EVALUATION OF UPLAND HARDWOOD PATCHES USING THREE TAXA IN DOUGLAS-FIR PRODUCTION FORESTS

by

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ABSTRACT

Evaluation of upland hardwood patches using three taxa in Douglas-fir production forests

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Forests are high-functioning and ecologically productive in both natural and managed settings. They provide ecosystem functions and societal benefits including water filtration, carbon storage, and habitat for fish and wildlife. More than 10 million acres of forestland are managed for forest products and ecosystem services in Washington State. Managers of forestlands have the opportunity, through intentional conservation and silvicultural practices, to manage for forest resiliency and biological diversity while maintaining alignment with business and societal objectives. Managed forests form a mosaic of habitat types across the landscape, many of which are conifer dominated with hardwood patches scattered throughout. For this study, I examined the use of conifer- and hardwood-dominated habitat types by three forest taxa (ground beetles, amphibians, and songbirds). Small, upland hardwood patches within the managed conifer matrix were high functioning, with utilization of both habitat types by all taxa. Of the 45 species that were included in the analysis, 14 (31%) were unique to one habitat type or the other, with four species unique to conifer-dominated habitats and ten species unique to hardwood-dominated habitats. The mean species richness of ground beetles and birds was similar in both conifer- and hardwood-dominated plots, while the mean species richness of the herpetofauna community was greater in hardwood-dominated plots. Forest structure and composition components were also evaluated. Across all surveys, significantly more plant species occurred in hardwood-dominated plots than in conifer-dominated plots. The results of this study suggest that upland hardwood patches within the managed forest setting provide conservation value for many species.

Table of Contents

List of Figures

List of Tables

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Chapter 1. Introduction

Pacific Northwest (PNW) forests provide ecosystem functions that filter and produce healthy air, regulate, filter and store water, prevent soil erosion and cycle nutrients, store carbon, and provide a diversity of wildlife habitats, while also providing aesthetic, medicinal, scientific, recreational, spiritual, and economic benefits to society (Carey et al., 1999; Carey, 2003; Gustafsson et al, 2012). More than 10 million acres of forestland are managed for forest products and ecosystem services in Washington State. Forestland managers have the opportunity, through intentional forest management activities, to manage for forest resilience and ecological function while also maintaining alignment with business and societal objectives.

Forest management has intensified on privately managed forestlands in order to maintain older habitats on publicly owned forestlands and to meet the growing demand for high-quality building materials. Overtime, the PNW managed forest landscape has shifted from naturally diverse second-growth forests to planted Douglas-fir dominated third-growth forests. As a result, the diversity of plants and habitat features in managed forests has declined in some areas and shifted spatially in others. The simplification of forest habitat structure has led to concerns about the impacts of forest management on forest resilience and biological diversity. Evaluation of the biological function of specific habitats will improve understanding of high functioning habitats and provide data to inform function-based conservation practices across the managed landscape.

Conservation of forest ecosystem functions, specifically biological diversity, has been identified as a global concern (Chandra & Idrisova, 2011). Healthy ecosystems contain diverse species that provide functional roles important to a systems stability and productivity, many of which are not well understood. Although forestland managers may desire to ensure optimal diversity across their ownership, it is difficult to know what strategies to employ to ensure the effectiveness of specific activities (Simberloff, 1999). Research that identifies high-functioning forest habitats can be used to guide successful conservation strategies.

Understanding the relative contribution of conifer- and hardwood-dominated forest habitats to species biodiversity has significance for forest managers. West of the Cascade mountains in Washington and Oregon, forests managed for forest products are primarily reforested with Douglas-fir *(Pseudotsuga menziesii)* tree species. Although it is known that these forests provide habitat important to an array of species, the relative contributions and required abundances of different elements of forest structure and composition are not well understood. Sustainable forest management is governed by state rules and regulations however, applying function-based protections requires site-specific planning to ensure important forest structural features are maintained at levels that support healthy ecosystem function.

Planted Douglas-fir dominated third-growth forests are regularly interspersed with patches of shrubs and hardwood vegetation. At the time of reforestation, Douglas-fir trees are planted densely and as they grow, they stretch out, blocking sunlight to the forest floor. Sunlight is a limited resource within closed-canopy, conifer dominated forests. Sunlight provides the energy to the subcanopy and forest floor that makes photosynthesis possible. Before sunlight becomes blocked and when sunlight is again made available, a variety of plant species will occur. Many plant species are adapted to thrive in the open forest and understory environments when the right conditions exist. The vegetation response to either access or denial of sunlight occur quickly and frequently in the managed forest setting and create a shifting mosaic of habitat types.

Douglas-fir trees grow better in some soils than others. They do not prefer habitats where the soil is seasonally inundated or occupied by pervasive root-rotting fungus. When the conifer trees die in these areas, canopy gaps are created. Gaps are also created in areas where wind causes trees to topple. The creation of canopy gaps restores sunlight access to the forest floor, creating opportunities for understory species to grow. Many hardwood species are quick to colonize areas where the soil has been disturbed. The result is a conifer-dominated forested landscape with patches of hardwood forest scattered throughout. The physical structure of hardwood vegetation (deciduous leaves instead of evergreen needles) allows sunlight to infiltrate through the canopy, supporting the development of understory vegetation.

Canopy density is an important driver for regulating forest floor light, moisture content, and temperature regimes (Gray et al., 2002; Muscolo et al., 2014). The timing of gap formation, variation in gap size, and differences in microsites within gaps contribute to the diversity of species within forests. Resources vary within and among canopy gaps and also by location and forest type (Gray et al., 2002). Hardwood patches in the PNW are often dominated by red alder (*Alnus rubra*) or bigleaf maple (*Acer macrophyllum*) and a variety of other forb and shrub species. They are utilized by canopy epiphytes and invertebrates within the conifer matrix and can serve as an important source of nitrogen in the nitrogen-limited forest ecosystem typical of the region (Kennedy and Spies, 2005).

Habitat patch size and species richness are associated. Larger patches can have greater resources to support the habitat needs of more kinds of species, even if they utilize similar resources (Andren, 1994). In smaller habitat patches, limited resources may more successfully support species that do not compete for the same habitat elements. Habitats become fragmented when there is a loss of cover type across the landscape. Isolated and fragmented habitats can provide critical habitat for specialized species and play key roles in maintaining biodiversity (Andren, 1994). For taxon that have very small home ranges, the hardwood patches within Douglas-fir production forests could perform similar ecosystem functions. For this study, small sized, upland hardwood patches were selected in order to assess canopy gaps at a scale that exists naturally in the managed forest setting.

A comparison of historic and contemporary hardwood patch dynamics in managed forests of the Oregon Coast Range indicate that hardwood patches have declined in size, number, and total area (Kennedy & Spies, 2005), with numerous implications for conservation. The study of gaps has contributed to our understanding of small-scale disturbance and their importance in maintaining habitat heterogeneity in managed forests (Coates & Burton, 1997). An evaluation of the biological function of small hardwood patches will improve understanding of the shifting patch dynamics and habitat utilization within managed forests.

Mature mixed conifer and hardwood stands provide habitat for a diversity of wildlife species in forested environments, where both coniferous and non-coniferous vegetation species make important contributions to forest biodiversity. Out of more than 430 species of forest-dependent wildlife on the west side of the Cascades, more than 200 species breed or rear young in hardwood-dominated forests (Allbriten & Bottorff, 2004) and 78 species have been associated with non-coniferous vegetation for food resources (Hagar, 2007). Upland hardwood stands provide habitat for cavity-nesting and upper canopy dwelling birds, food resources and nesting cavities for mammals, amphibians, and reptiles, and forage habitat for deer and other browsers (Allbriten & Bottorff, 2004). However, the biological function of small, upland hardwood patches within the context of Douglas-fir production forests is poorly documented.

Conservation of biological diversity requires an understanding of how habitat features, and species compositions vary within ecosystems. Different species select for and utilize different habitat features. These differences are related to evolutionary adaptations and preferences for foraging, breeding, and nesting habitats. Examination of specific taxa within specific forest types can be used to describe the relative contribution of those forest types to biodiversity and the ecosystem. Consideration of multiple taxa will provide a comprehensive evaluation of relative contributions of different forest types and features to biodiversity. For this study, ground beetles, amphibians and forest songbirds were evaluated within conifer- and hardwood-dominated forest habitats to examine the relative contribution of each habitat to biodiversity.

The study was conducted in the lowlands of western Washington. Five sites were selected based on their forest management history and the prevalence of upland hardwood patches. At each site, a 20- by- 20 meter paired-plot design was established, one plot in hardwood-dominated forest, and a paired plot in the adjacent conifer-dominated matrix. Three pitfall traps were created at each plot to survey for ground beetles, establishing a total of 30 traps. Amphibian surveys were conducted across 100%

of the plot area, three surveys were conducted at each site. Point count surveys for forest songbirds were conducted from each plot center and occurred during three occasions from late spring to early summer. Forest structure and composition data was collected at each site to describe the association of the dominant tree canopy type on various forest attributes and habitat conditions.

An assessment of the ground beetle, amphibian, and songbird communities, alongside the biotic and abiotic features within the conifer- and hardwood- dominated forest habitat types will provide for an improved understanding of the associations that exist between forest structural and compositional features and biological responses to forest management. Results may be used to better understand forest management contributions to biodiversity conservation, provide evidence for the conservation benefit of upland hardwood patches, and provide support to forestland managers who wish to quantify the effect of their conservation efforts. Additionally, the results of research like this could support the implementation of function-based forest management practices.

Chapter 2. Literature Review

Ecological Framework

Ecosystem diversity is the variation in ecosystems found in a region (or the whole planet) and includes variation in both terrestrial and aquatic ecosystems (Lapin, 1995). Ecosystem diversity considers the variation in the physical environment and the biological community. The physical environment of a forest is composed of elements such as mature forests, young forest, wetlands, streams, and meadows. Ecosystem diversity is the largest scale of biodiversity, and within each ecosystem, there is a great deal of both species and genetic diversity.

Forest biological diversity underpins the ecosystem's production, resilience, and stability (Thompson, 2011). Healthy ecosystem function relies in part on the health and function of each of the species within it (Luck et al., 2003). Forest biological diversity results from evolutionary processes that occur over thousands to millions of years which, in themselves, are driven by ecological forces such as climate, fire, competition, and disturbance (Carey and Curtis, 1996; Drever et al., 2006). The diversity exists at the ecosystem, landscape, species, populations, and genetics levels and complex interactions occur within and amongst these levels. In biologically diverse forests, the complexity allows organisms to adapt to continually changing environmental conditions and to maintain ecosystem functions. Within forest ecosystems, the maintenance of ecological processes is dependent upon the maintenance of their physical and biological diversity. Ecosystems are stable when mechanisms are in place that help them return to their original state after a disturbance has occurred (Connell & Sousa, 1983). Disturbances that occur too quickly or over too large an area can pose a threat to overall forest health and resiliency (Folke et al., 2004).

Loss of forest biological diversity can be the result of compounding or individual events. Historically, human demands on natural systems have resulted in modification and simplification of biological systems. The conversion of forests to alternate land uses, unsustainable forest management, introduction of invasive plant and animal species, infrastructure development, catastrophic forest fires, and climate change all can negatively impact forest biodiversity (Braunisch et al., 2014). These influences can decrease the resilience of forest ecosystems and make it more difficult for them to cope with changing environmental conditions. Some ecosystems experience tipping points in which significant environmental changes result in the inability to return to previous conditions.

Superimposed on the many anthropogenic impacts on forest ecosystems is global climate change. Climate has a major influence on rates of photosynthesis and respiration (Thompson, 2011), and on other forest processes, acting through temperature, radiation, and moisture regimes over medium and long time periods. Climate and weather conditions also directly influence shorter-term processes in forests, such as frequency of storms and wildfires, herbivory, and species migration (Gundersen et al., 2000). As the global climate changes, forest ecosystems will change because species' physiological tolerances may be exceeded and the rates of biophysical forest processes will be altered (Litten et al., 2010; Weed et al., 2013). Maintaining forest biodiversity is a key element to maintaining forest resiliency.

In forests managed for timber products, the heterogeneity of natural forests is minimized in order to produce consistent and high-quality building products (Drever et al., 2006). However, management of forests for forest products and biodiversity do not have to be mutually exclusive (Aubin et al., 2008; Bunnell & Dunsworth, 2010; Carey & Curtis, 1996; Lindenmayer et al., 2000 & 2012; Rapp, 2002). Understanding how biodiversity supports local forest resilience and resistance will provide important information that can be used to improve forest management.

Natural History

Present patterns of diversity represent the culmination of ecological, climatological, and geological processes spanning over several time scales. The PNW has been shaped by millions of years of glacial advances and retreats. Each episode shaped the landscape, creating ravines that formed the structure for many of our waterways and depositing deep beds of gravel in their wakes. The form and structure of plate tectonics in the PNW have worked over millions of years to create the volcanic ridge of the Cascade Mountains. The array of active volcanoes shaped the environment in ways that fostered resilient and adaptive ecosystems. As the mountains rose, they also functioned to shape weather patterns, creating a wet temperate environment that is conducive to a productive and diverse growing environment on their west side (Franklyn & Dyrness, 1973).

Fire has also been a dominant force in PNW evolution. The fire return interval based on forest age-class data shows that the Olympic Peninsula in Washington may have experienced fire once in several centuries, with the sporadic nature of the fires contributing to catastrophic events (Agee, 1993). The structure of the Siskiyou mountains of southern Oregon and norther California, however, suggest that they may have experienced low-severity fire every few decades, creating fire-dependent ecosystems (Martin, 1997). The forest response to these fire regimes created notable differences in structure and diversity. In the PNW, the catastrophic nature of historic fire regimes created stand renewing conditions, while the more frequent fires in the south were less catastrophic, leaving behind surviving trees while creating openings for new vegetation and creating multi-layered forests.

Forests have shaped the region's history for more than two million years; however, the remnants of the oldest forests are relatively young, having emerged in the past few thousand years following the retreat of the ice sheets of the last ice age. The ecosystems of the PNW are so productive and diverse that they contain more biomass than natural forests of equivalent size in tropical forests. In lower elevations of western Washington, Douglas-fir (*Pseudotsuga menziesii*), western hemlock (*Tsuga heterophylla*), western red cedar (*Thuja plicata*), and grand fir (*Abies grandis*) are common, while Pacific silver fir (*Abies amabilis*), mountain hemlock (*Tsuga mertensiana)*, lodge pole pine (*Pinus contorta*) and subalpine fir (*Abies lasiocarpa*) exist in higher elevations (Chappell et al., 2001). Canopy structure varies from single to multi-storied and tree size varies from small to large. Large snags and downed trees vary from uncommon to abundant based on past land practices and naturally occurring events.

Mid to lower forest canopies vary in structure and density. They are comprised of a variety of native species depending on sunlight penetration, elevation, and precipitation (McIntosh et al., 2009). Deciduous broadleaf shrubs are the most common understory dominants (Chappell et al., 2001). Primary understory coverage in middle-aged forests include vine maple (*Acer circinatum*), red huckleberry (*Vaccinium parvifolium*), dwarf

Oregon grape (*Mahonia nervosa*), oceanspray (*Holodiscus discolor*), and sword fern (*Polystichum munitum*). In younger, open forests, trailing blackberry (*Rubus ursinus*), snowberry (*Symphoricarpos albus*), and bracken fern (*Pteridium aquilinum*) are more common in the understory (McIntosh et al., 2009). Other understory vegetation often includes species such as hazelnut (*Corylus cornuta*), salal (*Gaultheria shallon*), dwarf rose (*Rosa gymnocarpa*), thimbleberry (*Rubus parviflorus*), salmon berry (*Rubus spectabilis*), elderberry (*Sambucus spp.*), and honeysuckle (*Lonicera ciliosa*).

Indigenous Land Management

Indigenous people influenced the shape and structure of their landscape through active management (Lepofsky & Lertzman, 2008). In the PNW, fire was utilized as a tool for manipulating and maintaining natural resources (Williams, 2002). It was used in low levels but often frequently to maintain a forested, meadow mosaic and prairies that encouraged the growth of food crops and augmented the amount of open land used by game species. The cumulative effects of thousands of years of burning altered the function and vitality of the habitat at a landscape level in ways that were mutually beneficial for plants, animals, and humans in the region. The careful application of fire also reduced the fuel load that could be burned by naturally occurring wildfires.

Tribal communities also managed the forests for wood to make harpoons, baskets, and mats. Western red cedar was especially important for the construction of homes, canoes, and totem poles (Williams, 2002). It also provided the raw material to make clothing and intricately carved masks. Plant resources were harvested from the forest that provided food, medicine, and material for mechanical and spiritual purposes (Berg, 2007).

Contemporary Forest Management and Regulatory Framework

Pacific Northwest forests changed rapidly following the westward advance of European descendants. The first significant investor in the region's timber resources, The Hudson's Bay Company in the mid-1800s, introduced drastically different practices of forest management. At first, there were no rules or regulations to dictate how the forests and its resources should be managed and over time it was realized that the health and integrity of the forests and streams were declining. As a result, Federal and State rules and regulations evolved to require protection of sensitive fish and wildlife species, their habitats, and forest ecosystem resources like water.

In 1973, the federal Endangered Species Act (ESA) was enacted with the intent of protecting and recovering imperiled plant and animal species and the ecosystems upon which they depend. It is administered by the U.S. Fish and Wildlife Service (USFWS) and the National Ocean and Atmospheric Administration Fisheries (NOAA Fisheries). Under the ESA, species may be listed as threatened or endangered. Once a species is listed, harassment or harm towards that species or its habitat is prohibited. As of 2020, 1,634 species were listed, several of which are known to reside within western Washington forests and forest streams (USFWS, 2020).

In 1994, the federal Northwest Forest Plan was adopted to establish an ecosystem and watershed-based management plan for federal lands in western Oregon, western Washington, and part of northern California. It is a series of policies and guidelines designed to govern long-term management of late-successional forest habitat in response to declines in Northern spotted owl (*Strix occidentalis*) populations. A multi-disciplinary team composed of tribes, federal agencies, scientists, and others worked together to develop the plan. The plan identified five major goals: 1) never forget the human and economic dimensions of the issue, 2) protect the long-term health of forests, wildlife, and waterways, 3) focus on scientifically sound, ecologically credible, and legally responsible strategies and implementation, 4) produce a predictable and sustainable level of timber sales and nontimber resources, and 5) ensure that the federal agencies work together (U.S. Forest Service et al., n.d.). This legislation caused a prominent shift in the distribution and intensity of forest management from public to private lands.

In 1999, the Washington Forest and Fish Report (FFR) was produced by a multi-stakeholder group composed of tribes, forest landowners, federal and state governments, counties, environmental groups, and others. The FFR was developed for non-federal landowners, in response to the federal listing of several species of Pacific salmon as well as the continued listing of surface waters under the federal Clean Water Act 303(d) list. To address these issues the FFR outlined protections for water quality and aquatic wildlife. The plan identified four key goals: 1) provide compliance with the federal Endangered Species Act for aquatic and riparian dependent species, 2) restore and maintain riparian habitat to maintain a harvestable supply of fish, 3) meet the requirements of the Clean Water Act for water quality, and 4) keep the forest products industry economically viable (Washington Department of Natural Resources et al., 1999). That same year, the Salmon Recovery Act of 1999, also known as the Forest and Fish Law was enacted. It directed the adoption of the Forest and Fish Report goals and protective strategies into State Forest Practice Rules. The Forest Practice Rules are governed by the state’s Forest Practices Board.

In 2006, the Washington Department of Natural Resources (WDNR), in collaboration with USFWS and NOAA Fisheries, completed the statewide Forest Practices Habitat Conservation Plan. This legislation endorsed the Forest and Fish Law and ensured that landowners who conducted their forest practices activities in compliance with the Forest Practices Act and rules, were also adhering to the requirements for aquatic species under the federal Endangered Species Act (WDNR, 2005).

Despite the regulatory framework that exists to manage and conserve the regions forests and forest waters, questions linger regarding the suitability of protective measures. Diverse interpretations of the science and disparate values contribute to an ongoing debate about how to approach conservation policy for managed lands. Some land managers are concerned about the reduced economic value that comes with setting land aside, while some members of the public are concerned that not enough is being done to conserve species and shared resources. Successfully integrating science and societal values will be necessary to develop effective environmental policy (Wilhere & Quinn, 2018).

Measuring Forest Diversity

Resilient forests are high-functioning and ecologically productive in both natural and managed settings, however the ecological and biological responses to management are not well understood. Biodiversity indicators are needed to measure and monitor changes and trends in aquatic and terrestrial habitats (Brown & Pollock, 2019). Conservation of biological diversity requires an understanding of how habitat features, and species compositions vary within ecosystems. Identifying landscape level forest cover and vegetation patterns can provide landowners a reference condition, from which they can develop landscape level biodiversity targets. Identifying habitat -species relationships within forested ecosystem at the local scale can help determine which forest features contribute to high-functioning wildlife habitat (Brown and Pollock, 2019). Combining landscape and local scale assessment methods can provide forestland managers tools to evaluate current conditions, compare them to regional and local biodiversity targets, implement plans, and measure change.

Landscape Scale Biodiversity Metrics

A landscape is an area of land with groups of vegetation communities or ecosystems forming an ecological 'unit' with distinguishable structure, function, geomorphology, and disturbance regimes (Gaines et al., 1999). Landscape diversity is the amount and percentage of different ecosystems within the landscape and to some degree also represents the interactions and/or disturbance regimes within it. Landscape features such as forest cover by size and percentage, wetland or aquatic habitat by size and percentage, habitat connectivity and ratio between interior and edges all describe elements of the landscape that affect species composition, viability, and distribution.

Assessments of the level of diversity that exists within areas of interest and an evaluation of how it compares with historic levels are important landscape scale analyses. Gaining an understanding of the trends in landscape composition, features, and specific habitats help shape landscape diversity targets. Developing a baseline is challenging but critical to efforts that aim to recognize patterns and changes in species populations, compositions, and distributions over time.

Biological indicators can be monitored for change to answer and validate specific species-habitat relationships and to measure the status of biodiversity from landscape to local scales (Noss, 1990). A multi-taxon framework is likely best suited to understand biological responses to management activity (Lelli et al., 2019) and could be applied at landscape and local scales. By assessing the patterns of many species in an area, a stronger correlation to the importance of certain habitat features can be drawn. However, consideration of the three attributes of biodiversity (composition, structure, and function) at the landscape, population, species, and genetic levels will help define more specific areas of biological significance (Noss, 1990).

Local Scale Biodiversity Metrics

Monitoring biodiversity at the local scale is important when trying to identify patterns and trends in ecosystem function and integrity. Important assessments at this scale include how management activities or natural disturbances affect species diversity and richness or rarity in localized areas, how species function in their ecosystems, and how well their associated habitats are perpetuated across the landscape. Forestland managers may be interested in the results of these types of assessments to better understand the impacts of forest management activities (positive or negative) and use results to guide best management practices.

Assessing the spectrum of species is complex and many researchers have attempted to group them into categories that are easier to assess. Common groupings are based on habitat preference, behavioral similarities, or by the functional role they perform in the ecosystem. For assessments based on ecosystem function, the relative importance of each guild (or kind of specie) is considered based on the relative importance of its role in the ecosystem (e.g., decomposer, seed disperser). However, this type of research can be flawed in its underlying assumption, if it assumes that each specie provides only one specific function with which its relative importance is solely based.

Many forest biodiversity indicators have been used to assess (or indicate) levels of conservation success and/or need, however evaluation and critical assessment of the scientific rigor at the individual indicator scale (species or habitat feature) as well as at the ecosystem scale is lacking (Gao et al., 2015). Additionally, potential gaps and overlaps exist that need evaluation. A review of 142 (European) published studies which included 83 indictor groups were analyzed (Gao et al., 2015). Of 412 indicators identified in the studies, 6 indicators were supported by strong evidence and scientific rigor. Species richness indicators were best represented when correlating deadwood volume with wood-living fungi and saproxylic beetle richness; when evaluating deadwood diversity with saproxylic beetle richness; and when relating canopy tree age with epiphytic lichen richness (Brin et al., 2009; Gao et al., 2015).

Habitat Indicators

High-functioning, diverse forested landscapes are ecologically complex and can be described as having vertical and horizontal heterogeneity which includes vegetation size and species mix, live and dead standing trees of various species, heights, and diameters, dead trees and limbs on the forest floor, and a healthy and diverse developed understory (Carey & Curtis, 1996, Lindenmayer et al., 2000 & 2012). Multiple dimensions within the forest landscape creates micro-habitats, each with its own light, temperature, and moisture conditions, which allows for a variety of flora and fauna to thrive. The structural complexity of any habitat can represent the diversity of wildlife habitat available.

Wildlife utilize a variety of habitat components. All forest types provide habitat to some species. Developing diverse and complex stand structures is key to providing for a diversity of species and processes and can be achieved at the stand and landscape levels in managed forests (Zobrist & Hinckley, 2005).

Wildlife Indicators

Suitable wildlife habitat provides connected feeding, roosting, breeding, nesting, and refuge habitat for a wide spectrum of native terrestrial and aquatic species. When habitats become highly fragmented, species that are habitat specialists (e.g., spotted owls) are vulnerable to becoming restricted to their isolated patch of habitat, limiting foraging and the potential for genetic mixing (Newmark et al., 2017). Species that are generalists (e.g., barred owls) have adapted to a wide variety of habitats and disturbance regimes, and therefore are often more resilient to disturbance imposed by human activity.

Each species requires a unique suite of habitat elements (i.e., snags, live trees, woody debris) which provides for its ability to thrive and perpetuate, therefore, a ‘one size fits all’ approach to biodiversity monitoring is ineffective at evaluating accurate baseline diversity metrics (Dale & Beyeler, 2001). The relative sensitivity of wildlife species to their environments can help to determine which species to focus on for measuring and monitoring biodiversity, whereas a suite of taxa may be most beneficial to understand broader ecosystem diversity (Dale & Beyeler, 2001). Indicators are most effective when they represent key attributes of the ecological functions of interest (Juutinen & Mönkkönen, 2004).

‘Representatives’ of Biodiversity

Habitat effectiveness is often evaluated by monitoring wildlife in one of three categories. Monitoring for the presence, absence, or relative well-being of ‘indicator’ species, ‘keystone’ species, or ‘umbrella’ species in a given environment is often utilized to represent larger groups of species and the overall health of the ecosystem. Emphasis on indicator, keystone, and umbrella species has illuminated the role of specific species within ecosystems. For example, indicator species are sensitive to foreign disturbance. By monitoring the condition of them, scientists can make correlations to the well-being of other species in the same habitat (e.g., Pacific salmon). They are chosen to assess how specific ecosystems are doing and to assess the effect of change (Carignan & Villard, 2001; National Geographic, n.d.).

Keystone species provide unique functions in their habitats, whereby few, if any other species in that habitat provide the same function (e.g., gray wolf). These species have notable influence on local food webs and without them, the local ecosystem is radically different (Mills & Doak, 1993; Society, 2017). Umbrella species are similar to keystone species except they are more likely to utilize large tracts of land (e.g., grizzly bear). They have a high degree of influence on local and disbursed food webs and therefore their value can cover broad geographic range (Roberge & Angelstam, 2004; Society, 2017).

Focus on specific species as representatives of entire ecosystems does not likely accurately represent all potential species and their functional relationships in the ecosystem (Simberloff, 1999; Dale & Beyeler, 2001), while focus on stand-level species richness or sensitive species habitat areas may not capture the full spectrum of biodiversity status and responses (Lelli et al., 2019). Management designed to protect one species are unlikely to be successful if conservation of the full range of species and their functions is desired. An functional approach which includes grouping species by 'kinds' rather than by abundance, could lead to a better understanding of land management activities that alter habitats and species assemblages, as long as species are not valued on a single functional trait. By evaluating the effects of forestry on a variety of species (that each provides a different function in the ecosystem), assessment of how the habitat has been altered and its effect on biodiversity may possibly be more easily understood.

Specific Taxa as Indicators

Several forest wildlife taxa have been identified as effective indicators of forest health and environmental change. The ground beetle, amphibian, and avian communities represent groups of wildlife that are reasonable to monitor and are often abundant, diverse, and sensitive to environmental change. They also utilize different resources in the forest, from the forest floor to the canopy, providing a unique perspective of the resource utilization by taxa. Evaluation of the species composition and abundance at each of the conifer and hardwood forest habitat types will aid understanding of the relative contribution of each habitat to biodiversity in managed forests.

Ground Beetles

Arthropods represent 65-70% of species in forests (Langor & Spence, 2006) and perform ecological functions such as pollinating, cycling nutrients, dispersing seeds, and controlling invertebrate populations. Carabid beetles are arthropods that make up a wide variety of ground-dwelling beetles with more than 2,000 known species in North America (Langor & Spence, 2006). They are among the best studied taxa regarding the effects of forest management on forest biota (Niemelä et al., 2006). They have been identified as good bioindicators of ecosystem disturbance in forested landscapes because they are diverse, abundant, sensitive to environmental change, and reasonably easy to monitor (Pearce & Venier, 2006). Some studies indicate they poorly represent the species abundance and richness of other taxa (Koivula, 2011), while other studies indicate they are well-suited as representatives (Pearce & Venier, 2006). However, many agree they are most effective as ecological indicators when evaluated in tandem with other environmentally sensitive taxa (Koivula, 2011; Pearce & Venier, 2006; Rainio & Niemelä, 2003).

The diversity of specialized habitat associations known to exist within the ground beetle community make them especially useful in evaluating changes in habitat conditions (Niemelä et al., 1996). In the forested environment, many species have demonstrated a sensitivity to changes in canopy cover (e.g., clearcut harvest), with observed shifts from forest habitat species to open habitat species once the canopy is removed (Pearce & Venier, 2006; Niemelä et al., 2006). Additionally, the creation of canopy gaps within the forested environment has been shown to increase the species richness in members of the ground beetle family (Perry et al., 2018). Small-scale forest disturbances are linked to greater vegetation diversity, influencing microhabitats and leaf litter composition that are known to be favored by members of the Carabidae family (Koivula et al, 1999). The use of ground beetles as bioindicators has led to greater understanding about the importance of heterogenous habitats in the forested environment.

Amphibians

Amphibians have been widely studied in ecological research to increase understanding of the changing environment, pollution thresholds, and climate change (Hopkins, 2007). Amphibians are critical components of both aquatic and terrestrial communities. They occupy diverse trophic niches and often serve as abundant prey sources for wildlife. In some environments, certain species compose the most abundant vertebrate in the population, forming important trophic roles up and down the food web. Their environmental sensitivity, trophic importance, and detectability make them well suited as bioindicators.

Amphibians have complex life cycles, often requiring both aquatic and terrestrial habitats. Environmental variables important to amphibians include suitable breeding, feeding, and resting habitats, which maintain suitable moisture and temperature regimes. The composition and density of the forest canopy is an important driver for controlling light availability to the forest floor, which in turn controls the composition and vigor of the understory vegetation community. In the forested environment, canopy density also regulates forest floor moisture content and temperature regimes. However very few studies describe the composition of the forest canopy or correlate canopy species composition with amphibian abundance and diversity (Bennett et al., 1980; Gomez & Anthony, 1996).

Amongst the amphibian community, the plethodontid salamanders are terrestrial salamanders that live in forested environments across North America. Their sensitivity to environmental change, along with their longevity, small territory size, site fidelity and tendency to occur in high densities have made them presumed indicators of biodiversity and ecosystem integrity in forested environments (Welsh & Lind, 1991). The plethodontid species known to occur in east coast forested habitats are known to thrive in hardwood forests whereas west coast plethodontid species occur primarily in conifer forests. Studies indicate their abundance and density are correlated with microhabitat variables such as moisture content, leaf litter depth, understory vegetation, woody debris, and temperature (Pough et al., 1987; Homyack & Kroll, 2014). Investigations have identified species-habitat relationships with forest edges, forest canopy removal, historical disturbance regimes, and various silvicultural practices.

Songbirds

The distribution and composition of birds across forested landscapes has been well documented. Songbird communities are a species-rich component of many forests. They facilitate important forest ecosystem processes such as nutrient cycling and transfer, seed dispersal, and maintaining balance in invertebrate communities. Abundances and populations of some species have been linked to changes in habitat quality including changes in nesting and foraging habitat. While many species are forest generalists, some are specialized, utilizing specific elements of forest structure and composition, and making them effective bioindicators of forest conditions (Gregory et al., 2003). These qualities make them an excellent taxonomic group for evaluating the difference between forest habitat types.

The managed forest environment is subject to shifting habitat conditions and disturbance. The landscape is primarily conifer dominated, with hardwood vegetation scattered throughout. Hardwood tree and shrub species have been identified as important contributors to avian habitat, either through structural cover and nesting habitat or as a source of prey (Ellis & Betts, 2011). Early seral hardwood cover provides critical nesting and foraging habitat for many neotropical migrants (Ellis & Betts, 2011). Some studies indicate that a decline in early seral habitats may be linked to the decline in populations of several avian species (Keller et al., 2003). Habitat relationships exist throughout natural and managed forests that benefit different species at different times (Hansen et al., 1995). However, the contribution that dispersed hardwood dominated patches, located within the managed forest matrix, make to avian richness and abundance has not been well evaluated.

Conclusion

Forests perform ecosystem functions critical to the health of our planet. Healthy ecosystem function relies in part on the health and function of each of the species within it. Determining scientifically rigorous methods to measure forest biodiversity is important for forest managers who strive to incorporate biodiversity goals into long-term forest management plans. Forestland managers have the opportunity, through intentional forest management activities, to optimize performance of ecological functions while also maintaining alignment with business and societal objectives. Although it is known that conservation of forest biodiversity is important, it is difficult to quantify baseline conditions and measure the effectiveness of efforts to maintain or increase it.

Conservation of biological diversity requires an understanding of how habitat features within ecosystems function to create the diversity of life. Ensuring diversity of habitat or structural features may serve as a suitable proxy for biological diversity however, increased evidence is necessary to understand the relationships and therefore ensure management efforts are appropriately prioritized and successfully meeting biodiversity goals.

Biological response to forest management is complex and mechanisms that shape responses in diversity are variable. Biological indicators can be monitored for change to answer and validate specific species-habitat relationships and to measure the status of biodiversity from landscape to local scales. Clearly articulated objectives are necessary to develop well-designed and effective research strategies. A multi-pronged approach is useful to accurately assess elements of biodiversity at landscape and local scales. However, focus on keystone, umbrella, or indicator species alone does not accurately represent the full spectrum of species within an ecosystem. Hundreds of studies have been conducted to evaluate and validate the use of additional biological indicators to assess ecosystem health (Gao et al., 2015).

Quantification of change in biodiversity, by species or ecosystem richness, can be determined once a reference condition has been established. Repeating methods overtime, providing for the same variables, will allow assessment of what has changed, although it may not allow for understanding why. If conducted in a methodical fashion, the examination of three taxa can be used to measure the relative contribution of forest habitats to the conservation of biodiversity. Ground beetles, amphibians, and forest songbirds have characteristics that make them useful biological indicators in forested habitats.

Chapter 3. Methods

Study Area

The study was conducted within the temperate forests of western Washington, specifically the westside lowland conifer-hardwood and montane mixed-conifer forest habitat types (Chappell, 2001, Figure 1). The topography in the study area is generally mild, with elevations ranging from 128 to 280 meters (Table 1). The climate is mild and typically comprised of wet winters and dry summers. Precipitation occurs most often as rainfall, with 81-114 centimeters in a typical year (Western Regional Climate Center, 2020). Dominant forest trees are Douglas-fir (*Pseudotsuga menziesii),* western hemlock (*Tsuga heterophylla*), western red cedar (*Thuja plicata),* red alder (*Alnus rubra*) and big leaf maple (*Acer macrophyllum).* The understory is often composed of a wide variety of woody shrubs and herbaceous forbs. Natural and anthropogenic influences on the forested landscape have created a mosaic of forest ages and habitat conditions. Private and publicly held Douglas-fir managed forests are intermixed with rural and agricultural communities.

Table 1. Summary of physical characteristics at each forest habitat type where: CON = conifer and HW = hardwood. Source HW = rational for why the hardwood patch exists, Dist. Water = the distance from plot center to the nearest source of water, and Dist. Forest = the distance from plot center to the nearest change in forest habitat. *Root rot, **Depressional feature.

Location	Age	Elevation (m)		Aspect		Source	Dist. Water (m)		Dist. Forest (m)	
		CON	HW	CON	HW	HW	CON	HW	CON	HW
Brooklyn	43	145	160	N	W	RR*	75	95	230	175
Lake Creek	43	280	280	E	E	Dep**	115	115	215	95
Langworthy	30	135	135	NE	N	RR*	120	130	170	120
Redfield	39	140	130	W	SW	RR*	120	120	175	145
Skookum	31	175	200	SW	SW	Dep **	70	85	115	160

Monitoring Design

Five paired plots were established in third-growth forests between the ages of 30 and 43 years old (Table 1). Paired plot locations were selected based on the prevalence of deciduous patches within third-growth conifer dominated forest matrices. Plots were randomly selected from a list of locations that had been assessed for stand characteristics. Stands were considered if they were dominated by Douglas-fir and had average stocking densities. Stands were disqualified if management activities were being conducted within close proximity.

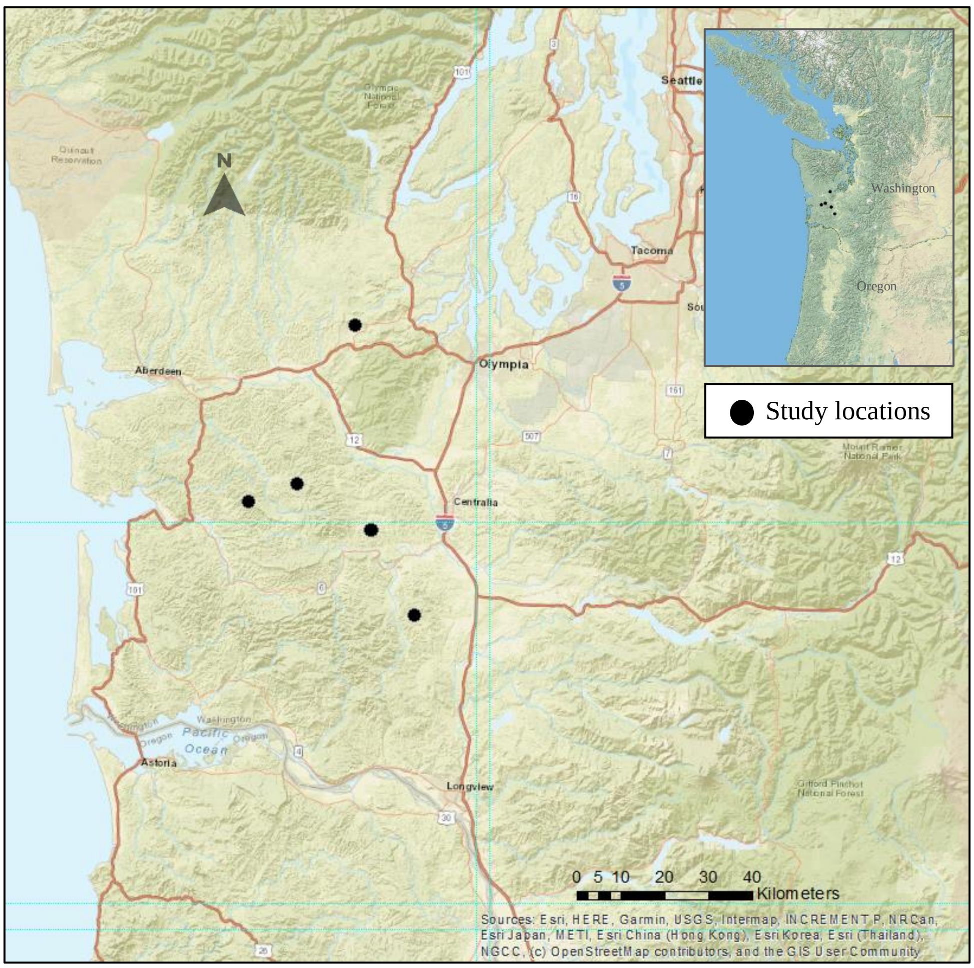

Figure 1. The study was conducted within the temperate forests of western Washington, specifically the westside lowland conifer-hardwood and montane mixed-conifer forest habitat types.

Hardwood dominated patches are the focal points of the study and thus determined the location of the first plot. Each hardwood plot was evaluated to determine the primary functions influencing the hardwood vegetation community and to ensure the presence of water was not persistent (Table 1). Plots were centered within the hardwood patch, with the second plot located within the adjacent conifer forest, located at least 250 meters away. Each plot was 20- by 20-meters in size (0.04 ha), measured from the plot center and oriented in cardinal directions (Figure 2). Hardwood patches were irregularly shaped, however, they all fit approximately within the 20- m by 20-meter plot.

From the hardwood patch, the location of the conifer dominated patch was determined using randomly generated compass bearings, but the end location had to fall at least 250 meters away and meet the conditions identified above. Both plots at a site were established within a forest of the same age and management history. All plots were between 95 and 230 meters away from a habitat boundary as indicated by a change in forest type or seral stage and were at least 70 meters away from aquatic features, such as streams or wetlands.

Forest Structure and Composition

Forest structure and composition variables related to leaf litter depth, woody debris volume, vegetation cover, vegetation composition, canopy cover, and basal area were evaluated across all plots. An assessment of soil moisture regimes was conducted during winter 2020, all sites were considered upland terrestrial habitats based on a lack of hydric soils, hydrology, and hydric vegetation.

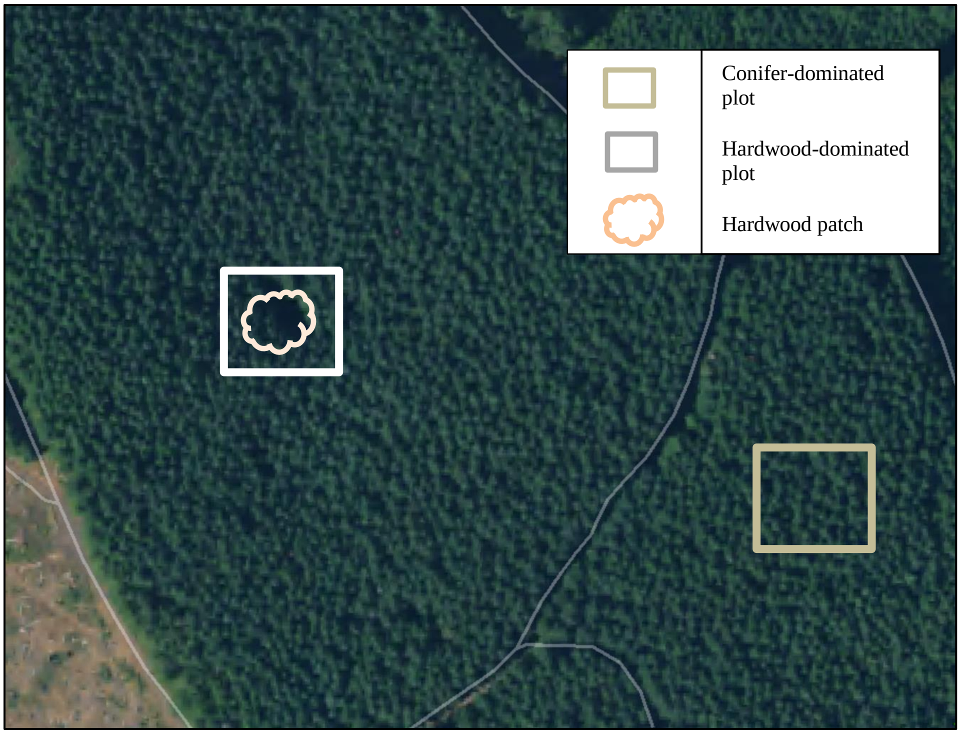

Figure 2. The paired study design was comprised of both conifer- and hardwood- dominated forest habitat types (diagram not to scale). The two forest habitat plots were located at least 250 meters apart from each other and at least 95 meters away from a habitat boundary, as indicated by a change in forest type or seral age.

A grid-based design was established, with the plot center located at the center of the grid and oriented to the north. The 20- by 20- meter plot was composed of 5-meter transect lines established longitudinally and latitudinally within it, creating a 5- by 5-meter grid with 16 subplots and establishing intervals 5 meters apart along each transect line, creating 25 intervals.

Leaf litter was recorded as the depth of the O soil horizon (cm). This was measured at three of the 25 intervals and reported as an average depth per site. Eight of the 16 subplots were randomly selected. Within those eight subplots, ground cover measurements (mosses, forbs, shrubs,) were estimated based on ocular assessments.

Overstory canopy cover was estimated based on the average of four measurements taken with a spherical densitometer at the center of each of the randomly determined subplots. Cover and canopy measurements were averaged across the subplots and reported as an average per site.

Woody debris volume was measured in each of the eight randomly determined subplots. Portions of pieces that spanned outside the boundary of the plot were not measured. The minimum diameter of the pieces measured was 3 cm, slightly smaller than the smallest size that an amphibian had been detected in association with (4 cm). Each piece of woody debris ≥ 3 cm wide and ≥ 10 cm long within the randomly determined subplot was measured and assigned to one of five decay classes as modified from Maser (1979) and identified to species if possible. Pieces were also assessed for if they had fallen naturally or had been cut, as all sites had been previously pre-commercial thinned. Woody debris was reported as the total volume of all pieces within the subplots, per site.

Vegetation composition was determined for the entire 20- by 20-meter plot using simple presence/absence. Mosses, grasses, and vetches were lumped by division or genus and not identified to species, therefore each of those categories counted as 1 if its type was represented. All trees 10 cm in diameter or larger were measured to determine a total basal area by species for each site.

Ground Beetle Surveys

Ground beetle diversity and abundance was measured according to existing protocols (Hoekman et al., 2014). Thirty un-baited pitfall traps were deployed across the ten sites, three each in every conifer and hardwood plot. Traps were deployed near the center of each plot and were equipped with cover boards positioned approximately 1.0 cm above them, so as to avoid being flooded by rainwater and to prevent inadvertent captures of amphibians or other taxa. Traps were deployed at the beginning of June and collected by mid- July. Traps were checked frequently throughout the duration of deployment to reduce specimen loss due to degeneration or predation within the trap. All specimens that were captured were retained for later species identification. Each plot was deployed between 35 to 42 nights, with equal nights for both plots at any given site. Traps were deployed for a combined total of 1158 nights.

Amphibian Surveys

Amphibian surveys were conducted in the spring when they were most likely to be surface active. A crew of two people conducted area-constrained, ‘light-touch’ searches consistent with existing protocols (Corn & Bury, 1990; Wilson, 2016). Surveys were conducted across 100% of the plot area, including all 16 plots generated by the grid design. Three separate surveys were conducted, with a minimum of four days between surveys. Search time per survey was not restricted and varied based on the level of effort required to search all potential habitat structures consistently.

Observers searched under all cover objects (woody debris, bark, and rocks), vegetation (mosses, forbs, and ferns), and probed all crevices. Objects were returned to their original position, and woody features were surveyed only when it could be accomplished without causing habitat destruction. Captured amphibians were identified to species, measured (snout to vent and total length, mm) and photographed. Reptiles that were detected during surveys were identified to species and photographed where possible, but measurement of length was not collected. All captured amphibians and reptiles were retained in conditions emulating the habitat they were found in until surveys were completed for all plots within a site. Once the survey was completed and measurements had been obtained, they were released at the location of capture.

Songbird Surveys

Auditory and visual surveys for forest songbirds were conducted in late spring when they were most likely to be breeding (late- May to early- July). Point-count surveys were conducted consistent with protocols identified by Ralph et al (1995). Surveys were conducted from each plot center and included all birds that could be heard or seen from the plot center. Three separate surveys were conducted, with a minimum of four days between surveys. All birds that were detected by sight or sound during 30-minute survey durations were recorded. Surveys were performed between sunrise and 10:00am, during calm weather conditions. All detected birds were allocated to a category of either ‘in’ the plot or ‘out’ of the plot depending on where the bird was active at the time it was observed.

Data Analysis

All individuals that were detected during surveys were identified and recorded (Appendix 1), however, only specific individuals and species were included in the analyses. For the beetle analysis, only species detected from the Carabidae family were included. For the amphibian analysis, an additional taxonomic group was detected while conducting surveys and was included (reptile). For the rest of this thesis, where amphibians and reptiles were combined in the analysis, they are collectively referred to as herpetofauna. For the bird analysis, only individuals that were observed utilizing forest habitats were included (individuals observed flying overhead were excluded). To aid visualization of species occurrences across locations and habitat types, a checkerboard plot was generated (Appendix 2).

A relative abundance index was generated for carabid beetles and herpetofauna to account for varying survey efforts. For beetles, because the duration of trap deployment varied from site to site, I divided the total number of detections at each site by the number of trap nights for that site. The number was then multiplied by 100 for easier visual interpretation. For herpetofauna, because the duration of the survey varied from visit to visit and from site to site, I divided the total number of detections at each site by the combined duration of survey time in hours at each site. Bird surveys were all conducted for 30-minutes per visit, and each site was visited three times, so abundance totals were not adjusted by effort. The relative abundance index results were used to develop a correlation matrix, and to calculate descriptive statistics and paired t-tests using Microsoft Excel (2009).

The correlation matrix was generated to measure the strength and direction of the relationship between communities (overall and by taxa) and habitat patch forest structure and composition variables (Appendix 3) as well as with forest stand scale physical characteristics (Appendix 4). Descriptive statistics and paired t-tests were calculated to evaluate the difference between richness and abundance means for all faunal categories and forest structure attributes at conifer and hardwood plots. Using the R statistical computing platform (R Core Team, 2020), c-scores were created using the package 'EcoSimR' (Gotelli et al., 2015) to evaluate species co-occurrence across all possible species pairs. Co-occurrence evaluations were also conducted using the package ‘cooccur’ (Griffith et al., 2016) to evaluate whether certain species were either more or less likely to occur in the same site compared to random chance alone.

I normalized species abundances to determine how common or rare a species was compared to other species that were detected across all taxa. To do this, I divided the number of detected individuals at a site by the maximum number of individuals detected across all sites. Once the abundances were normalized, I used the Bray-Curtis distance index (Bray & Curtis, 1957) to quantify taxonomic dissimilarity for each taxon. I then used nonmetric multidimensional scaling (NMDS) in the package ‘vegan’ (Oksanen, 2019) to characterize species associations with each of the conifer and hardwood plots and to visualize associations with forest structure and composition variables related to leaf litter, woody debris volume, vegetation cover, and basal area. NMDS was performed for all species combined as well as at the level of the taxonomic group (ground beetles, herpetofauna, birds). Additionally, using the package ‘lme4’ (Bates et al, 2014), I ran a mixed-effects model, with the habitat types treated as the fixed effect and the locations treated as the random effect. For this analysis, the 8 randomly selected subplots (see above) in a given plot were the individual sample units. To compare communities of all hardwood vs. all conifer plots, as well as communities by location, I performed a post-hoc analysis of similarity test (ANOSIM) in the package ‘vegan’.

With five paired plots, the analysis of the data was primarily exploratory and not focused solely on statistically significant relationships. Multiple approaches were used to determine if patterns or trends emerged that could be useful in future studies. An alpha of 0.10 was chosen to increase the power of the individual statistical tests (with a small sample size), and the *p*-values shown in individual scatter plots were not corrected for multiple comparisons.

Chapter 4. Results

Forest Structure and Composition

Leaf litter depth was similar between conifer- and hardwood-dominated plots (while controlling for the random effect of location ($p = 0.01$), fixed effect of habitat type: $F_{1, 34} = 1.80$, $p = 0.19$; Table 2). Percent of moss ground cover was also similar (while controlling for the random effect of location ($p > 0.99$), the fixed effect of habitat type: $F_{1, 78} = 2.73$, $p = 0.10$; Table 2). However, significantly more forb and shrub ground cover occurred at hardwood-dominated plots as compared to conifer-dominated plots (while controlling for the random effect of location forb: ($p < 0.001$) fixed effect of habitat type: $F_{1, 74} = 48.15$, $p < 0.001$; shrub: ($p = 0.73$) fixed effect of habitat type: $F_{1, 74} = 32.95$, $p < 0.001$; Table 2) and forb and shrub cover were positively correlated with plant richness (forb: Pearson's $r = 0.74$, $p = 0.02$; shrub: Pearson's $r = 0.62$, $p = 0.05$).

Woody debris volume in conifer- and hardwood-dominated plots was similar ($t_{(4)} = 0.35$, $p = 0.74$; Table 2, Figure 3). At the plot scale, an average of 9.8 cubic meters of woody debris was measured at conifer plots (range = 6.2-16.3, SD = 4.0) and an average of 9.1 cubic meters was measured at hardwood plots (range = 2.7-14.8, SD = 4.6; Table 2). All conifer plots had woody debris pieces that were sourced from cut pieces (from prior pre-commercial thin management). Of the woody debris pieces measured in conifer plots, an average of 2.5 cubic meters of woody debris were sourced from cut pieces (range = 1.1-3.6, SD = 1.0), representing 9 to 58 percent. In the hardwood plots, three out of the five plots had woody debris pieces that were sourced from cut pieces. Of those three plots, an average of 1.4 cubic meters were sourced from cut pieces (range = 0-1.7, SD = 0.2), representing 8 to 50 percent.

Tree basal area in conifer- and hardwood-dominated plots was similar ($t_{(4)} = 2.06$, $p = 0.11$; Figure 4). At the individual plot scale, an average of 2.5 square meters of basal area was measured at conifer plots (range = 1.8-4.3, SD = 1.0) and an average of 1.0 square meters was measured at hardwood plots (range = 1.2-1.6, SD = 0.6; Table 2).

Sixty plant species were identified across all surveys with 42 species observed in conifer-dominated plots and 54 species observed in hardwood-dominated plots (Appendix 1). More plant species were found in hardwood-dominated plots than conifer-dominated plots ($t_{(4)} = -4.63$, $p = 0.01$; Figure 5). Of the species observed in each habitat type, five were unique to conifer-dominated plots and 13 were unique to hardwood-dominated plots (Appendix 1). At the individual plot scale, an average of 16.6 species were observed at conifer plots (range = 8-21, SD = 5.4) and an average of 24.8 species were observed at hardwood plots (range = 22-29, SD = 2.6; Table 3).

Two Washington noxious weeds were identified during surveys. Robert geranium (*Geranium robertianum*), a Class B noxious weed, was observed in one plot each of both conifer and hardwood sites, while English holly (*Ilex aquifolium*), a species on the State monitor list, was observed at one hardwood site (Appendix 1).

Table 2. Summary of forest structure and composition attributes at each forest habitat plot. CON = conifer and HW = hardwood.

Location	Leaf Litter (cm)		Woody Debris (m^3)		Basal Area (m^2)	
	CON	HW	CON	HW	CON	HW
Brooklyn	3.0	2.6	11.1	7.5	4.3	0.0
Lake Creek	5.9	4.4	16.3	12.2	2.3	0.9
Langworthy	3.3	3.0	6.2	2.7	2.0	1.0
Redfield	2.0	2.5	7.5	14.8	1.8	1.6
Skookum	5.3	3.6	8.1	8.4	1.9	1.3

Location	Moss Cover (%)		Forb Cover (%)		Shrub Cover (%)		Canopy Cover (%)	
	CON	HW	CON	HW	CON	HW	CON	HW
Brooklyn	29.5	69.1	11.4	27.1	1.6	32.4	100	100
Lake Creek	1.6	68.8	3.7	68.8	5.5	62.5	100	100
Langworthy	68.1	16.3	44.4	74.4	7.0	50.0	100	99.4
Redfield	18.1	61.3	5.1	12.5	2.3	68.1	100	98.9
Skookum	54.4	13.1	48.1	76.3	45.6	17.1	100	100

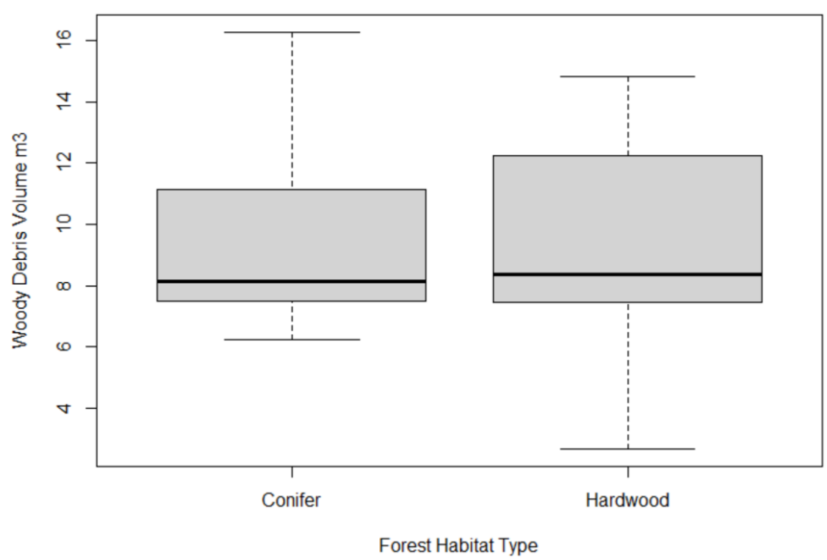

Figure 3. Woody debris volume (≥ 3 cm diameter by ≥ 10 cm length) by forest habitat type was similar ($t_{(4)} = 0.35$, $p = 0.74$).

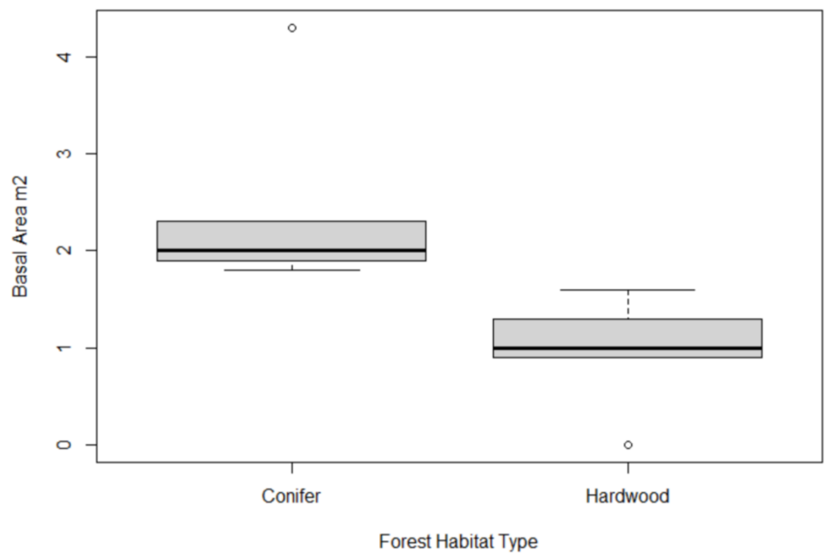

Figure 4. Tree basal area by forest habitat type was similar (conifer basal area > hardwood basal area, $t_{(4)} = 2.06$, $p = 0.11$).

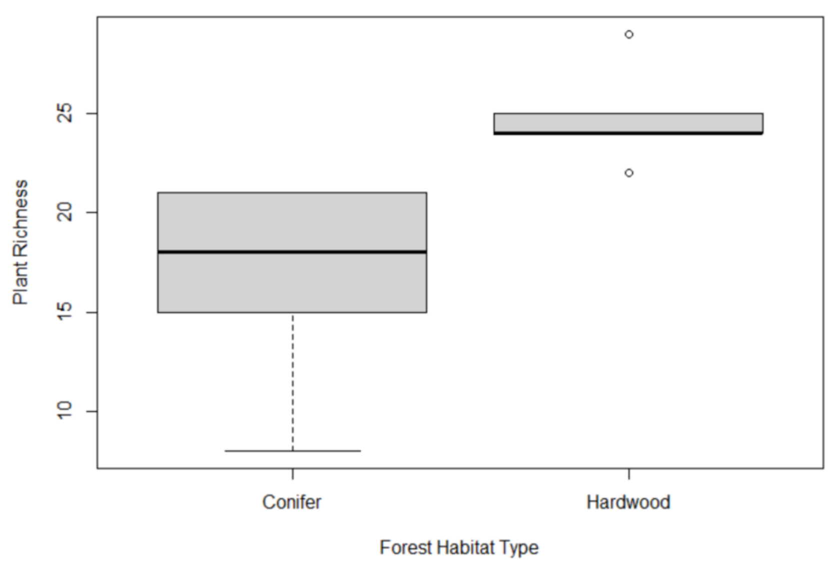

Figure 5. Plant richness (including forbs, shrubs, and trees) by forest habitat type was significantly different (hardwood richness > conifer richness, $t_{(4)} = -4.63$, $p = 0.01$).

Species Richness and Abundance

When individuals and species across all animal taxa were tallied, 388 individuals representing 45 species were observed (Appendix 1). The detection of herpetofauna and birds were not independent, as the same individuals may have been detected during more than one survey. Of the 45 species that were detected, 33 occurred in conifer-dominated plots and 41 occurred in hardwood-dominated plots (Appendix 2). Of those species, four were uniquely associated with conifer-dominated plots and ten were uniquely associated with hardwood-dominated plots.

When all animal taxa were combined, the mean species richness was similar when comparing conifer- to hardwood-dominated sites ($t_4 = -1.11$, $p = 0.33$; Table 3). When NMDS was performed across all taxonomic groups, communities were not significantly different from each other when grouped by habitat type (ANOSIM $R = -0.08$, $p = 0.69$), but *were* different when grouped by location (ANOSIM $R = 0.50$, $p = 0.02$; Figure 6). Pairs of sites tended to be closer to each other in the NMDS ordination plot than to other sites of the same habitat type.

For all taxa combined, there was significantly more species-pair segregation ($p = 0.03$; Figure 7) than expected by chance alone, using 'EcoSimR' to evaluate species co-occurrence. The observed c-score (2.46) is statistically higher than the mean simulated c-score (2.40), although this represents a very small absolute difference (0.06), so the biological meaning of that difference is unclear. For each taxon analyzed individually, ground beetles ($p = 0.48$) and birds ($p = 0.36$) were not significantly aggregated or segregated, but herpetofauna were ($p = 0.08$), compared to randomized occurrences. In the case of herpetofauna, the observed c-score (0.86) was higher than the mean simulated c-score (0.72) by 0.14, which is also a small absolute difference.

Table 3. Summary of species richness by taxa and plant richness detected in each forest habitat plot. CON = conifer and HW = hardwood.

Location	Ground Beetles		Herpetofauna		Birds		Plants	
	CON	HW	CON	HW	CON	HW	CON	HW
Brooklyn	0	2	3	3	12	12	18	25
Lake Creek	3	2	2	6	17	11	15	24
Langworthy	5	5	1	2	8	9	21	29
Redfield	2	3	2	4	9	12	8	22
Skookum	6	3	2	7	11	10	21	24

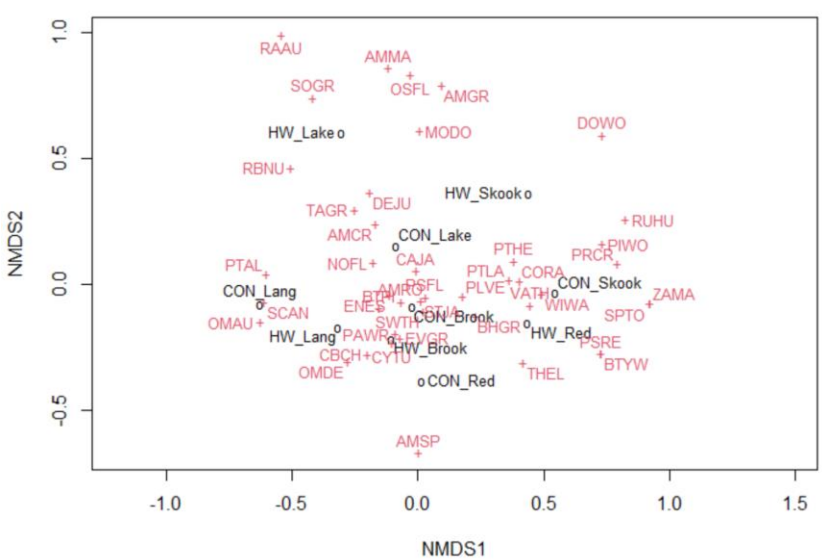

Figure 6. Nonmetric multidimensional scaling ordination performed across all animal taxa in conifer- (CON) and hardwood- (HW) dominated sites were not significantly different from each other when grouped by habitat type (conifer vs. hardwood; ANOSIM R = -0.08, p = 0.69) but *were* different when grouped by location (ANOSIM R = 0.50, p = 0.02). 4-letter species codes shown in Appendix 1. stress < 0.13

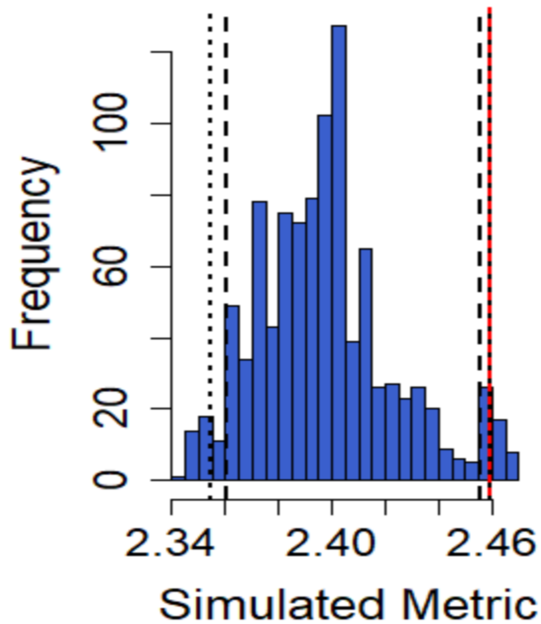

Figure 7. When all animals were combined, the observed c-score (2.46) was statistically higher (p = 0.03) than the mean simulated c-score (2.40), although this represents a very small absolute difference (0.06).

Ground Beetles

A total of 99 individual beetles were captured, representing 12 species and nine genera (*Amara, Cychrus, Necrophilus, Nicrophorous, Omus, Promecognathus, Pterostichus, Scaphinotus, and Zacotus*). Of the nine genera, seven represented the Carabidae family, one represented the Agyrtidae family *(Necrophilius hydrophiloides),* and one was from the Silphidae family *(Nicrophorous defodiens)* (Appendix 1). Members of the Carabidae family comprised 80% of total captures and were the focus of analysis. When all conifer and hardwood plots were combined, nine carabid species were detected in conifer plots and nine carabid species were detected in hardwood plots. Of the species observed, one was uniquely associated with conifer-dominated plots and one was uniquely associated with hardwood-dominated plots (Appendix 2).

Trapping effort and carabid beetle captures by forest habitat were similar (Table 4). The mean species richness of ground beetles captured at conifer and hardwood dominated forest habitats was virtually equal ($t_{(4)} = 0.00$, $p > 0.99$; Figure 8). At the individual plot scale, an average of 3.2 species were detected at conifer plots (range = 0-6, SD = 2.4) and an average of 3.2 species were represented at hardwood plots (range = 2-5, SD = 1.3; Table 4).

Zero to 14 individual carabid beetles were captured in conifer-dominated plots and two to 12 beetles were captured in hardwood-dominated plots (Table 4). Carabid beetles were captured at all sites except one, the Brooklyn conifer site. The mean relative abundance of captured ground beetles for each forest habitat type was similar ($t_{(4)} = 1.40$, $p = 0.24$). The relative abundance of beetles averaged 7.5 in conifer plots (range = 0-12.6, SD = 4.9) and 5.7 in hardwood plots (range = 1.8-9.5, SD = 3.7; Figure 9). NMDS ordination performed on carabid beetles in conifer- and hardwood- dominated sites show a minimal pattern of dissimilarity by habitat type (ANOSIM R = 0.009, p = 0.44; Figure 10).

Table 4. Summary of pitfall trapping effort and carabid beetle captures by forest habitat type. CON = conifer and HW = hardwood.

Location	Trap Nights	Species		Individuals		Relative Abundance	
		CON	HW	CON	HW	CON	HW
Brooklyn	111	0	2	0	2	0	1.8
Lake Creek	105	3	2	6	2	5.7	1.9
Langworthy	126	5	5	13	8	10.3	6.3
Redfield	111	2	4	14	10	12.6	9
Skookum	126	6	3	11	12	8.7	9.5

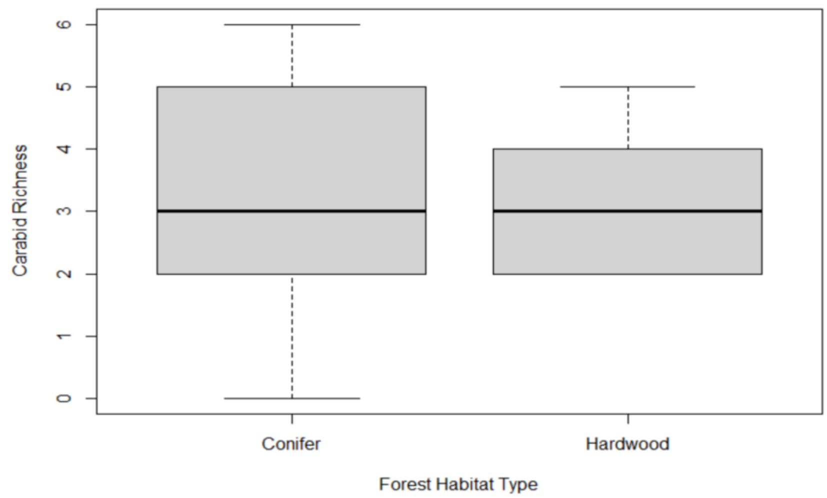

Figure 8. Carabid beetle species richness by forest habitat type was virtually equal ($t_{(4)} = 0.00$, $p > 0.99$).

Figure 9. Carabid beetle relative abundance by forest habitat type was similar ($t_{(4)} = 1.40$, $p = 0.24$).

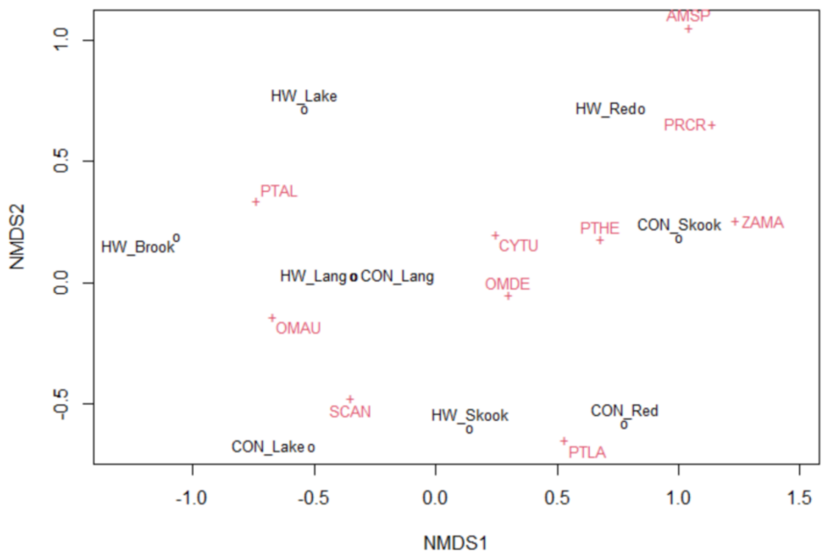

Figure 10. Nonmetric multidimensional scaling ordination performed using carabid beetles in conifer- (CON) and hardwood- (HW) dominated sites show a minimal pattern of dissimilarity by habitat type (ANOSIM R = 0.009, p = 0.44). 4-letter species codes shown in Appendix 1. stress < 0.08

Amphibians and Reptiles

A total of 102 amphibians and three reptiles were captured across all surveys. The detection of all individuals was not independent, as the same individuals may have been detected during more than one survey. Of the 105 individuals that were captured, eight species representing seven genera were detected (*Plethodon, Ensatina, Rana, Ambystoma, Taricha, Pseudacris, and Thamnophis*) (Appendix 1). When all conifer and hardwood plots were combined, three species were detected in conifer plots and eight species were detected in hardwood plots. All species that were detected within the conifer plots were also detected within the hardwood plots, but not the other way around (Appendix 2).

Survey effort by forest habitat type was similar (Table 5). The mean species richness of hardwood dominated plots was greater than in conifer dominated plots ($t_{(4)}$ = -2.59, p = 0.06; Table 5, Figure 11). At the individual plot scale, an average of 2.0 species were detected in conifer plots (range = 1-3, SD = 0.7) and an average of 4.4 species were detected in hardwood plots (range = 2-7, SD = 2.0; Table 5).

Four to 15 individual herpetofauna were captured in conifer-dominated plots and three to 26 individuals were captured in hardwood-dominated plots (Table 5). The mean relative abundance of herpetofauna was similar in both hardwood- and conifer-dominated plots ($t_{(4)}$ = 0.42, p = 0.70; Figure 12). The relative abundance of herpetofauna averaged 2.9 in conifer-dominated plots (range = 1.0-5.0, SD = 1.5) and 2.4 in hardwood-dominated plots (range = 0.8-5.1, SD = 1.8). NMDS ordination performed on herpetofauna in conifer- and hardwood- dominated sites show a minimal pattern of dissimilarity by habitat type (ANOSIM R = 0.14, p = 0.22; Figure 13).

Table 5. Summary of amphibian survey effort and herpetofauna detections by forest habitat type. CON = conifer and HW = hardwood.

Location	Total Survey Hours		Species		Individuals		Relative Abundance	
	CON	HW	CON	HW	CON	HW	CON	HW
Brooklyn	3.0	3.9	3	3	15	5	5.0	1.3
Lake Creek	3.4	4.8	2	6	9	17	2.6	3.5
Langworthy	3.9	4.0	1	2	4	3	1.0	0.8
Redfield	2.8	4.9	2	4	10	7	3.6	1.4
Skookum	3.9	5.1	2	7	9	26	2.3	5.1

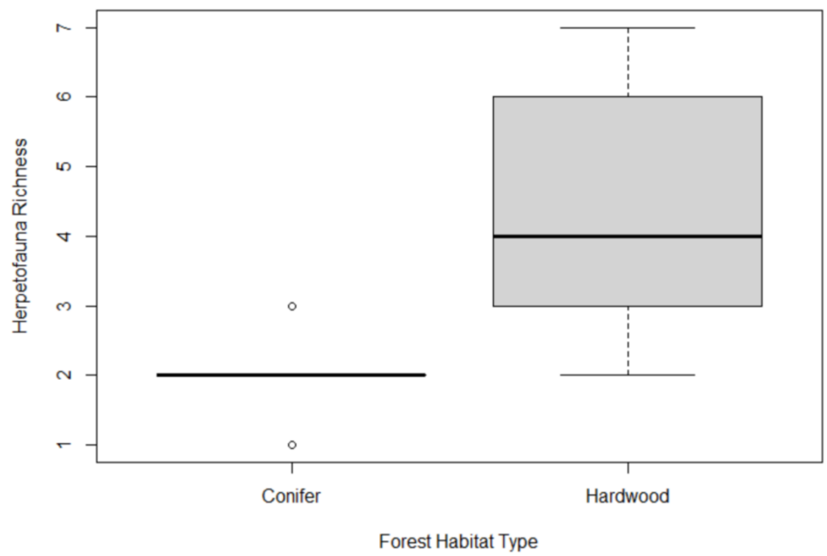

Figure 11. Herpetofauna species richness by forest habitat type was significantly different (hardwood richness > conifer richness, $t_{(4)} = -2.59$, $p = 0.06$).

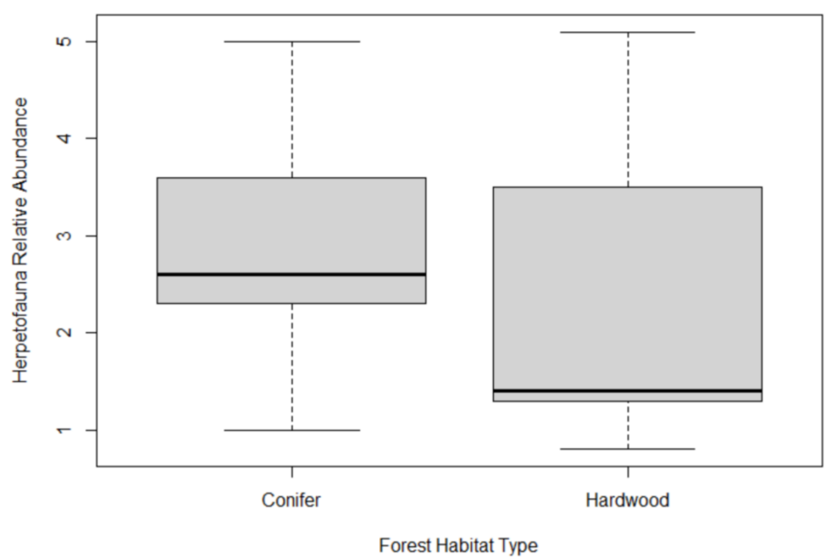

Figure 12. Herpetofauna relative abundance by forest habitat type was similar ($t_{(4)} = 0.42$, $p = 0.70$).

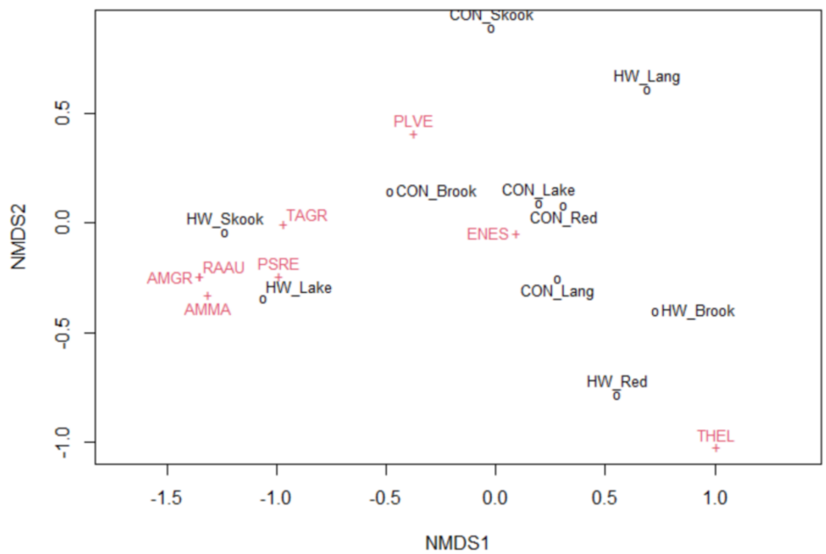

Figure 13. Nonmetric multidimensional scaling ordination performed on herpetofauna in conifer- (CON) and hardwood- (HW) dominated sites show a minimal pattern of dissimilarity by habitat type (ANOSIM R = 0.14, p = 0.22). 4-letter species codes shown in Appendix 1. stress < 0.09

Terrestrial Salamanders

Of the 105 individual amphibians and reptiles that were captured across all surveys, Ensatina and Western redback salamanders comprised 73% of the total. This provided the basis to evaluate more specific associations. The mean relative abundance of both taxa was similar across habitat types ($t_{(5)} = 1.90$, $p = 0.12$, $t_{(8)} = -0.78$, $p = 0.45$, respectively). For Ensatina salamanders, the relative abundance averaged 1.5 in conifer-dominated plots (range = 0.3-2.9, SD = 1.0) and 0.7 in hardwood-dominated plots (range = 0.3-1.0, SD = 0.3; Table 6). For Western redback salamanders, the relative abundance averaged 1.2 in conifer-dominated plots (range = 0.0-2.1, SD = 0.9) and 0.7 in hardwood-dominated plots (range = 0.2-2.4, SD = 0.9; Table 6).

The snout to vent length (SVL) measurement of Ensatina salamanders was similar by habitat type (while controlling for the random effect of location ($p > 0.99$), fixed effect of habitat type: $F_{1, 37} = 0.68$, $p = 0.42$; Figure 14). The average SVL was 43.8 mm in conifer-dominated plots (range = 18.4-59, SD = 9.7) and 40.8 mm in hardwood-dominated plots (range = 18.8-54, SD = 12.9). The snout to vent length measurement of Western redback salamanders was also similar by habitat type (while controlling for the random effect of location ($p > 0.99$), fixed effect of habitat type: $F_{1, 35} = 0.51$, $p = 0.48$; Figure 14). The average SVL was 40.8 mm in conifer dominated plots (range = 18-50.6, SD = 8.9) and 43.1 mm in hardwood dominated plots (range = 16.2-58.9, SD = 11.1).

Ensatina and Western redback salamanders were detected in association with various forest floor habitat features (Figure 15). 96% were detected under a forest floor feature (boulder, woody debris, moss, fern fronds, needles, or bark), while 4% (three Ensatina's) were detected roaming on the surface of the forest floor. Across all surveys, 59% of Ensatina's were detected in association with woody debris cover (n = 23), while 32% of Western redbacks were detected in association with woody debris cover (n = 12) and 42% were detected in association with sword fern frond cover (*Polystichum munitum)* (n = 16). In conifer dominated plots, 71% of Ensatina salamanders were detected under woody debris (n = 17), while in hardwood dominated plots, 40% of individuals were detected under woody debris (n = 6). For Western redback salamanders, 47% of individuals in conifer dominated plots were detected under woody debris (n = 9) and 31% were detected under fern fronds (n = 6), while in hardwood plots, 17% were detected under woody debris (n = 3) and 56% were detected under fern fronds (n = 10).

Table 6. Summary of Ensatina and Western redback salamander detections by forest habitat type. CON = conifer and HW = hardwood.

Location	Individuals				Relative Abundance			
	Ensatina		Western Redback		Ensatina		Western Redback	
	CON	HW	CON	HW	CON	HW	CON	HW
Brooklyn	6	3	6	1	2.0	0.8	2.0	0.3
Lake Creek	5	5	4	2	1.5	1.0	1.2	0.4
Langworthy	4	1	0	2	1.0	0.3	0.0	0.5
Redfield	8	3	2	1	2.9	0.6	0.7	0.2
Skookum	1	3	8	12	0.3	0.6	2.1	2.4

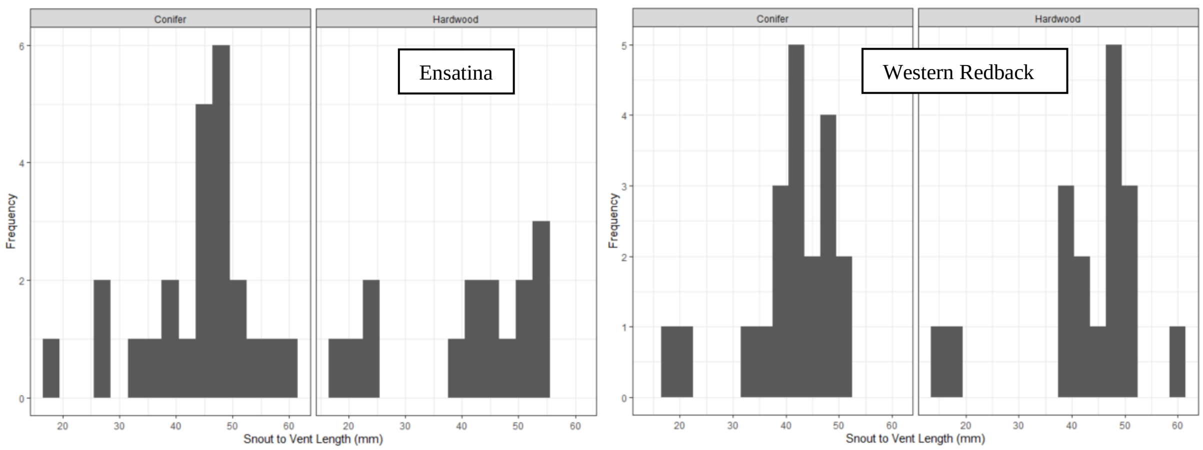

Figure 14. Ensatina (ENES) and Western redback salamander (PLVE) snout to vent length (SVL) measurements in conifer- and hardwood-dominated forest habitat types were similar (ENES: fixed effect of habitat type, $F_{1, 37} = 0.68$, $p = 0.42$; PLVE: fixed effect of habitat type, $F_{1, 35} = 0.51$, $p = 0.48$).

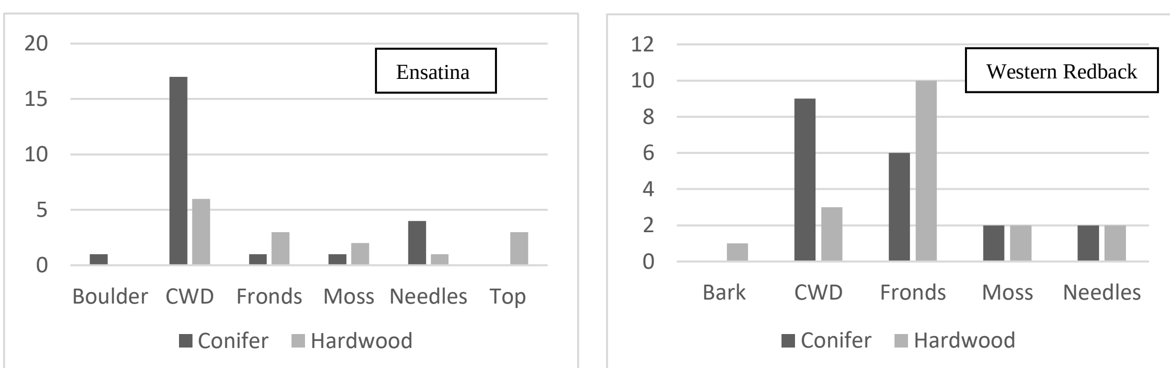

Figure 15. Ensatina and Western redback salamander cover types in conifer- and hardwood-dominated forest habitat types. Three Ensatina salamanders were detected on the surface of the ground (Top). CWD = coarse woody debris.

Woody Debris and Soil Temperature

Woody debris volume is a function of woody debris size. To understand the utilization and selection by salamanders of individual pieces, volume and size were evaluated separately. Woody debris piece sizes that salamanders utilized for cover were significantly different between habitat types when all pieces were considered (while controlling for the random effect of location ($p > 0.99$), fixed effect of habitat type: $F_{1, 33} = 3.08$, $p = 0.09$; Figure 16), however this result was driven by the presence of a single large-sized wood piece. Without that outlier value, piece size used by salamanders were similar between habitat types (while controlling for the random effect of location ($p > 0.99$), fixed effect of habitat type: $F_{1, 32} = 0.24$, $p = 0.63$). The mean diameter of woody debris pieces that salamanders utilized in conifer dominated plots was 12.8 cm (range = 4.5-50.6, SD = 9.8) and the mean diameter of pieces utilized in hardwood dominated plots was 26.7 cm (range = 4.0-120.0, SD =37.6). Of the nine woody debris pieces that salamanders were detected under in hardwood dominated plots, one piece was an outlier at 120 cm. When that piece was excluded from the analysis, the mean piece size in hardwood dominated plots was 15.0 cm.

Soil temperatures where herpetofauna were detected were slightly warmer in hardwood dominated habitat types than in conifer dominated habitats (while controlling for the random effect of location ($p = 0.10$), fixed effect of habitat type: $F_{1, 96.6} = 3.75$, $p = 0.06$; Figure 16). With this p-value there is a difference between the mean temperatures across habitat types while taking into the random effects of location. The mean temperature of the soil in conifer plots where herpetofauna were detected was 10.5 C°

(range = 8.7-13.6, SD = 1.1) and the mean temperature of the soil in hardwood plots where herpetofauna were detected was 11.1 C° (range = 8.8-14.6, SD = 1.5).

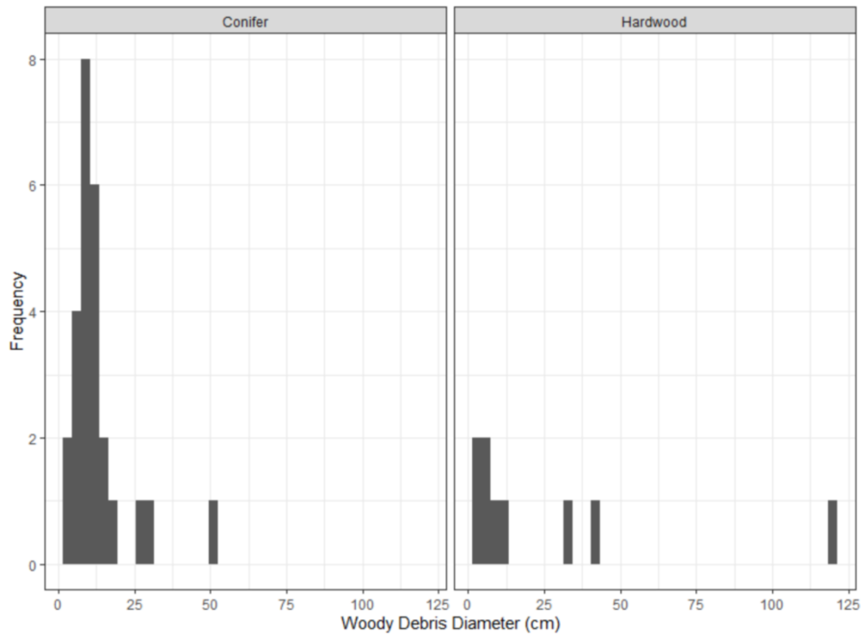

Figure 16. The diameter (cm) of woody debris pieces utilized by amphibians in conifer- and hardwood-dominated forest habitat types was similar when the single large piece in a hardwood plot (120 cm) was removed (fixed effect of habitat type, $F_{1, 32} = 0.24$, $p = 0.64$).

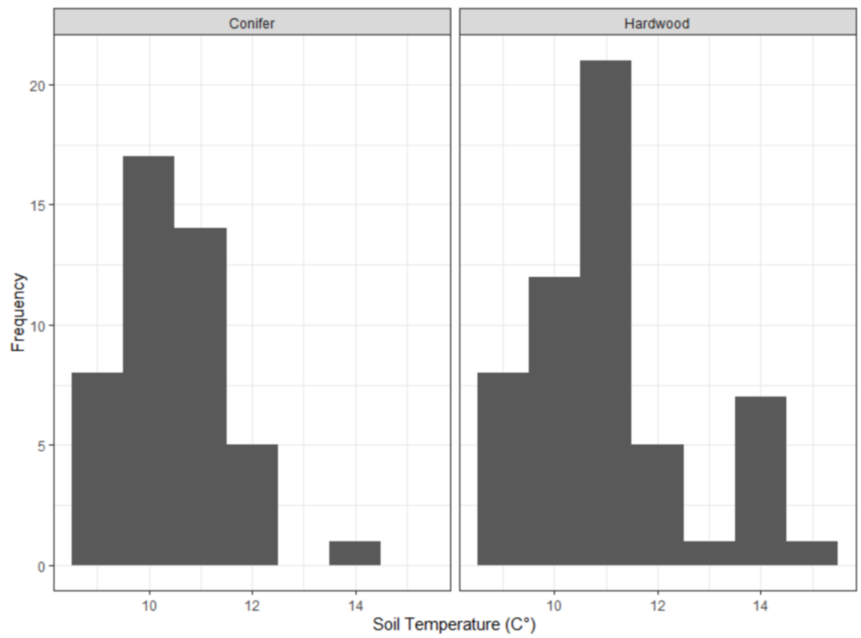

Figure 17. Soil temperatures (C°) measured at the sites of herpteofauna detections were significantly different (hardwood temperatures > conifer temperatures, fixed effect of habitat type, $F_{1, 96.6} = 3.75$, $p = 0.06$).

Songbirds

A total of 205 individual birds were seen or heard during surveys, representing 25 species. Of the 205 birds that were detected, 38 were detected inside the 20- by 20- m plots, while 167 were detected adjacent to, but outside the plots. When all conifer and hardwood plots were combined, 21 species were detected in or around conifer plots and 22 species were detected in or around hardwood plots (Appendix 1). Of all the birds that were detected, three species were uniquely associated with conifer plots and four species were uniquely associated with hardwood plots (Appendix 2). For the following analysis, birds that were detected inside the plots ('in') were evaluated separately from the combination of all birds detected in and around the plots ('all').

The mean bird species richness for detections that occurred in plots were similar by habitat type ($t_{(4)} = -1.17$, $p = 0.31$; Figure 18). For birds that were detected in the plots, an average of 2.6 species were detected in conifer plots (range = 2-4, SD = 0.9) and an average of 4.2 species were detected in hardwood plots (range = 1-6, SD =2.2; Table 7). The mean bird species richness for all birds that were detected was virtually equal by habitat type ($t_{(4)} = 0.00$, $p > 0.99$; Appendix 5, Figure 21). For all birds that were detected, an average of 11.8 species were detected in association with both conifer and hardwood plots (range = 8-16, SD = 3.0 and range = 10-13, SD = 1.3, respectively; Table 7).

Two to five individual birds were detected in conifer-dominated plots and one to six individuals were detected in hardwood-dominated plots (Table 7). Bird abundance for species that were detected in hardwood- and conifer-dominated plots was similar ($t_{(4)} = -0.72$, $p = 0.51$; Figure 19). An average of 3.2 birds were detected in conifer-dominated plots (range = 2-5, SD = 1.3) and average of 4.4 birds were detected in hardwood-dominated plots (range = 1-7, SD = 2.4; Table 7). Bird abundance for all birds detected at hardwood- and conifer-dominated plots was also similar ($t_{(4)} = -0.09$, $p = 0.94$; Appendix 5, Figure 22). The abundance of all birds detected averaged 20.4 in conifer-dominated plots (range = 14-27, SD = 4.7) and averaged 20.6 in hardwood-dominated plots (range = 18-23, SD = 2.3; Table 7).

Nonmetric multidimensional scaling ordination performed on birds detected in conifer- and hardwood- dominated sites show a minimal pattern of dissimilarity by habitat type (ANOSIM R = -0.12, p = 0.79; Figure 20). NMDS performed for all birds that were detected also were not significantly different from each other when grouped by habitat type (ANOSIM R = -0.15, p = 0.89), but *were* significantly different when grouped by location (ANOSIM R = 0.34, p = 0.07; Appendix 5, Figure 23). Pairs of sites tended to be closer to each other in the NMDS ordination plot than to other sites of the same habitat type (Appendix 5). Bird species richness for birds detected in conifer- and hardwood-dominated plots was positively correlated with the percent of shrub cover (Pearson's r = 0.69, p = 0.03) and distance to a change in forest habitat (Pearson's r = 0.59, p = 0.07). However, these p-values were not corrected for multiple comparisons.

Co-occurrence analysis of all taxa pairs in the R package 'cooccur' suggest the Red-breasted nuthatch *(Sitta canadensis)* and Varied thrush *(Ixoreus naevius)* were negatively associated, but there were no other species pairs with positive or negative associations. These two species occurred both in conifer- and hardwood-dominated sites, but never overlapped at sites where they were observed (Appendix 2).

Table 7. Summary of bird detections by forest habitat type. Individuals were tallied based on if they were inside or outside the 20- by 20-meter plot. The "inside plots" category includes birds that were detected utilizing habitat inside the survey plots, while the 'all detections' category includes birds that were detected inside and around the survey plots. Birds that were observed flying overhead were not included. CON = conifer and HW = hardwood.

	Inside Plots				All Detections			
Location	Species		Abundance		Species		Abundance	
	CON	HW	CON	HW	CON	HW	CON	HW
Brooklyn	2	6	2	6	13	13	21	23
Lake Creek	3	3	4	3	16	12	27	19
Langworthy	2	5	3	5	8	10	14	20
Redfield	2	6	2	7	10	13	21	23
Skookum	4	1	5	1	12	11	19	18

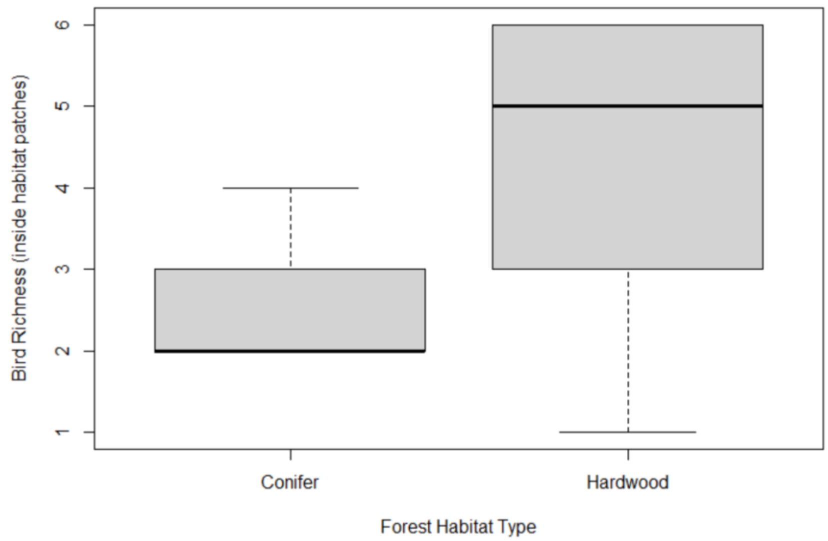

Figure 18. Songbird species richness (inside habitat patches) by forest habitat type was similar ($t_{(4)} = -1.17$, $p = 0.31$). See Appendix 5 for a bird richness plot for all birds that were detected.

Figure 19. Songbird species abundance (inside habitat patches) by forest habitat type was similar ($t_{(4)} = -0.72$, $p = 0.51$). See Appendix 5 for a bird abundance plot for all birds that were detected.

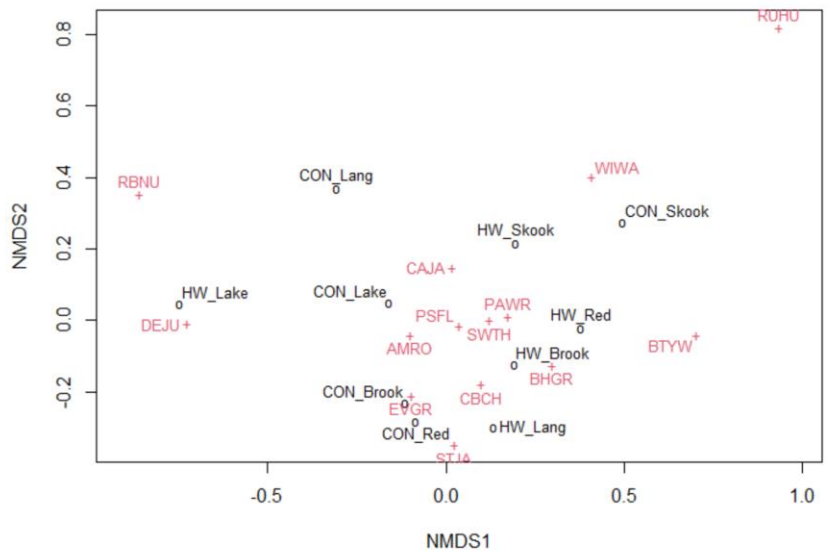

Figure 20. Nonmetric multidimensional scaling ordination performed using birds detected inside conifer- (CON) and hardwood- (HW) dominated sites show a minimal pattern of dissimilarity by habitat type (conifer vs. hardwood; ANOSIM R = -0.12, p = 0.71. 4-letter species codes shown in Appendix 1. See Appendix 5 for an NMDS plot for all birds that were detected. stress < 0.13.

Chapter 5. Discussion

Upland conifer- and hardwood-dominated forest habitat types contributed to species richness in managed forestlands in this study. Differences by habitat type were not significant across all taxa, but results indicate a contribution to both structural diversity and species richness by both habitat types. Overall, plant species and herpetofauna richness were greater in hardwood-dominated habitats, as compared to conifer-dominated habitats, and 14 species (31% of the total detected) were observed uniquely in either conifer- or hardwood-dominated habitat types.

The managed forest landscape is often a mix of habitat types, as indicated by forest age and composition, creating an interplay between forest interior and edge habitats. These areas are often highly diverse and productive. Although habitat patches that are located in the interior of some broader matrices appear (and potentially are) fragmented, the isolated patches do not function in isolation. They are functioning within the matrix of forestland that surrounds them and forming edges with adjacent habitats that create opportunities for additional species. Future research that quantified the habitat features and connectivity qualities within the conifer-dominated matrix may aid understanding of species occurrences and distributions in managed forests.

In Washington, forest practice rules require a minimum number of trees to be retained per acre of harvested land. Trees must be a minimum of 12” in diameter to count towards this requirement. Focus on the retention of trees provides a regulatory framework that incentivizes the reforestation of habitats that are not treed such as shrub dominated or seasonally wetted areas, although they may be providing valuable and limited habitat, and are biologically high functioning. Results of this type of research can help improve forest practices and encourage conservation efforts that are focused on maintaining high-functioning habitats across the landscape.

Forest Structure and Composition

Small, upland hardwood patches within the managed conifer matrix were high functioning, with utilization of both habitat types by all taxa. Forest habitat development and availability is intrinsically linked to forest structure and composition. Forests that are comprised of mixed tree species, including hardwoods, provide pathways for solar penetration into the subcanopy environment (Gray et al., 2002). Sunlight is a limited resource for plant growth in the understory of densely planted managed forests. The solar resources available in hardwood gaps provide energy for photosynthesis, encouraging understory plant development. Results in this study were consistent with the results in other studies where plant richness and understory cover were found to be significantly greater in hardwood- dominated habitats, rather than in conifer-dominated habitats (Figure 5).

Woody debris is known to provide important forage and cover habitat for insects and amphibians (Rose et al., 2001). Woody debris volume was similar in both conifer- and hardwood- dominated forest habitat types, however, the source of woody debris varied. Pre-commercial thinning (PCT) is a common forest management practice where densely planted forest stands are thinned to create more space for the remaining trees to grow (Reukema, 1975). This typically occurs in stands that are between 10 and 15 years old. Trees that are cut during the thinning process are retained on the forest floor. Areas that are hardwood dominated do not typically meet PCT management criteria (i.e., they are not overcrowded). In this study, forest floor woody debris that had been sourced from cut pieces represented 18% of the total volume and 14% of the total pieces, with the majority of cut pieces observed in conifer-dominated plots.

Species Richness and Abundance

Analysis of species richness when all four animal taxa were combined did not appear to have meaningful differences by habitat type, however, they did appear to have meaningful differences by location. When NMDS was performed across all taxonomic groups and was grouped by location, communities were significantly different from each other (Figure 6). This indicates that species compositions were more similar when they were located near to each other, rather than when evaluated by habitat type. Further analysis of this pattern suggested that the distribution of the bird community was primarily driving this result. When examining species co-occurrences there was significantly more species-pair segregation than expected by chance, but the absolute difference in the observed c-score (0.06 higher than the mean simulated score) made this of limited practical significance.

Measuring species presence and distribution allows for understanding how species are distributed across habitat types. Of the 45 species that were included in the analysis, 14 (31%) were unique to one habitat or the other, with four species unique to conifer habitats and ten species unique to hardwood habitats (Appendix 2). Although most of the species were widely distributed across the two habitat types, the presence of rarer species indicates some potentially unique habitat associations. Typically, these results would suggest that the widely abundant species are habitat generalists, adapted to depend on a wider range of habitat resources, while the rarer species may be more specialized and adapted to specific habitat resources. To understand the relative importance of maintaining upland hardwood habitats across the forested landscape, additional studies focused on habitat utilization by the species that were detected in association with one habitat type or the other would be beneficial.

Ground Beetles

Carabid beetle species richness and abundance at each of the conifer- and hardwood-dominated forest plots during 1158 trap nights was similar (Figures 8 & 9). Two carabid species (of the ten detected) occurred uniquely at either one site or the other. Previous research suggests we may have expected to see a difference in the species richness between the two forest habitat types due to differences in habitat preferences (Perry et al., 2018). A larger sample size and expanding the trapping effort to span greater seasonal variation may be valuable considerations during future studies.

Amphibians and Reptiles

The results of this study suggest that upland hardwood-dominated habitats are high-functioning for the herpetofauna community. Herpetofauna species richness was significantly greater in hardwood-dominated plots than in conifer-dominated plots (Figure 11), however, both habitats supported similar numbers of individuals (Figure 12). Using NMDS, there was significant overlap in conifer and hardwood herpetofauna communities where three species (of the eight detected) were common across both habitat types and the remaining five species were unique to hardwood-dominated habitats (Figure 13). An evaluation of woody debris volume (m^3), and the moss, forb, and shrub cover components did not help explain the variance. Differences in habitat utilization are possibly linked to seasonal moisture regimes and the diversity and abundance of prey species that exist in the microhabitats dominated by hardwood vegetation. These components may be useful co-variates to consider during future studies.

Soil temperatures measured at each herpetofauna detection site were significantly warmer in hardwood-dominated plots than in conifer-dominated plots, however the actual mean difference (0.6 C°) may be of limited practical significance. While the minimum temperature was nearly the same between both habitat types, the maximum temperature varied, with higher temperatures reached in hardwood-dominated plots. The variability in the maximum temperature is likely a function of the increased solar penetration typical of conditions in hardwood-dominated forest canopies (Gray et al., 2002). Variability in forest floor temperatures likely have important implications for species occurrences and seasonal distributions.

Terrestrial Salamanders

Ensatina and Western redback salamanders composed 73% of the herpetofauna detections. Abundances of both species were similar across habitat types. Locations of terrestrial salamander detections suggested potential preferences for habitat cover types. Although cover type results varied by species, woody debris and sword fern fronds comprised 69% of total occurrences. The relationship between terrestrial salamander abundance and woody debris volume has been widely recognized (Kluber et al., 2009; Aubry, 2000; Mutts & McComb, 2000). However, in addition to woody debris, the results of this study indicate that sword ferns provide an important source for terrestrial salamander cover, specifically for Western redback salamanders.

Songbirds

Songbird species richness was similar in both conifer- and hardwood-dominated forest habitat types (Figures 18 and 19). However, out of 25 detected bird species, seven were uniquely observed in one habitat type or the other, where three species were identified in association with conifer-dominated habitats and four were identified in association with hardwood-dominated habitats. NMDS performed for all bird species were significantly different from each other when grouped by location (Figure 23). This result indicates that bird communities were more similar by location than they were across habitat types. Based on previous studies, we may have expected to see a difference in the species richness between the two forest habitat types (Gregory et al., 2003; Ellis & Betts, 2011). Bird species richness for birds detected inside survey plots was positively correlated with the distance to a change in forest habitat type. This result indicates that habitat isolation and fragmentation may be important factors effecting bird richness and distribution.

Conclusions

Forestlands across the managed landscape form a mosaic of mixed-aged, conifer-dominated habitats, with hardwood-dominated habitats scattered throughout. The species richness and abundance of ground beetle, herpetofauna, and avian communities were evaluated to determine their use of conifer- and hardwood- dominated habitat types. Small, upland hardwood patches within the managed conifer matrix were found to be high functioning, with utilization of both habitat types by all taxa.

Canopy composition was found to be an important driver for regulating forest floor vegetation structure and temperature regimes. Hardwood patches allowed for sunlight infiltration resulting in a higher diversity of plants, while conifer forests were effective at blocking solar infiltration and therefore maintaining cooler soil temperatures. Each forest habitat type was composed of an arrangement of resources, based on vegetation composition, cover, and woody debris components. The diversity of resources available between the two types provided habitat elements beneficial to a diversity species.

Forest management alters the condition of the vegetation community and woody debris components, however, PNW native species are adaptable and have evolved in a disturbance rich environment. Managed forests perform important societal roles and have unique opportunities to maintain forest resiliency and conserve biodiversity. The results of this study indicate that upland hardwood and conifer habitats that exist in managed forests contribute to biodiversity. Maintaining diversity of forest habitats throughout production forests will ensure the ongoing resilience of native species and forest ecosystems. Balancing forest production with biodiversity conservation is a critical and achievable objective.

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Appendices

Appendix 1: Table of species associations per forest habitat type.

Taxa			Forest Habitat Type	
Scientific Name	Common Name	Specie Code	Conifer	Hardwood
Beetles				
Amara spp	Amara spp	AMSP		X
Cychrus tuberculatus	Tuberculate rare snail-eating beetle	CYTU	X	X
*Necrophilus hydrophiloides**	Flat brown scavenger beetle	NEHY		X
*Nicrophorous defodiens***	A sexton beetle	NIDE		X
Omus audouini	Andouin's night-stalking tiger beetle	OMAU	X	X
Omus degeanii	Greater night-stalking tiger beetle	OMDE	X	X
Promecognathus crassus	Smooth take-caution beetle	PRCR	X	X
Pterostichus algidus	No common name	PTAL	X	X
Pterostichus herculaneus	No common name	PTHE	X	X
Pterostichus lama	Giant striated ground beetle	PTLA	X	X
Scaphinotus angusticollis	Narrow-collared snail eating beetle	SCAN	X	X
Zacotus mathewsii	Matthews' angry gnashing beetle	ZAMA	X	
Herpetofauna				
Ambystoma gracile	Northwestern salamander	AMGR		X
Ambystoma macrodactylum	Long toed salamander	AMMA		X
Ensatina eschscholtzii	Ensatina	ENES	X	X
Plethodon vehiculum	Western redback salamander	PLVE	X	X
Pseudacris regilla	Pacific tree frog	PSRE		X
Rana aurora	Red legged frog	RAAU		X
Taricha granulosa	Rough-skin newt	TAGR	X	X
Thamnophis elegans	Western terrestrial garter snake	THEL		X
Birds				
*Accipiter striatus****	Sharp shinned hawk	SSHA	X	
Cardellina pusilla	Wilsons warbler	WIWA	X	X
*Cathartes aura****	Turkey vulture	TUVU	X	
Catharu ustulatus	Swainson's thrush	SWTH	X	X
Coccothraustes vespertinus	Evening grosbeak	EVGR	X	X
Colaptes auratus	Northern flicker	NOFL	X	X
Contopus cooperi	Olive sided flycatcher	OSFL		X
Corvus brachyrhynchos	American crow	AMCR	X	
Corvus corax	Common raven	CORA	X	X
Cyanocitta stelleri	Steller's jay	STJA	X	X
Dendragapus fuliginosus	Sooty grouse	SOGR	X	X
Dryobates pubescens	Downy woodpecker	DOWO		X
Dryocopus pileatus	Pileated woodpecker	PIWO		X
Empidonax difficillis	Pacific-slope flycatcher	PSFL	X	X
Ixoreus naevius	Varied thrush	VATH	X	X
Junco hyemalis	Dark eyed junco	DEJU	X	X

Patagioenas fasciata	Band tailed pigeon	BTPI	X	X
Perisoreus canadensis	Canada jay	CAJA	X	X
Pheuticus melanocephalus	Black headed grosbeak	BHGR	X	X
Pipilo maculatus	Spotted towhee	SPTO	X	
Poecile rufescens	Chestnut backed chickadee	CBCH	X	X
Selasphorus rufus	Rufous hummingbird	RUHU	X	
Setophaga igrescens	Black-throated gray warbler	BTYW		X
Sitta canadensis	Red-breasted nuthatch	RBNU	X	X
*Spinus pinus****	Pine siskin	PISI	X	X
*Spinus tristis****	American goldfinch	AMGO	X	
Troglodytes pacificus	Pacific wren	PAWR	X	X
Turdus migratorius	American robin	AMRO	X	X
*Unknown****	Unknown warbler	UNK	X	X
Zenaida macroura	Mourning dove	MODO	X	X
<u>Plants</u>				
Acer circinatum	Vine maple		X	X
Acer macrophyllum	Big leaf maple		X	X
Adenocaulon bicolor	Pathfinder		X	X
Adiantum pedatum	Maidenhair fern		X	
Alnus rubra	Red alder			X
Arunus dioicus	Goat's beard		X	X
Asarum caudatum	Wild ginger		X	X
Athyrium filix-femina	Lady fern		X	X
Blechnum spicant	Deer fern			X
Bryophyta	Mosses		X	X
Claytonia perfoliata	Miner's lettuce		X	X
Cornus nuttallii	Pacific dogwood			X
Corylus cornuta	Beaked hazelnut		X	X
Dicentra formosa	Pacific bleeding heart		X	X
Digitalis purpurea	Foxglove		X	X
Disporum hookeri	Hooker's fairybell		X	X
Galium spp	Galium spp		X	X
Gaultheria shallon	Salal		X	X
Geranium robertianum°	Robert geranium			X
Geum macrophyllum	Large leaved avens			X
Goodyera oblongifolia	Rattlesnake plantain			X
Holodiscus discolor	Oceanspray		X	
Hydrophyllum fendleri	Fendler's waterleaf		X	X
Ilex aquifolium°°	English holly		X	X
Lapsana communis	Nipplewort		X	X
Lonicera ciliosa	Western trumpet honeysuckle		X	
Mahonia aquifolium	Tall Oregon grape			X
Mahonia nervosa	Oregon grape		X	X
Maianthemum dilatatum	False lily of the valley			X
Marah oreganus	Manroot			X
Oemleria cerasiformis	Indian plum			X
Oxalis oregana	Redwood sorrel		X	X
Poaceae gen spp	Grasses		X	X

Polypodium glycyrrhiza	Licorice fern		X
Polystichum munitum	Sword fern	X	X
Prunus emarginata	Bitter cherry	X	
Pseudotsuga menziesii	Douglas-fir	X	X
Pteridium aquilinum	Bracken fern	X	X
Rhamnus purshiana	Cascara	X	X
Ribes spp	Gooseberry		X
Rosa spp	Rose	X	X
Rubus spectabilis	Salmonberry		X
Rubus ursinus	Trailing blackberry	X	X
Rumex spp	Dock spp		X
Sambucus racemosa	Red elderberry	X	X
Sambucus spp	Elderberry spp	X	X
Smilacina stellata	Star flowered false solomon's seal		X
Stachys spp	Hedge nettle	X	X
Stellaria media	Chickweed		X
Thalictrum occidentale	Western meadowrue	X	X
Thelypteris phegopteris	Narrow beech fern		X
Tolmiea menziesii	Youth on age	X	X
Trientalis latifolia	Broad leaved starflower	X	X
Trillium ovatum	Western trillium	X	X
Tsuga heterophylla	Western hemlock		X
Vaccinium parvifolium	Red huckleberry	X	X
Vancouveria hexandra	Inside out flower	X	
Vicia spp	Vetch spp	X	
Viola glabella	Stream violet	X	X
Viola sempervirens	Trailing yellow violet	X	X

*Agyritadae family

** Silphidae family
*** flyover observation
° noxious weed Class B
°° noxious weed monitor list

Appendix 2: Species occurrences by forest habitat type and location.

	CON Brook	CON Lake	CON Lang	CON Red	CON Skook	HW Brook	HW Lake	HW Lang	HW Red	HW Skook
<u>Beetles</u>										
Giant striated ground beetle		■		■	■		■			■
Greater night-stalking tiger beetle			■	■	■			■	■	
Tuberculate rare snail-eating beetle			■		■			■	■	■
Andouin’s night-stalking tiger beetle		■	■			■		■		
Narrow-collared snail eating beetle		■	■					■		■
No common name (PTHE)			■		■		■	■		
A sexton beetle**						■				■
No common name (PTAL)			■			■				
Smooth take-caution beetle					■				■	
Amara spp									■	
Flat brown scavenger beetle*										■
Matthews’ angry gnashing beetle					■					
<u>Herpetofauna</u>										
Ensatina	■	■	■	■	■	■	■	■	■	■
Western redback salamander	■	■		■	■	■	■	■	■	■
Rough-skin newt	■						■			■
Western terrestrial garter snake						■			■	
Red legged frog							■			■
Pacific tree frog									■	■
Northwestern salamander							■			■
Long toed salamander							■			■

* Agyritadae family

** Silphidae family

	CON Brook	CON Lake	CON Lang	CON Red	CON Skook	HW Brook	HW Lake	HW Lang	HW Red	HW Skook
<u>Birds</u>										
Swainson’s thrush	■	■	■	■	■	■	■	■	■	■
Pacific-slope flycatcher	■	■	■	■	■	■	■	■	■	■
American robin	■	■	■	■	■	■	■	■	■	■
Pacific wren	■	■	■		■	■		■	■	■
Evening grosbeak	■	■	■	■		■	■		■	■
Band tailed pigeon	■	■		■	■	■	■	■		■
Steller’s jay	■	■		■		■	■	■	■	
Black headed grosbeak	■	■		■	■	■		■	■	
Varied thrush	■			■	■	■			■	■
Chestnut backed chickadee	■	■		■		■		■		■
Wilsons warbler		■			■				■	■
Dark eyed junco	■		■				■	■		
Common raven	■	■			■				■	
Red-breasted nuthatch		■	■				■			
Sooty grouse		■					■			
Pine siskin***	■						■			
Northern flicker		■				■				
Mourning dove		■					■			
Canada jay			■			■				
American goldfinch***	■	■								
Turkey vulture***		■								
Spotted towhee					■					
Sharp shinned hawk***				■						
Rufous hummingbird					■					
Pileated woodpecker									■	
Olive sided flycatcher							■			
Downy woodpecker										■
Black-throated gray warbler									■	
American Crow		■								

*** flyover observation

Appendix 3: Correlation matrix – habitat patch structure and composition

Pearson correlation coefficients measuring the relationship between communities (overall and by taxa group) and habitat patch forest structure and composition variables. Some communities are subsets of others (i.e., carabids are a subset of beetles, and birds (inside habitat patches) are a subset of birds (all detections)). Bird observations do not include flyovers.

	All Fauna Richness	All Fauna Abundance	Beetle Richness	Beetle Rel Abundance	Carabid Richness	Carabid Rel Abundance	Herp Richness
All Fauna Richness	1.00	0.55	0.03	-0.09	-0.02	-0.21	0.57
All Fauna Abundance		1.00	-0.03	0.49	-0.20	0.27	0.66
Beetle Richness			1.00	0.65	0.95	0.58	-0.15
Beetle Rel Abundance				1.00	0.47	0.79	0.08
Carabid Richness					1.00	0.57	-0.33
Carabid Rel Abundance						1.00	-0.20
Herp Richness							1.00
Herp Rel Abundance							
Bird Richness (All)							
Bird Abundance (All)							
Bird Richness (In)							
Bird Abundance (In)							
Plant Richness							
Litter Depth cm							
Woody Debris m3							
Moss % Cover							
Forb % Cover							
Shrub % Cover							
Basal Area m2							

											1.00
										1.00	-0.52
									1.00	0.41	-0.45
								1.00	0.11	0.50	-0.33
							1.00	-0.02	-0.53	0.06	0.31
						1.00	0.40	-0.15	0.14	0.03	0.12
					1.00	0.02	-0.33	0.35	0.74	0.62	-0.48
				1.00	0.37	-0.02	0.12	0.43	-0.17	0.65	-0.40
			1.00	0.96	0.45	-0.16	0.01	0.41	-0.10	0.69	-0.49
		1.00	0.41	0.40	-0.27	0.18	0.60	-0.37	-0.62	0.02	0.03
	1.00	0.87	0.28	0.30	-0.13	0.48	0.78	-0.20	-0.47	0.06	0.16
1.00	0.18	-0.01	-0.72	-0.79	-0.36	0.08	0.27	-0.41	-0.02	-0.41	0.47
0.57	0.14	-0.07	-0.19	-0.29	0.33	0.01	0.26	0.03	0.45	0.33	-0.25
-0.16	-0.50	-0.30	-0.22	-0.10	-0.35	-0.14	-0.23	-0.19	0.00	-0.13	-0.11
-0.65	-0.35	-0.34	0.28	0.42	0.34	0.28	-0.33	0.16	0.39	0.38	-0.31
-0.13	-0.55	-0.41	-0.18	-0.14	0.06	-0.27	-0.45	-0.14	0.30	-0.15	-0.39
-0.60	-0.42	-0.40	0.26	0.37	0.48	0.13	-0.41	0.14	0.50	0.36	-0.41
0.66	0.16	0.09	-0.47	-0.49	-0.14	0.10	0.18	-0.51	0.15	-0.24	-0.06
0.26	0.71	0.44	0.03	0.09	0.19	0.65	0.68	-0.17	0.09	0.22	-0.09
Herp Rel Abundance	Bird Richness (All)	Bird Abundance (All)	Bird Richness (In)	Bird Abundance (In)	Plant Richness	Litter Depth cm	Woody Debris m3	Moss % Cover	Forb % Cover	Shrub % Cover	Basal Area m2

Appendix 4: Correlation matrix – forest stand physical characteristics

Pearson correlation coefficients measuring the relationship between communities (overall and by taxa group) and forest stand physical characteristics. Some communities are subsets of others (i.e., carabids are a subset of beetles, and birds (inside habitat patches) are a subset of birds (all detections)). Bird observations do not include flyovers.

	All Fauna Richness	All Fauna Abundanc	Beetle Richness	Beetle Rel Abundanc	Carabid Richness	Carabid Rel	Herp Richness
Distance to Forest Change (m)							
Distance to Water (m)							
Aspect							
Elevation (m)							
Age							
Plant Richness							
Bird Abundance (In)							
Bird Richness (In)							
Bird Abundance (All)							
Bird Richness (All)							
Herp Rel Abundance							
Herp Richness							1.00
Carabid Rel Abundance						1.00	-0.20
Carabid Richness					1.00	0.57	-0.33
Beetle Rel Abundance				1.00	0.47	0.79	0.08
Beetle Richness			1.00	0.65	0.95	0.58	-0.15
All Fauna Abundance		1.00	-0.03	0.49	-0.20	0.27	0.66
All Fauna Richness	1.00	0.55	0.03	-0.09	-0.02	-0.21	0.57

										1.00
									1.00	-0.09
								1.00	-0.22	0.12
							1.00	-0.10	-0.01	-0.11
						1.00	0.40	0.05	-0.01	0.41
					1.00	-0.35	-0.04	-0.23	-0.03	0.41
				1.00	0.37	0.08	-0.21	0.20	0.22	0.42
			1.00	0.96	0.45	0.15	-0.23	0.21	0.21	0.59
		1.00	0.41	0.40	-0.27	0.68	0.31	0.17	0.07	0.06
	1.00	0.87	0.28	0.30	-0.13	0.70	0.54	0.08	-0.25	0.07
1.00	0.18	-0.01	-0.72	-0.79	-0.36	0.26	0.31	0.04	-0.53	-0.17
0.57	0.14	-0.07	-0.19	-0.29	0.33	0.14	0.41	0.23	-0.23	0.20
-0.16	-0.50	-0.30	-0.22	-0.10	-0.35	-0.62	-0.32	0.42	0.29	-0.60
-0.65	-0.35	-0.34	0.28	0.42	0.34	-0.78	-0.19	0.05	0.20	-0.29
-0.13	-0.55	-0.41	-0.18	-0.14	0.06	-0.69	-0.34	0.46	0.13	-0.47
-0.60	-0.42	-0.40	0.26	0.37	0.48	-0.85	-0.26	0.11	0.19	-0.29
0.66	0.16	0.09	-0.47	-0.49	-0.14	-0.08	0.29	0.46	-0.34	-0.42
0.26	0.71	0.44	0.03	0.09	0.19	0.23	0.72	0.15	-0.25	-0.19
Herp Rel Abundanc	Bird Richness	Bird Abundanc	Bird Richness	Bird Abundanc	Plant Richness	Age	Elevation (m)	Aspect	Distance to Water (m)	Distance to Forest

Appendix 5. Songbird species richness and abundance for birds detected within and adjacent to survey plots during point-count surveys by forest habitat type (excluding birds observed flying overhead).

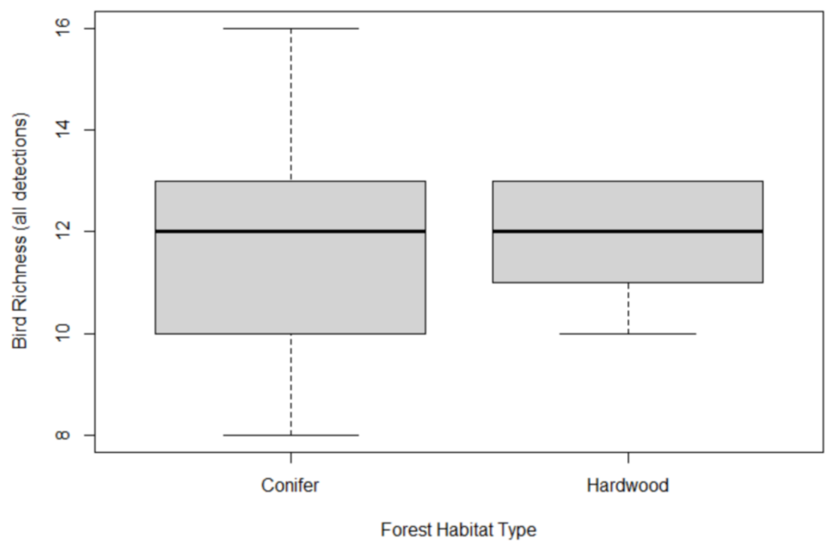

Figure 21. Songbird species richness by forest habitat type was virtually equal ($t_{(4)} = 0.0$, $p > 0.99$).

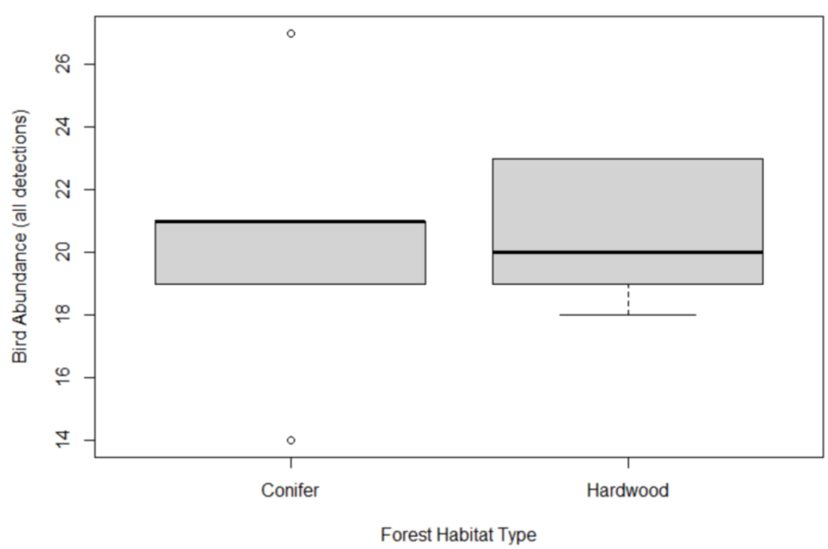

Figure 22. Songbird species abundance by forest habitat type was similar ($t_{(4)} = -0.09$, $p = 0.94$).

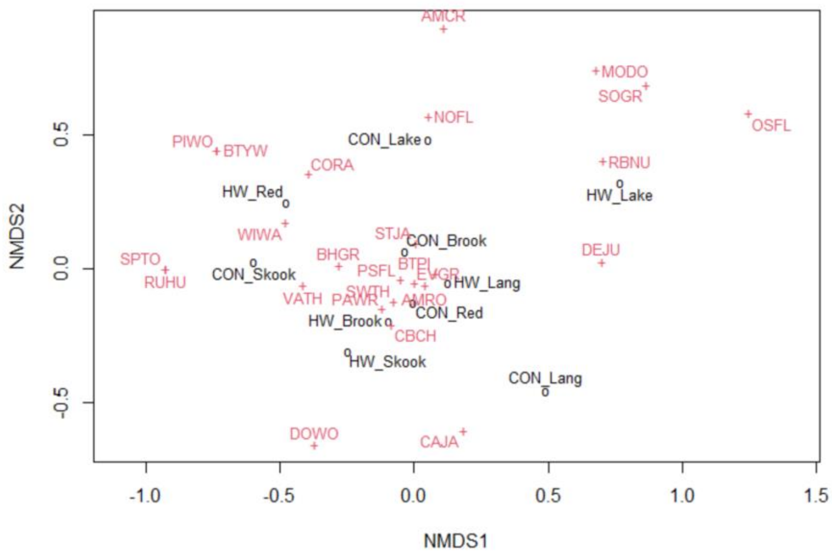

Figure 23. Nonmetric multidimensional scaling ordination performed on all birds detected in conifer- (CON) and hardwood- (HW) dominated sites show a minimal pattern of dissimilarity when grouped by habitat type (ANOSIM R = -0.15, p = 0.89), but were significantly different when grouped by location (ANOSIM R = 0.34, p = 0.07). 4-letter species codes shown in Appendix 1. stress < 0.10