COMPARISON OF SMALL MAMMAL AND HERPETOFAUNA COMMUNITY COMPOSITION IN NATURALLY REGENERATED CLEAR-CUTS, PRE-COMMERCIALLY THINNED, AND SOFT-WOOD PLANTATION FORESTS, AT TWO DEVELOPMENTAL STAGES

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ABSTRACT

A viable softwood forestry industry relies on intensive forest management practices to optimize yield, and reduce rotation time. In New Brunswick, Canada, the primary management strategies are plantations followed by herbicide spraying, or naturally regenerated stands that are selectively pre-commercially thinned. Forest managers have both economic and conservation targets so it is critical to understand (1) how managed stands provide habitat value to native biodiversity relative to natural Acadian mixed-wood forest and (2) how the succession since the management intervention (i.e. time since clear-cutting, planting, thinning, etc.) affects habitat quality. This thesis addresses these questions by estimating and comparing small mammal and herpetofauna abundance and taxonomic richness in plantations, thinned, and naturally regenerated stands at two different developmental stages following clear-cutting. Stand characteristics within these treatment and stage categories were surveyed, in order to develop hypotheses about the mechanisms underlying relationships between stand treatment and development stage, and taxonomic richness or individual species abundances.

Abundance and taxonomic richness of small mammals, and forest dependent amphibians (wood frog and red-backed salamander), were negatively affected by intensive forest management practices, with plantations having a greater effect than pre-commercially thinned stands. Small mammal richness and abundance, and specifically abundances of *Sorex*, red-backed voles (*Myodes gapperi*), wood frogs (*Lithobates sylvatica*), and red-backed salamanders (*Plethodon cinerus*), were greater in naturally regenerated mixed-wood stands than in pre-commercially thinned and planted stands. Associations with stand characteristics were species specific, but deciduous components (i.e. canopy cover, leaf litter, percentage of hardwood) were important for species negatively affected by management practices. This is particularly true of canopy cover, which was greatest in naturally regenerated mixed-woods, and lowest in plantations. The relationship between thinning and native small mammal and herpetofauna species was relatively subtle and should be further studied to address the question of whether there are critical thresholds. Contrary to previous studies, no overall effect of forest management was found for herpetofauna taxonomic richness, woodland jumping mouse (*Napeozapus insignis*), short-tailed shrew (*Blarina brevicauda*), or deer mouse (*Peromyscus maniculatus*) abundances, however, this study may not have had the power to detect small effects. Both small mammals and herpetofauna were more abundant in the earlier development stage than the later, but taxonomic richness was similar between stand stages. This highlights the importance of measuring abundance and taxonomic richness, as particular species may be present but reduced in abundance in managed stands. Wood frog and red-backed salamander, two species that are more dependent on a terrestrial life stage than other amphibian species in the region, were positively associated with stand stage. This is likely due to increased canopy closure and resulting higher moisture microclimate levels at later developmental stages. Effective management for small mammal and herpetofauna

habitat will require the conservation of mixed-wood stands with a high amount of closed canopy cover.

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TABLE OF CONTENTS

LIST OF FIGURES

LIST OF TABLES

LIST OF SYMBOLS, NOMENCLATURE OR ABBREVIATIONS

PCT.. Pre-Commercial Thinning

IFM .... Intensive Forest Management

DNR.. Department of Natural Resources

DBH.. Diameter at Breast Height

INTRODUCTION

Forest managers are faced with the daunting task of maximizing timber production (quality and quantity) while maintaining wildlife habitat integrity. Forests provide valuable ecological, recreational, and aesthetic services, as well as being essential to the local economy (Nyland, 1996). Intensive forest management (IFM) practices have been an integral part of the New Brunswick's sustainable management plan for its forests, supported by a process of adaptive management (i.e. these practices are modified in response to ongoing ecological research.) Within the last couple of decades, increased public scrutiny of practices has challenged forest managers to employ management techniques that reduce impact on native biodiversity (Freedman et al., 1994).

Most of the research examining the effects of IFM has focused on differences in taxonomic richness and abundances between naturally regenerated and managed stands (Halpern and Spies, 1995; Aubry, 2000; Dieterich et al., 2006). A review of approximately 50 papers on the effect of IFM on vertebrates suggests that a loss of forest structure commonly impacts abundance and taxonomic richness, and that these impacts are potentially cumulative over the long-term (Thompson et al., 2003). There are still gaps in our understanding of how different forest management strategies (e.g. pre-commercial thinning and planting) affect ecosystem services provided by naturally regenerating forests. In particular, little is known about the ability of intensively managed forests to provide suitable habitat for forest organisms. An ongoing debate in

New Brunswick as to whether or not intensively managed forests provide suitable habitat for the range of species that are associated with naturally regenerated forests suggests that further research is needed, both at the stand and landscape scales (Betts et al., 2005). Specifically, Betts et al. (2005) suggest that forest managers in New Brunswick take a closer look at how older managed stands are used by native species, as well as possible threshold responses of native species to habitat characteristics potentially lost to intensive management strategies.

Forest types in New Brunswick

Native Acadian forest in New Brunswick

The Acadian forest biome extends from the north-eastern United States, to Quebec, and throughout the Maritime provinces of eastern Canada. It consists of both coniferous and deciduous tree species, variable in assemblage, size, and age (Parks Canada, 2009) but often characterized by the presence of red spruce (*Picea rubens*) and yellow birch (*Betula alleghaniensis*). The forest assemblage has been influenced by a long history of clearing for agriculture, logging and subsequent plantations, fire, and pest outbreaks in New Brunswick, most recently an eastern spruce budworm (*Choristoneura fumiferana*) eruption in the 1980's. These disturbances have resulted in the majority of forest stands being less than 200 years old (Loo et al., 2010 in McAlpine and Smith, 2010) with indigenous species that are adapted to take advantage of early successional forest stand conditions.

Natural factors influencing the re-establishment of forest stands following a disturbance include disturbance type, neighbouring forest stands, facultative species presence, and site conditions -- thus stands are variable in species composition as well as rate of establishment (Nyland, 1996; Roberts and Zhu, 2002). Abundant deciduous tree species within the Acadian forest include red maple (*Acer rubrum*), sugar maple (*A. saccharum*), northern red oak (*Quercus rubra*), American beech (*Fagus grandifolia*), white ash (*Fraxinus americana*), and white/paper (*Betula papyrifera*), yellow , and grey (*B. populifolia*) birch. Common coniferous tree species include red , black (*P. mariana*), and white spruce (*P. glauca*), eastern white (*Pinus strobus*) and red pine (*P. resinosa*), balsam fir (*Abies balsamea*), eastern hemlock (*Tsuga canadensis*), and eastern white cedar (*Thuja occidentalis*) (Schneider, 2012). Currently, the Acadian forest biome contains plant species of conservation concern that require prudent management in order to ensure their preservation -- the majority of these species are deciduous (Mosseler et al., 2003).

Commonly found small mammals native to the Acadian forest include the snowshoe hare (*Lepus americanus*), northern flying squirrel (*Glaucomys sabrinus*), American red squirrel (*Tamiasciurus hudsonicus*), eastern chipmunk (*Tamias striatus*), woodland jumping mouse (*Napaeozapus insignis*), meadow jumping mouse (*Zapus hudsonius*), deer mouse (*Peromyscus maniculatus*), southern bog lemming (*Synaptomys cooperi*), southern red-backed vole (*Myodes gapperi*), star nosed mole (*Condylura cristata*), northern short-tailed shrew (*Blarina brevicauda*), masked shrew (*Sorex cinereus*), water shrew (*S. palustris*), smoky shrew (*S. fumeus*), pygmy shrew (*S.

hoyi), and less commonly, the maritime shrew (*S. maritimensis*), and long tailed shrew (*S. dispar*) (Forbes et al., 2010 in McAlpine and Smith, 2010). Herpetofauna species that have a dependence on permanent or temporary aquatic environments within the major forest region in New Brunswick include yellow spotted salamander (*Ambystoma maculatum*), blue spotted salamander (*A. laterale*), eastern newt (*Notophthalmus viridescens*), American toad (*Anaxyrus americanus*), spring peeper (*Pseudacris crucifer*), green frog (*Lithobates clamitans*), pickerel frog (*L. palustris*), bullfrog (*L. catesbeiana*), and wood frog (*L. sylvatica*). Terrestrial herpetofauna include the northern red-backed salamander (*Plethodon cinereus*), Maritime garter snake (*Thamnophis sirtalis pallidulus*), smooth green snake (*Liochlorophis vernalis*), Northern ring-necked snake (*Diadophis punctatus edwarsii*) and Northern red-bellied snake (*Storeria occipitomaculata occipitomaculata*) (Adams and Freedman, 1999; McAlpine, 2010 in McAlpine and Smith, 2010).

Managed forests in New Brunswick

Balsam fir, black, white, red, and Norway (*P. abies*)(a non-native species) spruce, eastern white cedar, white, red, and jack (*P. banksiana*) pine are selected for, and intensively managed by the softwood lumber and pulp and paper industries (J.D. Irving Ltd, 2007). IFM treatments employed by forest managers to obtain the desired species and stand densities include herbicide release, pre-commercial mechanical thinning, and planting (J.D. Irving Ltd, 2007). Pressure from the forestry industry has resulted in an increase in allowable cutting of Crown lands by 20 percent in 2014, with the license being renewable every 5 years. In order to achieve this quota, the provincial

Department of Natural Resources (DNR) has reduced the amount of Crown land that is protected (i.e. protected natural areas, unique sites, deer winter areas, old forest wildlife habitat, riparian buffers, wetland buffers, vegetative communities, steep slopes, inoperable/inaccessible areas, sugar and camp leases) from forestry operations from 31 to 23 percent (DNR, 2005; J.D. Irving Ltd., 2007). This increase in managed forest allocation has been met with fierce criticism by scientists, forest enthusiasts, conservationists, and some of the general public who maintain that a percentage closer to 30 percent of naturally regenerated stands is integral to maintaining native biodiversity (CCNB, 2015).

IFM stands are characterized as having a less complex understory stratum of deciduous vegetation (Roberts and Zhu, 2002), with a more uniform distribution of trees, and a canopy cover that develops more slowly than in naturally regenerated stands (Wuest and Betts, 2010). Currently in New Brunswick managed forests fall predominantly into two types post clear-cutting/harvest: those that regenerate naturally with a commercially desirable species by seed trees (i.e. desired species and phenotype left after cutting, as seedlings that were present in the understory, or in neighbouring stands and then subsequently pre-commercially thinned, PCT), and those that need to be planted in order to regenerate the desired species (plantations). Plantations are clear-cut, planted and then treated with herbicide (i.e. chemically thinned) to reduce competition from deciduous species, while PCT stands are naturally regenerated managed forests that are mechanically thinned to reduce competition between coniferous species as well as deciduous species (J.D. Irving, 2007).

There is concern amongst some of the general public and scientists that further reduction of the natural Acadian forest will not be adequate to sustain non-commercial attributes of New Brunswick forests (Nature NB, 2014). Because it is impossible to track the responses of all components of biodiversity, it is useful to identify indicators (i.e. characteristics that are correlated with other measures of overall biodiversity). Small mammals and herpetofauna are useful as indicator species in studying the long-term effect of IFM on native biodiversity because they are easily captured in large numbers on site, they have relatively short life spans and small migration ranges, and they are often both predators and prey in local food webs (Thompson et al., 2002).

Effects of forest management techniques on native Acadian forest flora and fauna.

Clear-cutting

Clear-cutting is a commonly used forest harvesting practice in New Brunswick, where all trees are removed and the site is subsequently planted, or the majority of the trees are cut but select seed trees are maintained to repopulate the site. Trees are harvested using tracked machinery, and removed with large trucks via a network of constructed dirt roads. Clear-cutting has been found to increase light penetration to the ground as a result of canopy loss (Etcheverry et al., 2005), decrease soil moisture and nutrients (Bormann et al., 1968; Carey and Johnson, 1995) and subsequently results in a loss or slow recovery of the understory of angiosperms (Duffy and Meier, 1992; Roberts and Zhu, 2004), and bryophytes associated with wooded moist habitats

(Ross-Davis and Frego, 2002; Fenton et al., 2003). A shift to shade intolerant, early successional, herbaceous species such as fireweed (*Chamerion angustifolium*), flat-topped white aster (*Doellingeria umbellata*), hay-scented fern (*Dennstaedtia punctilobula*), red raspberry (*Rubus idaeus*), common goldenrod (*Solidago canadensis*), swamp thistle (*Cirsium muticum*), and woolly grass bulrush (*Scirpus eriophorum*) (Freedman et al., 1993) are common characteristics of recently clear-cut stands within the Acadian forest. In the years following clear-cutting during which the canopy is open, predatory and insectivorous birds, large herbivores, pond-breeding amphibians, and insectivorous mammals prosper as a result of associations with shade intolerant vegetation and insects (Johnson and Freedman, 2003).

The direct effect of clear-cutting on herpetofauna has been found to be generally negative, mostly attributed to disruption of breeding ponds, death during the process of harvesting, decreased soil moisture and leaf litter, and increased temperature (Pough et al., 1987; Mitchell et al., 1997; Grialou et al., 2000). Small mammals tend to react in a species-specific manner with many species relatively un-impacted by clear-cutting (Kirkland, 1990; Etcheverry et al., 2005). When a decrease in small mammal populations in response to clear-cutting has been detected, it has been attributed to loss of predator protection, and lack of adequate nesting sites as a result of the removal of snags and canopy cover (Fuller and DeStefano, 2003). Suggestions have been made that the retention of mature forest attributes (i.e. coarse woody debris, snags, riparian buffer strips, etc.) within a clear-cut stand may be beneficial to small mammals, amphibians, and associated predators, and warrant further research

(DeMaynadier and Hunter, 1995; Dupuis et al., 1995; Cole et al., 1998; Wilson and Carey, 2000; Bowman et al., 2001; Fuller et al., 2004). Furthermore, certain species appear to be more sensitive to clear-cutting than others. For example, DeMaynadier and Hunter (1995) conducted a meta-analysis of studies examining the effects of clear-cutting on amphibians and noted that anurans appeared to be much more tolerant to increases in temperature than salamanders.

Long-term effects of clear-cutting

Previous studies examining the long-term effects of clear-cutting (10+ years) indicate that herpetofauna species' abundances were at least two times higher in older controls than in more recently clear-cut stands (DeMaynadier and Hunter, 1995), that abundance increases with age (i.e. time since disturbance), reaching full recovery at approximately 30-60 years in the Acadian and neighbouring boreal and deciduous forest regions of Eastern North America (Pough et al., 1987; Bonin, 1991; DeGraaf et al., 1992; Petranka et al., 1993; Petranka et al., 1994). Pough et al. (1987) compared several older managed and naturally regenerated stands that had originated by clear-cut with old growth forests and found that older (~25 years) naturally regenerated stands supported salamander abundances comparative to old growth, whereas the older (25 years) managed stands did not. This difference between how older managed stands and naturally regenerated stands recover as habitat for native species is attributed to site preparation activities such as herbicide application, burning, and scarification that are known to set back re-colonization time for certain species (DeMaynadier and Hunter, 1995; Nyland, 1996; Roberts et al.,2001; Thompson et al., 2003; Roberts and Ramovs, 2005). A critical research gap exists with respect to how managed stands that reach an over mature stage provide habitat relative to the naturally regenerated native habitat stands of the same development stage.

Herbicides

Herbicides, primarily glyphosate-based, are applied to reduce competition for sunlight, soil nutrients, space etc. from secondary succession deciduous vegetation, both persistent shrubs and tree seedlings and colonizing pioneers, following clear-cutting. Reduced competition improves survival, growth, and quality of the merchantable timber species (Freedman et al., 1993; Nyland, 1996). Following the application of herbicide, stands have shown changes in ground microclimate attributed to a reduction in leaf litter, shrub vegetation, and over-story vegetation (Sullivan et al., 1998; Prezio et al., 1999). A temporary reduction in small mammal abundance within two years of herbicide application has been attributed to habitat loss rather than to herbicide toxicity (Anthony and Morrison, 1985; Sullivan et al., 1998; Gagné et al., 1999; Sullivan and Sullivan, 2003; Fuller et al., 2004). In a study examining the effect of herbicides on small mammals 10 years following application, Sullivan et al. (1997) did not find any long-term effects on taxonomic richness or diversity. Common shrew species (masked, pygmy, smoky, short-tailed) have been found to be relatively unaffected by herbicide treatments (Lautenschlager et al., 1997) and to thrive in partially harvested mixed stands (Fuller et al., 2004) possibly because they are opportunistic insectivores and their prey is likely more common in open areas.

Direct toxicity of glyphosate-based herbicide spraying at amounts used for forest management has not been found in terrestrial or larval stages of pond-breeding herpetofauna (Cole et al., 1998; Harpole and Haas, 1999; Thompson et al., 2004; Edge et al., 2011; Gertzog et al., 2011). In the short term, terrestrial salamanders are able to find cover following herbicide application even though there was a loss of leaf litter (Harpole and Haas, 1999). In a study examining longer term (i.e. 7-13 years) effects, Homyack and Haas (2009) found that a resulting loss of canopy cover had a negative effect on salamander abundance. Further negative long-term effects of herbicides on terrestrial herpetofauna are attributed to indirect effects of loss of leaf litter, canopy cover, and stand structure (Cole et al., 1998).

Mechanical thinning

Pre-commercial thinning (PCT), which is achieved by selectively cutting deciduous trees and tightly spaced coniferous trees using saws, and leaving the slash on-site (Nyland, 1996; Ransome and Sullivan, 2002), is another common technique for encouraging growth of target species by removing competition. PCT is most commonly performed following the sapling stage (approx. 8-20 years) and before trees are large enough to warrant commercial harvest. Thinned stands have some characteristic structural components of a naturally regenerating forest (i.e. mature trees, snags, coarse woody debris, herbaceous cover, partial or complete canopy cover) but have reduced levels of other structural components (i.e. tree density, shrub cover, tree species diversity, multiple canopy layers) (McComb et al., 1993; Wilson and Carey, 2000; Etcheverry et al., 2005; MacCracken, 2005). A study in New Brunswick, Canada found that negative effects of thinning on native understory plant diversity was mitigated after fifteen years, suggesting that once a complete canopy cover is established there are little differences between naturally regenerated and thinned stand plant understories (Cole et al., 2008).

Small mammal response to PCT treatment is conflicting, and appears to be species specific (Etcheverry et al., 2005; Fuller et al., 2004; Homyack, 2005; Henderson, 2011; Dracup et al., 2015). Deer mice have been shown to be relatively unaffected by thinning (Homyack, 2005; Dracup et al., 2015), whereas there are conflicting reports on red-backed vole response to PCT. Etcheverry et al. (2005) and Henderson (2005) found red-backed voles to be less abundant in pre-commercially thinned stands at 5, 10, and 20 years after treatment, whereas other studies in Western and Eastern North America have found no effect of PCT on red-backed voles (Sullivan et al., 2001; Homyack et al., 2005) . A study by Henderson (2005) within the same forest region found that structures which provide cover, most specifically an adequate volume of coarse woody debris, can and should be maintained in managed stands as a potential limiting factor for small mammals such as red-backed voles. A study conducted 6-16 years post thinning suggests that thinning may accelerate the development of stand characteristics prevalent in mature stands (i.e. larger trees, uneven canopy cover) (Homyack et al., 2005) and therefore could be beneficial to species known to be associated with older stands, such as deer mice and red-backed voles (Lautenschlager et al., 1997). Coarse woody debris has been identified as an integral requirement for red-backed voles, and is maintained and sometimes increased following thinning thereby creating a beneficial habitat (Bowman et al., 2000; Dracup et al., 2015). These conflicting results may be explained by the fact that a tolerance for thinning is greatest when habitat requirements such a food sources, predator avoidance, and nesting sites remain intact following thinning treatments (Lautenschlager et al., 1997; Gagné et al., 1999; Wilson and Carey, 2000; Carey and Wilson, 2001; Fuller et al., 2004). For instance, a study by Dracup et al. (2015) found that stands that retained coarse woody debris following thinning as opposed to those stands where debris was removed, supported twice as many red-backed voles.

Short-term amphibian response to thinning has generally been negative and attributed to reduction of aquatic habitat and/or soil moisture (Pough et al., 1987; Messere and Ducey, 1998; Grialou, 2000; MacCracken, 2005). Grialou et al. (2000) found that the short term effect of thinning on terrestrial red-backed salamanders was negative, as a result of a short term reduction in canopy cover and subsequent changes in soil moisture. Brooks (1999) also concluded that conserving canopy cover can eliminate the negative effect of thinning on salamanders. The long-term effects of thinning on herpetofauna have not been well documented but a study by Karraker and Welsh (2006) did not detect an effect of thinning on abundance of amphibians 10 years after thinning had occurred.

Plantations

Planting trees is an IFM technique intended to replace the species that are likely to regenerate in that stand with a more commercially desirable species or combination of tree species. In the Acadian forest region, plantations produce trees destined for the pulp and paper industry which, depending on the site, are a combination of black spruce, Norway spruce, white spruce, red spruce, white pine, jack pine, red pine, and eastern white cedar (J.D. Irving, Ltd., 2007). Tree planting is preceded by clear-cutting and site preparation (i.e. scarification or burning), and is often followed by chemical and/or mechanical thinning: In comparison with naturally regenerated mixed-wood forests, coniferous plantations have fewer snags, leaf litter, and coarse woody debris, and have a less complex canopy cover and shrub cover because stands are evenly aged. This results in fewer native species that depend on structure complexity (DeMaynadier and Hunter, 1998; Waldick et al., 1999, Bowman et al., 2001; Johnson and Freedman, 2003; Thompson et al., 2003; Pearce and Venier, 2005).

There is convincing evidence that plantations support fewer and different bird (Thompson et al., 2003; Woodley et al., 2006), vascular plant (Ramovs and Roberts, 2005) and bryophyte species (Ross-Davis and Frego, 2002; Thompson et al., 2003) than naturally regenerated mixed-wood forests, but there is conflicting evidence about the effects of plantations on amphibian and mammal diversity (Mitchell et al., 1995; Waldick et al., 1999; Thompson et al., 2003). Furthermore, there are relatively few studies that explore the relationship between small mammal and herpetofauna and older (native species) plantations that replace mixed-wood stands. Bowman et al. (2001) found that small mammals were as abundant in New Brunswick plantations as in naturally regenerated mixed-wood stands, with the exception of red-backed voles, the lower abundance of which was attributed to lack of coarse woody debris in plantations. A study by Atkeson and Johnson (1979) found that a fifteen year old pine plantation in

Georgia, USA supported fewer small mammals than younger plantations. They proposed that older plantations support fewer individuals because they have increased canopy cover which results in a subsequent reduction in shade intolerant shrub cover- thought to be important to small mammals. However, no comparison was made to the native mixed-wood stands. A review by DeMaynadier and Hunter (1995) and a study by Pough et al. (1987) found that plantations support fewer amphibians than naturally regenerated mixed-wood forests, which they suggest could be a result of a reduced litter and coarse woody debris in plantations- characteristics that have been found to be positively associated with amphibians - and increased soil acidity. A New Brunswick study by Waldick et al. (1999) found that terrestrial amphibians (i.e. yellow spotted and red-backed salamanders, spring peeper, and wood frog) are particularly sensitive to the conversion of mixed-wood stands to plantations because it results in reduced hardwood debris.

Objective

My objective is to estimate the relative conservation value of intensively managed forests as habitat for species of small mammals and herpetofauna that are native to the Acadian forest. The objective of this study was to compare small mammal and herpetofauna taxonomic richness and abundance in plantations and pre-commercially thinned stands, relative to naturally regenerated stands of the same developmental stage in the Acadian forest. I also aimed to determine whether effects of forest management practices on small mammals and herpetofauna are mitigated as forests mature, by comparing plantations, pre-commercially thinned, and naturally regenerated stands at two developmental stages, the first stage having an average softwood tree height of 3-5 metres and between 10 and 26 years old, the second stage having an average tree height of 10-20 metres and between 17 and 78 years old (age variability dictated by the organic stand establishment process for naturally regenerated stands). I correlated data on small mammals and herpetofauna with stand structure characteristics in order to identify potentially critical stand characteristics. The results are designed to provide information to forest managers who are tasked with finding a balance between intensive silviculture practices (IFM) intended to increase forest productivity, and maintaining ecological integrity for future sustainability of indigenous species.

METHODS

Study area

This study was conducted in the Cole's Island area of central New Brunswick, Canada within latitudes of 45° 38' and 46° 10', and longitudes of 65° 11' and 65° 45' (Figure 1) in 2005 and 2006, between June and September. Annual precipitation is approximately 1000-1300mm, with an average summer temperature of 19.1^{0}C (Government of Canada: Climate, 2015). The study area falls within the Eastern Lowlands ecoregion, Castaway ecodistrict, characterized by a flat geology with valley slopes composed of carboniferous sedimentary sandstone, clay loam, with intermittent volcanic deposits (ref). Soils are relatively acidic, with poor drainage conducive to formation of wetlands and vernal pools (DNR, 2005; GNB, 2011). The landscape is a mosaic of managed and naturally regenerated Crown land composed of wetland tolerant mixed-wood species, with little residential development.

Stand selection

Stands were separated into three categories of forest management treatments (naturally regenerated, pre-commercially thinned, and plantation) and two successional stages of stand development (S1 and S2). Twenty-four stands, 4 of each treatment combination (i.e. naturally regenerated-S1, naturally regenerated-S2, thinned-S1, thinned-S2, plantation-S1, plantation-S2) were sampled during the summers of 2005 and 2006 (June-September). Stands ranged in size from 7-40 hectares, were accessible by vehicle, originated by clear-cut, and contained no large lakes or wetlands (i.e. nothing present on a 1:10,000 metre map)(Table A1). Some stands contained secondary logging roads. All stands were spatially separated by 1.4 kilometres or more (Figure 1), and were predominantly surrounded by either mid-rotation or mature (~30 years post clear-cut) stands of hardwood or softwood trees (Appendix table 2).

Stand development stages were categorized into stage 1 and stage 2 categories based on a combination of time since clear-cutting/thinning/planting, and average softwood tree height. This was done to ensure comparability of both stand age and stage of forest succession. Time since clearing/thinning/planting was extracted from JD Irving Geographical Information System (GIS) inventory data base and map layers, and average stand tree height was determined by measuring at least 10 haphazardly chosen trees (many more when there was variability) on site by the tangent method (Minnesota Department of Natural Resources, 2015) with a Suunto clinometer and a measuring tape.

Treatments

Naturally regenerated stands

Stands that were clear-cut and naturally regenerated without any subsequent management treatment (hereafter referred to as 'naturally regenerated') (n=8) were composed of native Acadian forest mixed-wood species representative of the Eastern Lowlands eco Region, that colonized the site naturally following clear-cutting. Stage 1 stands, representative of a 'stand initiation stage' (Oliver and Larson, 1996), had an average height of measured softwood trees between 3 and 5 metres and were clear-cut 10-26 years prior to the study. Stage 2 stands, representative of a 'stem exclusion stage' (Oliver and Larson, 1996), had an average height of measured softwood trees between 10 and 20 metres, and were clear-cut between 17 and 78 years prior to the study (Appendix table 1).

PCT stands

Pre-commercially thinned stands (n=8) regenerated by natural processes with a predominantly coniferous tree inventory as a result of intentionally retained seed trees or coniferous neighbouring stands. Mechanical thinning of competing hardwood trees and/or high density coniferous trees is conducted approximately 15 years post-harvest. Stage 1 stands were thinned 5 years prior to the study with measured softwood trees being between 3 and 5 metres in height at the time of study, while Stage 2 stands were thinned between 10 and 12 years prior to the study, with measured softwood trees being within a range of 10-20 metres in height. (Appendix table 1).

Plantations

Plantation stands (n=8) were planted with the desired harvest coniferous species because conditions were not favourable for the clear-cut stands to regenerate merchantable coniferous trees through natural processes. Sites that had been partially naturally colonized and fill planted were also considered plantations. Aerial herbicide application was done within 5 years of planting to reduce competition with shade intolerant hardwood species. Stage 1 plantations were established 10-12 years prior to the study, with measured softwood trees within a range of 3-5 metres in height, and Stage 2 plantations were established 21-23 years prior to the study with measured softwood trees within a range of 10-20 meters in height (Appendix table 1).

Small mammal and herpetofauna sampling

Both small mammals and herpetofauna were sampled in 2005 and 2005 from June to August using drift fence/ pit fall arrays for seven consecutive nights (4032 total trap nights, ~168 trap nights per stand). In addition, small mammals were sampled for seven consecutive nights using live traps (4368 total trap nights, ~182 trap nights per stand), and herpetofauna were sampled using natural cover searches (4320 objects, ~180 cover objects per stand) (Table 1).

Two sampling stations were established in each of the twenty four stands, with each fence greater than 30 metres from the stand edge. A drift fence and pit fall trap was constructed of silt fence fabric buried in an X formation (each arm of the fence was 7.62 metres) an average of fifteen centimetres into the ground. Ten litre buckets, punctured for drainage, were buried at the ends and centre of the silt fencing with the lip flush to the ground, and sponges in the buckets to retain moisture or to act as a raft if water accumulated (Figure 2). At each sampling station, twelve pitfall buckets and thirteen live traps were opened for seven consecutive nights. On twelve occasions, pit fall traps either filled with water or lifted above the ground due to excess water; these were not counted.

In the area of one of the two drift fence/pitfall traps, eighteen 23 L x 7.6W x 9W cm H.B. Sherman live traps were established in two 3 x 3 arrays with ~ 75 centimetres between traps. In addition, eight 50L x 15W x 15H cm Tomahawk live traps were established in two 2x2 arrays with ~1.5 metres between traps. The trap arrays were placed ~30 metres from the "crooks" of the drift fence array (Figure 2). Both Sherman and Tomahawk traps were baited using a mixture of seeds commonly used in commercial small mammal feed. Traps were also filled with natural cotton for comfort. Most animals were captured at night, and thus traps were checked early in the morning to minimize the length of time in the trap and to reduce stress. Traps were opened for 7 consecutive nights; closed empty live traps were given a value of 0.5 when calculating trap nights to account for the fact that we did not know when the trap was closed (Beauvais and Buskirk, 1999). Once trapped, mammals were identified via morphological characteristics to either genus (*Sorex* spp.) or species, marked with an individually numbered ear tag (except for shrews which were marked with nail polish), and released.

Six 5-minute natural cover searches were conducted in each stand for a total of half an hour per stand in September of 2005 and 2006. Natural objects were defined as any potential cover object (e.g. rocks, stumps, or decomposing logs) at least 20 by 25 cm in size. Decomposing logs or stumps were torn open, searched for salamanders, and then replaced to reduce habitat disturbance. Herpetofauna were identified to species and released.

For each species and for abundances of small mammals and amphibians an index of abundance was expressed as capture per 100 corrected trap-nights (Beauvais and Buskirk, 1999) by dividing the number of captures x 100 by the corrected (0.5 value for sprung traps) trap-nights. All taxa caught, irrespective of abundance, were used to calculate taxonomic richness. In addition, analyses examining treatment and/or developmental stage effects on total herpetofauna or mammal abundance, used captures across all taxa. However, analyses examining treatment and/or developmental stage effects on the abundance of individual taxa were only carried out for taxa with catch numbers > 20 individuals over both sampling years.

All individuals from the genus *Sorex* were treated as a single taxon because I was not confident in my ability to distinguish between species. Because small mammals were captured live and my intention was live release, I was not able to measure definitive species characteristics such as tooth and skull patterns (van Zyll de Jong, 1983; Churchfield, 1990), necessary for distinguishing species of *Sorex* shrews. Furthermore, because shrew species were producing cohorts at different times, both adults and juveniles were captured throughout the sampling period and size differences among species were not useful. Often, morphological differences such as colour were also difficult to distinguish, especially when shrews were wet. Examination of a subsample of individuals who did not survive the trapping using tooth and skull patterns indicated that I was only able to correctly identify 75% of shrews to species. I have therefore chosen to use the term “taxonomic richness” (rather than taxonomic richness) to describe small mammal and herpetofauna diversity among forest stands.

However, since the short-tailed shrew was readily distinguishable, captures of this species were analyzed separately from other shrew species. All terrestrial species of shrews present in New Brunswick are potentially sympatric within the Acadian forest and have varied, but generalist, diets of mainly of fungi and invertebrates (Whittaker and French, 1983; Mitchell et al., 1997; McCay et al., 2004).

Stand characteristic measurements

Canopy cover, leaf litter, needle litter, coarse woody debris cover, shrub cover, moss cover, percent hardwood, and density were measured at each of the sites. Depths to water table (i.e. depth at which soil is 100% saturated with water) were obtained from digital maps supplied by J.D. Irving Ltd. Areas (m^3) within the stand that had a depth to water table (i.e. distance from the surface to the water table underground) between 0 and 0.1 metres were classified as 'wet' (i.e. more likely to have seasonal vernal pools and damp soil), and areas that had a depth to water table between 0.5 and 10 metres were classified as 'dry'. Sampling of all but canopy cover, percent hardwood and tree density were done using 1 m^2 quadrats (n=40). Five quadrats were placed in a straight line at each of the four ends of the pitfall trap arrays with approximately 1 metre between quadrats (Figure 2). Leaf litter, needle litter, coarse woody debris, shrub cover, and moss cover were assigned to one of 6 categories in each quadrat: None = 0, >0-20% coverage = 1, 20-40% coverage=2, 40-60% coverage=3, 60-80% coverage=4, and 80-100% cover=5. The values in all 40 quadrats (i.e. 20 quadrats at each of the 2 sampling stations) were then averaged to provide a stand-level value. On-site data were collected in the late summers of 2005 and 2006, when leaves were still on the trees and leaf litter was a result of the previous year's abscission. Canopy cover, stand density, and percentage hardwood trees were sampled at both drift fence/pitfall trap areas outside a 15m radius from each drift fence. This was done to avoid bias, as areas of lower tree density areas were selected for drift fence installation. Canopy cover was measured using a spherical densitometer (n=32) at 4 different locations at each of the ends of the stand density transects for both fences (Figure 3). Tree density (trunks per hectare) was estimated as counts using a point-quarter centre method (Mitchell, 2010). Percentage of hardwood trees was determined by counting all trees (both softwood and hardwood) that fall on the circumference of the tree density transect, dividing the number of hardwood trees by the total of trees (Figure 3). Only trees $\geq$ 10cm diameter at breast height (DBH) were included in calculations of density and percentage of hardwood.

Stand information gathered by aerial photograph interpolation and historical geographic information systems (GIS) data (e.g. percentage of canopy cover, overall stand type, stand maturity stage) available from J.D. Irving Ltd. (land managers) was gathered in order to compare my collected stand characteristics with what is commonly used by land managers (Appendix Table 1).

Statistical analyses

Small mammal, herpetofauna, and stand characteristic versus stand management and stage

Randomization tests were used to test the null hypothesis that small mammal, herpetofauna, and stand characteristic (leaf litter, needle litter, coarse woody debris, shrub cover, moss cover, canopy cover, percentage of hardwood, density, area with depth to water table at 0-0.1 metres, and area with depth to water table at 0.5-10metres) did not differ with stand treatment types (i.e. plantations, pre-commercially thinned stands, and naturally regenerated controls) or stages (i.e., S1 and S2). A randomization approach was taken because the data did not always meet the assumptions (i.e. homogeneous variance and normally distributed error) of parametric tests. Post hoc pair-wise comparisons, and stand stage and treatment interactions were also done using randomizations. I conducted post hoc power analyses using the program *G*Power 3.1* (Faul, 2009), which was especially relevant in determining the chances of a type II error for taxa that had lower catch numbers. I estimated power for 4 different effect sizes: very low effect (f=0.1), low effect (f=0.25), moderate effect (f=0.4), and large effect (f=0.6) (Appendix figure 1). Alpha was set equal to 0.10 rather than the traditional 0.05 to minimize the probability of Type II errors. Power analyses should be treated as approximate because they were estimated based on parametric tests (t-tests and ANOVA rather than randomizations).

Small mammal and herpetofauna versus stand characteristics

Simple linear regressions were used to model relationships between stand characteristics and small mammals and herpetofauna taxonomic richness and abundances.

RESULTS

Small mammal and herpetofauna trapping results

Eleven small mammal taxa (including pooled *Sorex* spp.) and eight herpetofauna species were captured (Tables 2,3). Shrews were the most commonly captured small mammal, followed by red-backed voles, woodland jumping mice, and deer mice. I do not have reason to believe that woodland jumping mice were able to jump out of the pit fall traps. Fewer herpetofauna than small mammals were captured, with the most abundant herpetofauna captured being the southern red-backed salamander.

Pit fall traps were the most successful trapping method for both small mammals and herpetofauna (Table 1). Herpetofauna were never caught in live traps, while Tomahawk traps captured only larger small mammals (i.e. snowshoe hare, northern flying squirrel, and red squirrel). Short-tailed shrew (0 of 26 individuals), meadow jumping mouse (0 of 9 individuals), and *Sorex* spp. (6 of 609 individuals) were rarely, if ever captured in Sherman traps.

There was a high instance of shrew capture in the pit fall traps. This may have greatly reduced the recorded capture numbers of herpetofauna as there was evidence of shrews having eaten other individuals. Carnivorous shrews are known to have a very high metabolic rate and commonly eat amphibians and other small mammals (Churchfield, 1990), there was evidence that shrew predation did take place. Nevertheless, pit fall arrays were the most successful method of capturing herpetofauna with the exception of red-backed salamanders. This wholly terrestrial species was most readily caught via quadrat natural cover searches, as was also noted in previous studies (Adams and Freedman, 1999; Smith and Petranka, 2000).

Because of low capture rates, snowshoe hare, eastern chipmunk, meadow jumping mouse, woodland jumping mouse, star nosed mole, red squirrel, northern flying squirrel, American toad, eastern newt, yellow spotted salamander, spring peeper, and garter snake were not used in individual species analyses, but were considered in taxonomic richness and herpetofauna/mammal abundance analyses.

Stand-level forest characteristics

My results indicate that there are clear differences in the stand characteristics resulting from the three management strategies examined (i.e. naturally regenerated, pre-commercially thinned, and planted).

Canopy cover, leaf litter and percent hardwood were highest in naturally regenerated stands, lowest in planted stands, and intermediate in thinned stands (Table 4, Figure 4). Post hoc analyses indicate that canopy cover was influenced by an interaction between stand stage and treatment, being significantly different between stand types at stage 1 but not stage 2 stands (Tables 4, 5). Post hoc analyses indicate that differences in leaf litter, and percentage of hardwood, were statistically significant between naturally regenerated stands and thinned stands, and between naturally regenerated stands and planted stands, but not between planted and thinned stands (Table 4). There were no interactions between the stand stage and treatment for leaf litter and canopy cover. Leaf litter, canopy cover, and percentage hardwood were also found to be positively correlated (Appendix table 3).

Naturally regenerated stands had 3x more area that was wet (depth to water table at 0-0.1 metres) than plantations, and had 1.8x higher wet area than thinned stands although this difference was not significant (Table 4, Figure 4).

Needle litter and shrub cover were influenced by an interaction between stand stage and type, being significantly different between stage 2 pre-commercially thinned stands and stage 2 naturally regenerated stands (needle litter higher in thinned stands, shrub cover higher in naturally regenerated stands), but not significantly different between stage 2 plantations and the other stand types, nor significantly different among any of the stand types in the stage 1 category (Table 5).

Coarse woody debris was also influenced by an interaction between stand stage and type. Stage 1 plantations had twice the coarse woody debris as thinned and naturally regenerated stands of the same stage, but there was no difference in coarse woody debris between stand types in stage 2 stands. Both moss cover and stand density were influenced by an interaction between stand stage and type, however, post hoc analyses did not indicate any significant differences between stand types in either stand stages (Table 5).

Small mammal and herpetofauna relationships with forest management treatments

Taxonomic richness

Combined and small mammal taxonomic richness was significantly higher in naturally regenerated stands than in planted stands and thinned stands were intermediate (Table 4) (Figure 6). The same rank order was found for amphibian taxonomic richness, but the differences were not significant for stand stage or treatment. Total, small mammal, and herpetofauna taxonomic richness were not different between stand stages, however, a post hoc power analysis determined that the probability of detecting a small or medium effect was very low (Appendix figure 1). Taxonomic richness was calculated using all taxa caught, regardless of abundance levels.

Leaf litter, canopy cover, and percentage of hardwood were positively associated with total and small mammal taxonomic richness (Table 4). Naturally regenerated forest stands had significantly higher canopy cover, leaf litter, and percentage of hardwood than the other two forest types (Table 4). Total taxonomic richness was negatively associated with coarse woody debris but this pattern did not apply to either small mammal or herpetofauna taxonomic richness (Table 4) (Appendix figure 2). Small mammal taxonomic richness was also positively associated with stand density and wetness (depth to water table 0-0.1m) (Table 4) (Appendix figure 3).

Abundance

Abundance versus treatment and stand stage

Small mammal abundance was influenced by both stand stage and stand treatment separately, being higher in stage 1 than stage 2 stands and also higher in naturally regenerated stands, followed by thinned and planted stands respectively(Table 4)(Figure 7,8). Total and herpetofauna abundances were influenced by an interaction between stand stage and treatment (Table 4, Figure 7). Total abundance, like taxonomic richness, was statistically significantly higher in naturally regenerated stands than planted stands, however, only in the stage 1 category (Tables 4, 5). Herpetofauna abundance, in contrast, was not significantly different among treatments in stage 1, but was lower in stage 2 thinned stands.

Abundance versus stand characteristics

Leaf litter and percentage of hardwood were positively associated with total abundance of combined small mammals and herpetofauna, and of small mammals alone, comparable to taxonomic richness trends (Table 4) (Appendix figure 4,5). Small mammal abundance, as with small mammal taxonomic richness, had a positive relationship with stand wetness (Table 4) (Appendix figure 4). Total, small mammal and herpetofauna abundances were negatively associated with moss cover (Table 4, Appendix figures 4,5). Herpetofauna abundance was negatively associated with stand density and positively associated with canopy cover (Table 4)(Appendix figure 5).

Individual taxa

Individual taxon versus treatment and stand stage

The *Sorex* spp. category (most likely including sympatric species of masked, pygmy, and smokey shrews) comprised 73% of all small mammal captures. As for both taxonomic richness and small mammal abundances, *Sorex* spp. abundance was higher in naturally regenerated stands followed by pre-commercially thinned stands and then plantation stands (Figure 9). I found a statistically significant difference in abundance between naturally regenerated stands and thinned stands, but not naturally regenerated stands and plantations, although the measured abundance was higher in naturally regenerated stands than plantations (Tables 3, 4).

Red-backed vole abundance was highest in pre-commercially thinned stands, followed by naturally regenerated and then plantations; there was a statistical difference between naturally regenerated and thinned stands, and plantations but not between naturally regenerated and thinned stands (Table 4)(Figure 9).

Deer mouse was influenced by an interaction between stand stage and treatment, being greatest in naturally regenerated stands followed by thinned and planted stands respectively but only in stage 1 stands (Tables 4, 5).

The remaining small mammal species with abundances >20 (i.e. woodland jumping mouse, short-tailed shrew, etc.) showed no significant effect of treatment or stage.

Red-backed salamander was most abundant in naturally regenerated stands, followed by thinned stands, and then least abundant in planted stands (Tables 3, 4)

(Figure 10), and also positively associated with stand development stage (Table 3, 4) (Figure 11). Wood frog was more abundant in naturally regenerated stands than both thinning and plantation managed stands but only in stage 2 stands (Tables 4, 5).

Individual taxa versus stand characteristics

Sorex spp. abundance was negatively correlated with both needle cover and shrub cover, and positively with leaf litter and hardwood (Table 4) (Appendix figure 6). Red-backed vole abundance was positively correlated with both percentage of hardwood forest and stand density (Table 4) (Appendix figure 7). Woodland Jumping mouse abundance, although not significantly different between stand types or stage, did have positive associations with leaf litter and canopy cover (Table 4) (Appendix figure 8). Short-tailed shrew abundance was positively associated with stand wetness and density, and negatively associated with shrub cover (Table 4) (Appendix figure 9). Deer mouse abundance was positively associated with canopy cover, wetness (depth to water table between 0 and 10 cm), as well as dryness (depth to water table between 0.5m and 10 m) (Table 4) (Appendix figure 10).

Both wood frog and red-backed salamander abundances were positively correlated with leaf litter, canopy cover, and percent hardwood (Table 4) (Appendix figures 11,12) in keeping with their positive associations with naturally regenerated stands which have higher levels of these characteristics than the other two stand types (Figure 4). Red-backed salamander abundance was also negatively associated with shrub cover and stand dryness (Table 4) (Appendix figure 12).

DISCUSSION

I found that managed forests, particularly plantations, had negative impacts on diversity and total abundance of small mammals. The results were similar, although not as conclusive, for herpetofauna abundance. These effects are likely due to fundamental differences in stand characteristics in plantations and stands that have been pre-commercially thinned. The effect of management appeared to be reduced in stage 2 managed stands for small mammal abundance, and deer mouse, but not for small mammal taxonomic richness, *Sorex* spp., or red-backed voles. No effect of treatment or stand stage was detected for abundances of woodland jumping mice or short-tailed shrews. Wood frogs were negatively impacted by both pre-commercial thinning and plantations, and were 2.5 times more abundant in stage 2 than stage 1 stands. Red-backed salamanders were less abundant in both thinned and planted stands, and also positively associated with stand development. Taxonomic richness results suggest that naturally regenerated stands support, on average, 1.7 times more taxa than plantations, and 1.1 more taxa than thinned stands (and undoubtedly greater numbers of species). These results suggest that, for some taxa, there are critical stand attributes that are lost during the conversion of the native Acadian mixed-wood forest to managed coniferous stands.

Effect of management treatments on stand characteristics

Leaf litter and percentage of hardwood

My results were consistent with previous studies which found lower levels of leaf litter and a lower percentage of hardwood trees in pre-commercially thinned stands (Etcheverry et al., 2005; Henderson, 2011) and plantations (Waldick et al., 1999; Ross-Davis and Frego, 2002; Johnson and Freedman, 2003; Ramovs and Roberts, 2003) than in naturally regenerated stands. These effects are attributed to tree species selection (PCT), herbicide treatments (plantations), and site preparation (plantations).

There was no difference in leaf litter or percent hardwood between stages, suggesting that the conversion of the native mixed wood forest to managed stands, both thinned and planted, results in a permanent loss of deciduous habitat components.

Depth to water table between 0 and 0.1m

My results were consistent with previous research that found plantations to have fewer damp (i.e. depth to water table 0 – 0.1m) areas (Bliss and Comerford, 2002; DeMaynadier and Houlahan year, in Calhoun and DeMaynadier, 2007). This stand characteristic is important because it would likely result in a drier microclimate, which could have affected the moisture content and therefore the value as habitat for small mammals and amphibians, of other stand characteristics such as leaf litter and coarse woody debris.

Needle litter

Needle litter was influenced by both stand stage and treatment. I found it to be highest in later, stage 2 pre-commercially thinned stands compared to both naturally regenerated stands and plantations. It is likely that stage 2 stands had more litter because there has been more time for needles to accumulate, furthermore I suspect that the difference in needle litter between thinned stands and the other two stand types is a result of the stand composition prior to clear-cutting. Thinned stands were regenerated by natural processes with desired softwood species because they were present as seed trees or in the vicinity of the stand and so likely have had long-term accumulation of needle litter. By contrast, plantations require planting in order to achieve the desired softwood species and naturally regenerated unmanaged stands had a mixed-wood composition and so the long-term contribution of needle litter is likely lower.

Canopy cover

Within the early stage 1 development stands I found canopy cover to be lowest in plantations followed by thinned stands but did not detect a difference between stand types in stage 2. Previous studies have noted a reduction in cover in thinned stands versus naturally regenerated stands (Fuller et al., 2004; Homyack et al., 2004; MacCracken, 2005), however a long-term study by Henderson (2011) also found the difference in canopy cover between stand types to be mitigated by 10 years post thinning. These results are not surprising as the intent of pre-commercial thinning is to reduce competition by removing hardwood and overly dense coniferous trees; it therefore stands to reason that canopy would be reduced post-thinning.

My results echo a study by Veinotte et al. (2003), which found canopy cover in early stage plantations (3 to 8 years) to be lower than reference mixed-wood stands but equal or greater in percent cover than reference stands at later stage plantations (13 to 21 years). I suspect that this shift from a comparatively more open canopy to a closed one in plantations is a result of the accelerated rate at which plantations are managed to grow.

Shrub cover

There was no difference in shrub cover among treatments in stage 1, but there was a difference in stage 2 stand. Shrub cover was lowest (overall for both stages) in stage 2 thinned stands and significantly lower than naturally regenerated stands of the same stage but not significantly lower than plantations. This is contrary to previous studies, which found understory cover to have a positive response to thinning (Hayes et al., 1997; Thomas et al., 1999; Sullivan et al., 2000; Fuller et al., 2004; Sullivan et al., 2005). It does not appear that the observed difference in canopy closure among these stands is the primary driver of shrub abundance.

Coarse woody debris

My results supported the conclusions of 2 previous studies conducted in Maine (Fuller et al., 2004; Homyack et al., 2004) that did not find a significant difference in coarse woody debris between naturally regenerated and thinned stands, but are in contrast to other studies that suggest that thinning has a negative effect on coarse woody debris (Carey and Johnson, 1995; Etcheverry et al., 2005; MacCracken, 2005) in all studies, slash was left in-situ. However, these studies often had reference stands that were much older than the managed stands. I suspect the similarity in CWD between thinned and naturally regenerated stands in this study is related to the fact that both stand types had similar origins prior to thinning, were relatively young (i.e. not to an over-mature stage where natural mortality occurs), and the slash from thinning (fine woody debris) was not considered in the CWD measurement.

Percentage of CWD was 2 times higher in stage 1 plantations than naturally regenerated and pre-commercially thinned stands of the same stage, but there was no significant difference among stand types within stage 2 stands. Presumably, high CWD in early stage plantations is due to site preparation but this difference is not maintained in later stages due to decay (Harmon et al., 1986).

Stand density

I did not find a difference in stand density between naturally regenerated and plantation stands. Ross-Davis and Frego (2002) reached similar conclusions in this region. By contrast, some previous studies have found tree density to be lower in plantations (Hansen et al., 1991; Roberts and Ramovs, 2003) and thinned stands (Hayes et al., 1997; MacCracken, 2005) relative to naturally regenerated reference stands. However, I did find that stand density was greater at stage 2 for plantation and PCT stand types, but not for naturally regenerated stands (Table 5). This is probably because I only included trees with DBH >10 cm in my stand density estimates, and thus many smaller diameter trees would not have been included in stage 1 stand density estimates.

Moss cover

There was a statistically significant interaction between stand stage and stand treatment for moss cover but there were no significant differences when I examined the stages and stand types separately. This can happen when there isn't enough power to detect an effect when stand stages are analyzed separately (Table 5).Moss cover was higher in the stage 2 stands than stage 1 stands (Table 3), which is in contradiction to Ross-Davis and Frego (2002) who found no relationship between bryophyte cover and time since disturbance. My results are in keeping with Hill (1979), who suggests moss cover tends to be greater in a closed canopy, a characteristic that was significantly greater in stage 2 stands than stage 1 stands in my study. Although not statistically significant, moss cover was much higher in PCT, and to lesser extent plantations, than naturally regenerated stands in stage 2 stands, but was similar across all treatments in the early stage. I found moss cover to be negatively associated with percentage of hardwood, (Appendix table 4)- which was lowest in plantations (Table 4), which is consistent with Ross-Davis and Frego (2002), who found bryophyte cover to be higher in plantations than naturally regenerated clear-cuts. They suggest however, that estimate of moss cover is largely driven by a subset of robust perennial bryophyte species, which are likely able to thrive in stands that may not have been ideal habitat for other moss and vascular plant species.

Effect of intensive forest management on small mammals

Community-level effects

Taxonomic richness and abundance in PCT stands

Small mammal taxonomic richness and small mammal abundance were not significantly different between naturally regenerated stands and thinned stands. These results are consistent with previous studies, which found no relationship between thinning and small mammal taxonomic richness (Lautenschlager et al., 1997; Sullivan et al., 2002; Verschuyl et al., 2011) and abundance (Fuller et al., 2004; MacCracken, 2005). This is attributed to the fact that the majority of small mammals are resource generalists (Parker, 1989; Mitchell et al., 1997; Bowman et al., 2001; Sullivan et al., 2002; Sullivan et al., 2005).

My results are consistent with many previous studies in both eastern and western North America which found no long-term effect of thinning (10 -16 years post thinning) on small mammals, compared to both softwood and mixed-wood control sites (Sullivan et al., 2001; Homyack et al., 2004, 2005; Sullivan et al., 2005). Homyack et al. (2004) found no detectable effect of PCT during 1-11 years post thinning on small mammals and suggested that thinning may actually accelerate the development of habitat characteristics, such as a reduced stand and understory density, more common in old growth stands. However, Etcheverry et al. (2005) did find a negative long-term effect of thinning on small mammal taxonomic richness and abundances of many small mammal species (>10 years post thinning) when compared to mixed-wood stands.

Despite reduced leaf litter and percentage of hardwood in stage 2 thinned stands and identified positive associations of small mammals to these stand characteristics, I did not detect a long-term effect of thinning on small mammal abundance. It is possible that the small mammal associations with these stand characteristics are, in fact, a determining factor of small mammal abundance and taxonomic richness but that the effect of stand treatment was not sufficient to be detectable, given low power and high variability in the stand characteristics within thinned stands. It is also possible that there are multiple factors that influence abundance and diversity and that only some of those are associated with treatments such as planting and pre-commercial thinning. In addition, it is possible that small mammals are able to adapt to changes caused by different treatments. For example, needle litter, which was higher in thinned stands than naturally regenerated stands, may have offered a substitute for leaf litter in terms of habitat (e.g. seed caches) for small mammals, as suggested by Abbott and Quink (1970). I did not detect a relationship between small mammals and needle litter but if small mammals are able to use both leaf and needle litter we would not expect to see correlations.

Canopy cover, which was significantly different between stage 1 thinned and naturally regenerated stands but not stage 2 stands, should be further studied as a critical limiting factor for small mammals in thinned forests. Etcheverry et al. (2005) suggest that canopy cover is reduced following thinning, which is essential in providing cover from predation for small mammals. I detected a positive relationship between small mammal taxonomic richness and canopy cover, but not abundance. This is probably because shrews (*Sorex* spp and *Blarina brevicauda*) comprised 72.5% of all small mammal individuals and were not among the species that appear to be affected by canopy cover. That is, the taxa that contributed to reduced taxonomic richness when canopy cover was reduced were not taxa that I captured in high abundance.

This difference between taxonomic richness and abundance results highlights the importance of using both measurements to fully understand the impacts of plantations and pre-commercial thinning. For example, it is possible that the impacts of plantation and/or PCT may not cause local extirpations, but may lead to declines in abundance – such impacts would not be detected if abundance was ignored.

Taxonomic richness and abundance in plantations

I found both small mammal taxonomic richness and abundance to be lower in plantations when compared to naturally regenerated stands. Previous studies have found small mammal abundance to be lower in plantations than in naturally regenerated stands (Langley and Shure, 1980; Gagné et al., 1999; Cameron and Puddister, 2000; Mengak and Guynn Jr., 2003) attributed to a reduction in stand characteristics such as shrub/canopy complexity (Carey and Johnson, 1995; Bowman et al., 1999; Bowman et al., 2001; Ramirez and Simonetti, 2011), leaf litter (Kaminski et al., 2007), and moistness (Miller and Getz, 1977) all of which have been identified in this study as characteristics that are positively associated with small mammals. However, two studies in Virginia, USA and New Brunswick, Canada (both of which have native mixed-wood forests) found that small mammals were as diverse and abundant in plantations as in native mixed-wood controls (Mitchell et al., 1997; Bowman et al., 2001). The authors attributed this to broad habitat and resource adaptability, but they also reported variation in species' abundances among stand types, suggesting that individual species differ in their tolerance for plantation treatments.

Although few studies have specifically compared plantations to both naturally regenerated and PCT stands, Mengak and Guynn Jr. (2003) suggested that plantations were more deleterious for small mammals than thinned stands because herbicide use reduces deciduous understory plant diversity while thinning increases it. I found a similar trend of lower small mammal abundance and taxonomic richness in plantations than PCT stands but the difference was not statistically significant (Table 3).

Structural complexity (e.g. shrub cover, canopy cover, coarse woody debris, native tree species) has been associated with an increase in small mammal abundance and taxonomic richness, and is often negatively associated with plantations (DeGraaf and Rudis, 1990; Hartley, 2002; Ramirez and Simonetti, 2011). The degree to which these characteristics change with stand stage is dependent on the intensity of forest management practices such as site preparation, use of non-native species, and herbicide applications (Carey and Johnson, 1995; Lindenmeyer, 2003). The fact that I found taxonomic richness to be lower in plantations than naturally regenerated stands at both early and later stages suggests that there are critical factors missing for a subset of taxa in plantations even as they develop. These results are in keeping with Johnson and Freedman (2003), who compared various ages of plantations in New Brunswick to naturally regenerated (and un harvested) mixed-wood stands in New Brunswick, and found the comparative loss of characteristics such as snags and percentage of hardwood remained constant as plantations developed. They did, however, find that canopy cover increased with time since initiation in plantations while shrub cover and coarse woody debris decreased. Despite this, I found that effects on small mammal abundance were no longer detectable by the stage 2 of development. Studies that compared small mammal abundance in early stage plantations (10 years or younger) with mature plantations (~20 years old) are scarce. Previous research into stand preparation management techniques indicates that the use of herbicides, does not have a long-term (10+ years) direct effect on small mammal taxonomic richness (Sullivan et al., 1997; Gagné et al., 1999; Sullivan and Sullivan, 2003) but, long-term species specific effects are present, likely as a results of a negative effect on habitat and food resources (Thompson et al., 2003). A review by Thompson et al. (2003) noted that there are no long-term studies of the effect of plantations on small mammals, but that it is theorized that the reduction in a mature forest stage as a result of IFM rotations would be detrimental to species that require mature forest characteristics.

My results suggest that CWD and canopy cover are potentially critical factors for small mammal abundance, which have previously been identified as important factors for small mammals because they provide shelter from predators, habitat for invertebrate food sources, as well as a source of moisture (Carey and Harrington, 2001; Bowman et al., 2000). These characteristics were similar between stage 2 but not stage 1 plantations and naturally regenerated stands.

Species-level effects

Shrews in PCT stands

Shrews (*Sorex* spp. and *Blarina brevicauda*) were more abundant in naturally regenerated stands relative to PCT stands. This observation conflicts with studies which found no difference (Henderson, 2011) or a positive relationship between shrews and thinning, attributed to reduced canopy which allows for more insects and understory vegetation (Wilson and Carey, 2000; Homyack at al., 2005; MacCracken, 2005), but consistent with Fuller et al. (2004) in the Acadian forest in Maine, who found deciduous stands to be most suitable for shrews, and attributed this to the qualities and quantities of the leaf litter. Homyack et al. (2004) compared thinned stands to clear-cut and stands treated with herbicides, while Wilson and Carey (2000) and MacCracken (2005) compared thinned stands to naturally regenerated coniferous stands, and therefore the proportion of deciduous components present in my study may not have been present in these studies. I conclude that there were likely several factors at play which contribute to shrew abundance, and that shrub cover may have been more important in stands that are softwood dominated.

Shrew abundance was positively associated with percentage of hardwood and leaf litter, stand characteristics that are generally greater in naturally regenerated stands than PCT stands. A positive association between shrews and leaf litter has been well documented (Getz, 1961; Merritt, 1986; DeGraaf and Healy, 1992; Wilson and Carey, 2000; DeGraaf and Yamasaki, 2001) as leaf litter maintains a moist microclimate. In addition, leaf litter is positively associated with invertebrate populations (Haskell

2000), and previous studies demonstrated that shrews are positively associated with food availability (Innes et al., 1990; Bellocq et al., 1994). Like Craig, 1995, McKay and Komorski, 2003 and Thompson et al., 2003, I found no association between shrew abundance and coarse woody debris, in contrast with other studies that have found coarse woody debris to be important as a retainer of moisture (e.g. Davis et al., 2010 and DeGraaf and Yamasaki 2001). McKay and Komorski (2003) suggested that coarse woody debris may be more necessary in predominantly softwood stands, where its capacity for moisture retention compensates for the absence of leaf litter.

Shrews in plantations

Shrew (*Sorex* spp. and *Blarina brevicauda*) abundance was lower in plantations than naturally regenerated stands. This is in contrast to previous studies which found that plantations supported more shrews than mixed-wood controls (Naylor and Bendell, 1983 in Bellocq and Smith, 2003; Parker, 1989). Hartling (2004) suggested that shrew abundance was better predicted by stand characteristics such as canopy cover, moisture, temperature, and coarse woody debris than stand type, which may explain why my results are different than some previously published findings. I did not find an association between shrews and canopy cover. These results are consistent with a review by Zwolak (2009), who found no difference in shrew abundance in plantations that were >10 years since initiation and those that were 10-20 years old, suggesting that because shrews are generalist insectivores they are not directly affected by stand development. I found a positive relationship between shrews and leaf litter and percentage of hardwood which was more abundant in managed mixed-wood stands than plantations. As was the case with PCT stands, I suspect that reduced leaf litter in managed stands is a driving factor in the reduced abundance of shrews. Furthermore, there was a significant difference in the depth to water table (i.e. stand moisture) between plantations and naturally regenerated mixed-wood stands, the latter having a higher amount of water closer to the surface (although regression results did not show an association between shrews and stand wetness). Vernal pools are associated with an increase in invertebrates and amphibians (Brooks and Doyle, 2001), which are important food sources for shrews (van Zyll de Jong, 1983; DeGraaf and Yamasaki, 2001).

Although the naturally regenerated stands had greater mean canopy cover, they had many more canopy gaps than the plantations (pers. obs.) and therefore provided greater habitat heterogeneity, including open areas suitable for invertebrates. A more complex canopy provides for cool moist habitat as well as warmer dryer areas, and large invertebrate populations which are favourable for shrews (Banfield, 1974; Innes et al., 1990; DeGraaf and Yamasaki, 2001; Whittaker Jr., 2004).

Red-backed voles in PCT stands

My results were consistent with several studies which have documented red-backed voles to be generally unaffected, or to prefer pre-commercial thinning treatments to reference stands (Sullivan et al., 2000; Sullivan et al., 2001; Homyack et al., 2004; Hanley et al., 2005, MacCracken, 2005; Kaminski et al., 2006; Dracup et al., 2015), but conflicted with others that suggest that red-backed voles are more abundant in naturally regenerated stands than PCT stands (Etcheverry et al., 2005;

Henderson, 2011). Fuller et al. (2004) found that thinning did not negatively affect red-backed vole density within the Acadian forest in Maine because the thinned stands had ample amounts of coarse woody debris and downed logs, previously identified as important to red-backed voles (Carey and Johnson, 1995; Bowman et al., 2000; Etcheverry et al., 2005; Kaminski et al., 2006; Sullivan et al., 2013; Dracup et al., 2015). I did not find a difference in coarse woody debris between treatment types nor did I find an association with red-backed voles and coarse woody debris, however, my measurements of coarse woody debris did not consider fine woody debris, differences in decay class, nor the vertical volume of downed wood and therefore my results must be interpreted with caution. A study in New Brunswick, Canada by Henderson (2011) suggests that a threshold amount of CWD may be required before it can be of habitat value to red-backed voles. Furthermore, several studies that did not find association between red-backed voles and coarse woody debris (Carey and Harrington, 2000; Sullivan and Sullivan, 2001; Homyack et al., 2004; Henderson, 2011) hint at the generalized resource use by red-backed voles. It appears that several habitat characteristics can provide the food, nesting sites, and shelter from predators necessary for voles (Sullivan et al., 2000; Etcheverry et al., 2005). I found positive associations between voles and stand characteristics previously identified as important to voles: adequate canopy cover layer (Pearce and Venier, 2005; Vanderwel et al., 2010), moist microclimate (Miller and Getz, 1977), and stand density typical in old growth forests (Klenner and Sullivan, 2003; Pearce and Venier, 2005). These results suggest that there are several factors at play in predicting red-backed vole abundance, and that it may be possible to maintain an adequate amount of these characteristics within PCT stands to support red-backed voles.

Red-backed voles in plantations

Current research into the effects of plantations on red-backed voles is contradictory, suggesting that voles may be more affected by the absence of specific stand characteristics than by stand type. Based on a review of 50 papers with an emphasis on the boreal forests of Ontario, Thompson (2003) found that the overall effect of intensive forest management on red-backed voles was negative, in response to reduction in coarse woody debris and ground vegetation cover. However, other studies in the northeast of North America suggest that red-backed vole abundance is unaffected by plantations (Parker, 1989) and/or are abundant in a variety of stand types, both mixed-wood and coniferous (DeGraaf et al., 1992; Forbes et al, 2010). I found red-backed vole abundance to be at least 2 times higher in naturally regenerated and PCT stands than plantations across stage 1 and stage 2 stands, suggesting that there is a loss of a critical habitat component in the conversion of native stands to plantations. This is consistent with the reduced red-backed vole abundance reported in the New Brunswick Acadian forest in landscapes with relatively large amounts of plantation (Bowman et al., 2001).

One possible explanation for the negative relationship between red-backed voles and plantations is reduced tree canopy, which has previously been identified as important to red-backed voles (Parker, 1989; McLaren et al., 1998; Thompson et al., 2003; Pearce and Venier, 2005) because it provides cover from predators and increases stand moisture and subsequent fungal food sources (Miller and Getz, 1977; Carey and Johnson, 1995; Fuller et al., 2004). Canopy cover was significantly lower in plantations than in thinned stands, and naturally regenerated mixed-wood stands. Red-backed vole abundance was strongly correlated with canopy cover, suggesting, as other studies have, that cover is an integral stand characteristic for voles and may be more important than disturbance type (Keinath and Hayward, 2003; Pearce and Venier, 2005). Red-backed vole abundance was positively associated with stand wetness (indicative of a moist microclimate), which was significantly lower in plantations than both PCT and naturally regenerated stands. Further research should investigate the relationship between depth to water table (i.e. saturation of soil) and stand characteristics such as coarse woody debris and fungal food sources as it is possible that stand wetness could be an integral factor in providing stand characteristics required by red-backed voles. Furthermore, although I did not find any associations between red-backed voles and deciduous characteristics such as leaf litter, shrub cover, or percentage of hardwood, previous studies found red-backed voles to be dependent on shrub cover for survival from predators and smaller shrubs as a food source (Allen, 1983; Hanley et al., 2005), and thus herbicides, commonly used on young plantations, have been found to have a temporary negative effect (D'Anieri et al., 1987; Runciman and Sullivan, 1996; Gagné et al., 1999; Bowman et al., 2001). It is possible that although I did not find a difference in shrub cover among treatment types, the negative effect of plantations on red-backed voles could be a residual effect of herbicide treatments and subsequent loss of earlier shrub cover.

Contrary to previous studies that have identified red-backed voles as positively associated with stand stage (Zwolak, 2009), I did not detect a difference in red-backed vole abundance between stage 1 and 2 stands. Stage 2 stands within this study did not contain trees in an over-mature stage, and therefore were likely not representative of the 'old growth' stands which red-backed voles have been previously identified as being positively associated with (Sullivan et al., 2000).

Deer mouse versus stand stage and treatment

I found deer mouse abundance to be influenced by both stand stage and treatment, being 9 times greater in naturally regenerated stands than plantations. Likewise, deer mouse abundance was greater in PCT stands than plantations, but only in the stage 1 stands (Table 5), suggesting that the negative effects of management may be mitigated by stand development. My results are consistent with a study by Henderson (2011) in New Brunswick, Canada, who found that deer mice are positively associated with canopy cover, which was significantly different between treatment types in stage 1, but not stage 2, stands.

Effects of intensive forest management treatments on herpetofauna

Community-level effects

Herpetofauna taxonomic richness and abundance versus stand stage and treatment

I did not find herpetofauna taxonomic richness to be affected by stand stage, but I did find abundance to be influenced by an interaction between treatment and stage, being different among treatments in stage 2 stands but not in stage 1 (Table 5). This suggests that the effects of treatment on herpetofauna do not manifest themselves until later in stand development. The negative impacts of stand management at early stages may have been offset by the open canopy of early plantations and PCT stands, which are ideal habitat for migrating juvenile pond-breeding herpetofauna species because of a resulting increase in macro invertebrates as well as air and pond temperature (Bury and Corn, 1988; Skelly et al., 1999; Palik, 2001). Skelly et al. (1999) found that forest succession had a negative effect on hydro-period, reducing breeding habitat for some amphibian species. Furthermore, Aubry (2000) found that a partially open canopy cover may be beneficial to amphibians by allowing understory shrub cover, which provides important foraging opportunities for food sources. This contradicts some previous studies that found an increase in herpetofauna abundance (mostly salamanders) with stand stage: these studies looked at various ages following clear-cutting or burning with no subsequent management comparative to non- disturbed stands of 30 years and up to 100 years (Pough et al., 1987; Bonin, 1991; Petranka et al., 1993; Duguay and Wood, 2001; Hicks and Pearson, 2003; Thompson et al., 2008)- an age generally older than stands at commercial maturity.

Taxonomic richness (although not statistically significant) as well as wood frog and red-backed salamander abundances, indicated a preference for older stands. This is attributed to increased coarse woody debris, lower temperatures, and increased stand moisture (Dupuis et al., 1995; Martin and McComb, 2003; Loehle et al., 2005).

This difference between total and species-specific abundance as well as taxonomic richness responses highlights the importance of using both measurements to assess forest suitability- a subset of species may thrive in specific environments, but all possible species may not be represented (Aubry, 2000).

Herpetofauna taxonomic richness and abundance in PCT stands

A review by Verschuyl et al. (2011) of 33 studies concluded that herpetofauna response to thinning is variable and species specific, but generally neutral within North America. I did not detect a statistically significant difference in herpetofauna taxonomic richness between thinned and naturally regenerated stands. Low statistical power, however, prevented detection of any potential small and moderate effects. Herpetofauna taxonomic richness trended toward being higher in naturally regenerated mixed-wood stands than PCT stands, supporting suggestions that loss of deciduous trees and leaf litter are detrimental, particularly over time to terrestrial salamanders (Waldick et al., 1999; Ryan et al., 2002; Semlitsch et al., 2009).

Abundance of herpetofauna, conversely, was affected by an interaction between stand stage and treatment, being significantly higher in older naturally regenerated stands than PCT stands (yet not plantations), but not statistically different between treatment types in the stage 1 stands. I suspect that total herpetofauna abundance (most likely in stage 1 stands) was in part influenced by green frogs, accounting for 35% of herpetofauna captures. This species favours a more open canopy (prevalent in younger stands), as it increases pond temperature and invertebrate food sources (Skelly et al., 2014).

Because pre-commercial thinning removes the competitive deciduous tree component of stands, as canopy cover closes in older stands, shrub cover is reduced relative to stage 1 stands. The loss of leaf litter and lower percentage of hardwood have been identified as important to herpetofauna. These factors are likely the mechanisms behind the reduction of herpetofauna in stage 2 thinned stands, suggesting herpetofauna could be negatively affected by stand succession. For many native herpetofauna species, especially salamanders, thinning practices that maintain a partial canopy and allow leaf litter to persist have been shown to promote the moisture retention and protection from predators herpetofauna require (Pough et al., 1987; Harpole and Haas, 1999; Grialou et al., 2000; Karraker and Welsh, 2006; Verschuyl et al., 2011). The long-term effects of pre-commercial thinning on herpetofauna have not been well documented. I was not able to locate any previous studies that evaluated herpetofauna abundance or taxonomic richness in multiple age stands since thinning. Of the two studies published that explicitly look at amphibian response to thinning after at least 10 years, neither found a negative effect of thinning. A previous study in California found no difference in amphibian abundance between previously clear-cut stands that were either thinned 10 years prior or un-thinned. They suggest that the thinning treatment in primarily softwood forest types does not have negative long-term effect on amphibian abundance but may affect body condition (Karraker and Welsh, 2006). In New England Brooks (1999) found no long term effect of thinning on red-backed salamanders in oak forest that had been thinned 12-21 years previously when compared to un-thinned controls. Brooks (1999) attributes an increase in understory vegetation as a result of thinning disturbance to mitigate any effect on red-backed salamanders.

Herpetofauna taxonomic richness and abundance in plantations

In contrast to several other studies (e.g. Bennett et al., 1980; DeMaynadier and Hunter, 1998; Waldick et al., 1999; Hanlin and Martin, 2000; Ryan et al., 2002), and two reviews (DeMaynadier and Hunter, 1995; Palacios et al., 2013), I found no statistically significant decline in herpetofauna richness or abundance in plantations, nor did I find any positive associations with stand characteristics. Stand characteristics previously identified as important to herpetofauna, such as reduction in leaf litter (Pough et al., 1987; Waldick et al., 1999), canopy cover (Harmon et al., 1984; Dupuis et al., 1995; Harpole and Haas, 1999), percent hardwood (Degraaf and Rudis, 1990; Thomspon et al., 2003; Loehle et al., 2005), coarse woody debris (Dupuis et al., 1995; Waldick et al., 1999) and stand wetness or proximity to vernal pools/ (Fleet and Autrey, 1999; Ryan et al., 2002) were all lower in plantations compared to naturally regenerated mixed-wood stands. Herpetofauna taxonomic richness and abundance were reduced in plantations relative to naturally regenerated stands such that that statistical power prevented the detection of an effect. However, I did find herpetofauna abundance to be negatively associated with moss cover, stand density, and canopy cover. This may be because a preference for increased temperature and reduced canopy cover (Skelly et al., 2002; Skelly et al., 2014) offset some of the deleterious effects of plantations.

The long-term effects of plantations on herpetofauna have not been well documented. A study by Means et al. (1996) in Florida over 22 years found a decline in amphibian abundance over 10 generations, and theorized that mechanical alteration of the terrestrial habitat, loss of vernal pools, and the consistently closed canopy associated with older plantations, are important factors. It is possible that site preparation led to a reduction in vernal pools as a result of fewer depressions (pers. obs.).Combined with older trees requiring more water this may lead to a gradual loss of available habitat for pond-breeding species. A comparison between means in the different stand types suggests that herpetofauna were less abundant than unmanaged naturally regenerated stands among stage 2 stands (although not statistically significantly different). I suspect that I did not have enough statistical power to detect a potential effect, but it is possible that older plantations provide a less favourable habitat for herpetofauna when compared to naturally regenerated stands of the same age.

Species-level effects

Wood frog versus stand stage and treatment

Wood frogs breed in ponds but then disperse to forests in following the aquatic larval stage. Previous studies have identified wood frogs as being particularly sensitive to terrestrial habitat characteristics in their non-water life phase when compared to other anuran species (DeMaynadier and Hunter, 1998; Guerry and Hunter, 2002; Patrick et al., 2006; Popescu et al., 2012). I found wood frog abundance to be influenced by an interaction between stand stage and treatment, with abundances being highest in stage 2, naturally regenerated stands (Table 5). These results are in keeping with previous research in eastern North America, which found that adult wood frogs are associated with old stand characteristics – a closed canopy (Gibbs, 1998; Baldwin et al., 2006) presence of leaf litter and higher percentage of hardwood, all of which are generally associated with old stands.

Wood frogs in PCT stands

I found wood frogs to be negatively affected by PCT, although literature examining the effect of thinning on wood frogs is scarce, previous studies suggest the severity of the effect is dependent on the intensity of reduction in canopy cover (Gibbs, 1997; Patrick et al., 2006; Blomquist and Hunter, 2010; Todd et al., 2014; Semlitsch et al., 2009). In addition, I found positive relationships between wood frogs and canopy cover, leaf litter, and percent hardwood (Heatwole, 1961; Skelly et al., 2002; Baldwin et al., 2006; Thompson et al., 2008; Blomquist and Hunter, 2010), probably because these characteristics contribute to a comparatively cooler microclimate and increased moisture than more open stands. Thinned stands had a lower canopy cover, lower percentage of hardwood, and less leaf litter than naturally regenerated mixed-wood stands, but did have an average canopy cover of 61%. This is above the previously suggested critical threshold of 50% cover necessary to maintain wood frog abundance (Patrick et al., 2006; Blomquist and Hunter, 2010). A review by Semlitsch et al. (2009) suggests that thinning has the potential to be positive for wood frogs where there is an increase in shrub cover, with this leading perhaps to increased invertebrate food sources and a favourable microclimate. However, I was not able to confirm this theory as I did not detect a difference in shrub cover between the thinned stand and the naturally regenerated mixed-wood stands in this study. Like Patrick et al. (2006), I did not find coarse woody debris to be important for wood frogs. However, this contradicts findings by Blomquist and Hunter (2010) who suggested coarse woody debris is important as it retains moisture. As was noted for red-backed voles, it is possible that coarse woody debris is more important where characteristics such as leaf litter, vernal pools, and canopy cover are not providing adequate moisture retention. My findings suggest that there are potentially several factors at play. Presence of a deciduous component seems to be important, and the suggested threshold of 50% canopy cover should be considered conservative for maintaining wood frog populations. Future research should compare the degree of canopy cover and its impact on shade intolerant shrub cover.

Wood frogs in plantations

My results are consistent with studies which found wood frogs to be more abundant in naturally regenerated mixed-wood stands than plantations (Degraaf and Rudis, 1990; DeMaynedier and Hunter, 1995; Waldick et al., 1999; Baldwin et al., 2006). Higher wood frog abundance was associated with greater canopy cover and deeper leaf litterboth of which are features of forests others have found important for wood frogs (Heatwole, 1961; DeMaynadier and Hunter, 1999; Waldick et al., 1999; Baldwin et al., 2006; Blomquist and Hunter, 2010). The importance of percent canopy cover, which differed significantly among naturally regenerated ($\bar{x}$=89%), thinned ($\bar{x}$=61%), and planted stands($\bar{x}$=33%), is consistent with the suggestion that a threshold of 50% cover is necessary for wood frogs (Patrick et al., 2006; Blomquist and Hunter, 2010).

Red-backed salamanders versus stand stage

DeMaynadier and Hunter (1995, 1998) suggest that because red-backed salamanders are non-migratory terrestrial breeders and rely on cutaneous respiration, they are potentially more sensitive to, and a good indicator of, forest management practices. My results are consistent with current research, which suggests that salamanders are positively associated with time since clear-cutting (Welsh, 1990; Bonin 1991; DeMaynadier and Hunter, 1995; Dupuis et al., 1995; Herbeck and Larsen, 1999; Grialou et al.. 1999). Homyak and Haas (2009) suggest that 60+ years are required following a harvesting disturbance to adequately return the litter layer to a state suitable for salamander habitat, as opposed to DeMaynadier and Hunter (1995) who suggests that timeline to be closer to 25 years post-harvest. My study did not have a comparable 'old growth' stand, but does suggest that habitat characteristics previously identified as important to red-backed salamanders, such as leaf litter (Bonin, 1991; Pough, 1987) and canopy cover (DeMaynadier and Hunter, 1998; Brooks, 1999; Harpole and Haas, 1999), are more prevalent in stage 2. Forest managers should therefore maintain this later stage of forest stand in order to provide habitat for red-backed salamanders.

Red-backed salamanders in PCT stands

I found red-backed salamander abundance to be 2 times greater in naturally regenerated stands than PCT stands; however, this result was not statistically significant. Current research in coniferous-dominant stands suggests that red-backed salamander abundance is reduced initially following thinning, but recovers once a deciduous shrub cover is established (Brooks, 1999; Grialou et al., 2000). Short-term and long-term studies in the Appalachian region suggest that, because salamanders depend on a moist microclimate (Todd et al., 2014), that the strength and duration of effects are related to the degree to which canopy cover is reduced (Knapp et al., 2003; Homyak and Haas, 2009). By contrast, Messere and Ducey (1998) found no effect of thinning on red-backed salamanders, which they attributed to adequate residual leaf litter. I found red-backed salamanders to be positively associated with increased canopy cover, increased percent hardwood, and increased leaf litter, which is consistent with other studies which have identified shaded, moist habitats (Dupuis et al., 1995; Clay and Brownlie, 1997; Morneault et al., 2004), and leaf litter (Renaldo et al., 2011) as a critical requirements, all of which were lower in PCT stands than naturally regenerated ones. Harpole and Haas (1999) noted that thinned stands provide adequate canopy cover for red-backed salamanders much sooner than stands completely harvested. My results suggest that forest managers should maintain at least a 66% (mean PCT canopy cover) cover, and that 33% (mean plantation canopy cover) cover is inadequate at providing habitat for red-backed salamanders. Future research should examine the threshold at which canopy cover maintains an equivalent habitat to naturally regenerated stands.

Red-backed salamanders in plantations

The conversion of previously mixed-wood stands to plantations has been identified as detrimental to red-backed salamanders as a result of a decrease in pH (Clay and Brownlie, 1997), leaf litter (Pough et al., 1987; Waldick et al., 1999), canopy cover and moisture (Clay and Brownlie, 1997; Mitchell et al., 1997; Fleet and Autry, 1999). I found that the density of red-backed salamanders was 86% lower in plantations relative to naturally regenerated mixed-wood stands. Previous studies found the impacts of harvesting on terrestrial salamanders are temporary and abundances will be equivalent to naturally regenerated controls over time (Harper and Guynn, 1999; Duguay and Bohall Wood, 2002). However, my results suggest that the cumulative effect of initial harvesting, site preparation, and plantations result in a permanent loss of habitat features integral to re-backed salamander survival. Red-back salamander abundance was positively associated with leaf litter, canopy cover, and percentage of hardwood and negatively associated with stand dryness and shrub cover. The importance of coarse woody debris has been highlighted by several studies (Freedman et al., 1996; Waldick et al., 1999; Hartley, 2002; Ross-Davis and Frego, 2002; Johnson and Freedman, 2003; Betts et al., 2005) as an important habitat characteristic often lost by plantation management and shorter harvest rotations. However, I did not find significant differences in coarse woody debris among stand types in this study. Previous studies have noted that coarse woody debris must be moist in order to be suitable habitat for red-backed salamanders (Dupuis et al., 1995; Waldick et al., 1999). A negative association with stands that had a depth to water table greater than 50 cm suggests that perhaps a greater in moister in stands (and therefore moist substrate) may be required, along with adequate coarse woody debris. DeMaynadier and Hunter (1995) expressed concern about the comparison of relatively young plantations to natural stands that are much older. The work reported here addresses this concern by comparing forest stand types that are similar in both relative and seral age. In spite of this, and after controlling for stage I, I still found effects on salamander abundance.

I was not able to locate any long-term studies on the effect of plantations on red-backed salamanders, however Pough (1987) noted that red-backed salamander abundances were lower in older plantations when compared to naturally regenerated mixed-wood controls. Previous studies have found that abundance was positively associated with time since clear-cutting (Welsh, 1990; Bonin 1991; DeMaynadier and Hunter, 1995; Dupuis et al., 1995; Herbeck and Larsen, 1999; Grialou et al., 2000). In light of the fact that managed forests operate on a ~30 year growth to harvest rotation, it is recommended that forest managers maintain older stands as part of the forest landscape in order to sustain red-backed salamander populations. My results suggest that a deciduous component (i.e. leaf litter and percentage of hardwood) and an adequate canopy cover should be maintained in order to provide habitat for red-backed salamanders. Because red-backed salamanders are non-migratory terrestrial breeders and rely on cutaneous respiration (DeMaynadier and Hunter 1995, 1998), they are potentially more sensitive to, and a good indicator of, forest management practices. In light of this reliance on cutaneous respiration, I suspect that depth to water table (i.e. stand wetness), which was greatest in naturally regenerated stands, followed by PCT, and plantations, is a critical factor for red-backed salamanders.

Future research

Plantations were found to have reduced habitat value for small mammals and some herpetofauna relative to naturally regenerated areas. Pre-commercially thinned stands were found to be intermediate in habitat value. In order to identify when/if pre-commercially thinned stands provide equal habitat value to naturally regenerated stands, the intensity of thinning should be further studied to assess potential thresholds of tolerance for species that require deciduous components to forests. Because all three of the forest types sampled had been clear-cut, further research to evaluate whether the naturally regenerated stands at both stages provide equivalent habitat for native small mammals and herpetofauna than undisturbed old growth Acadian forests is important. Further comparisons between the end of 'management rotation' in previously clear-cut, naturally regenerated, stands and old growth Acadian forests should be performed to address whether each provides habitat features necessary to support similar taxonomic richness and abundance of small mammals and herpetofauna. Finding a term that best described stand 'age' required a compromise between age since clear-cutting and age of establishment. This accommodation posed significant difficulty for naturally regenerated stands in particular, because of their variable rate of re-establishment. Future research should study differences in biodiversity (mycology, flora, fauna) between stands that are established immediately following clear-cutting and those that have taken longer to establish. This will provide a better perspective as to whether seral stage or actual age is the more appropriate measure of stand ‘stage’.

Adjacent site context has been identified as important when considering loss of mature forest habitats as a result of shorter age rotation (Freedman et al., 1994; Bowman et al., 2000; Thompson et al., 2003) because small mammals and herpetofauna may be foraging in one stand type but breeding in another, or using habitat beyond the sampled area. Future research might include a landscape level approach considering both stand type and stage when considering the effects of forest management on small mammals and herpetofauna.

I did not have confidence in my identification of live shrews, which, is a weakness in this study. Future research should address the fact that traps designed for the live capture and release of individuals present issues of species identification in the genus *Sorex*. While euthanizing animals to identify via traditional skull pattern methods is an option, such an approach may influence the measurement of density dependent features of small mammal populations (e.g. abundance, age-structure, associated species, etc.) and increasingly is considered ethically unacceptable. Genetic approaches involving tissue sampling may be a solution.

I used several sampling techniques (i.e. pitfall traps, cover object searches, Tomahawk trap and Sherman traps) but it was clear that there were species that were not well-sampled by any of these techniques. For example, red squirrels may be more common at these sites than my sampling indicated. Red squirrels were captured more frequently in other studies even where those studies sampled similar forest types with similar live traps (Etcheverry, 2005; Henderson, 2005; Homyack, 2005). Research examining how robust my results are to alternative sampling techniques (e.g. alternative bait type or trap placement) would be a useful complement. That said, I do not believe that there was any bias in the trapping techniques I employed that would result in any species being more likely to be trapped in a particular stand type or stage.

My stand characteristics measurements of coarse woody debris and canopy cover were coarse. When coupled with low power, I cannot discount the possibility of a type 2 error. Further studies should not only consider overall percentage of canopy cover, but also canopy structure (composition and layers) and size of gaps, coarse woody debris should be measured to account for volume and decay class, and small mammal and herpetofauna trapping effort should be increased to gain further data (especially important for taxonomic richness results).

Taxonomic richness counts were taken from all species captured in each stand type irrespective of the abundance of that species. Although I have no reason to believe there is a sampling bias between sites, there may be concern that rarely caught species can disproportionately affect the results. In order to address this concern, I compared taxonomic richness counts including rarely caught species and taxonomic richness counts using only species with a capture rate of more than 20 individuals, and compared both sets of figures to stand treatment and stage (Appendix table 5). However, a comparison between the two methods of calculating taxonomic richness suggests that total taxonomic richness is negatively affected by plantations, irrespective of how taxonomic richness is calculated. However, the relationship between small mammal taxonomic richness and treatment is less evident when considering only species with capture rates greater than 20, whereas herpetofauna taxonomic richness and treatment is shows greater differences. Method of calculating taxonomic richness did change the fact that taxonomic richness was not affected by stand stage.

Conclusions and management implications

Both abundance and taxonomic richness of small mammals, and specifically *Sorex* spp., red-backed voles, and red-backed salamanders were found to be negatively impacted by plantations and to a lesser extent pre-commercial thinning in New Brunswick. These effects were found in both early and late stage stands. I did not find an effect of forest management on the woodland jumping mouse, or the short-tailed shrew.

Total abundance and deer mouse abundance were influenced by an interaction between stand stage and treatment, being dissimilar within treatment types in stage 1 but not stage 2, suggesting that the effects of treatment may have been mitigated over time. Herpetofauna and wood frog abundance were also influenced by an interaction between stand treatment and stage, however, the effect of treatment is significant at the stage 2 development stage. I suspect that forest management practices do have a negative impact on herpetofauna but that the effects were skewed by pond-breeding amphibians, which prefer an open canopy as it increases invertebrate food sources and pond temperature (Skelly et al., 2002). Both wood frog and red-backed salamander species, which are known to be more reliant on terrestrial habitat, were negatively affected by forest management, most strongly by plantations. Contrary to previous research, no relationship between herpetofauna taxonomic richness and intensive forest management practices or stand stage was detected. However, I saw trends that were consistent with a negative effect and my study did not have good power to detect small and/or moderate effects.

The effects of stand management were at least partly due to changes in stand characteristics- of particular importance were leaf litter, percentage of hardwood and stand wetness (depth to water table at 0-10cm). Wood frogs and red-backed salamanders were positively associated with leaf litter, canopy cover, and percentage of hardwood all of which are lowest in plantations, followed by thinned stands. Small mammal taxonomic richness also had a positive relationship with canopy cover and stand density- characteristics that have been previously identified as being important at maintaining shelter from predators as well as providing a moist microclimate integral in providing food supply of invertebrates and fungi.

Red-backed salamanders were positively associated with stand development as well as most abundant in naturally regenerated stands, suggesting that forest managers should maintain older staged naturally regenerated forests in order to conserve this species. Small mammal abundance was influenced by stand stage, being higher in stage 1 than stage 2 stands. By contrast, small mammal taxonomic richness was not influenced by stand stage. This difference between abundance and taxonomic richness illustrates the importance of both measures when assessing the relative value of intensively managed forests as habitat for small mammals.

The loss of canopy cover resulting in a dryer microclimate, and a reduced deciduous component that provides leaf litter has been identified as integral for a subset of both small mammal and herpetofauna species. Trends indicate that stage 2 successional (>10 years post thinning or planting) managed stands do not provide a habitat similar to controls for small mammals and herpetofauna, with the exception of pond-breeding amphibians. This suggests that although canopy cover is greater in stage 2 stands, a deciduous component to stands is more important than canopy cover or time since disturbance. Forest managers should maintain documented habitat associations in managed stands in an effort to provide habitat for a greater number of species endemic to the Acadian forest ecosystem. The challenge of finding a way to maintain a deciduous component within managed stands, while at the same time maintaining harvest levels, is significant. This may require a landscape level approach integrating patches of intensive forest management with patches of naturally regenerated areas.

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TABLES

Table 1. Total abundance of individual small mammal and herpetofauna species caught for the three sampling methods during 2005-2006 sampling periods. PFT=pit fall trap; LT Tom= Tomahawk live trap; LT Shrm= Sherman live trap; NCS= natural cover search. All species with a total capture of less than 20 individuals were not considered for further individual analysis, but were included in taxonomic richness and total, small mammal, or herpetofauna abundances.

	Total number	PFT	LT Tom	LT Shrm	NCS
Small Mammals	840	771	18	51	0
Herpetofauna	244	137	0	0	107
***Sorex* spp.**	609	603	0	6	0
Red back vole	92	78	0	14	0
Woodland jumping mouse	45	37	0	8	0
Short tailed shrew	26	26	0	0	0
Deer mouse	24	12	0	12	0
Snowshoe hare	15	2	13	0	0
Chipmunk	11	1	0	10	0
Meadow jumping mouse	9	9	0	0	0
Star nosed mole	3	3	0	0	0
Red squirrel	4	1	3	0	0
Flying squirrel	2	0	2	0	0
Red back salamander	102	2	0	0	100
Green frog	86	86	0	0	0
Wood frog	21	21	0	0	0
American toad	16	16	0	0	0
Newt	7	5	0	0	2
Yellow spotted salamander	6	1	0	0	5
Peeper	4	4	0	0	0
Garter snake	2	2	0	0	0

Table 2. Abundance of individual small mammal and herpetofauna species caught in three stand types with differing management at two age classes, during the 2005-2006 sampling periods. Values are totals? 2005 capture numbers are in the top left corner, 2006 are in the top right and total captures are on the bottom. S1.NR= Stage 1, naturally regenerated sites; S2.NR= Stage 2 naturally regenerated sites; S1.PCT= Stage 1 pre-commercially thinned sites; S2.PCT= Stage 2 pre-commercially thinned sites; S1.RP=Stage 1 plantation; S2.RP=Stage 2 plantation.

	S1.NR			S2.NR			S1.PCT			S2.PCT			S1.RP			S2.RP		
	2005	2006	Total	2005	2006	Total	2005	2006	Total	2005	2006	Total	2005	2006	Total	2005	2006	Total
Small mammal abundance	120	113	233	74	61	135	80	69	149	72	32	104	69	36	105	50	64	114
Herpetofauna abundance	30	11	41	25	14	39	41	45	86	11	1	12	8	40	48	11	7	18
***Sorex* spp.**	89	83	172	68	39	107	44	27	71	56	32	88	68	23	91	26	54	80
Southern red-backed vole	11	12	23	1	16	17	10	4	14	9	14	23	2	3	5	6	4	10
Woodland jumping mouse	7	6	13	3	5	8	0	3	3	1	8	9	0	1	1	11	0	11
Short tailed shrew	4	4	8	5	0	5	3	1	4	3	2	5	1	1	2	2	0	2
Deer mouse	3	6	9	0	0	0	3	1	4	2	5	7	1	0	1	0	3	3
Snowshoe hare	5	0	5	1	2	3	3	0	3	1	0	1	0	3	3	0	0	0
Eastern chipmunk	0	0	0	1	7	8	0	0	0	0	0	0	0	0	0	0	3	3
Meadow jumping mouse	0	0	0	0	0	0	6	0	6	0	0	0	0	0	0	3	0	3
Star nosed mole	1	1	2	0	0	0	0	0	0	0	0	0	0	1	1	0	0	0
American red squirrel	0	1	1	1	0	1	0	0	0	2	0	2	0	0	0	0	0	0
Northern flying squirrel	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0	2	0	2
Northern red-backed salamander	16	5	21	27	14	41	4	0	4	20	7	27	1	0	1	7	1	8
Green frog	6	4	10	2	19	21	2	40	42	2	3	5	6	0	6	2	0	2
Wood frog	2	0	2	9	4	13	0	0	0	2	3	5	1	0	1	0	0	0
American toad	0	2	2	1	4	5	1	0	1	0	0	0	1	1	2	1	5	6
Eastern newt	2	0	2	0	2	2	0	0	0	1	0	1	0	0	0	1	1	2
Yellow spotted salamander	0	0	0	2	1	3	1	0	1	0	0	0	2	0	2	0	0	0
Spring peeper	4	0	4	0	0	0	0	0	0	0	0	0	0	0	0	0	0	0
Garter snake	0	0	0	0	1	1	0	0	0	0	1	1	0	0	0	0	0	0

Table 3. Mean (per stand) and standard error of small mammal and herpetofauna per hundred trap nights, and stand level vegetation samples within two stand stages, and three stand treatment types. S1= stage 1 stands, had coniferous trees with an average height between 3-5 metres , whereas S2= stage 2 stands, stands had coniferous trees with an average height of 10-20 metres. NR= Clear-cut and naturally regenerated; PCT= Clear-cut, naturally regenerated, and mechanically pre-commercially thinned; RP= clear-cut, planted with coniferous trees, and sprayed with herbicide. Letters indicate statistically significant (α 0.1) differences between stage or treatment categories.

	Stage 1		Stage 2		NR		PCT		RP	
	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.	Mean	S.E.
Total Taxonomic Richness	**6.17**	0.58	**6.92**	0.89	**8.25^{a}**	0.65	**6.25^{ab}**	0.94	**5.125^{b}**	0.83
Mammal Taxonomic Richness	**4.17**	0.34	**4.17**	0.44	**5.125^{a}**	0.30	**4.00^{ab}**	0.42	**3.375^{b}**	0.50
Herpetofauna Taxonomic Richness	**2.00**	0.37	**2.75**	0.52	**3.13**	0.65	**2.25**	0.90	**1.75**	0.90
Total Abundance	**55.17^{a}**	5.75	**35.17^{b}**	4.71	**56.00**	7.55	**43.88**	6.73	**35.63**	6.39
Mammal Abundance	**40.58^{a}**	4.68	**29.42^{b}**	4.25	**46.00^{a}**	6.94	**31.625^{b}**	3.41	**27.375^{b}**	4.53
Herpetofauna Abundance	**14.58^{a}**	2.97	**5.75^{b}**	1.34	**10.00**	1.50	**12.25**	4.41	**8.25**	3.24
Sorex spp.	**8.28**	1.40	**6.82**	1.04	**10.38^{a}**	1.53	**5.92^{b}**	0.94	**6.36^{b}**	1.56
Shorttailed shrew	**0.35**	0.11	**0.30**	0.12	**0.48**	0.19	**0.33**	0.12	**0.15**	0.06
Southern red-backed vole	**1.04**	0.21	**1.24**	0.26	**1.49^{a}**	0.32	**1.38^{a}**	0.26	**0.56^{b}**	0.16
Deer mouse	**0.35**	0.10	**0.25**	0.11	**0.33**	0.15	**0.41**	0.11	**0.15**	0.11
Northern red-backed salamander	**0.64^{a}**	0.28	**1.88^{b}**	0.46	**2.31^{a}**	0.57	**1.15^{ab}**	0.46	**0.33^{b}**	0.16
Wood frog	**0.07^{a}**	0.05	**0.45^{b}**	0.14	**0.56^{a}**	0.20	**0.19^{b}**	0.10	**0.04^{b}**	0.04
Leaf litter	**1.82**	0.46	**2.29**	0.49	**3.78^{a}**	0.56	**1.07^{b}**	0.16	**1.32^{b}**	0.33
Needle litter	**0.71^{a}**	0.22	**1.77^{b}**	0.30	**0.69^{a}**	0.25	**1.85^{b}**	0.46	**1.18^{ab}**	0.27
Coarse woody debris	**1.08**	0.17	**1.22**	0.13	**0.91**	0.17	**1.16**	0.19	**1.38**	0.17
Shrub cover	**2.17^{a}**	0.26	**1.18^{b}**	0.22	**1.60**	0.21	**1.61**	0.43	**1.81**	0.38
Moss cover	**0.67^{a}**	0.15	**1.60^{b}**	0.37	**0.73**	0.27	**1.68**	0.45	**0.98**	0.35
Canopy cover	**44.94^{a}**	9.68	**77.26^{b}**	5.26	**88.88^{a}**	2.26	**61.22^{b}**	10.13	**33.20^{c}**	8.30
% hardwood	**27.63**	10.55	**26.42**	8.40	**65.14^{a}**	10.09	**11.82^{b}**	2.96	**4.12^{b}**	1.66
Tree density	**0.92^{a}**	0.05	**1.11^{b}**	0.07	**1.04**	0.07	**1.07**	0.10	**0.94**	0.07
Depth to water 0-10cm	**20864.58**	6541.67	**17899.49**	5455.14	**31394.62^{a}**	9273.96	**16600.55^{ab}**	6991.42	**10150.93^{b}**	1782.66
Depth to water 50cm-10m	**143836.68**	27696.87	**158108.92**	37326.60	**161269.53**	59699.72	**165012.44**	28386.12	**126636.43**	24828.12

Table 4. Probability (p) values for randomization, interactions between stand types, post hoc analyses, and regressions of small mammal, herpetofauna, and stand characteristics between stand treatment variables. Stand variables are stage, S1=stage 1 (coniferous trees 3-5 metres) and S2=stage 2 (coniferous trees 10-20 metres); and type, NR= Clear-cut and naturally regenerated; PCT= Clear-cut, naturally regenerated, and mechanically pre-commercially thinned; RP= clear-cut, planted with coniferous trees, and sprayed with herbicide. All p values of 0.1 or less are in bold. **'.'** indicates that the test was not applicable.

Dependent Variable	Randomization		Interaction	Post Hoc			Regression									
	Stage	Treatment	Stage&Treatment	PCT-NR	RP-NR	RP-PCT	Leaf	Needle	CWD	Shrub	Moss	Canopy	%HW	Density	Water0-10cm	Water50-10m
TAXONOMIC RICHNESS																
Total	0.528	**0.042**	0.924	0.154	**0.028**	0.448	**0.018**	-0.056	**-0.099**	-0.310	-0.1270	**0.025**	**0.044**	0.429	0.129	-0.6090
Small Mammal	1	**0.018**	0.859	0.128	**0.014**	0.41	**0.007**	-0.377	-0.152	-0.3090	-0.1170	**0.013**	**0.012**	**0.098**	**0.026**	0.5830
Herpetofauna	0.330	0.214	0.713	0.300	0.097	0.592	0.1230	-0.8480	-0.1480	-0.4330	-0.2550	0.1300	0.2430	0.9340	0.5340	0.1870
ABUNDANCE																
Total Abundance	**0.023**	0.138	**0.043**	0.260	0.117	0.447	**0.101**	0.1350	-0.3900	0.5370	**-0.026**	0.781	**0.074**	**-0.077**	0.221	-0.571
Small Mammal	**0.095**	**0.042**	0.117	**0.066**	**0.018**	0.582	**0.021**	0.3120	0.4140	0.7860	**-0.106**	0.203	**0.011**	-0.207	**0.109**	-0.567
Herpetofauna	**0.008**	0.718	**0.005**	0.668	0.732	0.392	-0.779	-0.1160	-0.6270	0.3570	**-0.038**	**-0.097**	0.754	**-0.088**	0.957	0.793
Sorex spp.	0.451	**0.060**	0.457	**0.028**	**0.071**	0.841	**0.044**	-0.3960	0.5250	-0.6210	-0.312	0.214	**0.011**	-0.990	0.580	-0.264
Southern red-backed vole	0.617	**0.041**	0.52	0.852	**0.020**	**0.031**	0.7020	0.9510	0.4310	-0.4680	**0.052**	**0.005**	0.178	**0.059**	**0.044**	**0.014**
Woodland jumping mouse	0.351	0.753	0.465	0.464	0.480	0.956	**0.093**	-0.9300	0.9780	-0.2820	-0.9200	**0.082**	0.2390	0.7310	0.7690	0.5900
Shorttailed shrew	0.888	0.222	0.860	0.528	**0.087**	0.271	0.4150	-0.9020	0.9550	**-0.069**	-0.3010	0.195	0.415	**0.089**	**0.017**	0.279
Deer mouse	0.428	0.270	**0.033**	0.59	0.252	0.174	0.5750	-0.6740	0.7450	-0.2110	0.6770	**0.087**	0.575	-0.675	**0.020**	**0.028**
Wood frog	**0.0100**	**0.0110**	**0.005**	**0.054**	**0.006**	0.522	**0.075**	0.470	0.580	-0.202	0.8890	**0.008**	**0.093**	0.529	0.565	-0.728
Northern red-backed Salamander	**0.0470**	**0.0170**	0.157	0.108	**0.003**	0.266	**0.010**	0.677	-0.399	**-0.050**	-0.3610	**0.000**	**0.008**	0.384	0.274	**-0.040**
STAND CHARACTERISTICS																
Leaf litter	0.492	**0**	0.708	**0.001**	**0.002**	0.497	.	.	.	.	.	.	.	.	.	.
Needle litter	**0.012**	**0.086**	**0.025**	**0.028**	0.357	0.201	.	.	.	.	.	.	.	.	.	.
Coarse Woody Debris	0.544	0.202	**0.035**	0.349	0.075	0.405	.	.	.	.	.	.	.	.	.	.
Shrub cover	**0.013**	0.904	**0.003**	0.979	0.683	0.699	.	.	.	.	.	.	.	.	.	.
Moss cover	**0.022**	0.200	**0.081**	0.077	0.642	0.196	.	.	.	.	.	.	.	.	.	.
Canopy cover	**0.009**	**0.000**	**0.02**	**0.067**	**0.001**	**0.061**	.	.	.	.	.	.	.	.	.	.
% Hardwood	0.886	**0.000**	0.962	**<1e-04**	**<1e-04**	0.651	.	.	.	.	.	.	.	.	.	.
Tree density	**0.044**	0.535	**0.046**	0.799	0.415	0.259	.	.	.	.	.	.	.	.	.	.
Depth to water 0-10cm	0.763	**0.078**	0.548	0.165	**0.035**	0.56	.	.	.	.	.	.	.	.	.	.
Depth to water 50cm-10m	0.764	0.768	0.818	0.951	0.550	0.533	.	.	.	.	.	.	.	.	.	.

Table 5. Means for abundances of small mammals and herpetofauna and forest stand characteristics that were significantly influenced by an interaction between treatment and stand age. S1= stage 1 stands, coniferous trees with an average height between 3-5 metres, S2= stage 2 stands, coniferous trees with an average height of 10-20 metres. NR= Clear-cut and naturally regenerated; PCT= Clear-cut, naturally regenerated, and mechanically pre-commercially thinned; RP= clear-cut, planted with coniferous trees, and sprayed with herbicide.

	S1			S2		
	NR	**PCT**	**RP**	**NR**	**PCT**	**RP**
Total abundance	**68.5[a]**	**58.75[ab]**	**38.25[b]**	43.50	29.00	33.00
Herpetofauna abundance	10.25	21.50	12.00	**9.75[a]**	**3.00[b]**	**4.50[ab]**
Deer mouse	**0.67[a]**	**0.30[ab]**	**0.07[b]**	0	0.52	0.22
Wood frog	0.15	0.00	0.07	**0.97[a]**	**0.37[b]**	**0.00[b]**
Needle litter	0.56	0.92	0.66	**0.82[a]**	**2.77[b]**	**1.70[ab]**
Coarse woody debris	**0.83[a]**	**0.75[b]**	**1.66[b]**	0.98	1.57	1.10
Moss cover	0.60	0.86	0.54	0.86	2.51	1.42
Shrub cover	1.58	2.66	2.28	**1.63[a]**	**0.57[b]**	**1.34[ab]**
Canopy cover	**85.89[a]**	**36.63[b]**	**12.30[b]**	91.86	85.81	54.10
Density	1.04	0.85	0.86	1.03	1.28	1.02

FIGURES

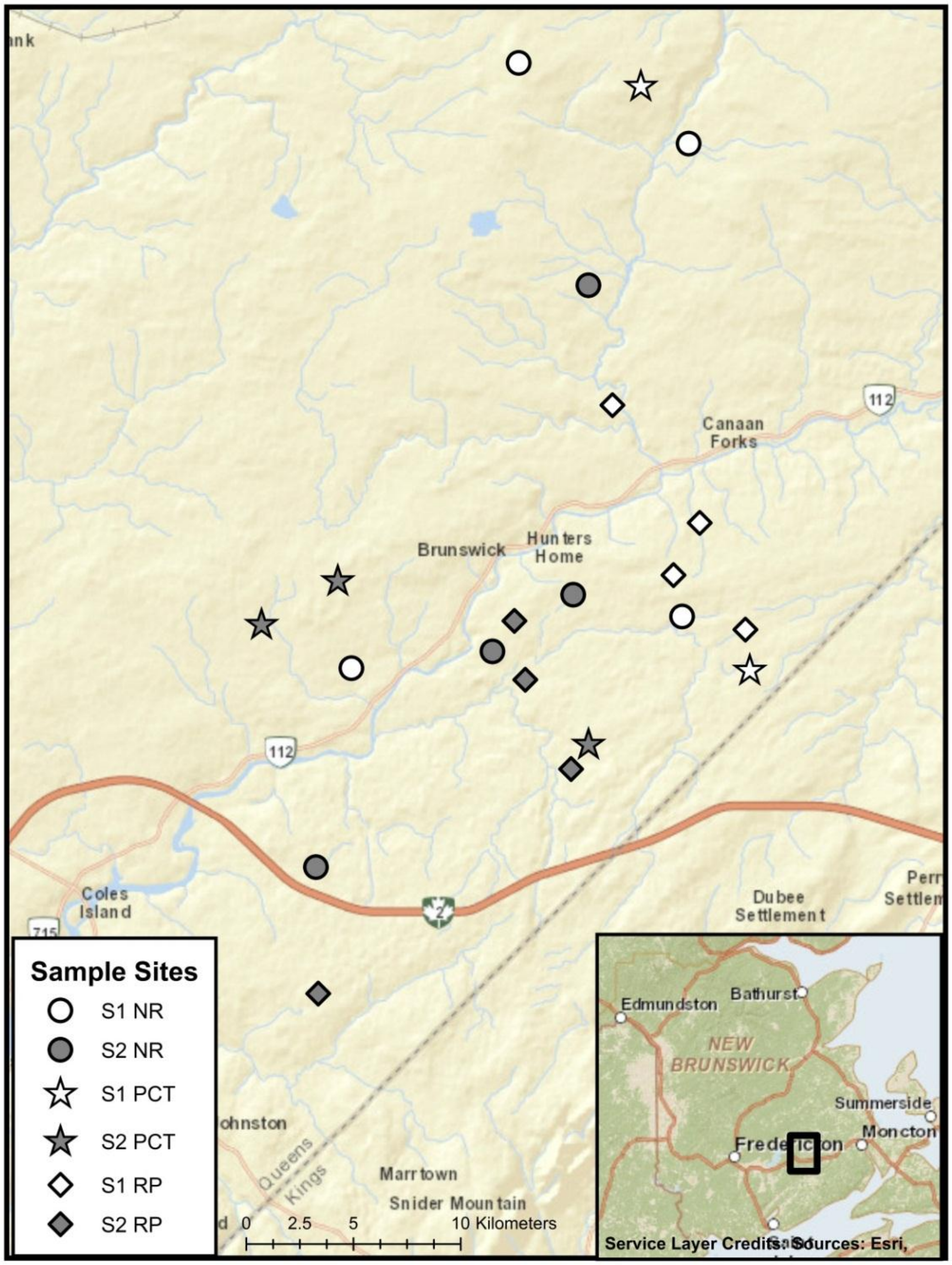


Figure 1. Sample sites in the Cole's Island Area of Central New Brunswick. S1.NR= Stage 1, naturally regenerated sites; S2.NR= Stage 2 naturally regenerated sites; S1.PCT= Stage 1 pre-commercially thinned sites; S2.PCT= Stage 2 pre-commercially thinned sites; S1.RP=Stage 1 plantation; S2.RP= Stage 2 plantation.

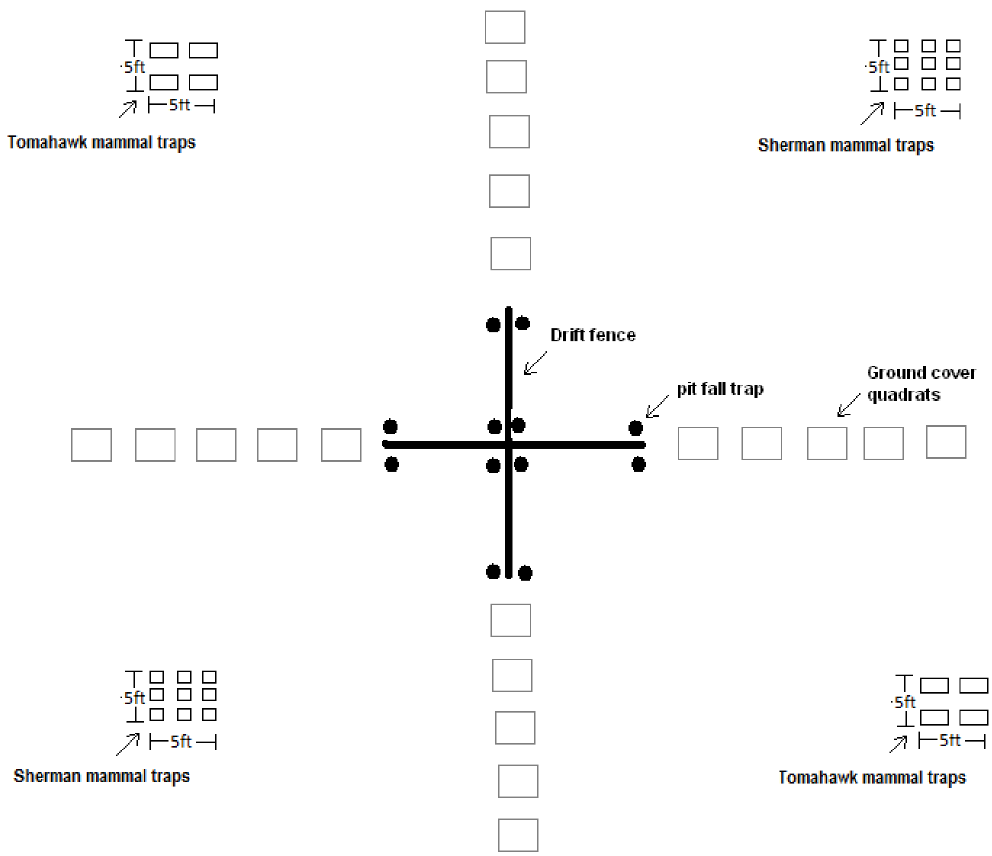


Figure 2. Drift fence pit fall trap, Sherman and Tomahawk live traps, and ground cover quadrats. Not to scale.

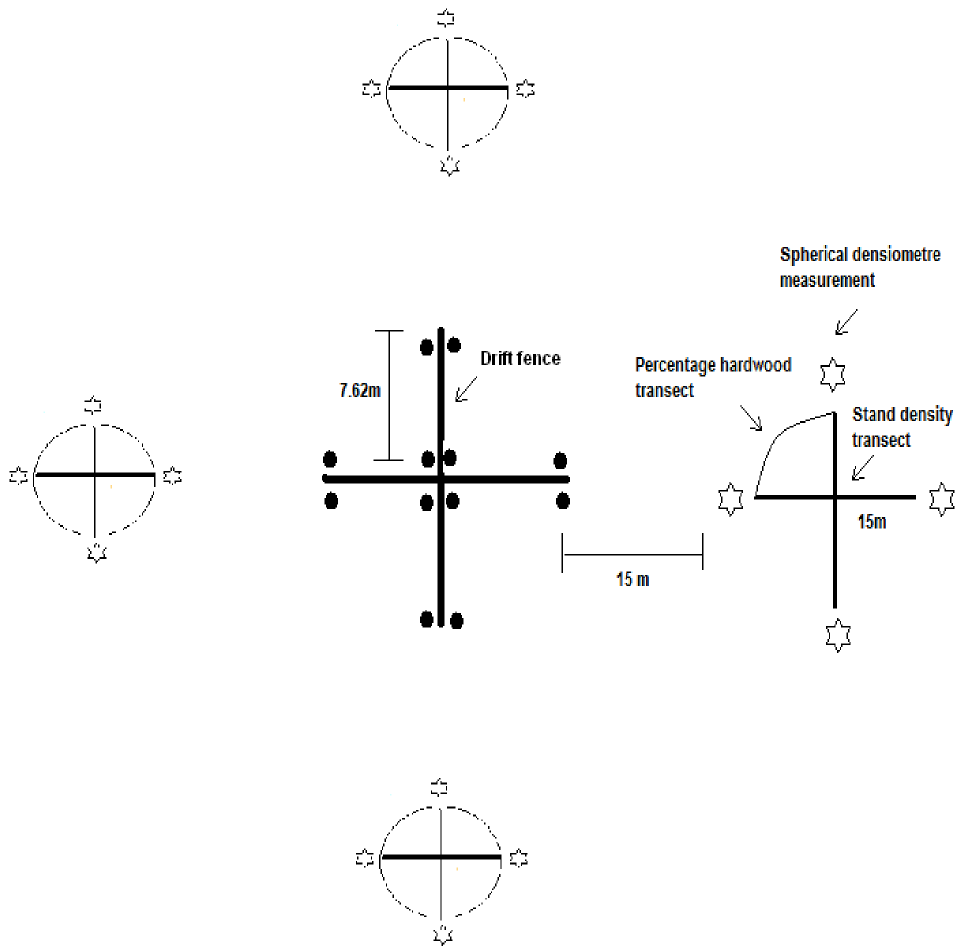


Figure 3. Drift fence pit fall trap, point quarter method used to describe stand density, and spherical densitometer used to describe canopy cover percentage. Not to scale.

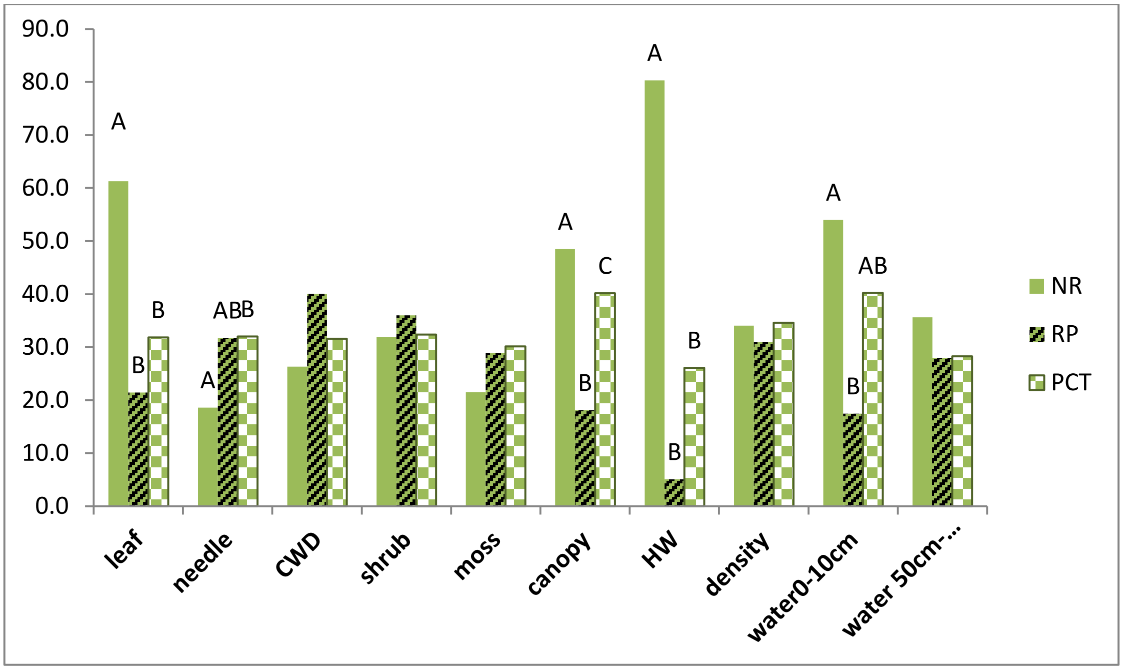


Figure 4. Forest stand characteristics within stand types. Bars are mean percentages. NR=Naturally regenerated stands; PCT=pre-commercially thinned stands; RP= planted stands; Leaf, needle, shrub, moss, CWD (coarse woody debris)= rating of cover within a 1m^2 quadrat (0 =none ,5= complete cover); Canopy=percentage of canopy cover; HW= percentage of hardwood; density=stand density; water 0-1cm= amount of water that was present between 0 and 10 cm from the surface; water 50cm-10m= amount of water that was present between 50cm and 10m from the surface.

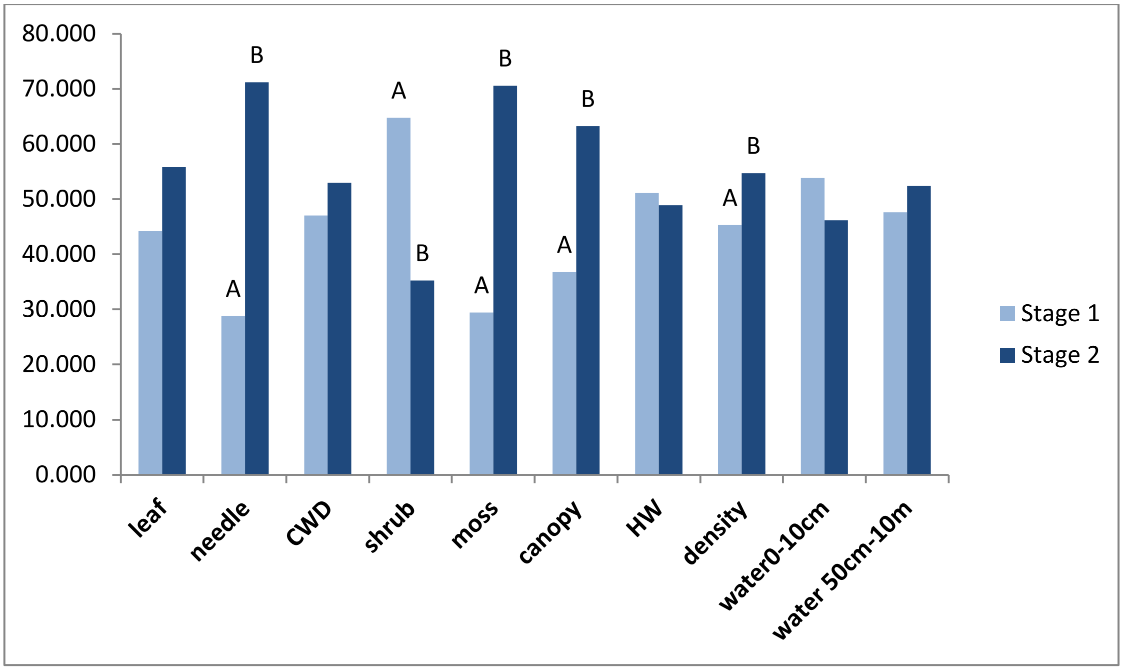


Figure 5. Mean percentage of forest stand characteristics within stage 1 and stage 2 stand development classes. Stage 1 stands had coniferous trees with an average height between 3-5 metres , whereas stage 2 stands had coniferous trees with an average height of 10-20 metres. Leaf, needle, shrub, moss, CWD (coarse woody debris) = rating of cover within a 1m^2 quadrat (0 = none , 5 = complete cover); Canopy = percentage of canopy cover; HW= percentage of hardwood; density=stand density; water 0-1cm = amount of water that was present between 0 and 10 cm from the surface; water 50cm-10m = amount of water that was present between 50cm and 10m from the surface.

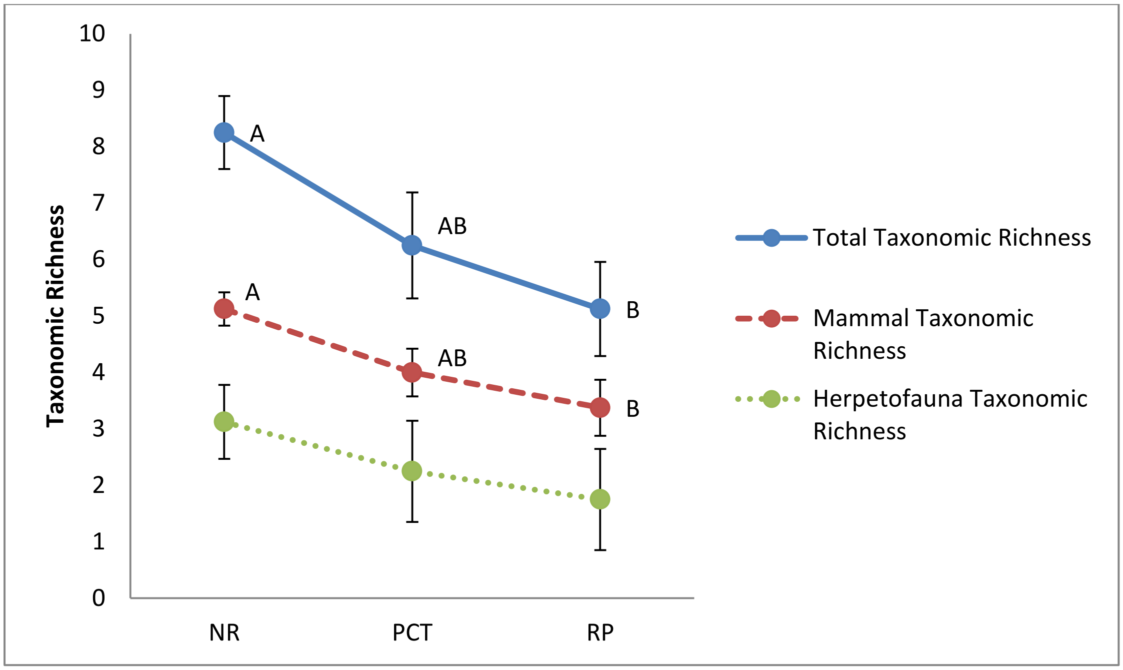


Figure 6. Taxonomic richness (# taxa) for total (small mammals and herpetofauna), small mammal, and herpetofauna with standard error, versus silviculture treatments. NR=naturally regenerated; PCT= pre-commercially thinned; RP=plantation. Values with the same letter are not significantly different (α=0.1).

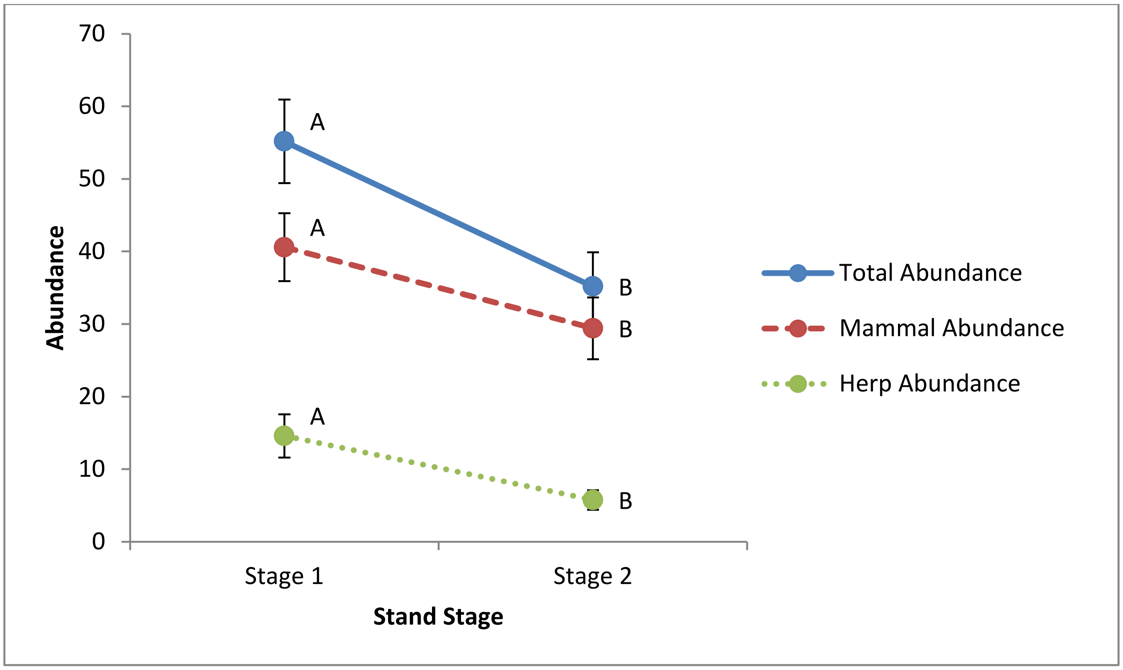


Figure 7. Abundance (# individuals): total (small mammal and herpetofauna), small mammal, and herpetofauna (mean ±standard error) versus stand stage. Letter symbols indicate statistically significant differences (α=0.1). Stage 1 stands had coniferous trees with an average height between 3-5 metres, whereas stage 2 stands had coniferous trees with an average height of 10-20 metres.

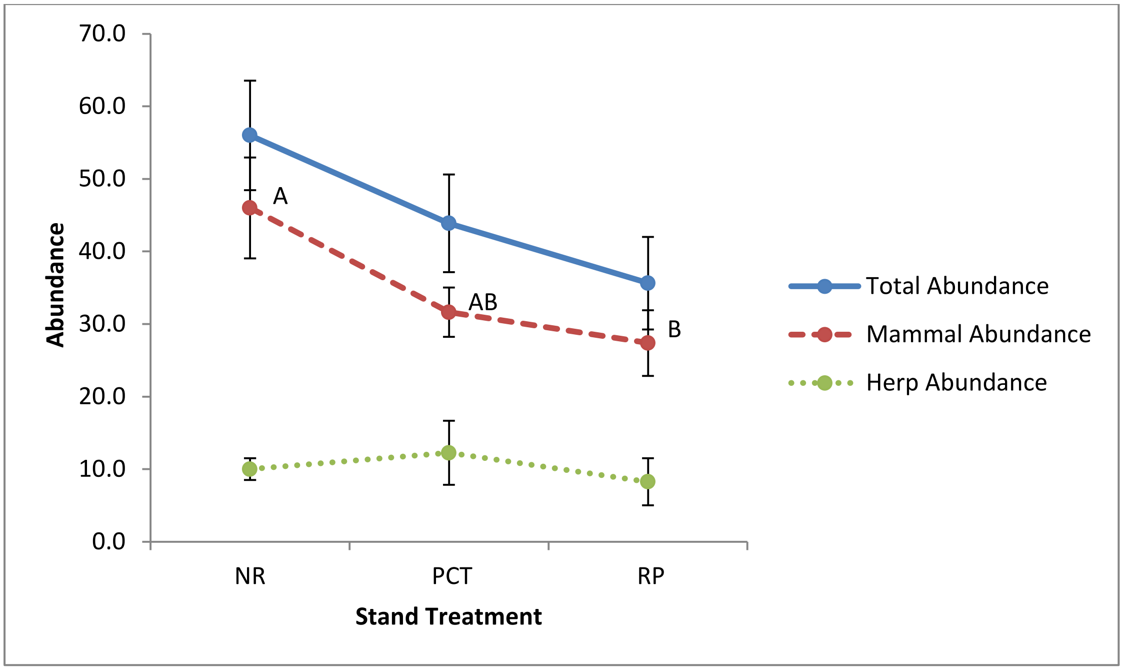


Figure 8. Abundance (# individuals): total (small mammal and herpetofauna), small mammal, and herpetofauna (mean ± standard error) versus silviculture treatments. Letter symbols indicate statistically significant differences (α=0.1). NR=naturally regenerated; PCT= pre-commercially thinned; RP=plantation.

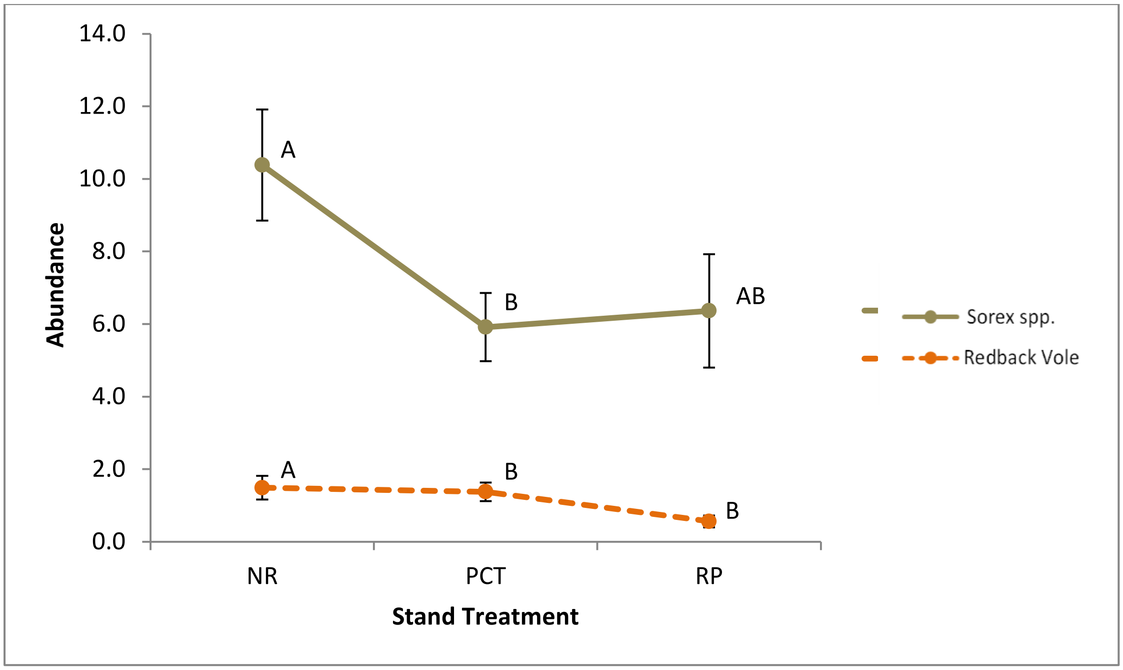


Figure 9. Abundances of *Sorex* spp. and red-backed vole (mean ± standard error) versus stand treatment. Letter symbols indicate statistically significant differences (α=0.1). NR=naturally regenerated; PCT= pre-commercially thinned; RP=plantation.

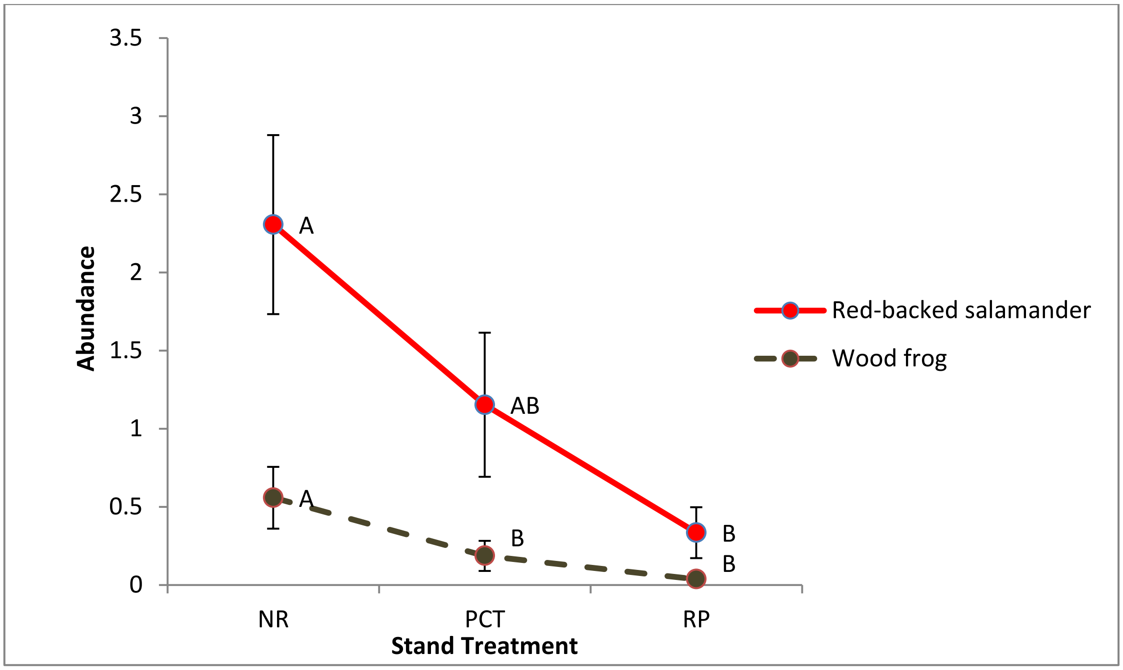


Figure 10. Abundances of northern red-back salamander and wood frog (mean ± standard error) versus stand treatment. Letter symbols indicate statistically significant differences (α=0.1). NR=naturally regenerated; PCT= pre-commercially thinned; RP=plantation.

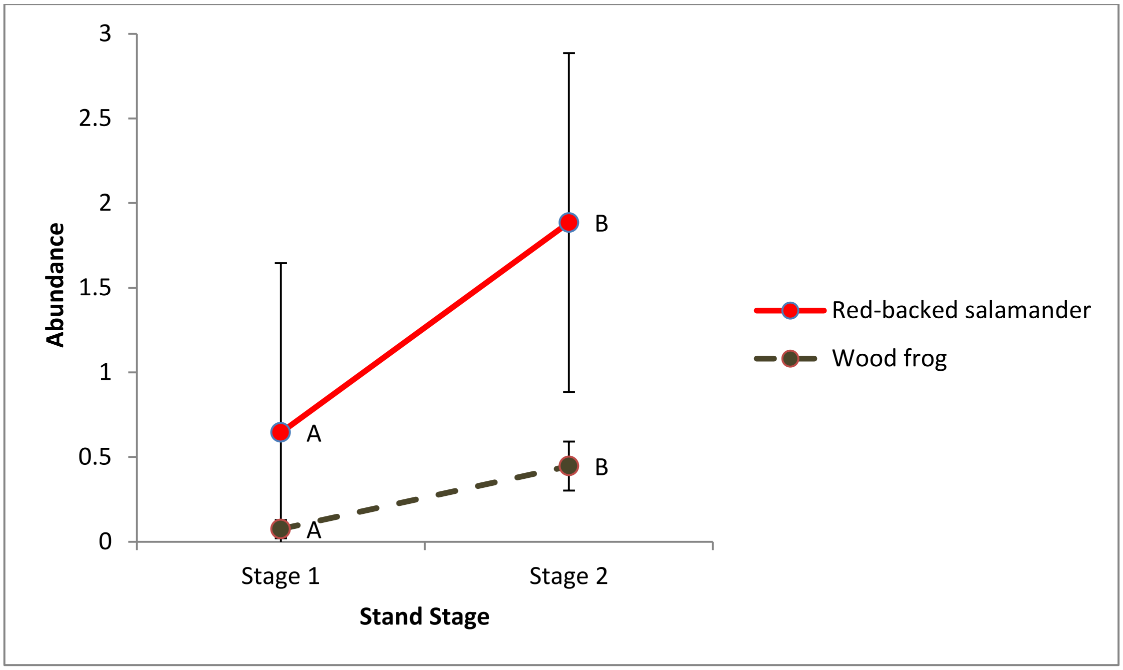


Figure 11. Abundances of northern red-back salamander and wood frog (mean $\pm$ standard error) versus stand development stage. Letter symbols indicate statistically significant differences (α=0.1). Stage 1 stands had coniferous trees with an average height between 3-5 metres, whereas stage 2 stands had coniferous trees with an average height of 10-20 metres.

APPENDICES

Appendix Table 1. Individual forest stand sizes, overall forest stand type, year of establishment, treatment year (if any) as provided by J.D. Irving Ltd., and distance from sample sites to nearest forest stand edge. S1= stage 1 stands, had coniferous trees with an average height between 3-5 metres, whereas S2= stage 2 stands, stands had coniferous trees with an average height of 10-20 metres. NR= Clear-cut and naturally regenerated; PCT= Clear-cut, naturally regenerated, and mechanically pre-commercially thinned; RP= clear-cut, planted with coniferous trees, and sprayed with herbicide; HS=Hardwood dominated mixed-wood stand; SW= predominantly softwood stand; SH=softwood dominated mixed-wood stand; na= information not available. * indicates that there was a discrepancy between the geographic information system data provided by the New Brunswick Department of Natural Resources, and the land manager J.D. Irving, Ltd.. In these instances I deferred to the information provided by J.D. Irving Ltd.

Stand age	Stand treatment type	Stand size (m^2)	Overall stand type	Year Clear Cut	Treatment year	Fence 1- distance to nearest edge (m)	Fence 2- distance to nearest edge (m)
YOUNG	NR	152520.8	HS	1979	na	109	104
		235198.7	SW	1985	na	40	50
		211805.7	SH	1988	na	94	70
		127157.2	HS	1984	na	71	41
	PCT	258768.4	SW	1989	2000	74	42
		150052.1	SH	1989	2000	35	40
		166259.5	SW	1985	2000	93	68
		174665.3	SW	1993	2000	118	87
	RP	240775.6	SH	1993	1993	69	53
		137910.9	SH	1993	1993	156	35
		104160	SH	1994	1994	63	53
		105654.6	SH	1995	1995	59	47
OLD	NR	218771	SH	1967	na	120	264
		32911.41	UN	na	na	68	54
		144771.5	HW	1927	na	33	99
		529217.5	SH	1957	na	105	77
	PCT	134959.6	SW	1982	1982	81	66
		139129.6	SW	1980	1995	215	191
		503131.3	SH	1988	1993	48	37
		163140	SW	1979	1993	43	41
	RP	178590.2	SH	1988	2001	48	33
		142957.4	SW	1988	1988	58	34
		410190.1	SH	1987	1998	62	71
		77034.6	SH	1985	1985	98	40

Appendix Table 2. Forest stand types and ages within a 150 metre buffer surrounding sampled stands as provided by J.D. Irving, Ltd. Stand types include: RD= road, S= softwood, and softwood dominated mixed-wood stand, H= shade intolerant and tolerant harwood, and hardwood dominated mixed-wood stand, UN= unknown stand type, WL= wetland, RV= river, AP=alders growing on a plain, GP= gravel pit, OC=occupied habitat. Stand stage categories are:Y= young stands (0-10 years), I= immature stand (10-20 years), and M=mature stand (20-40 years), including also anything that was considered 'overmature' (40+ years). S1= stage 1 stands, had coniferous trees with an average height between 3-5 metres , whereas S2= stage 2 stands, stands had coniferous trees with an average height of 10-20 metres. NR= Clear-cut and naturally regenerated; PCT= Clear-cut, naturally regenerated, and mechanically pre-commercially thinned; RP= clear-cut, planted with coniferous trees, and sprayed with herbicide.

Stand age	Stand treatment type	Buffer stand type	Buffer Stand Age	Proportion (%) of buffer	Stand age	Stand treatment type	Buffer stand type	Buffer Stand Age	Proportion (%) of buffer
young	NR	RD	N/A	10	old	NR	H	M	28
		S	M	90			RD	N/A	4
young	NR	H	M	34			S	Y	3
		RD	N/A	2			S	I	2
		S	I	14			S	M	32
		S	M	35			UN	Y	21
		UN	Y	12			UN	M	3
		UN	I	2			WL	N/A	7
young	NR	H	I	7	old	NR	H	I	1
		H	M	43			H	M	0
		RD	N/A	3			RD	N/A	0
		S	I	20			S	Y	1
		S	M	26			S	M	96
young	NR	H	I	0			UN	Y	1
		H	M	65	old	NR	H	M	22
		RD	N/A	6			RD	N/A	14
		S	Y	11			S	Y	15
		S	M	17			S	I	1
young	PCT	H	I	1			S	M	47
		H	M	7	old	NR	IH	M	0
		RD	N/A	6			RD	N/A	5
		S	Y	2			RV	N/A	2
		S	I	51			S	Y	25
		S	M	24			S	I	0
		RD	N/A	8			S	M	48
		UN	I	1			UN	Y	13
young	PCT	H	M	0			UN	I	4
		RD	N/A	8			WL	N/A	3
		S	Y	14	old	PCT	AP	N/A	10
		S	I	70			RD	N/A	8
		S	M	0			S	Y	6
		UN	Y	8			S	I	5
		WL	N/A	0			S	M	71
young	PCT	RD	N/A	1	old	PCT	H	I	8
		S	y	24			H	M	20
		S	I	24			RD	N/A	4

		S	M	26			S	I	53
		UN	Y	15			S	M	16
		WL	N/A	9			UN	I	0
young	PCT	H	M	3	old	PCT	GP	N/A	1
		RD	N/A	5			RD	N/A	11
		S	Y	5			S	M	81
		S	I	43			WL	N/A	7
		S	M	31	old	PCT	RD	N/A	7
		UN	y	0			S	Y	3
		UN	I	5			S	I	22
		WL	N/A	7			S	M	69
young	RP	H	I	4	old	RP	H	M	6
		H	M	1			RD	N/A	6
		RD	N/A	4			S	Y	0
		S	Y	39			S	I	54
		S	I	35			S	M	9
		S	M	14			UN	Y	5
		UN	Y	2			WL	N/A	19
young	RP	H	I	7	old	RP	H	O	16
		H	M	6			OC	N/A	12
		RD	N/A	6			RD	N/A	7
		S	Y	47			S	Y	0
		S	I	21			S	I	20
		S	M	10			S	M	24
		UN	N/A	3			UN	Y	12
young	RP	I	M	21			WL	N/A	10
		RD	N/A	6	old	RP	H	M	20
		S	Y	8			RD	N/A	7
		S	I	15			RV	N/A	3
		S	M	35			S	Y	1
		UN	Y	1			S	I	34
		WL	N/A	14			S	M	25
young	RP	H	M	54			WL	N/A	10
		RD	N/A	19	old	RP	H	M	4
		S	I	4			RD	N/A	9
		S	M	23			S	I	45
							S	M	42

Appendix Table 3. Mean and standard deviation of forest stand characteristics within the two stand development stages. S1= stage with an average coniferous tree height between 3-5 metres, S2=stands with an average coniferous tree height of 10-20 metres. S1= stage 1 stands, had coniferous trees with an average height between 3-5 metres , whereas S2= stage 2 stands, stands had coniferous trees with an average height of 10-20 metres.

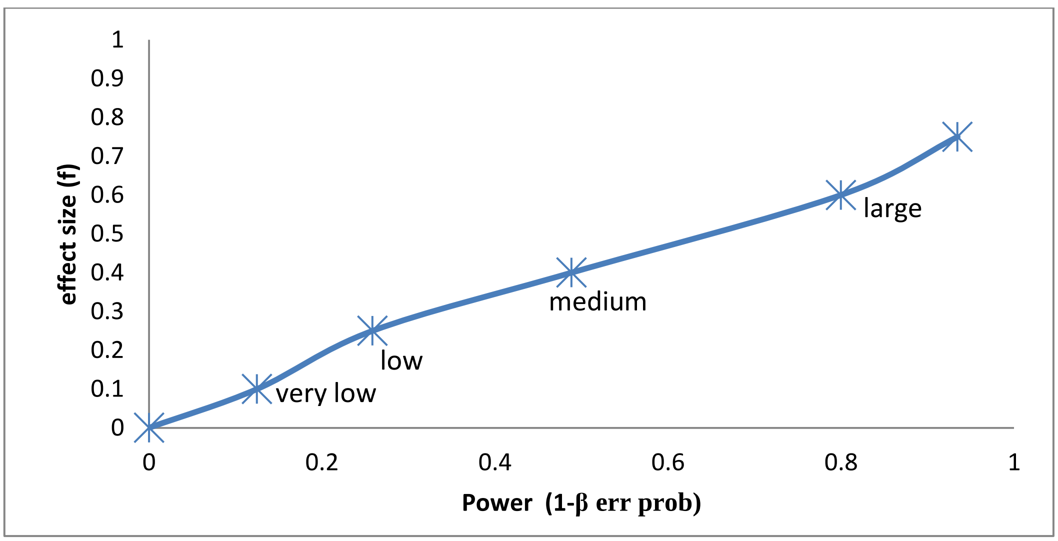
	S1		S2	
	mean	standard deviation	mean	standard deviation
leaf	1.82	1.60	2.20	1.70
needle	0.71	0.75	1.36	1.09
CWD	1.08	0.58	1.09	0.67
Shrub	2.17	0.90	1.18	0.76
Moss	0.67	0.52	1.55	1.28
Canopy	44.94	33.53	77.29	18.22
Percent Hardwood	27.63	36.54	25.86	29.11
Density	0.92	0.17	1.10	0.24

Appendix Table 4. Correlation matrix of independent forest stand characteristic variables measured at each site. Leaf litter=proportion leaf litter; needle litter=proportion needle cover; Coarse woody debris= proportion of coarse woody debris; shrub cover= proportion shrub cover; moss cover=proportion moss cover; canopy cover= percent canopy cover; density= stand density around trapping sites;depth to water 0-10cm= percent water from 0-0.1 m underground; depth t water 50 cm-10 m=percent water from 0.5-10m underground.). *** indicates a p-value of less than 0.01, ** indicates a p-value of less than 0.05, and * indicates a p-value of less than 0.1.

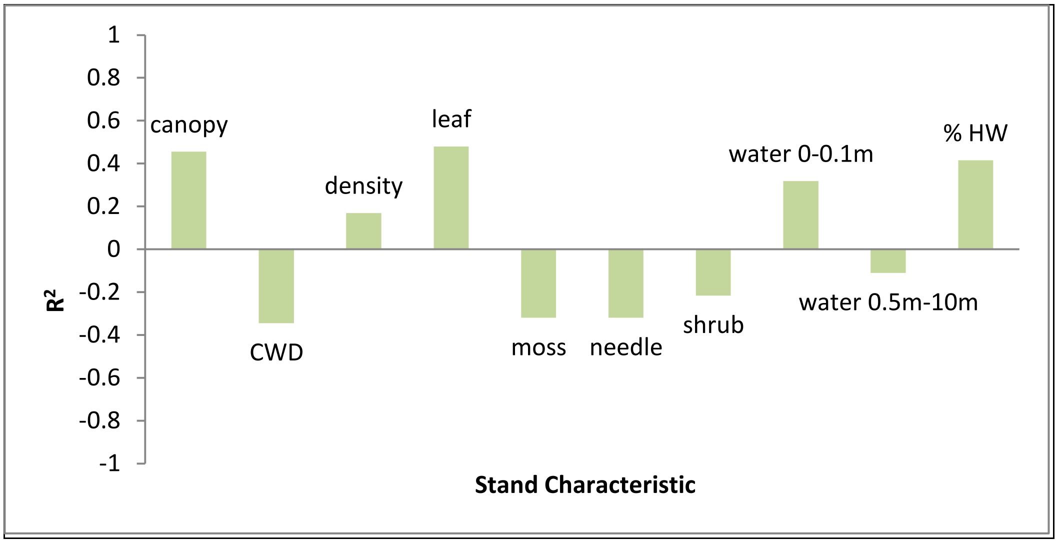
Dependent variable	Leaf litter	Needle litter	Coarse woody debris	Shrub cover	Moss cover	Canopy cover	% hardwood	Density	Depth to water 0-10cm	Depth to water 50cm-10m
Leaf litter	1									
Needle litter	**-.416****	1								
Coarse woody debris	-0.343	0.28	1							
Shrub cover	-0.004	-0.33	**-0.394***	1						
Moss cover	-0.319	**.689*****	0.261	-0.194	1					
Canopy cover	**.565*****	0.123	-0.178	**-.542*****	0.218	1				
% hardwood	**.889*****	**-.485****	-0.325	-0.069	-0.303	**.658*****	1			
Density	-0.068	0.309	0.066	**-.632*****	**0.355***	**.505****	0.013	1		
Depth to water 0-10cm	0.22	0.048	0.192	-0.322	0.123	**.455****	0.303	0.211	1	
Depth to water 50cm-10m	-0.119	0.173	0.071	-0.012	**.558*****	0.211	-0.068	**0.361***	0.341	1

Appendix Table 5. P values of small mammal and herpetofauna taxonomic richness relationships with stand stage and treatment with all species captured, and excluding rare captures of less than 20 individuals. P values in bold are less than 0.1. TR= taxonomic richness.

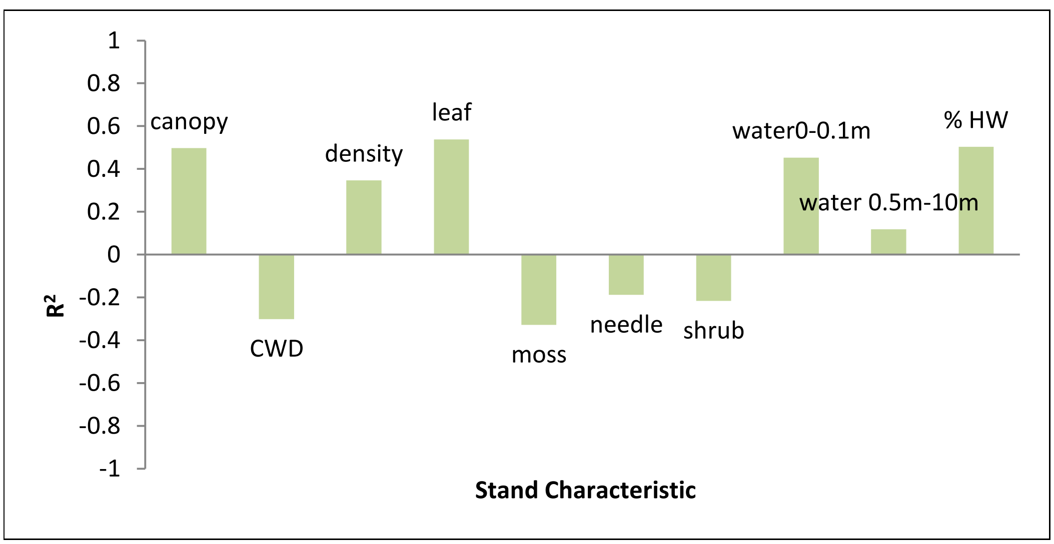
	Dependent Variable	Randomization		Interaction	Post Hoc		
		Stage	Treatment	Stage&Treatment	PCT-NR	RP-NR	RP-PCT
All species captured	Total TR	0.528	**0.042**	0.924	0.154	**0.028**	0.448
	Small Mammal TR	1.000	**0.018**	0.859	0.128	**0.014**	0.410
	Herpetofauna TR	0.330	0.214	0.713	0.300	0.097	0.592
Capture rates over 20 individuals	Total TR	0.568	**0.025**	0.339	**0.051**	**0.011**	0.626
	Small Mammal TR	0.758	0.128	0.151	0.148	0.177	1.000
	Herpetofauna TR	0.970	**0.038**	0.258	0.192	**0.021**	0.424



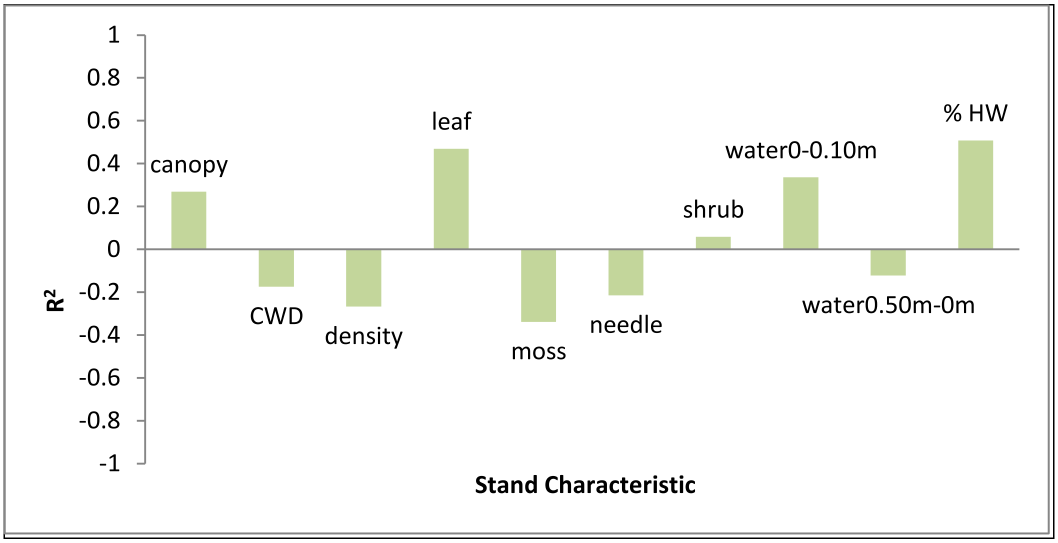
Appendix Figure 1. Post hoc power analysis, effect sizes that can be expected based on various statistical power levels. Effect size (f), or Cohen's f is defined as $f=\sigma_{means}/\sigma$ and power is defined as 1=β error of probability.



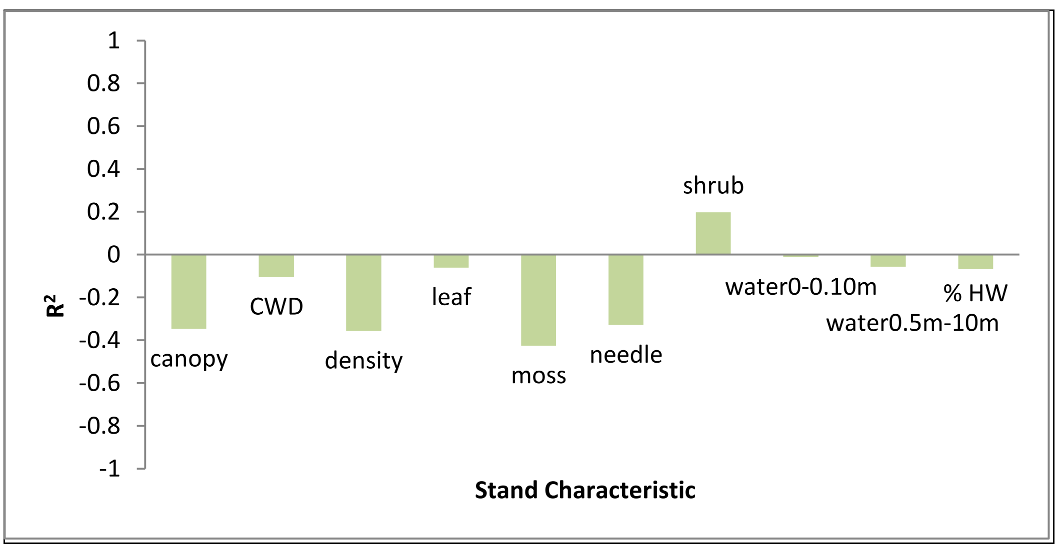
Appendix Figure 2. Plot of univariate relationships between forest stand characteristics and total small mammal and herpetofauna taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1m= percent ground water from 0-0.1 m; water 0.5m-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



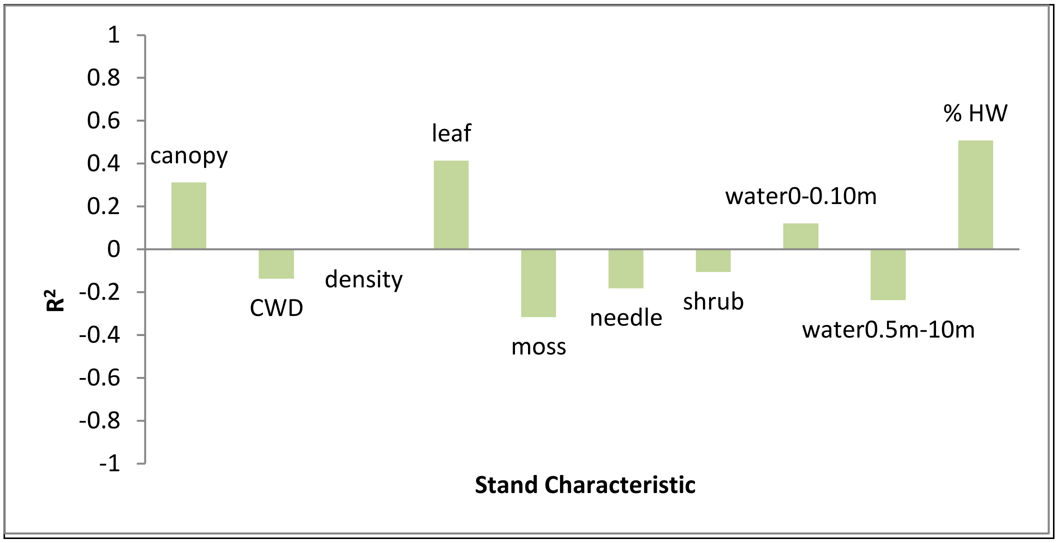
Appendix Figure 3. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



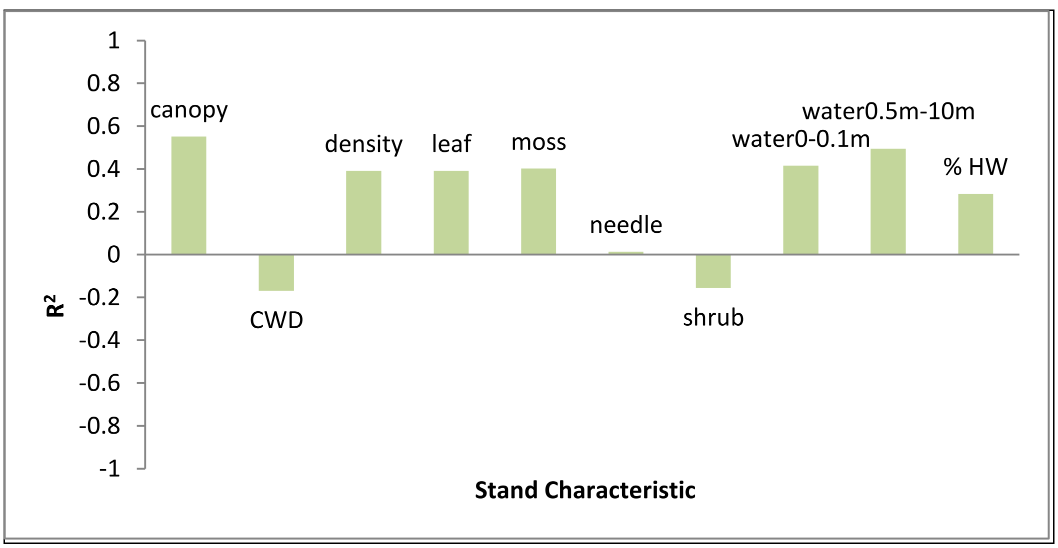
Appendix Figure 4. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



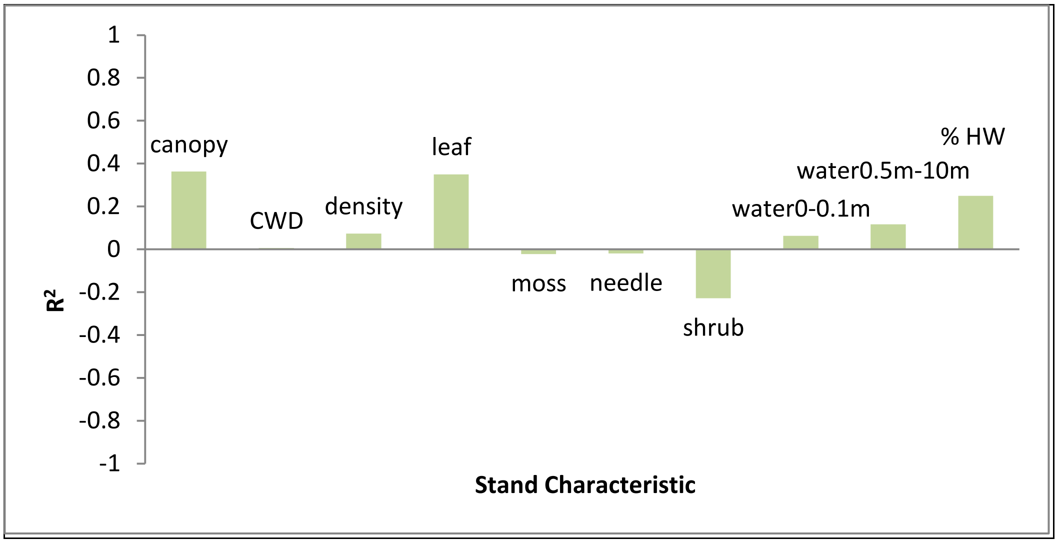
Appendix Figure 5. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



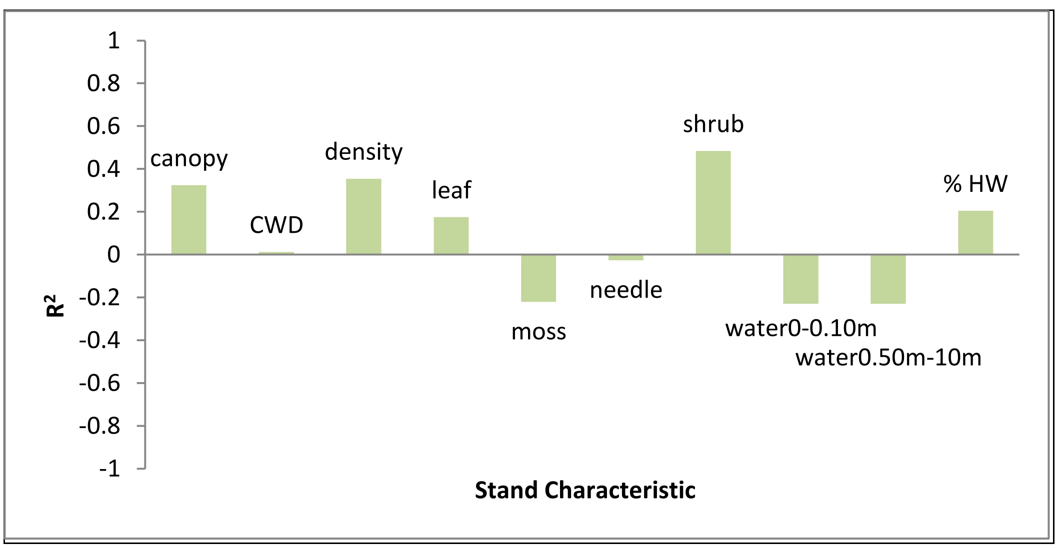
Appendix Figure 6. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



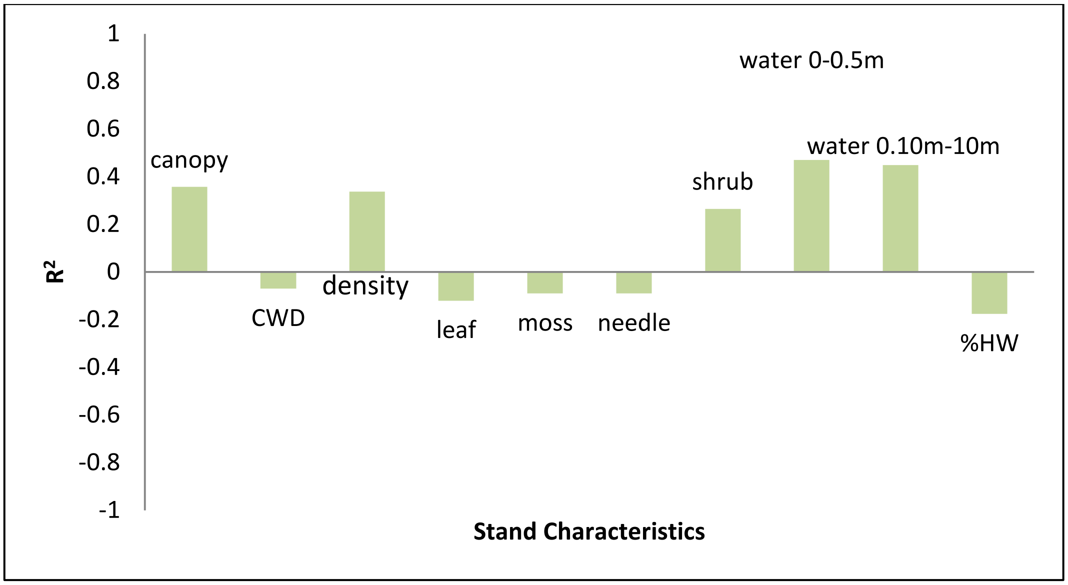
Appendix Figure 7. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



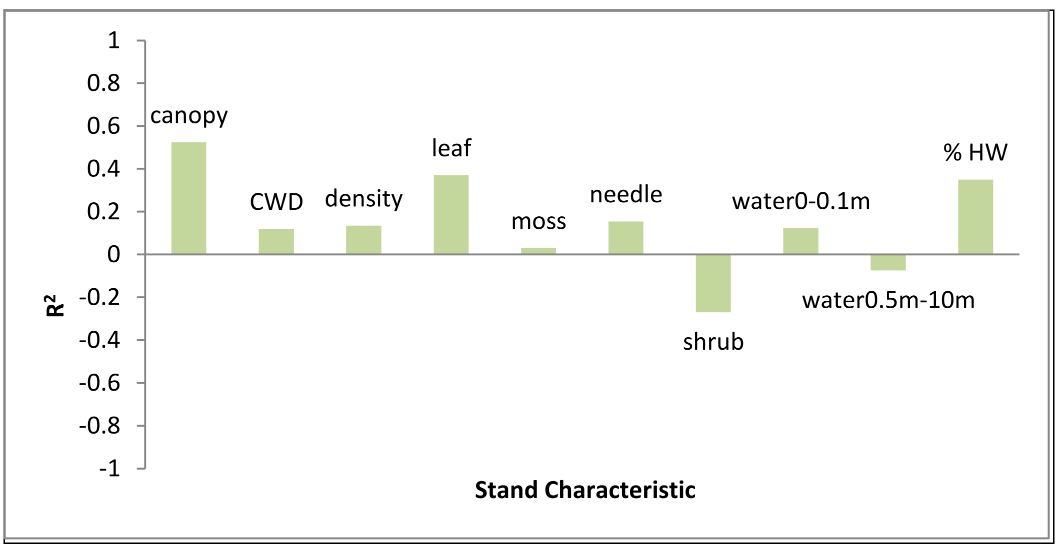
Appendix Figure 8. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



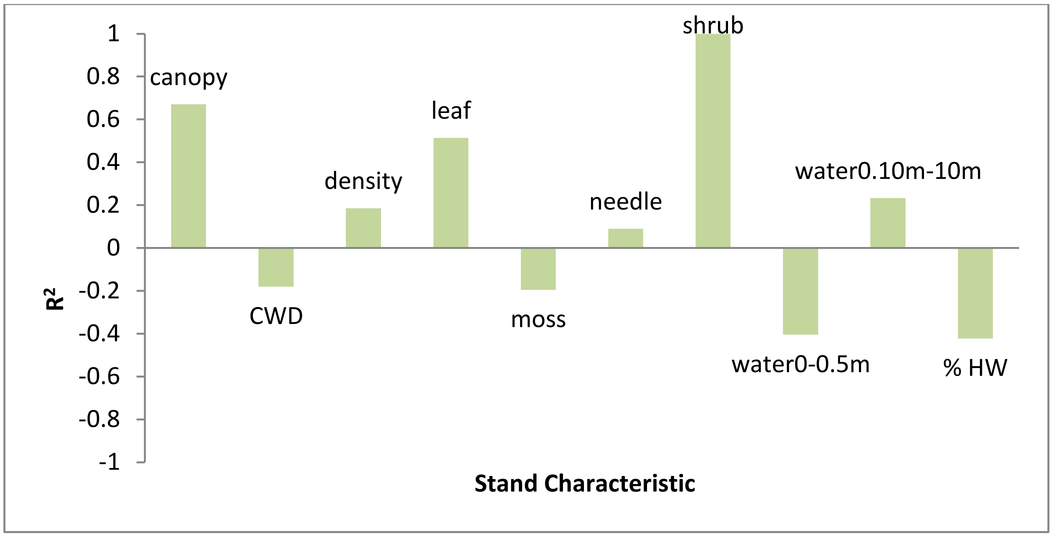
Appendix Figure 9. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



Appendix Figure 10. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



Appendix Figure 11. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.



Appendix Figure 12. Plot of univariate relationships between forest stand characteristics and small mammal taxonomic richness. canopy= percent canopy cover; CWD= proportion of coarse woody debris; density= stand density around trapping sites; water 0-0.1cm= percent ground water from 0-0.1 m; water 10cm-10m =percent ground water from 0.5m-10m; leaf=proportion leaf litter; moss=proportion moss cover; needle=proportion needle cover; shrub= proportion shrub cover.

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