

USING AIRBORNE LIDAR TO MAP HABITAT STRUCTURE AND  
CONNECTIVITY ACROSS ALBERTA'S MANAGED FOREST FOR  
BIODIVERSITY CONSERVATION

by

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## **Abstract**

Vegetation structure is an important biodiversity indicator providing biological and physical environment that supports and maintains forest biodiversity. The airborne lidar (Light Detection and Ranging) systems have the advantage of directly measuring three-dimensional vegetation structure, and have been widely used in wildlife habitat mapping and species distribution modeling at the local scales. As lidar data are increasingly compiled into broad spatial coverage, the development of structural inventory and indicators to categorize habitat types and identify important patches would be beneficial to regional-level conservation planning and biodiversity monitoring. However, this area of research has not been adequately explored. Large-area mapping of critical habitat patches is also a fundamental step towards modeling habitat connectivity. Quantification and dynamic modeling of habitat connectivity under long-term influence of land cover change events provide insights into forest management and conservation planning, and including climate change constraints into the modeling framework also helps maintain ecosystem integrity and improve conservation effectiveness.

Therefore, the objectives of this thesis are to 1) characterize vegetation structure and identify important habitat patches with critical structural traits using regional lidar dataset, and 2) build habitat networks to model connectivity dynamics under land cover change events. To do this, first, a novel approach using cluster analysis to process large-area lidar data into categorical classes representing natural groupings of habitat structure was applied to derive eight unique structure classes in the managed forested area in Alberta, Canada. Second, the structure classes indicating high levels of structure complexity combined with Landsat-derived forest cover types were used

to identify important habitat patches to develop habitat networks. Lastly, spatial prioritization schemes based on different aspects of connectivity and climate constraints were generated and implemented through scenario-based simulations of land cover change events. Connectivity dynamics through the simulations were assessed and compared between scenarios. The result showed that the conservation strategies considering both habitat area and habitat spatial configuration were best at maintaining habitat connectivity, and taking climate constraints into consideration didn't affect overall connectivity. Overall, this research provides an integrated approach to characterize habitat structure using large-area lidar data for dynamic connectivity modeling following land cover change simulations.

## **Lay Summary**

Vegetation structure as the backbone of forest ecosystem, provides the physical and biological environment to support forest biodiversity. Structurally complex stands could potentially provide more habitat niches and have been considered as suitable habitat for many forest species. Landscape-level, spatially-connected habitat patches with high biodiversity potential can sustain ecological processes and species persistence, which are essential for regional-level conservation planning and the maintenance of biodiversity. This research used a regional airborne lidar (light detection and ranging) dataset to identify habitat patches with critical vegetation structure for creating habitat networks and applied network analysis to assess different spatial prioritization strategies in mitigating connectivity loss under forest management activities. The result indicates that both habitat area and their spatial configuration should be considered into conservation planning, and long-term monitoring of connectivity dynamics during land cover change is important to ensure a connected landscape for biodiversity conservation.

## Preface

This thesis is based on two scientific papers for publication in which I am the lead author. The lidar data used in this research were prepared by Ministry of Agriculture and Forestry in Alberta. I am responsible for the primary research development, data processing and analysis, interpretation of the results, and authoring both manuscripts. Dr. Nicholas Coops provided project supervision, guidance on methodology development and result interpretation, as well as editorial assistance throughout my research. Dr. Sarah Gergel provided thoughtful advices around network analysis and modeling development in Chapter 4; Dr. Gergel also offered detailed manuscript writing and editing assistance. Dr. Mark Drever offered consistent advice in habitat connectivity analysis in addition to his work in manuscript editing. Chris Bater from Ministry of Agriculture and Forestry in Alberta provided the pre-processed lidar dataset and constant data support for my research, as well as editorial comments and assistance for both manuscripts. Dr. Scott Nielson and Dr. J. John Stadt provided guidance and editorial assistance for both of the manuscripts. Dr. Piotr Tompalski contributed ideas and discussions in lidar data processing and statistical analysis of the lidar-derived structure classification. Publications arising from this thesis include:

- Chapter 3: Guo, X., Coops, N.C., Tompalski, P., Nielsen, S.E., Bater, C.W. and Stadt, J.J., 2017. Regional mapping of vegetation structure for biodiversity monitoring using airborne lidar data. *Ecol. Inform.* 38, 50–61.
- Chapter 4: Guo, X., Coops, N.C., Gergel, S.E., Bater, C.W.; Nielsen, S.E., Drever, M and Stadt, J.J., 2017. Applying lidar remote sensing data to model habitat connectivity dynamics for spatial conservation prioritization.

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## **List of Abbreviations**

ABMI – Alberta Biodiversity Monitoring Institute

ALS – Airborne Laser Scanning

CanESM – Canadian Earth System Model

CBD – Convention of Biodiversity

CCCma – Canadian Climate Centre’s Modeling and Analysis

CHM – Canopy Height Model

DEM – Digital Elevation Model

EOSD – Earth Observation for Sustainable Development

GIS – Geographic Information System

GPS – Global Positioning System

IIC – Integrated Integral Connectivity

IMU – Inertial Measuring Unit

IPCC – International Panel of Climate Change

NTS – National Topographic System

SD – Standard Deviation

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## **Dedication**

To my family and friends.

我爱你们.

# **Chapter 1: Introduction**

## **1.1 Forest management of structure and connectivity**

Forest stand structure provides the physical environment that maintains and supports forest biodiversity (Moritz, 2001; Marchese, 2015). Forest canopies mediate microclimate, provide perching, nesting, foraging, and covering habitats for many animal species, and influence food quality, diversity and accessibility (Hamer and Herrero, 1987; Johnson et al., 2002). Vertical stratification and structural heterogeneity of forest stands provide ecological niches for species of various habitat specializations, resulting in a variety of bird, mammal and plant species thriving in different layers of the forest canopy where habitat resources are optimized (MacArthur, 1958; Hunter, 1999; Culbert et al., 2013). The energy gradient along the vertical dimension of forest canopy creates a range of microclimates with species diversity positively associated with vegetation structural complexity. As a result, forest horizontal and vertical structure have been identified as essential biodiversity indicators across a broad range of the world's forested ecosystems (Ozanne et al., 2003; Chirici et al., 2011; Gao et al., 2014).

Forest structure is inextricably affected by natural and anthropogenic disturbance regimes, including wildfire, insects, harvesting, and road development, leading to a transformed landscape where habitat loss and fragmentation are inevitable (Tews et al., 2004; Devictor et al., 2008; Boutin et al., 2009; Desrochers et al., 2012). The loss of high-quality habitat, such as habitat with a high degree of structural complexity, can reduce the resources available to wildlife species, impairing forest biodiversity and ecosystem services. The linkage between forest biodiversity and forest structure is a central assumption in ecosystem-based management approaches, where forest

managers aim to maintain the diversity of forest structural attributes at both landscape and stand scales in order to maintain forest biodiversity (Hunter, 1993). Sustainable forest management aims to create stand conditions within the natural range of variability, thereby attempting to mitigate the impact of forest operations on ecosystem integrity (Lindenmayer et al., 2000). For example, wildlife retention trees and thinning practices can create compositionally and structurally complex stands that are important to wildlife habitat.

More importantly, the protection and maintenance of critical forest patches within the existing matrix of forest structure can be an effective way to combat the loss of biodiversity and habitat degradation. The ecological impact of habitat loss depends on not only the reduced habitat area but also the geometry and spatial pattern of habitat fragmentation (Haddad et al., 2015; Keil et al., 2015). Landscape connectivity of suitable habitat patches is important to maintain ecological processes and services such as pollination, seed dispersal, gene flow, wildlife movement and migration (García-Feced et al., 2011). The ability of wildlife species to disperse between available habitat patches, i.e. habitat connectivity, enhances utilization of environmental niches and promotes species diversity. For example, habitat connectivity is essential for maintaining metapopulations through species persistence mechanisms such as species sorting and mass effect, where productivity is mediated and stabilized through energy flow (Thompson et al., 2016). Well-connected habitat patches can also reduce the negative impact of environmental fluctuations and climate change on biodiversity by facilitating species movement, migration and colonization (Thompson et al., 2009). Therefore, biodiversity conservation needs to include the protection of habitat types and their appropriate geographic extents, as well as the connectivity among habitat patches. In sustainable forest management, maintaining a well-connected forest landscape through

carefully designed harvesting patterns and the protection of critical wildlife areas may be among the best strategies to mitigate the potentially negative effects of land cover change on forest biodiversity loss (Bailey, 2007; García-Feced et al., 2011).

## **1.2 Airborne Laser Scanning**

The identification of high-quality habitat patches with valuable structural traits is a fundamental step in biodiversity conservation and lays the foundation for the understanding of habitat connectivity between these patches. Airborne laser scanning (ALS) or light detection and ranging (lidar) is an active remote sensing technology that characterizes habitat structure more efficiently than conventional field assessment or aerial photo-interpretation of vegetation structure, the latter two of which can be spatially limited, labour-intensive and time-consuming (Lefsky et al., 2002; Goodwin et al., 2006; Clawges et al., 2008). Airborne lidar, however, can acquire spatial information in a consistent and systematic manner over a broad spatial coverage. The instrument usually contains a scanning system to emit a laser pulse to the object, a receiver to measure the proportion of returned energy, a global positioning system (GPS), and an inertial measuring unit (IMU) to record the location and orientation of the system (Gatziolis et al, 2008). Lidar systems directly measure the three-dimensional structural information of objects by emitting a laser pulse which strikes the target surface and measures the returned portion of the laser energy to the sensor (Lefsky et al., 2002; Virling et al., 2008). Compared to passive remote sensing technology which records the radiative energy reflected by the surface of the target object from an external illuminating source to the sensor, airborne lidar remote sensing has the advantage of being the emitting source of energy and directly measuring the location of a target object in three

dimensional environment, which allows more control and assurance of the data quality and credibility.

The accurate assessment of vegetation structure using airborne point cloud lidar data is a relatively mature application of the technology in forestry (Wulder et al., 2012). Point cloud data are filtered and classified into ground returns and non-ground returns from which a raster-based digital elevation model (DEM) and canopy height model (CHM) can be generated separately. Lidar-derived structural metrics have been widely used in forestry to directly measure forest height, estimate forest volume, biomass and carbon storage, and assess forest ecological traits (e.g. wood quality, stand structure and development), dramatically improving forest inventory, growth and yield modeling and ecological monitoring (Maltamo et al., 2005; Falkowski et al., 2009; Morsdor et al., 2010; Zald et al., 2014).

In biodiversity monitoring, airborne lidar has been used more recently to derive terrain and habitat structure variables for species occurrence, richness and abundance at various scales. Canopy height, canopy cover, and height distribution and variation are some of the most frequently used variables in habitat modeling. A brief review of literature using airborne lidar data in wildlife and biodiversity monitoring is provided in Table 1-1. Most of the studies focused on relational modeling using lidar-derived variables to predict species distribution and diversity at local scale, whereas regional-level, coarse-filter inventory of habitat structure types is not as well-explored. As airborne lidar data acquisitions have been increasingly compiled into regional coverages, the availability to characterize habitat structure at landscape level to improve biodiversity monitoring

has improved (Asner et al., 2011; Bolton et al., 2013; Hansen et al., 2014). Forest patches with spatially-explicit structural traits of high conservation value can be identified across large landscapes based on lidar-derived structural metrics, providing a unique opportunity to understand the spatial patterns and connectivity dynamics of these potential habitat patches. Therefore, as a fundamental step, forest structure and structure classes need to be mapped at a scale that is suitable for regional-level biodiversity monitoring and forest management.

### **1.3 Biodiversity monitoring and assessment**

Species-based biodiversity monitoring at regional scales requires comprehensive, consistent species data with acceptable quality, which are generally unavailable (Wang et al., 2016). Information on species movement, distribution, morphological and physiological traits, and their response to environmental changes can be gathered within local study areas; however, they are very difficult to obtain at broad spatial scales (Gaston 1996, Green et al., 2005). Indicator species, defined as a small number of species with distribution patterns correlated with species richness of a larger group of organisms, have been used as surrogates in biodiversity monitoring (Fleishman et al., 2005). However, the limited list of indicator species may not represent the full range of trophic levels and habitat specializations, and therefore provide an inadequate measure for biodiversity (Marcot et al., 1994; Boutin et al., 2009; Marchese 2015). An ecosystem-based approach using habitat structure as an indicator to assess biodiversity can provide an alternative way to infer landscape-level, multi-species biological conservation (McCleary & Mowat, 2002). Of particular note, in taking advantage of regional airborne lidar-derived structural metrics, detailed attributes of habitat structure can be characterized to reflect biodiversity potential over broad spatial areas.

Depending on spatial prioritizations, the accumulated effect of land cover change reduces the number of available habitat patches and results in different spatial configurations of habitat, changing the dynamics of landscape connectivity. A graph-based approach depicts habitat patches as a relational network through a collection of nodes and edges (Urban and Keitt, 2001). In this way, isolated habitat patches are spatially-conceptualized into a habitat network where the connectivity between habitat patches can be quantified and evaluated. To monitor landscape-level connectivity dynamics caused by land cover change events, the simplicity and flexibility of graph-based network connectivity analysis allows for the modeling of connectivity loss through a series of patch removals in the network (Calabrese & Fagan, 2004, Bodin & Saura, 2010; Albert et al., 2017). In habitat connectivity analysis, the connectivity is determined by patch area, patch distribution and the distance threshold considered for species dispersal (Araújo et al., 2004; Hodgson et al., 2009; Bergsten et al., 2009). Depending on the spatial configuration of habitat networks, the loss of habitat patches could have different impacts on landscape connectivity. The loss of highly connected network patches can lead to more severe losses in network connectivity than the removal of poorly connected patches (Thompson et al., 2014). Modeling the dynamics of network connectivity through simulations of land cover change evaluates the conservation effectiveness of different spatial prioritization schemes and will provide insights into conservation and forest management planning.

Table 2-1. Summary of studies that use lidar data in wildlife and biodiversity monitoring

<b>Taxon</b>	<b>Location</b>	<b>Scale</b>	<b>Major habitat type</b>	<b>lidar metrics</b>	<b>Major findings</b>	<b>Reference</b>
<i>Vegetation</i>						
Species richness	USA	Sample plots	Coastal marsh, meadow and woodland habitat	Vegetation height	Importance of vegetation height and structural complexity in estimating plant species richness	Lucas et al., 2010
Species richness; community composition	Germany	Local	Mixed montane forests	Vegetation height and height distribution	Lidar predictors were suitable for species richness and community composition prediction complementary to optical remote sensing data	Leutner et al., 2012
Species diversity; community type and structure	South Portugal	Sample plots	Mediterranean oak forests	Canopy cover, vegetation height and height distribution	Lidar-derived structural metrics can be used to map plant species diversity and composition	Simonson et al., 2012
Species abundance	Canada	Local	Mixedwood forest	Terrain, canopy height and height variation	Lidar-derived terrain and structural metrics can explain some drivers of tree species abundance	Ewijk et al., 2014
<i>Bird</i>						
Species occurrence	Switzerland	Landscape	Montane and subalpine forest	Canopy cover and height	Canopy cover was the most important variable indicating species presence and absence	Graf et al., 2008
Species richness	USA	Local	Conifer forests with scattered deciduous stand	Canopy height and height distribution	Foliage layers near the forest floor were strong indicators of bird species richness	Clawges et al., 2008
Multi-year species occurrence	USA	Local	Northern hardwoods forest	Canopy height, cover, vertical complexity derived for waveform lidar	Vertical distribution and canopy complexity were important predictors of multi-year habitat uses	Goetz et al., 2012

<b>Taxon</b>	<b>Location</b>	<b>Scale</b>	<b>Major habitat type</b>	<b>lidar metrics</b>	<b>Major findings</b>	<b>Reference</b>
Species occurrence and abundance	Germany	Sample plots	Mixed montane temperate forests	Canopy height and height distribution	Lidar metrics better explained bird diversity and composition than aerial photo and field assessment	Mueller et al., 2009
Species richness	USA	Local	Two mixed conifer forests	Vertical, horizontal and topographic structural variables	The importance of lidar-derived structural and terrain metrics varied between study areas and among nesting guilds	Vogeler et al., 2014
Organism-Habitat Relationship	England	Local	Ash-oak woodland	Canopy height, height distribution and variation	Strong relationship was found between mean nestling body mass and mean canopy height	Hill and Hinsley, 2015
<i>Beetles</i> Species richness and community composition	Germany	Local	Montane forest	Canopy height and its vertical distribution, height variation	High predictive power using lidar variables to predict mean body size and species composition and assemblage	Muller and Brandl, 2009
<i>Spider</i> Species composition, richness and distribution	Germany	Local	Montane forest	Lidar-derived environmental variables, vegetation height and canopy cover	Lidar variables were comparable or superior than ground-based assessment of spider community characteristics and species distribution	Vierling et al., 2011
<i>Mammal</i> Species occurrence, activity and composition	Germany	Local	Uneven-age, multi-species managed forest stands	Lidar-derived structural metrics of canopy height, gaps, edges and height variations	The occurrence and activity of bat species were positively associated with structural heterogeneity of the managed forest	Jung et al., 2012

<b>Taxon</b>	<b>Location</b>	<b>Scale</b>	<b>Major habitat type</b>	<b>lidar metrics</b>	<b>Major findings</b>	<b>Reference</b>
Habitat classification	Canada	Landscape	Montane to alpine forest	Digital elevation model, canopy height model, canopy density	More detailed, accurate habitat mapping and land cover classification for grizzly bear when incorporating habitat structure and topography	Nijland et al., 2015
Predator-prey interaction	Africa	Local	Succulent ticket vegetation interspersed with open grass land	Digital elevation model and canopy height model to generate viewshed	Sites in dense vegetation were twice as likely to be lion kill sites compared to open areas and vegetation structure is an important factor in predator-prey interaction	Davies et al., 2016

## **1.4 Research questions**

The objectives of this thesis are to address the opportunities presented in section 1.2 and 1.3 to explore the utilization of regional airborne lidar datasets in assessment of habitat structure and connectivity. The airborne lidar point cloud data for the managed forest in the Province of Alberta, Canada were used to characterize vegetation structure and identify important habitat patches for creating habitat networks and modeling their connectivity. Two research questions are posed:

- 1) How can we utilize a regional lidar dataset to characterize vegetation structure over a broad spatial area for biodiversity monitoring? What is the distribution pattern and forest attributes of various identified vegetation structure types?
  
- 2) What is the spatial configuration of habitat patches with important vegetation structure? And how does land cover change based on different spatial prioritization schemes affect habitat connectivity?

This thesis is structured in the following four chapters:

Chapter 2 describes the details of the study area including climate, topography, regionization and species composition, and all the datasets used in the subsequent chapters.

Chapter 3 addresses research question 1. Vegetation structure is characterized through lidar-derived structural metrics. The details about data processing, statistical analysis, results and implications are discussed in this chapter.

Chapter 4 addresses research question 2. Landscape connectivity is modeled through graph-based network analysis. The detailed description of constructing habitat networks, the quantification of network connectivity, and the simulation of land cover change events are discussed in this chapter.

Chapter 5 concludes the thesis by summarizing key results and highlights from Chapter 3 and 4, also discussing limitations of the research and recommendations for future studies.

## Chapter 2: Study Area and Data

### 2.1 Study area

Alberta's landscape is divided into six natural regions based on the influence of climate, topography, and geology (Natural Regions Committee, 2006). The study area is located in the managed forest in Boreal Forest and Foothills Natural Regions, which together cover > 33 million ha of the provincial land base (50%), and support Alberta's biodiversity by providing critical seasonal and permanent habitats to numerous wildlife species (Brandt, 2009).

The Boreal Forest Natural Region (Figure 2-1) is the largest natural region in Alberta. This region is influenced by short summers and long, cold winters, with deciduous, mixedwood and coniferous forest intertwined with extensive wetlands (Natural Regions Committee, 2006). Eight natural subregions occur within this region, with elevations ranging from 150 m to 1100 m above sea level. Upland forests are composed of trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*), white spruce (*Picea glauca*), and jack pine (*Pinus banksiana*), with more pine-dominant stands in areas with higher elevation. Treed wetlands are characterized by black spruce (*Picea mariana*) and eastern larch (*Larix laricina*), with both bogs and fens being common across the landscape.

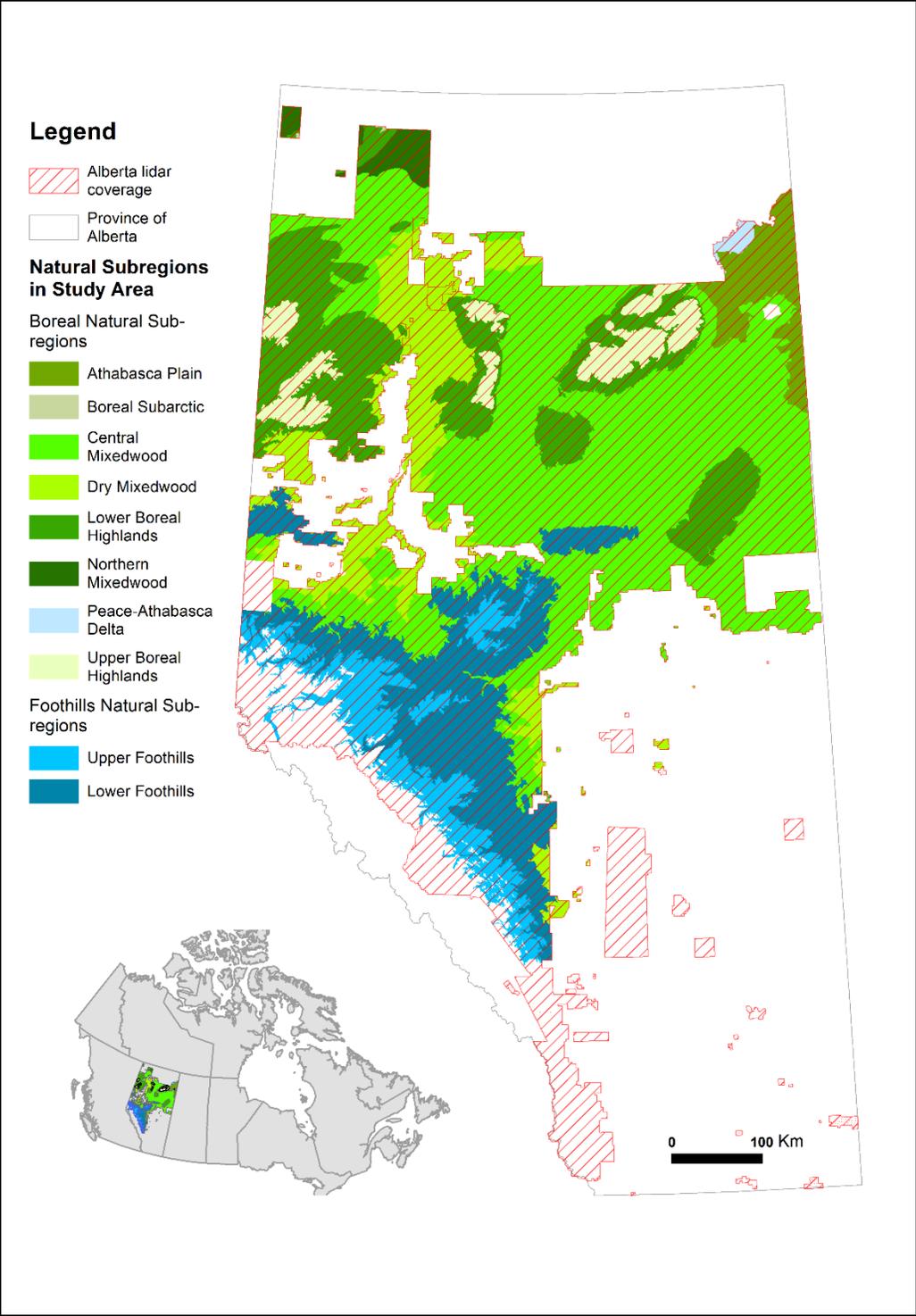


Figure 2-1. Study area: managed forested area in Boreal Forest and Foothills Natural Regions within the boundary of airborne lidar coverage.

Prominent wildlife species of the Boreal Forest Natural Region include moose (*Alces alces*), caribou (*Rangifer tarandus*), black bear (*Ursus americanus*), wolf (*Canis lupus*), beaver (*Castor canadensis*), snowshoe hare (*Lepus americanus*), and many migratory birds such as whooping crane (*Grus americana*). About 110 rare plant species were reported in this natural region across a range of upland and wetland ecosystems (Natural Regions Committee, 2006). Major land uses in this region include agricultural cultivation, oil and gas extraction and harvesting of timber.

The Foothills Natural Region (Figure 2-1) extends east from the flank of the Rocky Mountains, north from the Bow River Valley (51.176°N, 115.570°W) to just south of Grande Prairie (55.170°N, 118.799°W), covering 10% of the province. The region is characterized by a cool, moist climate and long growing seasons, with gently undulating to rolling terrain ranging from 700 m to 1700 m above sea level into the Rocky Mountains, and is divided into two natural subregions. Dominant vegetation cover in the Lower Foothills Natural Subregion is mixedwood forest, consisting of trembling aspen, white spruce, lodgepole pine (*Pinus contorta*) and balsam poplar, whereas the Upper Foothills Natural Subregion is more dominated by uniform lodgepole pine and black spruce stands. This region contains some of the most productive timber areas in Alberta.

Variable topography, surface and groundwater regimes create diverse plant and wildlife communities in the Foothills Natural Region. This region provides critical habitats for grizzly bear (*Ursus arctos*), woodland caribou (*Rangifer tarandus caribou*), wolverine (*Gulo gulo*), and many song bird species, and about 80 rare plant species (Natural Regions Committee, 2006). Major land

uses in this region include open-pit mining, oil and gas extraction, and commercial harvesting of timber.

Within recent decades, the boreal and foothill forests have experienced marked land-use changes with escalated development pressure from forestry and energy exploration sectors (Linke et al., 2005). Development activities have resulted in substantially transformed forested landscapes in the province. For example, the oil sand operations under Lower Athabasca Land Use Framework is one of the largest fossil fuel energy resource development and extraction sites in the world, drawing a great many concerns regarding biodiversity conservation in the province (Kelly et al., 2010). Developing biodiversity indicators and monitoring framework to meet the conservation needs in the province is urgent and necessary.

## **2.2 Data**

### **2.2.1 Airborne lidar**

The Government of Alberta has acquired airborne discrete-return lidar data for over 33 million hectares of managed forested landscape in the province, covering 75% of the Alberta foothills and boreal regions (Figure 2-1). The data used in this study were acquired between 2003 and 2014, with more than half of the data obtained in 2007 and 2008. Multiple lidar contractors contributed to data acquisition with an agreement on general specifications: scan angle of  $< 25$  degrees from nadir, the pulse density ranging from 1 to 4 returns per square meter, and a vertical accuracy of no larger than 30 cm root mean square error (Alberta Environment and Sustainable Resource Development, 2013). Bare-Earth products derived from these data were used to normalize the point

elevations to height above ground level. A suite of forest canopy metrics was generated at 30-meter resolution based on normalized first returns of the lidar point cloud data using a combination of software packages specialized for lidar data processing, including FUSION (McGaughey, 2015) and LAStools (Isenburg, 2016).

### **2.2.2 Land cover map**

The land cover classification for the forested areas of Alberta (Figure 2-2) derived from the EOSD (Earth Observation for Sustainable Development) raster dataset produced by the Canadian federal government (Wulder et al., 2008) was used to distinguish different forest cover types in the study area. The EOSD land cover map was based on image segmentation and class assignment of cloud-free 30 m orthoimages from Landsat-7 satellite between 1999 and 2002, and was later updated to 2010 conditions with ancillary GIS information on natural and anthropogenic disturbance through change detection techniques (Castilla et al., 2014). Eleven classes with an overall accuracy of 75% were achieved, among which coniferous forest, broadleaf forest, and mixedwood forest represent the three upland forest cover types used in this research. A binary wetland inventory, derived from the ecosystem classification system developed by Corns and Annas (1986) and Beckingham et al. (1996), was used to identify wetland forest patches for the study area.

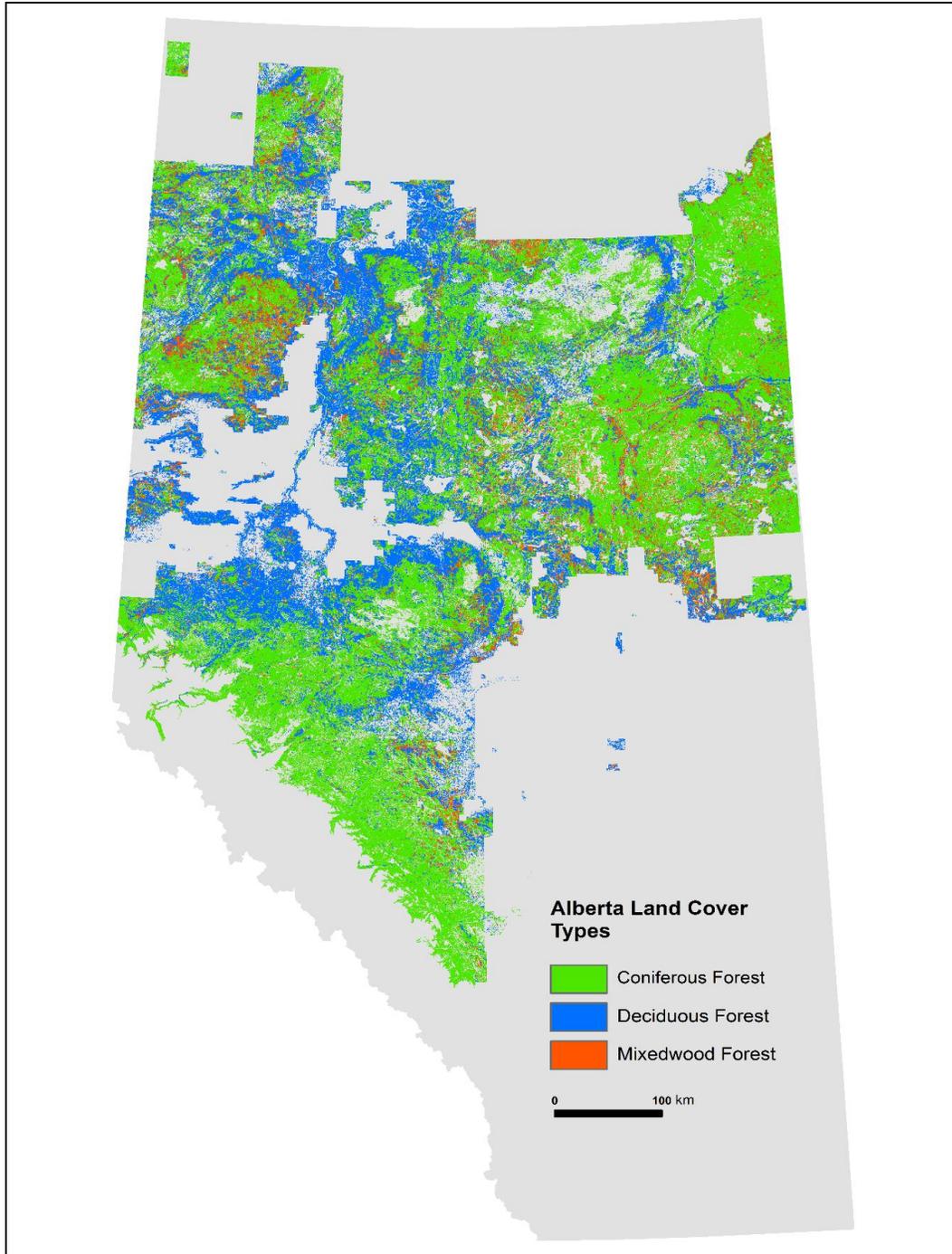


Figure 2-2. Coniferous, Deciduous and Mixedwood forest cover types derived from the EOSD land cover classification for study area.

### **2.2.3 Climate data**

Current climate data were obtained from ClimateWNA (<http://cfcg.forestry.ubc.ca/projects/climatedata/climatebcwna/>), a program that downscales PRISM (Parameter-elevation Regressions on Independent Slopes Model, dataset Norm71m) dataset to a 1-km resolution to provide climate information in Western North America (Daly et al., 2008; Wang et al., 2012). The future climatic condition was projected using the Canadian Climate Centre's Modeling and Analysis (CCCma) CanESM2 model for the year 2050 based on medium carbon emission scenario (RCP4.5) (Arora et al., 2011; Amanda et al., 2016). The CanESM2 model is an improvement over the first generation of Canadian Earth System Model (CanESM1) and has been used in the IPCC (International Panel of Climate Change) fifth climate change assessment report (Chylek et al., 2011). Based on this model, the majority of the boreal and foothill regions of Alberta will experience increased temperature, precipitation and drought in the summer, and declined snow cover and frost in the winter (MacDonald et al., 2012; Schneider 2013).

### **2.2.4 Disturbance data**

The Alberta historical wildfire database is a polygon-based GIS (geographic information system) layer with spatial information of large fires (> 200 ha) since 1961 (Alberta Environmental Protection, 1991). Wildfire information from this database between 1980 and 2010 was used in this research. An anthropogenic disturbances database was derived from a nationwide inventory of polygonal disturbance types across the Canadian boreal ecosystem based on the interpretation of 2008 - 2010 Landsat images at 1:50,000 viewing scale (Pasher et al., 2013) (Figure 2-3).

Cutblocks, agricultural development, oil and gas exploration, and urban settlement were the major disturbance types recorded in the database.

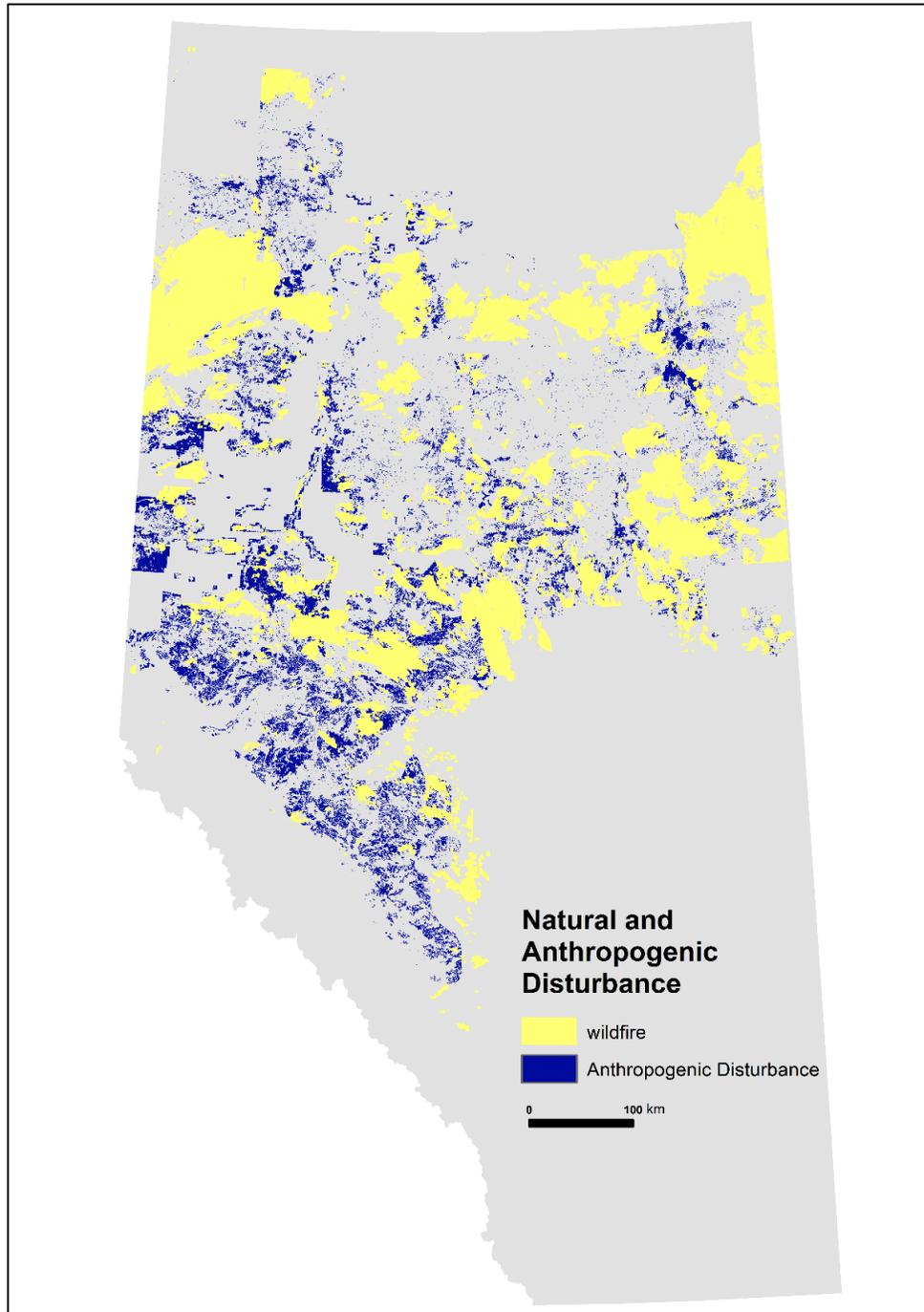


Figure 2-3. Wildfire and anthropogenic disturbance for area within the lidar coverage

### **2.2.5 Vegetation inventory plot**

The Alberta Biodiversity Monitoring Institute (ABMI) has established a provincial monitoring system with 1,656 rectangular permanent photo plots (3 km north to south, 7 km west to east) distributed in a 20-km systematic grid covering 5% of Alberta's land base to evaluate and report biodiversity status in the province (Castilla et al, 2013). Images acquired between 2008 and 2011 were used for analysis to generally match the period during which the anthropogenic disturbance layer was created (Figure 2-4). Within each photo plot, detailed vegetation inventory and land use classification with a minimum mapping unit of 2 ha were produced based on high-resolution (0.30 m spatial resolution) aerial photo-interpretation.

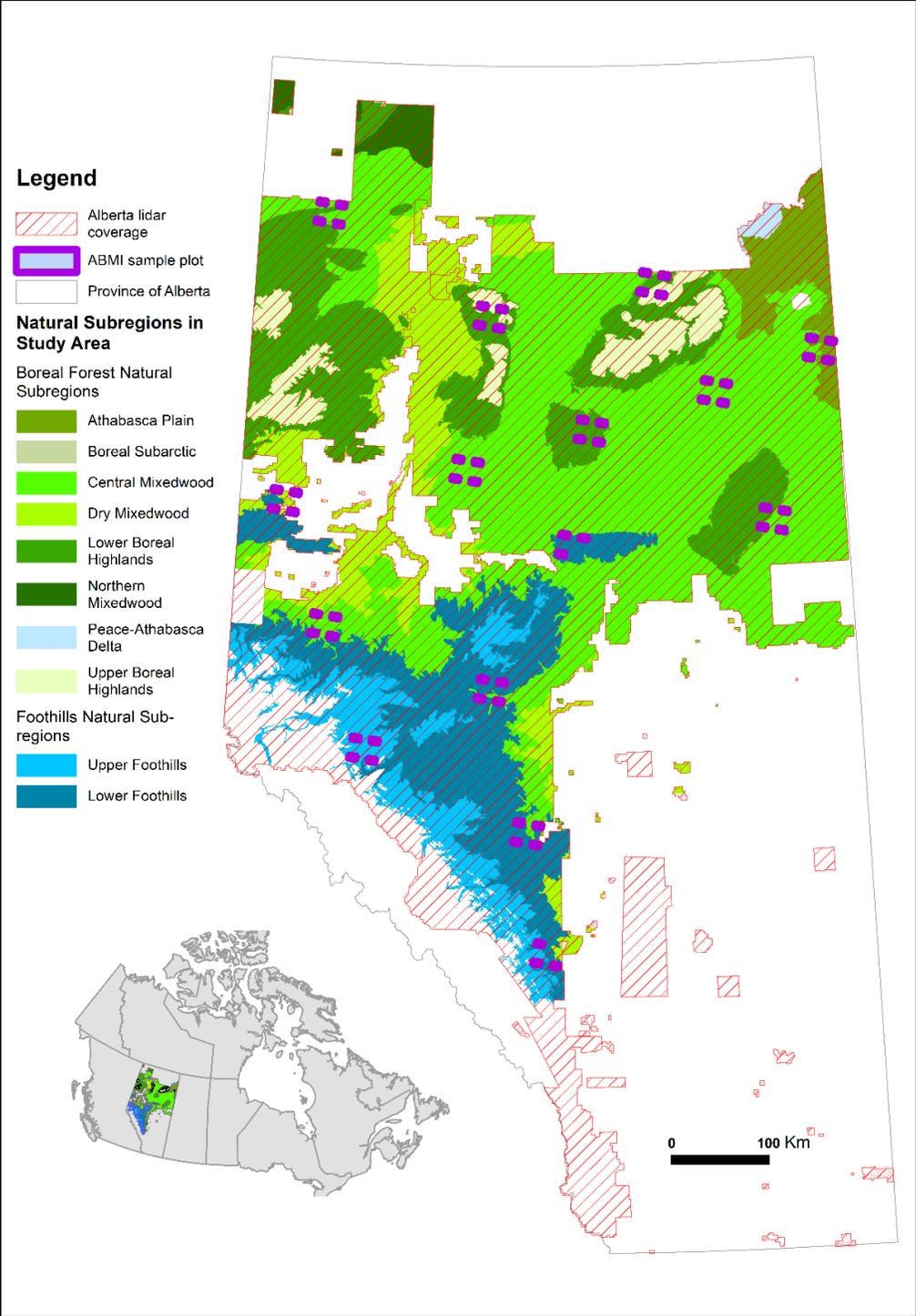


Figure 2-4. Spatial distribution of the sampled ABMI (Alberta Biodiversity Monitoring Institute) vegetation inventory plots.

# **Chapter 3: Regional Mapping of Vegetation Structure for Biodiversity Monitoring Using Airborne Lidar Data**

## **3.1 Introduction**

MacArthur (1972) identified productivity, climatic stability, and habitat structure as three primary drivers of biodiversity whose effects can be reflected in three aspects of the ecosystem: composition, structure, and function (Franklin et al., 1981; Noss, 1990). Spectral information acquired from optical remote sensing data has been widely used to assess the compositional and functional components of biodiversity over broad spatial scales (Cohen and Goward, 2005; Duro et al., 2007; Coops et al., 2008; Schuster et al., 2015). Habitat classifications based on land cover types (Wessels et al., 2000; Franklin et al., 2001; McDermid et al., 2009; Riggio et al., 2013), and habitat suitability indices derived from vegetation productivity and seasonality (Nilsen et al., 2005; Coops et al., 2008) have contributed significantly to species distribution models and animal movement studies. As opposed to correlations with spectral indices, the structural component of biodiversity has principally been assessed through forest resource inventories that require labor-intensive field surveys and/or aerial photo-interpretation (Fensham et al., 2002; Hyde et al., 2006; Clawges et al., 2008; Nijland et al., 2015b). In addition, resource inventories submitted by multiple forest management stakeholders may lack consistency in interpretation standards, update schedule, and aerial coverage when collectively used for large-area habitat mapping (McDermid et al., 2009). Moreover, vegetation height estimates from photo-interpretation are reported at the polygon level where the within-polygon variations in height and structure are not readily assessed

(McDermid et al., 2009). More detailed, fine-scale mapping of vegetation structure is needed to allow a broader range of biodiversity values to be included in forest management planning.

Airborne lidar (light detection and ranging) serves as an excellent tool that can accurately measure three-dimensional vegetation structure (Lim et al., 2003). Lidar-derived structural metrics have been used widely in forest ecological studies to determine or predict a number of important forest attributes, including: forest vertical layering and overall architecture (Maltamo et al., 2005); forest successional stages (Falkowski et al., 2009); vegetation strata and forest genera (Morsdorf et al., 2010; Kim et al., 2011); tree species abundance (Ewijk et al., 2014); forest volume, biomass and carbon storage (Zald et al., 2014); vegetation regeneration; and response after timber harvesting (Nijland et al., 2015b).

Although lidar technology cannot directly measure forest biodiversity, previous studies have examined the hypothesis that vegetation structure, as measured by lidar, can be an important indicator of species diversity as postulated by MacArthur and MacArthur (1961) and Erdelen (1984). Species distribution and habitat models built on lidar-derived structural variables indicate strong relationships between bird species occurrence, richness, and canopy height and cover metrics (Wulder et al., 2008; Graf et al., 2008; Goetz et al., 2012; Hovick et al., 2014; Hill and Hinsley, 2015). For example, positive correlations have been found between avian species richness and foliage height diversity (Clawges et al., 2008; Bergen et al., 2009), as well as canopy height (Goetz et al. 2007). Coops et al. (2016) found that adding variables correlated with vegetation structure to existing bird richness models improved model predictive power dramatically. Lidar-derived

structural metrics have also been used to examine habitat suitability for plant, bird and mammal species, including grizzly bears (*Ursus arctos*) and african lions (*Panthera leo*) (Bergen et al., 2009; Mueller et al., 2009; Lucas et al., 2010; Simonson et al., 2012; Jung et al., 2012; Gao et al., 2014; Nijland et al., 2015; Davies et al., 2016).

As most of these studies are species- and area-specific, broad-scale, multi-species, and multi-dimensional biodiversity monitoring is needed. However, this endeavor is difficult and costly (Gaston 1996, Green et al., 2005). An ecosystem-based approach may better lend itself to the conservation of plant and wildlife species diversity than species-specific monitoring of biodiversity. Noss (1990) emphasized that a top-down assessment of biodiversity status should start with a broad-scale inventory of vegetation composition, habitat structure, and their landscape patterns. As vegetation structure is an important indicator of forest species diversity (Moritz, 2001; Marchese, 2015), an inventory of vegetation structure using regional lidar data over broad area can provide base-layer information for ecosystem-based biodiversity monitoring. Without a comprehensive understanding of what types of habitat structure exist over the landscape, biodiversity monitoring based solely on relationships between species diversity, environmental gradients, vegetation composition, and productivity developed from localized studies could be biased and incomplete.

In this chapter, a large-area, discrete-return lidar dataset covering the managed forests of Alberta, Canada was used to develop an inventory of vegetation structure that efficiently synthesizes forest vertical variation into distinct classes for use in conservation and management planning. To do so,

six structure-related lidar metrics that measure canopy height, cover, and height variation at a 30-m spatial resolution (i.e. grid cell size) covering the study area were selected. Second, a two-step cluster analysis was applied to classify forest structure into eight classes and used discriminant function analysis to contrast variable importance within and between identified classes. Finally, forest attributes of species and age composition, wetland presence, and the impact of disturbance (both anthropogenic and non-anthropogenic) in the derived classes were interpreted and compared.

## **3.2 Materials and methods**

### **3.2.1 Alberta natural regions and subregions**

The study area covers more than 33 million ha of the managed boreal and foothills forest in the province of Alberta. A detailed description of the climate, topography, land uses, vegetation and wildlife composition can be found in chapter 2 section 2.1.

### **3.2.2 Lidar data and derived metrics**

The Government of Alberta, Canada has provided a multi-year lidar data compilation covering the majority of the managed forest in the province. A detailed description of the lidar data can be found in Chapter 2 Section 2.2.1.

Based on existing literature, six lidar-based metrics that were commonly related to forest biodiversity (Table 3-1) were selected. As height-related metrics were among the most widely used lidar metrics for habitat monitoring and assessments, four height strata (1.37 m to 5 m, 5 m to 10

m, 10 m to 20 m, 20 m to 30 m) were chosen to calculate canopy height density (percentage of lidar first returns) within each height stratum (Næsset & Gobakken, 2008; Nijland et al., 2015). A suite of metrics on canopy height density can reveal more details about canopy height distribution along the vertical dimension than direct lidar height measurements, thus a combination of canopy height metrics was used in this study instead. Short shrubs below 1.37 m were not considered. Previous studies have found strong correlations between bird species diversity and the proportional lidar returns near the forest floor (<5 m) (Clawges et al., 2008; Bergen et al., 2009; Mueller et al., 2009). Tall shrub and tree saplings below 5 m provide important habitat for many bird and mammal species. Therefore, canopy height density between 1.37 m to 5 m was used. The other three height strata used in this study represented the range of lower (5 - 10 m), middle (10 - 20 m) and upper (20 - 30 m) canopy height distributions in Alberta which might be utilized by wildlife species with varying habitat needs (Coops et al., 2016).

Beyond canopy height density, standard deviation (SD) of height has also been used as an important predictor of bird richness and abundance (Bergen et al., 2009; Mueller et al., 2009; Culbert et al., 2013; Vogeler et al., 2014). Forest stands with low standard deviations of height tend to have less structural complexity than stands with high standard deviations of height, therefore providing fewer niches for plant and wildlife species (August, 1983). Canopy heterogeneity has also been used as an indicator of the presence of large snags for cavity-nesting species (Martinuzzi et al., 2009; Bader et al., 2009).

Canopy cover was measured as the total percentage of lidar returns above a minimum tree height threshold (1.37 m in this study). Open canopy stands with strong light penetration near ground

level are associated with high exposure to solar radiation in shrub and understory layers. These stands generally have a high volume of understorey plant biomass and plant species diversity providing important habitat for understory and shrub nesting birds species and arthropod communities (Blakely et al., 2010). In contrast, medium to closed-canopy stands can help maintain a suitable moisture content for potential lichen growth, which furthermore provide critical winter habitat for caribou and other ungulate species (Kershaw, 1977; Apps et al., 2001; Coops et al., 2010 ).

Table 3-1. Six selected lidar variables on canopy height, canopy height density and variation

<b>lidar metrics</b>	<b>Explanation</b>	<b>Reference</b>
SD of Height	Standard deviation of vegetation height	Clawges et al., 2008
Canopy Cover	% of total first returns above 1.37 m	Vogeler et al.,2014
Canopy Height Density - 1.37 to 5 m	% of first return between 1.37 and 5 m	Bergen et al., 2009
Canopy Height Density - 5 to 10 m	% of first return between 5 and 10 m	Næsset & Gobakken,2008
Canopy Height Density - 10 to 20 m	% of first returns between 10 to 20 m	Næsset & Gobakken,2008
Canopy Height Density - 20 to 30 m	% of first return between 20 and 30 m	Næsset & Gobakken,2008

### **3.2.3 Vegetation inventory plots and disturbance data**

Three data sources were used in the study to interpret vegetation attributes and disturbance regimes for lidar-derived structure classification: the ABMI vegetation inventory plots, Alberta historical wildfire database and an anthropogenic disturbance database. A detailed description of the three data layers can be found in Chapter 2 Section 2.2.4 and 2.2.5.

### **3.2.4 Data analysis**

Cluster analysis is an exploratory, unsupervised classification technique that agglomerates objects based on perceived intrinsic similarity of the data to identify natural groupings (Kaufman and Rousseeuw, 2009). It is used to reveal the general characteristics and underlying structure of the data. An ideal cluster contains a set of objects which are similar to each other within the cluster, but heterogeneous or distinct from objects in other clusters. This method has been used in multivariate-based ecosystem classifications (Fitterer et al., 2012; Thompson et al., 2016) and ecoregion zoning (Coops et al., 2009). In our analysis, a two-step clustering analysis (Chiu et al., 2001) was used to accommodate the large amount of data and associated computing time. A k-means clustering algorithm was applied as the first step to classify the lidar data based on the six variables into pre-clusters, replacing raw data. These pre-clusters were then grouped using a commonly-used agglomerative hierarchical clustering algorithm (Unweighted Pair Group Method with Arithmetic Mean) based on Euclidean distance.

Before cluster analysis, lidar data points were filtered to remove vegetation heights above 45 m (air points) and below 1.37 m (which corresponds to the height at which stem diameters are

measured in the field). All data were standardized to be within the same range by multiplying a scalar. Pearson's correlation test (at P-value = 0.05) was performed to assess correlated relationships among the six variables. The two-step clustering process was conducted in R software with the packages "bigmemory" and "biganalytics" (Kane et al., 2010).

To test the uniqueness of the derived structure classes, a non-parametric Kruskal-Wallis rank sum test (Breslow 1970), followed by a Dunn's post hoc multiple comparison test, was used to examine if the overall clustering and pairwise difference between clusters were significant for individual lidar variable. When considering all variables together, a two-sample location test based on marginal ranks (Sen and Puri, 1977) was used as a non-parametric multivariate statistical test to determine if the clustering results were significant. Linear discriminant function analysis was applied on 10% random sample of the data to verify the separability of clustering results and assess variable importance from a modeling perspective. The predicted structure classes and the six variables used in the discriminant function analysis were plotted against the first two axes of the discriminant coefficients to contrast structural characteristics within and between structure classes and assess the strength of each variable in discriminating these classes.

To interpret the dominant forest attributes in each structure class, such as dominant species, layering architecture, density class, age composition, and wetland coverage, 58 ABMI photo plots containing > 15,000 polygons were sampled and overlaid with our classification. To examine the influence of anthropogenic and non-anthropogenic disturbance on each of the structure classes, the historical wildfire and anthropogenic disturbance databases were overlaid with the structure classification as well. At the pixel level, the information in each polygon of ABMI photo plots and

the two disturbance inventories were applied to each pixel in each structure class in the overlaid area.

### **3.3 Results**

#### **3.3.1 Validation and interpretation of structure classification**

Pearson's coefficients for the six lidar variables were significantly lower than 0.7 ( $p$  value = 0.05), indicating no significantly strong linear correlation between selected variables (Table 3-2). The k-mean clustering algorithm generated 100 pre-clusters, which were in turn further reduced to form the final eight structure classes through the hierarchical clustering process. Figure 3-1 shows the dendrogram from the hierarchical clustering and highlights where the natural breaks for eight structure classes were apparent in terms of maximizing the dissimilarity between an appropriate numbers of structure classes. Both the Kruskal-Wallis rank sum test and the two-sample location test based on marginal ranks indicated a significant difference between the eight classes for each individual variable and for all six variables ( $p$  value = 0.05). The Dunn's post-hoc test between paired samples was also significant for all variables ( $p$  value = 0.05), except for the standard deviation of height between classes 4 and 5, and the canopy height density between 20 and 30 m for class 3 and 7 as demonstrated in Figure 3-2.

Table 3-2. Pearson's correlation coefficients for six lidar variables (p value = 0.05; n = 3372180)

	SD of height	Canopy Cover	Canopy Height Density			
			1.37 to 5 m	5 to 10 m	10 to 20 m	20 to 30 m
SD of height	1.00					
Canopy Cover	0.49	1.00				
Canopy Height Density - 1.37 to 5 m	-0.34	0.02	1.00			
Canopy Height Density - 5 to 10 m	0.05	0.52	0.14	1.00		
Canopy Height Density - 10 to 20 m	0.47	0.68	-0.49	0.06	1.00	
Canopy Height Density - 20 to 30 m	0.53	0.36	-0.30	-0.22	0.19	1.00

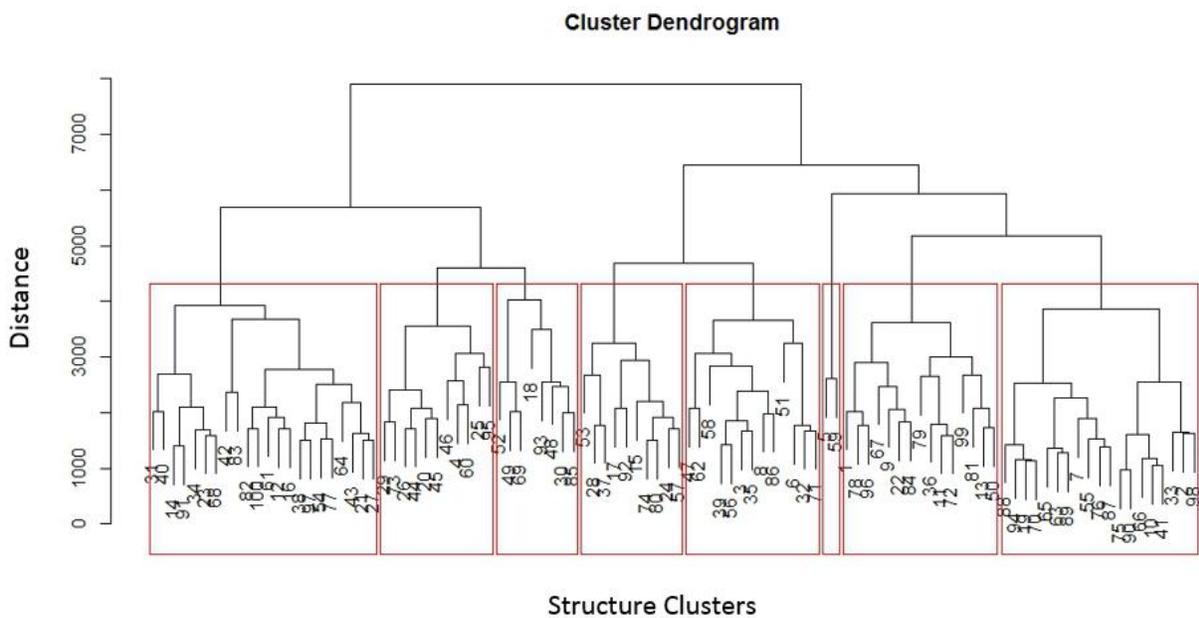


Figure 3-1. Eight classes identified from hierarchical clustering of the 100 pre-clusters

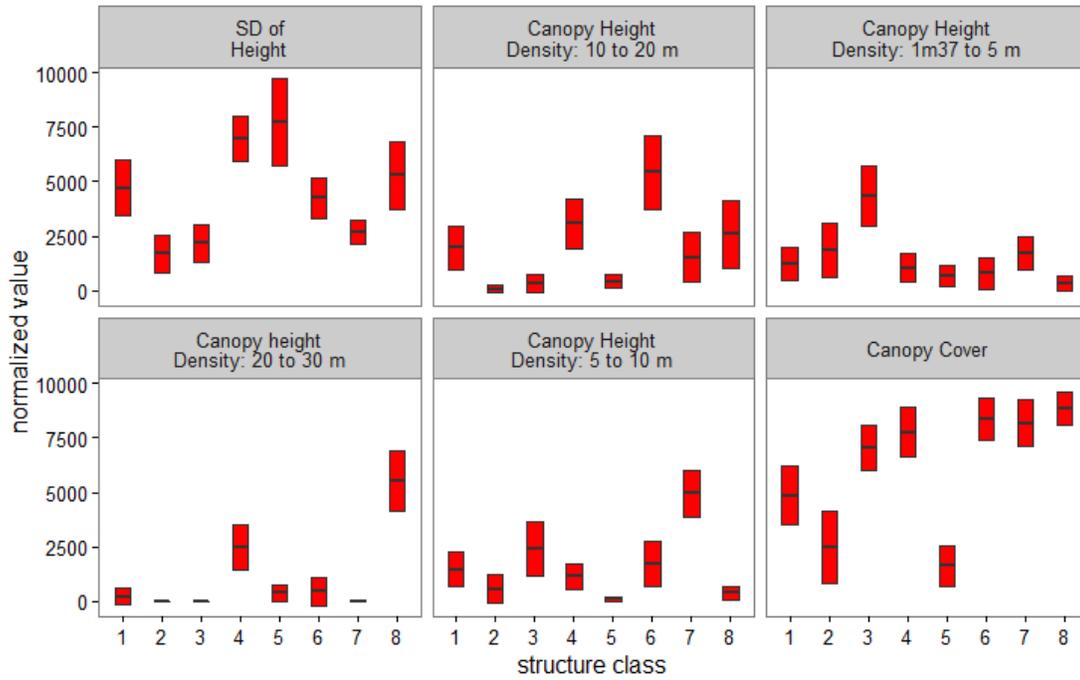


Figure 3-2. Comparisons between the eight structure classes for the six selected lidar variables which was normalized between 0 and 10000

Table 3-3. Confusion matrix for discriminant analysis of the eight structure classes

Class	Description	1	2	3	4	5	6	7	8	Total Count	User's Accuracy
1	Short, medium canopy cover stand	29914	3533	203	1225	2838	1118	1468	0	40299	0.74
2	Short, open canopy stand	3586	95782	12660	1	1483	0	574	0	114086	0.84
3	Very short, dense canopy cover stand	1473	0	31761	0	0	1	3788	0	37023	0.86
4	Very tall, complex stand	413	0	0	20709	398	1192	0	1381	24093	0.86
5	Very tall, open canopy stand	0	0	0	0	1975	0	0	0	1975	1
6	Tall, dense canopy cover stand	5946	0	88	3300	0	63765	3338	5	76442	0.84
7	Short, closed canopy stand	203	0	86	0	0	108	27222	0	27619	0.99
8	Very tall, closed canopy stand	0	0	0	1881	0	3	0	13797	15681	0.88
	Total Count	41535	99315	44798	27116	6694	66187	36390	15183	337218	
	Producer's Accuracy	0.72	0.96	0.71	0.76	0.3	0.96	0.75	0.91		Average Accuracy: 84%

The discriminant function analysis on the test dataset indicated an accurate classification (average accuracy: 84%) for all the classes except class 5 which was often misclassified into classes 1 or 2 (Table 3-3). The eight structure classes were all well separated when plotted against the first two discriminant coefficient axes (Figure 3-3) which explained 80% of the total variance. Total canopy

cover and canopy height density between 20 - 30 m were the most influential variables in terms of discriminating different structure classes.

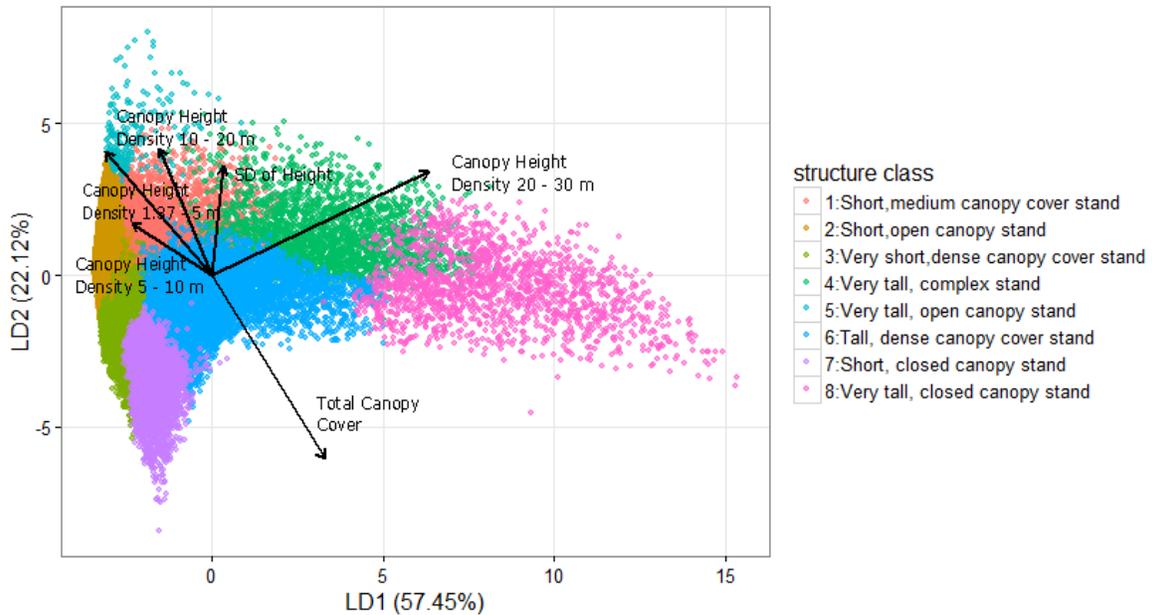


Figure 3-3. Structure classes plotted against the first two discriminant axes with the strength and direction of each variable indicated.

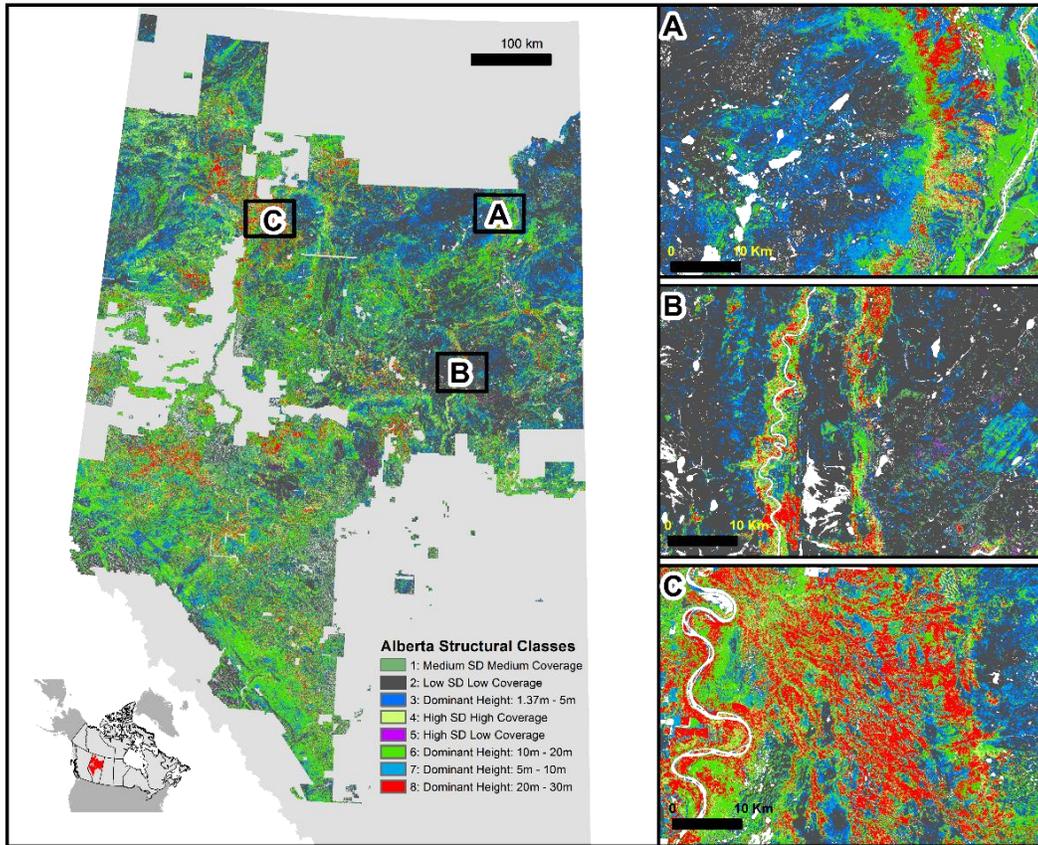
### 3.3.2 Spatial distribution and forest attributes of derived structure classes

The spatial distribution and stand profile of the eight structure classes are shown in Figure 3-4. As indicated by Figure 3-4(2), four of the eight classes were driven by differences in canopy height density in each height stratum (classes 3, 6, 7 and 8) and the other four classes were differentiated by canopy cover and standard deviation (SD) of height (classes 1, 2, 4, and 5). In Figure 3-4(1), different natural subregions exhibited different spatial patterns of structure classification. For example, black spruce-dominated wetland forest type associated with structure class 2 and 3 was more commonly found in the Upper Boreal Highland Natural Subregion and Central Mixedwood

Natural Subregion (A and B), whereas aspen-dominated upland forest type associated with structure class 8 and 6 more frequently occurred in the Dry Mixedwood Natural Subregion (C).

Short and open canopy stands with low standard deviations (SD) of height and low canopy cover (class 2) were the most common stand structure type in all natural subregions, except the Dry Mixedwood, Lower Foothills and Upper Foothills Natural Subregions (Figure 3-5). This class was characterized by black spruce-populated wetlands with wildfire as the most common disturbance agent (Table 3-4). The multimodal age distribution of this class might indicate cohorts of regenerated upland forest stand type dominated by aspen species after disturbance. Stands with tall, dense canopy cover and dominant heights between 10 - 20 m (class 6) had the second largest aerial extent and were most commonly found in the Central and Dry Mixedwood, Lower and Upper Foothills Natural Subregions, and in the higher elevated areas in the Lower Boreal Highlands Natural Subregion (Figure 3-4, Figure 3-5). Based on the ABMI photo plots, this class was characterized by dense, aspen-dominated upland forest stands with the year origin around 1950 (Table 3-4). Two aspen dominated structure classes (class 4 and class 5) with high standard deviations of height, but contrasting canopy cover, were the most structurally complex stands in the study area. In fact, the stand type represented by structure class 5 accounted for < 1% of the study area and was scattered throughout the Central and Dry Mixedwood, Lower and Upper Foothills Natural Subregions (Figure 3-5), with strong associations with harvesting activities (40%) and recent wildfires (15%) (Figure 3-6).

1)



2)

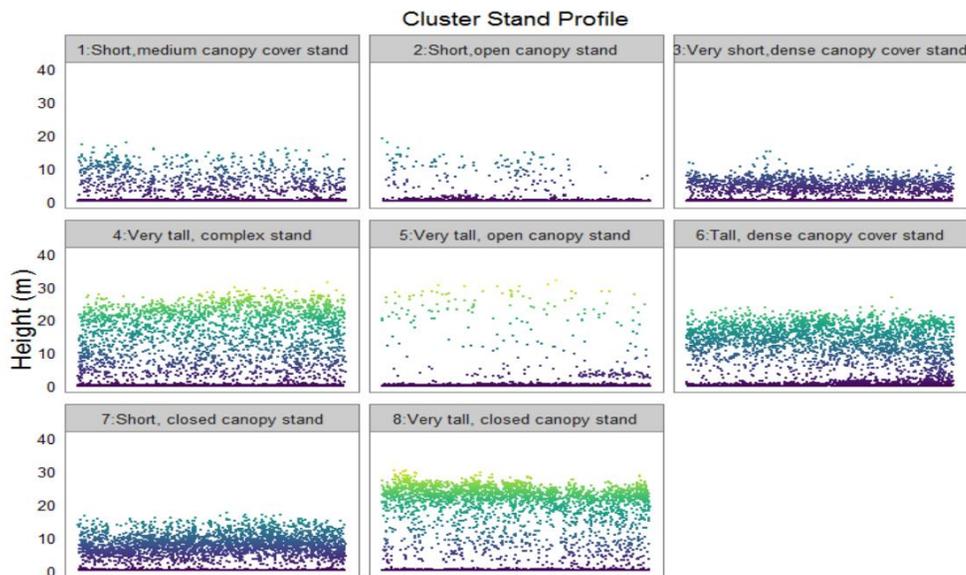


Figure 3-4. 1) Spatial distribution of the eight structure classes (A: transitional area between the Lower Boreal Highlands and Central Mixedwood Subregions; B: riparian zone along Athabasca River; C: gentle slope in transitional area between the Dry Mixedwood to Lower Boreal Highlands Subregions); 2) illustration of the stand profile for each structure class.

Table 3-4. Description of each structure class related to forest composition and disturbance (Aw: Aspen; Sb: Black Spruce; Pj: Jack Pine).

Class	Description	Leading Species	Age Composition	Density Class	Percent wetlands	Percent Fire
1	Short, medium canopy cover stand	Aw	Multimodal (1950,2000)	A/-	16%	13%
2	Short, open canopy stand	Sb	Multimodal (1950,2000)	A/-	64%	28%
3	Very short, dense canopy cover stand	Sb	Multimodal (1950,1980)	A/B	44%	21%
4	Very tall, complex stand	Aw	Unimodal (1910)	C/A	2%	2%
5	Very tall, open canopy stand	Aw	2000	A/-	5%	17%
6	Tall, dense canopy cover stand	Aw	Unimodal (1950)	C/-	4%	2%
7	Short, closed canopy stand	Aw & Sb & Pj	Unimodal (1950)	A/-	18%	18%
8	Very tall, closed canopy stand	Aw	Unimodal (1910)	C/-	1%	1%

As indicated from Figure 3-4(1), tall and complex, aspen-dominated, upland stand type (class 8) were spatially concentrated along river floodplains (A), at a lower slope position along transitional areas between the Dry Mixedwood and Lower Boreal Highland Natural Subregions (B), and between the Central Mixedwood and Lower Boreal Highland Natural Subregions (C). Forest

stands with dense canopy cover and very short vegetation heights ranging from 1.37- 5 m (class 3) were mostly black spruce-dominated wetlands with wildfires being a common disturbance agent (Table 3-4). This stand type was most abundant in the Central Mixedwood and Lower Boreal Highlands Natural Subregions (Figure 3-5). Closed-canopy stands with dominant tree heights ranging from 5 - 10 m (class 7) were the most encompassing class with respect to species and age compositions (Table 3-4). Structure class 1 associated with stands of short tree height and medium canopy cover was mostly aspen-dominated forest type with moderate influence of human and wildfire disturbance (Table 3-4, Figure 3-6). The multimodal age composition of structure class 1 and 3 might indicate cohorts of recently generated forest stands following disturbances (Table 3-4).

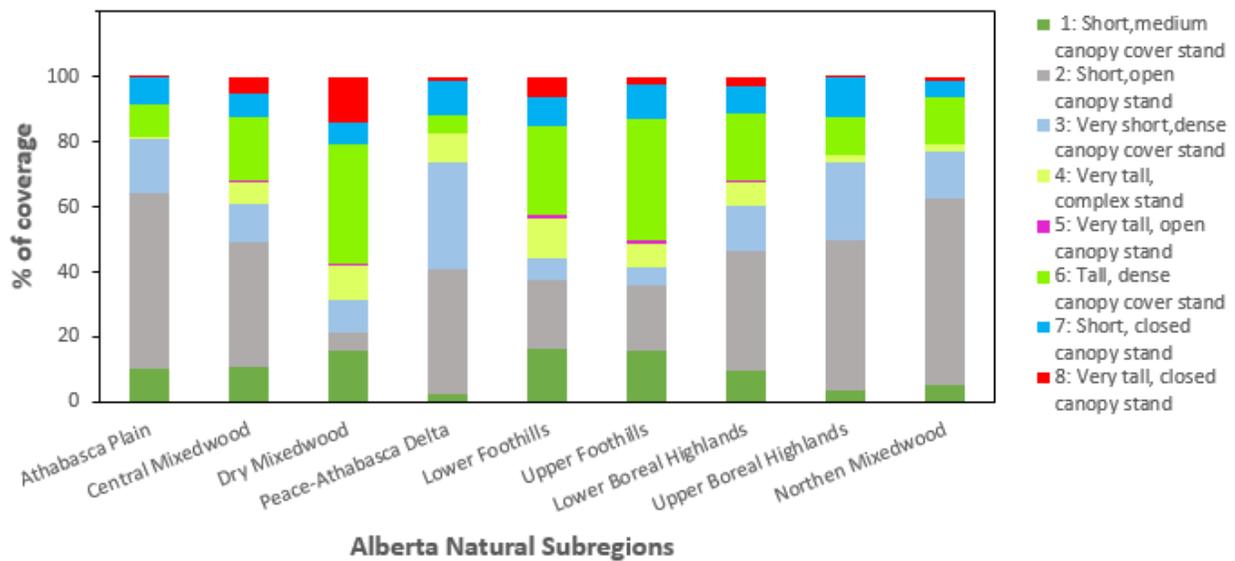


Figure 3-5. Composition of structure classes in each natural subregion.

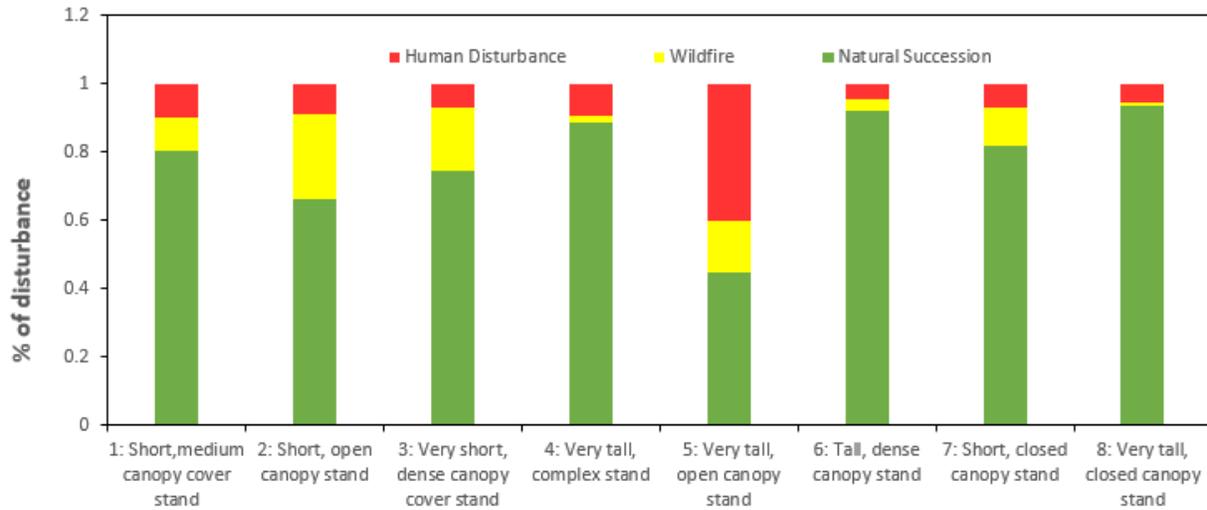


Figure 3-6. Composition of anthropogenic and non-anthropogenic disturbance in each structure class.

### 3.4 Discussion and conclusion

#### 3.4.1 Factors affecting structure classification result

Canopy height density and canopy cover are the two major factors discriminating different structure classes. The contribution of standard deviation of height is less prominent indicating some degree of multicollinearity between canopy cover, canopy height and standard deviation of height. In general, tall stands tend to have high standard deviations of height. Also, single-layer stands with dense canopy cover usually have lower standard deviations in heights than that of more complex stands. The low predictive accuracy of structure class 5 in discriminant function analysis may be due to the extremely low areal coverage of this class and large variations in heights. With canopy height distributed evenly along the vertical dimension, this structure class was more likely to be misclassified with other classes, whose dominant height is within the range of height variations of the class 5 forest structure type.

### 3.4.2 Critical stand structure and associated ecological function

Vegetation height variation in different natural subregions is predominately influenced by elevation, climate regimes and tree species distribution (Hasen et al., 2014). Optimal slope position, soil nutrients and drainage condition (Little et al., 2002, Zhang et al., 2011) also facilitated the formation of spatial clusters of tall and dense forest stands (class 8) across the study area (Figure 3-4(1)), which could provide cover and protection for certain wildlife species such as great gray owl (*Strix nebulosa*) and woodland caribou (Servheen and Lyon, 1989; Duncan, 1997). Large trees, the presence of which are also positively related to tree height, may form snags and cavities, providing important habitat for many cavity-nesting species (Conner and Adkisson, 1977; Nelson et al., 2005; Jung et al., 2012). Stands of tall and dense forest cover (class 8) could provide important habitat value to forest specialist species.

Dense forest stands with the most complex vegetation structure (class 4) were highly abundant in the Lower Foothills Natural Subregion where rolling terrain and various topographic features were found. These areas span a broad range of environmental gradients creating diverse microhabitats for vegetation structural development (Opedal et al., 2015). Similarly, Hansen et al. (2014) confirmed that mature forest with fully-developed canopy structures were more likely found in mountainous areas than on flat landscapes in southeastern US. Areas of rough terrain are often associated with a high level of plant species richness (Opedal et al., 2015) with terrain shading as an important factor affecting the occurrence of individual plant species (Nijland et al., 2014). As structurally complex areas tend to accommodate a higher level of biodiversity (Vierling et al., 2008), this structure class (class 4) with its topographic context may be of high interest for biodiversity monitoring and conservation.

Fully developed, semi-open forest stands were previously reported to have high levels of plant, bird and mammal species diversity (Hobson and Bayne, 2000; Gil-Tena et al., 2007; Smart et al., 2012). Semi-open stands (class 1, 2 and 5) with different levels of light penetration allow both light-adapted and shade-adapted species to flourish, and also provide a variety of food resources and sufficient open space for wildlife species to forage (Nielsen et al., 2004). However, with different disturbance regimes altering vegetation structure pattern, these types of structure classes could be the result of disturbances. Further research is need to quantitatively assess the impact of disturbance on forest ecology in terms of canopy cover, vertical architecture, stand dynamics, and landscape patterns. This analysis result indicated that human disturbance contributed to the development of open, however complex vegetation structure (class 5) which was geographically rare across the landscape.

### 3.4.3 Implication and utilization of the data product

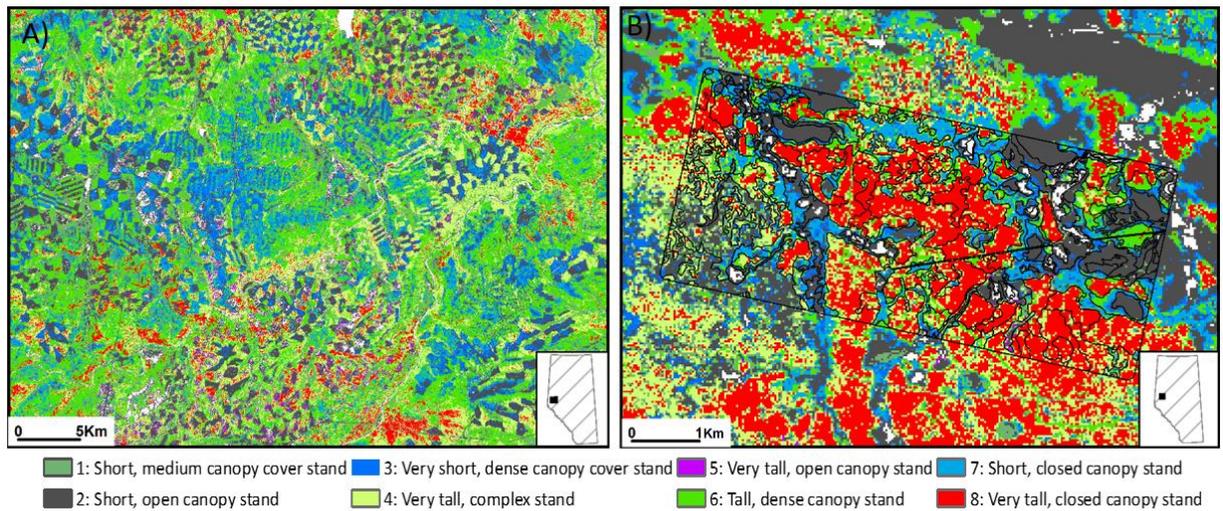


Figure 3-7. A) Fragmented landscape caused by human disturbance in Lower Foothills Natural Subregion; B) Comparison between the lidar-derived structure classes and ABMI photo plot.

This study integrates canopy cover, canopy height and height variation into a single classification scheme based on natural variation in vertical structure of Alberta's boreal and foothills forest. The 30-m resolution (grain or cell size) of the structure classification is sufficient to reflect subtle differences in vegetation structure, especially in fragmented landscapes such as cutblocks and other human disturbed areas (Figure 3-7). Satellite-based optical land cover mapping is relatively insensitive to this level of structure variation (Wulder, 1998; Lim et al., 2003; Geotz et al., 2007). As opposed to polygon-based vegetation inventory plots, this approach captures within-polygon variations in terms of tree height and canopy gaps, which can be used to reflect fine-scale habitat values (e.g. snags and cavity trees) (Figure 3-7). Furthermore, this study follows a consistent and objective classification scheme that does not require costly, labor-intensive manual digitalization and photo-interpretation. In addition, the qualitative structural inventory and stratification can serve as a descriptive tool to augment quantitative resource inventories when accurate estimation of timber attributes (e.g. volume, stocking) is not required for management activities, such as wildlife conservation and forest protection where timber harvesting is not the top priority (Reque and Bravo, 2008).

The derived structure classification has the capacity to help conservation planners identify areas of high conservation value for follow-up remote sensing or ground data acquisition and in-depth species modeling. This classification of vegetation structure describes habitat-related vegetation features independent of a specific wildlife species, adding flexibility to wildlife monitoring programs across different taxonomic groups. However, each structure class is likely to encompass a variety of biological and ecological conditions, making labeling classes with accurate biodiversity traits difficult. As a result, this classification should be considered as a base layer used

to generate a series of value-added data products when integrated with spatial information of wildlife species distribution, land cover types, forest inventories, ecosystem classification and human disturbance data. Without this additional information, the vegetation structure classification alone may lack biological and ecological relevance. For example, a black spruce-dominated bog could have a similar stand structure to that of a burned or harvested aspen stand with vegetation regeneration, but would provide different habitat values.

The lidar data used in this study is a provincial compilation collected through multiple years and seasons by various data providers using different sensors, however following the same data acquisition standards. Wildfire and land use changes during the period of data collection may not be fully represented in our analysis due to temporal misregistration. However, with the majority of the data acquired in 2007 and 2008, this discrepancy should be alleviated. Also, time series of image products containing information about land cover change locally or regionally can be overlaid with the classification to update the spatial distribution of vegetation structure to a more contemporary status. Photogrammetric point cloud data can also be used to develop similar structural variables for classifying vegetation structure at local areas with fine-resolution, cost-efficient sensors. This study demonstrates how regional lidar data can be utilized to classify and map vegetation structure for large-area wildlife and biodiversity monitoring initiatives.

# **Chapter 4: A Network-based Approach to Estimate Landscape Forest Structure Connectivity Using Lidar Remote Sensing**

## **4.1 Introduction**

Forest ecosystems provide vegetation structure supporting a wide range of forest species, as well as related ecological processes such as pollination, seed dispersal, gene flow, wildlife movement and migration (Forman, 1995; Rosenvald & Lohmus, 2008; Wang et al., 2010; Marchese, 2015). Forest biodiversity is highly dependent on the availability and accessibility of high quality habitat (Bodin, 2009). In forested ecosystems, structurally complex forests are hypothesized to have higher amounts of biodiversity by providing more niches for species with different habitat specializations (Lindenmayer et al., 2000; Grelle, 2003; Culbert et al., 2013). In forested wetlands, large patches with undisturbed, dense vegetation cover serve as critical nesting and foraging habitat for many species, as well as a primary source of wetland biomass, net primary production, and carbon storage (Burkett and Kusler, 2000; Erwin, 2009). In biodiversity conservation, both habitat quality and connectivity should be considered to make sure that high-quality habitat patches are accessible to a wide range of species in a connected habitat network.

Vegetation structure is an important indicator of both floral and faunal species diversity, and can be well characterized and quantified through lidar-derived structural metrics, such as canopy cover, canopy height and canopy vertical complexity at stand and local levels (Bergen et al., 2009; Apps et al., 2001; Coops et al., 2016). Landscape-level inventory, categorization and quantification of forest structural attributes are now possible as local lidar acquisitions are increasingly compiled into regional coverages (Asner et al., 2011; Hansen et al., 2014). Therefore, spatially-explicit

habitat patches with structural traits of high biodiversity value can be identified across large landscapes using lidar-derived structural metrics as a fundamental step to build regional-level habitat networks towards better forest management and biodiversity monitoring. The application of lidar in facilitating regional-level conservation planning has been largely unexplored.

Habitat loss and fragmentation due to anthropogenic and natural disturbances can change the spatial arrangement of habitats in landscapes making them highly vulnerable to loss in connectivity (Saura et al., 2011; Dilts et al., 2016; Albert et al., 2017). Habitat patches should be spatially connected in a manner that ensures energy flow and movements of organisms, thus promoting species persistence (García-Feced et al., 2011). To promote habitat connectivity, habitat area and the spatial configuration are two important criteria in conservation planning which have been considered to develop effective conservation strategies (Hodgson et al., 2009; Mortelliti et al., 2010; Sverdrup-Thygeson et al., 2014). Habitat area can be considered as an indicator of available resources and habitat quality, whereas the spatial configuration of habitat patches affect how well individual habitats are connected. Forest management planning considers these two criteria through the maintenance and placement of large forest patches containing interior habitat (von Sacken 1998) and through harvest patterns that preserve stepping-stone patches between high-quality habitats (Araújo et al., 2004; Wikramanayake et al., 2004; Alberta Sustainable Resource Development, 2006; Ontario Ministry of Natural Resources, 2010). Network analysis approaches to evaluate connectivity can help balance these two needs (Minor and Urban, 2007).

A network-based approach depicts habitat patches as a relational network through a collection of nodes and links (Urban and Keitt, 2001). The simplicity and flexibility of network-based connectivity analysis allows for a spatially-conceptualized representation of habitat patches and their spatial relationships across a landscape, and has been used in a variety of ecological settings (Calabrese and Fagan, 2004, Bodin and Saura, 2010; Albert et al., 2017). Patches differentially contribute to overall network connectivity depending on not only habitat area but also habitat locations (Minor and Urban, 2007). Large forest patches may serve as a single core habitat containing a high level of resources for wildlife species, whereas small forest patches could play an irreplaceable role as stepping stones to help species disperse between core habitat patches. The loss of key connector patches, even those small in size, may lead to significant connectivity losses and increased vulnerability to landscape fragmentation (Rothley and Rae, 2005; Bodin and Saura, 2010). Network-based modelling approaches are therefore useful in partitioning the different roles that patches can play and improving our understanding of their contributions to landscape connectivity, thus prioritizing important patches into conservation planning.

In Canada, additional protected areas are required for biodiversity conservation reaching the 17% national target set by the Convention of Biodiversity (CBD, 2010). To balance the demand from forest operation and allocate conservation efforts to areas with high priorities, spatial prioritization is necessary (Moilanen et al., 2009), however, challenging over a landscape covering broad spatial area with diverse topography, disturbance regimes and climatic conditions. To be specific, firstly, forest harvest and wildfire have created mosaic patterns of land cover change, impacting connectivity of remaining habitat across the landscape (Saura et al., 2011; Dilts et al., 2016; Albert et al., 2017). In most cases of spatial prioritization, however, the cumulated effects on habitat

connectivity have not been well-addressed (Rubio et al., 2014; Blazquez-Cabrera et al., 2014). Therefore, modeling the long-term effect of spatial prioritization is important in tracking the spatial and temporal dynamics of landscape connectivity to prevent large connectivity losses, and to evaluate the effectiveness of conservation strategies. Secondly, the effect of climate change could influence regional-level conservation planning. Climate change is projected to increase the frequency of forest fires, insect attacks, and disease in northwest North America, thus reducing the habitat suitability for many plant and wildlife species (Thompson et al., 2009; Mathys et al., 2017). To mitigate the negative effects of climate change, important habitat patches in regions of relatively stable climatic conditions (i.e. high climate stability) could be considered an additional priority in spatial prioritization for conservation planning (Bailey, 2007; Heller and Zavaleta, 2009; Beier et al., 2011). Such areas can act as refugia and facilitate species migration and colonization to reduce the vulnerability to biodiversity loss (Iwamura et al., 2010; Thompson et al., 2009).

In this chapter, a novel approach was demonstrated to conservation planning, employing forest structural indicators derived from airborne lidar data to assess habitat connectivity dynamics through changing landscapes in the managed forest in the province of Alberta, Canada. We construct habitat networks based on lidar-derived habitat patches and model connectivity through simulated land cover change scenarios with spatial conservation prioritization. Specifically, we aim to 1) compare spatial patterns of habitat networks and assess connectivity dynamics based on different conservation prioritization schemes and 2) evaluate which conservation strategy is most effective in reducing losses in landscape connectivity in light of changing landscapes. Results can provide insights into the location and quality of habitat patches and serve as an example that

leverages the use of spatially-explicit lidar remote sensing data and network analysis tools to improve spatial conservation prioritization, forest management and conservation planning.

## **4.2 Materials and methods**

In the managed forested area in Alberta, a regional lidar dataset combined with information on land cover types were used to identify key habitat structure and derive four habitat networks for deciduous, coniferous, mixedwood and wetland forest patches, respectively. Connectivity metrics reflecting habitat area and habitat configuration were evaluated. We generated five land cover change scenarios according to four prioritization schemes based on connectivity metrics and climate constraints. We then applied the five scenarios on 10 replicas of each habitat network and compared the connectivity losses between scenarios for each habitat network.

### **4.2.1 Study area**

The study area encompasses the managed boreal and foothill forests in the province of Alberta, covering more than 33 million hectares and including a range of climatic and environmental conditions. A detailed description of the climate, topography, land uses, vegetation and wildlife composition can be found in chapter 2 section 2.1.

## **4.2.2 Data**

### **4.2.2.1 ALS data**

A regional classification of forest structure based on lidar-derived metrics was used to identify important habitat structure (see Chapter 3 for a detailed description) (Figure 4-1). In brief, six structure-related lidar metrics were summarized at a 30-meter spatial resolution incorporating canopy height, height variation, and canopy cover. Vegetation structure was clustered into eight unique structure classes based on the six structural metrics representing natural groupings of vegetation structure. The broad spatial coverage of the lidar dataset resulted in a continuous inventory of vegetation structure across Alberta's managed forested area. Based on canopy density and height distribution, two of the classes (classes 4 and 8) with high levels of structural complexity were considered key habitat structure types in deciduous, coniferous and mixedwood forest. Structure classes 3 and 7 represented short and dense forest coverage, and served as indicators of important wetland habitat types.

### **4.2.2.2 Land cover map**

A land cover classification for Alberta was used to identify four forest cover types: deciduous-dominated forest, coniferous-dominated forest, mixedwood-dominated forest, and wetland dominated forest. A detailed description of the land cover layer can be found in Chapter 2 Section 2.2.2.

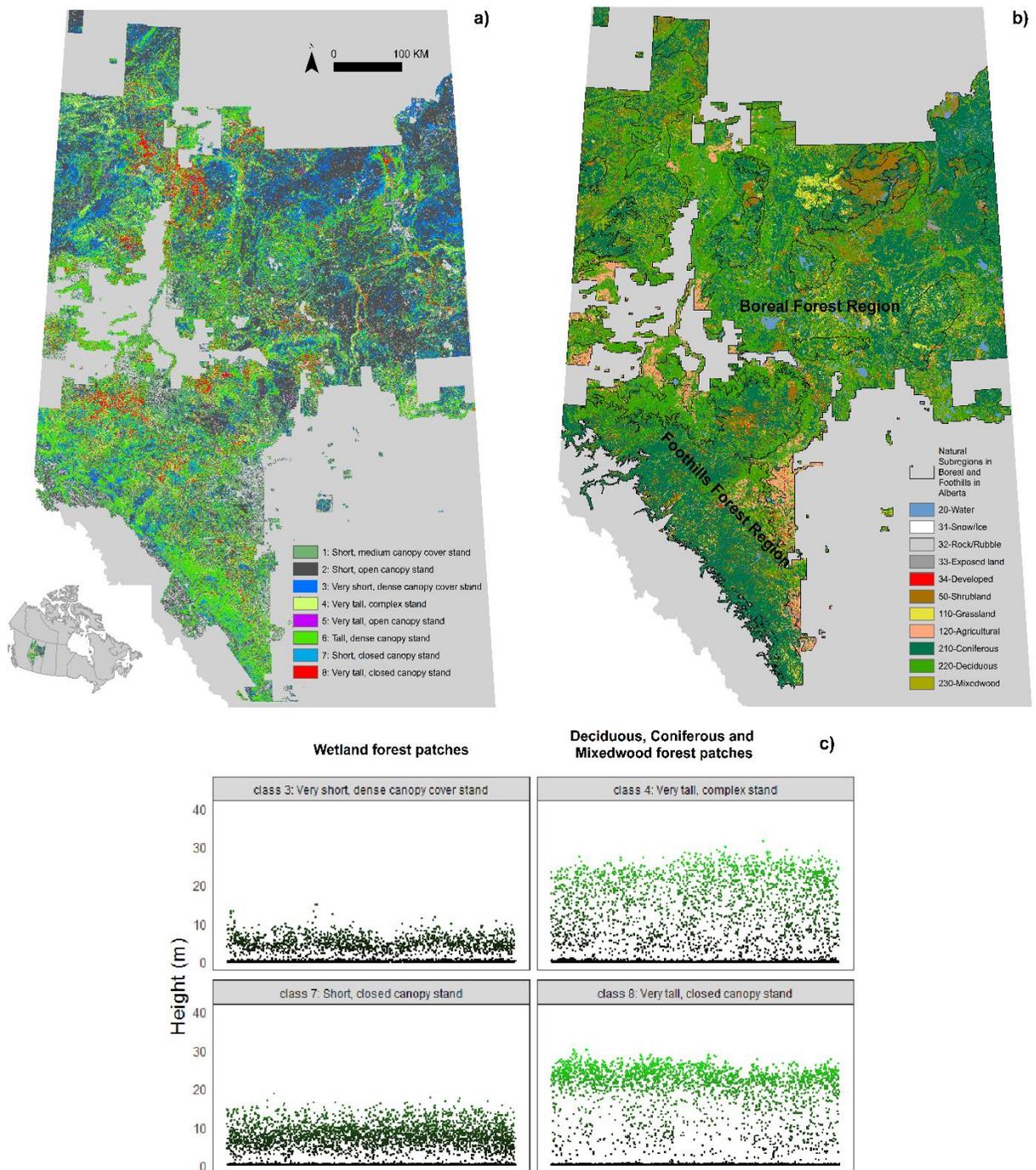


Figure 4-1. Characterization of potential habitat patches: a) a structural inventory across managed forested area in Alberta (Guo et al., 2017); b) rasterized land cover classification for the province of Alberta (Wulder et al., 2008); c) structure classes combined with land cover map to identify potential habitat patches: class 4 and 8 for deciduous, coniferous and mixedwood dominated patches, and class 3 and 7 for wetland dominated patches.

### 4.2.2.3 Climate data

Current and projected climate data were used in the analysis to identify area with high climatic stability. A detailed description of the climate data can be found in Chapter 2 Section 2.2.3.

## 4.2.2 Network-based models and connectivity metrics

Network-based modeling approaches can be used to assess the possibilities of species movement or ecological process among spatially isolated habitat patches across the landscape (Urban and Keitt, 2001; Pascual-Hortal and Saura, 2006). A graph-based network conceptualizes habitat patches and their spatial relationships, as nodes and links, respectively. In our study, we used a simplified binary connection model where habitat patches are considered connected if the link distance is below a specified distance threshold related to species dispersal (Saura & Rubio, 2010). Node importance is usually assessed by removing individual nodes from the network and quantifying the corresponding loss in connectivity. As landscape-level, species-specific connectivity monitoring is difficult to due to a lack of data availability on distributions and movements of individual species, generalized connectivity metrics which avoid species-specific parameterizations are used as an indicator of network connectivity at different distance thresholds (Liu et al., 2001; Calabrese, J. M. & Fagan; 2004).

The Integral index of connectivity (IIC) metric (Pascual-Hortal & Saura, 2006) is a habitat availability index that measures patch importance based on both the connected area existing within the habitat patch as well as its contribution to connectivity between patches:

$$IIC = \sum_{i=1}^n \sum_{j=1}^n \frac{a_i * a_j * (1/(1+d_{ij}))}{A_L^2} \quad (1)$$

where  $n$  is the number of habitat patches across the landscape with a habitat area of  $a_i$  for each patch and  $A_L$  is the total landscape area. The  $d_{ij}$  represents the link distance between patch  $i$  and  $j$  (minimum number of links that required to move from  $i$  to  $j$ ). When  $i = j$ , link distance  $d_{ij}=0$ . When there is no possible link between patch  $i$  and  $j$ ,  $d_{ij}$  is infinite. IIC result is usually quantified as Equivalent Connected Area (EC (IIC)) which is described as the size of a single habitat patch that would provide the same IIC value. The connectivity loss of each individual patch  $k$  can be addressed by calculating the IIC metrics before and after the removal of patch  $k$ :

$$dIIC_k = 100 * \frac{IIC - IIC_{remove}}{IIC} \quad (2)$$

The three partitions of IIC index (Saura & Rubio, 2010) reflect different contributions of habitat patches to landscape connectivity and can be calculated for each patch  $k$  before and after it was removed from the landscape network:

$$dIIC_k = dIICintra_k + dIICflux_k + dIICconnector_k \quad (3)$$

Among the three partitions, the *dIICintra* and *dIICconnector* are the two most distinguishing metrics. The *dIICintra* corresponds to the intra-patch habitat availability of patch  $k$  represented by its area, which is then used to quantify patch importance based on habitat area. The *dIICconnector<sub>k</sub>* indirectly measures the inter-patch connectivity by calculating the connectivity loss of all other habitat patches if patch  $k$  was removed from the landscape. It is used to quantify patch importance based on habitat configuration.

### **4.2.3 Generation of habitat networks**

Forest patches were delineated based on the vegetation structure classes defined by Guo et al., (2017) (see Chapter 3 for detailed description) spatially overlaid with the forest cover types and the wetland inventory, resulting in four habitat patch configurations for patches > 25 ha. As species dispersal occur at different dispersal distances, habitat connectivity is highly scale dependent (Metzger and Décamps, 1997; Maciejewski and Cumming, 2016). Therefore, *dIIC* metric was evaluated at a variety of dispersal distances to identify the distance threshold as the most appropriate scale for our study. Specifically, network connectivity was quantified as the sum of *dIIC* metric at a variety of distance thresholds between 5 and 100 km representing dispersal capacities of different vertebrate species. The distance threshold  $D$  was defined that all patches separated by a Euclidean distance smaller than  $D$  were considered connected (links assigned between these patches). Overall, the distance at which the sum of the *dIIC* was at its maximum was used as the distance threshold to construct habitat networks. At this distance threshold, the landscape was sufficiently connected and highly sensitive to connectivity loss caused by patch removal (Bodin & Saura, 2010).

### **4.2.4 Conservation prioritization and scenario-based land use change simulations**

Three criteria (habitat area, habitat configuration measured as inter-patch connectivity, and climatic stability) were used as guiding principles to create alternative conservation prioritization schemes. The first two criteria were quantified independently through IIC-based patch importance: *dIICintra* and *dIICconnector*. Climatic stability was quantified by computing the relative differences in climate variables between current and projected future climatic conditions. Current annual minimum and maximum temperature, annual precipitation and adjusted annual number of

frost days were compared to future projected climatic conditions for the year 2050. The relative change between each variable was calculated and averaged to a 1-km spatial resolution. Each habitat patch was assigned the relative difference and ranked with lower values, indicating higher climatic stability.

Five conservation prioritization schemes in light of land cover change were simulated based on combinations of the three criteria for each of the four habitat networks over time (Table 4- 1). To evaluate how effective the five prioritization schemes were at mitigating future connectivity loss caused by land cover change over the long-term, change events were simulated over 30 years (from 2020 to 2050). The national biodiversity conservation objective (CBD, 2010) of 17% remaining forest patch area was used as the 2050 target. Over 30 annual time steps, 2.7% of the habitat area was removed per annum, which approximates the average annual harvesting rate in Forest Management Agreement Units in Alberta. The five prioritization schemes were implemented as the following scenarios:

1. **Area-only scenario:** habitat area was considered as the only criterion for patch protection; thus, habitat patches with the lowest *dIICintra* in each time (patch removal) step were removed first.
2. **Area-climate scenario:** climate change constraints were added to the *area-only scenario* where conservation preference was given to patches that remained relatively stable under future climate projections.
3. **Area-connector scenario:** both habitat area and habitat configuration were considered; thus, habitat patches with the lowest *dIICintra* and *dIICconnector* in each time (patch removal) step were removed first.

4. **Area-connector-climate scenario:** climate change constraints were added to the *area-connector scenario* where conservation preference was given to patches that remained relatively stable under future climate projections.
5. **Business-as-usual scenario:** A randomized land use change simulation where no spatial prioritization rules were considered; thus, patches were removed randomly.

A further simplifying assumption was made that no additional forest patches were added to the landscape network over the ensuing 30 years. For each scenario of each forest type, simulations were conducted to generate temporal trajectories for 10 replicate habitat graphs by randomly sampling 30% of each habitat network for each replicate. The *IIC* metrics was evaluated using Conefor Sensinode 2.6 software package (Saura and Torne, 2009) command line version and the simulation process was programmed in R 3.3.1 (R Core Team, 2012). The total connectivity losses quantified by *IIC* metrics before and after each simulation scenario were calculated for each habitat network. Connectivity losses were also monitored for long-term dynamics of connectivity metrics during the process of simulations to evaluate the effectiveness of each prioritization scheme in mitigating connectivity loss. The average patch importance was calculated as the mean number of simulation time steps that patches were removed from the landscape and summarized to the National Topographic System (NTS) at a 30-km resolution.

Table 4-1. Summary of the scenario-based land use change simulations based on four conservation priorities.

Prioritization Schemes	Criteria	Conservation Scenarios	Simulations
Protect the largest habitat patches	Habitat area	Area-only	Remove the least important habitat patches in terms of habitat area for 2.7% of land base in each step for 30 steps
Protect the largest habitat patches in areas with stable climatic conditions	Habitat area and climate stability	Area-climate	Similar as area-only prioritization, but patches were protected if they remained stable between current and future climate conditions
Protect large habitat patches, as well as connector patches	Habitat area and habitat configuration	Area-connector	Remove the least important habitat in terms of habitat area and spatial configuration for 2.7% of the land base in each step for 30 steps. 70% weight was given to intra-patch connectivity and a 30% weight given to inter-patch connectivity
Protect large habitat patches, as well as connector patches in areas with stable climatic conditions	Habitat area, habitat configuration and climate stability	Area-connector-climate	Similar to area-connector prioritization, but patches were protected if they remained stable between current and future climate conditions
No priorities	Not applied	Business-as-usual	Randomly remove 2.7% of the land base in each step for 30 steps

## 4.3 Results

### 4.3.1 Habitat patch distribution and landscape structural connectivity

The four habitat networks with coniferous, deciduous, mixedwood and wetland forest cover types resulted in different patch distributions (Table 4-2, Figure 4-2). Deciduous-dominated forest patches were well distributed over the northern boreal forests and the northern areas of the Foothills forest, and had the highest number of patches, largest patch size and shortest inter-patch distance compared to other forest cover types. Coniferous-dominated patches were distributed more evenly over the landscape with more patches in the Lower and Upper Foothills Natural Subregions. Mixedwood-dominated patches were the least abundant patch type across the landscape with the smallest patch size and lowest number of patches. Wetland patches were mostly concentrated in the northern boreal plains where the flatter terrain is dominated by wetlands and peatlands.

Table 4-2. Patch characteristics for the four habitats network composed of coniferous, deciduous, mixedwood and wetland dominated patches.

Forest Types	Structure Class	Structure Types	Number of Patches	Average size (ha)	Total area	Maximum distance between two nearby patches (km)
Coniferous	4, 8	tall and complex	2347	70.6	165,532	57
Deciduous	4, 8	tall and complex	3812	129.1	491,647	42
Mixedwood	4, 8	tall and complex	967	55.9	54,024	68
Wetland	3, 7	short and dense	2038	70	142,77	67

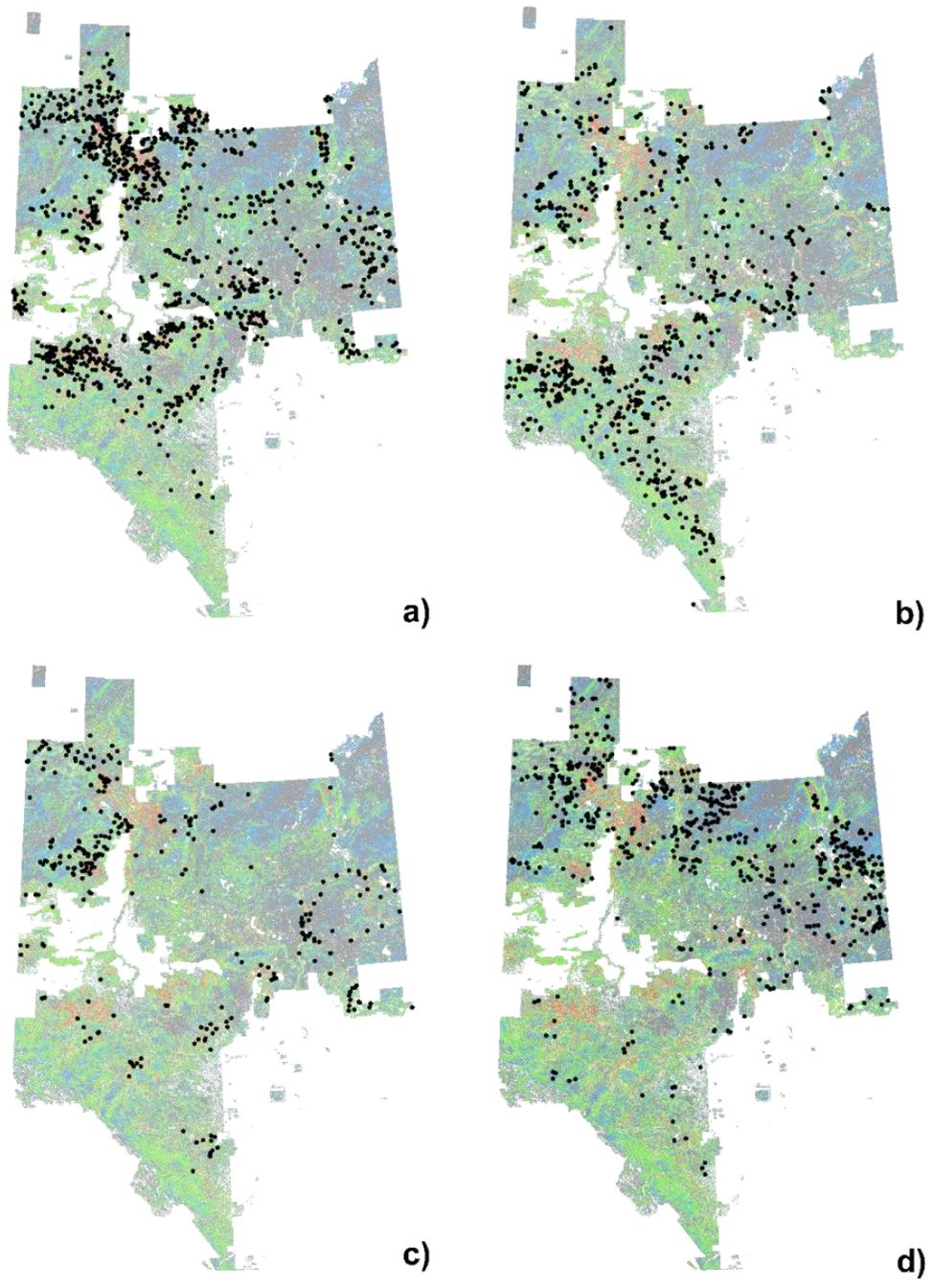


Figure 4-2. A demonstration of four habitat patch networks (habitat patches are represented by black dots): a) deciduous-dominated patches; b) coniferous-dominated patches; c) mixedwood-dominated patches; d) wetland-dominated patches.

### 4.3.2 Climate stability

The calculated climate suitability resulted in an uneven distribution pattern across the province (Figure 4-3). The Foothills Nature Region is predicted to experience increased minimum and maximum temperature and decreased frost period, whereas the boreal plains in the northern part of the province will have relatively stable climatic conditions.

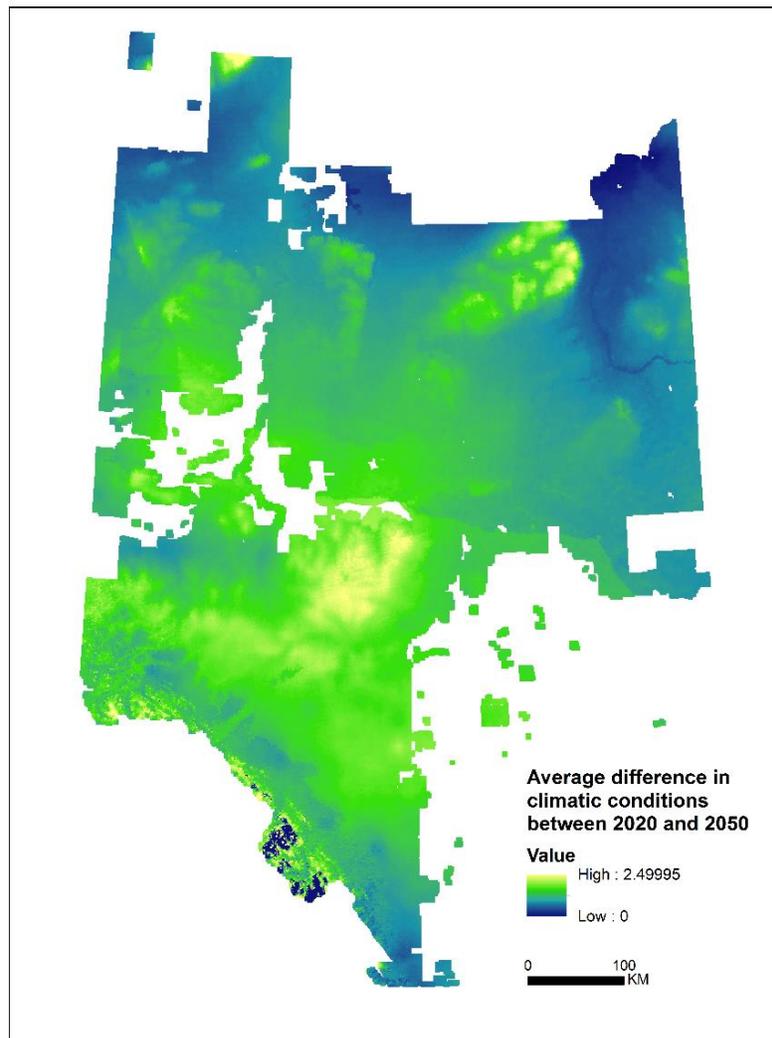


Figure 4-3. Climatic stability in the study area calculated as the weighted average differences of four climatic variables between current and future conditions: maximum temperature, minimum temperature, precipitation and frost.

### **4.3.3 Scales for constructing habitat networks**

The distance thresholds selected for networks of different forest types ranged from 35 to 80 km. The distance threshold for deciduous and coniferous dominated habitat network was 35 km (Figure 4-4). To be specific, the removal of individual patches in these two forest types did not cause major changes in landscape connectivity when networks were designed using distance thresholds greater than 35 km as alternative connector patches could easily supplement connectivity along alternative routes. At distance thresholds less than 35 km, stepping-stone patches played an irreplaceable role in connecting patches. The distance threshold for wetland-dominated networks was 30 km. Multiple peaks in connectivity loss with different distance thresholds for mixedwood-dominated networks likely reflected the dispersed distribution and local maxima of sub-components of the patch network, however 80 km was deemed the most sensitive distance (Figure 4-4). Based on these initial analyses, distance thresholds of 35 km, 35 km, 80 km, and 30 km were selected as the scales for building deciduous, coniferous, mixedwood and wetland-dominated networks, respectively.

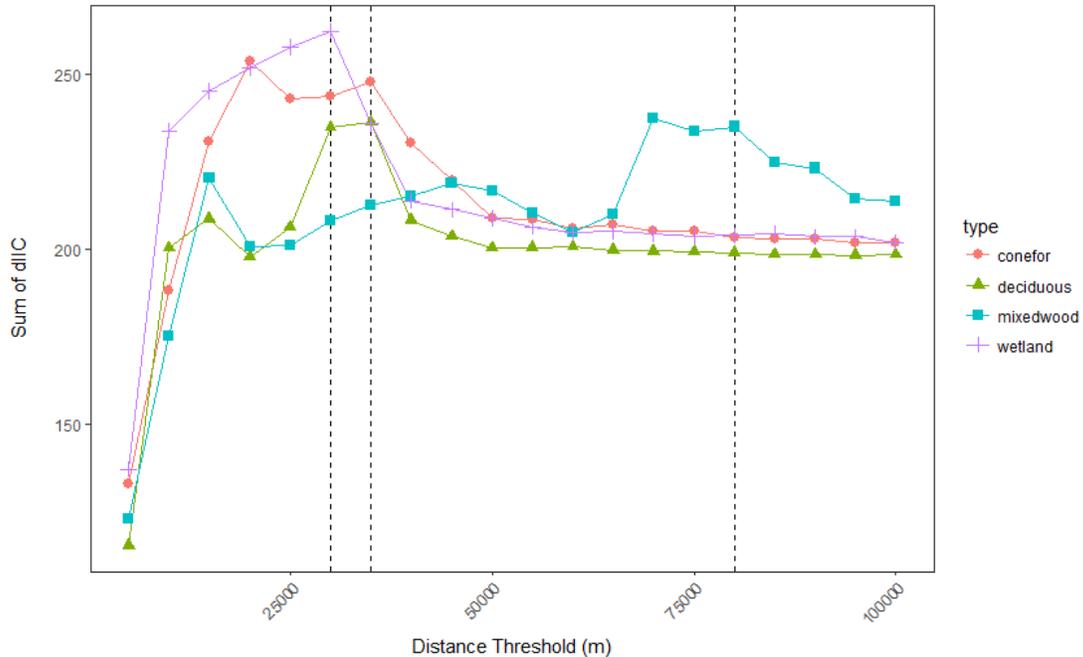


Figure 4-4. Sum of dIIC (Integral index of connectivity) using varying distance thresholds (0.5 - 100 km) to define patch/node connections in networks composed of four different forest types. Distances at 30 km, 35 km and 80 km (indicated by dashed lines) were used in building the network graphs for the simulation processes.

#### 4.4.3 Land use change simulations

For the four habitat networks, the business-as-usual scenario with no spatial prioritization demonstrated the most severe losses in connectivity (Figure 4-5) leading to highly fragmented landscapes by 2050. There were no significant differences in total connectivity loss between different conservation prioritization schemes which however, resulted in different change dynamics (Figure 4-6). When climatic stability criteria were added, almost no difference occurred between different scenarios.

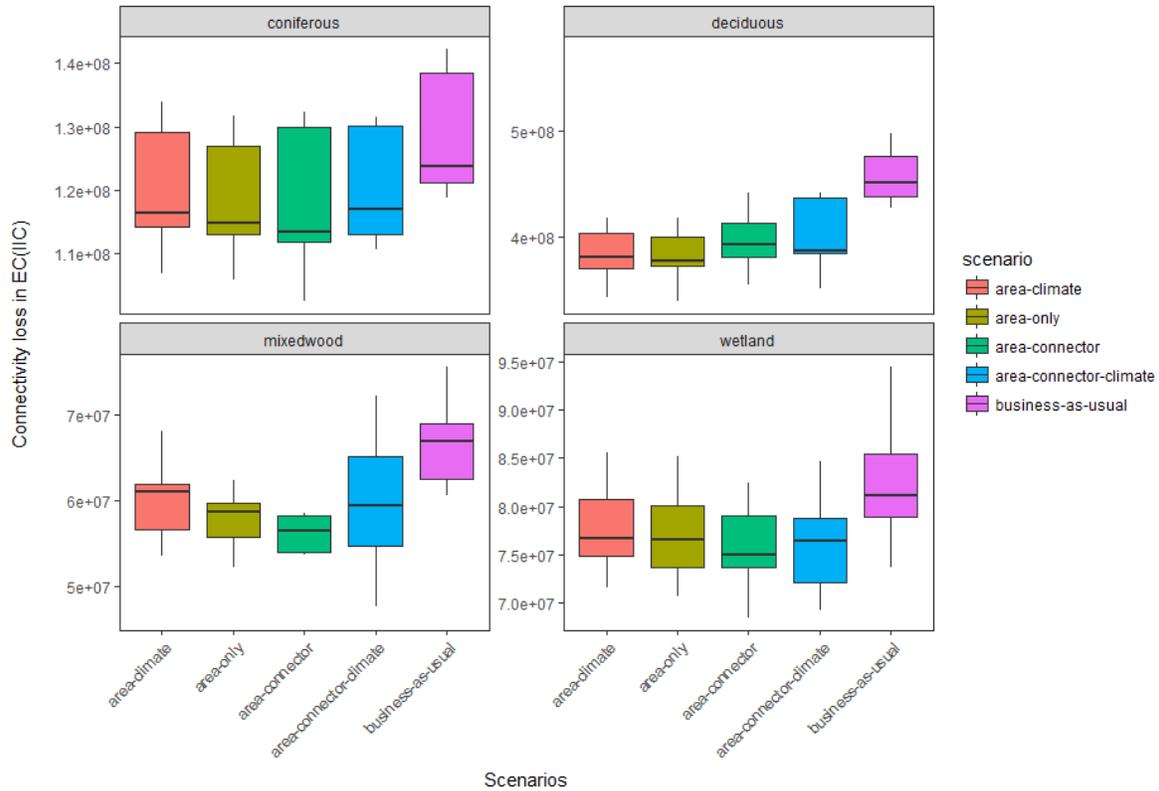


Figure 4-5. Total connectivity loss for coniferous, deciduous, mixedwood and wetland dominated patch landscape under 5 simulation scenarios. (Boxplot indicates the mean + SD, mean, mean – SD and the overall range for the result of 10 sub-graphs).

Changes in connectivity metric EC(IIC) were plotted for each time step of the area-only, area-connector and the randomized scenarios (Figure 4-6) by each forest type. During the initial five years in the simulations, the area-only scenarios resulted in a 17% - 23% decrease in landscape connectivity measured by EC(IIC), which were comparable to the business-as-usual scenario. The dramatic decrease in the landscape connectivity metric for the area-only simulation scenarios at the beginning of the simulation indicated the loss of important stepping stone patches and inverse relationship between patch size and their contribution to inter-patch connectivity. As shown in Figure 4-7, nodes A1 – A4 were scheduled for removal in the area-only scenario due to their small patch size. However, these nodes were important stepping stones for the connection of distant

nodes: node A2 shortened the link steps needed to traverse between node C1 and C2; similarly, node A3 and A4 were critical connector elements shortening the distance required to traverse between node pair C3 and C4, and node pair C5 and C6 dramatically. In comparison, the nodes removed in area-connector scenario (in purple) were on the periphery of the network configuration and were more replaceable than nodes removed in area-only scenario (in red) as alternative pathways could be identified.

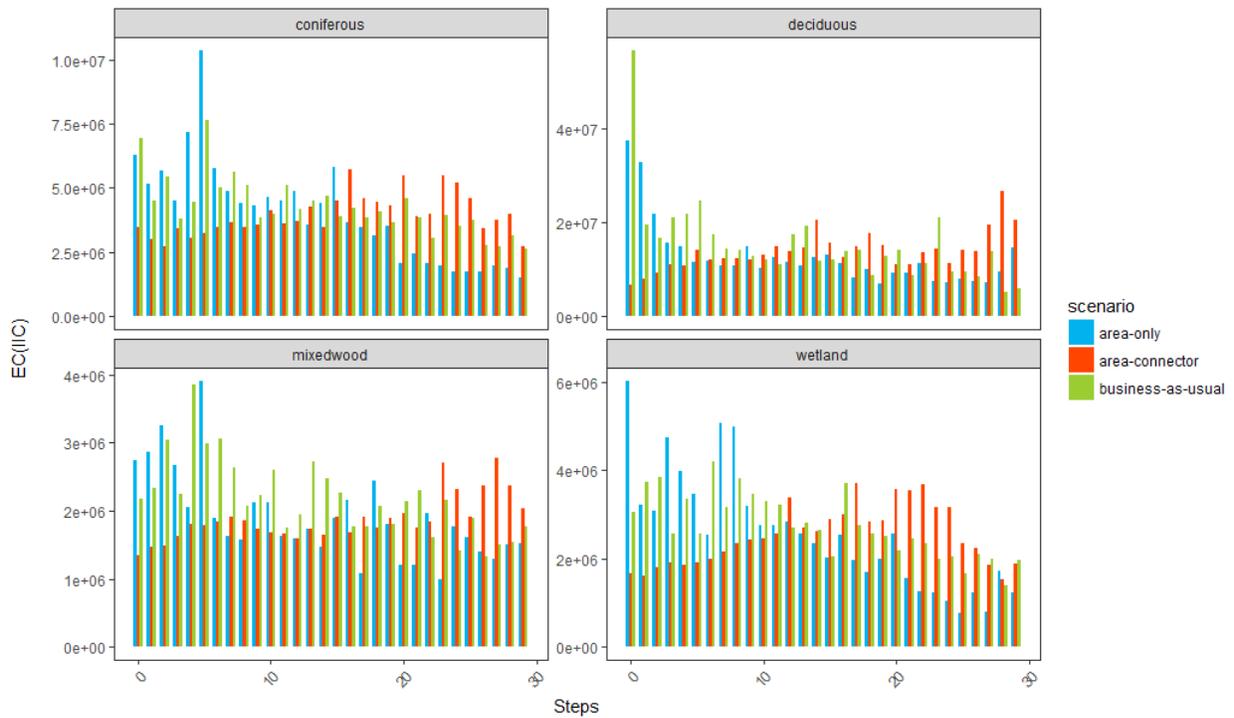


Figure 4-6. Changes in connectivity metric EC (IIC) between each step of the area-only, area-connector and business-as-usual simulation scenarios for coniferous, deciduous, mixedwood and wetland dominated patch.

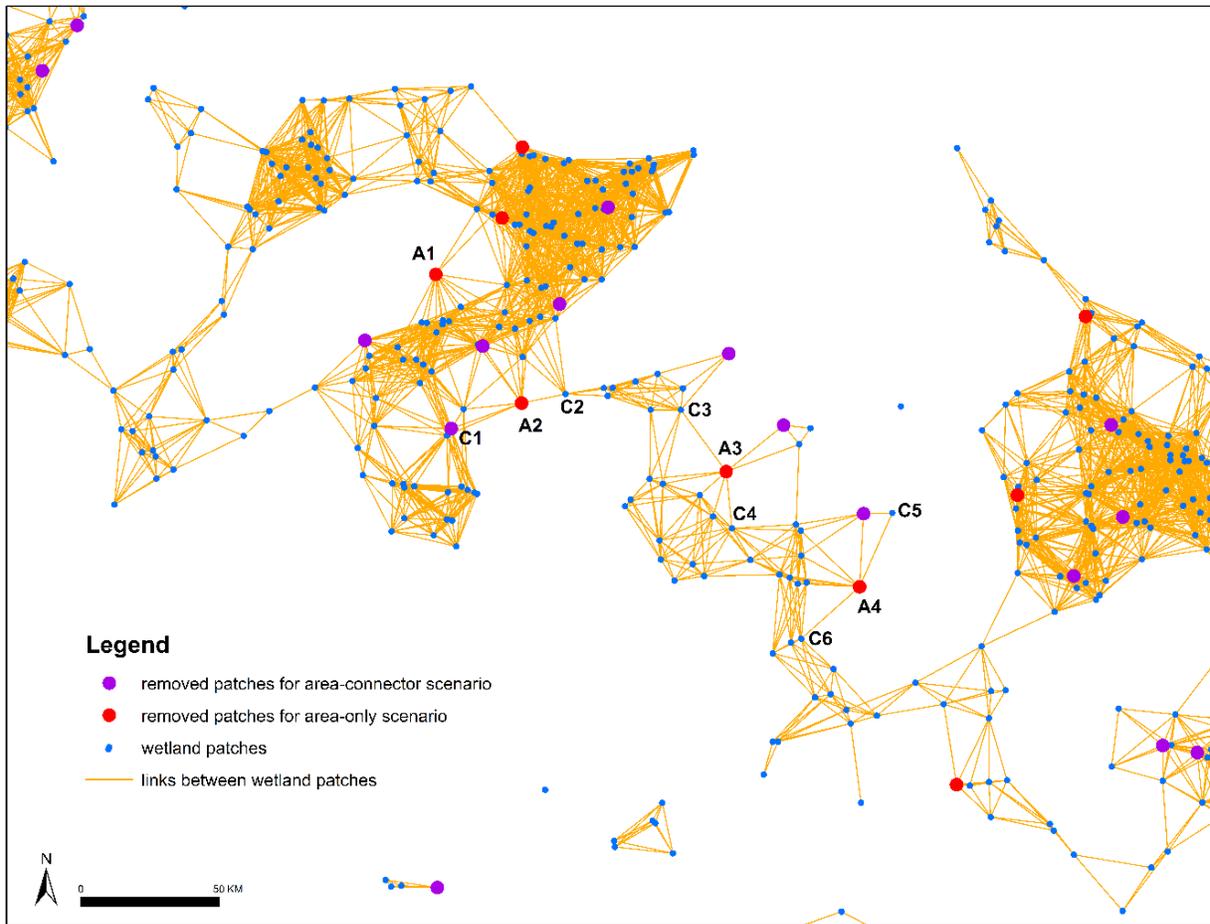


Figure 4-7. A fine-scale view of the network configuration before first step of patch removal for wetland-dominated habitat patches.

During the 30-year time steps of simulated land cover change, the area-connector scenario maintained highest overall connectivity relative to other scenarios by protecting both the core habitat and critical connector elements of the landscape (Figure 4-8). Average patch importance indicated areas of high conservation priorities for different habitat networks across the province (Figure 4-9). The Dry Mixedwood Natural Subregion had the highest concentration of critical habitat patches for deciduous-dominated landscapes, whereas the Lower and Upper Foothills Natural Subregions were saturated by important patches for coniferous-dominated landscapes. The

wetland-dominated landscape had an uneven distribution of patches, with important patches aggregated in the boreal plains in the north. Some of the most critical patches for the mixedwood-dominated landscape were isolated and could play important stepping-stone roles connecting other components of the graph.

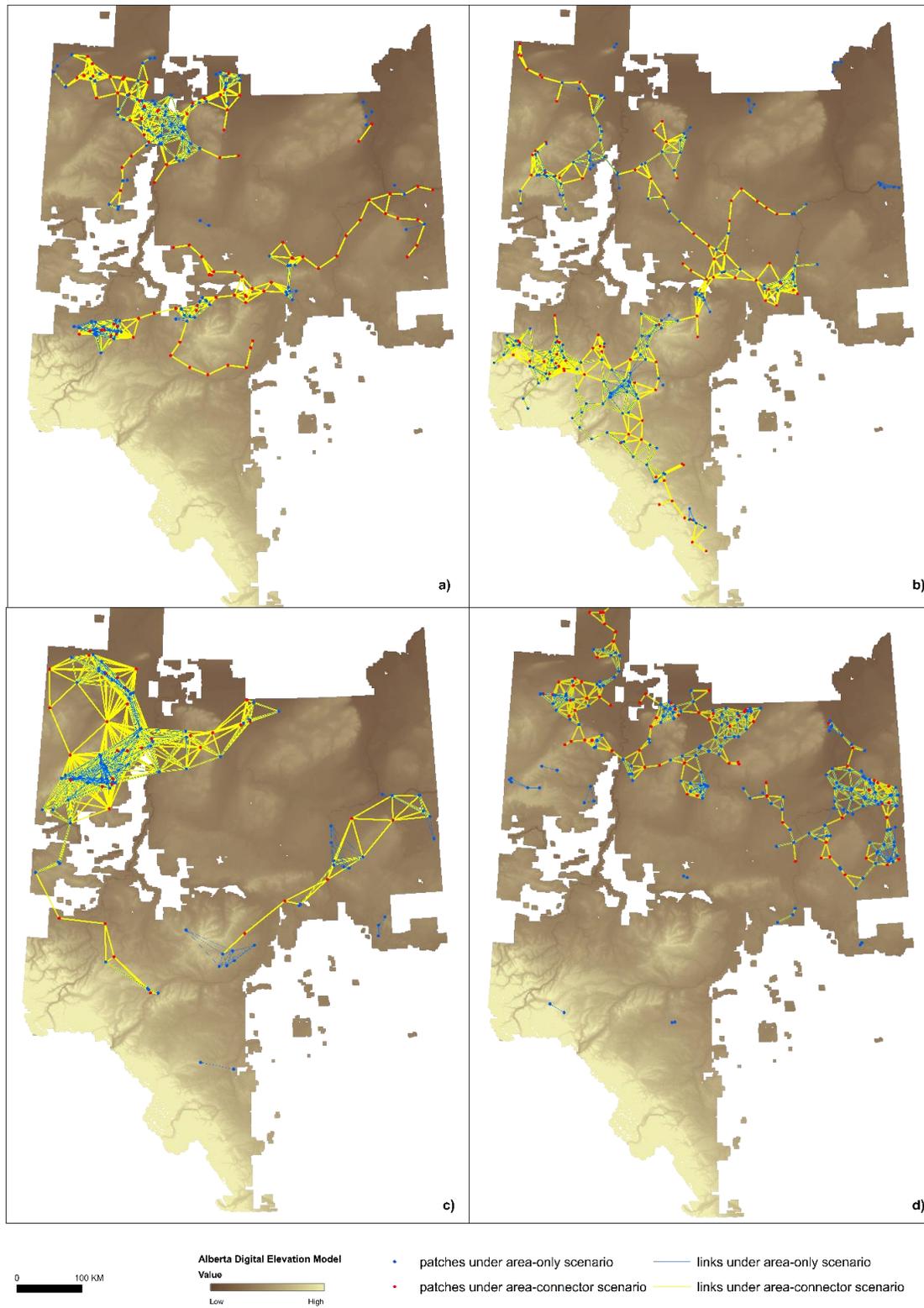


Figure 4-8. Examples of network spatial configurations at step 15 (the midpoint of the simulations) to compare area-only and area-connector scenarios for a) deciduous-dominated patches; b) coniferous-dominated patches; c) mixedwood-dominated patches; d) wetland-dominated patches.

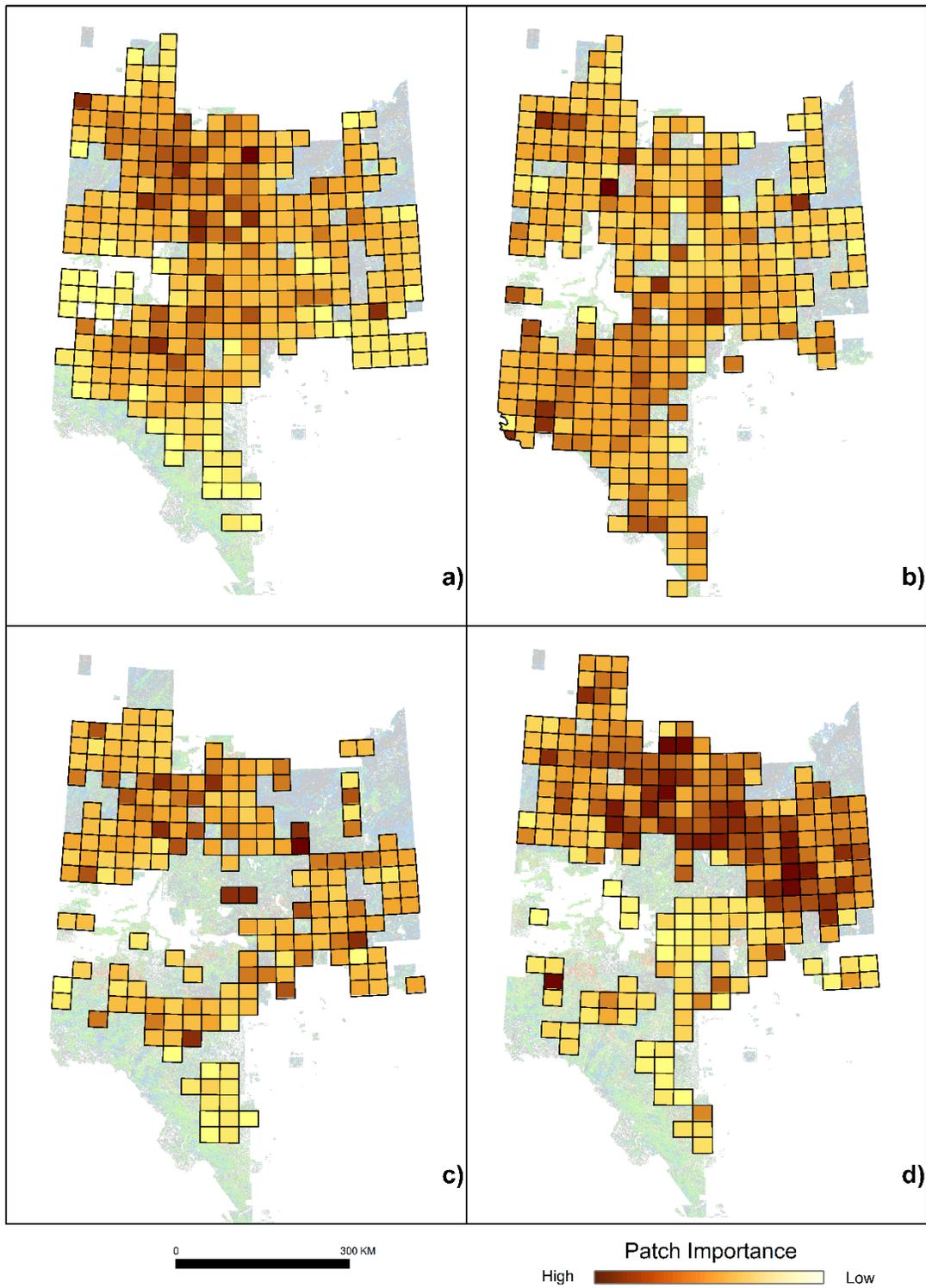


Figure 4-9. Summarized patch importance based on NTS (National Topographic System) tiles for a) deciduous-dominated patches; b) coniferous-dominated patches; c) mixedwood-dominated patches; d) wetland-dominated patches.

## 4.5 Discussion and conclusion

### 4.5.3 Land cover change simulations indicate trade-off between conservation criteria

Disentangling different landscape properties, such as the amount of habitat area and habitat configuration, is important for assessing the effectiveness of different conservation and management strategies. Preventing loss of large habitat patches (area) has been argued as the most effective way to maintain biodiversity and promote species persistence (Uezu et al., 2005; Hodgson et al., 2009; Sverdrup-Thygeson et al., 2014). In contrast, despite the well-known benefits of structural connectivity, it remains a secondary concern of many conservation prioritization exercises (Mortelliti et al., 2010). It is true that without the presence of core habitat patches, conserving important connector habitat is less effective; however, there are potential trade-offs between the role of a habitat patch as a resource-provider versus connectivity-provider, which have not yet been well explored.

In this research, the decrease in connectivity in the initial zero to five years seen in the area-only land cover change scenario helped emphasize the substantial contributions made by the smallest forest patches. Despite their small size, the overall reduction in habitat connectivity upon their removal was notable. In addition, the negative relationship between habitat area and inter-habitat connectivity for all forest types (Figure 4-6) revealed interesting landscape patterns of forest patches. In fact, García-Feced et al. (2011) found the contribution to inter-patch connectivity (as measured by *dIICconnector*) was unevenly distributed across the landscape as only 11 key connector patches were identified in the highly important category. Partitioning of the habitat availability index (IIC) to quantify different contributions of a habitat patch to connectivity

provides a more balanced approach to rank patches by considering their importance both in terms of habitat area and habitat configuration for conservation planning.

Adding climate stability constraints to either area-only or area-connector scenario did not substantially reduce connectivity. Although the spatial configurations between scenarios with and without climate constraints were different, the overall connectivity loss remained the same (Figure 4-10). This lack of impact on connectivity may be due to the homogeneous distribution of forest patches in areas with high climatic stability, which were well-connected to the rest of habitat networks. The replacement of patches with different climate stability levels did not hinder overall network connectivity. A similar result was found by Albert et al. (2017) who determined that spatial prioritization of suitable habitat patches under stable climatic conditions did not make a difference in the effectiveness of maintaining connectivity.

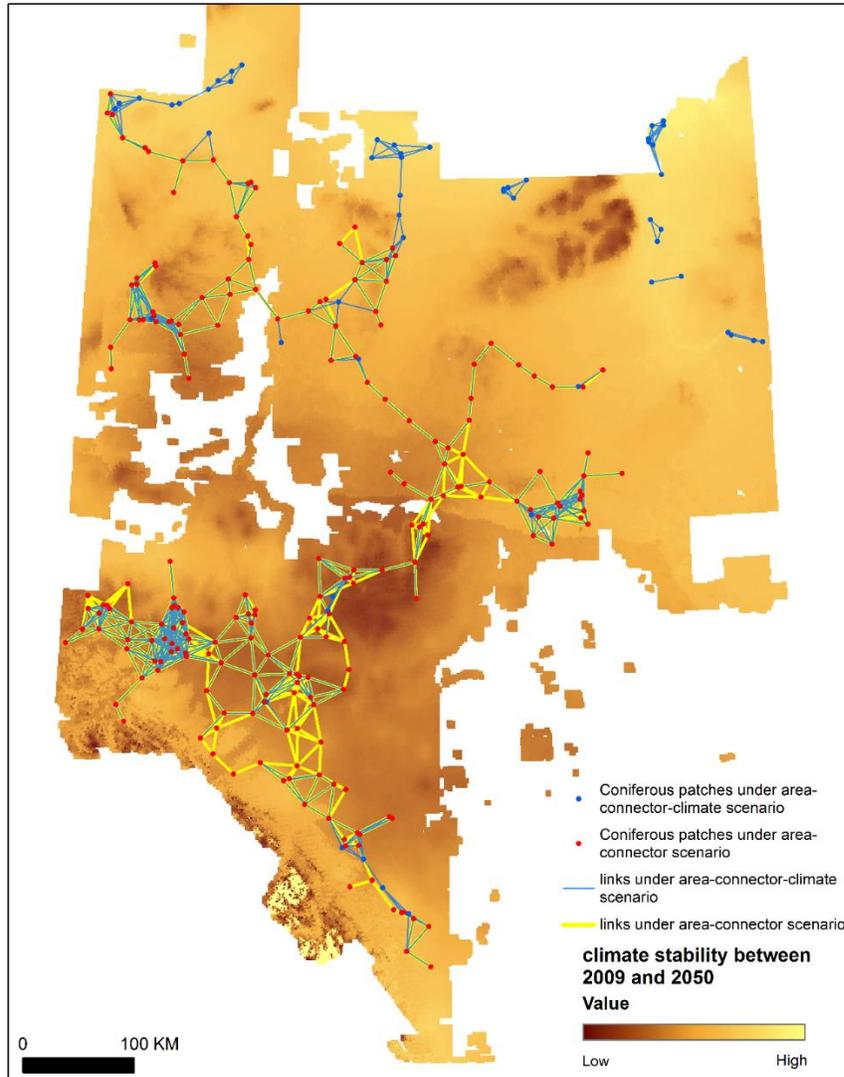


Figure 4-10. An example of network spatial configurations at step 15 (the midpoint of simulations) to compare coniferous-dominated patches distribution in area-connector and area-connector-climate scenarios.

#### **4.5.4 Enhance conservation planning by expanding protected area network and assessing structural connectivity**

Currently, the protected areas of Alberta comprise national and provincial parks, provincial recreational and natural areas where highly productive forest stands with complex stand structure are relatively uncommon (Young et al., 2006). Expanding the existing protected-area network to connect with surrounding areas of continuous, dense and complex forest cover is arguably essential to support a full representativeness of biodiversity (Joppa and Pfaff, 2009). Lidar-based forest structural assessments can enhance the identification of forest patches which are important in terms of habitat structure. The direct measurement of vegetation structure over broad areas can improve the data quality and consistency used for biodiversity monitoring and conservation planning at regional levels (Graf et al., 2009).

This study conceptualized habitat networks where only the physical distance between forest patches was used to determine if nodes (or patches) were connected. As in many other studies (Adriaensen et al., 2003; Albert et al., 2017; Xun et al., 2017), the least cost paths accounting for heterogeneity in the landscape matrix were not considered here because of the limited information available on species-specific movement, and also because of our desire to explore a more generalized, multi-species approach. As such, we focused on quantifying structural connectivity of suitable forest patches to infer potential habitats using a coarse-filter approach to conservation. Structurally connected forest landscapes provide the physical continuity of the living environment that may in turn facilitate and enhance functional, species-dependent connectivity (King and With, 2002; Saura et al., 2011). Although structural connectivity cannot guarantee functional connectivity, the chance of reaching high quality habitat through stepping-stone patches and

structural corridors can be largely improved (Bergsten et al., 2013). For example, in this analysis, maintaining a connected habitat network via key connector patches in mixedwood-dominated patch landscape would likely be beneficial for species sensitive to habitat fragmentation with medium to long distance dispersal capabilities, such as grizzly bear and woodland caribou. Our modelling framework can be easily adopted for a particular focal species with well-documented distribution patterns and dispersal behaviors (García-Feced et al., 2010). With additional species-specific information in regards to habitat requirements, dispersal capacity, and tailored climate constraints, our general modelling process could be adapted to provide optimal conservation strategies for the preservation of habitat connectivity of focal species.

#### **4.5.3 Model change dynamics to improve conservation effectiveness**

This paper provides a framework using habitat area and habitat configuration as criteria to prioritize potential forest habitat patches for conservation under future land cover change simulations. Instead of identifying critical habitat patches based on current landscape structure and climate, the simulations track long-term connectivity reductions by recalculating connectivity metrics for the remaining habitat patches in each subsequent network after successive patch removals. The dynamic modelling of changes in landscape connectivity can help resource managers forecast the most likely consequence of habitat losses under different management scenarios, evaluating both near-term and the long-term effectiveness of different conservation prioritizations (Bodin and Saura, 2010; Bergsten et al., 2013; Rubio et al., 2015). Although all prioritization scenarios met the 17% protected area target, habitat configurations varied widely across natural subregions, and the change dynamics in connectivity were different throughout the 30-year simulation timeframe. To maintain ecological integrity and long-term sustainability, a

change-dynamic modelling framework can provide in-depth, detailed information of landscape connectivity change through time.

Our evaluation of structural connectivity of potential habitat patches emphasizes the need for ecosystem-based, multi-species management approaches to conservation (McCleary and Mowat, 2002). Using forest structure-based indicators of biodiversity, regional-scale lidar remote sensing provides an efficient assessment at high spatial resolution capturing the full extent and range of forest patches with structural traits of high biodiversity potential (Guo et al., 2017). Compared to the conventional species sampling of biodiversity monitoring at local scales, our approach linked stand-level mapping of habitat quality (such as canopy density and canopy height distributions), to region-wide connectivity of available habitats, thus increasing the level of detail and improving the accuracy of landscape-level connectivity monitoring (Mortelliti et al., 2010).

## Chapter 5: Conclusion

### 5.1 Overview

Vegetation structure as an important indicator of biodiversity can be mapped over broad spatial extents using airborne lidar data. As lidar acquisitions are progressively compiled into regional coverage, characterizing fine-scale vegetation structure over broad spatial extents can serve as an inventory of vegetation structural traits for landscape-level habitat suitability mapping and species distribution modeling. Conservation of high-quality forest patches with high structural complexity provides an alternative pathway to biodiversity conservation, as these patches can provide various niches to support a diversity of forest species. In addition, the connectivity of the habitat patches should be maintained through a well-connected habitat network in to sustain energy exchange and species persistence. The capacity for species to disperse and migrate between high-quality habitats is also important to combat the negative effect of climate change. The research in this thesis identified high-quality habitat patches in managed forested areas in Alberta, Canada using regional lidar remote sensing in order to describe their spatial pattern and model the connectivity dynamics under changing landscapes. Key findings of the two research questions are:

- 1) How can we utilize a regional lidar dataset to characterize vegetation structure over a broad spatial area for biodiversity monitoring? What is the distribution pattern and forest attributes of various identified vegetation structure types?

In chapter 3, eight classes reflecting the natural grouping and general types of vegetation structure in the study area were identified using cluster analysis of six lidar-derived structural metrics in

managed forested area in Alberta, Canada. The spatial distribution of these eight structure classes varied across natural subregions. Lower Foothills and Dry Mixedwood Natural Subregions had the largest area dominated by stands of highly-complex vegetation structure. Recent anthropogenic and natural disturbance have created a unique structure class (structure class 5) with less than 1% spatial coverage. Two classes (structure classes 4 and 8) with high levels of structural complexity were used to locate high-quality habitat patches to build landscape-level habitat networks allowing investigation of their spatial pattern and connectivity. The inclusion of lidar data in vegetation structure classification can improve our ability to describe forest biodiversity patterns in addition to the information derived from passive optical remote sensing data.

- 2) What is the spatial configuration of habitat patches with important vegetation structure?  
And how does land cover change based on different spatial prioritization schemes affect habitat connectivity?

Chapter 4 built habitat networks using lidar-derived habitat patches and examined network connectivity dynamics following different conservation scenarios. Key criteria in spatial prioritization, such as habitat area, habitat configuration and climate constraints were evaluated in terms of reducing connectivity losses. Habitat networks of different forest types had varied patch distributions. The modeling result showed that conservation scenarios considering both habitat area and habitat configuration (as assessed by inter-patch connectivity) can best preserve habitat connectivity and adding climate stability did not improve the result.

Overall, the research in this thesis explored the utilization of lidar data and the derived data product for characterizing habitat structure, mapping and modeling habitat connectivity. The result of the research can provide spatially-explicit information for spatial prioritization of forest harvesting and biodiversity conservation. The processing of regional lidar dataset and its utilization in habitat structure mapping are applicable to other regions where broad-area lidar coverages are available, and regional habitat mapping and conservation prioritization are needed.

The result of this study contributes to the development of structure-based indicators and connectivity modeling framework to support decision making in biodiversity monitoring. The near wall-to-wall lidar coverage over the managed forested area of Alberta captures the full range and extent of vegetation structure and resulted in a detailed inventory of available forest patches. The derived classification of vegetation structure, when combined with species and land cover information, can be used for forest management planning, biodiversity monitoring and prioritization of conservation programs. The connectivity modeling framework demonstrated one application of the data product to model habitat connectivity change and prioritize conservation efforts to important habitat patches. As habitat loss and degradation caused by anthropogenic disturbance significantly reduce habitat area and connectivity, the modeling of network connectivity through simulations can provide insights into the protection of critical habitat patches and choosing the most effective conservation strategies at mitigating the negative impact of long-term anthropogenic disturbance. The fusion of lidar remote sensing data with landscape connectivity modeling techniques opens the door for more interdisciplinary research aimed at addressing complex issues in ecology where integrated data sources and robust modelling methods are required.

## **5.2 Key findings and research innovations**

- Using two-step cluster analysis to process the large amount of regional lidar dataset resulted in eight unique structure classes caused by different drivers of natural and anthropogenic disturbances. The structure types with high structural complexity were used to identify important habitat patches with which to construct habitat networks.
- Processing large, regional lidar data highlighted the maturity of the technology and its widespread uptake and utilization.
- Vegetation structure-based indicators were proposed as surrogates to reflect ecosystem-based, multi-species biodiversity potential, representing the full range of trophic levels and habitat specializations of biodiversity.
- Conservation strategies considering both habitat area and habitat configuration were demonstrated to be the most effective way of mitigating connectivity loss of habitat networks during land cover change processes.
- This modeling approach not only prioritized critical habitat areas for biodiversity conservation and identified best conservation strategies, but also monitored the change dynamics of landscape structural connectivity in face of habitat losses during long-term land cover change events, which had rarely been done previously.

## **5.3 Limitations**

The lidar dataset used in this study is a provincial compilation acquired over multiple years and seasons by various data providers. Mixed seasonalities and phenological growth stages of the forest could affect the accuracy of structure classification. Although leaf-on and leaf-off conditions

may not have a significant impact on lidar data processing (White et al., 2015), the lack of knowledge about the species composition and stand conditions at the time data were acquired prevents more accurate and precise interpretation of the classification result. In addition, the impact of subsequent natural and anthropogenic disturbances since lidar acquisition was not present in the data; therefore, current conditions of vegetation structure cannot be accurately described. Furthermore, the broad spatial coverage of the study area restricted our ability to obtain species-specific habitat and matrix information. Rather, structural complexity was applied as the biodiversity indicator with which to select habitat patches with high biodiversity potential. Validation of this assumption remains to be examined with adequate and comprehensive field survey data. Species-dependent distribution data can also describe habitat suitability and dispersal capacity more precisely and in detail for constructing habitat network and modeling network connectivity.

## **5.4 Directions for future studies**

### **5.4.1 Improve classification of vegetation structure**

A variety of lidar-derived structural metrics measuring canopy heterogeneity, surface roughness and layering architecture (Goetz et al., 2007; Vogeler et al., 2013) can be used as substitutes of standard deviation of height to represent structural complexity. In this way, complexity in the upper, middle and lower strata of the canopy could be described separately for species dwelling in different layers of the forest canopy. More detailed structure classification could be achieved for a refined representation of habitat structure. Vegetation structure across natural subregions and districts of land use frameworks can be different because of various climatic conditions,

topography, and land use policies. Classification of vegetation structure in the same subregion with consistent land use policies could potentially increase the robustness and certainty of the classification regime. A more consistent and accurate classification result could be produced.

As forest landscape is changing under various natural and anthropogenic disturbance regimes, it is necessary to track the changes of vegetation structure through time and space to inform management decisions. Time-series land cover changes derived from Landsat satellite data can be used to update the status of vegetation structure described in this research and build a temporal trajectory of structural changes for monitoring habitat suitability and constructing dynamic habitat networks.

#### **5.4.2 Strengthen modeling of habitat connectivity**

Species-specific information on habitat selection including forest species, age and height composition, moisture regime and terrain feature could also be considered when identifying important habitat patches. Data on species distribution and movement can help evaluate and rank habitat suitability and accessibility to build a more complex and realistic habitat network for connectivity modeling. The matrix resistance to dispersal between suitable habitat patches could also be quantified to improve the accuracy of connectivity estimation. In this way, species-specific habitat network models can be generated to monitor connectivity dynamics based on in-situ data.

Forest harvest planning normally considers several ecological, topographical and operational constraints into harvest design and cutblock layout. Factors such as slope, species composition, soil nutrient and moisture, distance to road and social-economic concerns could be incorporated into land cover change simulations to model realistic land cover change events. Sophisticated multi-criteria optimization of forest management and conservation activities could be explored to generate spatial prioritization schemes that preserve habitat connectivity under changing climate while maximizing the value of forest operations.

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