitable cavities were in snags showing an advanced degradation stage. This contrasts with other studies conducted on *Bucephala* species in North America, which mostly described nesting sites as excavated cavities or natural hollows occurring in living or slightly decaying hardwood species (Prince 1968; Peterson and Gauthier 1985; Gauthier and Smith 1987; Evans 2003). Several studies conducted on Wood Ducks (*Aix sponsa*) in riparian forests also reported similar descriptions for natural nesting sites (Gilmer et al. 1978; Soulliere 1988; Lowney and Hill 1989; Robb and Bookhout 1995; Yetter et al. 1999). Processes involved in the ontogeny of cavities are not well understood and comprise stochastic factors which make them difficult to predict (Lindenmayer et al. 1993). We could not identify a tree species with a higher potential to support cavities, yet we found that larger trees were more susceptible to bear natural cavities. Large decaying trees are therefore essential structural attributes of forest stands for Barrow's Goldeneyes.

Within a conservation perspective, reliance on highly degraded snags implies two important elements that could influence availability of potential nesting sites for Barrow's Goldeneyes. First, advanced decayed snags may not remain standing for several years; therefore lifespan of natural cavities is probably not very long. Additional studies on recruitment rate of snags and on formation of hollow trees during decay process are needed to understand the dynamics of natural cavities. Determining transition rates between decay-classes with dendrochronological measurements of time since death (Kruys et al. 2002; Aakala 2005)

could refine our projections through time and provide a more dynamic and precise forecast than our coarse estimate of potential densities provided in our projections of potential cavity trees. Second, apical cavities occur in degraded snags that are in old-growth forests where canopy closure may vary with regards to the size of the gap created by tree death, leading to a variable detectability to prospecting goldeneye females. Gaps and edges are common in our study area because natural old-growth forest stands make up an important proportion of the land base (De Grandpré et al. 2000). For prospecting females, these canopy openings probably represent essential stand features that enhance cavity detection through visual and physical access. Pham et al. (2004) have shown that small gaps are far more numerous than large gaps created by the death of multiple trees in the coniferous cover types in the Québec North Shore region. Variability in gap size was higher in balsam fir stands than in black spruce stands (Pham et al. 2004). However, black spruce stands showed a more clustered pattern of mortality (Aakala et al. 2007) which probably explains the higher occurrence of larger gaps in these stands (Pham et al. 2004). But since black spruce stands do not provide large quantities of large diameter trees (Aakala et al. 2007; Vaillancourt, Chapter 1), these larger openings may not lead to a higher availability of cavities for large cavity nesters. Hence, more research on tree mortality and gap characteristics related to detectable cavity trees and on prospecting behaviour constraints are necessary for increasing our understanding of Barrow's Goldeneye efficiency in detecting cavity trees in this old-growth forest cover.

For the purpose of this study, all cavity trees above 1 m high were considered. Evans (2003) reported that Barrow's Goldeneyes selected cavities that were higher above the ground than available ones and no nesting cavities were found so far in trees shorter than 3.1 m in our study area (MR, unpublished data). It is therefore possible that cavities located close to the ground might not be selected by goldeneye females because of higher risk of predation. Moreover, the majority of cavities defined as suitable in our study showed parameters close to minimal ranges identified by other studies (see methods). Hence, cavity size is known to influence reproductive output in cavity nesting passerines (Karlsson and Nilsson 1977; Rendell and Robertson 1989, 1993). However, relationship between cavity size and reproductive success has not been investigated so far in cavity nesting-ducks and would help refining our knowledge on cavity quality. Moreover, suitability, as defined in the current

study, is strictly based on size parameters. Obviously, local and contextual variables (e.g. distance to water, fragmentation) and behavioural constraints (e.g. opening of flypath to cavity, intra or inter specific competition) may also influence availability and selection of cavities by nesting goldeneyes. Pöysä et al. (1999) have shown that prospecting Common Goldeneye females visited equally nest-boxes at shoreline and inside forest, but chose preferentially those located near water. A study conducted by Dow and Fredga (1985) failed to identify significant influence of contextual variables in the choice of nesting sites in a nest-boxes network (see also Savard and Robert 2007), but showed that females tended to visit preferentially nest-boxes in which there was a successful breeding the previous year, a result also reported by Savard (1988) and Zicus and Hennes (1989).

Forest management implications

Our results clearly show that current forest management based on short rotations with clearcutting as the main silvicultural practice is likely to strongly reduce the availability of cavities for large secondary cavity nesters such as Barrow's Goldeneyes. The lowest densities of potential cavity trees were observed in clearcut areas but remnant linear strips also had a lesser amount of suitable trees than their natural counterparts, even though no partial harvesting was conducted in these remnants. A lower abundance of standing dead trees in linear forest strips has been observed in several other studies (Whitaker and Montevecchi 1997; Gagné 2006; Vaillancourt, Chapter 1). Even though we found presumably highly detectable cavities in clearcuts, the absence of recruitment in dead trees will eventually lead to a deficit in cavities and potential cavity trees, as dead decaying trees will collapse. Given that managed forests will make up the major part of Québec's eastern boreal forest in the very near future, reduction in nesting tree availability is likely to exert an additional pressure on this population at risk throughout its range. This large scale shift from an old-growth forest landscape mosaic to a landscape dominated by forests in early serial stage requires the implementation, on a permanent basis, of variable retention strategies (*sensu* Franklin et al. 1997) of biological legacies (including large live and dead trees) (Franklin et al. 2000, 2002).

Regulations in Québec public lands with regards to forest cover retention in cutover areas are not oriented towards maintaining biological legacies of older forests with the perspective of biodiversity maintenance. Instead, the rationale for setting aside remnant linear habitats (riparian and upland strips) is either for water quality protection (riparian buffers) or for facilitating game wildlife movements (upland strips). Furthermore, these habitats can be partly or totally harvested. First, harvesting trees is allowed in riparian buffers up to the level of leaving 500 stems/ha of commercial size (> 10 cm DBH). Second, upland strips are entirely harvested when regenerating cutover areas reach 3 m in height. Clearly, management guidelines for implementing permanent retention of biological legacies in managed landscapes are critically needed, and this even more in a regional context such as the one within Barrow's Goldeneye range where most of natural forests are dominated by old-growth stands with large live and dead trees.

Within Barrow's Goldeneye range, in clearcut areas, single tree and patch retention of large living and dead trees, particularly balsam fir, should be applied in order to ensure recruitment of dead trees while surrounding forests regenerate. Riparian strips should be enlarged enhance large dead trees availability and maintain interior forest conditions (see Mascarúa López et al. 2006). Because of their higher densities of large potential trees, old-growth balsam fir stands should also be prioritized when deciding the location upland forest strips. These remnant habitats should be left on a permanent basis and thus be excluded from annual allowable cut (AAC) estimates.

Finally, another management approach that should be developed to favor the retention of key structural attributes such as large live and dead trees in the eastern boreal forest would be to shift conventional forest management based on even-aged management with single clearcutting practice to an ecosystem-based approach that diversifies forestry practices by including partial and selection cutting to generate structural conditions of older forests (Bergeron et al. 1999, 2002; Gauthier et al. 2001; Bergeron 2004). Such new management approach should be based on prior knowledge of natural disturbance dynamics (Attiwill 1994; Angelstam 1998; Delong and Tanner 1996; Bergeron and Harvey 1997; Bergeron et al. 2002; Delong 2002). Targets for the supply of habitat attributes at the stand and landscape

levels could include knowledge on Barrow's Goldeneye snag requirements such as the one obtained through our projections. Within this ecosystem-based management perspective, preventing temporal and spatial discontinuity of large dead trees availability may well be a prerequisite for long-term maintenance of Barrow's Goldeneye's eastern population. This will require major changes in Québec's forest management planning process that must simultaneously plan timber retention and harvest at the landscape scale.

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List of figures

Figure 1. Location of the study area, north of Saguenay River, Québec, Canada, where woodpecker surveys and cavity searches were conducted to document cavity availability and suitability for Barrow's Goldeneye.

Figure 2. Decay-stage classification used for living and dead trees sampling (modified from Imbeau and Desrochers 2002) within the 116 transect lines of our sample set. Decay-classes 4 to 8 represent dead trees.

Figure 3. Pileated Woodpecker's occurrence at 75 playback stations during May and June 2005 and lakes where Barrow's Goldeneyes was observed according to an helicopter survey conducted in may 2001 by the Canadian Wildlife Service over 424 lakes of the study area. This figure shows that both species are not sympatric in the study area.

Figure 4. Number of cavities found for each cavity type in Zec Chauvin and ZEC Martin-Valin, Québec, Canada. Filled bars correspond to excavated cavities and shaded bars to naturally formed cavities.This figure shows that majority of excavated cavities found were medium sized and no very large cavities were encountered. Natural cavities were as abundant as excavated cavities. For details on cavity types, see methods.

Figure 5. Proportion of dead trees (filled bars) and cavity trees (shaded bars) available in Zec Chauvin and ZEC Martin-Valin, Québec, Canada. No tree species provided significantly more cavities then its availability across the study area.

Figure 6. Mean densities of potential cavity trees with regards to stand type in unharvested stands, remnant stands, and cutblocks in Zec Chauvin and ZEC Martin-Valin, Québec, Canada. Filled bars correspond to natural stands and shaded bars to managed stands. This figure shows that potential cavity tree availability is significantly reduced in managed stands. Different letters indicate a significant difference between treatments (see Table 4).

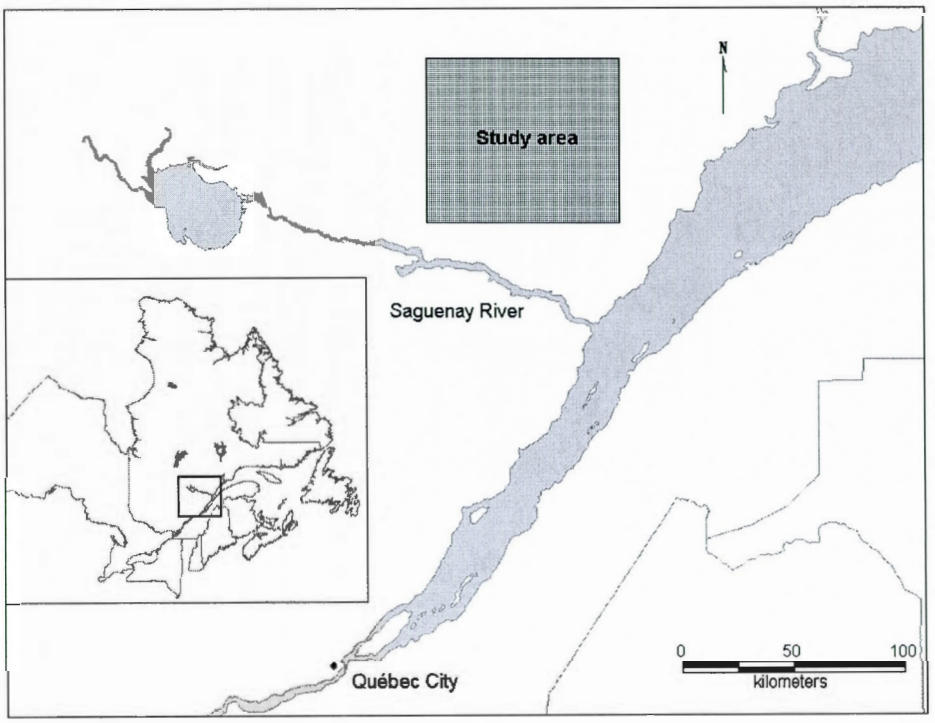
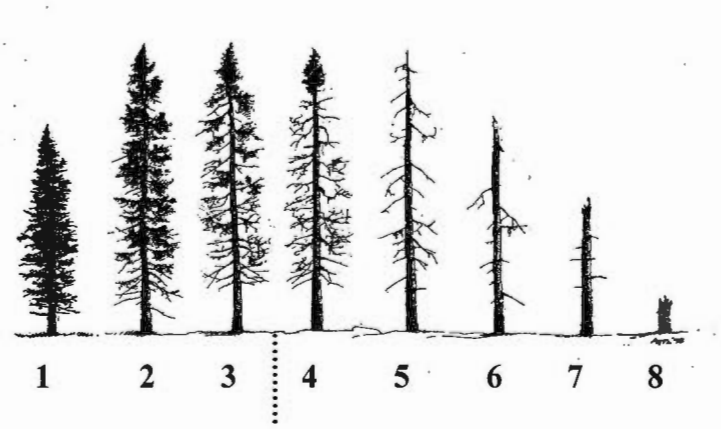


Figure 1. Vaillancourt et al.



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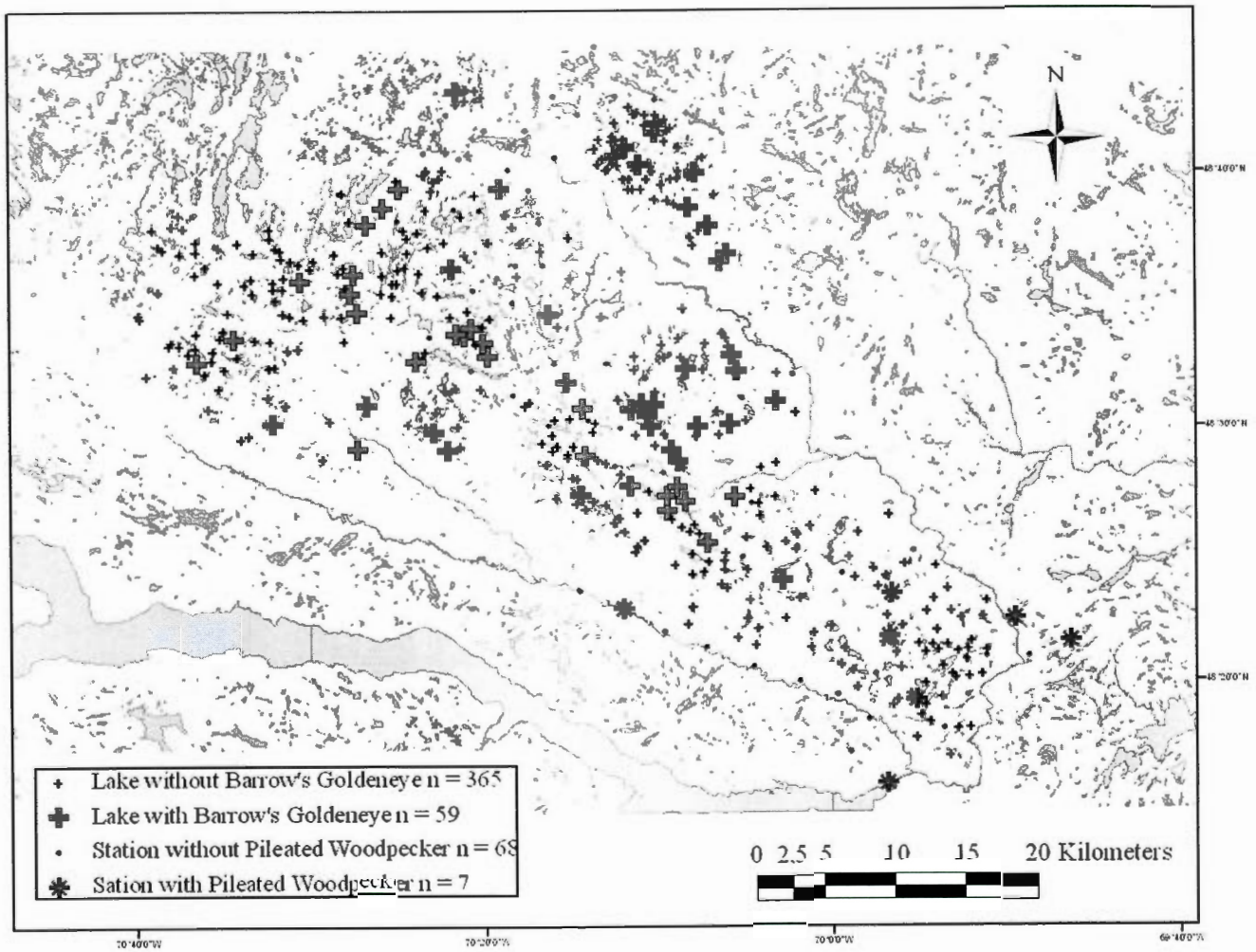

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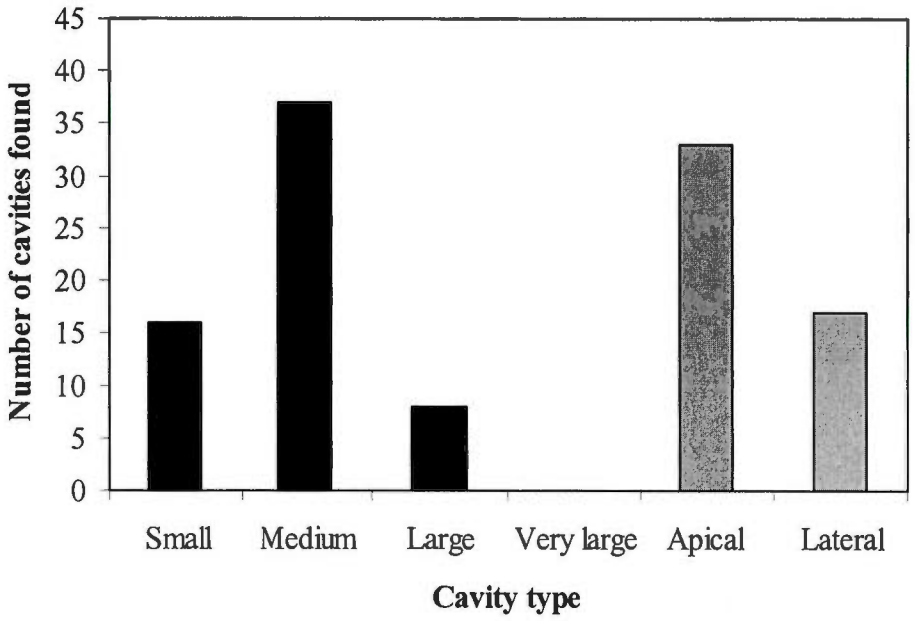

Figure 4. Vaillancourt et al.

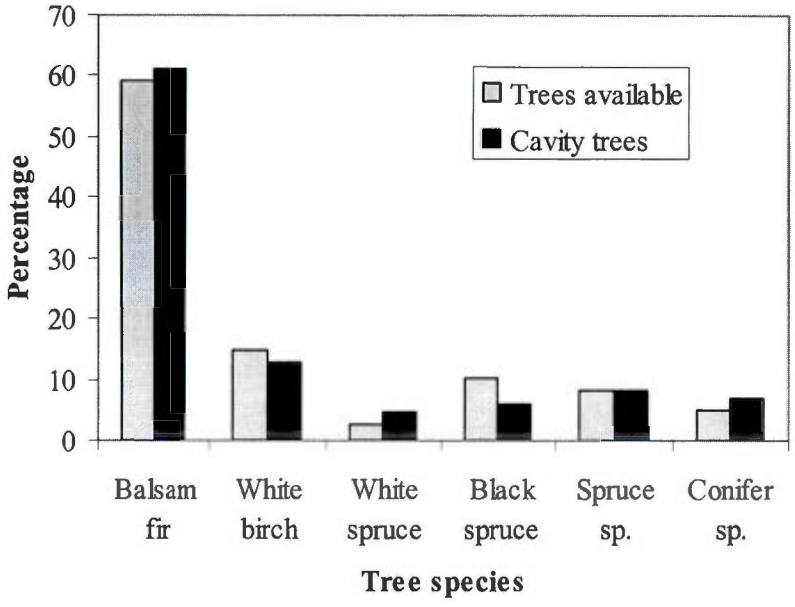

Figure 5. Vaillancourt et al.

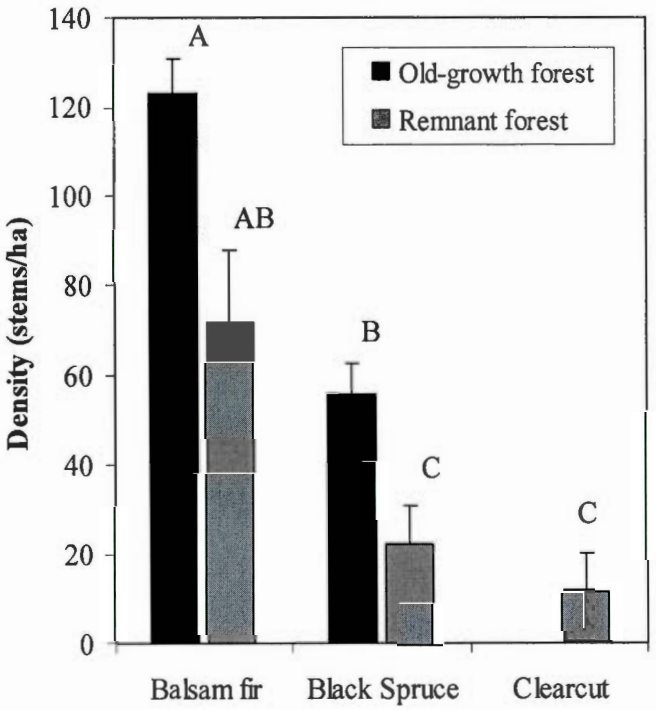

Figure 6. Vaillancourt et al.

Table 1. Stand and cavity tree characteristics in spruce, balsam fir and clearcut stands in ZEC Chauvin and ZEC Martin-Valin, Québec, Canada, 2004-2005.

Characteristics	*Spruce stands*	*Balsam fir stands*	*Clearcuts*	*P*
No. living trees sampled	2224	2532	124	
No. dead trees sampled	1188	1113	565	
No. of cavities found	21	59	30	
Proportion of cavity trees (%)	1.4	3.6	4.6	
Cavity tree density (no./ha)*				
Total	1.7 (0.5) a	6.5 (1.1) b	1.5 (0.3) a	<0.0001
Excavated by woodpecker	0.9 (0.4) a	3.3 (0.7) b	0.9 (0.4) b	0.003
Naturally formed	0.8 (0.2) a	3.2 (0.7) b	0.7 (0.2) c	0.006
Mean cavity tree DBH (cm)*				
Total	25.7 (1.7)	30.2 (1.1)	30.8 (1.5)	0.0558
Excavated by woodpecker	24.4 (2.3) a	28.3 (1.5) ab	33.7 (1.9) b	0.013
Naturally formed	26.6 (2.3)	32.0 (1.6)	27.9 (2.2)	0.0956
Mean cavity tree height (m)*				
Total	5.5 (0.7)	5.1 (0.5)	4.9 (0.7)	0.350
Excavated by woodpecker	6.9 (1.2)	7.1 (0.8)	6.3 (1.0)	0.310
Naturally formed	4.4 (0.6)	3.2 (0.4)	3.4 (0.7)	0.328
Distribution of canopy closure indice (%)†				0.005
Clear	35.3	23.7	78.6	
Partially closed	11.8	47.4	14.3	
Closed	52.9	28.9	7.1	
Distribution of decay-class (%)†				0.08
5	4.3	0	11.5	
6	21.7	38.1	38.5	
7	54.3	42.9	19.2	
8	19.6	19.0	30.8	
Distribution of cavity tree species (%)				
Excavated by woodpecker	n=9	n=22	n=13	
Balsam fir	66.7	61.5	23.1	
White birch	11.1	0	46.2	
White spruce	0	0	0	
Black spruce	22.2	15.4	7.7	
Spruce sp.	0	7.7	23.1	
Conifer sp.	0	15.4	0	
Naturally formed	n=12	n=24	n=13	
Balsam fir	61.5	54.2	69.2	
White birch	0	12.5	7.7	
White spruce	0	8.3	0	
Black spruce	15.4	0	0	
Spruce sp.	7.7	4.2	15.4	
Conifer sp.	15.4	20.8	7.7	

* Standard errors are in parentheses; non parametric Kruskal-Wallis test was conducted; pairwise comparisons were determined using Mann-Whitney tests.

† Chi-sqaure test were conducted for differences among distributions

Table 2. Stand type and physical characteristics of cavities considered suitable for the breeding of Barrow's Goldeneyes found in linear transects located in natural and managed forests in ZEC Chauvin and ZEC Martin-Valin, Québec, Canada, 2004-2005.

	Stand type	*Cavity type*	*Tree species*	*DBH (cm)*	*Decay class*	*Canopy closure*	*Cavity height (m)*	*Floor diameter (cm)*	*Cavity depth (cm)*
1	Balsam fir	Apical	White birch	31	7	2	1.8	23	43
2	Balsam fir	Apical	Balsam fir	51	5	1	2.0	36	99
3	Cutblock	Lateral	White birch	51	6	1	2.0	28 x 18	79
4	Cutblock	Apical	Balsam fir	22	7	1	1.7	18	51
5	Cutblock	Apical	Balsam fir	28	8	1	1.3	18	69
6	Cutblock	Apical	Conifer sp.	29	8	1	2.2	18	132
7	Black spruce	Apical	Spruce sp.	32	8	1	3.3	18	79
8	Cutblock	Apical	Balsam fir	30	8	1	1.3	15	53
9	Cutblock	Apical	Balsam fir	24	8	2	1.8	15	66
10	Black spruce	Apical	Balsam fir	29	7	2	2.1	15	53
11	Balsam fir	Apical	Balsam fir	21	8	3	1.5	17	56
12	Black spruce	Apical	Balsam fir	36	8	3	1.8	25	81
13	Balsam fir	Apical	Balsam fir	31	8	3	1.0	n/a	n/a
14	Cutblock	Apical	Spruce sp.	26	8	3	2.0	17	81
15	Cutblock	Apical	Balsam fir	34	8	3	3.5	20	119
16	Balsam fir	Apical	Balsam fir	36	7	3	1.5	24	41
17	Cutblock	Apical	Balsam fir	20	8	3	1.8	15	76
18	Black spruce	Apical	Conifer sp.	35	8	3	3.4	84	14

Note: Entrance dimensions are not included in the table since only one lateral cavity was considered suitable and its entrance dimensions met the criteria fixed.

Table 3. DBH threshold values (x_t) for the occurrence of natural cavity in different tree species. β_1 and β_0 corresponds to the parameter estimate and intercept of the logistic regression between DBH and occurrence of cavity. AUC is the area under the ROC curve and c is the cut-off value obtained from the ROC curves (see method).

Tree species	β_1	β_0	*AUC*	*c*	x_t
Balsam fir	0.107	-7.34	0.63	0.0122	27.6
White birch	0.111	-8.36	0.69	0.0069	30.5
Spruce sp.	0.077	-7.96	0.71	0.0032	28.8

Table 4. Summary of one-way ANOVA results for the effect of forest type on potential cavity trees (see Fig. 6).

Source of variation	*DF*	*MS*	*F ratio*	*Prob > F*
Forest type	4	251.075	36.9371	< 0.0001
Error	111	754.5071	6.797	

CONCLUSION GÉNÉRALE

2.1 Disponibilité de gros arbres en forêt boréale conifériennne

Bien qu'il existe une panoplie d'études documentant l'abondance d'arbres morts au sein de différents types de couvert forestier en zone boréale (Lee et al. 1997; Clark et al. 1998; Greif et Archibold 2000; Hély et al. 2000; Hennon et McClelland 2003), peu de travaux ont établi la disponibilité et la qualité des arbres morts sur pied dans les forêts naturelles de l'Est du Québec qui dépassent l'âge d'exploitabilité commerciale (voir Desponts et al. 2002; Aakala et al. 2007). Afin d'être à même de mesurer l'impact de l'aménagement des forêts sur la diversité biologique et plus particulièrement sur la faune associée aux arbres morts, il était opportun d'évaluer le potentiel des différents peuplements à offrir des arbres à valeur faunique à l'intérieur de forêts non perturbées par les opérations forestières.

La proportion d'arbres morts obtenue en forêt boréale de l'Est du Québec s'est avérée considérablement élevée en comparaison à ce qui a été rapporté dans plusieurs études effectuées en forêt tempérée nord-américaine et en forêt boréale scandinave (revues par Nilsson et al. 2002). La répartition de ces arbres morts sur pied n'est toutefois pas uniforme dans la mosaïque de peuplements âgés constituant la forêt boréale de l'Est. Nos résultats ont montré que les forêts dominées par le sapin baumier offrent une plus grande densité de tiges vivantes et mortes comparativement aux peuplements d'épinettes noires, et ce, de façon beaucoup plus marquée lorsque l'on considère les arbres qualifiés de gros diamètre (DHP $\geq$ 30 cm). Conséquemment, les peuplements de sapins baumiers offrent une densité accrue de tiges mortes pour chaque stade de dégradation. Ces résultats suggèrent que les sapinières pourraient jouer un rôle clé au sein de la forêt boréale conifériennne en assurant un bon recrutement de grosses tiges et en offrant des conditions d'habitats diversifiées pouvant vraisemblablement combler les différentes utilisations de la grande faune cavicole.

2.2 Qualité des forêts résiduelles

Au Québec, la forêt boréale est presque entièrement vouée à la production commerciale de matière ligneuse (Ministère des Ressources naturelles et de la Faune 2006). Les forêts linéaires laissées sur pied dans les secteurs aménagés représentent une proportion importante du couvert forestier susceptible d'abriter la faune associée aux forêts matures et constituent le principal bassin de rétention et de recrutement d'arbres morts durant les années qui suivent la coupe. Bien que, à l'instar des travaux de Mascarúa López et al. (2006) et Gagné (2006), nos résultats aient montré que ces forêts linéaires contiennent des quantités d'arbres morts (DHP $\geq$ 20 cm) comparables à celles retrouvées dans les peuplements naturels équivalents, la densité de tiges de plus gros diamètre (DHP $\geq$ 30 cm) sont fortement réduite dans les forêts linéaires de notre aire d'étude. De plus, les forêts résiduelles diffèrent de la forêt naturelle environnante à l'égard de leur composition en espèces. Alors que la matrice forestière est constituée d'une importante proportion de peuplements dominés par le sapin baumier, les forêts résiduelles sont davantage dominées par l'épinette noire, et donc constituée de peuplements qui n'offrent pas une grande disponibilité en tiges de gros diamètre ni un potentiel à en générer dans le futur. Ces résultats suggèrent que les espèces utilisatrices de gros arbres morts risquent d'être affectées non seulement par la perte nette d'habitat engendrée par la récolte de forêts matures, mais également par la dégradation des forêts résiduelles des secteurs aménagés.

2.3 Cavités disponibles pour le Garrot d'Islande

Les informations concernant les sites naturels de nidification utilisés par le Garrot d'Islande proviennent quasi exclusivement d'une étude effectuée pour la population de l'Ouest de l'Amérique du Nord, où l'espèce niche principalement dans de grandes cavités excavées par le Grand Pic en forêt boréale mixte (Evans et al. 2002; Evans 2003). Les résultats présentés dans le second chapitre de ce mémoire sont donc les premiers à dresser un portrait des sites de nidification potentiels pour la population de l'Est du Garrot d'Islande. La situation qui prévaut en forêt boréale coniférienne dans l'Est du Canada diffère considérablement de celle décrite dans l'Ouest puisque nos résultats montrent que les excavateurs primaires ne

contribuent pas à la disponibilité de grandes cavités dans cet écosystème. Par conséquent, la population de l'Est dépend vraisemblablement de la disponibilité de grandes cavités naturelles pour sa nidification.

L'utilisation de cavités naturelles par les canards cavicoles est un phénomène bien documenté dans les forêts riveraines tempérées (Prince et al. 1968; Gilmer et al. 1978; McComb et Noble 1981; Soulliere 1988; Lowney et al. 1989; Yetter et al. 1999; Roy Nielsen 2007) alors qu'en forêt boréale, il existe très peu de publications sur le sujet. C'est pourquoi notre démarche consistait principalement à documenter les cavités naturelles en forêt boréale mature ainsi qu'à caractériser celles qui sont théoriquement disponibles à la nidification du Garrot d'Islande. Nos résultats révèlent que les cavités naturelles dans les forêts coniférienes sont rares et, par conséquent, que les cavités adéquates pour le Garrot d'Islande le sont d'autant plus. Il s'est avéré que la majorité des cavités naturelles étaient supportées par des arbres très décomposés. Bien que les facteurs responsables de la formation des cavités naturelles soient complexes et fortement aléatoires (donc peu prévisibles) (Lindenmayer et al. 1997), il demeure évident que la formation de ces cavités est un processus qui prend du temps. De plus, vu leur dégradation avancée, le temps de résidence des arbres porteurs doit probablement être court, ce qui souligne l'importance d'assurer un recrutement constant en arbres morts de gros diamètre, lesquels sont plus susceptibles de développer des cavités.

La densité d'arbres susceptibles de porter des cavités naturelles est la plus élevée dans les peuplements de sapins baumiers et se trouve passablement réduite dans les forêts linéaires et les parterres de coupe. Les pratiques sylvicoles actuelles, qui sont basées sur la maximisation de la récolte de matière ligneuse au sein d'un territoire, laissent très peu d'arbres sur pied dans les secteurs aménagés et favorisent la rétention de peuplements résiduels n'ayant pas tous les attributs des forêts d'intérieur. Ce type d'aménagement forestier appliqué à grande échelle en Scandinavie a contribué à mettre en danger plusieurs organismes associés aux arbres morts (Berg et al. 1994), dont plusieurs espèces de pics (Angelstam et Mikusiński 1994), et de vastes réseaux de nichoirs contribuent aujourd'hui à maintenir les populations de canards cavicoles qui, autrement, pourraient devenir menacées. Même si, jusqu'à maintenant, il existe peu d'évidences au Canada quant au rôle de la foresterie à engendrer un déclin chez

certaines espèces (voir Thompson et al. 2003b), les changements créés par l'aménagement forestier quant à la structure des forêts résiduelles et la disponibilité des arbres morts pourraient sans doute exacerber les menaces qui pèsent sur le Garrot d'Islande ainsi que sur d'autres espèces associées aux arbres morts.

Dans les régions les plus affectées par la coupe forestière et où les substrats de nidification sont susceptibles d'être limitants pour le Garrot d'Islande, l'installation de réseaux de nichoirs, à l'instar des pays européens, est certes une solution d'appoint pour prévenir une baisse de population locale. Toutefois, la vastitude de l'aire de nidification de la population de l'Est de l'Amérique du Nord et le faible taux d'occupation de ce territoire rend virtuellement impossible le suivi d'un tel réseau sur la totalité de l'aire fréquentée à long terme. Des solutions provenant des méthodes d'aménagement forestier sont davantage à préconiser puisqu'elles s'appliquent sur l'ensemble des forêts publiques et risquent également de favoriser les nombreuses autres espèces associées aux arbres morts en forêt boréale.

2.4 Méthodes alternatives d'aménagement forestier

Afin de pallier aux effets négatifs que pourraient entraîner les régimes forestiers traditionnels à courtes rotations, Lindenmayer et al. (2006) ont récemment suggéré plusieurs principes de base à appliquer pour favoriser la conservation de la diversité biologique à l'intérieur des paysages aménagés, tels que la rétention de composantes clés dans la complexité structurale des forêts (e.g. arbres morts et vivants de gros diamètre) et l'augmentation des cycles de rotation de récolte afin de permettre le recrutement de ces structures. Dans les secteurs fréquentés par le Garrot d'Islande, ces stratégies alternatives de pratiques forestières doivent être appliquées afin d'assurer la disponibilité et le recrutement de substrats de nidification. En plus de favoriser la rétention d'arbres morts et vivants dans les aires de coupes agglomérées (rétention par bouquet), les peuplements laissés sur pied (bandes riveraines et séparateur secs) devraient être élargis afin de maintenir les attributs de forêts d'intérieur (voir Mascarúa López et al. 2006) et les peuplements matures dominés par le sapin baumier devraient être favorisés lors de la décision prise pour l'emplacement des séparateurs secs.

L'accélération de la perte de biodiversité a favorisé l'émergence de l'aménagement écosystémique des forêts, une nouvelle conception de l'aménagement forestier ayant pour objectif de préserver l'intégrité écologique des écosystèmes (Grumbine 1994; Galindo-Leal et Bunnell 1995; Christensen et al. 1996). Une des assises de ce récent paradigme est de recréer les conditions générées par les perturbations naturelles d'un système par une varité de méthodes de récolte de matière ligneuse, de manière à préserver les processus écologiques et, par le fait même, la biodiversité adaptée aux régimes de perturbation qui prévalent (Franklin 1993; Gauthier et al. 1996). Au Québec, plusieurs auteurs ont suggéré différentes stratégies d'aménagement forestier basé sur les perturbation naturelles ayant cours en forêt boréale (Bergeron et Harvey 1997; Bergeron et al. 1999; Harvey et al. 2002). Puisque la forêt boréale de l'Est du Québec est surtout modifiée par des perturbations dont la sévérité est intermédiaire (épidémies d'insectes, chablis) qui permettent le maintien d'une importante proportion de vieilles forêts à structure inéquienne (De Grandpré et al. 2000; Boucher et al. 2003), on doit envisager une diminution de la proportion du territoire aménagé en coupes agglomérées (aménagement équienne) pour favoriser des pratiques sylvicoles inéquiennes (i.e. coupes partielles, coupes sélectives) (Bergeron et al. 1999; Harvey et al. 2002). L'adoption de telles pratiques ajustées à des enjeux spécifiques de maintien de la diversité biologique (rétention de tiges de gros diamètre) pourrait permettre de maintenir une plus grande quantité de bois sur pied, notamment des arbres de gros diamètre qui offriront un recrutement continu en gros bois mort dégradé.

2.5 Travaux futurs

Nous avons mis en évidence l'importance des vieilles sapinières dans la disponibilité locale d'arbres de gros diamètre. Bien que la région couverte par cette étude comprenne une importante proportion de sapinières, l'aire de nidification du Garrot d'Islande chevauche également la région bioclimatique de la pessière à mousse de l'Est (Robert et al. 2000), où les peuplements dominés par le sapin sont moins prépondérants. De futurs travaux sur la distribution de ces peuplements à l'échelle de l'aire de nidification permettraient de vérifier si l'espèce se répartit en fonction des peuplements à haut potentiel identifiés lors de cette étude.

En outre, ceci faciliterait la planification de zones de conservation de l'habitat de nidification du Garrot d'Islande dans les portions non touchées par l'aménagement forestier à ce jour.

Étant donné le peu de connaissances concernant la nidification du Garrot d'Islande dans l'Est de l'Amérique du Nord, la présente étude aura permis de jeter les bases sur la nature des sites de nidification disponibles. Les arbres porteurs de cavités adéquates pour le Garrot d'Islande se sont avérés être des arbres très dégradés, donc morts depuis plusieurs années. Davantage de connaissances doivent être acquises sur les taux de mortalité des arbres de gros diamètre pour les essences d'arbres en forêt boréale ainsi que sur leur taux de transition à travers les différentes classes de dégradation afin de connaître la dynamique temporelle de recrutement des arbres très dégradés et ainsi prévoir avec une plus grande précision le potentiel de ces forêts à offrir des arbres à cavités. Enfin, de futurs travaux portant sur la sélection de sites de nidification et sur la productivité du Garrot d'Islande en forêt boréale de l'Est du Canada selon les différents contextes (e.g. forêts naturelles vs forêts aménagées) permettront un raffinement dans la détermination des cavités adéquates et donc une évaluation plus précise de la disponibilité de sites de nidification et des impacts de l'aménagement forestier sur cette espèce au statut précaire.

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