SNAG LONGEVITY AND AVAILABILITY FOR CAVITY-NESTERS IN POST-HARVEST LANDSCAPES IN WESTERN NEWFOUNDLAND

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Abstract

Standing dead trees (hereafter snags) are a valuable component of forest systems, comprising habitat for wildlife, nursery sites for forest regeneration, and as stores of biomass, nutrients, and carbon. I examined populations of snags present following harvest in a chronosequence of balsam fir (*Abies balsamea*) dominated boreal forests in western Newfoundland to assess: (1) snag density, longevity and biomass in a post-harvest landscape, and (2) the availability and quality of snags for cavity-nesting birds and other wildlife. Average snag longevity was indicated by a sharp decline in snag numbers 10 to 15 years after harvest, followed by low snag densities until the regenerating forest reached senescence. A smaller but more immediate snag density decline was observed 1 to 4 years after harvest when residual snags and live trees were prone to windfall and domestic harvesting. Snag longevity was positively correlated with stem diameter at breast height (dbh). A large portion (55%) of 1260 snags encountered were balsam fir which have a small average dbh. These small-diameter snags were short-lived, contributing low biomass and a long period of low snag abundance. White birch (*Betula papyrifera*) produced the largest diameter and longest standing snags.

Cavity nesters used snag species based on availability with 41% of the 81 cavities identified being in balsam fir. Cavity presence was most strongly positively correlated to snag diameter at breast height (dbh), followed by decay class, time since harvest and height. Less than 40% of snags available throughout the chronosequence had large enough dbh for cavity nesters. Downy Woodpeckers (*Picoides pubescens*) were responsible for 47% of all cavities identified, excavating balsam fir ~50% of the time. Two larger cavity nesters present, Northern Flickers (*Colaptes auratus)* and Three-toed Woodpeckers (*P. tridactylus*), were preferentially using large-diameter white birch snags. Northern Flickers excavated the largest cavities, potentially providing habitat for the greatest variety of secondary cavity-nesters. Managing for snags with >30 cm dbh, which flickers target for excavation, has the greatest potential to enhance the broader snag cavity-based community in western Newfoundland.

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Co-authorship Statement

I conducted this research independently but under the co-supervision of Drs. Ian Warkentin and Martin Moroni. I was responsible for substantial components of project development and design in collaboration with my committee members and representatives of Corner Brook Pulp & Paper (Ltd.). I completed the field studies associated with this project with the assistance of Ms. Jennifer Beaton, who worked under my supervision.

I collated, entered, and analyzed all data based on consultation with Drs. Ian Warkentin, Martin Moroni and Edward Reid. In addition, I interpreted all of the results and wrote the manuscripts that constitute the chapters of this thesis. I have revised the manuscripts based on the advice and comments of my supervisor, supervisory committee, and journal reviewers. The second chapter, Smith C.Y., Moroni, M.T. and I.G. Warkentin, Snag dynamics in post-harvest landscapes of western Newfoundland balsam fir-dominated boreal forest, has been submitted to Forest Ecology & Management. The third chapter, Smith, C.Y., Warkentin, I.G. and M.T. Moroni, Snag availability for cavity nesters across a chronosequence of post-harvest landscapes in western Newfoundland, has been published in Forest Ecology & Management 256(4):641-647.

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1.0 Introduction

Standing dead trees (hereafter snags) are a valuable component of forest systems, comprising habitat for wildlife, nursery sites for the regenerating forest, and as stores of biomass, nutrients, and carbon (C) (McClelland and Frissel 1975, Raphael and White 1984, Machmer and Seeger 1995, Yatskov et al. 2003, Storaunet 2004). When these snags eventually fall, they fulfill other roles, becoming a substantial source of woody debris in some forest systems, with associated habitat and nutrient cycling benefits (Storaunet 2004; Everett et al. 1999). Snags are particularly important as habitat for cavity-nesting species such as woodpeckers, providing opportunities for nesting, foraging, perching, and drumming (Cline et al. 1980, Cimon 1983, Morrison and Raphael 1993, Martin et al. 2004). Additionally, the cavities excavated in snags by these primary cavity-nesters subsequently support a wide variety of secondary cavity-nesters (Bull et al. 1980, Zack et al. 2002, Martin et al. 2004, Remm et al. 2006). With approximately 85 species of both primary and secondary cavity-nesting birds in North America (Spiering and Knight 2005) the availability of suitable habitat for these species is ecologically important.

Snags result from tree mortality and become a dominant feature following anthropogenic (harvesting) and natural (insect and fire) disturbances. Forest managers once considered snags to be of little value, representing both a safety and a fire hazard (Thomas 2002). Emerging knowledge of the role played by snags in forest communities has led managers to be more conscious of snag retention and longevity (McComb and Lindenmayer 1999). Increasing interest in

C cycling has emphasized the understanding of how much C can be found in the snag pool and its residency time. In order to judiciously manage snags for longevity and habitat availability, forest managers require estimates of snag longevity on post-harvest landscapes specific to their region. Managers also need to quantify the availability of snags with characteristics most useful to wildlife for habitat requirements such as nesting and foraging (Lehmkuhl et al. 2003, Spiering and Knight 2005). Cavity-nesters tend to use snags with a larger diameter at breast height (dbh) and bias snag use based on snag species, state of decay, percent bark coverage, and the presence of broken tops (Mannan et al. 1980, Raphael and White 1984).

Timber harvesting has been a century-long industry on the west coast of Newfoundland, Canada (Graphic 1). Harvesting has become a widespread disturbance type in this region. The dominant tree species is balsam fir (*Abies balsamea*) and forests are managed under 80-120 year rotation periods (Thompson et al. 2003). The area is characterized by wet boreal forests with a low level of fire activity and the dominant natural disturbance being insect defoliation. Two major defoliators drive western Newfoundland balsam fir forests dynamics, hemlock looper (Lambdina fiscelleria fiscelleria [Guen.]) and spruce budworm (Choristoreura fumiferana [Clem]). These insects have acted simultaneously (Hudak and Raske 1981, Hudak 1996) in western Newfoundland since the beginning of the 20th century (Hudak 1996). Recent insect outbreaks have defoliated an average of 28 000 ha of Newfoundland forests annually (Average values from 'The State of Canada's Forests' publications 1995-2002), that typically result in reductions in growth and high levels of mortality (Karsch 1994, Bergeron 1995). For example, spruce budworm outbreaks between 1971 and 1979 killed 23% of the islands merchantable tree volume (Hudak and Raske 1981), and hemlock looper defoliated 73.6 million ha, killing 330 000 ha between 1983 and 1995 (Hudak 1996). Balsam fir forests in eastern Canada readily re-establish after insect mortality due to advanced regeneration (Morin 1994, Kneeshaw 1999).

Land managers are now trying to preserve appropriate representation of habitat types and biodiversity following harvest, incorporating the preservation of snags as one component in this approach. A number of studies have examined snag dynamics (decomposition rates and longevity [the length of time a tree remains standing after it has died]) in post-fire landscapes (e.g., Everett et al. 1999 and Russell et al. 2006) but fewer studies have examined snag dynamics in post-harvest landscapes (e.g., Garber et al. 2005, Moroni 2006). Snag densities post-harvest differ from snag densities after natural disturbance where higher densities of snags are left immediately following the disturbance (e.g. following fire Schieck and Song 2006). Hence mimicking natural disturbance regimes requires alternative forest management strategies than previously practiced to maximize snag availability over time. Cline et al. (1980) suggested leaving 10 snags/ha following harvest, a recommendation that has been incorporated into forest management policies for Newfoundland (Corner Brook Pulp and Paper Ltd. 2004). However, it is not clear that 10 snags/ha present initially after harvest provides sufficient snag densities to ensure the continued presence of snags in the initial post-harvest period and for extended periods into the forest rotation. Nor does the current requirement consider snag recruitment after the initial post-harvest period. The requirements also make no stipulations as to the characteristics of the snags retained following harvest, and thus the suitability of those snags in terms of their potential longevity or usefulness in accommodating wildlife.

Working in balsam fir-dominated boreal forests of western Newfoundland, Canada, the aims of this project were to assess snag density, longevity and biomass in a post-harvest landscape (Chapter 2) and also to determine the availability and quality of snags for cavity-nesting birds and other wildlife (Chapter 3). I conducted these studies using a chronosequence of balsam fir (*Abies balsamea*)-dominated boreal forests that encompassed one full rotation of the forest following harvest.

Both chapters were written in the format of the peer-reviewed journal Forest Ecology & Management. The first chapter is currently under review and the second is now published.

1.1 References

Bergeron, Y. Luduc, A., Morin, H., Jayal, C. 1995. Balsam fir mortality following the last spruce budworm outbreak in northeastern Quebec. Canadian Journal of Forest Research 25, 1375-1384.

Bull, E.L., Twombly, A.D., Quigley, T.M. 1980. Perpetuating snags in managed mixed conifer forests of the Blue Mountains, Oregon. P. 325-336. *In* Management of western forests and grassland for nongame birds. Workshop proceedings. USDA Forest Service General Technical Report INT-86. Intermountain Forest and Range Experiment Station, Ogden, Utah.

Cimon, N. 1983. A Simple model to predict snag levels in managed forests. *In* Snag Habitat Management: Proceedings of the Symposium. USDA Forest Service General Technical Report RM-99.

Cline, S.P., Berg, A.B., Wight, H.M. 1980. Snag characteristics and dynamics in Douglas-fir forests, western Oregon. Journal of Wildlife Management, 44, 773–786.

Corner Brook Pulp and Paper Ltd. 2004. Sustainable Forest Management Plan. Version 2. 151 p.

Dixon, R.K., Brown, S., Houghton, R.A., Solomon, A.M., Trexler, M.C., Wisniewski, J. 1994. Carbon pools and flux of global forest ecosystems. *Science*, 263, 185–190.

Everett, R., Lehmkuhl, J., Schellhaas, R., Ohlson, P., Keenum, D., Riesterer, H., Spurbeck, D. 1999. Snag dynamics in a chronosequence of 26 wildfires on the east slope of the Cascade Range in Washington State, USA. International Journal of Wildland Fire, 9, 223-234.

Garber, S.M., Brown, J.P., Wilson, D.S., Maguire, D.A., Heath, L.S. 2005. Snag longevity under alternative silvicultural regimes in mixed-species forests of central Maine. Canadian Journal of Forest Research, 35, 787–796.

Hudak, J. 1996. The eastern hemlock looper, *Lambdina fiscellaria (Guen.)*(Lepidoptera: Geometridae) in Newfoundland, 1983-1995. Canadian Forest Service Information Report No. N-X-302.

Hudak, J., Raske, A.G. 1981. Review of the spruce budworm outbreaks in Newfoundland – its control and forest management implications. Canadian Forest Service Information Report No. N-X-205.

Karsch, M.B., Lavigne, M.G, Donnelly, J.G. 1994. Growth responses of the balsam fir and black spruce spacing trials. Canadain Forest Service Information Report N-X-291.

Kneeshaw, D., Bergeron, Y. 1999. Spatial and temporal patterns of seedling and sapling recruitment within canopy gaps caused by spruce budworm. Ecoscience 6, 214-222.

Lehmkuhl, J.F., Everett, R.L., Schellhaas, R., Ohlson, P., Keenum, D., Riesterer, H., Spurbeck, D. 2003. Cavities in snags along a wildfire chronosequence in eastern Washington. Journal of Wildlife Management, 67, 219-228.

Machmer, M.M., Steeger, C. 1995. The ecological roles of wildlife tree users in forest ecosystems. B.C. Ministry of Forests. Victoria, British Columbia.

Mannan, R.W., Meslow, E.C., Wight, H.M. 1980. Use of snags by birds in Douglas-fir Pseudotsuga-menziesii forests western Oregon, USA. Journal of Wildlife Management, 44, 787-797.

Martin, K., Aitken, K.E.H., Wiebe, K.L. 2004. Nest sites and nest webs for cavity-nesting communities in the interior British Columbia Canada: Nest characteristics and niche portioning. The Condor, 106, 5-19.

McClelland, B.R., Frissell, S.S. 1975. Identifying forest snags useful for hole-nesting birds. Journal of Forestry, 73, 414-417.

McComb, W., Lindenmayer, D. 1999. Dying, dead and down trees. In: Hunter, M.L. Jr. (ed.). Maintaining biodiversity in forest ecosystems. Cambridge University Press.

Morin, H. 1994. Dynamics of balsam fir forests in relation to spruce budworm outbreaks in the boreal zone of Quebec. Canadian Journal of Forest Research 24, 730-741.

Moroni, M.T. 2006. Disturbance history affects woody debris dynamics of Newfoundland balsam fir (*Abies balsamea*) stands. Canadian Journal of Forest Research, 36, 3194–3208.

Morrison, M.L., Raphael, M.G. 1993. Modeling the dynamics of snags. Ecological Applications, 3, 322-330.

Raphael, M.G., White, M. 1984. Use of snags by cavity-nesting birds in the Sierra Nevada. Wildlife Monographs 86.

Remm, J., Lohmus, A., Remm, K. 2006. Tree cavities in riverine forests: What determines their occurrence and use by hole-nesting passerines? Forest Ecology and Management, 221, 267-277.

Russell, R.E., Saab, V.A., Dudley, J.G., Rotella, J.J. 2006. Snag longevity in relation to wildfire and postfire salvage logging. Forest Ecology and Management, 232, 179-187.

Schieck, J., Song, S.J. 2006. Changes in bird communities throughout succession following fire and harvest in boreal forests of western North America: literature review and meta-analyses. Canadian Journal of Forest Research, 36, 1299-1318.

Spiering, D.J., Knight, R.L. 2005. Snag density and use by cavity-nesting birds in managed stands of the Black Hills National Forest. Forest Ecology and Management, 214, 40-52.

Storaunet, K.O. 2004. Models to predict time since death of *Picea abies* snags. Scandanavian Journal of Forest Research, 19, 250-260.

Thomas, J.W. 2002. Dead wood: from forester's bane to environmental boon. *In* Proceedings of the Symposium on Ecology and Management of Deadwood in Western Forests, 2-4 November 1999, Reno, Nev. *Technical coordinators:* W.F. Laudenslayer, Jr., P.J. Shea, B.E. Valentine, C.P. Weatherspoon, and T.E. Lisle. U.S. For. Serv. Gen. Tech. Rep. PSW-GTR-181. pp. 3-9.

Thompson, I.D., Larson, D.J., Montevecchi, W.A. 2003. Characterization of old “wet boreal” forests, with an example from balsam fir forests of western Newfoundland. Environmental Reviews, 11, S23-S46.

Yatskov, M., Harmon, M.E., Krankina, O.N. 2003. A chronosequence of wood decomposition in the boreal forests of Russia. *Canadian Journal of Forest Research*, 33, 1211–1226.

Zach, S., George, L., Laundenslayer, W.F. Jr. 2002. Are there snags in the system? Comparing cavity use among nesting birds in “Snag-rich” and “Snag-poor” eastside pine forests. USDA Forest Service General Technical Report PSW-GTR-181. 279-291.

2.0 Snag dynamics in post-harvest landscapes of western Newfoundland balsam fir-dominated boreal forests

2.1 Abstract

We examined the longevity of standing dead trees (snags) and changes in snag volume and biomass with time following harvesting in a chronosequence of balsam fir (*Abies balsamea*) dominated boreal forests in western Newfoundland. The aim of our study was to determine whether the current snag management practices in western Newfoundland would be able to maintain the recommended minimum of 10 snags/ha on the post-harvest landscape. Snags displayed two distinct periods of decline following harvest: the first, 1–4 years after harvest, when residual snags and live trees are prone to windfall and domestic harvesting; the second, 10–15 years following harvest, was followed by low snag densities, indicating the average lifespan of snags. Snag densities then increased with forest age, reaching the highest level in 81- to 100-year-old forests that had become senescent. Snag longevity was positively correlated with stem diameter. White birch (*Betula papyrifera*) produced the largest diameter and longest standing snags. Small-diameter snags of western Newfoundland boreal forests are short lived, contribute low biomass and exhibit a long period of low snag abundance.

2.2 Introduction

Standing dead trees (hereafter snags) are increasingly recognized as a crucial ecological component of post-harvest, managed and unmanaged forest landscapes. Snags are important for biodiversity and the natural cycling of nutrients in forest ecosystems (Garber et al., 2005; Marage and Lempérière, 2005). The benefits of snag retention in managed forests for nutrient cycling, wildlife habitat and carbon (C) storage have been well documented (e.g., Cline et al., 1980; Peng et al., 2002; Yatskov et al., 2003; Howard et al., 2004; Yamasaki and Leak 2006) and guidelines have been formulated by resource managers aimed at maintaining snags on post-harvest landscapes. Cline et al. (1980) suggested leaving 10 snags/ha, a recommendation adopted as the legal requirement for harvesters in the Canadian province of Newfoundland and Labrador (NL) (Corner Brook Pulp and Paper Ltd., 2004). To manage snags in post-harvest landscapes, snag dynamics (e.g., decomposition rates and longevity—the length of time a tree remains standing after it has died) must be understood (Garber et al., 2005). To date, studies of dead wood in Newfoundland boreal forests have focused either on woody debris (e.g., Sturtevant et al., 1997) or on snags present at only three age classes following natural and anthropogenic disturbance (e.g., Moroni 2006). Snag dynamics over an entire forest rotation have not been documented in Newfoundland and are poorly documented elsewhere.

Snag dynamics and snag biomass are highly influenced by anthropogenic disturbances (harvesting) and land management regimes (Krankina and Harman, 1994). Snags represent 5–10% of trees in most mature forest stands (Hunter, 1990), but as many as 90% of the trees found on recently harvested landscapes. Overall, the effects of clearcut harvesting are few snags left on the landscape and few live trees left for snag recruitment. Snags have always been inconvenient for harvesters, obstructing access to merchantable trees and constituting health hazards (Thomas, 2002). However, regardless of the method of harvesting, harvesters are unlikely to allocate time or resources to cut snags or unmerchantable trees unnecessarily. Thus, in the absence of legal requirement, unless a snag prevents access to merchantable trees or forms a hazard to harvesters, we would expect snags to typically be left standing. Here, we assumed that the practice of leaving snags in place has been followed since harvesting commenced in Newfoundland, and based on existing literature, typical snag fall rates would suggest the effects of past harvesting methods would no longer be measurable since current use of short wood mechanical harvesters have been adopted for ~20 years, longer than most snags in Newfoundland are expected to stand. Therefore, even without the recent emphasis on snag retention, snag removal from past harvesting methods is likely to have been minor and not a limiting factor, allowing us to conduct our study using a chronosequence sampling technique.

The forest of western Newfoundland is described as “wet boreal” and is dominated by balsam fir (*Abies balsamea*), which is a short-lived tree species with a relatively small diameter at breast height (dbh; 1.3 m) (Thompson et al., 2003). Snag decomposition and longevity are influenced by climate and species-specific patterns of decay rates and morphology (Conner and Saenz, 2005; Garber et al., 2005). However, snag longevity is also influenced by a host of other factors, including dbh, top presence and exposure to wind (Bull, 1983; Morrison and Raphael, 1993; Garber et al., 2005). Among these, dbh is typically a strong positive correlate of snag longevity (Cline et al., 1980). Bull's (1983) assessment of snag longevity in an experimental mixed conifer forest indicated that snag fall rates were low until approximately 6–8 years following tree death, after which rates increased. Everett et al. (1999) found that most snag fall occurred in the first 15 years following disturbance on a post-fire lodgepole pine (*Pinus contorta*) and Douglas-fir (*Pseudotsuga menziesii*) landscape. Longevity of snags in post-harvest landscapes should be very different from that of snags in mature forests because the surrounding live trees that protect snags from wind and sun exposure are removed. Garber et al. (2005) suggested that snag fall rates should be relatively low immediately following harvest, but then increase until about 15 years after disturbance.

For this study, we assessed snag density, longevity, and biomass in a post-harvest chronosequence of balsam fir-dominated boreal forests in western Newfoundland that encompassed one full rotation of the forest. We predicted that current snag management strategies for harvesting the small-dbh forests of western Newfoundland would not be able to maintain the recommended minimum of 10 snags/ha on the post-harvest landscape. In particular, we predicted that snag density would be >10 snags/ha following harvest, but would decrease rapidly and fall below the required density within 20 years, and would remain low until snag creation through senescence occurred late in the natural forest succession. We also predicted that large snags would be most persistent and important to snag density 20 years after harvest.

2.3 Methods

2.3.1 Study area

Our study was conducted in the Western Newfoundland Ecoregion (Forest Section B28b) of the Canadian Boreal Forest (Rowe, 1972) on the west coast of Newfoundland. Mean July and January daily temperatures are 17 °C and -6 °C, respectively, with a mean precipitation of ~1200 mm/year, of which a large proportion occurs as snow (Damman, 1983; Snyder, 1984). The region supports wet boreal forest with high precipitation due to the humid maritime climate; thus, there is limited forest fire activity and insect activity is the dominant natural disturbance (Thompson et al., 2003). Study sites were located within Forest Management District 15 (Fig. 2.1), which covers a 562 533-ha area (Government of Newfoundland and Labrador, 2003; S. Balsom, Corner Brook Pulp and Paper, Ltd. (CBPP Ltd.), personal communications 2006). Forests in the study area are dominated by conifers, mainly balsam fir, with areas of low-moderate (up to 49.9%) densities of black spruce (*Picea mariana*), depending on site characteristics (Damman, 1983). Other species—including white spruce (*Picea glauca*), white birch (*Betula papyrifera*), white pine (*Pinus strobus*), red maple (*Acer rubrum*) and eastern larch (*Larix laricina*)—occur at low densities, except where succession or site characteristics are particularly favourable. The forest floor/understory ranges from moss-dominated, poor quality sites to rich sites dominated by ferns and other herbaceous plants. The dbh in balsam fir-dominated forests is generally <30 cm and senescence usually begins before stands reach 100 years old (Moroni, 2006). The forest rotation in Newfoundland is 80–120 years (S. Balsom, CBPP Ltd., personal communications 2006).

Approximately 1 million ha of Canadian forest are harvested each year, mostly from the boreal forest (Howard et al., 2004). In Newfoundland, 17 500 ha/year of boreal forest are harvested (Newfoundland Department of Forest Resources and Agrifoods, 1998), and about 2500 ha/year in Forest Management District 15 (CBPP Ltd., 2002). Corner Brook Pulp and Paper Ltd. manages 72% (372 054 ha) of the inventoried land base in this district (CBPP Ltd., 2002; Government of Newfoundland and Labrador, 2003), and clearcutting is the primary method of harvesting in the region (Whitaker and Montevecchi, 1999).

2.3.2 Site selection

Selected sites formed a chronosequence ranging in age from 1 year since harvest (YSH) to 81 to 100 YSH. We randomly selected 15 sites in each of 15 age classes for a total of 225 sites; all sites were >2 ha in size. Age classes were divided into three stages of the forest rotation (and reflect the nature of stand-aging data available from provincial agencies): recently harvested from 1–7 YSH in 1-year increments; early regeneration from 9–20 YSH, with four age classes in 2-year increments; and mid- to late-rotation from 21–100 YSH, with four age classes in 20-year increments. An emphasis was put on the years immediately following harvest to focus on the dynamics in the first 20 years. Forest harvesting in District 15 targets stands aged 80 to 120 years at time of harvest, but some cutting occurs before that age in productive second-rotation forests (S. Balsom, CBPP Ltd., personal communications 2006). We selected harvested sites formerly occupied by forest composed of >75% balsam fir, rated as medium or good yielding (for merchantable volume), and with 50–100% crown closure from across the district. All sites chosen were adjacent to forest access roads. Two age classes, 4 YSH and 81 to 100 YSH, were clustered geographically because of limited site distribution; sites for all other age classes sampled were scattered throughout the district.

2.3.3 Field measurements

Fieldwork was conducted from 1 June to 31 August 2006. At each site, a plot was created measuring 100 m long and 20 m wide. This plot was situated 50 m from the access road at a harvest boundary and ran toward the interior of the site. Data were recorded for all snags with a dbh >9 cm and a height >1.5 m within 10 m on either side of the 100-m transect. Snags were tallied within the plot to determine snag density per hectare for each site. Data recorded for each snag included: GPS waypoint (UTM), species, dbh, height (measured with clinometer), decay class (Table 2.1; Natural Resources Canada, 2008), bark cover and top presence. Percentage bark cover was estimated visually to the nearest 10%, and the top was recorded as broken or intact. Original height of snags with broken tops was estimated based on a regression of height vs. dbh for intact live trees (Moroni, unpublished data).

Snag volume was estimated using published regression equations based on height and dbh (Page et al., 1971; Warren and Meades 1986). Overall snag bole volume was converted to biomass by multiplying snag volume by density based on species and decay classes assigned to individual trees (Table 2.2; Natural Resources Canada, 2008). We assumed that the biomass was 50% C (Moroni, 2006). Bark and branch volumes were estimated based on relationships between bark, branch and stem biomass determined for live trees of each species (Moroni, unpublished data). We assumed that decay classes 1, 2 and 3 snags retained 100%, 50% and 25% of original branches, respectively.

2.3.4 Statistics

We ran a correlation matrix (Minitab Inc., 2006) to determine the single variable that was most correlated to the variation in total snag biomass, using the snag species identified as the predictor variable. After finding which variable was most correlated to total biomass we ran a linear regression to determine the R^2 value.

2.4 Results

2.4.1 Snag density

Snag density (snags/ha) decreased by 50% in the first 4 years following harvest (Fig. 2.2), and then rose to an intermediate plateau averaging 25 snags/ha between 6 and 9 to 10 YSH. Snag densities further declined to eight snags/ha at 20 YSH and remained low until 61 to 80 YSH, when snag density increased again, reaching a high of 44 snags/ha in the 81 to 100 YSH age class.

Balsam fir and white birch were the most abundant snag species. However, only balsam fir snag density changed with time in a pattern similar to that of total snag density (Fig. 2.3). Balsam fir contributed 55% of all snags sampled and comprised 25–65% of snags in any given age class. Birch, the only hardwood sampled, and the only study species not taken during harvesting, showed a very different pattern. Birch snag density was low immediately following harvest, but increased slightly over the next 3 years. Thereafter, birch density remained relatively constant from 5 YSH until 41 to 60 YSH, when it decreased to near 0 snags/ha by 81 to 100 YSH (Fig. 2.3). The other three species sampled (black spruce, white pine and white spruce) followed an intermediate pattern, initially declining over the first 4 YSH, remaining a small but constant presence until 81 to 100 YSH, when their densities reached a level comparable to that of balsam fir (Fig. 2.3). The rise in the density of other study species in the final age class was largely attributable to white pine.

2.4.2 Stem Diameter

Average snag dbh remained relatively constant at ~17 cm from 1 YSH to 20 to 21 YSH, after which it increased to >25 cm by 41 to 60 YSH, then decreased again to ~17 cm by 81 to 100 YSH (Fig. 2.4a). Average balsam fir dbh was 15 to 20 cm, except at 41 to 60 YSH when it reached 22 cm, after which it returned to <20 cm (Fig. 2.4b). In contrast, birch dbh was 18–25 cm up to 20 to 21 YSH, after which it increased to 35 cm by 61 to 80 YSH, followed by a sharp decline (Fig. 2.4b).

2.4.3 Snag volume and biomass

Patterns in total snag volume and biomass (reflected by total height of histogram columns; Fig. 2.5a, b) had a similar temporal pattern as snag density (Fig. 2.2). Balsam fir was the best species predictor for variability of overall snag biomass through a correlation matrix ($p < 0.001$, Minitab Inc., 2006). Balsam fir accounted for 41.5% of the variability ($p < 0.001$). By 15 to 16 YSH, balsam fir had reached a low of 4% of snag biomass (24 kg/ha), whereas white birch made its maximum contribution of 87% of snag biomass (446 kg/ha) (Fig. 2.5b). Conversely at 81 to 100 YSH, balsam fir contributed 46% of snag biomass (940 kg/ha) and white birch only contributed 2% (30 kg/ha) (Fig. 2.5b). Although balsam fir had higher densities than birch for most age classes, white birch constituted a large percentage of snag volume and snag biomass until the final age class. White pine made a substantial contribution to overall volume and snag biomass at 81 to 100 YSH.

2.5 Discussion and conclusions

There were two periods of snag density decline observed in this study but only the second decline, which occurred from 9 to 10 YSH to 15 to 16 YSH, seemed to reflect snag longevity. The snag longevity we observed was comparable to results in a variety of other studies. Garber et al. (2005) predicted that balsam fir snags with a 15-cm dbh would have a <15% probability of standing 15 years following harvest. This is because of the short lifespan and small dbh of balsam fir (Hunt et al., 1999; Thompson et al., 2003). A similar time frame for longevity was estimated for small-diameter (10–30 cm) Douglas-fir and ponderosa pine (*Pinus ponderosa*) snags (Cline et al., 1980; Russell et al., 2006). Hardwood snags of similar dbh tend to persist longer than softwood snags (Cline, 1977) due to higher wood density (Garber et al., 2005). This short overall longevity we observed leaves a large period of low snag abundance (<10 snags/ha) between 15 and 60 YSH (Fig. 2.2). It also diminishes any possible effects of changing harvesting practices because any differences in snag abundance would not be detectable 20 years following harvest, and all sites that had been cut using rudimentary harvesting methods were aged >20 years. Snag density is expected to be higher over time in unmanaged sites (Marage and Lempérière, 2005), but the natural variation in snag densities in pre-harvest landscapes has not been examined in Newfoundland.

The initial period during which snag density declined encompassed the period up to 4 years following harvest and is likely the result of two factors: domestic harvesting for firewood and wind-throw. Snags and live trees considered valuable for firewood (largely hardwoods) are more visible and easier to access after harvest, making them a target for domestic harvesters (Manning et al., 2006). Domestic harvesting is comparable to the salvage logging seen in studies of post-fire disturbance sites; it decreases the average diameter and density of remnant snags, which in turn decreases their overall longevity. Russell et al. (2006) found that salvage logging after fire increased the average rate of fall in remaining snags. The second factor, which may have a greater effect on snag loss than domestic harvesting, is the increased incidence of wind-throw (Lohmand and Helles, 1987; Morrison and Raphael, 1993). Snags surviving or created by harvest become more vulnerable to wind-throw and stem breakage, which facilitates increased decay rates (Stevenson et al., 2006). Russell et al. (2006) found that more exposed snags fell sooner than snags that occurred in clusters.

There were also two periods of snag density increase observed. The initial period of snag density increase occurred from 4 to 6 YSH, following the initial snag density decrease; the second increase occurred at 81 to 100 YSH (Fig. 2.2). The second increase in snag density (≥61 to 80 YSH; Fig. 2.2) was expected due to senescence of regenerating stands (Thompson et al., 2003), but the first increase in snag density between 5 and 9 to 10 YSH occurred during a period of anticipated reductions in snag density. There is little information in the literature about snag recruitment in post-harvest landscapes. Trees left alive on cutblocks following harvesting in western Newfoundland are typically hardwoods, dominated by unmerchantable white birch (Moroni, 2006). These trees do not survive long without the protection of adjacent trees. Solitary trees are especially susceptible to wind-throw when in full leaf because of the “sail” effect (Bebber et al., 2005), thus many residual hardwoods are broken or uprooted when in leaf, with many being blown down in the spring following winter harvesting. Large uprooted hardwoods with fresh leaves were encountered most often during this study on recently harvested sites (C. Smith, personal observations). Residual live trees on cutblocks are likely killed by exposure, potentially resulting in an increase in snag density between 5 and 10 YSH (Fig. 2.2). However, live trees that are left exposed after harvesting and subsequently blown down do not contribute to snag density during any part of the forest rotation. Softwoods are less susceptible to wind-throw than hardwoods (Bebber et al. 2005), but few living softwoods >9 cm dbh are left unharvested in western Newfoundland. Bull and Partridge (1986) recommended that, if live trees were left in clumps or topped and branched to lessen the effects of wind-throw, they could form a post-harvest snag source.

Many studies have concluded that larger diameter snags are the most valuable for wildlife habitat because of their longevity (Morrison and Raphael, 1993; Ganey and Vojta, 2005; Russell et al., 2006). Likewise, longer-standing snags contribute more significantly to biomass pools than smaller snags. Evidence of this can be seen in our data for the period after 15 to 16 YSH, when snag density reached its lowest and average snag dbh increased. The remaining snag biomass is mostly attributable to large-dbh white birch snags (Fig. 2.4b). Decomposition is slower in snags than in fallen coarse woody debris (Zielonka, 2006), and larger-diameter snags are less prone to wind-throw (Ganey and Vojta, 2005) and decompose more slowly because of their larger surface area:volume ratios and increased proportion of decay-resistant heartwood (Bull, 1983; Garber et al., 2005).

Snag biomass followed the same temporal pattern as snag density, although snag biomass did not have the pronounced minima of snag densities at 15 to 20 YSH (Figs. 2.2 and 2.5). White birch dominated snag biomass in many age classes, even though it was consistently present at relatively low densities (snags/ha). A small number of large-dbh white birch snags, individually containing a large biomass due to their size (Fig. 2.4b) and high wood density (Table 2.2), maintained snag biomass when snag densities were low. As a stand ages, the mean snag dbh and range of dbh increases, with large hardwoods persisting for longer periods (Cline et al., 1980; Greif and Archibold, 2000) and new small-dbh snags join the population in the shelter of the growing forest, a pattern typical of boreal forests (Lee et al., 1997). Snag biomass at 81 to 100 YSH in our study was significantly lower than that found in older Newfoundland forests (Sturtevant et al., 1997; Thompson et al., 2003; Moroni, 2006). This was likely because our 81 to 100 YSH sites were at the younger end of this age range with less advanced senescence. The dominance of the last age class by white pine snags likely results from the limited geographic distribution of 81 to 100 YSH sample sites to a location with relatively large numbers of white pine.

Based on current clearcut harvest practices and average tree size distribution, it is unlikely that >10 snags/ha could be maintained through a full forest rotation. This is especially the case where second-rotation forests are harvested before they senesce and thus contain few pre-harvest snags for the post-harvest landscape to inherit. Most snags in post-harvest landscapes of balsam fir-dominated forests in western Newfoundland are short lived and make only a minor contribution to biomass. However, low snag density and biomass could be raised through appropriate management of non-merchantable white birch. The large volume, high average dbh, and high wood density of white birch snags mean they persist longer and contribute a large amount of biomass. By enhancing the presence of white birch snags and subsequent woody debris, snag density over the mid to late rotation could be increased and other forest values (e.g., wildlife habitat) would be addressed. We found that these large, long-lived snags may be of particular importance to enhancing populations of Northern Flickers (*Colaptes auratus*), a key-stone primary cavity excavator in this setting (Smith et al. 2008; see chapter 3).

2.6 References

Bebber, D.P., Cole, W.G., Thomas, S.C., Balsillie, D., Duinker, P., 2005. Effects of retention harvests on structure of old-growth *Pinus strobus* L. stands in Ontario. Forest Ecology and Management 205, 91–103.

Bull, E.L., 1983. Longevity of snags and their use by woodpeckers. In: Davis, J.W., Goodwin, G.A., Ochenfels, R.A. (Eds.), Proceedings of the Snag Habitat Management Symposium. USDA Forest Service General Technical Report RM-GTR-99, pp. 64–67.

Bull, E.L., Partridge, A.D., 1986. Methods of killing trees for use by cavity nesters. Wildlife Society Bulletin 14, 142–146.

Cline, S.P., 1977. The characteristics and dynamics of snags in Douglas-fir forests of the Oregon Coast Range. PhD Dissertation, Oregon State University, Corvallis, OR. 106 p.

Cline, S.P., Berg, A.B., Wight, H.M., 1980. Snag characteristics and dynamics in Douglas-fir forests, western Oregon. Journal of Wildlife Management 44, 773–786.

Conner, R.N., Saenz, D., 2005. The longevity of large pine snags in eastern Texas. Wildlife Society Bulletin 33, 700–705.

Corner Brook Pulp and Paper Ltd., 2002. Five Year Operating Plan, Forest Management District 15 2003–2007. 50 p.

Corner Brook Pulp and Paper Ltd., 2004. Sustainable Forest Management Plan. Version 2. 151 p.

Damman, A.W.H., 1983. An ecological subdivision of the island of Newfoundland. In: South, G.R., Biogeography and Ecology of the Island of Newfoundland. Junk Publishers, The Hague, The Netherlands, pp.163–206.

Everett, R., Lehmkuhl, J., Schellhaas, R., Ohlson, P., Keenum, D., Riesterer, H., Spurbeck, D., 1999. Snag dynamics in a chronosequence of 26 wildfires on the east slope of the Cascade Range in Washington State, USA. International Journal of Wildlife Fire 9, 223–234.

Ganey, J.L., Vojta, S.C., 2005. Changes in snag populations in northern Arizona mixed-conifer and ponderosa pine forests, 1997–2002. Forest Science 51, 396–405.

Garber, S.M., Brown, J.P., Wilson, D.S., Maguire, D.A., Heath, L.S., 2005. Snag longevity under alternative silvicultural regimes in mixed-species forests of central Maine. Canadian Journal of Forest Research 35, 787–796.

Government of Newfoundland and Labrador, 2003. Five-year operating plan for Crown land in forest management district 15 [2003–2008]. NL Department of Environment and Conservation. (online) http://www.env.gov.nl.ca/env/Env/EA%202001/Project%20Info/1054.htm. (Accessed October 2006).

Greif, G.E., Archibold, O.W., 2000. Standing-dead tree component of the boreal forest in central Saskatchewan. Forest Ecology and Management 131, 37–46.

Howard, E.A., Gower, S.T., Foley, J.A., Kucharik, C.J., 2004. Effects of logging on carbon dynamics of a jack pine forest in Saskatchewan, Canada. Global Change Biology 10, 1267–1284.

Hunt, E.R. Jr., Lavigne, M.B., Franklin, S.E., 1999. Factors controlling the decline of net primary production with stand age for balsam fir in Newfoundland assessed using an ecosystem model. Ecological Modelling 122, 151–164.

Hunter, M.L., 1990. Wildfire, forests and forestry: principles of managing forests for biological diversity. Prentice Hall, Englewood Cliffs, NJ. 370 p.

Krankina, O.N., Harmon, M.E., 1994. The impact of intensive forest manag*ement on carbon stores in forest ecosystems.* World Resource Review 6, 161–177.

Kurz, W.A., Apps, M.J., 1993. Contribution of northern forests to the global C cycle: Canada as a case study. Water, Air, and Soil Pollution 70, 163–176.

Lee, P.C., Crites, S., Nietfeld, M., van Nguyen, H., Stelfox, J.B., 1997. Characteristics and origins of deadwood material in aspen-dominated boreal forests. Ecological Applications 7, 691–701.

Lohmand, P., Helles, F., 1987. Wind throw probability as a function of stand characteristics and shelter. Scandinavian Journal of Forest Research 2, 227–238.

Marage, D., Lempérière, G., 2005. The management of snags: a comparison in managed and unmanaged ancient forests of the Southern French Alps. Annals of Forest Science 62, 135–142.

Manning, A.D., Fischer, J., Lindenmayer, D.B., 2006. Scattered trees are keystone structures—implications for conservation. Biological Conservation 132, 311–321.

Minitab Inc., 2006. MINITAB Release version 14.1, State College, Pennsylvania.

Moroni, M.T., 2006. Disturbance history affects woody debris dynamics of Newfoundland balsam fir (*Abies balsamea*) stands. Canadian Journal of Forest Research 36, 3194–3208.

Morrison, M.L., Raphael, M.G., 1993. Modeling the dynamics of snags. Ecological Applications 3, 322–330.

Natural Resources Canada (NRCan), 2008. Canada's National Forest Inventory and online database. NRCan, Ottawa, ON. URL: http://www.pfc.forestry.ca/monitoring/inventory/nfi_e.html. Accessed September 2007.

Newfoundland Department of Forest Resources and Agrifoods (NLDFRA), 1998. 20-Year Forestry Development Plan 1996-2015. NLDFRA, Newfoundland Forest Service, St. John's, NL. 165 p.

Page, G., Robinson, A.J., van Nostrand, R.S., Honer, T.G., 1971. Volume Tables for the Major Tree Species of Newfoundland. Information Report N-X-77, Natural Resources Canada, Canadian Forest Service – Atlantic Forestry Centre, Corner Brook, NL. 21 p.

Peng, C., Jiang, H., Apps, M.J., Zhang, Y., 2002. Effects of harvesting regimes on carbon and nitrogen dynamics of boreal forests in central Canada: a process model simulation. Ecological Modelling 155, 177–189.

Rowe, J.S., 1972. Forest Regions of Canada. Natural Resources Canada, Canadian Forest Service, Ottawa, ON. Publ. 1300. 172 p.

Russell, R.E., Saab, V.A., Dudley, J.G., Rotella, J.J., 2006. Snag longevity in relation to wildfire and postfire salvage logging. Forest Ecology and Management 232, 179–187.

Smith, C.Y., Warkentin, I.G., Moroni, M.T. 2008. Snag availability for cavity nesters across a chronosequence of post-harvest landscapes in western Newfoundland. Forest Ecology and Management 256, 641-647.

Snyder, J.E., 1984. Marten use of clear-cuts and residual forest stands in western Newfoundland. M.Sc. Thesis, University of Maine, Orono, ME. 31 p.

Stevenson, S.K., Jull, M.J., Rogers, B.J., 2006. Abundance and attributes of wildlife trees and coarse woody debris at three silvicultural systems study areas in the interior Cedar–Hemlock zone, British Columbia. Forest Ecology and Management 233, 176–191.

Sturtevant, B.R., Bissonette, J.A., Long, J.N., Roberts, D.W., 1997. Coarse woody debris as a function of age, stand structure, and disturbance in boreal Newfoundland. Ecological Applications 7, 702–712.

Thomas, J.W., 2002. Dead Wood: from Forester's Bane to Environmental Boon. In: Laudenslayer, W.F. Jr., Shea, P.J., Valentine, B.E., Weatherspoon, C.P., Lisle, T.E. (Tech. Coords.), Proceedings of the Symposium on

Ecology and Management of Deadwood in Western Forests, 2–4 November 1999, Reno, NV. USDA Forest Service General Technical Report PSW-GTR-181, pp. 3–9.

Thompson, I.D., Larson, D.J., Montevecchi, W.A., 2003. Characterization of old “wet boreal” forests, with an example from balsam fir forests of western Newfoundland. Environmental Reviews 11, S23–S46.

Warren, G.R., Meades, J.P., 1986. . Wood defect and density studies II. Total and net volume equations for Newfoundland’s forest management units. Information report N-X-242, Natural Resources Canada, Canadian Forest Service – Atlantic Forestry Centre, Corner Brook, NL.

Whitaker, D.M., Montevecchi, W.A., 1999. Breeding bird assemblages inhabiting riparian buffer strips in Newfoundland, Canada. Journal of Wildlife Management 63, 167–179.

Yamasaki, M., Leak, W.B., 2006. Snag Longevity in Managed Northern Hardwoods. Northern Journal of Applied Forestry 23, 215–217.

Yatskov, M., Harmon, M.E., Krankina, O.N., 2003. A chronosequence of wood decomposition in the boreal forests of Russia. Canadian Journal of Forest Research 33, 1211–1226.

Zielonka, T., 2006. Quantity and decay stages of coarse woody debris in old-growth subalpine spruce forests of the western Carpathians, Poland.

Canadian Journal of Forest Research 36, 2614–2622.

Table 2.1. Descriptions of decay classes used to classify snags within the studied sites (NRCan 2006)

	Decay Class		
Attribute	1	2	3
Wood texture	Intact, hard	Intact, hard to partly decaying	Hard, large pieces, partly decaying
Portion on ground	Elevated on support points	Elevated but sagging slightly	Sagging near ground, or broken
Twigs <3 cm (if originally present)	Twigs present	No twigs	No twigs
Bark	Bark intact	Intact or partly missing	Trace bark
Shape	Round	Round	Round
Invading roots	None	None	In sapwood

Table 2.2. Snag density (kg/m^3) by species and decay class (Moroni 2006)

Species	Decay Class		
	1	2	3
Abies balsamea	340	271	202
Picea mariana	410	341	272
Picea glauca	350	281	212
Betula papyrifera	506	424	342
Pinus strobus	360	291	222

Figure 2.1. Newfoundland, with Forest Management District 15 shaded.

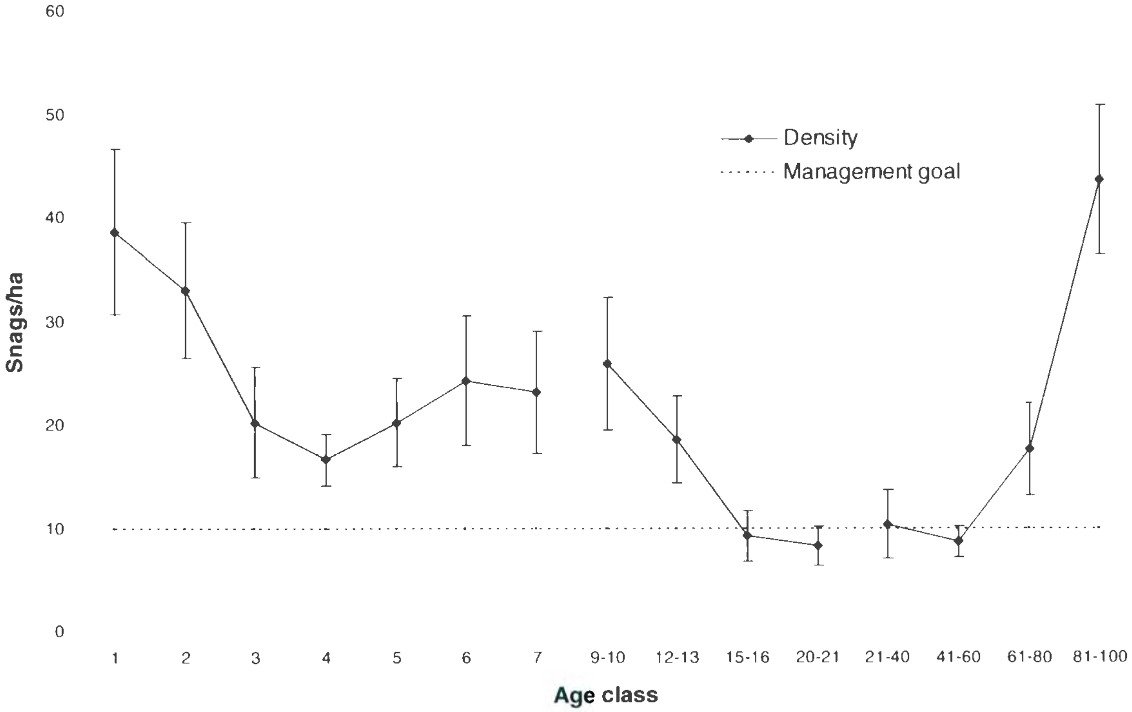


Figure 2.2. Snag density (snags/ha) within study sites by age class (mean ± SE; *n* = 15; dotted line indicates management goal; note changing x-axis scale)

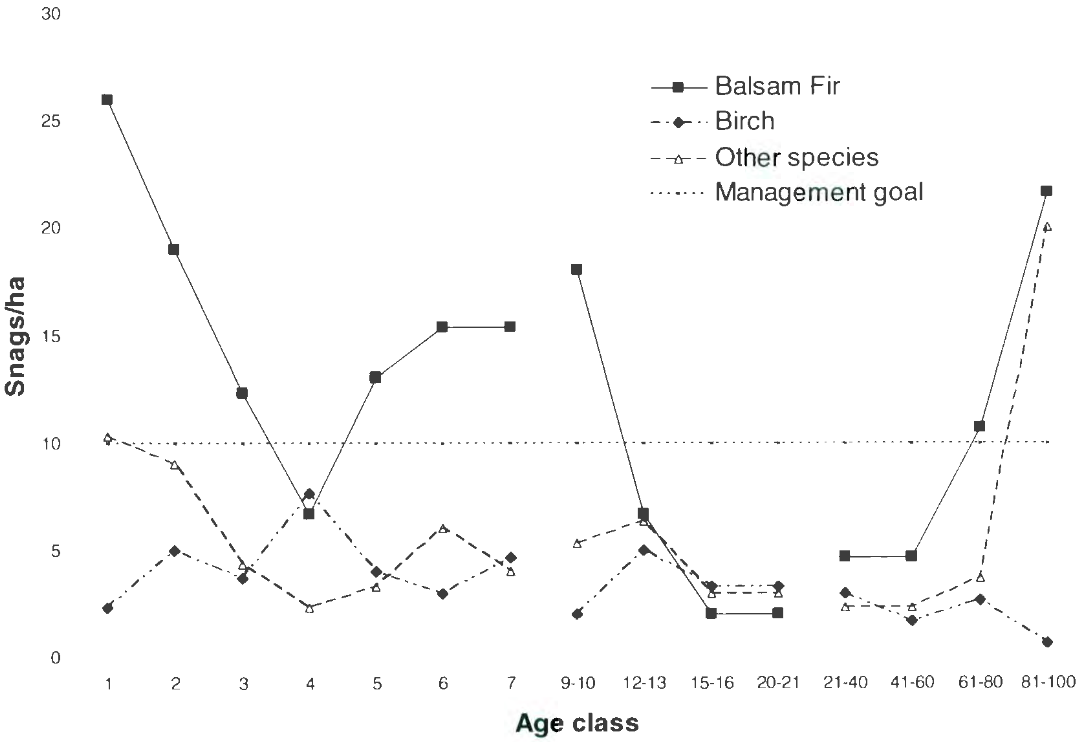


Figure 2.3. Snag density (snags/ha) by species and age class (mean ± SE; *n* = 15; note changing x-axis scale).

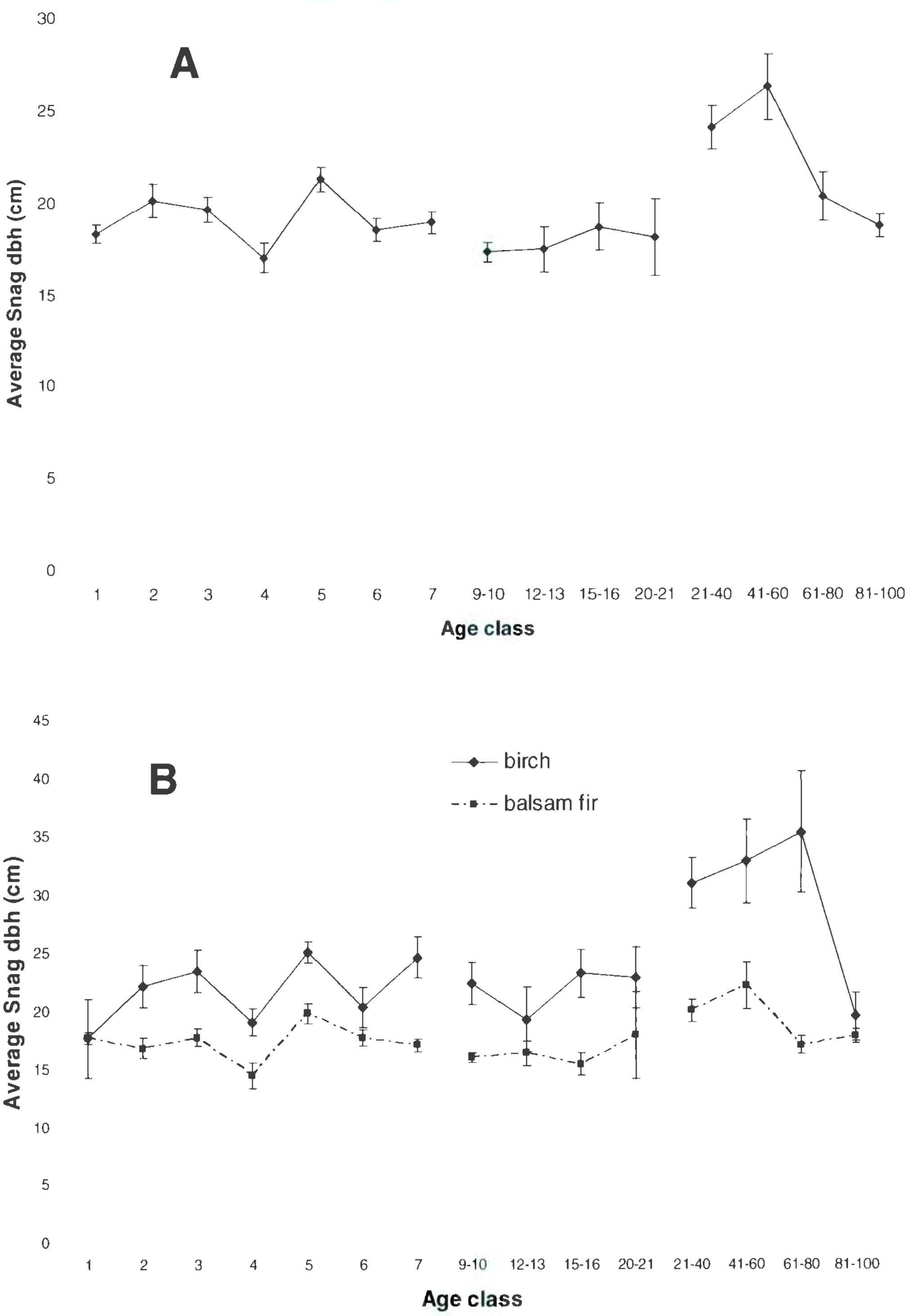


Figure 2.4. Average snag dbh by a) age class; b) species and age class (mean ± SE; *n* = 15; note changing x-axis scale).

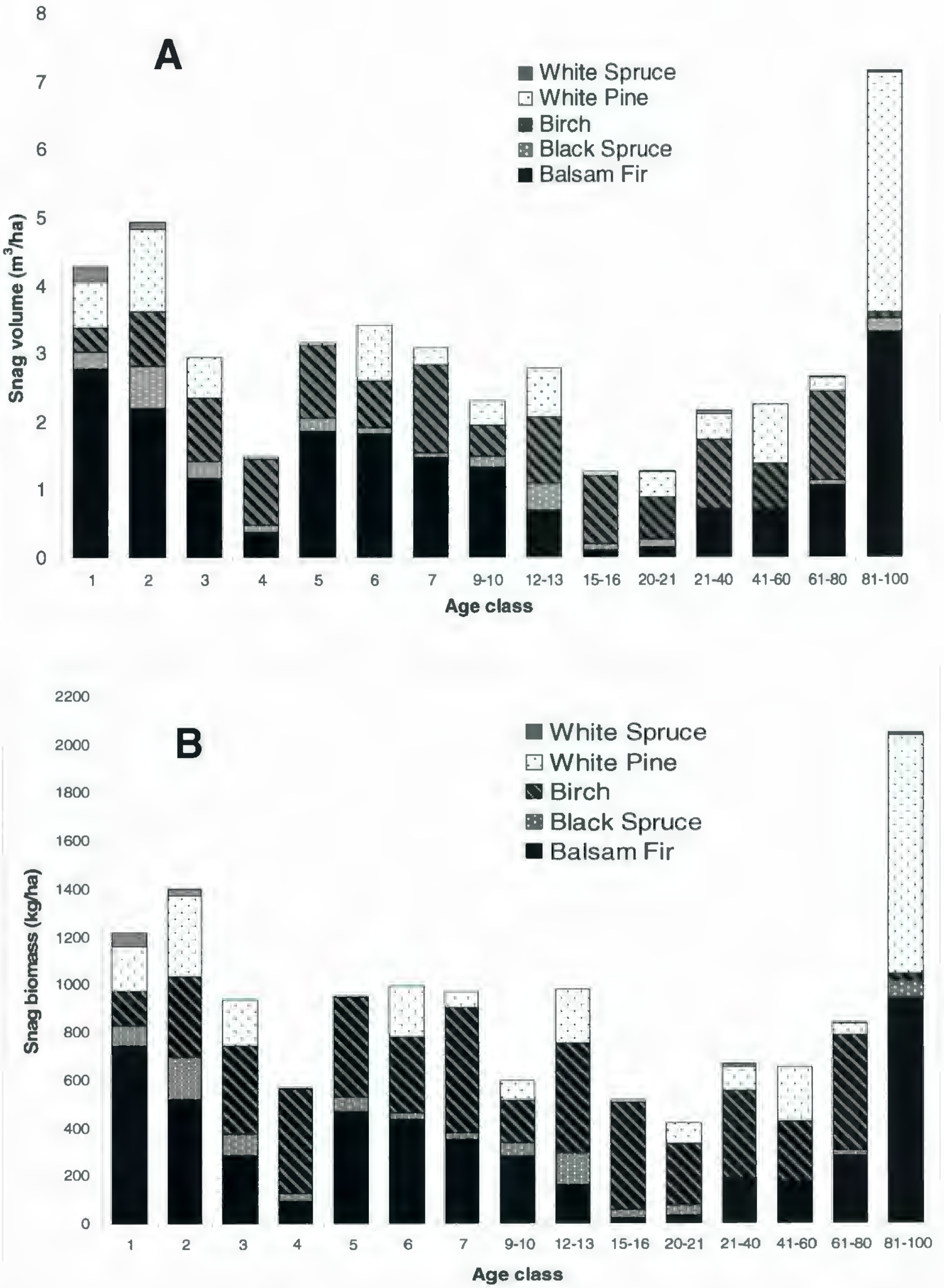


Figure 2.5. a) Snag volume (m^3/ha) by a) species and age class ($n = 15$), and b) snag carbon (kg/ha) by species and age class ($n = 15$).

3.0 Snag availability for cavity nesters across a chronosequence of post-harvest landscapes in western Newfoundland

3.1 Abstract

We examined the availability and quality of standing dead trees (snags) for nesting habitat in a harvest chronosequence of boreal forests dominated by balsam fir (*Abies balsamea*) in western Newfoundland. Snag density declined substantially 10–15 years after harvest, then increased to reach its highest level in 81- to 100-year-old forests that had become senescent. Most (55%) of 1260 snags encountered were balsam fir, which contained 41% of the 81 cavities identified. Cavity presence was most strongly positively correlated to Snag diameter at breast height (dbh), followed by decay class, time since harvest and height. Less than 40% of snags available throughout the chronosequence had large enough dbh for cavity nesters. Downy Woodpeckers (*Picoides pubescens*) were responsible for 47% of all cavities identified, excavating balsam fir ~50% of the time. Two larger cavity nesters present, Northern Flickers (*Colaptes auratus)* and Three-toed Woodpeckers (*Picoides tridactylus*), were more likely to use large-diameter white birch (*Betula papyrifera*) snags. Northern Flickers excavated the largest cavities, potentially providing habitat for the greatest variety of secondary cavity-nesters. Managing for snags with >30 cm dbh, which flickers target for excavation, has the greatest potential to enhance the broader snag cavity-based community in western Newfoundland.

3.2 Introduction

Standing dead trees (hereafter snags) are an important but variably available resource for a range of forest birds and mammals (Bull, 1983; Morrison and Raphael, 1993; Martin et al., 2004). Snags support a diverse food base for a host of insectivorous vertebrates, and provide roosting and nesting opportunities for many wildlife species (Bunnell et al., 1999; Spiering and Knight, 2005). Snag availability for wildlife has become a component of forest management decisions (Bull et al., 1997; Garber et al., 2005), yet few studies have reported data on snag availability across periods of time reflecting entire forest rotations. Such decisions require an understanding of snag dynamics and usage by wildlife over time (Garber et al., 2005), especially following disturbances that lead to a major shift in forest composition and structure.

Prominent among vertebrates using snags are cavity-nesting birds, particularly the woodpeckers (*Picidae*), which forage for insects often found in snags but also excavate cavities in both live and dead trees (Raphael and White, 1984; Farris et al., 2004; Remm et al., 2006). As primary cavity nesters, woodpeckers create habitat for other species and play roles in seed dispersal, soil aeration, organic decomposition, and pest control (Everett and Otter, 2004; Farris et al., 2004). Because they excavate new nest cavities each breeding season, there may be a rapid increase in habitat for non-excavating secondary cavity nesters in communities where woodpeckers are present (Martin et al., 2004; Walter and Maguire, 2005; Remm et al., 2006), especially where there is a high density of suitable snags available.

Managing forests with the intent to encourage woodpeckers and secondary cavity nesters involves providing snags appropriate for cavity excavation (Bull et al., 1980; Cimon, 1983). Targeting mature, but pre-senescent, stands for harvest reduces the number and size of snags left on the post-harvest landscape (Cline et al., 1980; Bull et al., 1997; Thompson et al., 1999); this reduces eventual snag availability and, therefore, wildlife abundance and diversity (Haney and Schaadt, 1996; Zack et al., 2002). Woodpeckers select snags with a large diameter at breast height (dbh) to support their nests (Swallow et al., 1986). Although cavity nesters display species-specific differences in their snag size preferences, larger snags are used by more species and tend to stand longer than smaller snags (Bull et al., 1980; Schieck and Song, 2006). Cavity-nesting birds also select snags for excavation based on decay class, species, bark cover, and whether the top of the snag is still intact or broken off (Mannan et al., 1980; Raphael and White, 1984; Moorman et al., 1999; Farris et al., 2004). Studies of snags as wildlife habitat have suggested that snag abundance may not be as important as the quality (possessing characteristics necessary for cavity excavation) of snags available (Bull et al., 1997; Imbeau and Desrochers, 2002). Thus, cavity-nesting bird densities are dependent on the availability of high-quality snags and trees (Runde and Capen, 1987).

Disturbance can greatly alter the landscape and, consequently, the availability of snags for cavity-nesting birds, however depending on the source of the disturbance (natural vs. human caused), the outcome may be very different in terms of stand structure. Fire is the main natural disturbance in many forest systems, but in much of the boreal region of North America, harvesting is becoming increasingly prevalent (Simon et al., 2002). Concomitant with the rising demand for wood products, harvesting levels in Canadian forests (largely boreal) have increased to approximately 1 million ha per year (Howard et al., 2004). Until recently, the prominent use of clearcutting in these harvest operations left landscapes with very few snags or living trees (Simon et al., 2002), creating post-harvest landscapes very different from post-fire systems in terms of the amount of snags and deadwood remaining (Schieck and Song, 2006). In their review of studies comparing post-harvest and post-fire landscapes in western Canada, Schieck and Song (2006) identified significant differences between these landscapes in bird community composition during the years immediately following disturbance. Cavity-nesting birds were abundant in post-fire disturbance sites while the bird communities of post-harvest landscapes were dominated by open-meadow or shrubby habitat bird species. However, they also found that with time the physical and community differences between disturbance types lessened, leading to old-forest stands with similar structural and avifaunal community characteristics (Schieck and Song, 2006). These findings indicate that management for cavity nesters and other wildlife in post-harvest landscapes may be most critical in the years immediately following harvesting, rather than later successional stages, at least in boreal North America. It remains unclear whether attempts to mimic natural disturbance with modified forest harvest techniques will help retain the expected early seral-stage avifaunal community in general (Schieck and Song, 2006). Nor is it known what the general nature of the post-harvest cavity-nesting community is in the forests of eastern boreal North America (e.g., Newfoundland), where fire is less prevalent and the scale of natural disturbance is smaller. The aim of this study was to determine the availability and quality of snags for cavity-nesting birds and other wildlife in post-harvest balsam fir-dominated boreal forests of western Newfoundland, Canada. We predicted that snag availability would decrease below 10 snags/ha around the mid-rotation stage following harvesting. We also predicted that very few large-diameter snags would be available for cavity nesting species.

3.3 Methods

3.3.1 Study area and site selection

Our study was conducted in the Western Newfoundland Ecoregion (Forest Section B28b) of the Canadian Boreal Forest (Rowe, 1972) on the west coast of Newfoundland, Canada. The mean July and January daily temperatures are 17°C and -6°C, respectively, with a mean precipitation of ~1200 mm/year, of which a large proportion occurs as snow (Damman, 1983; Snyder, 1984). Fire in this region is limited, and insect outbreaks are the dominant natural disturbance (Thompson et al., 2003). Study sites were located within forest management district 15 (Fig. 2.1), which covers an area of 562 533 ha (Govt. of Newfoundland and Labrador, 2003). Selected stands were dominated by balsam fir, with low-moderate densities and scattered stands dominated by black spruce (*Picea mariana*), depending on site characteristics (Damman, 1983). Other species, including white spruce (*Picea glauca*), white birch (*Betula papyrifera*), white pine (*Pinus strobus*), red maple (*Acer rubrum*), and eastern larch (*Larix laricina*), occur at low densities except at scattered locations where site characteristics favor greater abundance. The dbh (at 1.3 m) in balsam fir-dominated forests is generally <30 cm, and stands rarely reach 100 years old before senescence begins (Moroni, 2006). The forest rotation in Newfoundland has been 80–120 years (Setterington et al., 2000), but the harvest rotation is currently as short as 60 years where productivity is high (S. Balsom, Corner Brook Pulp and Paper, Ltd. pers. comm. 2006) and clearcutting is the primary method of harvesting in the region (Whitaker and Montevecchi, 1999).

We selected 15 sites for each of 15 age classes in a chronosequence ranging from one year since harvest (YSH) to 81–100 YSH, for a total of 225 sites. Age classes were divided into three stages of the forest rotation (and reflect the nature of stand-aging data available from provincial agencies): recently harvested from 1–7 YSH in 1-year increments; early regeneration from 9–20 YSH, with four age classes in 2-year increments; and mid- to late-rotation from 21–100 YSH, with four age classes in 20-year increments (Fig. 3.1). All sites selected were on harvest blocks >2 ha in size, composed of >75% balsam fir, with 50–100% crown closure, and having medium or good site quality (based on merchantable volume). For two age classes, 4 YSH and 81–100 YSH, the sites chosen were clustered geographically because of limited availability in the forest management district.

3.3.2 Field measurements and data collection

Fieldwork was conducted from 1 June to 31 August 2006. At each site, beginning 50 m from the access road and at the cut block boundary, a 100 m long transect was established running toward the interior of the cut block to create a plot 20 m wide. Data were recorded for all snags inside the plot having a dbh ≥9 cm and a height >1.5 m. Snags were tallied within the plot to determine snag density per hectare for each site. Data recorded for each snag included GPS waypoint (UTM), species, dbh, height (measured with clinometer), decay class, bark cover, and top presence (See chapter 2; Smith et al. submitted). Bird and other wildlife data recorded for each snag included number of cavities present, species of woodpecker that created each cavity, presence of foraging and excavation, and any animals occupying the cavity.

Nest cavities were attributed to primary excavators based on cavity dimensions characteristic to each species. The species of woodpeckers found in western Newfoundland include Downy Woodpeckers (*Picoides pubescens*), Hairy Woodpeckers (*Picoides villosus*), Black-backed Woodpeckers (*Picoides arcticus*), Three-toed Woodpeckers (*Picoides tridactylus*), and Northern Flickers (*Colaptes auratus*). Downy Woodpeckers are commonly found in deciduous, riparian, and mixed forest, and nest in either snags or live trees with advanced heart rot and an average dbh >25 cm (Setterington et al., 2000; Jackson and Ouellet, 2002). Downy Woodpecker cavities were easily distinguished from other potential excavators by the circular shape of the entrance compared with the distinctly oval cavity openings of other species; the entrance has an average diameter of 3 cm

(Jackson and Ouellet, 2002). Hairy Woodpeckers are known to use both deciduous and coniferous trees for nesting (Saab et al. 2004); they will nest in snags but prefer live trees with advanced heart rot and a dbh >25 cm (Jackson et al., 2002). The dimensions of a Hairy Woodpecker cavity entrance average 4.8 cm high and 3.8 cm wide (Jackson et al., 2002). Three-toed Woodpeckers are associated with mature forest and nest in either coniferous or deciduous snags, but prefer a dbh >27 cm; the average diameter of their cavity entrance, which may be irregularly shaped, is between 3.8 and 4.5 cm (Leonard, 2001). Black-backed Woodpeckers are strongly associated with fire-disturbed forest in most parts of the boreal as well as in the forests of the western United States (Hutto 1995; Hoyt and Hannon 2002) but are also known to use old-growth forest in places such as western Newfoundland (Thompson et al. 1999). They nest in live trees and snags of various species having an average dbh of approximately 27 cm (Dixon and Saab, 2000; Setterington et al., 2000). Black-backed Woodpeckers have a cavity entrance that is 4.4 cm in diameter, with a flattened sill-like spot on the bottom (Dixon and Saab, 2000). Northern Flickers prefer open forest landscapes, and excavate their cavities in dead trees with sufficient decay to provide soft wood and an average dbh >30 cm. Flickers have the largest cavity entrance among these species, approximately 8 cm high and 7.4 cm wide (Moore, 1995).

3.3.3 Statistics

We used backward stepwise selection to determine the best model with an α value of 0.15 being required to enter or leave the model; no interactions qualified to be in the model, nor did snag species. We then ran a binary logistic multiple regression to assess which parameter(s) (from among snag dbh, decay class, age class, and height) were the best predictor of cavity presence in a snag (Minitab Inc., 2006). Logistic regression applies maximum-likelihood estimation after transforming the dependent into a logit value; it does not require normally distributed variables. An α level of 0.05 was used to determine significance for all statistical tests.

3.4 Results

Data for 1260 snags from 225 sites covering 15 age classes were collected; there were a total of 81 cavities contained in 45 snags. Cavity presence was positively correlated with snag dbh (binary logistic regression: $z = 8.47$, $p < 0.0001$), increasing in occurrence as dbh increased. Cavity presence was also positively correlated with, decay ($z = 2.98$, $p = 0.003$), and age class ($z = 2.13$, $p = 0.033$), reaching its highest level in the oldest age class (Fig. 3.2). There was a negative correlation between cavity presence and height ($z = -2.13$, $p = 0.033$), with cavities occurring less often in taller snags. There were no significant interaction effects.

Average snag dbh remained relatively constant at ~17 cm throughout the rotation, with an increase of about 10 cm by 41–60 YSH (Fig. 3.3). However, 62% of all snags had a dbh between 10 and 20 cm, and average dbh was 19.4 ± 0.72

cm (mean ± SD; Fig. 3.4). In contrast, snags with cavities had an average dbh of 30.7 ± 1.37 cm, and cavity nesters seem to prefer snags with a larger dbh (Fig. 3.4). Approximately 70% of cavities recorded were in decay class 3 snags; but only 463 of the 1260 snags encountered had decayed enough for potential excavation (decay class 2 or 3), and in addition, had a dbh >20 cm. Thus, there were fewer than two snags/ha on average per study site with a dbh >20 cm available for cavity excavation. The average height of snags containing cavities was 5.68 ± 0.28 m, compared with 5.34 ± 0.07 m for all other snags.

Snag species was not a useful predictor of cavity presence in a snag and woodpeckers appeared to create cavities in snags based largely on availability. Balsam fir was the most abundant snag-forming species (669, 53%) followed by white birch (266, 21%), white pine (175, 14%), black spruce (134, 11%), and white spruce (16, 1%). Overall, 41% of cavities were in balsam fir, 29% in white pine, 21% in birch, and 9% in black spruce. The largest proportion of cavities was excavated by Downy Woodpeckers (47%), and although most prominent on recently harvested (1-7 YSH) sites, Downy Woodpeckers were responsible for about one third of cavities in early regeneration, as well as mid-to-late successional stands (Fig. 3.2). Hairy Woodpecker cavities were the second-most abundant and showed little change in abundance over time, a pattern similar to Black-backed woodpecker cavities, which were the least abundant overall. Northern Flicker cavities reached a maximum in early regenerating forest, and were least abundant in the older sites where canopy closure inhibited their activities and snags containing flicker cavities excavated early in the rotation fell to the forest floor. Three-toed Woodpecker cavities were rare early in the forest rotation, but became the second-most abundant during the middle to late phase of the forest rotation.

Downy Woodpecker and Hairy Woodpecker cavities were most common in balsam fir snags, which contained 50% of these cavities (Fig. 3.5). Northern Flickers and Three-toed Woodpeckers constructed their cavities in birch snags ~50% of the time. Most cavities that were formed naturally or of unknown creation were found in birch snags. Black-backed Woodpeckers excavated four of the cavities recorded, two of which were found in white pine snags.

Of the 1260 snags recorded in the study, only 12% showed no evidence of foraging, with high levels of foraging occurring throughout all age classes.

3.5 Discussion and conclusions

The loss of snags soon after harvest typically leads to a dramatic change on the landscape for cavity-nesting species (Haney and Schaadt, 1996; Zack et al., 2002; Schieck and Song, 2006). Previously (Smith et al., submitted; see chapter 2), we found that snags in post-harvest landscapes of western Newfoundland were generally short lived. Many snags (either existing before, or created during, harvest) appear to be quickly lost to windthrow and domestic harvest for firewood, with the few remaining individuals falling within 10–15 years after harvest (Smith et al., submitted; see chapter 2). The small stature of the predominant but short-lived balsam fir contributed to the overall short lifespan of snags on this landscape. The most common hardwood in the study area, white birch (Smith et al., submitted; see chapter 2), would be expected to have enhanced longevity because of its dense tissue (Cline, 1977; Garber et al., 2005). However, in western Newfoundland, the targeting of birch for domestic harvest and its apparent heightened susceptibility to windthrow immediately after harvest (Smith et al., submitted; see chapter 2) have meant a relatively limited presence among snags during the early post-harvest period. Similar to other studies (see Schieck and Song, 2006), we found that snag density increased in the latter stages of the forest rotation when senescence occurred. Understanding the time frame of these processes is important for managing habitat availability for cavity-nesting species (Walter and Maguire, 2005). Snags left at the time of harvest may only be available for a short period in the entire forest rotation (Bull et al., 1980) unless care is taken to protect snags with characteristics that enhance their potential longevity.

Larger-diameter snags are the most valuable for wildlife habitat because of their longevity and ability to support a wider range of animal species (Morrison and Raphael, 1993; Ganey and Vojta, 2005; Russell et al., 2006). When snag density was at its lowest during our study (15–60 YSH), the average dbh was at its highest point, indicating that larger snags survived longest after harvest and consequently provided habitat for the greatest period of time (Fig. 3.3). Consistent with this finding, over half (62%) the snags encountered in our study were <20 cm in diameter, whereas 90% of the snags containing cavities were >20 cm in diameter (four Downy Woodpecker cavities were in trees with dbh between 15 and 20 cm). Thus, similar to other studies (e.g., Swallow et al. 1986), not only do large dbh snags last longer, but they have a greater likelihood of being chosen for cavity excavation. Similar results have been found for woodpeckers excavating in lodgepole pine *(Pinus contorta) and* ponderosa pine (*Pinus ponderosa*) forests, with no cavities found in snags <25 cm dbh (Bull, 1983). Hairy Woodpeckers have been reported to require a minimum dbh of 25 cm whereas Black-backed woodpeckers and Three-toed woodpeckers have been reported to require a minimum dbh >30 cm (Bull et al., 1980; Leonard, 2001; Jackson et al., 2002). We would caution however that while small-diameter snags (<20 cm) would appear to have little value for nesting, they were still heavily used by woodpeckers for foraging (for which evidence was found on ~90% of snags in this study).

The largest dbh trees observed during this study were white pine and white birch. The white pine is a slow-growing species that is no longer common in Newfoundland because of intensive harvesting in the past and expansion of the white pine blister rust disease (*Cronartium ribicola*) (Rajora et al., 2002), making it difficult to manage. In contrast, white birch is common on the landscape and is frequently left standing after harvest. The potential contribution of these trees to the snag population would be enhanced if birch were managed to: (1) survive their increased susceptibility to windthrow (Bebber et al., 2005), and (2) exclude them from domestic harvest (Smith et al., submitted; see chapter 2). Managing white birch to maximize snag recruitment would require selective retention of large dbh individuals that were already snags at the time of harvest, but also the creation of snags through the topping and trimming of live trees (e.g., Bull and

Partridge, 1986). Thus, wildlife habitat availability could be improved both immediately after harvest through retention/protection of the extant white birch snag population, and (based on the prominence of birch snags in the low-density 21–40 YSH age class; Smith et al. submitted; see chapter 2) perhaps beyond 20 years into the post-harvest rotation based on the creation of new snags at the time of harvest.

Maintaining white birch on the landscape would directly support species associated with deciduous trees such as Downy and Hairy Woodpeckers, as well as Northern Flickers (Setterington et al., 2000). The increase of large diameter trees left behind on the landscape would also increase overall cavity excavation. Given that ~50% of Northern Flicker cavities were in white birch, and flickers used birch for cavities roughly in proportion to availability, we would predict that cavity creation (particularly in the more open landscapes favored by this species) would increase along with Northern Flicker numbers. As a keystone excavator (Martin et al., 2004), greater Northern Flicker populations would have the potential to provide more breeding habitat for a variety of secondary cavity-nesting species, including large-bodied species such as Northern Hawk Owls (*Surnia ulula*), American Kestrels (*Falco sparvenius*), and Boreal Owls (*Aefolius funereus*), all of which make use of cavities in snags left behind on clearcuts in Newfoundland (Gosse and Montevecchi, 2001).

3.5.1 Management implications

The quality (both diameter and state of decay) of snags on the landscape may be more important than the overall abundance in terms of supporting cavity-nesting birds (Farris et al., 2004). In western Newfoundland, however, forest management practices have provided an average of <2 snags per cutover with adequate dbh (>20 cm) and sufficient decay for cavity excavation. Two factors acting against the creation of such quality snags are clearcut harvesting, which tends to decrease the number of large trees and snags retained on the landscape (Simon et al., 2002), and short harvest rotations (<100 years), which prevent forests from senescing and producing larger snags (Imbeau and Desrochers, 2002). Together, these factors may be the reason why post-harvest landscapes in Newfoundland have limited cavity-nesting populations immediately following harvest. It may also explain the absence of Black-backed Woodpeckers during late-successional stages on these post-harvest landscapes due to their preference in western Newfoundland for old-growth forest (80+ years) (Thompson et al. 1999; Setterington et al., 2000). Implementing management to retain greater numbers of quality snags on post-harvest landscapes would narrow the gap in differences between natural disturbances and harvesting (Schieck and Song, 2006). Although, as evident in the widespread occurrence of foraging activity on snags of all sizes, caution must be exercised to leave not only potential nesting habitat but also foraging trees.

By focusing on the protection or creation of high-quality snags at the time of harvest, it may be possible to greatly extend the availability of nesting habitat for primary and secondary cavity nesters. Live white birch could be killed to create snags that would remain on the landscape longer into the forest rotation (Moorman et al., 1999) and live trees could also be left in clusters to promote snag recruitment. Bull and Partridge (1986) found that topping was the most effective method for creating snags as it reduces susceptibility to wind and allows faster fungal and decomposing bacteria invasion. Snags created by this method were also most frequently used for nesting and foraging (Bull and Partridge, 1986; Hallet et al., 2001).

Managing for large-cavity excavators, such as Northern Flickers, and the associated range of secondary nesters would be greatly facilitated through the retention and/or creation of larger-diameter white birch snags in Newfoundland. Current forestry guidelines in this jurisdiction suggest that ten snags/ha are to be left following harvest (following Cline et al., 1980). It would be more appropriate to leave ten high-quality snags/ha, i.e., snags with a large enough dbh to be potential cavity-nesting trees for the largest species in the region.

3.6 References

Bebber, D.P., Cole, W.G., Thomas, S.C., Balsillie, D., Duinker, P., 2005. Effects of retention harvests on structure of old-growth *Pinus strobus* L. stands in Ontario. Forest Ecology and Management, 205, 91–103.

Bull, E.L., Longevity of snags and their use by woodpeckers, 1983. In: Davis, J.W., Goodwin, G.A., Ochenfels, R.A., (Eds.), Proceedings of the Snag Habitat Management Symposium. USDA Forest Service General Technical Report RM-GTR-99, pp. 64–67.

Bull, E.L., Parks, C.G., Torgersen, T.R., 1997. Trees and logs important to wildlife in the interior Columbia River basin. USDA Forest Service General Technical Report PNW-GTR-391. Pacific Northwest Research Station, Portland, Oregon.

Bull, E.l., Partridge A.D., 1986. Methods of killing trees for use by cavity nesters. Wildlife Society Bulletin, 14, 142-146.

Bull, E.L., Twombly, A.D., Quigley, T.M., 1980. Perpetuating snags in managed mixed conifer forests of the Blue Mountains, Oregon. P. 325-336. In: Management of western forests and grassland for nongame birds. Workshop proceedings. USDA Forest Service General Technical Report

INT-86. Intermountain Forest and Range Experiment Station, Ogden, Utah.

Bunnell, F.L., Kremaster, L.L., Wind, E., 1999. Managing to sustain vertebrate richness in forests of the Pacific Northwest: relationships within stands. Environmental Revues, 7, 97-146.

Cimon, N., 1983. A simple model to predict snag levels in managed forests, In: Snag Habitat Management: Proceedings of the Symposium. USDA Forest Service General Technical Report RM-99. Flagstaff, Arizona.

Cline, S.P., 1977. The characteristics and dynamics of snags in Douglas-fir forests of the Oregon Coast Range, Ph.D. Thesis. Oregon State University, Corvallis, OR. 106pp.

Cline, S.P., Berg, A.B., Wight, H.M., 1980. Snag characteristics and dynamics in Douglas-fir Forests, Western Oregon. Journal of Wildlife Management, 44, 773-786.

Damman, A.W.H., 1983. An ecological subdivisions of the island of Newfoundland. In: Biogeography and ecology of the island of Newfoundland. *Edited by* G.R. South. Junk Publishers, The Hague, Netherlands. 163-206.

Dixon, R.D., Saab, V.A., Black-backed woodpecker (*Picoides arcticus*), 2000. In: Poole, F., Gill, F. (eds.), The Birds of North America, No. 509. The Birds of North America, Inc., Philadelphia, PA.

Everett, K.T., Otter, K.A., 2004. Presence of cavities in snags retained in forest cutblocks: Do management policies promote species retention? Canadian Field-Naturalist, 118, 354-359.

Farris, K.L., Huss, M.J., Zack, S., 2004. The role of foraging woodpeckers in the decomposition of ponderosa pine snags. The Condor, 106, 50-59.

Ganey, J.L., Vojta, S.C., 2005. Changes in snag populations in Northern Arizona mixed-conifer and Ponderosa Pine Forests, 1997-2002. Forest Science, 51, 396-405.

Garber, S.M., Brown, J.P., Wilson, D.S., Maguire, D.A., Heath, L.S., 2005. Snag longevity under alternative silvicultural regimes in mixed-species forests of central Maine. Canadian Journal of Forestry Research, 35, 787-796.

Gosse, J.W., Montevecchi, W.A., 2001. Relative abundance of forest birds of prey in western Newfoundland. Canadian Field-Naturalist, 115, 57-63.

Govt. of Newfoundland and Labrador, 2003. Five year operating plan for crown land in forest management district 15 [2003-2008]. Environment and Conservation. http://www.env.gov.nl.ca/env/Env/EA%202001/Project%20Info/1054.htm. (Accessed October 2006).

Hallet, J.G., Lopez, T., O'Connell, M.A., Borysewicz, M.A., 2001. Decay dynamics and avian use of artificially created snags. Northwest Science, 75, 378-386.

Haney, J.C., Schaadt, C.P., Functional roles of eastern old growth in promoting forest bird diversity, 1996. In: Davis, M.B. (ed.), Eastern old growth forests: prospects for rediscovery and recovery. Island Press, Washington D.C., USA, 76-88.

Howard, E.A., Gower, S.T., Foley, J.A., Kucharik, C.J., 2004. Effects of logging on carbon dynamics of a jack pine forest in Saskatchewan, Canada. Global Change Biology, 10: 1267–1284.

Hoyt, J.S., Hannon, S.J., 2002. Habitat associations of black-backed and three-toed woodpeckers in the boreal forest of Alberta. Canadian Journal of Forest Research, 32: 1881-1888.

Hutto, R.L., 1995. Composition of bird communities following stand-replacement fires in Northern Rocky Mountain (USA) conifer forests. Conservation Biology, 9: 1041-1058.

Imbeau, L., Desrochers, A., 2002. Foraging ecology and use of drumming trees by three-toed woodpeckers. Journal of Wildlife Management, 66, 222-231.

Jackson, J.A., Ouellet, H.R., 2002. Downy Woodpecker (*Picoides pubescens*). In: Poole, A., Gill, F., (eds.), The Birds of North America, no. 613. The Birds of North America, Inc., Philadelphia, PA.

Jackson, J.A., H.R. Ouellet, Jackson, B.J.S., Hairy Woodpecker (*Picoides villosus*), 2002. In: Poole, A., Gill, F. (eds.), The Birds of North America, No. 702. The Birds of North America, Inc., Philadelphia, PA.

Leonard, Jr., D.L., Three-toed Woodpecker (*Picoides tridactylus*), 2001. In: Poole, A., Gill, F. (eds.), The Birds of North America, No. 588. The Birds of North America, Inc., Philadelphia, PA.

Mannan, R.W., Meslow, E.C., Wright, H.M., 1980. Use of snags by birds in Douglas-fir forests, western Oregon. Journal of Wildlife Management, 44, 787-797.

Martin, K., Aitken, K.E.H., Wiebe, K.L., 2004. Nest sites and nest webs for cavity-nesting communities in the interior British Columbia Canada: Nest characteristics and niche partitioning. The Condor, 106, 5-19.

Minitab Incorporated 2006. MINITAB Release version 14.1. State College, Pennsylvania.

Moore, W.S., Northern Flicker (*Colaptes auratus*), 1995. In: Poole, A., Gill, F. (eds.), The Birds of North America, No. 166. The Academy of Natural Sciences, Philadelphia, and the American Ornithologists' Union, Washington, D.C.

Moorman, C.E., Russell, K.R., Sabin, G.R., Guynn, Jr., D.C., 1999. Snag dynamics and cavity occurrence in the South Carolina Piedmont. Forest Ecology and Management, 118, 37-48.

Morrison, M.L., Raphael, M.G., 1993. Modeling the dynamics of snags. Ecological Applications, 3, 322-330.

Moroni, M.T., 2006. Disturbance history affects woody debris dynamics of Newfoundland balsam fir (*Abies balsamea*) stands. Canadian Journal of Forest Research, 36, 3194-3208.

Rajora, O.P., Mosseler, A., Major, J.E., 2002. Mating system and reproductive fitness traits of eastern white pine (Pinus strobus) in large, central versus small, isolated, marginal populations. Canadian Journal of Botany, 80, 1173-1184.

Raphael, M.G., White, M., 1984. Use of snags by cavity-nesting birds in the Sierra Nevada. Wildlife Monographs, 86.

Remm, J., Lohmus, A., Remm, K., 2006. Tree cavities in riverine forests: What determines their occurrence and use by hole-nesting passerines? Forest Ecology and Management, 221, 267-277.

Rowe, J.S., 1972. Forest Regions of Canada Publ. 1300, Canadian Department of the Environment, Canada Forest Service, Ottawa, 172p.

Runde, D.E., Capen, D.E., 1987. Characteristics of northern hardwood trees used by cavity nesting birds. Journal of Wildlife Management, 51, 217-223.

Russell, R.E., Saab, V.A., Dudley, J.G., Rotella, J.J., 2006. Snag longevity in relation to wildfire and postfire salvage logging. Forest Ecology and Management, 232, 179-187.

Saab, V.A., Dudley, J., Thompson, W.L., 2004. Factors influencing occupancy of nest cavities in recently burned forests. The Condor, 106, 20-36.

Schieck, J., Song, S.J., 2006. Changes in bird communities throughout succession following fire and harvest in boreal forests of western North America: literature review and meta-analyses. Canadian Journal of Forest Research, 36, 1299-1318.

Setterington, M.A., Thompson, I.D., Montevecchi, W.A., 2000. Woodpecker abundance and habitat use in mature balsam fir forests in Newfoundland. Journal of Wildlife Management, 64, 335-345.

Simon, N.P.P., Stratton, C.B., Forbes, G.J., Schwab, F.E., 2002. Similarity of small mammal abundance in post-fire and clearcut forests. Forest Ecology and Management, 165, 163-172.

Smith, C.Y., Moroni, M.T., Warkentin, I.G. submitted. Snag dynamics in post-harvest landscapes of western Newfoundland balsam fir-dominated boreal forest.

Snyder, J.E., 1984. Marten use of clear-cuts and residual forest stands in western Newfoundland. M.Sc. Thesis, University of Maine, Orono, ME. 31p.

Spiering, D.J., Knight, R.L., 2005. Snag density and use by cavity-nesting birds in managed stands of the Black Hills National Forest. Forest Ecology and Management, 214, 40-52.

Swallow, S.K., Gutierrez, R.J., Howard, Jr., R.A., 1986. Primary cavity-site selection by birds. Journal of Wildlife Management, 50, 576-583.

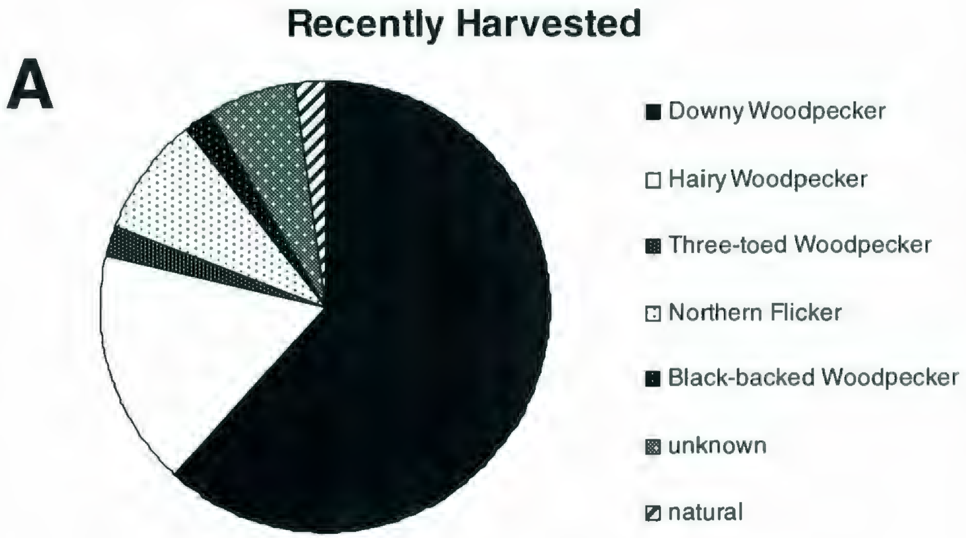
Thompson, I.D., H.A. Hogan, W.A. Montevecchi, 1999. Avian communities of mature balsam fir forests in Newfoundland: Age-dependence and implications for timber harvesting. The Condor, 101(2), 311-323.

Thompson, I.D., Larson, D.J., Montevecchi, W.A., 2003. Characterization of old “wet boreal” forests, with an example from balsam fir forests of western Newfoundland. Environmental Reviews, 11, S23-S46.

Walter, S.T., Maguire, C.C., 2005. Snags, cavity-nesting birds, and silvicultural treatments in western Oregon. Journal of Wildlife Management, 69, 1578-1591.

Whitaker. D.M., Montevecchi, W.A., 1999. Breeding bird assemblages inhabiting riparian buffer strips in Newfoundland, Canada. Journal of Wildlife Management, 63, 167-179.

Zack, S., George, L., Laundenslayer, W.F., Jr., 2002. Are there snags in the system? Comparing cavity use among nesting birds in “Snag-rich” and “Snag-poor” eastside pine forests. USDA Forest Service General Technical Report PSW-GTR-181. 279-191.


Recently Harvested
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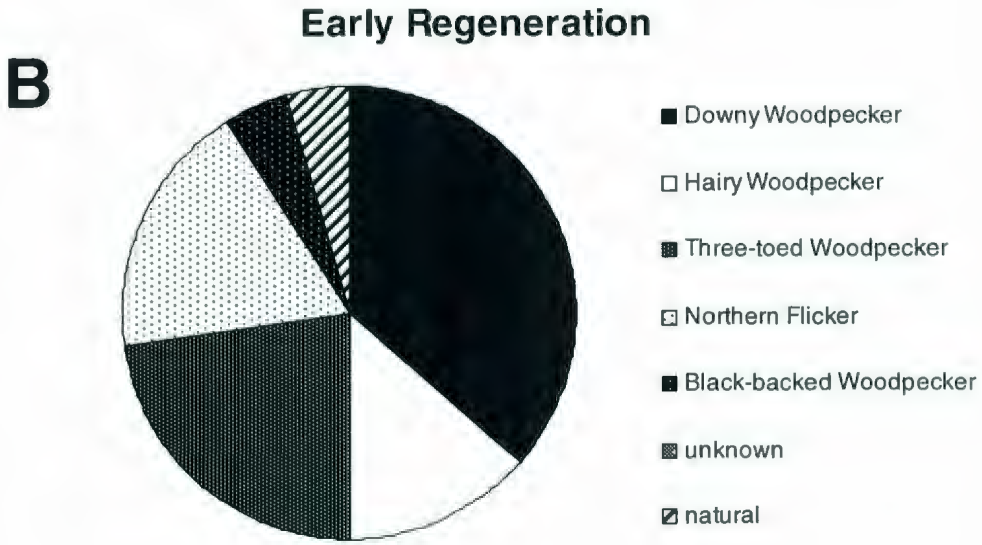
Downy Woodpecker
Hairy Woodpecker
Three-toed Woodpecker
Northern Flicker
Black-backed Woodpecker
unknown
natural


Early Regeneration
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Downy Woodpecker
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Northern Flicker
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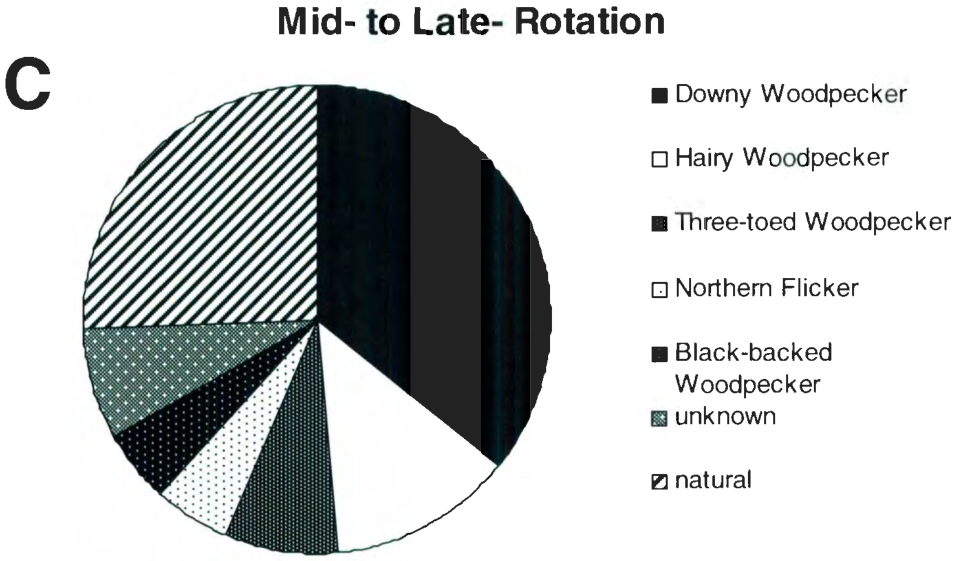

Figure 3.1. Breakdown of abundance of each species recorded in recently harvested (1–7 YSH), early regeneration (9–20 YSH) and mid- to late-rotation (21–100 YSH).

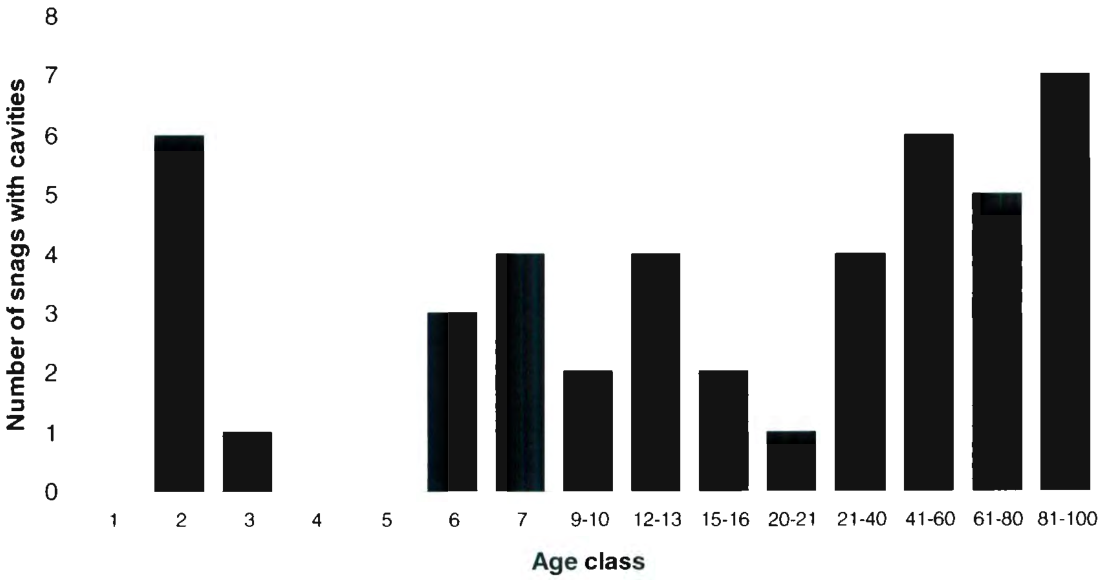

Figure 3.2. Total of snags with cavities found in each age class; total snags = 45 (note changing scale of x-axis).

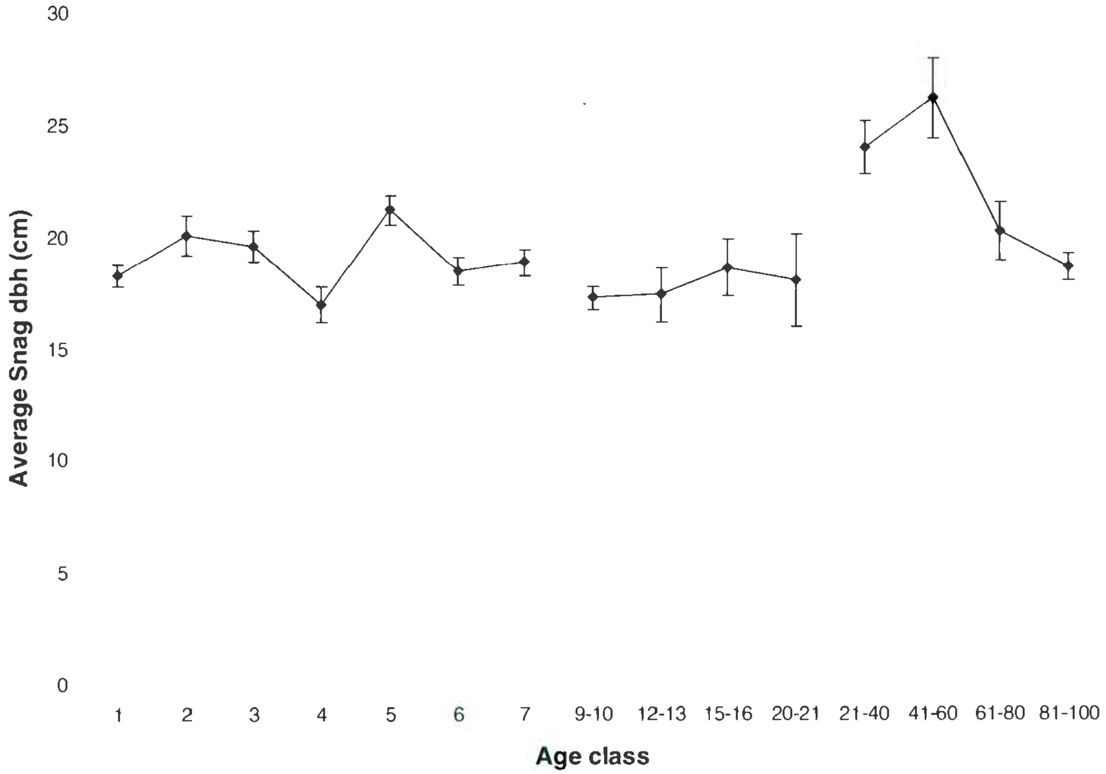

Figure 3.3. Average snag dbh by age class (mean ± SE; $n = 15$; note changing scale of x-axis).

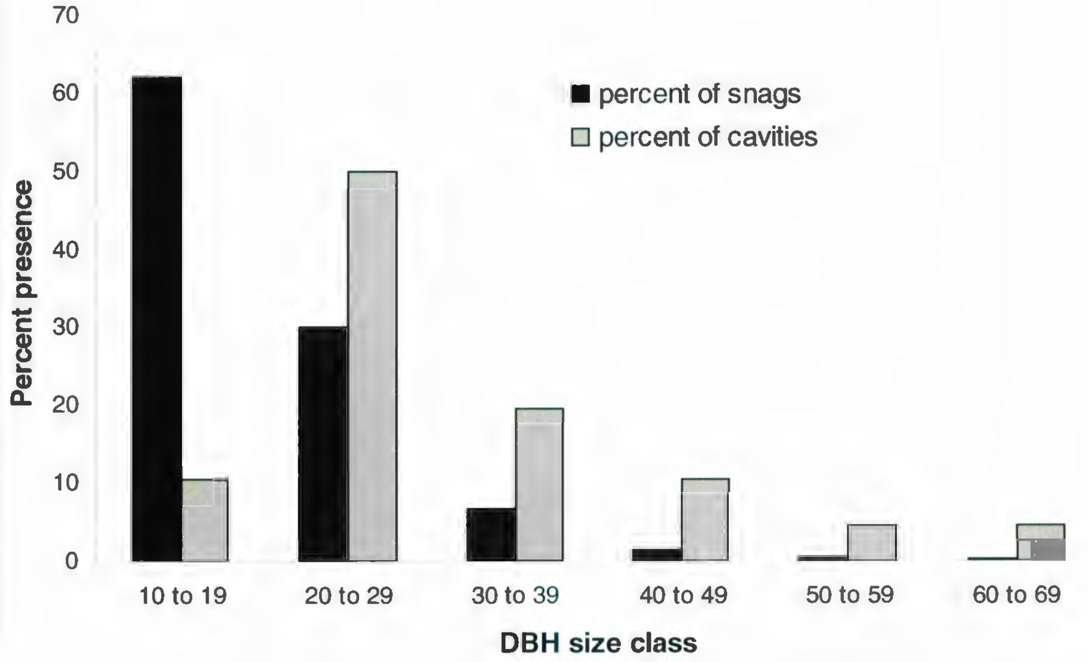

Figure 3.4. Percentages of overall snags and cavity snags by dbh size class.

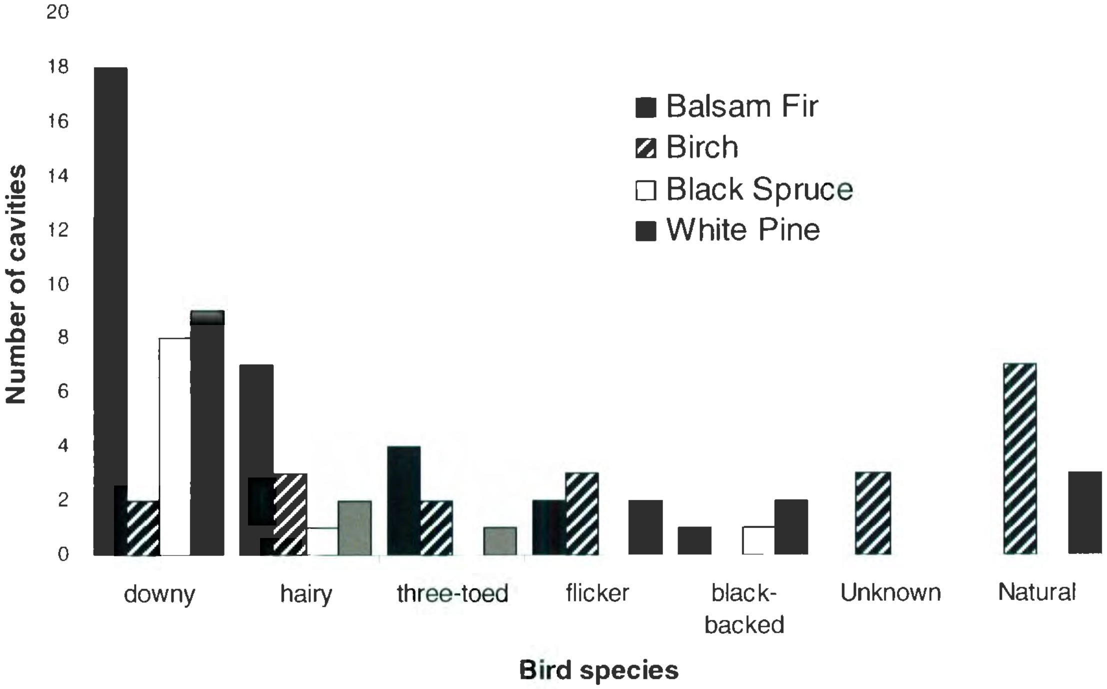

Figure 3.5. Totals of cavity snags of each tree species as used by each woodpecker species.

4.0 Summary

Snags in post-harvest landscapes of balsam-fir (*Abies* balsamea) dominated boreal forests in western Newfoundland are generally small in diameter and remain standing for only brief periods following their creation. Consequently, there are low snag densities for long periods of the forest rotation beginning shortly after harvest, with limited recruitment of new snags into the population. Consistent with other studies in similar systems (e.g., Cline et al. 1980, Everett et al. 1999, Garber et al. 2005, and Russell et al. 2006), I found that snags on the landscape remained standing for a maximum of ~10-15 years following harvest. Limited longevity of snags in western Newfoundland then leads to a long period of low snag density (<10 snags/ha) commencing mid-rotation, lasting until senescence increased snag numbers during late-rotation. Domestic harvesting and wind-throw further decreased snag density and availability immediately following harvest, to the detriment of wildlife populations dependent upon cavity-nesting habitat. In addition, many residual live trees are blown down after harvest, eliminating their potential to contribute to the snag population. Thus the small amount of snag recruitment from live trees left behind in cutovers does not maintain sufficient snag densities beyond 20 years since harvest (YSH) to meet the requirements of 10 snags/ha throughout the rotation (Cline et al. 1980).

I found that larger snags (graphic 2) were present on the landscape longer into the forest rotation. This was evident as the average snag diameter at breast height (dbh) increased when the average snag density decreased beyond 20 YSH. The average dbh of balsam fir is generally <30 cm (Thompson et al. 2003) so there are generally few large snags available to remain on the landscape. Over half (62%) of the snags observed in this study were <20 cm diameter whereas 90% of snags found to contain cavities were >20 cm diameter. While small snags are useful to cavity-nesters for foraging, the absence of large snags limits the presence of cavity-nesters on the landscape by reducing the availability of suitable nesting sites. With the harvesting of second-rotation forests occurring before senescence, there will be even fewer large snags available following harvest when compared to forests that are left to follow succession into senescence.

White birch (*Betula papyrifera*) trees are common to western Newfoundland and were frequently left standing following harvest. Larger diameter white birch snags contained a high proportion of snag biomass during periods of low snag density resulting in the large average dbh during these periods. Snag density and biomass in post-harvest landscapes would be increased by retaining more non-merchantable white birch post-harvest. White birch trees have high wood density and attain large individual volumes and diameters, resisting decay longer than smaller trees with softer wood. Thus birch snags were found to persist longer on the landscape than most of the balsam fir and other softwood snags. White birch snags supported cavity nesting species that utilize deciduous trees such as Downy (*Picoides pubescens*) and Hairy (*Picoides villosus*) Woodpeckers and Northern Flickers (*Colaptes auratus*) (Setterington et al. 2000). Northern Flickers were shown to use white birch for cavity excavation roughly in proportion to availability, thus increased white birch snag numbers would likely increase Northern Flicker habitat and numbers. Northern Flickers are considered keystone excavators (Martin et al. 2004) creating large cavities that provide habitat for a large variety of secondary cavity-nesting species.

I found that in western Newfoundland forest management practices produced post-harvest landscapes with low snag densities containing <2 snags per site, on average, with adequate dbh (>20 cm) and sufficient decay for cavity excavation. With current clearcut harvest practices and average tree size it is unlikely that >10 snags/ha could be maintained through the forest rotation until natural stand development generated increased numbers of snags. Retaining greater numbers of quality snags on post-harvest landscapes would narrow the gap in differences between natural disturbances and harvesting (Schieck and Song, 2006). By managing white birch to maximize snag recruitment and retain extant white birch snags, cavity-nester habitat availability and snag longevity could be improved both immediately after harvest and potentially beyond 20 years post-harvest. Bull and Partridge (1986) suggest that leaving live trees in clumps following harvest lessens the threat of wind-throw. They also found removing branches and tops of trees was the most effective method of snag creation, reducing windthrow and increasing initial rates of snag decay. Snags created as above were frequently used by cavity-nesters for foraging and nesting (Bull and Partridge 1986, Hallet et al. 2001). Improvement in the availability of large snags would enhance the potential for populations of large-cavity excavators such as Northern Flicker to increase, which would in turn expand habitat availability for secondary cavity-nesting species.

4.1 References

Bull, E.L., Partridge, A.D. 1986. Methods of killing trees for use by cavity nesters. Wildlife Society Bulletin, 14, 142–146.

Cline, S.P., Berg, A.B., Wight, H.M. 1980. Snag characteristics and dynamics in Douglas-fir forests, western Oregon. Journal of Wildlife Management, 44, 773–786.

Everett, R., Lehmkuhl, J., Schellhaas, R., Ohlson, P., Keenum, D., Riesterer, H., Spurbeck, D. 1999. Snag dynamics in a chronosequence of 26 wildfires on the east slope of the Cascade Range in Washington State, USA. International Journal of Wildland Fire, 9, 223-234.

Garber, S.M., Brown, J.P., Wilson, D.S., Maguire, D.A., Heath, L.S. 2005. Snag longevity under alternative silvicultural regimes in mixed-species forests of central Maine. Canadian Journal of Forest Research, 35, 787–796.

Hallet, J.G., Lopez, T., O'Connell, M.A., Borysewicz, M.A., 2001. Decay dynamics and avian use of artificially created snags. Northwest Science, 75, 378-386.

Martin, K., Aitken, K.E.H., Wiebe, K.L. 2004. Nest sites and nest webs for cavity-nesting communities in the interior British Columbia Canada: Nest characteristics and niche portioning. The Condor, 106, 5-19.

Russell, R.E., Saab, V.A., Dudley, J.G., Rotella, J.J. 2006. Snag longevity in relation to wildfire and postfire salvage logging. Forest Ecology and Management, 232, 179-187.

Schieck, J., Song, S.J. 2006. Changes in bird communities throughout succession following fire and harvest in boreal forests of western North America: literature review and meta-analyses. Canadian Journal of Forest Research, 36, 1299-1318.

Setterington, M.A., Thompson, I.D., Montevecchi, W.A. 2000. Woodpecker abundance and habitat use in mature balsam fir forests in Newfoundland. Journal of Wildlife Management, 64, 335-345.

Thompson, I.D., Larson, D.J., Montevecchi, W.A. 2003. Characterization of old “wet boreal” forests, with an example from balsam fir forests of western Newfoundland. Environmental Reviews, 11, S23-S46.

Graphic 1. Four year post-harvest study site near Pynn's Brook, Newfoundland.

Graphic 2. Twenty-one meter tall white pine (*Pinus strobus*) snag near Pynn's Brook, Newfoundland.