

Effects of clear-cut created ecotones on nocturnal macromoths in western Newfoundland,
Canada

by

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A Thesis submitted in partial fulfillment
of the requirements for the degree of
Master of Science

Boreal Ecosystems and Agricultural Sciences, Grenfell Campus

Memorial University of Newfoundland

May 2020

Corner Brook

Newfoundland

Grenfell Campus
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The undersigned certify that they have read, and recommend to the Boreal Ecosystems and Agricultural Science Unit (School of Graduate Studies) for acceptance, a thesis entitled “Effects of clear-cut created ecotones on nocturnal macromoths in western Newfoundland” submitted by Jasmine R. Pinksen in partial fulfillment of the requirements for the degree of Master of Science.



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May 7, 2020

Acknowledgements

Thank you to the many people who assisted with the strenuous field work necessary to collect data for this project: Lorne Pinksen, Tammy Pinksen, Nathan Pinksen, David Alexander, Dr. Joe Bowden, Dr. Julie Sircom, Jens Fiegler, and Inga Borisenoka. Thank you to the Canadian Forest Service, NRCAN, for allowing me to use the equipment necessary for data collection. Thank you to Heather Spicer and Katherine Casey, CFS, for various research planning, including maps of the logging areas around Camp 180 Resource Road, NL. Thank you to Jim Edsall who assisted in moth species identification. Special thank you to my supervisors Dr. Julie Sircom and Dr. Joe Bowden for their feedback and support.

Abstract

Human land use practices, especially clearing of native vegetation, tend to increase edges, creating ecotones. Ecotones are important transition zones for conservation, as they are species rich and promote biological diversity, thus they have become a focus of habitat management. Arthropod groups are known to be significantly more abundant and/or taxonomically diverse at the edges compared to the interior of the adjacent habitats. There is, however, a knowledge gap with respect to lepidopteran diversity across anthropogenic ecotones in boreal forests.

To address one of these knowledge gaps, I asked how forestry-created ecotones influence nocturnal macromoth assemblages in the eastern boreal forest. I sampled moths in western Newfoundland, Canada, to determine their responses at the local spatial scale. Sampling took place in four replicate sites, each with light traps placed in a clear-cut, adjacent forest, and the edge between them. Edges supported the highest abundance of nocturnal macromoths, forests supported an intermediate amount, and clear-cuts supported the lowest. The difference between the habitat that supported the highest abundance and the habitat that supported the lowest, i.e. edges and clear-cuts, was significant. While boasting many unique species, edges represented a macro-moth assemblage composition that was intermediate between clear-cuts and forests, and all habitats differed significantly. While composition differed significantly between the three habitats, there was no significant difference in species diversity among the three habitat types. The results of my study suggest that, like natural disturbance created edges, forest-clear-cut edges may serve as unique habitats for macro-moth assemblages in the boreal

forest. These results also have important implications for the maintenance of biodiversity in the boreal and forestry in the context of natural disturbance emulation.

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Literature Review

Introduction

Forests provide a plethora of valuable natural resources for people, but also serve as important habitats that sustain biodiversity and ecosystem functioning. While over 70% of Canada's managed forests are third party certified (SFMC, 2016), understanding forest insect biodiversity may provide another piece of valuable insight into forest management and the maintenance of biodiversity in the boreal. In order to help inform meaningful sustainable forest management strategies, we need to understand the impacts of timber harvest methods on patterns of biodiversity, which include terrestrial forest arthropods (Simberloff, 1999). Quite often we see that forest management plans do not place priority on monitoring non-pest insect species (e.g. Morsek, 2001). Insects, however, are one of the most diverse and critical components of forest ecosystems (Stork, 1988), serving important functions as predators of irruptive or non-native insects, herbivores of various plants, food for many migratory birds, bats and other mammals, nutrient cycling, and as pollinators for many plant species in the forest. Wild pollinator populations are declining (e.g., Potts et al., 2010; Soroye et al. 2020), and thus have increased the interest in pollinator conservation plans within the management of human-modified landscapes (Steinert et al., 2020). Since pollinators are critical for global food security and human health (Klein et al., 2007; Gallai et al., 2009), understanding how they are affected by forest management practices is important.

Lepidopterans, while important as forest pollinators, also serve critical roles as herbivores, as detritivores, and as prey for many birds and bats (Ober & Hayes, 2010). While there are many past studies showing effects of timber harvest on various taxa,

including Lepidoptera (Sillett, 1994; Gehlhausen et al., 2000; Magura, 2002; Esseen, 2006; Summerville & Crist, 2008), little attention has been given specifically to how edge affect Lepidoptera in the boreal forest. Some studies have shown that timber harvest may actually have benefits to certain insect taxa (Didham, 1997; Heliölä et al., 2001; Magura, 2002; Mathe, 2006; Steinert et al., 2020), including Lepidoptera (Bergman, 1999; Viljur & Teder, 2016; Williams & Jonusas, 2019). More specifically, this trend has been documented in macromoth families within the order Lepidoptera (Fluentes-Montemayor et al., 2012). Yet, there is little understanding of how these clear-cut-forest ecotones influence macromoth communities.

Forest practices can affect lepidopteran communities, resulting in trophic consequences due to diminished prey availability or shifts in consumers of primary producers (Summerville, 2011). For example, reduced lepidopteran populations could have knock-on effects for forest birds and bats, and limit shifts in consumer richness needed for primary producers. One way we can test how harvesting might affect moth assemblages is through multi-site monitoring in a replicated design that includes forests, harvested plots and the intermittent transition zone (edge). Edges between adjacent habitats contain characteristics of both habitats; therefore, the microclimate in edges may support species from both habitats. This may lead to higher insect abundances and/or greater insect species diversity (Magura, 2002; Mathe, 2006). In temperate and tropical forests, edges have been well studied, but less so in less productive, disturbance-adapted boreal forests (Harper et al., 2015). Given that managed conifer forests are essential to produce wood and fibre needed by society (Carle & Holmgren, 2008; FOA, 2016),

studying insect populations along forest edges can help in developing better forest management plans for insect conservation.

Biodiversity

The almost 1 billion hectares of circumboreal forest accounts for nearly one quarter of the world's closed canopy forest and contains half of the world's unexploited frontier forests (Burton et al., 2010). Canada accounts for 12% of the global boreal biome, with over 53% of the country's area in the boreal zone (Burton et al., 2010). Boreal forests are generally considered to have low species richness. This is likely because they are located at high latitudes, having cool climates (DeGrandpre et al., 2003). In North American boreal forests, wildfire, insect outbreaks, and forest harvest (in the form of large-scale clear-cutting) represent major disturbances (Bergeron et al., 2002; Rees & Juday, 2002). The idea of developing forest management strategies that attempt to emulate natural disturbance has long been discussed by forest managers and scientists (Hunter, 1993; Haila et al., 1994; Wuerther, 1995; Bergeron et al., 2002, Venier et al. 2017).

Recently, from the perspective of biodiversity conservation, boreal forests have been considered as quite unique with respect to the important role that disturbance plays in maintaining biodiversity (Bradshaw et al., 2009). All stages of forest succession are important for maintaining biodiversity and need to be considered for conservation management purposes. Old-growth boreal forests maintain a diverse understory community that is not present in younger stands, however, early and intermediate stages of forest development are just as significant to biodiversity conservation (Bergeron & Fenton, 2012). Forest managers should not focus exclusively on timber harvest when developing management plans. The vitality and health of forests must also be considered

along with other factors such as biological diversity, environmental protection and social benefits (e.g. Degraaf & Miller, 1996; Kolm & Franklin, 1997), *sensu* a holistic consideration. In order to ensure the long-term survival of naturally occurring species, in viable populations, we must be able to sustain the forest ecosystem and maintain its biodiversity (Angelstam, 1998).

Biodiversity changes, for a range of taxa, have been linked to timber harvest in eastern North American forests (Lousier, 2000). Given the value of biodiversity in providing numerous ecosystem services (Winfrey et al., 2007), biodiversity conservation needs to be considered as a component of forest management. Fortunately, forest biodiversity and critical ecosystem services have been known to be supported in forested areas that have undergone management for wood production (Hartley, 2002; Paquette & Messier, 2010; Irwin et al., 2014). This is likely due to the openness, light distribution, humidity, temperature, wind, and overall microclimatic stability of recently harvested stands (Forman, 1995; Larrivéé et al., 2005). Each biotic characteristic can have different effects on different species, so two adjacent habitats with different biotic characteristics, and an intermediate edge habitat may support greater biodiversity (Kuussaari et al., 2007).

Boreal forests harvested by clear-cutting that result in open habitats are potentially suitable for species that thrive in open landscapes (Viljur & Teder, 2016). These areas also have an increase in floral resources, nectar and pollen availability, and nesting substrates that are required for sustaining wild pollinators (Korpela et al., 2013; Taki et al., 2013; Rubene et al., 2015; Hanula et al., 2016). Although recent literature has indicated that some managed forests support wild pollinators (timber harvest can

positively affect abundance and diversity, Taki et al., 2013; Hanula et al., 2016), much remains to be learned about the diversity of pollinators which inhabit these managed forest areas (Rivers et al., 2018). Forests adjacent to open fields serve important roles as shelter and dispersal routes to other suitable habitats for insects (Merckx et al., 2010; Bailey et al., 2014), which can be essential for specialist species (Kleijn et al., 2011). Hence, well-planned logging that creates clear-cut-forest ecotones could support biodiversity.

Lepidoptera

Danks (1988) estimated over 6000 species of Lepidoptera in Canada, with only 4192 species actually described. Globally, the number of described Lepidoptera exceeds 160,000 species, 95% of which are moths (Scoble, 1995; New, 2004; Kristensen et al., 2007). The diversity of both moths and butterflies decreases latitudinally, with lower diversity at high latitudes (Kerr et al., 1998). Yet, in northeastern forests in North America, moths are species-rich, with the most diverse families being Noctuidae and Geometridae (Summerville & Crist, 2008). Moths are one of the most functionally important taxa in forests, with species playing important roles as herbivores, detritivores, pollinators, prey for birds and bats, and aiding in nutrient cycling (Holmes et al., 1979; Summerville & Crist, 2002; Fox, 2013; Summerville & Marquis, 2017).

Forest management activities can affect moth diversity and composition as vegetation structure is a good predictor of macromoth assemblages (e.g., Ober & Hayes, 2010; Highland et al. 2013). Within a single eco-region, moth community structure differs based on tree species composition, stand age, composition of the shrub community, and structure of the surrounding landscape matrix (Bailey, 1995). For

example, older, less disturbed forest stands in eastern deciduous forests have greater moth species richness than disturbed forests, as they tend to contain larger trees and relatively lower cover of invasive shrubs (Summerville & Crist, 2003).

A study conducted in mixed pine-spruce stands in Finland found that clear-cuts created by logging supported bumblebees and butterflies more obviously than diurnal moths, due to their attraction to increased abundance of flowering plants (Korpela et al., 2015). While moths also visit flowers for nectar to fuel their own activities, their responses are largely dependent upon changes in habitat openness and associated changes in microclimate (Korpela et al., 2015). Bees also nest in clear-cuts and collect nectar and pollen from the highly abundant flowering plants to feed their brood, unlike Lepidoptera which only visit the flowers, hence reflecting the more crucial role of clear-cuts. There are exceptions however, for example, burnet moths (*Zygaena* spp.) utilized clear-cuts with an abundance of host plants as fully functional habitats, and not just a supporting habitat in the sense of providing only nectar (Bergman et al., 2019). Parrish and Summerville (2015) confirmed that Lepidoptera species richness and composition are significantly impacted by timber harvest, due to the altered microclimate, with significantly fewer species in harvested stands than unlogged.

Some moth families/subfamilies (i.e. Catocalinae, Noctuidae, Hermeniidae) respond positively to logging disturbances, while others (i.e. Geometrinar, Lymantriidae) respond negatively to disturbance (Kitching et al., 2000). Disturbance, however, may actually benefit some rare species in a community (Summerville & Crist, 2008). Following disturbance, colonization processes are critical to the meta-population structure of moth species (Gripenberg et al., 2008; Jiang et al., 2010). For example, in a

landscape analysis of long-distance dispersal, populations of two species of Noctuidae were able to establish communities more than 25km from their natal habitat when suitable patches existed (Chapman et al., 2010). Similarly, when new forest openings are created via harvest, new moth species colonized them, even though forest openings were present less than 1km from non-timber matrix habitat (Summerville et al., 2013). Butterfly and moth species that feed on forest plants as larvae may fail to colonize habitat patches located far from forest edges (Alanen et al., 2011), but newly created clear-cuts and forest edges may provide adult Lepidoptera with nectar sources and favourable microclimates for larval development (Dennis et al., 2004; Jonason et al., 2014; Korpela et al., 2015; Viljur & Teder, 2016).

Ecotones/Edge Effects

The edge consists of both cover types from interacting adjacent cover types, across which both biotic and abiotic components change (Murcia, 1995). Clear-cuts can create more abrupt edges than wildfires (Forman, 1995; Harper et al., 2004). The hard edge along clear-cuts creates a transition between the two adjacent habitat types with very limited penetration of edge effect into the adjacent area, while a soft edge from wildfires is more permeable to edge effects and penetrate further into the adjacent area (Forman, 1995; Harper et al., 2004). Therefore, these edges may experience different ecological changes along the area between undisturbed and disturbed habitats based on the disturbance type (Larrivéé et al., 2008).

Ecotones were initially viewed as edges that define the extent of an assemblage, but later it became apparent that they may represent unique habitats, high in species richness, and therefore important for conservation (Dangerfield et al., 2003). The

function of ecotones as environmental gradients has since become a focus for habitat management. As human land use increases, especially the clearing of native vegetation, there tends to be an increase in the proportion of edges. The term ‘ecotone’ can then be used to describe these edges that mark the ecological change between the two environments (Marshall & Moonen, 2002), with small and linear fragments having proportionally more edge than large and round fragments (Saunders et al., 1991). Meffe and Carroll (1994) defined habitat fragmentation as the conversion of a formerly continuous habitat into small and isolated remnant patches, which results in the creation of biologically relevant edge effects (Silllett, 1994; Esseen & Renhorn, 1998).

Timber harvesting yields warmer microclimates and flower availability along clear-cut edges (Korpela et al., 2015). Along forest edges, the microclimate often includes increased amounts of sunlight, high wind speeds, and larger fluctuations in temperature and humidity, in comparison to forest interiors (Chen et al., 1995; van Wilgenburg et al., 2001). Murcia (1995) states that for a ‘true edge effect’, the physical conditions at the edge affects the abundance and distribution of a species. The ecological and environmental effects of edges on the spatial distribution of animals and plants have been described and discussed mostly from the perspective of how ecosystem fragmentation affects biodiversity (Ries et al., 2004; Rossetti et al., 2014; Stangler et al., 2015). Many studies have studied how the presence of edges can induce changes in vegetation and wildlife, (Murcia, 1995; Lidicker, 1999; Laurence et al., 2002; Ries et al., 2004; Harper et al., 2005), which can be linked to insect assemblages. For example, remaining forest fragments have high levels of edge related effects that can have a significant impact on many forest-dwelling assemblages (Murcia, 1995).

Along edges, assemblages of invertebrate species can be similar to the adjacent habitats (Kotze & Samways, 1999). A study in the boreal found that forest edges play a key role in supporting farmland insect diversity (Kuussaari et al. 2007). It has also been reported that there is an increase in richness and abundance of generalist insect species near edges, while the interior of fragmented habitats tends to favour specialized and competitor insects (Didham, 1997). Due to the historic availability of edge habitats, created by wildfires, especially in boreal forests, edge specialists have arisen overtime and benefit more from clear-cut created edges than other types of specialist species (Larrivéé et al., 2008). Several studies have found that arthropod groups were significantly more abundant or richer at the edges compared to the interior of the adjacent habitats (Jokimaki et al., 1998; Molnár et al., 2001; Magura, 2002; Pearce et al., 2005; Mathe, 2006). Similarly, the spatial distribution of a species of nymphalid butterfly, showed a distinct edge-biased distribution via highest abundance of preferred ground cover near forest edge (Bergman, 1999). Another group of insects showing edge preference are carabid beetles, and they have been known to have edge assemblages more similar to that of forest interiors than clear-cuts (Heliölä et al., 2001). Such studies strongly suggest that edge-biased distributions are often explained by microclimate differences between edge and interior (Nguyen & Nansen, 2018).

In an ‘edge-effects trial’ by Williams & Jonusas (2019), more moths were captured in pheromone traps positioned on the edges of oak woodland blocks than traps positioned within the oak woodland itself. More specifically, Arctiidae moths have been found to be significantly more species rich in recovering secondary forests, which are edge-like habitats, compared to mature forest (Fiedler et al., 2007; Noske et al., 2009).

Fuentes-Montemayor et al. (2012) discovered that the abundance of macromoths was higher along edges than in forest interior. Past studies by Summerville and Crist (2003; 2004) attributed observations like this to the species replacement hypothesis, whereby a few well adapted species are most likely the driving factor. An earlier study made similar conclusions with specific moth species along pine forest edges, as they also support higher abundance of moths than forest interior (Hourí & Doughan, 2006). The importance of clear-cutting in providing temporary habitats for lepidopterans may be beneficial for red listed species as well (Bergman et al., 2019). Increased rates of mate-finding success are more likely along forest edges than the interior of the adjacent matrix. Gypsy moths have the highest mate-finding success along edges, reduced rates in open fields and lowest rates in forest interiors (Thompson et al., 2016). These moths were also found in higher densities along edges than in forest interior. Consequently, it is understandable that nests and egg masses are more abundant along forest edges (Hauck et al., 2008; Dulaurent et al., 2012; Régolini et al., 2014). The plume is likely to disperse more readily and further when positioned on the forest edge rather than within the interior (Murlis et al., 2000). Fortin and Mauffette (2001) found larger egg masses and better overall performance including bigger pupae, higher larval and pupae survivorship along edge habitats. Caterpillar abundance and diversity show a general neutral or positive trend along edge habitats which can be attributed to a plethora of proposed mechanisms including niche availability, host plant diversity, host plant nutritional quality, and parasitoid/pathogen pressure (White et al., 2011).

From a landscape conservation perspective, measuring edge effects on species associated with interior forest conditions will better assess the impact of silviculture

practices on the landscape and organisms (Murcia, 1995; Baker et al., 2007). The transition zone created between harvested and unlogged forests may differ in structure and composition from the adjacent areas for several years. As human activity in natural environments increases, understanding the effects of these forest edges becomes increasingly important to better assess the impacts of forestry on biodiversity. There is a gap in knowledge about macromoth use of edges in boreal forest, as the existing knowledge fails to address this concept specifically, which I am filling by looking at how they respond to clear-cut ecotones.

Introduction

Human land use practices, especially clearing of native vegetation, tend to increase edges, creating ecotones. Ecotones are important for conservation, as they are species rich, thus they have become a focus of habitat management (Dangerfield et al., 2003). Edges have been an important part of boreal ecosystems for millennia, typically arising from recurrent wildfires that create edges between burned areas and unburned areas (Niemela, 1999; Harper et al., 2004). The reduction of wildfires in black spruce boreal forest is of concern as the loss of naturally created edges can impact biodiversity. Understanding how species assemblages change across environmental gradients is central to biodiversity science (Smith et al. 1997). Given the value of biodiversity in providing numerous ecosystem services (Winfrey et al., 2007), biodiversity conservation needs to be considered as a component of forest management.

Habitat edges represent rapid changes in landscape physiognomy, with concurrent changes in environmental conditions (Ries et al. 2004). These ecotones are important transition zones that can promote biological diversity (e.g., Korpela et al., 2015). Edge effects have been identified as one of the main driving forces behind changes in insect populations in forest fragments (Ewers et al., 2007; Ewers and Didham, 2008). Knowledge of how transitions such as those created by natural disturbance and forestry practices influence biodiversity is a critical contribution to sustainable forested management (e.g., Ries et al., 2004; Phillips et al., 2006; Rossetti et al., 2014; Stangler et al., 2015). While forest management practices do not typically consider the impact on non-pest insects (e.g. Holloway, 1989; Mrosek, 2001), there have been calls for the consideration of terrestrial arthropod groups in forest management (Niemela 1997, Pearce

and Venier 2006). Insects respond rapidly to changes in the landscape (e.g., Venier et al. 2017) and their species composition and diversity changes are often correlated with the fraction of trees that is removed during harvest (Forkner et al., 2006; Summerville, 2011).

Nocturnal macromoths were the focus of this study for several reasons, the most evident reason being their size, making them more simple to identify than micromoths. Nocturnal macromoths are abundant, diverse, and are among the most functionally important taxa in forests, with species playing key roles as pollinators, herbivores, detritivores, and as an important food source for bird and bat species (Holmes et al., 1979; Burford et al., 1999; Ober and Hayes, 2010; Summerville, 2011; Highland et al., 2013). Forests support considerable lepidopteran biodiversity (e.g., Summerville and Crist, 2008), hence, one of the most important threats to forest moths is habitat loss and fragmentation as deforestation continues rapidly (Conrad et al., 2004). Given that nocturnal macromoths can be trapped relatively easily and in large numbers by using light traps, they are an effective group to assess effects of human disturbance (Young, 2005; Chaundy-Smart et al., 2012). In deciduous forest of North America, Lepidoptera are species rich, especially for Noctuidae and Geometridae, but individual forest communities are often dominated by a small subset of species (Summerville and Crist, 2008). They have also shown promise as forest indicator taxa (Kitching et al., 2000), making them perfectly suitable to study community differences between logging transitions. There is, therefore, to be a knowledge gap with respect to lepidopteran diversity in the boreal forest. From a sustainable forestry perspective, implementing harvest strategies that lepidopteran communities can tolerate, or are promoted by, reduces the risk of broader trophic consequences. Forest practices that harm lepidopteran

communities may result in trophic consequences either because of diminished prey availability or because of shifts in consumers of primary production (Summerville, 2011), as they are a vital food source.

Forest macromoths are a pragmatic group with which to ask questions about how species respond to environmental gradients, such as edges because they can be trapped relatively easily and in large numbers by using light traps (Young, 2005; Chaundy-Smart et al., 2012). Furthermore, like beetles and spiders (Pearce and Venier 2006), they have also shown promise as forest indicator taxa (Kitching et al., 2000). While there have been a plethora of studies showing that clear-cuts support significantly fewer moth species than areas of later succession (e.g. Summerville, 2011; Summerville and Crist, 2008; Forkner et al., 2006; Summerville and Crist, 2002), Little work has been done on how forest edges affect these taxa in boreal forests, as studies focused more on comparing lepidopteran communities in logged areas to unlogged forests (e.g. Summerville and Crist, 2002; Summerville and Crist, 2008; Parrish and Summerville, 2015; Summerville and Marquis, 2017). Studies have found an increase in lepidopteran abundance along forest edges (e.g. Fuentes-Montemayor et al., 2012; Didham, 1997). More specifically Bergman (1999) found a species of butterfly that showed a distinct edge-biased distribution, driven by microclimatic conditions. The boreal forest is the largest forest biome in the world, representing some 1.9 billion hectares globally. Since much of the boreal forest is managed, there is ample opportunity to gain knowledge about how forest edges influence nocturnal macromoth assemblages.

I sampled nocturnal macromoths in western Newfoundland, Canada to determine their responses across forest edges at the local spatial scale. I predicted that:

1. Edge habitats will support the highest moth abundance compared to clear-cut and forest habitats;
2. Community composition will differ between clear-cut and forest habitats, with edge habitats supporting an intermediate community composition;
3. Edge habitats will support the highest diversity, including specialist species from both adjacent habitats.

Materials and Methods

Study Location

I conducted this study in 2018 in an active commercial forestry area in western insular Newfoundland and Labrador (48.19°N, 58.75°W; Figure 1). Softwood is the primary target of industrial logging in the region, for paper production. The forest in the region is dominated by two softwood species, balsam fir (*Abies balsamea* (L.) Mill. (Pinaceae)) and black spruce (*Picea mariana* Mill. (Pinaceae)), with limited hardwood stands dominated by white birch (*Betula papyrifera* Marshall (Betulaceae)). Clear-cut areas selected for this study were less than 5 years post-harvest and have very little regrowth of trees or shrubs, including, but not limited to, *Cornus* spp. L. (Cornaceae), *Rubus* spp. L. (Rosaceae), and *Vaccinium* spp. L. (Ericaceae).

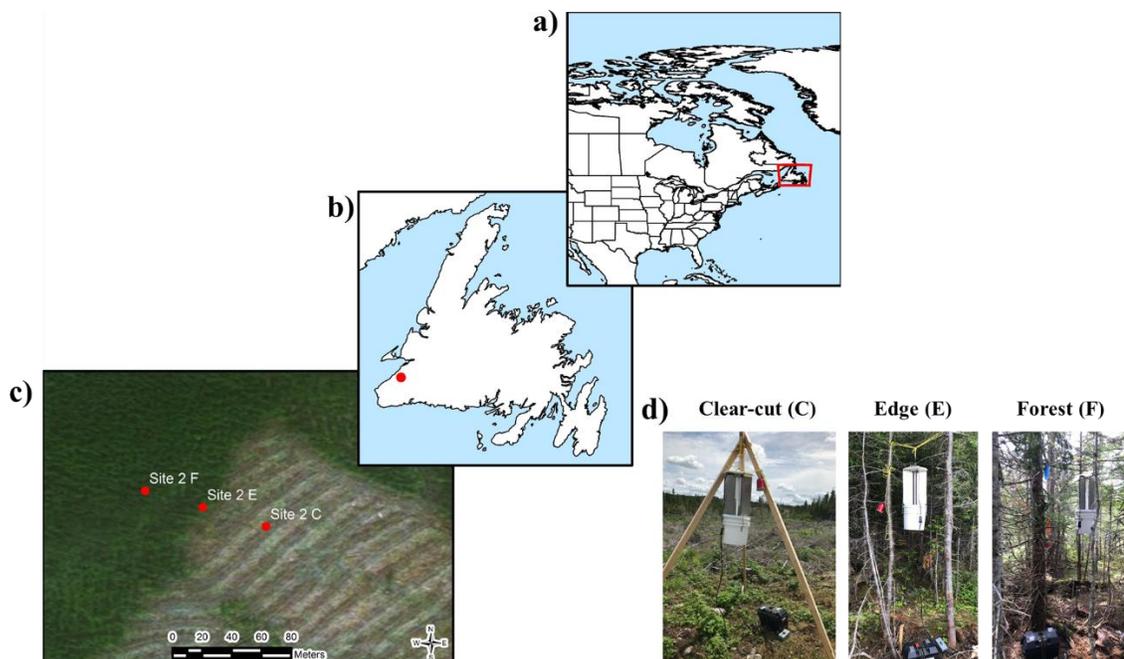


Figure 1. a) Newfoundland, Atlantic Canada. b) Location of study area in western Newfoundland (Camp 180 Resource Road). c) Site 2 of the 4 sites, illustrating the locations of light traps in Forest (F), Edge (E), and Clear-cut (C) habitats. d) Traps hung in Clear-cut (C), Edge (E), and Forest(F).

I selected and established sites at four locations along Camp 180 Resource road, locations were at least 1 km apart. These locations had both clear-cut and forested areas >50 m in diameter, and no logging was scheduled during our sampling period. At each site location, I established a light-trap within a clear-cut (C), at the edge (E), and within forest (F), resulting in a total of twelve traps. Forest sites were mature, consisting mostly of balsam fir and spruce, and clear-cuts had little regrowth in terms of trees but contained large amounts of small flowering shrubs (e.g. blueberries). Edge sites, being characterized as the boundary line where clear-cut and forest meet, had characteristics of both clear-cut and forest.

Light Trapping

I used flight intercept traps to capture nocturnal macromoths. Each trap had a 12-W fluorescent black tube light, powered by an automotive battery with a light sensor, with a photocell that triggered on at dusk and off at dawn. Each trap contained three insecticide strips (Hercon® Vaportape II™) as a killing agent. I placed traps approximately 2 metres off the ground, using available trees in forested and edge sites, and on tripods made of 3 pieces of softwood lumber (2 by 4 inches, by 8 feet long) in clear-cuts (Figure 1d). I placed forest and clear-cut traps thirty meters into the respective habitats from the edge, to limit interference between traps (Figure 1a). Past studies have shown that moths can be attracted to UV lights up to a distance of 30–40 m, but often below 10 m (Truxa and Fiedler, 2012; Merckx and Slade, 2014).

Moth Collection

I sampled during the three nights bracketing the new moon, with collections of specimens taking place each morning on 13-15 June, 12-14 July, and 10-12 August, for a

total of twelve trapping nights over three sampling periods. For each sampling period I hung light traps with new insecticide strips, then removed the light traps after the three-day sampling period. I stored all collected specimens in a container by site and date and placed them in a freezer until they were pinned. All moths that were considered macro (large and within a family classified as macro) were pinned within the following six months and identified individuals to the lowest possible taxonomic unit, using various literature (Beadle and Leckie, 2012; Handfield, 2011; Moth Photographers Group). For the purpose of this study only macromoths were assessed, included all families of macromoths, the remaining micromoths were stored in freezers at Natural Resources Canada, Canadian Forest Service, Corner Brook, NL. Pictures were taken of any species I was unable to identify on my own and uploaded to the Insects of Atlantic Canada Facebook group and identified by Jim Edsall. Integrated Taxonomic Information System (2020) was used to confirm the most up to date taxonomic name of each species. A voucher collection has been created and is stored at collections with Natural Resources Canada, Canadian Forest Service, Corner Brook, Newfoundland and Labrador, Canada.

Temperature and Light

I placed HOBO Pendant® temperature/light loggers (Onset Computer Corporation, Bourne MA) at each light trap on a nearby branch or tripod leg, under a red plastic drinking cup to avoid high temperature readings due to direct sunlight. While this may slightly limit light intensity readings, measurements are still useful for comparison among sites. These loggers recorded temperature and light intensity every hour during each sampling period. I calculated mean temperature for daytime (defined as the three hours before and after solar noon, i.e. between 10:00 and 16:00 h, Newfoundland

Daylight Time) and nighttime (the corresponding hours at night, i.e. 22:00 – 04:00 h). As there were no major differences in temperature, data analysis using this data was not completed (Appendix 1).

Data Analysis

I completed all data analysis using R (version 3.5.2, R Core Team). First, I used analysis of variance (ANOVA) to test for the effect of habitat on overall nocturnal macromoth abundance, with month and site as random effect. In the case of significant differences ($p < 0.05$), I used a Tukey post-hoc test to determine how abundance differed significantly between habitats. Using the same methodology, I analyzed the effect of habitat on each of the top five most abundant species, as they accounted for the majority of the collection and were the only species with abundance counts over 100 individuals.

I used non-metric multidimensional scaling (NMDS) ordination (Legendre and Legendre, 1998) using function *metaMDS* in R package *vegan* (Oksanen et al., 2015) to visualize species composition. In order to create the NMDS plot, I used the function *ggplot* in R package *ggplot2* (Wickham, 2008), excluding singletons and using transformed species data ($x' = \log(x+1)$). The approach uses Bray-Curtis dissimilarity scores, allowing the visualization of the similarity between moths collected among habitats. To make statistical conclusions from the NMDS, I performed permutational multivariate analysis of variance (PERMANOVA) using function *adonis2* in R package *vegan* (Oksanen et al., 2015).

I used a Venn diagram to visualize raw species richness in each habitat by showing the number of species that were unique to each habitat and the number that were shared between habitats. Using the *iNext* package (Chao et al., 2014; Hsieh et al., 2016), I

calculated diversity using Hill numbers, calculated at each habitat type, with site amalgamated and date as replication, and then averaged ($q = 0$; species richness, $q = 1$: Shannon diversity, $q = 2$; Simpson diversity). I created diversity profiles with error bars using the *ggiNEXT* function, using the rarefied and extrapolated species diversity determined from running *iNext*.

I ran indicator species analysis for each habitat using the *multipatt* function in the package *indicspecies* (Dufrene and Legendre, 1997). The *multipatt* function calculates two values, specificity and sensitivity. Specificity (A) is the probability that a trap belongs to a certain habitat, based on the species collected in it. Sensitivity (B) is the number of traps in a certain habitat in which the target species was found. Based on running 999 random permutations the resulting p -values represent how significantly a species is associated with a habitat.

Results

I collected and identified a total of 3486 individual moths, constituting 187 species in seven families (Drepanidae, Erebiidae, Geometridae, Noctuidae, Notouidae, Saturniidae, Sphingidae), over the twelve nights of sampling in 2018 (Appendix 2). Of the total individuals, 87% were identified to species, with the remaining 13% identified to the family or genus level. The majority of individuals identified to the species level were previously known from Newfoundland, except for seven species (see Appendix 2) which are confirmed in other Atlantic provinces in Canada. 83% of the collection was made up of Noctuidae (42%) and Geometridae (41%). The ten most abundant species, which made up 50% of the collection, consisted of Geometridae and Noctuidae and one Erebiidae (*Lophocampa maculata*; Figure 2). Only the top five most abundant species were represented by >100 individuals each.

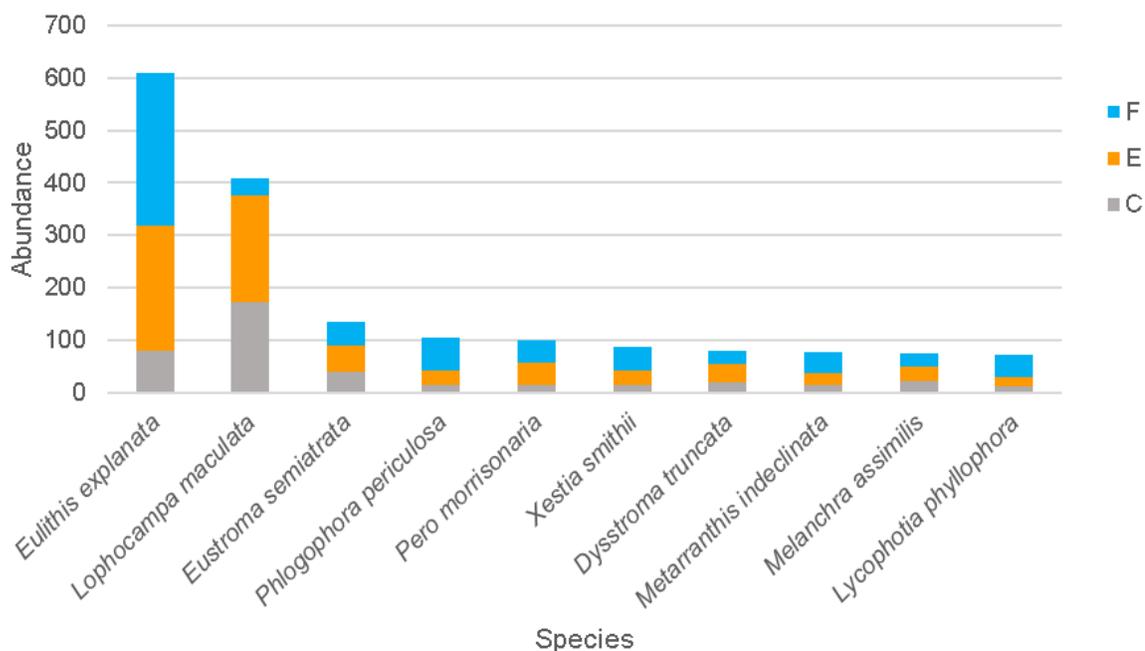


Figure 2. Total abundance of 10 most abundant macromoth species collected by habitat, F (forest), E (edge), C (clear-cut).

Moth Abundance

There was a significant effect of habitat on overall moth abundance ($F_{2,102} = 3.871$, $p = 0.024$), whereby clear-cuts supported significantly fewer moths than edges (TukeyHSD, $p = 0.024$, Figure 3). Among habitats, the lowest moth abundance occurred in clear-cuts (24% of the total collection), whereas edges supported 40% of the collection, and forests 36%. Moth abundance differed among months ($F_{2,102} = 52.773$, $p < 0.001$), but there were no differences among sites.

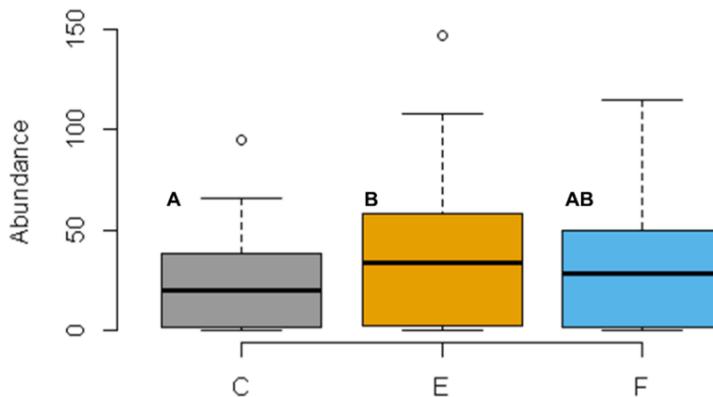


Figure 3. Box plot for the effect of habitat (clear-cut, edge, forest) on overall macromoth abundance. Letters *A* and *B* indicate the significance of differences between habitats for macromoth abundance.

Of the five species represented by >100 individuals, the abundance of *Lophocampa maculata* ($F_{2,103} = 4.171$, $p = 0.018$), *Phlogophora periculosa* ($F_{2,103} = 3.826$, $p = 0.025$), and *Pero morrisonaria* ($F_{2,103} = 3.724$, $p = 0.027$), differed in abundance among the three habitat types. Significantly more individuals of *Lophocampa maculata* were found along edges than in forests (TukeyHSD, $p = 0.072$). By contrast, *Phlogophora periculosa* and *Pero morrisonaria*, were significantly more abundant in forests than clear-cuts (Tukey-test, $p = 0.027$ and $p = 0.042$, respectively). *Eulithis explanata* and *Eustroma semiatrata* abundances did not differ significantly across

habitats. There were no significant differences in temperatures among habitats (see Appendix 1), or light, as levels were always zero due to the lack of light during the sample period around the new moon.

Species Composition

Of the total 187 species, 81 (43%) were collected in all three habitat types. There were 30 (16% of total) found only in edges, 19 (10%) only in clear-cuts, and 14 (7%) only in forests. Edges shared 20 species with clear-cuts and 18 with forests, and only 5 were shared between clear-cut and forest (Figure 4a). The NMDS ordination (method = ‘Bray-Curtis’, stress = 0.11) revealed distinct nocturnal macromoth assemblages between habitats (Figure 4b). Based on the 95% confidence ellipses, clear-cuts and forests were clearly segregated on opposite sides of the ordination. Edge habitats overlapped with clear-cut and forest and fell in-between the two habitats. Results from the PERMANOVA confirmed that macromoth communities differed significantly among habitats ($F = 2.0255$, $p = 0.01$).

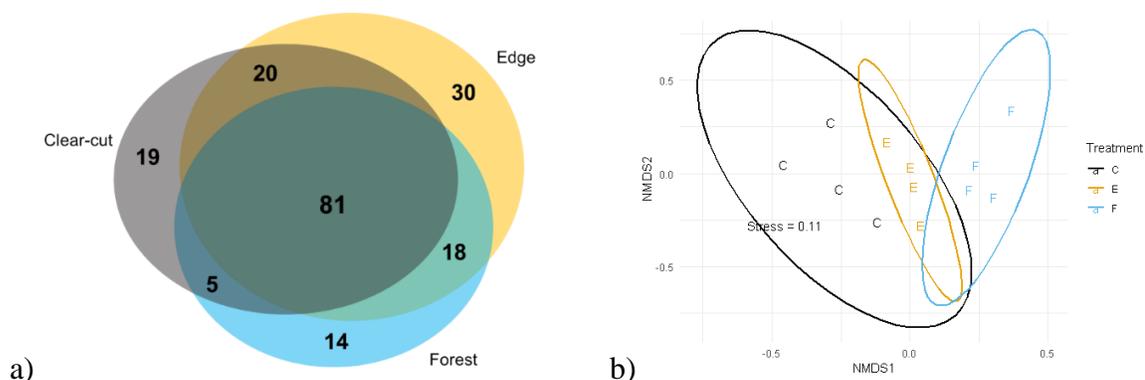


Figure 4. a) Venn Diagram representing raw species richness by habitat, showing the number of unique and shared species of nocturnal macromoths within clear-cut (grey), edge (orange), and forest (blue). b) Nonmetric multidimensional scaling of species abundances pooled across all dates, log transformed ($x' = \log(x+1)$) among habitats using Bray-Curtis dissimilarity method, with 95% confidence ellipses. Each individual letter on the plot represents a replicate sample site for clear-cuts (C), edges (E), and forests (F).

Species Diversity

There were no significant differences in species richness among habitats. Species diversity did not differ significantly among the habitat types as per overlapping confidence intervals. This was the case for species richness (Hill number, $q = 0$), Shannon diversity ($q = 1$), and Simpson diversity ($q = 2$; Figure 5). There were some diversity differences between clear-cut and forest habitats, as indicated by non-overlapping confidence intervals for Simpson diversity and Shannon diversity. Edges supporting 10% more species than clear-cuts and 15% more than forests. Edge habitat diversity was intermediate between the adjacent habitats by both measures. This demonstrates the similarity of edge diversity with its surrounding habitat, with clear-cuts and forests being most different from one another.

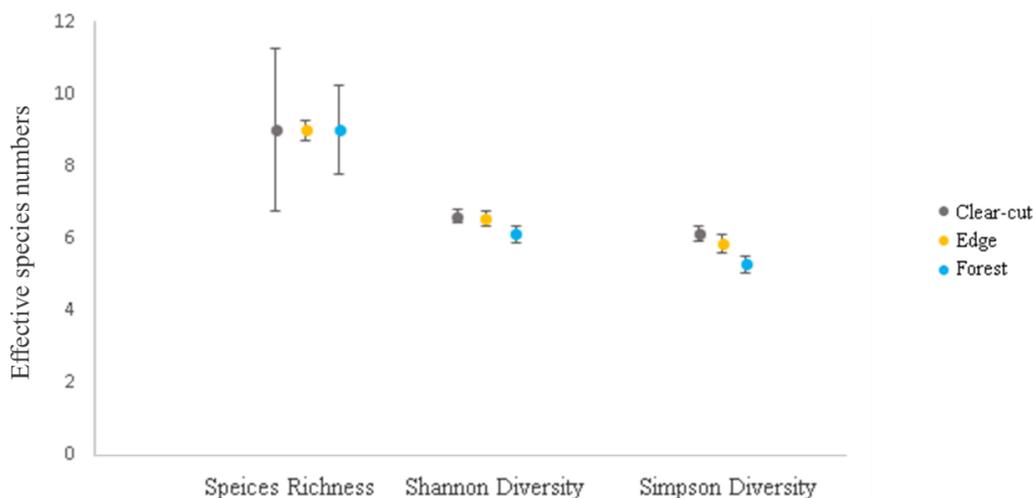


Figure 5. Diversity profiles, characterized by effective number of species (\pm Standard error) for Hill number order $q = 0$ (species richness), $q = 1$ (Shannon diversity), and $q = 2$ (Simpson diversity), at each habitat.

Using species indicator analysis, I identified one species, *Mythimna oxygala*, as being a significant indicator of the edge habitat ($A = 0.8$, $B = 1.0$, $p = 0.023$) with 80% of individuals found along edges. Two species were significantly associated with forest

habitats, *Campaea perlata* ($A = 0.77$, $B = 1.0$, $p = 0.044$) and an unknown Geometrid species ($A = 0.79$, $B = 1.0$, $p = 0.044$), with 77% and 79% of individuals found in forests, respectively. None of the species were significant indicators of clear-cut habitats.

Discussion

In this study evaluating the effects of clear-cut created ecotones on macromoth assemblages in the eastern boreal forest, I found significant differences in nocturnal macromoth abundance, with edges supporting significantly more moths than clear-cuts. I did not detect any significant differences in diversity but did find significant differences in the composition and species that were significant indicators of edges and forests.

Korpela et al. (2015) found that clear-cutting can have positive effects on some groups of pollinators such as bumblebees and butterflies, but that it is not necessarily the case for moths. Within Canadian black spruce boreal forests, stands are often shorter and more dense than broadleaf forests (Harper et al., 2005). Therefore, there is typically less difference between forest and adjacent non-forest system, leading to weaker edge influence, i.e. less difference between the edge and the interior (Harper et al., 2005).

Moth Abundance

I predicted the highest macromoth abundance along edges *sensu* edge effect, which was the case as edges supported a significantly higher abundance of moths than did clear-cuts. This was expected for two reasons: higher visibility in the clear-cut, and the edge microclimate resembling both adjacent habitats. High visibility could lead to high capture rates, since moths are attracted to light, and edge microclimate could create a more favourable habitat. The physical conditions at edges are known to affect the abundance and distribution of many species (Murcia, 1995). Given that the distance between traps was 30 m, it is likely that there was minimal light interference between light traps at each habitat. While higher light penetration could influence capture rates in

clear-cut areas, it is alternatively possible that the highly visible nature of clear-cut areas could allow for more efficient predation by crepuscular and nocturnal insectivores (Kilgo, 2005). Since moths are prey to birds and bats, higher predation in open clear-cuts than along edges or within forests is plausible. For example, a study on warblers foraging along forest edges found increased predation on arthropods as distance from forest edge increased (Kilgo, 2005). While predation may play a role, many studies conclude that low macromoth abundance within clear-cuts is largely due to unfavorable habitat conditions, (Summerville and Crist, 2008; Summerville, 2011; Fuentes-Montemayor et al., 2012).

There was no significant difference between macromoth abundance between edges and forests. I predicted that edges would have significantly more moths due to the microclimate and overall ecotone, since a similar study on forest moths found higher abundances of macromoths along forests edges than in forest interior (Fuentes-Montemayor et al., 2012). The most probable cause for the difference in observed abundance patterns is that the five most abundant species consisted of 39% of total macromoths and were found in greatest abundance along edges (41%), and lowest in clear-cuts (24%). Sharp transitions between forest and clear-cut can cause abrupt changes in microclimate, with rapidly changing air temperatures (van Wilgenburg et al., 2001). The microclimate and physiognomy of the forest edge can support macromoth species from both clear-cuts and forests. Other Lepidoptera such as butterflies, have also shown distinct edge-biased distributions, explained by microclimatic conditions that resulted in highest abundance near forest edges (Bergman, 1999). In this study, I measured two microclimate variables, light intensity and temperature. During the night hours when moths are most active, the light intensity was always zero and played no role in capture

rates, and this was, in part, controlled by my selection of sampling dates (about the ‘new moon’). While temperature has been known to affect species abundance (Jonason et al., 2014), there were no significant differences in temperature among habitats in this study. As would be expected moth abundance differed among months, but because the entire flight season of the moth was not covered, data were pooled over this period.

Species Composition

I predicted that the composition of nocturnal macromoths would differ between clear-cuts and forests, with edge assemblages resembling that of both adjacent habitats. Clear-cut harvesting is correlated with significant reductions in moth species richness, and changes in community composition (Summerville and Crist, 2008). Timber harvest has been found to have a significant impact on richness and composition, with approximately 50% fewer species in harvested stands compared to unlogged forests (Parrish and Summerville, 2015). Since plant species richness is higher along edge environments than in forest interior, pollinating insects’ benefit (Gehlhausen et al., 2000; Korpela et al., 2015). Lepidopterans that feed on forest plants as larvae may fail to colonise habitat patches located far from forest edges (Stasek et al., 2008; Alanen et al., 2011). Our study showed ample overlap in similar species among all three habitats, with edge habitats supporting the greatest number of unique species. Ordination showed that compositionally, edges supported communities that were intermediate between clear-cut and forest communities, representing species from adjacent habitats. The greatest difference in moth communities occurred between clear-cuts and forests. Specialists have been found to be particularly sensitive when logging removes >70% of the standing bole

(Summerville, 2013). Similar effects on species in this study are plausible as areas were clear-cut, leaving only a scattered birch behind.

Species Diversity

Finally, I predicted that edges would support the greatest diversity as they would support specialists from the adjacent habitats. The number of species supported by edges in comparison to the adjacent habitats did not differ greatly, with edges supporting 10% more species than clear-cuts and 15% more than forests. Despite good evidence from other studies that forests support a good deal of macromoth biodiversity (e.g., Summerville and Crist 2008; Korpela et al., 2015) and that clear-cuts support fewer species than unlogged stands (Summerville and Crist, 2002, Franklin et al. 2003), I found that edges supported slightly more than forests, but not more than clear-cuts. This could very well be indicative of a lagged-effect. The stands in which I established traps had all been harvested within the past five years and therefore enough time may not have passed for the assemblage to turnover and reflect the current habitat. Similarly, any species that are associated with early succession plants in clear-cuts may not have had enough time to colonise these areas. A longitudinal study of these sites or chronosequence could elucidate some of these outstanding questions.

Conclusion

Edges created by logging practices in western Newfoundland supported significantly higher abundances of nocturnal macromoths than did clear-cuts. Forested habitats supported an intermediate level of abundance compared to edges or clear-cuts, but with no significant differences. Edges also supported a community composition that was largely intermediate of the two adjacent habitats. While forests and clear-cuts support different compositions, there was no significant difference in diversity among the three habitat types. The results of my study suggest that, like natural disturbance created edges, forest-clear-cut edges may serve as unique habitats for macro-moth assemblages in the boreal forest. These results also have important implications for the maintenance of biodiversity in the boreal and forestry in the context of natural disturbance emulation.

It has been shown that the fraction of trees removed during harvest is often correlated with changes in insect species diversity and composition (Forkner et al., 2006; Summerville, 2011). Harper et al. (2004) suggested that forest management strategies such as ‘feathered’ edges or partial cuts near the edges should be added to the current practices as it could help alleviate the difference between recent cuts and wildfire edges, i.e. hard vs soft transitions. Maintaining a compliment of complex disturbance regimes including fire and non-fire disturbances as well as structural succession is important for preserving biodiversity (Bergeron and Fenton, 2012). While natural disturbance emulation is one of the goals when managing forests, the impacts of percent removal was not in the scope of our study but should be further investigated in the boreal. Maintaining biodiversity and sustaining forest ecosystems means assuring long-term survival of naturally occurring species in viable populations and preserving important processes that

encourage sustainability of the ecosystem (Angelstam, 1998). I provide here a strong baseline for macromoth biodiversity in the boreal shield east.

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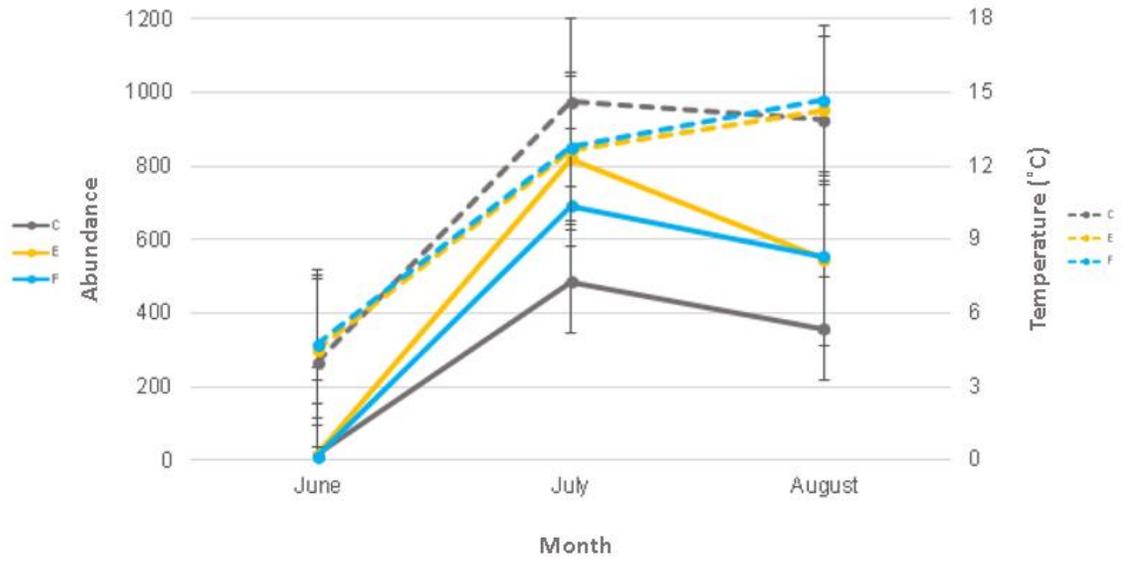
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Appendix 1. Total nocturnal macromoth abundance (solid line) and temperature (dotted line) for each month (June, July, August) by treatment (C, E, F). Error bars represent standard error.

Appendix 2. Raw moth species abundance, alphabetically by species. Collection days
amalgamated per month. Sites amalgamated by habitat type.

Species	June			July			August			Species	June			July			August		
	C	E	F	C	E	F	C	E	F		C	E	F	C	E	F	C	E	F
<i>Abagrotis placida</i> Grote									1	<i>Feralia jocosa</i> Guenée	1								
<i>Acronicta dactylina</i> Grote				8	13	6			1	<i>Graphiphora augur</i> Fabricius								4	5
<i>Acronicta fallax</i> Herrich-Schäffer									2	<i>Habrosyne scripta</i> Gosse		4	6	5					
<i>Acronicta fragilis</i> Guenée					1					<i>Harrisimemna trisignata</i> Walker		1							
<i>Acronicta grisea</i> Walker				3	16	24			1	<i>Hemipachnobia monochromatea</i> Morrison		1	6						
<i>Acronicta impressa</i> Walker					1					<i>Heterocampa biundata</i> Walker			2						
<i>Acronicta innotata</i> Guenée				1	7	2			1	<i>Heterocampa umbrata</i> Walker									1
<i>Acronicta oblongata</i> J. E. Smith				3	1					<i>Hyppa xylinoides</i> Guenée		1	4	6					2
<i>Acronicta</i> sp. (Unknown Noctuidae 1)					5					<i>Iridopsis larvaria</i> Guenée			7	16					
<i>Agrotis venerabilis</i> Walker				2	4		1	1		<i>Lacanobia grandis</i> Guenée								1	
<i>Antheraea polyphemus</i> Cramer				1	3	3	1			<i>Lacinipolia lorea</i> Guenée		5	2	3					1
<i>Anticlea vasiliata</i> Guenée	2									<i>Lacinipolia olivacea</i> Morrison							37	11	1
<i>Apamea impulsata</i> Guenée					2	1				<i>Lacinipolia renigera</i> Stephens									1
<i>Apamea indocilis</i> Walker					1					<i>Leucania commoides</i> Guenée		3	2						
<i>Apamea</i> sp. (Unknown Noctuidae 2)							1	1		<i>Lithophane pexata</i> Grote	1	1							
<i>Aplectoides condita</i> Guenée				1	19	40				<i>Lophocampa maculata</i> Harris		172	203	32	1	1	1		
<i>Autographa flagellum</i> Walker								1		<i>Lycophotia phyllophora</i> Grote		9	13	41	5	3	3		
<i>Autographa mappa</i> Grote and Robinson				1	2	1				<i>Macaria aemulataria</i> Walker		1	2	1					
<i>Autographa precationis</i> Guenée					2			1		<i>Macaria oweni</i> Swett		1	8	18					1
<i>Besma quercivoraria</i> Guenée					2	3				<i>Melanchnra adjuncta</i> Guenée		7	23	36					1
<i>Biston betularia</i> Linnaeus				8	18	2				<i>Melanchnra assimilis</i> Morrison		24	28	25					
<i>Cabera variolaria</i> Guenée					6					<i>Metarranthis duaria</i> Guenée	1	1		1					
<i>Caenurgina crassiuscula</i> Haworth								2		<i>Metarranthis indeclinata</i> Walker		17	21	41					
<i>Callopietria cordata</i> Ljungh						1				<i>Mythimna oxygala</i> Grote			4			3	8		
<i>Campaea perlata</i> Guenée							1	2	10	<i>Mythimna unipuncta</i> Haworth									1
<i>Caripeta divisata</i> Walker					6	21				<i>Nadata gibbosa</i> J. E. Smith		6	7		1	1			
<i>Chrysanympha formosa</i> Grote								1	3	<i>Nemoria mimosaria</i> Guenée		1		2					
<i>Clostera apicalis</i> Walker					1					<i>Nephelodes minians</i> Guenée							3	2	
<i>Coenophila opacifrons</i> Grote								3	8	<i>Ochropleura implecta</i> Lafontaine		4	12	5					1
<i>Colocasia propinqua</i> Grote				4	7	1	1			<i>Oligia bridghamii</i> Grote and Robinson								1	4
<i>Cryptocala acadensis</i> Bethune				1		2	27	23	13	<i>Oligia chlorostigma</i> Harvey				5	14				
<i>Ctenucha virginica</i> Esper				1	1	2				<i>Oligia obtusa</i> Smith								1	1
<i>Cucullia florea</i> Guenée					1	2				<i>Oreta rosea</i> Walker								1	1
<i>Dart</i> sp. (Unknown Noctuidae)	3	3	2							<i>Orthosia revicta</i> Morrison		2	2						
<i>Dart</i> sp. 2 (Unknown Noctuidae)					2	2	2	2	1	<i>Papestra biren</i> Groeze	1	2	3	1	1				
<i>Dart</i> sp. 3 (Unknown Noctuidae)				3	6	7	2	2	8	<i>Pero morrisonaria</i> H. Edwards			16	42	44				
<i>Diachrysa aereoides</i> Grote								1		<i>Phlogophora iris</i> Guenée		4	7	4					
<i>Diarsia rubifera</i> Grote				3	3	2	1	3	3	<i>Phlogophora periculosa</i> Guenée		1	1		16	25	63		
<i>Drepana arcuata</i> Walker				2	6	4	3	3		<i>Plagodis alcoolaria</i> Guenée				2					
<i>Drepana bilineata</i> Packard					2	2				<i>Plagodis phlogosaria</i> Guenée				3	2				
<i>Dysstroma citrata</i> Linnaeus								4	5	<i>Plagodis</i> sp.				1					
<i>Dysstroma truncata</i> Hufnagel				1	2	3	20	32	23	<i>Platartia parthenos</i> Harris		22	21	7				1	
<i>Dysstroma walkerata</i> Pearsall				1	6	3				<i>Plusia putnami</i> Grote			1	1	1	1	1		
<i>Ecliptopera silaceata</i> Denis and Schiffermüller					1					<i>Polia imbrifera</i> Guenée		2	3	5					1
<i>Ectropis crepuscularia</i> Denis and Schiffermüller					1					<i>Polia propodea</i> McCabe			6	6				4	8
<i>Elaphria versicolor</i> Grote					1	1				<i>Probole amicaria</i> Herrich-Schäffer		5	12	17					
<i>Erebidae</i> sp. 1						1				<i>Prochoerodes lineola</i> Goeze									2
<i>Erebidae</i> sp. 2							1			<i>Pseudeva purpurigera</i> Walker								2	7
<i>Eremobina claudens</i> Walker				1	3	2				<i>Pyrrhia exprimens</i> Walker		1							
<i>Eueretagtrotis perattentus</i> Grote					2			2	2	<i>Rheumaptera hastata</i> Linnaeus				1					
<i>Eulithis explanata</i> Walker							2	81	238	<i>Scoliopteryx libatrix</i> Linnaeus								1	
<i>Eulithis propulsata</i> Walker								1		<i>Selenia alciphearia</i> Walker	1								
<i>Euplexia benesimilis</i> McDunnough					4	6				<i>Sicya macularia</i> Harris							4	10	5
<i>Eurois occulta</i> Linnaeus				2	1	1	1	1	3	<i>Smerinthus cerisyi</i> Kirby				1					
<i>Eustroma semiatrata</i> Hulst					42	48	44			<i>Speranza pustularia</i> Guenée									1
<i>Feltia herilis</i> Grote					1	2	1	1	1	<i>Sphinx kalmiae</i> J. E. Smith			2	1	1	1	1		1
<i>Feralia comstocki</i> Grote	1	1	2							<i>Sphinx poecila</i> Stephens	1	10	13	3					

