

**Patterns of Vegetation Structural Diversity Across Heterogeneous
Landscapes in Southwestern Nova Scotia**

By

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Abstract

Forest edges, including transitional areas between forest and non-forest areas, outline the overall structure of the landscape. To assess and quantify patterns of structural diversity across natural and harvested landscapes in southwestern Nova Scotia, I used field-based structural diversity metrics and UAV imagery along two 1250 m transects to examine different aspects of the pattern of structural diversity across transitions in forested landscapes. For traditional field metrics, tree structural diversity had more success in determining transitions than functional plant group diversity, as tree structural diversity detected all edge types compared to just anthropogenic edges when using functional plant group diversity. For photogrammetrically derived metrics, no metric detected transitions at all edges and overall UAV metrics were incompatible with field sampling. Future studies should examine the compatibility of LiDAR and structural diversity metrics.

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Chapter 1: Introduction

My research aims to increase the understanding of patterns of vegetation structure in Nova Scotia's western ecoregion. This forested landscape houses a large portion of natural protected areas in the province as well as anthropogenically disturbed landscapes as a result of the province's forestry industry. Many of these areas play a role in the conservation of species at risk within the province. My primary goals focus on quantifying structural diversity at transition zones or edge ecosystems in natural and harvested landscapes using a variety of methodological approaches from traditional field sampling to utilizing drones for imagery collection and sampling.

1. Forest Edges and Landscape Patterns

The transition zone or edge ecosystem is an integral part of landscapes as it forms the overall mosaic structure. A forest edge is a transitional boundary between forest and non-forest ecosystems, and it can be either natural or anthropogenic (Harper et al., 2005). Natural edges are caused by disturbances such as wind, insect outbreaks or fire as well as inherent natural edges such as wetlands. In contrast, anthropogenic edges are caused by anthropogenic disturbances such as clear-cutting forest stands or linear disturbances in forests created by roads and power lines. Recent research has identified gaps in edge studies across different geographic locations (Franklin et al., 2021) and found that anthropogenic edges were more frequently studied than natural forest edges. Previous edge studies mostly examined forest structure and composition by comparing edge influence between two distinct types of disturbance-based boundaries, a natural boundary, such as those created by wildlife and an anthropogenic boundary, such as those created by clear-cutting (Harper et al., 2004).

Examining vegetation patterns with regard to vegetation structure presents an opportunity for further research and the importance of forested wetlands, which are inherent edges and crucial habitats. In the Acadian forest, forested wetland edges have been previously determined to have few clear patterns in vegetation structural diversity other than lower structural diversity at edges (Harper et al., 2021).

The spatial pattern of plant communities is crucial for ecosystems as it forms the overall patch mosaic of structure on the landscape (Johnston et al., 1992). Most natural landscapes are heterogeneous in structure. These landscapes are composed of edges and transition zones between patches of varied vegetation structure. As such, the forest edges house elements from either side of the patch it delineates (McIntire & Fajardo, 2014).

2. Conservation Value of Southwestern Nova Scotia's Forested Landscapes

As identified by Harper and Staicer (2021), there is a lack of knowledge of forested wetlands in Atlantic Canada. Forested wetlands are important for conservation as they provide valuable habitats for many at-risk species (Harper & Staicer, 2021). In Nova Scotia, these species include lichens such as the endangered boreal felt lichen (*Erioderma pedicellatum*) (Cameron & Toms, 2016) to songbirds such as the threatened Canada Warbler (*Cardellina canadensis*) (Westwood et al., 2020).

Boreal felt lichen or *E. pedicellatum* is a foliose cyanolichen (Padgett & Wiersma, 2020). The Atlantic population is located within the Atlantic coastal forest of Nova Scotia in the western ecoregion. Early habitat modelling utilizing large-scale habitat mapping on a landscape level identified areas within the Atlantic coastal forest suitable for *E. pedicellatum* (Cameron & Neily, 2008). Nevertheless, these models did not consider the potential for a

patchy distribution of forested wetlands as suitable habitats for this species. Conservation management is aimed to favour adult survival, and identifying suitable habitats for this species contributes to ongoing management (Power et al., 2018). Newer predictive models use structural diversity indicators such as tree diameter at breast height (DBH), tree basal area, crown closure, and site conditions to map wetlands and predictive habitats (Power et al., 2018). *Erioderma pedicellatum* only sometimes fell within mapped wetland areas and predictive habitat; it was found in areas with poor drainage and that had significantly higher tree density but a lower basal area of living trees and a lower crown closure (Power et al., 2018). This conclusion suggests that smaller patches of forested wetlands across the landscape could be suitable habitat areas for *E. pedicellatum* within the western ecoregion.

Forested wetlands within the Atlantic Acadian Forest, which is a mixed wood forest spanning most of Nova Scotia, New Brunswick and Maine, is a habitat for songbirds in Nova Scotia. The Olive-sided Flycatcher (*Contopus cooperi*) and the Canada Warbler (*Cardellina canadensis*) are both threatened bird species in Atlantic Canada (Westwood et al., 2019). Protected areas, such as national parks (i.e. Kejimikujik National Park), play a crucial role in aiding habitat connectivity for breeding populations of these threatened species (Westwood et al., 2019). Canada warblers utilize forested wetlands and young post-harvest forests, which are both areas with higher structural complexity. The studies suggest that comparison data and future breeding studies are needed for future conservation management decision-making (Westwood et al., 2020). Thus, identifying and understanding the patterns of structural diversity of these critical blocks of natural and anthropogenic habitat provides information for action by various land conservancies and government managers at the regional and

provincial levels and contributes to a further understanding of the suitable habitats for multiple species at risk.

3. Structural Diversity

Structural diversity is defined as how complex or diverse vegetation is in a given geographic area (LaRue et al., 2019; Storch et al., 2018). Structural diversity can be seen as an alternative to species diversity as it is a component of biological diversity. In the past, stand structural diversity, using species, diameter and height to assess vertical and horizontal stand structure, has been examined by creating classes and evaluating the dataset using the Shannon index (Staudhammer & LeMay, 2001). A recent study by LaRue et al. (2019) highlighted how structural diversity metrics, compared to species richness, are better methods for examining ecosystem functions such as productivity. The results concluded that using multiple levels of scale to assess structural diversity could provide a new lens for studying vegetation structure and function across the landscape (LaRue et al., 2019). Another study breaks structural diversity into three distinct categories, species diversity, tree size diversity and spatial diversity, but excludes patterns of structural diversity from its objectives (McRoberts et al., 2008).

As many heterogenous landscapes have varied vegetation structure due to the nature of their composition, many studies examining structural diversity do so through the lens of examining their spatial pattern across the landscape. For example, the spatial pattern of structural diversity has been examined across the boreal forest-tundra ecotone using five types of structural diversity: ground cover, herbaceous plants, woody plants, all live plants,

and deadwood (Dodonov & Harper, 2022), as well as between forested wetlands and upland forest using vegetation structural categories (Harper et al., 2021).

Vegetation structural diversity can be measured as field-based metrics such as stand structure or functional plant groups as vegetation categories or by using remote sensing practices such as light detection and ranging (LiDAR) or aerial photography for measures of canopy complexity (LaRue et al., 2019). Previously mentioned studies examined structural diversity at the stand level, using the number of tree species of different sizes (Godlee et al., 2021; Staudhammer & LeMay, 2001) and vegetation structural components such as shrubs, herbs and trees to determine patterns of structural diversity between forested wetlands and upland forests (Harper et al., 2021) and on a lakeshore to forest gradient to examine endangered lakeshore flora (Dazé Querry & Harper, 2017).

While many studies utilize field or ground-based sampling, structural diversity, given its intrinsic ties to vertical or spatial complexity, can also be evaluated by remote sensing practices such as drone imagery, LiDAR and aerial photography. Early studies aimed to determine the spatial distribution of trees from aerial photos by identifying tree crowns (Utterera et al., 1998). As technology advances and the use of drones becomes more accessible, the accuracy and quality of image collection and processing allow for a more precise set of imagery to be collected; 3D point clouds can be composed and used for analysis, such as estimating tree height (Jensen & Mathews, 2016; Panagiotidis et al., 2017) which can be considered a metric of vertical complexity or structural diversity. Alternate remote sensing methods such as LiDAR can assess both spatial and vertical complexity as it creates not only a picture of height but of the ground as well. Aerial LiDAR has had more

success in quantifying broader scale variation in structural diversity compared to terrestrial LiDAR, which was better at quantifying on a finer site-specific scale (LaRue et al., 2020).

4. Research Objectives

The overarching goal of this master's project was to assess and quantify patterns of structural diversity across heterogeneous landscapes in southwestern Nova Scotia. Chapter 2 aimed to locate changes in structural diversity patterns in forested landscapes, while Chapter 3 aimed to assess and determine landscape structure at multiple scales.

In Chapter 2, I asked research questions such as: Are there abrupt changes in structural diversity at edges or peaks at edges? Are there distinct patches on the landscape, or are there distinct boundaries or gradual transitions? I hypothesized that structural diversity is highest at and around edges. My objectives were 1) to determine if there are abrupt or gradual changes in structural diversity at edges or other areas of the landscape, 2) to compare patterns of vegetation structure and structural diversity in natural and harvested landscapes, and 3) to compare locations of significant transitions to other variables. Other variables included elevation, canopy cover, stand density, basal area and tree species diversity.

In Chapter 3, my objectives were to 1) describe the pattern of structural diversity across transitions in forested landscapes with drone imagery, 2) compare indices of structural diversity from drone imagery, and 3) compare indices between drone imagery and field measures. I aimed to compare the number of distinct boundaries for both harvested and natural landscapes using different indices of structural diversity derived from image-based point clouds, a canopy height model, and field sampling of vegetation structural elements.

5. Study Area

My study was located in the Western ecoregion of Nova Scotia. Nova Scotia is one of the eastern maritime provinces of Canada and the western ecoregion is located in the southwestern portion of the province (Figure 1-1). This region is considered mild in temperature compared to the rest of Nova Scotia and houses many of the province's forested areas (Nova Scotia Department of Natural Resources, 2017). This region's proximity to the Atlantic Ocean and the Bay of Fundy shows milder temperatures and increased precipitation, giving rise to extensive wetland areas. The climate is classified as a Dfb under the Koppen Geiger classification system. Nearby Bridgewater's average daily temperature is - 5 °C in January to 19 °C in July, with precipitation ranging from 96 mm to 165 mm in its wettest month (Environment and Climate Change Canada, 2023).

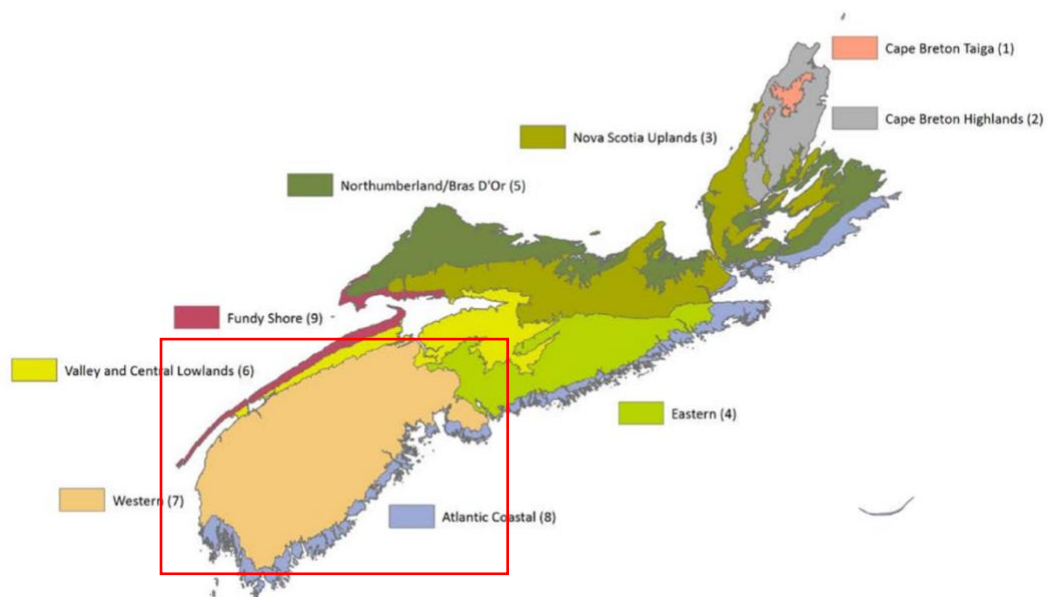


Figure 1-1: Ecoregions of Nova Scotia. The study location is found in the western ecoregion highlighted by the red box. Figure adapted from Government of Nova Scotia (2016).

Much of the western ecoregion has historically been impacted by harvesting (Nova Scotia Department of Natural Resources, 2017). Additional land use in this ecoregion includes the province's largest protected area, Kejimikujik National Park. As seen in most of Nova Scotia, this ecoregion is considered part of the Acadian Forest, with surrounding coastal areas part of the Maritime Boreal Forest (Nova Scotia Department of Natural Resources, 2017). The Acadian Forest houses several different forest types, but spruce hemlock or mixed wood forests are most frequent in the western ecoregion (Nova Scotia Department of Natural Resources, 2017).

Kejimikujik National Park was established in 1974 and protected some of Nova Scotia's old-growth hemlock forests from harvesting occurring in the region. Currently, the Hemlocks and Hardwoods trail takes visitors through a hemlock stand that is over 300 years old (Parks Canada, 2022). In contrast, the Medway Lakes Wilderness Area was established in 2015 in southwestern Nova Scotia and combined previous nature reserves in the area and the former Bowater Mersey woodlands (Department of Environment and Climate Change, 2015). One of our study transects (transect 2— see below) in the old Bowater Mersey woodland was harvested prior to acquisition by the province and has forested stands composed of plantation and harvest. Other areas of this region and along the transect have also been impacted by pre-commercial thinning and herbicide use.

6. Methodological Approach

Most studies on structural diversity use a quadrat and transect method to collect data, but they often use short-length transects to focus on only one edge per transect (Dazé Querry & Harper, 2017). However, some previous studies have examined landscape structure using

longer transects and multiple scales. For example, Saunders et al. (1998) studied the spatial distribution of patches related to temperature and abiotic features using a long transect of 3820 m, while Brosofske et al. (1999) examined plant species distribution along a 3575 m transect in the pine barrens landscape of northern Wisconsin, USA. Using longer transects with contiguous quadrat sampling allows for the assessment of structural and edge influences across multiple spatial scales, which is a finer scale than typically used in landscape ecology and larger than typically seen in plant ecology. Although using these longer-than-normal transects allows a more robust examination of landscape structure, it is harder to sample given the extended need for time and higher cost to sample at this length than shorter transects. In this study, I used two long, 1250 m transects (Figure 1-1). There was one transect placed in each landscape type, natural and harvested, using a contiguous quadrat sampling method.

Table 1-1: Coordinates of the start and end quadrat for each transect

	Transect 1: Natural Landscape	Transect 2: Harvested Landscape
Start	65.2515060°W 44.4518710°N	65.2300710°W 44.5480720°N
End	65.2665080°W 44.4548060°N	65.2288060°W 44.5369190°N

Technological advancement supporting aerial photography introduces the possibility of assessing spatial heterogeneity across the landscape at a new level of scale (Turner & Gardner, 2015). With the ability to use and apply remote sensing principles, the study of landscapes on a large scale allows for not only an abundance of widely available data such as that from light detection and ranging (LiDAR) (LaRue et al., 2020; Ozdemir & Donoghue,

2013; Zellweger et al., 2013) but also for study on finer scales than typically utilized in landscape ecology, such as with UAV or drone imagery (Jensen & Mathews, 2016; Li et al., 2017).



Figure 1-2: Photo of DJI Phantom 4 RTK used for data collection.

As a platform for remote sensing in plant ecology, UAVs present a cost-effective method for evaluating the landscape. The primary applications of UAV remote sensing for plant ecologists are for vegetation mapping, monitoring and measuring, ranging from the species level of scale to the landscape level of scale (Koh & Wich, 2012; Sun et al., 2021). An abundance of data can be collected in a short period of time when using drones compared to the time it would take to collect data from the ground. However, a cost-benefit trade-off exists between certification and legal regulations for operating UAVs (Werden et al., 2015). The highest potential cost is becoming a certified pilot, often remedied by a community or industry partner (Werden et al., 2015). In my case, the fee of pilot certification was waived by utilizing services from the Maritime Provinces Spatial Analysis Lab (MP_SpARC Lab) in

the Department of Geography at Saint Mary's University. This lab provides technician services and discounted equipment rental fees for students of the university, which removes the barrier associated with training and insurance costs. The drone used for this project's data collection can be seen in Figure 1-2 above.

Each of the two sites were chosen to maximize the number of visually determined transitions across the landscape within 1250 m. Each site examines a type of landscape, natural or harvested. Site 1, which examines the natural landscape, was within Kejimikujik National Park. This site started in Nixon Meadows Brook and went from the wetland surrounding the brook through a multi-age forest into the old-growth hemlock forest located on the Hemlocks and Hardwoods Trail. Site 2, which examines the anthropogenically disturbed or harvested forest, was within the Medway Lakes Wilderness Area. This site started in a clear-cut near Frog Lake and went through a multiage spruce forest that was planted and precommercial thinned, crossing a former logging road and into a bog. Both 1250 m transects were sampled using field sampling and UAV imagery in 2022.

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Chapter 2: Patterns of vegetation structural diversity across forested landscapes

Abstract

Structural diversity is an alternative metric to species diversity when examining the biological diversity of vegetation, particularly in more temperate climates with lower species diversity. To gain a more robust understanding of the pattern of structural diversity in the Acadian Forest, a type of temperate forest, I examined two indices of structural diversity across various edge types within two distinct forested landscapes in southwestern Nova Scotia, Canada. My research objectives were 1) to determine if there are abrupt or gradual changes in structural diversity at edges or other areas of the landscape, 2) to compare patterns of vegetation structure and structural diversity in natural and harvested landscapes, and 3) to determine if locations of significant transitions align for different variables. I contiguously sampled vegetation in 2.5 x 5 m quadrats along two 1250 m transects in harvested and natural landscapes in southwestern Nova Scotia. I used two indices of structural diversity: functional plant group diversity with 14 distinct structural groups of vegetation and tree structural diversity categorized using living vs. dead, four height categories, four decay stages for dead trees and DBH classes for all trees. Functional plant group diversity had abrupt transitions at anthropogenic edges but more gradual transitions at natural edges, whereas tree structural diversity showed significant transitions at all edge types. The natural forested landscape had fewer significant transitions than the harvested landscape for functional plant diversity. Tree structural diversity showed many significant transitions throughout the planted and natural spruce dominant stands, likely linked to changes in canopy cover. Changes in elevation, basal area, stand density and tree species diversity did not align with significant transitions. Further research should focus on expanding this

analysis to include more edge types and more environments. Ultimately, tree structural diversity is the best metric of field sampling for detecting edges using vegetation structural diversity.

1. Introduction

Structural diversity represents the level of complexity of vegetation in a given geographic area (LaRue et al., 2019; Storch et al., 2018). It is an alternative to species diversity; instead of looking at individual species, it groups vegetation as pseudo-species based on similarity in their structural characteristics. Most natural landscapes are heterogeneous in structure, with edges and transition zones between various structurally unique areas. A forest edge can be considered a transitional boundary between forest and non-forest ecosystems and can be natural or anthropogenic in origin (Harper et al., 2005). A heterogeneous landscape harbours a more varied vegetation structure at an edge as it houses elements from both sides of the patch it delineates (McIntire & Fajardo, 2014). The transition zones or edge ecosystems define and compose the overall arrangement of the landscape and play an essential role in forming the mosaic structure. I hypothesize that structural diversity should peak at and around edges, given that it has elements from patch on either side.

Previous studies examined structural diversity at the stand level, using the number of tree species of different sizes (Godlee et al., 2021; Staudhammer & LeMay, 2001). Other field-based measures include an abundance of vegetation structural components such as shrubs, herbs and trees to determine patterns of structural diversity between forested wetlands and upland forests (Harper et al., 2021) and on a lakeshore to forest gradient (Dazé Querry & Harper, 2017). While most of these studies employed a quadrat and transect method for data

collection on structural diversity, they are usually placed at a single edge (Dazé Querry & Harper, 2017) and utilize short transects. In previous decades, limited studies examined landscape structure in the context of temperature (Saunders et al., 1998) and vegetation response (Brosofske et al., 1999) using multiple levels of scale with long transects over 3 kilometres in length. When using a more extended transect, assessing the pattern of landscape structure at multiple spatial scales is possible at a scale larger than typically seen in plant ecology and much finer than is common in landscape ecology. My research objectives were 1) to determine if there are abrupt or gradual changes in structural diversity at edges or other areas of the landscape, 2) to compare patterns of vegetation structure and structural diversity in natural and harvested landscapes, and 3) to determine if locations of significant transitions align with other variables.

2. Methodology

a. Study Area and Site Selection

The study area is in southwestern Nova Scotia (eastern Canada) within the province's western ecoregion. This ecoregion is part of the Acadian Forest and is comprised primarily of spruce hemlock or mixed wood forests (Nova Scotia Department of Natural Resources, 2017). This region is considered mild in temperature compared to the rest of Nova Scotia and houses many of the province's forested areas (Nova Scotia Department of Natural Resources, 2017). Given this region's proximity to the Atlantic Ocean and the Bay of Fundy, it has milder temperatures and increased precipitation compared to the rest of the province, giving rise to extensive wetland areas. Bridgewater's average daily temperature is - 5 °C in January

to 19 °C in July, with precipitation ranging from 96 mm to 165 mm in its wettest month (Environment and Climate Change Canada, 2023).

Historically much of the area has been impacted by historical and commercial harvesting, planting, pre-commercial thinning, and herbicide use, such as in the Medway Lakes Wilderness Area. Southwestern Nova Scotia also houses the province's largest protected area, Kejimkujik National Park.

Two sites were selected using Google Earth imagery and the Nova Scotia Provincial Landscape Viewer (Government of Nova Scotia, 2022). The Landscape Viewer is an online ESRI-based geographic information system (GIS) interface that houses the province's publicly available geographic information. Using these platforms, each site was chosen to maximize the number of visually determined transitions across the landscape within 1.25 km. Each site examines a type of landscape, natural or harvested. Site 1, which examines the natural landscape, was within Kejimkujik National Park. This site started in Nixon Meadows Brook (Figure 2-1: B) and went from the wetland surrounding the brook through a multi-age forest into the old-growth hemlock forest (Figure 2-1:C). Site 2, which examines the anthropogenically disturbed forest, was within the Medway Lakes Wilderness Area. This site started in a clear-cut near Frog Lake (Figure 2-1: E) and went through a multiage spruce forest, crossing a former logging road and into a bog (Figure 2-1:D).

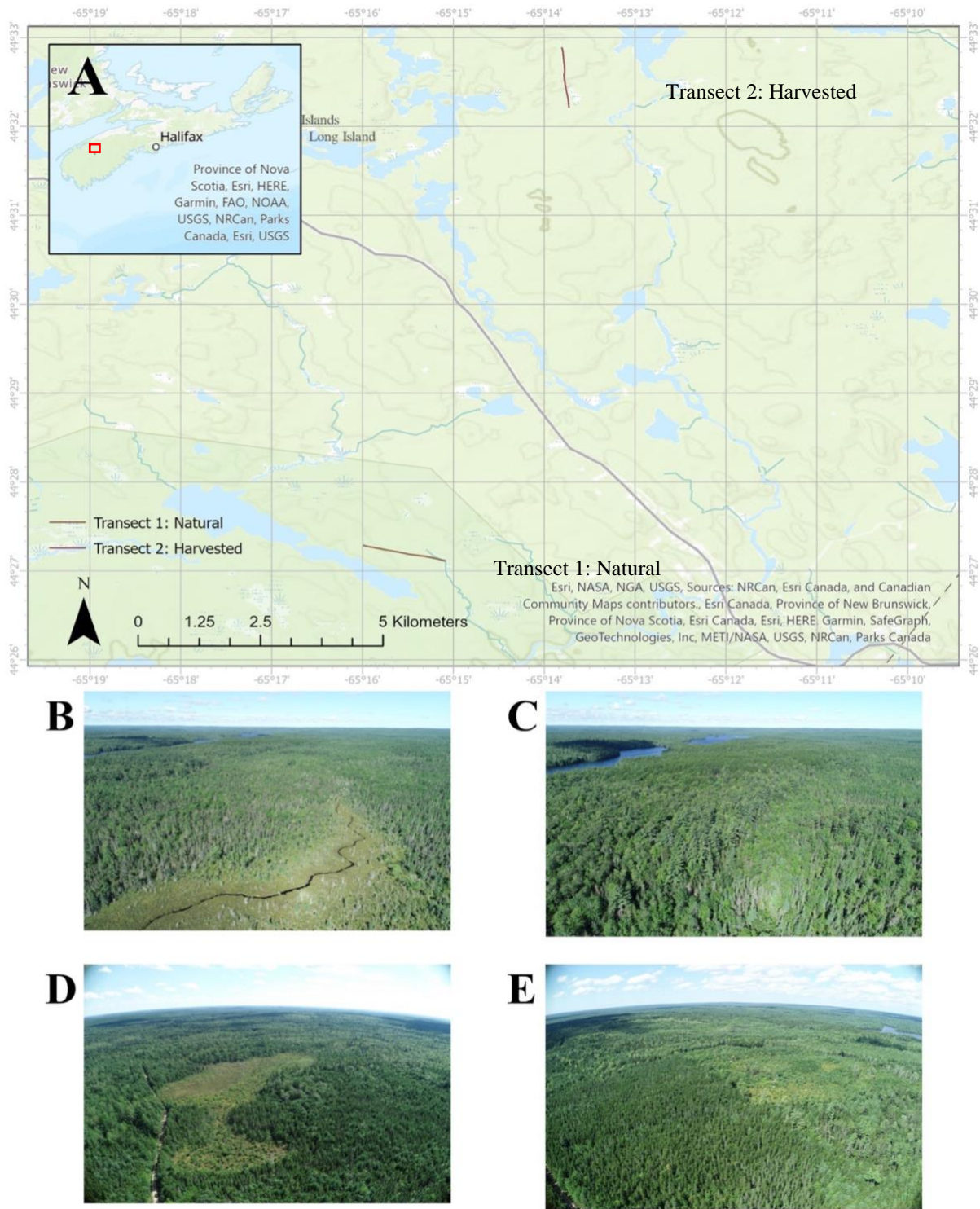


Figure 2-1: Map (A) and aerial photos of (B) Nixon Meadows Brook, (C) the transition between black spruce-dominated forest to old growth hemlock, (D) old growth patches and bog, and (E) a clear-cut stand and surrounding planted spruce forest. Map (A) has an inset of the province of Nova Scotia to reference the general location, and the enlarged area shows the location of each transect. Photos (B) and (C) are of the natural landscape in Kejimikujik National Park, and (D) and (E) are of the anthropogenically disturbed landscape in the Medway Lakes Wilderness Area.

b. Data Collection

Two 1250 m transects were sampled contiguously using 2.5 m by 5 m quadrats. The first transect was set up in a natural landscape in Kejimikujik National Park, and the second in an anthropogenically disturbed harvested landscape in the Medway Lakes Wilderness Area.

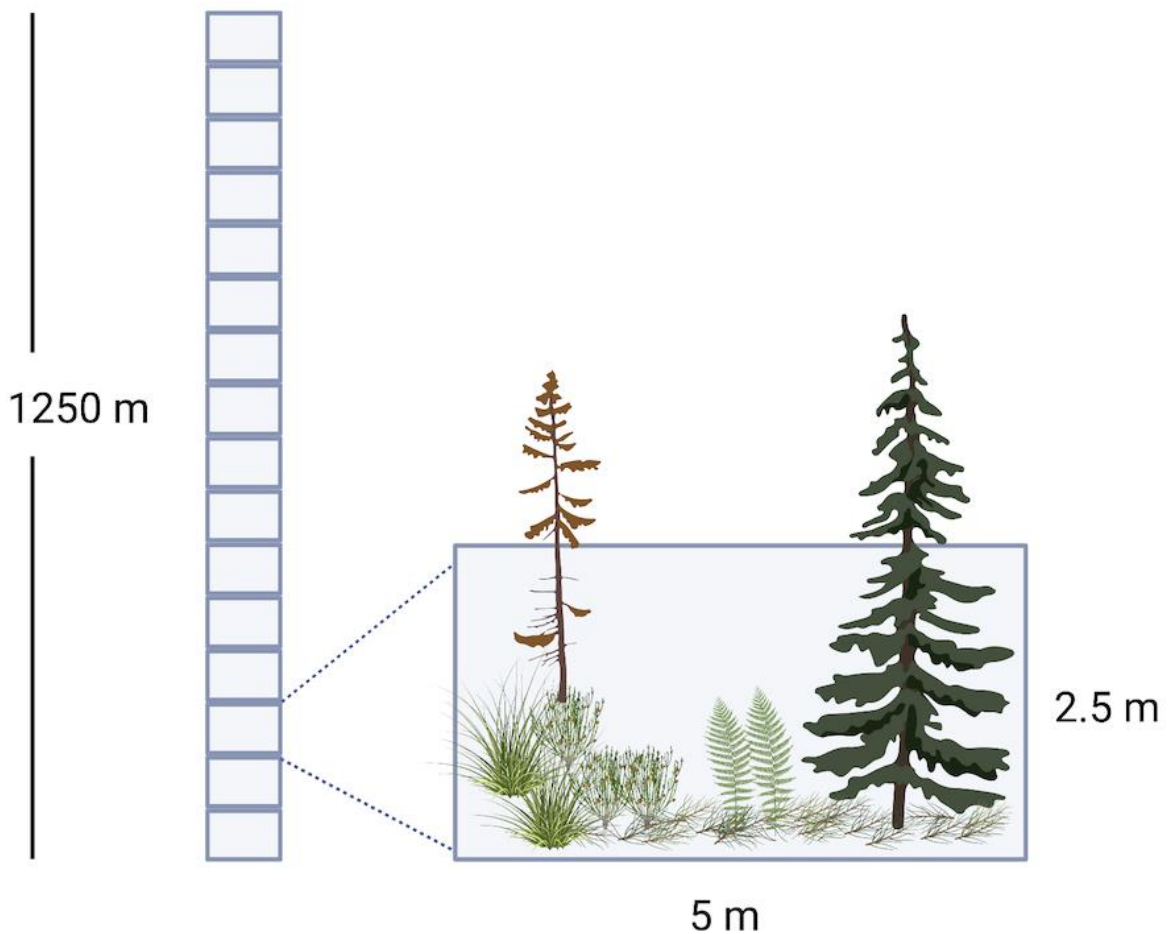


Figure 2-2: Transect sampling design. Quadrats were 2.5 m by 5 m along a 1.25 km (1250 m) transect. Plants within the depicted quadrat represent the varying types of structural diversity sampled, such as functional plant group diversity (i.e. trees, graminoids, ferns, litter) or tree structural diversity (i.e. trees of different heights and decay stages). Figure created with BioRender.com.

Along each 1250 m transect, data to inform two indices of structural diversity were measured by sampling vegetation structural categories and tree structural diversity. I recorded the percent cover of 14 total categories: trees, saplings, shrubs of varying height classes (<1 m, 1-2 m, >2 m), ferns, herbs, graminoids, bryophytes, *Sphagnum*, deadwood, stumps, ground lichen and litter (Table 2-1) to the nearest 10% over 5% and the nearest 1% under 5%. For each tree in the quadrats, I recorded the species, diameter at breast height (DBH), relative height, whether the tree was living or dead, and decay stage for all dead trees for (Table 2-2). Relative height categories were assigned as: suppressed — any tree below half the height of the canopy; intermediate — any tree between half of the canopy and under the canopy; co-dominant — any tree as tall as the majority of the canopy; and dominant — any tree above the height of the canopy. Canopy cover was estimated by averaging four readings per quadrat using a spherical crown densiometer in the field. Additional data, such as elevation, were sourced from the Nova Scotia Elevation Explorer, an ESRI-based web platform with provincial light detection and ranging (LiDAR) tiles for the Digital Elevation Model (DEM) at a 1 meter resolution.

Table 2-1: Structural categories of functional plant groups and description.

Structural Categories	Description
Trees	Woody plants with a DBH above 5 cm
Saplings	Woody plants with a DBH below 5 cm
Shrubs <1	Woody vegetation less than 1 m in height
Shrubs 1-2	Woody vegetation between 1-2 m in height
Shrubs >2	Woody vegetation greater than 2 m in height
Ferns	Flowerless plants, reproducing by spore
Herbs	Small, non-woody vegetation, excluding ferns, graminoids and bryophytes
Graminoids	Grasses, sedges, and rushes
Bryophytes	Non-vascular plants, mosses, liverworts and hornworts (excluding <i>Sphagnum</i> spp.)
<i>Sphagnum</i>	Type of bryophyte, indicative of wetlands and moister habitats
Stumps	The remaining portion of tree still in the ground (under 1.4 m in height)
Deadwood	Deadwood or branches with a diameter greater than 1 cm
Ground lichen	Any species of lichen growing from the forest floor
Litter	Leaf litter or deadwood less than 1 cm in diameter
Bare ground*	Exposed soil without litter, mud, or water
Rock*	Exposed rock

*Data collected but not used in the analysis

Table 2-2: Tree structural diversity categories.

Category	Relative height	Diameter (cm)	Decay stage
Live trees			
LS1	Suppressed	5-10	
LS2	Suppressed	10-20	
LS4	Suppressed	30-40	
LI1	Intermediate	5-10	
LI2	Intermediate	10-20	
LI3	Intermediate	20-30	
LI4	Intermediate	30-40	
LI5	Intermediate	40-50	
LI7	Intermediate	60-70	
LC1	Co-Dominant	5-10	
LC2	Co-Dominant	10-20	
LC3	Co-Dominant	20-30	
LC4	Co-Dominant	30-40	
LC5	Co-Dominant	40-50	
LC6	Co-Dominant	50-60	
LC7	Co-Dominant	60-70	
LC8	Co-Dominant	70+	
LD1	Dominant	5-10	
LD2	Dominant	10-20	
LD3	Dominant	20-30	
LD4	Dominant	30-40	
LD5	Dominant	40-50	
LD6	Dominant	50-60	
LD7	Dominant	60-70	
LD8	Dominant	70+	
Snags			
RSS1	Suppressed, Intermediate	5-10	1
RSS2	Suppressed, Intermediate	10-20	1
RSS3	Suppressed, Intermediate	20-30	1
RSS5	Suppressed, Intermediate	40-50	1
RSS6	Suppressed, Intermediate	50-60	1
RTS1	Co-Dominant, Dominant	5-10	1
RTS2	Co-Dominant, Dominant	10-20	1
RTS3	Co-Dominant, Dominant	20-30	1
ISS1	Suppressed, Intermediate	5-10	2
ISS2	Suppressed, Intermediate	10-20	2
ISS3	Suppressed, Intermediate	20-30	2
ISS4	Suppressed, Intermediate	30-40	2
ITS2	Co-Dominant, Dominant	10-20	2
ITS3	Co-Dominant, Dominant	20-30	2
OSS1	Suppressed, Intermediate	5-10	3-4
OSS2	Suppressed, Intermediate	10-20	3-4
OSS3	Suppressed, Intermediate	20-30	3-4
OSS4	Suppressed, Intermediate	30-40	3-4
OTS2	Co-Dominant, Dominant	10-20	3-4
OTS3	Co-Dominant, Dominant	20-30	3-4
OTS4	Co-Dominant, Dominant	30-40	3-4

*Abbreviations for live trees stand for living, relative height, and DBH range (e.g., LS1 = Living suppressed 5-10 cm DBH). For snags, the abbreviations R, I and O stand for recent, intermediate and old in reference to how long the trees have been dead (representing decay stage), and the second letter represents short or tall (combining suppressed and intermediate relative heights for short = S and co-dominant and dominant relative heights for tall = T), the third letter is S for snag and numerical values are assigned as for live trees based on DBH value.

c. Data Analysis

To quantify structural diversity for all functional plant groups and tree structural groups in each quadrat, I used the Shannon Diversity Index or the Shannon-Wiener Index, defined as,

$$H' = - \sum_i p_i \log_b p_i$$

where p_i is the proportional abundance of structural component i and b equals two and is the base of the logarithm, using the R *vegan* package (Oksanen et al., 2019) in the R environment (R Core Team, 2022). The Shannon Diversity Index is commonly used for quantifying species diversity in a particular area or community. However, I used each structural category as a pseudo-species when calculating a value of structural group diversity per quadrat for both the functional plant group diversity and tree structural diversity metrics. For functional plant group diversity, there were 14 pseudo-species (Table 2-1) and 46 for tree structural diversity (Table 2-2).

I used wavelet analysis to assess the spatial pattern of functional diversity and structural diversity on the landscape to determine where boundaries are present. Wavelet analysis consists of moving a template along the transect and assesses the similarity between the template and the data at each position along the transect at several scales by increasing the size of the template (Dale & Mah, 1998). The template represents the shape of a spatial structure. For example, the Mexican Hat template can detect the pattern of patches, whereas the Haar template can determine transitions across the landscape (Bradshaw & Spies, 1992;

Dale & Mah, 1998). The Haar position variance was used to determine if abrupt changes in structural diversity occurred at edges. A high wavelet transform indicates that the template matches the data and that the spatial structure is present (Dale & Mah, 1998), in this case, the location of abrupt transitions. Wavelet analysis was performed using the software PASSaGE (Rosenberg & Anderson, 2011). To identify significant transitions, I used the position variance of the Haar wavelet with 999 permutations, a maximum scale of 10% or 125 m and a 90% confidence interval. The position variance refers to summing the squared continuous wavelet transform across scales at each position and subsequently dividing them by the number of scales, allowing for the detection of abrupt change in spatial pattern at certain positions (Dale & Mah, 1998). The 90% confidence interval was chosen over the 95% confidence interval, given the exploratory nature of this analysis to capture more transitions that might be missed when using a smaller confidence interval. All variations of trials tested to determine the most representative metric to detect edges are in Appendix 2-1 & Appendix 2-2. I defined significant transitions as when the position variance exceeded that of the 90% confidence interval for at least two consecutive quadrats or where there were two or more quadrats in a row; any single outliers that fell within one quadrat of the significant transition were also included. These significant transitions were compared at edges in each environment and edge type.

Other variables explored in this study include changes in elevation, canopy cover, stand density, basal area, and tree species diversity. Data were averaged per quadrat using ArcGIS Pro's Zonal Statistics as a Table geoprocessing tool for elevation and in the field for canopy cover. The averaged values were graphed along with the location of significant transitions to compare changes in the variables to locations of significant transitions. The

additional variables of basal area and stand density were calculated using Excel. Basal area was calculated from the measured DBH of each tree based on the formula for the area of a circle and then converted into meters squared per hectare. Stand density was counted as the number of trees per quadrat. Tree species diversity was calculated the same way as the structural diversity indices using the Shannon diversity index in R (R Core Team, 2022)

3. Results

a. Functional Plant Group Diversity

Functional plant diversity showed more significant transitions in harvested areas than in natural areas. When using a 90% confidence interval, there were only two transitions in the natural landscape (Figure 2-3); compared to a 95% confidence interval, there were no significant transitions (Appendix 2-1). There were visual changes in structural diversity from the wetland to the spruce-dominant forest and again from the spruce-dominant forest transition into old-growth hemlock forest. However, none of these edges were significant boundaries with this analysis.

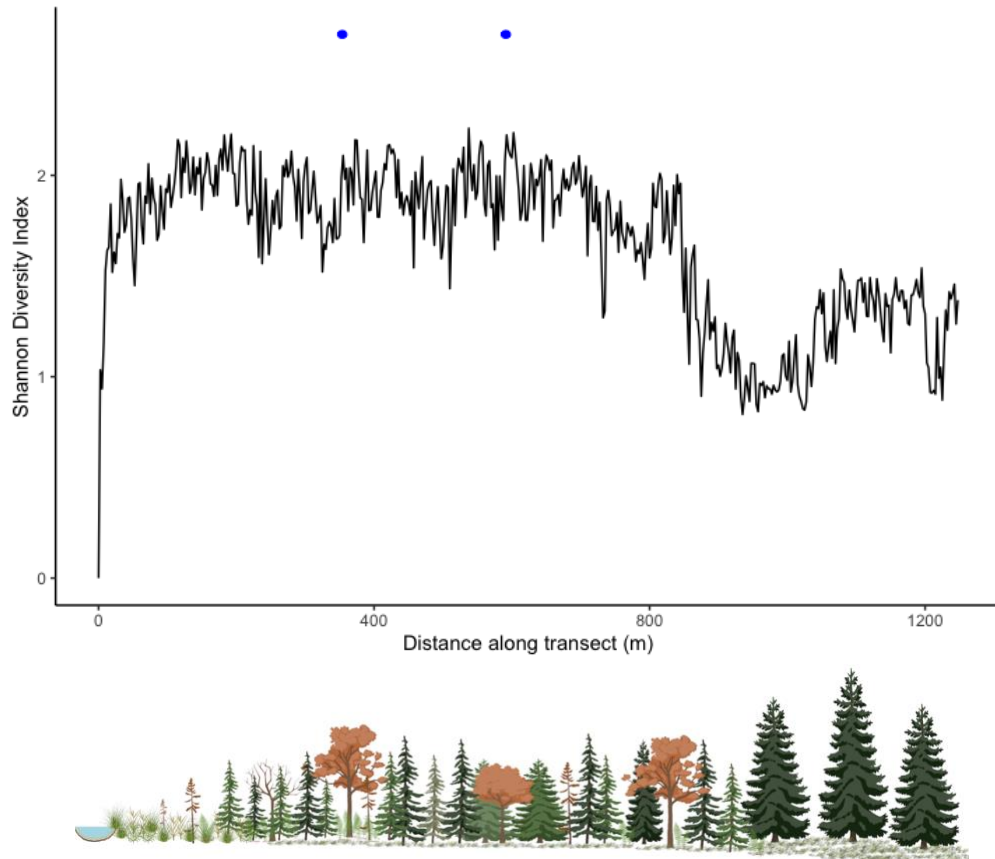


Figure 2-3: Functional plant diversity per quadrat and the location of significant transitions along the natural transect. Blue circles represent the significant transitions on the landscape. The black line represents the Shannon diversity value per quadrat. The graphic represents the type of forest present on the landscape to visually determine edges. Graphic created with BioRender.com

In the harvested landscape, when using a 90% confidence interval, there were eight significant transitions (Figure 2-4). Similar results were also seen using a 95% confidence interval (Appendix 2-2). Four of these transitions occurred at anthropogenic edges, the transition from the mature forest into the clear-cut, the transition from the clear-cut into the planted spruce forest and on both sides of the road. The other four significant transitions were located within the clear-cut, likely due to the regeneration of the stand. However, there was no significant transition present at the natural wetland edge.

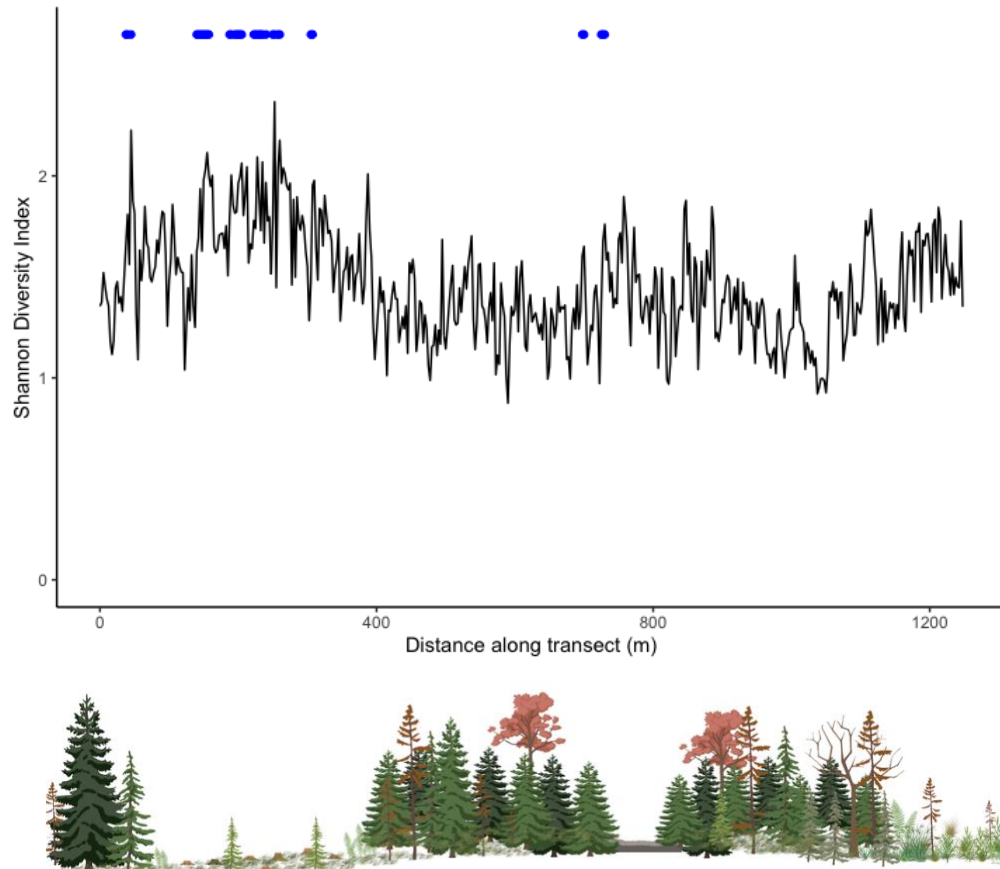


Figure 2-4: Functional plant diversity per quadrat and the location of significant transitions along the harvested transect. The blue circles represent the significant transitions on the landscape. The black line represents the Shannon diversity value per quadrat. The graphic represents the type of forest present on the landscape to visually determine edges. Graphic created with BioRender.com.

b. Tree Structural Diversity

Tree structural diversity had nine significant transitions across the natural landscape using a 90% confidence interval (Figure 2-5). Significant transitions occurred at visually determined natural edges, such as next to the wetland and the old-growth forest. Additional significant transitions occurred within the natural black spruce-dominated forest.

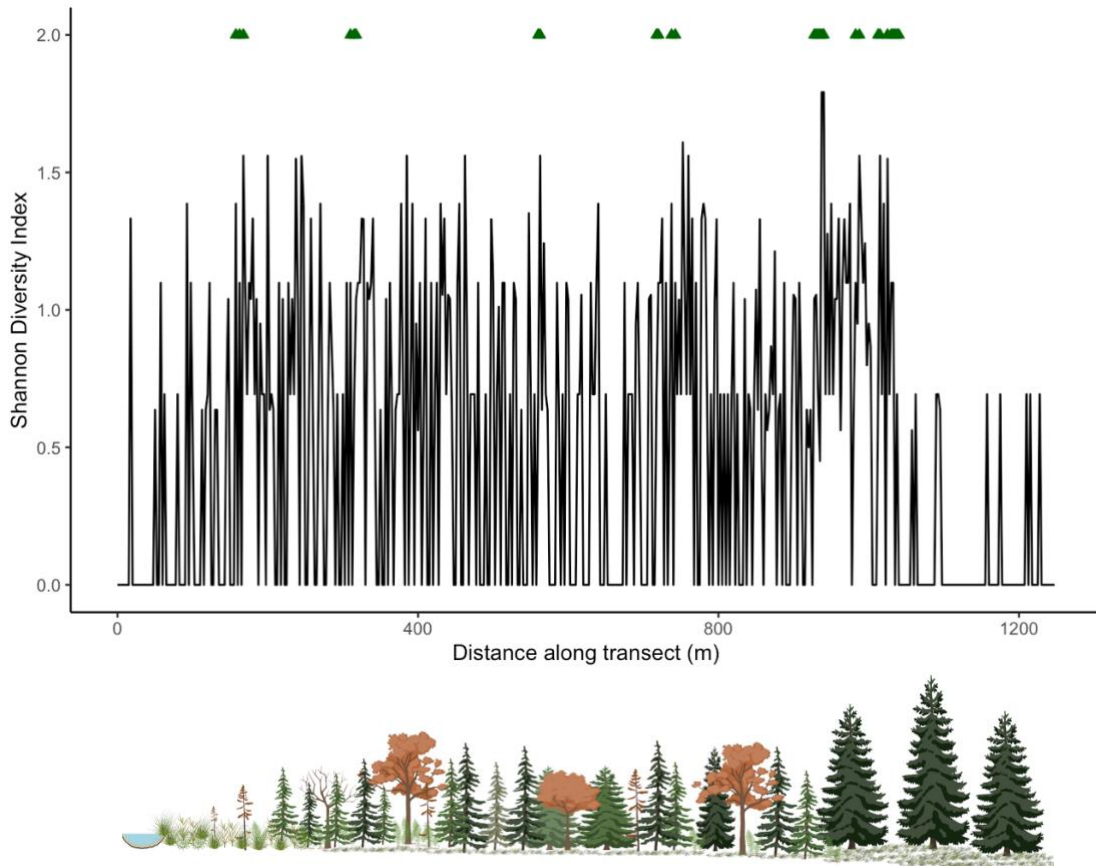


Figure 2-5: Tree structural diversity per quadrat and the location of significant transitions along the natural transect. The green triangles represent the significant transitions on the landscape. The black line represents the Shannon diversity value per quadrat. The graphic represents the type of forest present on the landscape to visually determine edges. Graphic created with BioRender.com

In the harvested landscape, there were a total of 11 significant transitions (Figure 2-6). Significant transitions occurred at the four anthropogenic edges, on either side of the clear-cut and either side of the road, but also occurred at natural edges, the remaining patch of natural older growth forest and into the wetland. Additional significant transitions occurred in the planted red spruce forest but not within the clear-cut as they did for functional plant diversity.

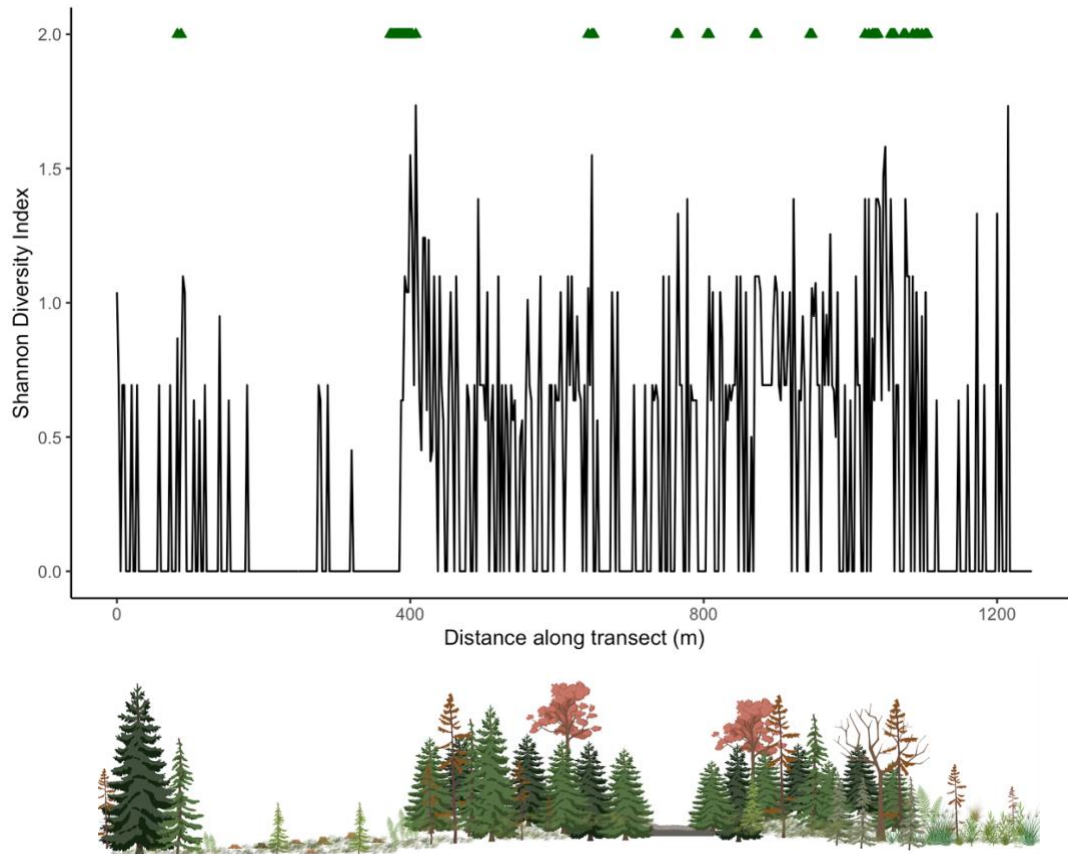


Figure 2-6: Tree structural diversity per quadrat and the location of significant transitions along the harvested transect. The green triangles represent the significant transitions on the landscape. The black line represents the Shannon diversity value per quadrat. The graphic represents the type of forest on the landscape to visually determine edges. Graphic created with BioRender.com.

c. Explanatory Variables

In the natural landscape, significant transitions did not align with the changes in elevation, canopy cover, stand density, basal area, or tree species diversity. Many of the related tree variables, such as stand density, basal area and tree species diversity, had significant transitions at areas with average values in these variables (Figure 2-7: C, D&E).

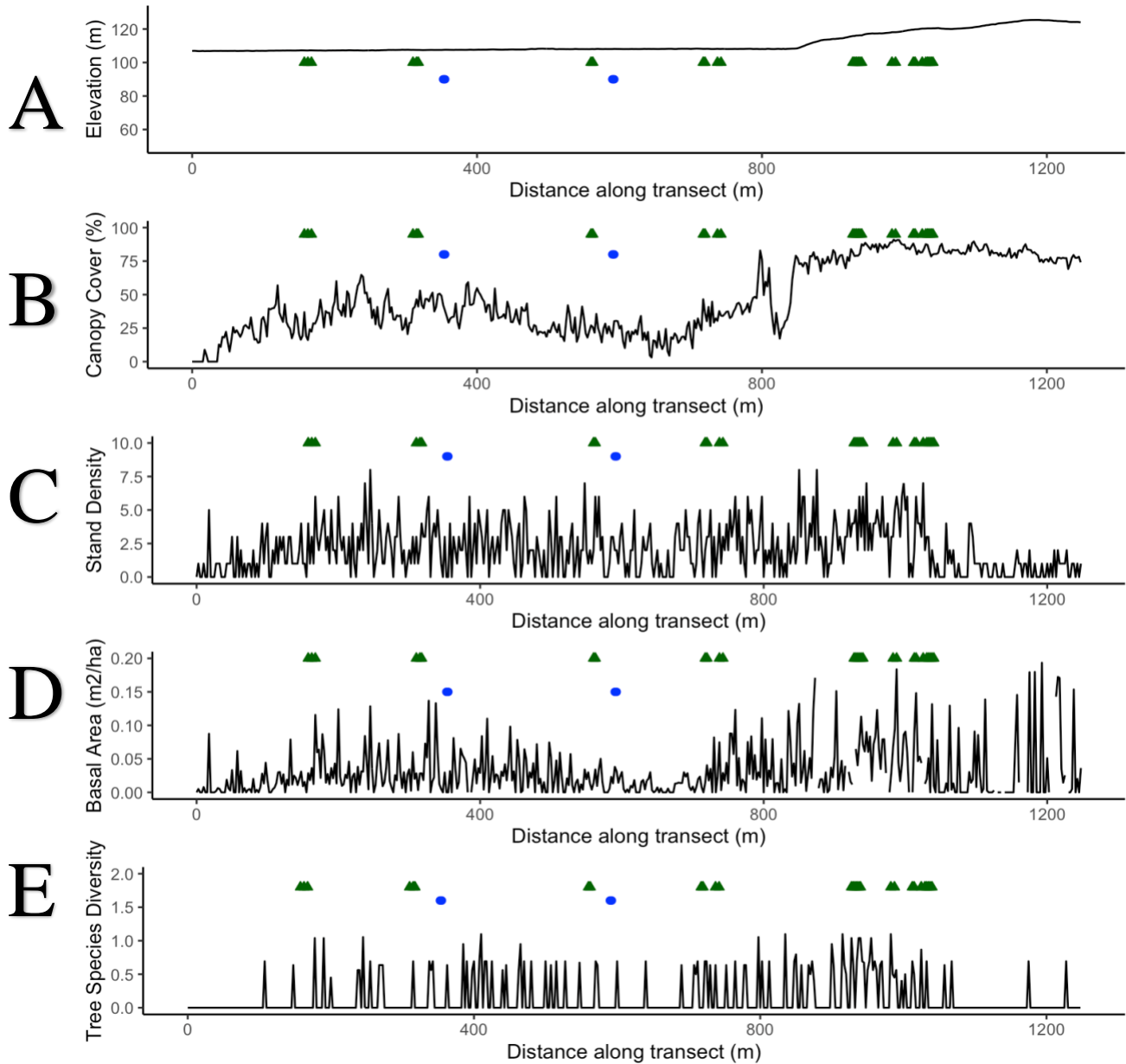


Figure 2-7: Location of significant transitions in functional plant diversity (blue circles) and tree structural diversity (green triangles) in the natural landscape compared with A) average elevation, B) average canopy cover, C) stand density, D) basal area, and E) tree species diversity per quadrat.

Significant transitions in the harvested landscape did not occur at elevation, stand density, basal area, or tree species diversity changes. However, where there was low canopy cover, there was a high proportional change in functional plant diversity, with most of the significant transitions found where there was low canopy cover (Figure 2-8: B). The opposite is seen with

tree structural diversity. Where there was a high canopy cover, there was little change in tree structural diversity.

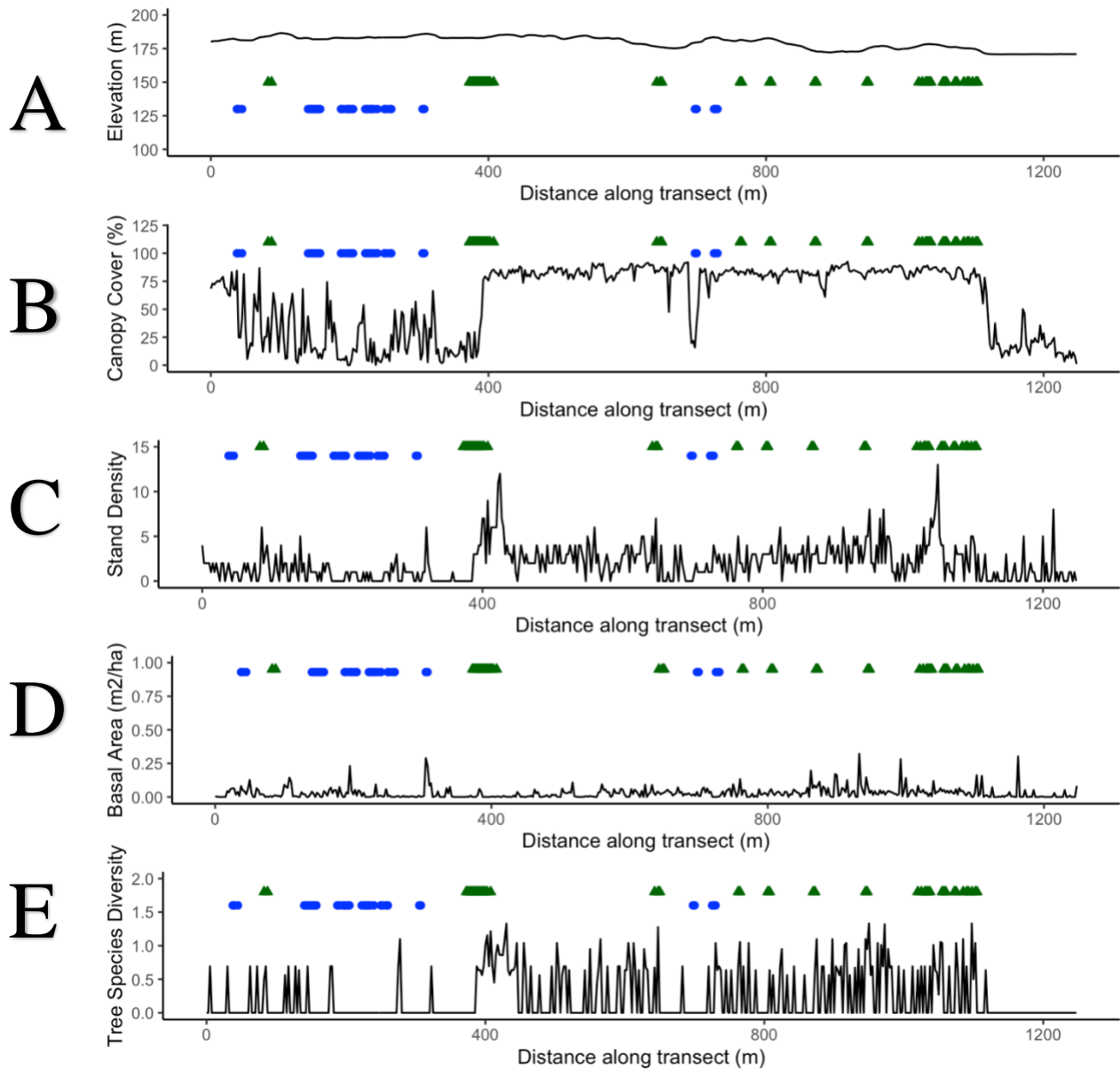


Figure 2-8: Location of significant transitions in functional plant diversity (blue circles) and tree structural diversity (green triangles) in the harvested landscape compared with A) average elevation, B) average canopy cover, C) stand density, D) basal area, and E) tree species diversity per quadrat.

4. Discussion

Significant transitions occurred for functional plant group diversity at anthropogenic edges but not at natural edges, and the natural landscape had fewer significant transitions than the anthropogenically disturbed landscape. When using tree structural diversity, significant transitions occurred at all edge types. Additional significant transitions were detected and occurred throughout planted and natural spruce dominant stands. Significant transitions were difficult to clearly determine at forest edges for both indices of structural diversity. The significant transitions did not align with peaks in other variables except for canopy cover in the anthropogenically disturbed landscape.

I hypothesized that structural diversity should peak at edges, given that it has elements from either side of the patch. However, when examining edges across the landscape, peaks in diversity were seen at all edges in tree structural diversity but not functional plant group diversity. Similar results were seen at forested wetland edges in the Atlantic Canadian Acadian forests (Harper et al., 2021). In forest types such as the Acadian Forest, there is lower diversity with many generalist and well-adapted understory species throughout the upland forest and edges as opposed to other landscape types, which house edge-specific vegetation (Erdős et al., 2019).

There were a variety of additional significant transitions throughout the natural spruce-dominant forest. These transitions could be due to the structure of the forest as spruce forests are variable, with frequent gaps in the canopy, leading to more microclimatic variation as the heterogeneity of understory vegetation is linked to gaps in the canopy (Bradshaw & Spies, 1992). Given the scale of the analysis using 125 m windows for comparison or a scale of 10% of the transect, the variation in the canopy can

create smaller significant transitions that are not easily distinguished as natural edges, which could account for some of additionally detected boundaries found within spruce dominant stands.

Significant transitions occurred at anthropogenic edges and were more frequent in anthropogenically altered landscapes in functional plant and tree structural diversity. The structure of anthropogenic edges, such as recent clear cuts, logging roads or linear disturbances such as power lines, are abrupt in structure (Baker et al., 2013; Eldegard et al., 2015; Harper et al., 2004). The logging road edges are more similar in structure to a power line as its vegetation is limited to an early successional stage, which will have less regrowth than a clear-cut edges and change in structure over time (Eldegard et al., 2015). Similarly, linear changes or disturbances such as roads can have effects along the landscape with regards to plant diversity when using a wavelet transform (Brosofske et al., 1999). Given the abrupt overall change in vegetation structure created by an anthropogenic edge, it is well suited to using the Haar wavelet to determine a significant transition. However, after analysis, it was determined that in the natural landscape, there were less frequent significant transitions and many natural edges in functional plant group diversity did not yield significant transitions. Significant transitions were seen at the anthropogenic edges for functional plant group diversity on the harvested landscape but not at the natural edges. Thus, determining transitions in functional plant group diversity in natural landscapes is more difficult due to the more gradual structure of the vegetation. The change in vegetation is visible but not obvious or as abrupt in structure for functional plant group diversity as for tree structural diversity, for which both edge types were detected using this analysis.

When examining the structural diversity of trees, there were significant transitions at both anthropogenic edges and natural edges in both harvested and natural landscapes. Thus, an index of tree structural diversity detects more edges in general when using this analysis than compared to functional plant group diversity. The additional transitions not at edges were detected across both landscapes in spruce-dominant forests, natural and planted. These additional significant transitions did not align with changes in elevation, canopy cover, stand density, basal area, or tree species diversity.

Neither index of structural diversity detected significant boundaries at elevation changes. This lack of change supports previous findings in the subarctic tundra that elevation at this scale does not impact structural diversity (Dodonov & Harper, 2022). The elevation change was only 18.6 meters for transect one and 15.8 meters for transect two over 1250 m. It would likely need to be a larger gradient to impact vegetation structural diversity.

The result in the harvested landscape supports previous findings in harvested landscapes as plant diversity decreases with canopy closure after logging or harvesting as light does not reach the ground (Bricca et al., 2023). As seen in my results, where there is low canopy cover, there was a high proportional change in functional plant diversity, primarily in the areas of regeneration in the more recent clear-cut. Conversely, where there was high canopy cover, there were fewer changes in functional plant diversity as the diversity in the understory decreased with the increase in canopy cover. This supports other studies in anthropogenically altered landscapes where the landscape is managed to support high canopy cover (Brosofske et al., 2001). This result was not seen in the natural

landscape, as changes in structural diversity did not correspond with changes in canopy cover.

While stand density is shown to be a driver of understory vegetation, it is linked to soil nutrient properties which were not included in the scope of this study (Zhang et al., 2021). In undisturbed forests, intermediate-density stands show the highest availability of nutrients and light. In low-density stands, there was limited litter to increase soil diversity and high competition with low light availability to the understory in high-density stands (Zhang et al., 2021). Significant structural diversity transitions were primarily located in areas with average stand density and not high or low stand density.

Basal area is an alternative indicator of stand density and is often used in forestry for harvesting decision-making (Kimsey et al., 2019). It is often positively correlated with canopy cover and did not provide a substantial connection with either metric of structural diversity. However, there was a slightly higher basal area at some significant transitions in tree structural diversity, but as these are related metrics, it does not provide proof of correlation between the two interconnected variables.

Species diversity is often linked to productivity. However, recent studies in temperate forests show that structural diversity may better indicate forest productivity than tree species diversity (LaRue et al., 2019; Park et al., 2019). I compared the significant transitions in plant functional diversity and tree structural diversity to the changes in tree species diversity. While tree species diversity can be linked to tree structural diversity, it did not show the same pattern. Overall, tree species diversity was relatively low across both transects, given the proportionate homogeneity when compared to the possible

structural categories, which far exceeds the number of tree species possible in the Acadian Forest.

As many explanatory variables are correlated, they do not provide the ability to explain the pattern and processes on the landscape. However, other studies have found diversity patterns linked to canopy (Bricca et al., 2023; Brosofske et al., 2001) or soil properties (Zhang et al., 2021; Kimsey et al., 2019), which were not examined in this study. Additionally, given the gradual structure of natural vegetation boundaries, it is difficult to distinguish natural edges using wavelet analysis with a metric of functional plant diversity. Other analyses to determine natural boundaries using gradient analysis by detrended correspondence analysis (DCA) and the moving split-window (MSW) technique also did not identify boundaries at the hypothesized visual boundaries from aerial photos (Choesin & Boerner, 2002). There was more success when using tree structural diversity to determine vegetation boundaries at edges across landscape types within this study.

These results add to the understanding of forest edges in Atlantic Canada and in the Acadian Forest, which can contribute to identifying and understanding critical blocks of natural and anthropogenic habitat. This knowledge also provides information for action by various land conservancies and government managers at the regional and provincial levels with regards to suitable habitats for multiple species at risk in the province such as the Canada Warbler (*Cardellina canadensis*) and Boreal Felt Lichen (*Erioderma pedicellatum*). However, our ability to conduct a large study at this larger-than-normal scale was limited due to the monetary and temporal cost of ground sampling using extended transects to view landscape changes. As such, there is a limited number of

edges and landscape types. Further studies could build upon this base and explore alternative methods for sampling structural diversity, such as drone imagery.

5. References

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Appendix 2-1: Transect 1 - Natural Landscape

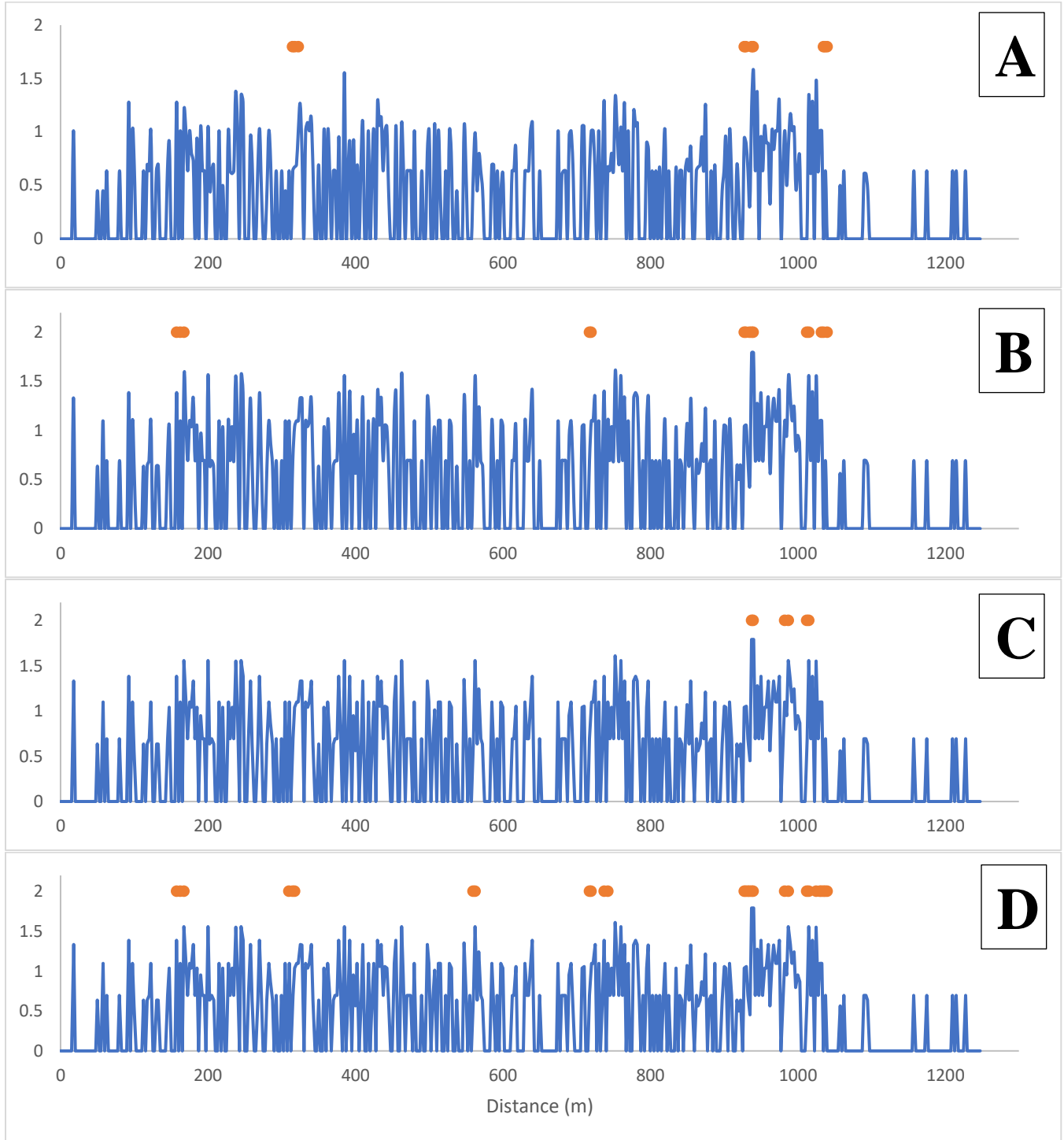


Figure 2-9: Location of significant transitions (orange circles) in tree structural diversity (blue lines); A) without DBH classes, B) with DBH classes, C) with DBH and a scale of 1% or 10 m, D) with DBH classes and a 90% confidence interval. If not specified, the scale is 10% or 125 m, the confidence interval is 95%, and all iterations use the Haar position wavelet.

Appendix 2-2: Transect 2 - Harvested Landscape

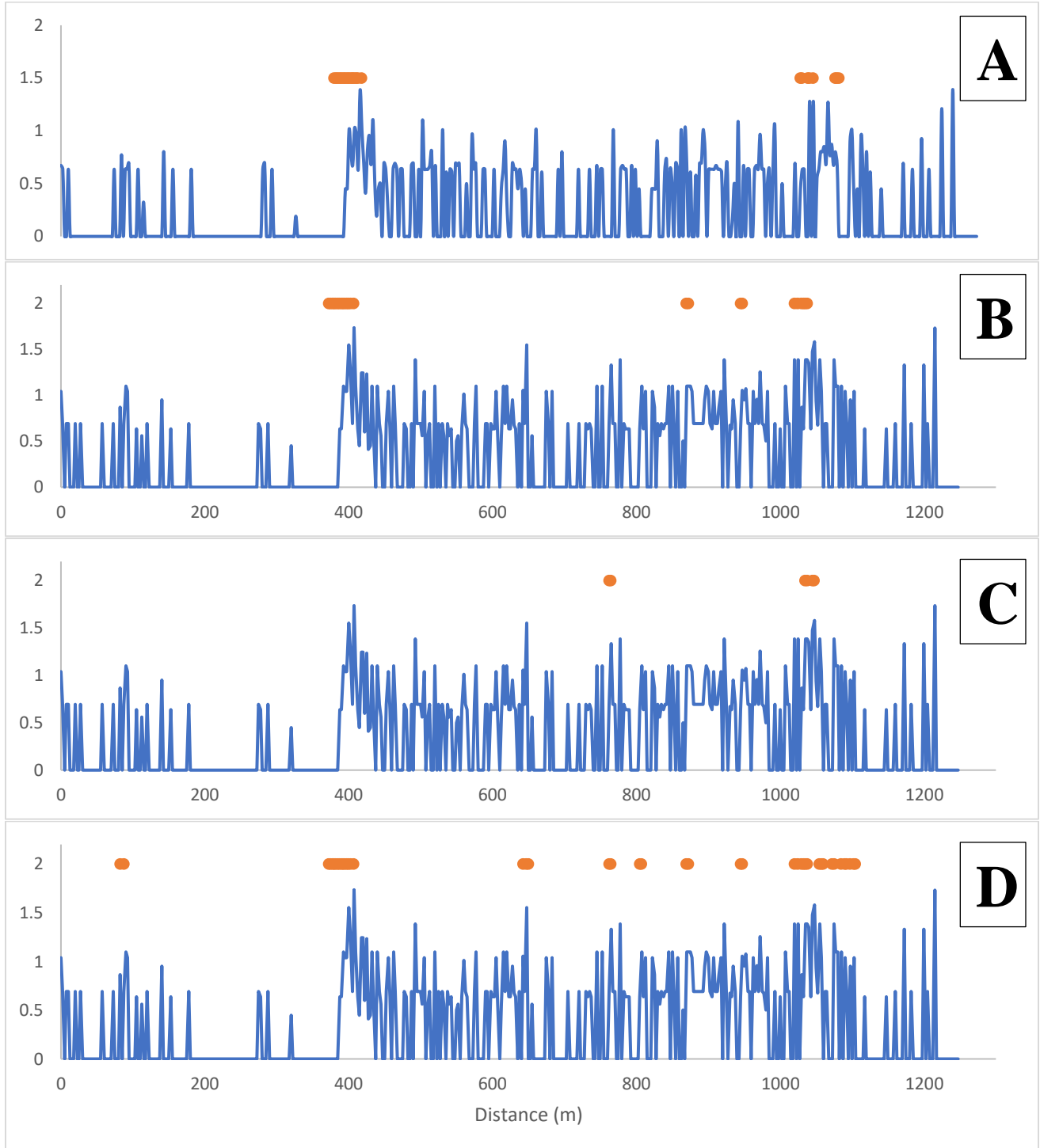


Figure 2-10: Location of significant transitions (orange circles) in tree structural diversity (blue lines); A) without DBH classes, B) with DBH classes, C) with DBH and a scale of 1% or 10 m, D) with DBH classes and a 90% confidence interval. If not specified, the scale is 10% or 125 m, the confidence interval is 95%, and all iterations use the Haar position wavelet.

Chapter 3: Can drone imagery replace field sampling when examining patterns of vegetation structural diversity in forested landscapes?

Abstract

Structural diversity can be used as an alternative metric to species diversity and can be examined using a variety of methods, from field sampling to aerial imagery. My objectives were 1) to describe the pattern of structural diversity across transitions in a forested landscape with drone imagery, 2) to compare indices of structural diversity obtained from drone imagery, and 3) to compare indices between drone imagery and field measures. The photogrammetrically derived imagery was sampled to estimate vegetation height differences as vertical heterogeneity or canopy complexity and overall diversity using the densified point cloud. The three metrics of structural diversity using photogrammetrically derived imagery were 1) the variation in canopy height per quadrat using standardized height data, 2) raw height data and 3) the concentration of points per quadrat from the densified point cloud. The georeferenced aerial photography was collected using a DJI Phantom 4 RTK model drone with an RGB camera and was processed using Pix4Dmapper. Significant transitions were not obvious at forest edges for any of the indices of structural diversity using drone imagery. When comparing the three indices of structural diversity using drone imagery, the Digital Surface Model (DSM) and the Canopy Height Model (CHM) were most similar. However, the points per meter squared showed the most success in detecting edges. The results from the imagery was compared with previously sampled 2.5 x 5 m quadrats along two 1250 m transects in harvested and natural landscapes in southwestern Nova Scotia and showed that metrics of drone imagery were more analogous to tree structural diversity than to functional plant group diversity. Overall, the most successful metric in detecting edges was tree structural diversity.

1. Introduction

Vegetation structural diversity can be measured using field-based metrics such as stand structure or remote sensing such as light detection and ranging (LiDAR) or aerial photography to measure canopy complexity (LaRue et al., 2019). Some studies examine structural diversity at the stand level using the number of tree species of different sizes (Godlee et al., 2021; Staudhammer & LeMay, 2001). Other field-based measures include the abundance and distribution of vegetation structural components such as shrubs, herbs and trees, for example, to determine patterns of structural diversity between forested wetlands and upland forests (Harper et al., 2021) or along a lakeshore to forest gradient (Dazé Querry & Harper, 2017).

Most of these studies employ a quadrat and transect method for data collection on structural diversity; they are usually placed at a single edge (Dazé Querry & Harper, 2017) and utilize short-length transects, as discussed in previous chapters. However, this is costly in both time and money. Technological advancement supporting aerial photography introduces the possibility of assessing spatial heterogeneity across the landscape (Turner & Gardner, 2015). With the ability to use and apply remote sensing principles, the study of landscapes on a large scale allows for not only an abundance of widely available data such as from LiDAR (LaRue et al., 2020; Ozdemir & Donoghue, 2013; Zellweger et al., 2013) but also study on finer scales than typically utilized in landscape ecology, such as with unmanned aerial vehicles (UAV) or drone imagery (Jensen & Mathews, 2016; Li et al., 2017). The primary applications of UAV remote sensing for plant ecologists are for vegetation mapping, monitoring and measuring, ranging from the individual level to the landscape scale (Koh & Wich, 2012; Sun et al., 2021).

The original use of image-based point clouds was to supplement forest monitoring and inventory produced from airborne laser scanning (ALS) LiDAR data (White et al., 2013). High-

resolution images with a multi-image overlap are needed to generate image-based point clouds or a digital terrain model (DTM) (White et al., 2013; Jensen & Mathews, 2016). Canopy height and density can be retrieved from image-based point clouds to quantify vegetation structure. Thus, point clouds can be used as an alternative quantification of vegetation structural diversity across long transects. My objectives were to 1) describe the pattern of structural diversity across transitions in forested landscapes with drone imagery, 2) compare indices of structural diversity from drone imagery, and 3) compare indices between drone imagery and field measures. I aimed to compare the number of distinct boundaries for both harvested and natural landscapes using different indices of structural diversity derived from image-based point clouds, a canopy height model, and field sampling of vegetation structural elements.

2. Methodology

a. Study Area and Site Selection

The study area is in southwestern Nova Scotia within the province's western ecoregion (Figure 3-1). This ecoregion is part of the Acadian Forest and is comprised primarily of spruce hemlock or mixed wood forests (Nova Scotia Department of Natural Resources, 2017). This region is mild in temperature, with nearby Bridgewater's average daily temperature ranging from 5 °C in January to 19 °C in July (Environment and Climate Change Canada, 2023) and is rich in forested areas (Nova Scotia Department of Natural Resources, 2017) including old-growth hemlock forests, forested wetlands, and spruce-dominant forests.

I selected two sites using Google Earth imagery and the Nova Scotia Provincial Landscape Viewer, to locate a 1250 m transect in each site, as discussed in previous chapters. Each site was chosen to maximize the number of visually determined transitions across the landscape along the transects. However, as this study requires drones, each site needed

appropriate areas for take-off and landing. For site one, flights were done from the wetland and windthrow stand at the ends of the transect. For site two, flights were done from the logging road that intersects the middle of the transect.

Transect one, the natural transect, was in the province's largest protected area, Kejimikujik National Park, and crossed a natural, undisturbed landscape (Figure 3-2). This transect went from Nixon Meadows Brook through the wetland surrounding the brook, through a multi-age forest and into an old-growth hemlock forest. Transect two, the harvested transect, was in the Medway Lakes Wilderness Area and was in an anthropogenically disturbed landscape (Figure 3-3). This location was impacted by commercial harvesting, planting, pre-commercial thinning, and herbicide use. This transect started in a clear-cut near Frog Lake and went through a multi-age spruce forest, crossing a former logging road and into a bog.

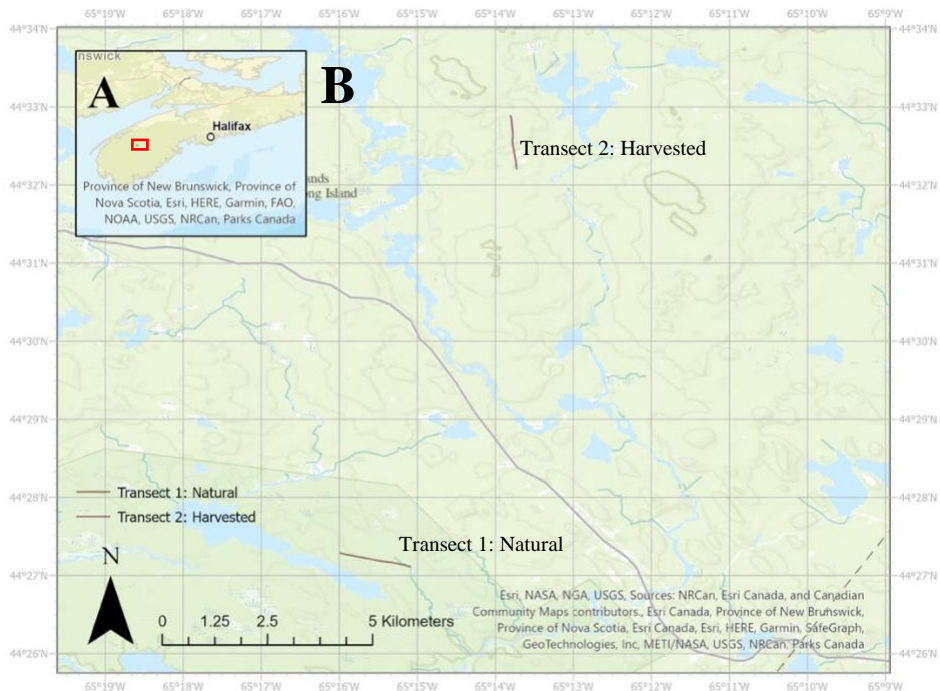


Figure 3-1: Maps of the study sites in southwestern Nova Scotia: A) overview showing the location of the study area within the province and B) the location of the two transects.

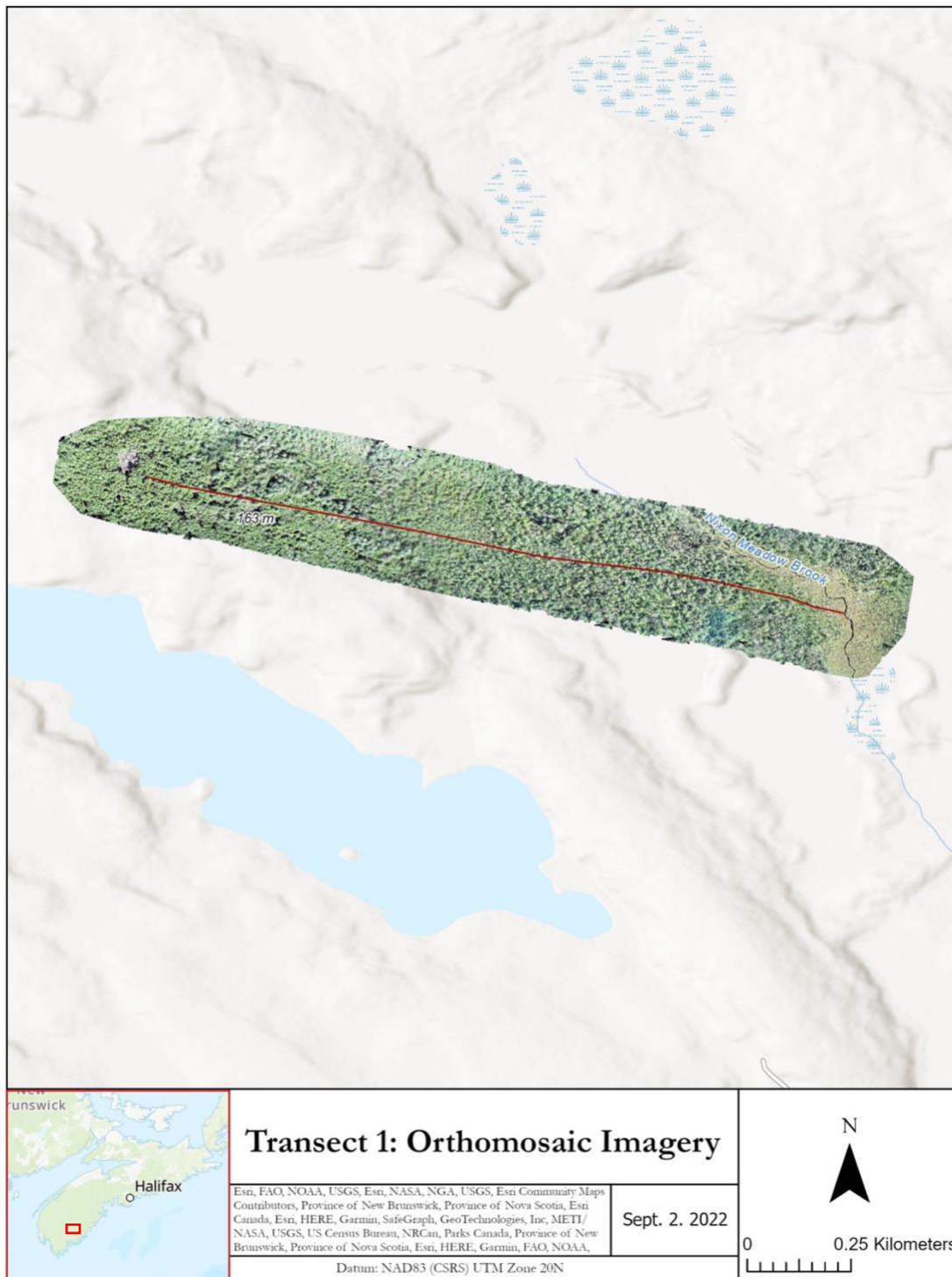


Figure 3-2: Map with orthomosaic imagery of transect one across the natural landscape in Kejimikujik National Park. This transect starts in Nixon Meadow Brook and ends in the old growth hemlock forest (left to right) to match the original ground sampled quadrats.



Figure 3-3: Map with orthomosaic imagery of transect two, the harvested landscape in the Medway Lakes Wilderness Area. This transect starts in just outside the clear-cut and runs into the bog (top to bottom) to match the original ground sampled quadrats.

b. Geospatial Data Collection and Processing

Georeferenced aerial photography was collected on Sept. 1 and 2, 2022, using a DJI Phantom 4 RTK with an RGB camera. The aerial survey was conducted 90-100 m above ground level. Ground control points (GCPs) were placed and collected using a Leica Geosystems GC14 dual-frequency GNSS receiver that uses RTK to improve the accuracy of the collected imagery during processing. This type of receiver is a survey-grade positioning unit that provides real-time corrections to up to low centimetre accuracy based on the number of available satellites. For transect one, the natural transect, there were 6 GCPs; for transect, the harvested transect two, there were 7 GCPs.

The collected aerial imagery was processed in the Maritime Provinces Spatial Analysis Research Centre using Pix4Dmapper, a photogrammetry software for drone imagery. I used its three structured steps: 1) The initial processing extracted key points from each image, subsequently matching them between overlapping images and entering the GCPs. 2) The point cloud densification took the original sparse point cloud to a densified final point cloud used for analysis. 3) Final outputs were produced from the digital surface model (DSM) and orthomosaic imagery.

c. Data Analysis

Data analysis was completed using ArcGIS Pro 2.9.0. GPS waypoints were logged in the centre of each quadrat in the field using a handheld Garmin 64SX model GPS. These waypoints were imported into Garmin Basecamp, the desktop program associated with the device, and logged as a GPX file. To create a transect in ArcGIS Pro, the GPX file was imported and converted to a feature class using the GPX to features geoprocessing tool to

create a new polyline. This shapefile was populated with a new line from the imported points. With a 5 m margin of error, I decided to use a line of best fit but edit the line's vertices by modifying the feature. As the waypoints were collected at the centre of each quadrat, I needed to extend the line by 1.25 m at each end using the continue features tool to have the total 1250 m transect for analysis. From a line of best fit, I created quadrats to match our previous analyses from Chapter 2. Using the divide tool, I divided the line of best fit into 500 sections and created a 2.5 m buffer on either side of the line to represent the previously sampled quadrats.

The digital surface model, DSM, is the *z-value* of the drone imagery, which represents the first return, or the highest point in space recorded by the imagery; this is usually the top of a tree or a building. The DSM was then sampled per quadrat using the Zonal Statistics as Table Geoprocessing tool to gain the standard deviation per quadrat, which was then exported to Microsoft Excel. The DSM is the uncorrected photogrammetrically derived canopy height value (Appendix 3-1 & Appendix 3-2) versus the CHM which is the LiDAR corrected canopy height value (Appendix 3-3 & Appendix 3-4). Similarly, to create the canopy height model (CHM), the digital elevation model (DEM) that was sourced from the Nova Scotia Elevation Explorer, an ESRI-based web platform with provincial LiDAR tiles covering the province, was subtracted from the *z-values* of the DSM, using the raster calculator. The CHM output was then sampled per quadrat using the Zonal Statistics as Table Geoprocessing tool to gain the mean tree height per quadrat and exported to Excel. This process was repeated for each transect.

Other sampling included counting the number of points per meter squared of the densified point cloud. Points per meter squared represent the density of vegetation based on

the number of individual points detected in the image, theorizing where there is higher point density, there is more vegetation structural diversity. Each quadrat was sampled using the LAS point Statistics by Area geoprocessing tool to count the total number of points per quadrat, which was then divided by the total area of the quadrat using the field calculator, as quadrats located at vertices did not have a standardized area to produce the number of points per meter squared. Field sampling metrics, tree structural diversity and functional plant diversity were sourced from Chapter 2 data analysis.

All significant boundaries were determined using wavelet analysis, which consists of moving a template that assesses the similarity between the template and the data at each position along the transect at several scales by increasing the size of the template (Dale & Mah, 1998). As determined in the Chapter 2 methods, I used the position variance of the Haar wavelet with 999 permutations, a maximum scale of 10% or 125 m and a 90% confidence interval. Significant transitions were defined as when the position variance exceeded that of the 90% confidence interval for at least two quadrats in a row or where there were two or more quadrats in a row separated by an insignificant quadrat. This was done for all indices of photogrammetrically derived imagery as well as previously for the field sampling metrics. Wavelet analysis was performed using the software PASSaGE (Rosenberg & Anderson, 2011).

To address objective three, I quantified the co-occurrence of significant boundaries to determine if field sampling metrics from Chapter 2 were comparable to drone metrics of structural diversity. Similar to the use of an overlap statistic to measure the amount of direct spatial overlap in two dimensions used in forested landscapes in southwestern Quebec (Fortin et al., 1996), I used the number of overlapping significant quadrats between metrics in one

dimension to compare indices of structural diversity. All comparisons for overlap statistics were completed using the select duplicates and highlight tool in Excel.

3. Results

Chapter 2 results show, that on the natural landscape, significant boundaries were not generally located at edges, or detected when using functional plant group diversity. On the harvested landscape, functional plant group diversity showed significant boundaries within the harvested area, and tree structural diversity showed significant boundaries at edges and throughout the planted spruce forest.

There were 14 significant transitions along the natural transect using the standard deviation of the DSM z -values as a metric of structural diversity (Figure 3-4:A). They were spaced along the transect, and most were only a few quadrats wide. There was also a total of 14 significant transitions along the harvested transect when using the standard deviation of the DSM z -values as a metric of structural diversity (Figure 3-4:B). These transitions were clustered near edges but were not observed at all edges along the transect.

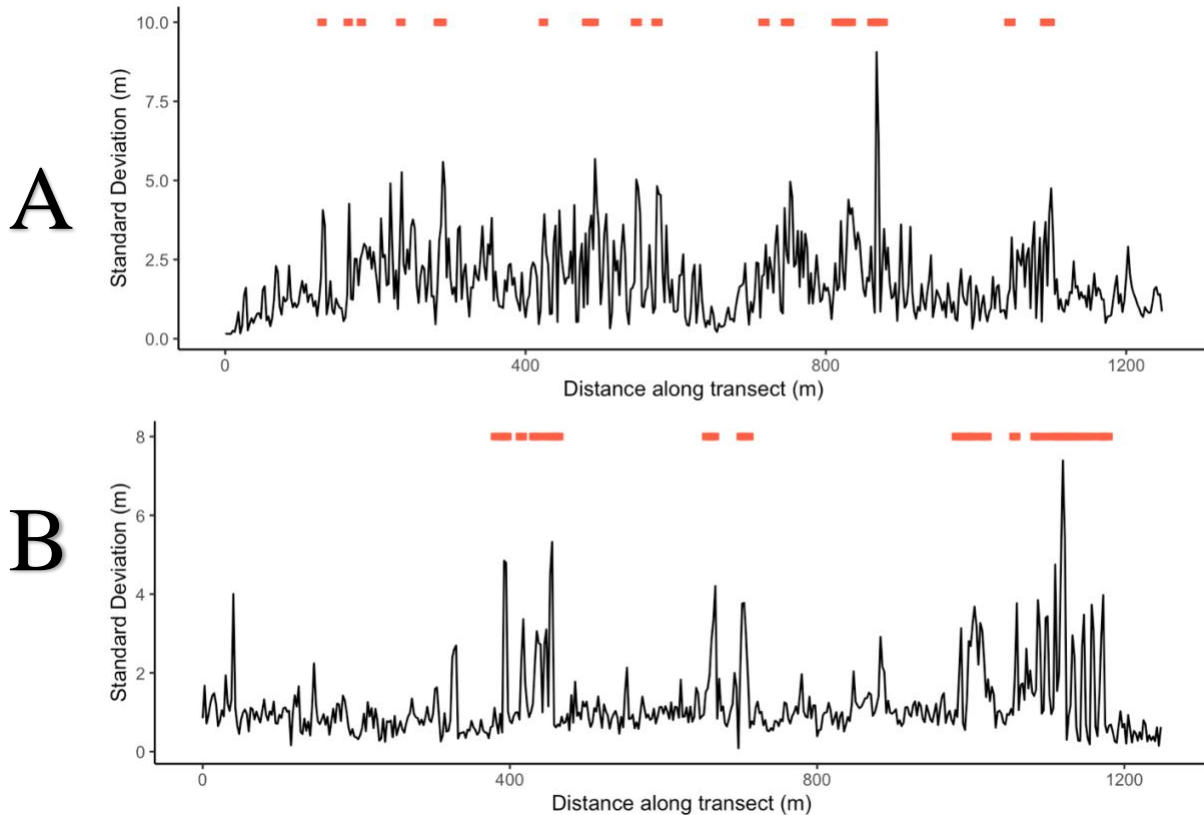


Figure 3-4: The digital surface model (DSM) standard deviation of the z-values, which represents the variation in canopy height per quadrat (solid line), and the location of significant transitions (orange squares) along the natural transect (A) and the harvested transect (B).

There were 10 significant transitions along the natural transect when using the standard deviation per quadrat of the canopy height model as a metric of structural diversity (Figure 3-5:A). The transitions were spaced out along the transect, and most were only a few quadrats in length and not located at edges across the transect. There were 12 significant transitions along the harvested transect when using the standard deviation per quadrat of the canopy height model as a metric of structural diversity (Figure 3-5:B). These transitions were clustered near edges but were not observed at all edges along the transect.

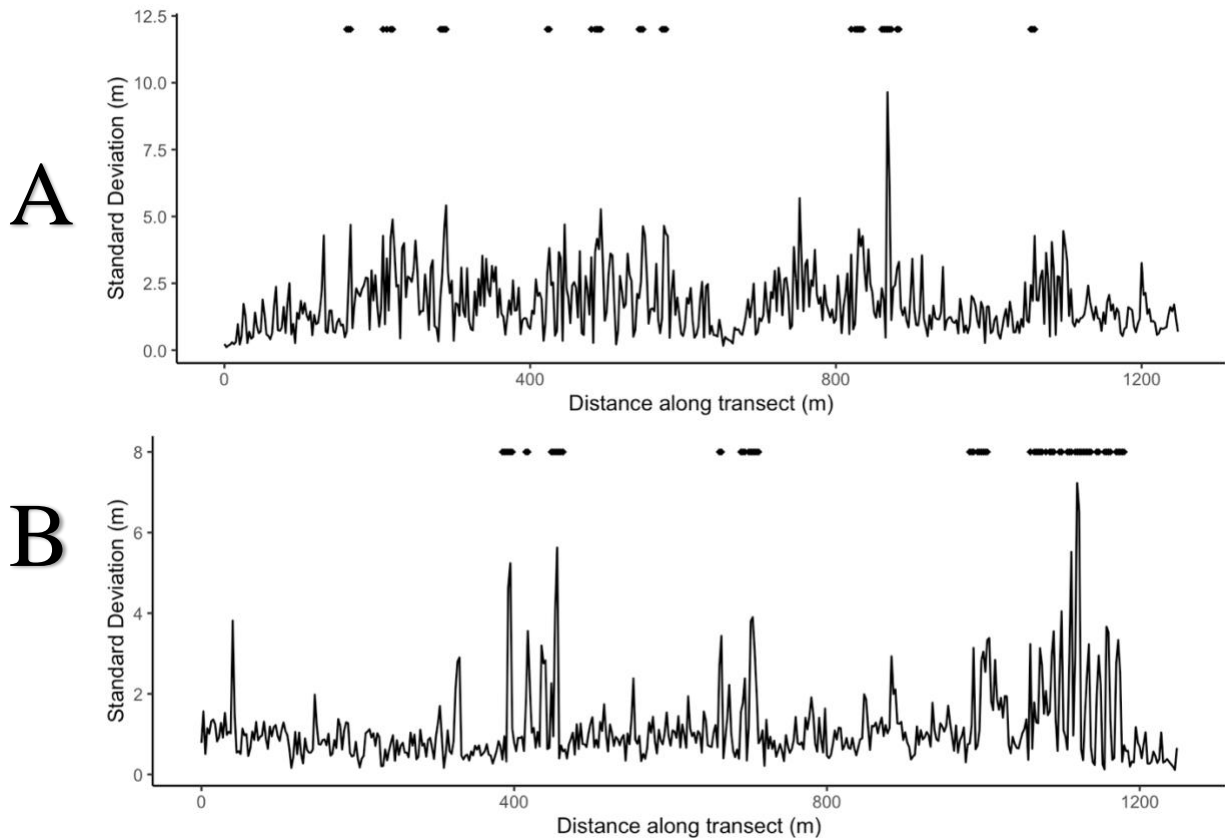


Figure 3-5: The canopy height model (CHM) standard deviation of the standardized z-values, which represents the variation in canopy height in meters per quadrat (solid line), and the location of significant transitions (black diamonds) along the natural transect (A) and the harvested transect (B).

There were 15 significant transitions along the natural transect when using the densified point cloud for each quadrat as a metric of structural diversity (Figure 3-6:A). The transitions were spaced out across the transect, and most were only a few quadrats in length. There were six significant transitions along the harvested transect when using the averaged points per meter squared for each quadrat as a metric of structural diversity (Figure 3-6:B). These transitions were located near edges but only at some edges along the transect. A large transitional zone was present in the centre of the transect.

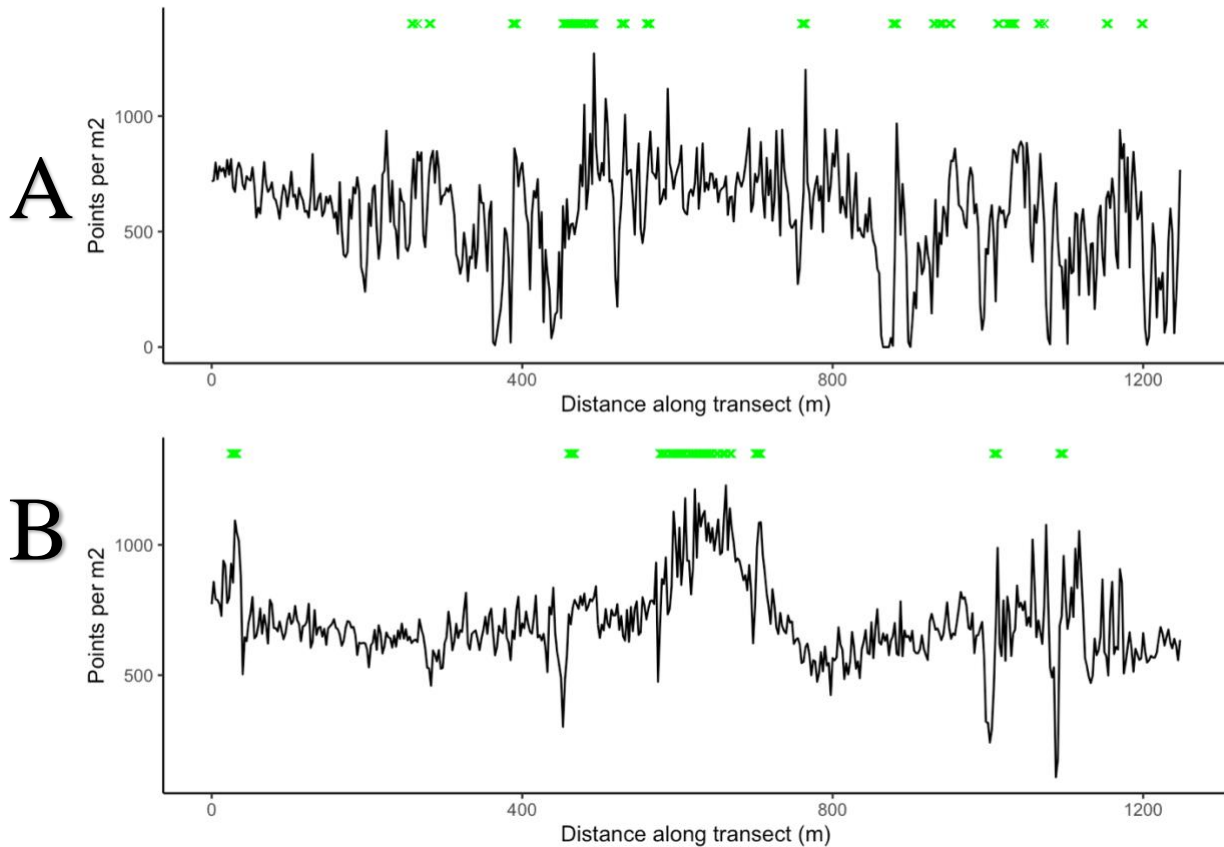


Figure 3-6: Averaged points per meter squared per quadrat (solid line) and the location of significant transitions (light green) along the natural transect (A) and the harvested transect (B).

Significant transitions in functional plant diversity were not located at the same places as the photogrammetrically derived metrics of structural diversity. While all variations of tree structural diversity, CHM, DSM, and the densified point cloud had a similar distribution of significant transitions along the natural transect, many transitions did not overlap at the same location (Figure 3-7). The highest amount of overlap occurred between the DSM and the CHM, as they were the most similar metrics for both the natural and harvested transects (Figure 3-1 & Figure 3-2). The second highest overlap occurred between the points per meter squared and tree structural diversity on the natural transect (Table 3-1), whereas between the DSM and tree structural diversity on the harvested transect (Table 3-2). While tree structural diversity had

significant transitions located primarily at edges and throughout the planted spruce forest, the CHM and DSM had a similar distribution of significant transitions across the harvested transect. In comparison, the densified point cloud significant transitions were distributed at some edges and had a large transition in the centre of the transect. The only significant transition detected by all analysis occurred at around 600 m on the natural transect.

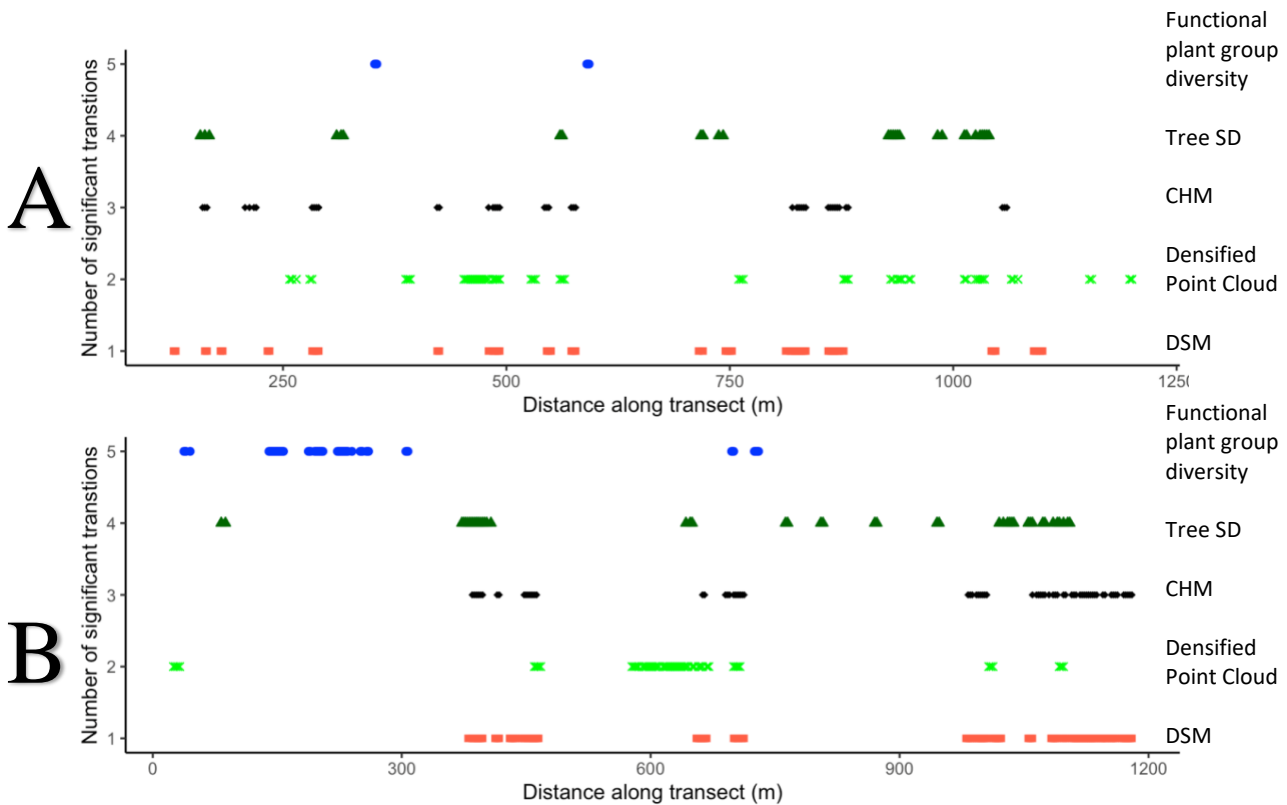


Figure 3-7: Significant transitions of each metric along the natural transect (A) and harvested transect (B). Points represent functional plant group diversity (blue circles), tree structural diversity (green triangles), the canopy height model (black diamonds), densified point cloud (light green crosses), and the DSM (orange squares).

Table 3-1: The overlap statistics for the natural transect. Values represent the number of quadrats that were part of significant transitions between pairs of methods.

Method	DSM	Densified Point Cloud	CHM	Tree Structural Diversity	Functional Plant Diversity
DSM	-	7	30	2	0
Points per m ²	7	-	8	12	0
CHM	30	8	-	1	0
Tree Structural Diversity	2	12	1	-	0
Functional Plant Diversity	0	0	0	0	-

Table 3-2: The overlap statistics for the harvested transect. Values represent the number of quadrats that were part of significant transitions between pairs of methods.

Method	DSM	Densified Point Cloud	CHM	Tree Structural Diversity	Functional Plant Diversity
DSM	-	12	58	14	1
Points per m ²	12	-	8	4	1
CHM	58	8	-	12	1
Tree Structural Diversity	14	4	12	-	0
Functional Plant Diversity	1	1	1	0	-

4. Discussion

Significant boundaries were not located at edges in the natural landscape for the three metrics of structural diversity using photogrammetrically derived imagery (CHM, DSM and densified point cloud). The significant boundaries that were detected throughout the landscape were small and frequent across the landscape. On the harvested landscape, significant boundaries were more clustered and occurred near some edges but were inconsistent between different metrics.

When comparing the three indices of structural diversity using drone imagery, the DSM and CHM were most similar, but the densified point cloud showed the most success in detecting edges. As the DSM is the uncorrected photogrammetrically derived canopy height value and the CHM is the LiDAR corrected canopy height value, the similar pattern of significant boundaries is not surprising. LiDAR-derived canopy height and photogrammetrically derived canopy height have previously been determined to be interchangeable when estimating vegetation canopy height in forested landscapes (Jensen & Mathews, 2016). As such, the DSM may be sufficient, but where available, the LiDAR corrected CHM functions may be a more reliable metric of height as it has been sampled to the exact height based on the landscape topography as opposed to a single height.

When using the densified point cloud as a metric of structural diversity, the transitions were spaced out across the natural transect, and most were only a few quadrats in length. There were six significant transitions across the harvested transect, and most transitions were located near edges. This metric of structural diversity had the most success detecting edges but only in the harvested landscape. Using wavelet analysis for all metrics showed more success in detecting edges on the harvested landscape.

As seen in Chapter 2, there were few significant boundaries in the harvested landscape when looking at functional plant group diversity, while multiple significant boundaries occurred at edges when examining tree structural diversity. Drone imagery metrics had a similar pattern to tree structural diversity. This pattern is expected as drone imagery does not penetrate the canopy to obtain values of structural diversity below; it simply captures the point of first return. This point represents a metric of canopy complexity rather than overall vegetation complexity, as would be described by the functional plant group diversity. The photogrammetrically derived indices present a more robust and informative way to examine canopy complexity and overall vertical complexity as a metric of structural diversity compared to that of ground sampling of trees (Lines et al., 2022), which relies on DBH or relative heights. Thus, given a more accurate representation of tree height and complexity at a low temporal cost, drone imagery may present a better avenue for studies of structural diversity over time or studies focused on metrics directly related to the canopy. This representation could allow the primary applications of UAV remote sensing for plant ecologists to expand from vegetation mapping, monitoring and measuring (Koh & Wich, 2012; Sun et al., 2021) into evaluating research questions such as in this study.

The metrics using drone imagery present a similar pattern to tree structural diversity, but the significant transitions of the three different metrics do not align. The DSM and CHM have clusters of significant boundaries near edges but do not occur at all edges, while the points per meter squared show significant boundaries close to all edges. While the amount of overlap was higher on the anthropogenically disturbed transect than on the natural transect, the significant transitions still do not align perfectly or appear in the same locations along the transect. Tree structural diversity and points per meter squared for the natural transect had the second-highest overlap and a low amount of overlap between the DSM and tree structural diversity on the

natural transect. For the harvested transect, the second highest overlap occurs between tree structural diversity and the DSM. There is a relatively low overlap between points per meter squared and the tree structural diversity on this transect. Thus, results differ between the two landscape types.

As all drone metrics are based on canopy height differences or vertical complexity, they are more comparable with tree structural diversity compared to functional plant group diversity, as drone imagery cannot penetrate the canopy. A more comparable metric for functional plant group diversity may be that of LiDAR. Some examples of using LiDAR include using structural diversity to evaluate landscapes with frequent disturbances regimes to evaluate changes in structural complexity over time to develop management plans (Listopad et al., 2015) and looking at how the variation in canopy gap structure at forest edges influences edge effects in fragmented landscapes (Blanchard et al., 2023).

As determined by the field sampling measures, the gradual structure of natural vegetation boundaries makes it difficult to distinguish natural edges using wavelet analysis with a metric of functional plant group diversity. This trend of difficulty identifying natural boundaries is also seen in previous studies when using gradient analysis by detrended correspondence analysis (DCA) and the moving split-window (MSW) technique with hypothesized visual boundaries from aerial photos (Choesin & Boerner, 2002). Alternatively, studies have examined landscape structure using longer transects and multiple scales (Brososke et al. 1999; Saunders et al., 1998). For example, Saunders et al. (1998) studied the spatial distribution of patches related to temperature and abiotic features using a long transect of 3820 m, while Brososke et al. (1999) examined plant species distribution along a 3575 m transect in the pine barrens landscape of northern Wisconsin, USA. Using these longer transects with contiguous quadrat sampling allows

for the assessment of structural and edge influences across multiple spatial scales. However, it is harder to sample given the extended need for time and higher cost to sample at this length than shorter transects. Thus, finding a comparable metric using remote sensing could allow for more data in shorter timeframes to be able to examine patterns across the landscape. In this study, no metric using drone imagery aligned exactly with field metrics. The most successful metric using the drone-collected data to detect edges was using points per meter squared in the harvested landscape. Future studies should look at testing alternate metrics.

In conclusion, field and aerial sampling can be compared based on the similarity in tree metrics and canopy complexity from aerial sampling. However, functional plant group diversity is not comparable to that of photogrammetrically derived imagery based on its inability to penetrate the canopy; future studies should examine its compatibility for comparison with LiDAR as it can provide a more robust picture as it can reach the ground. There is more success when using tree structural diversity as a field metric overall to determine vegetation boundaries at edges across landscape types. Remote sensing can be applied in plant ecology to fill knowledge gaps when using tree structural diversity and can provide some insights with regards to functional plant group diversity in less studied areas but ultimately field sampling for functional plant group diversity is needed until an appropriate metric of remote sensing can be found.

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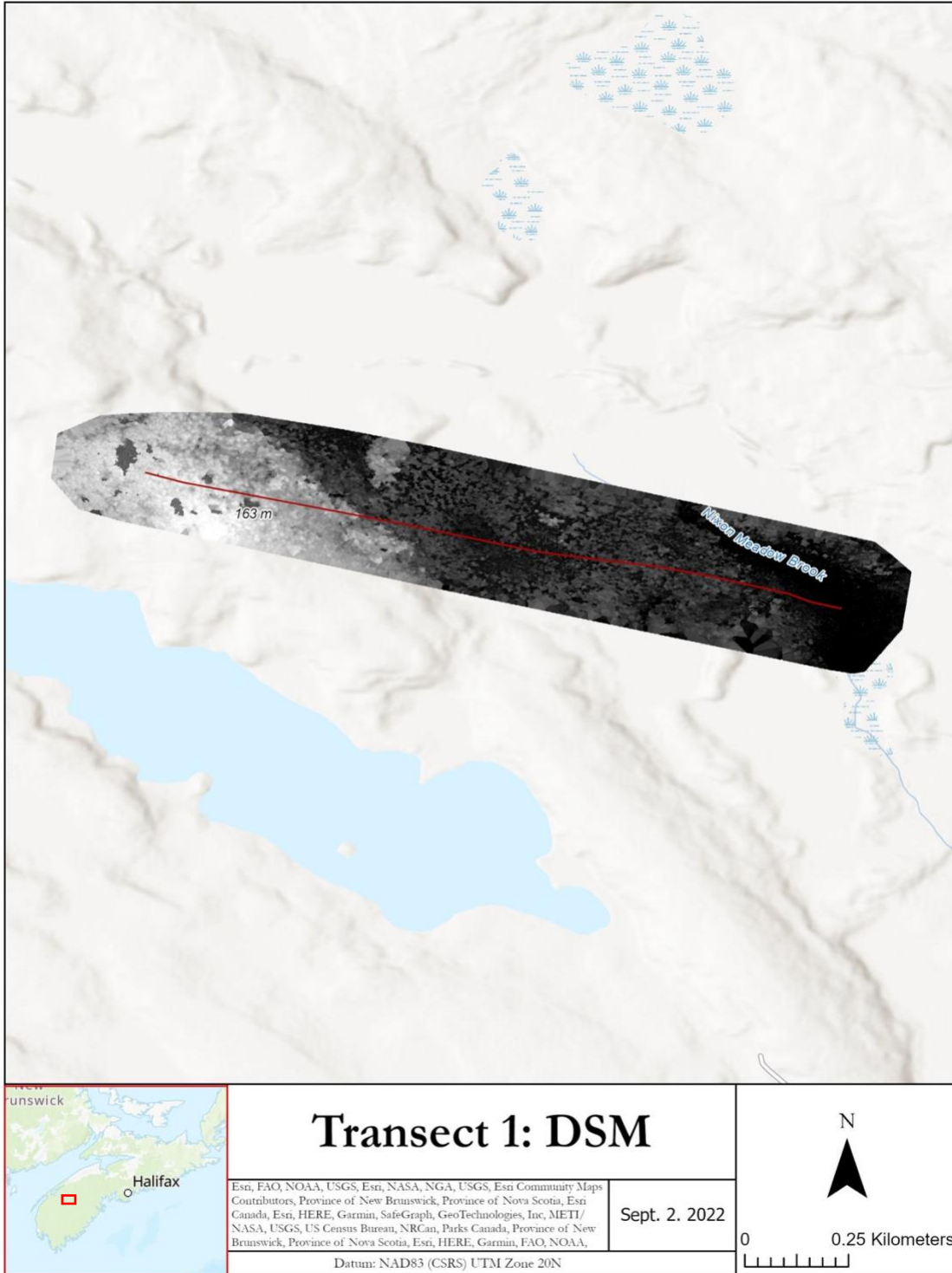
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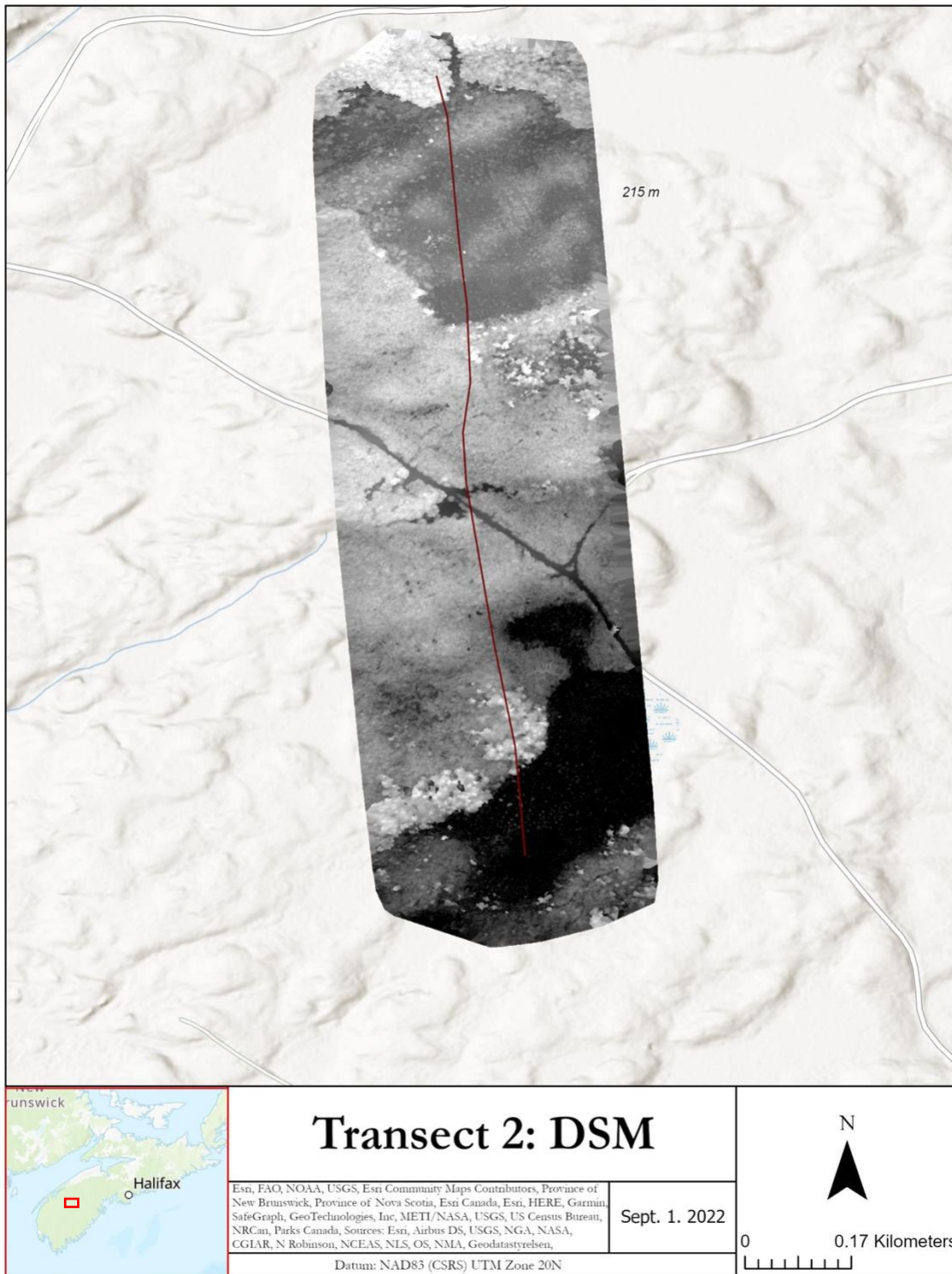
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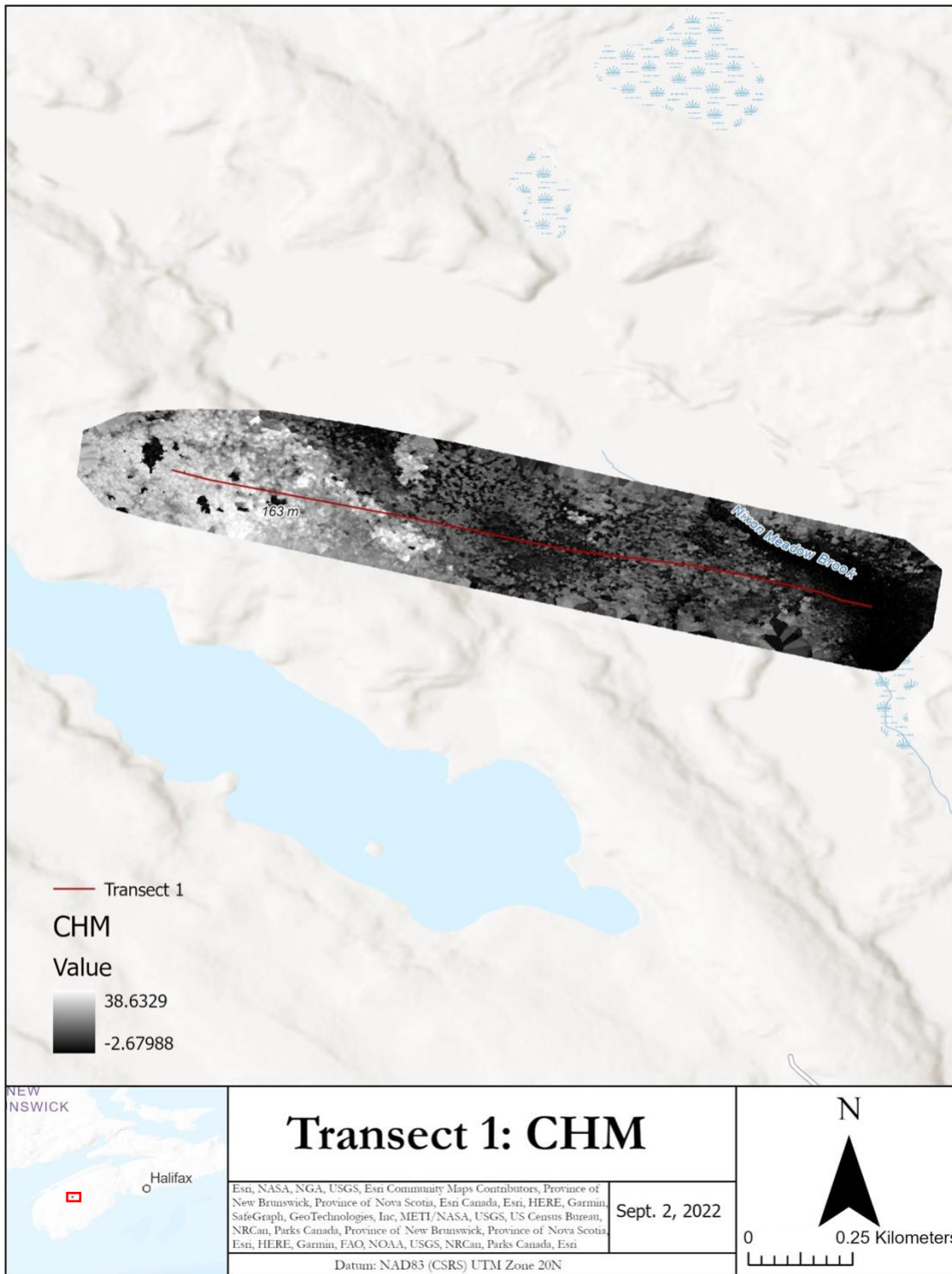
Appendix 3-1: Map with DSM of transect one, the natural landscape in Kejimikujik National Park.



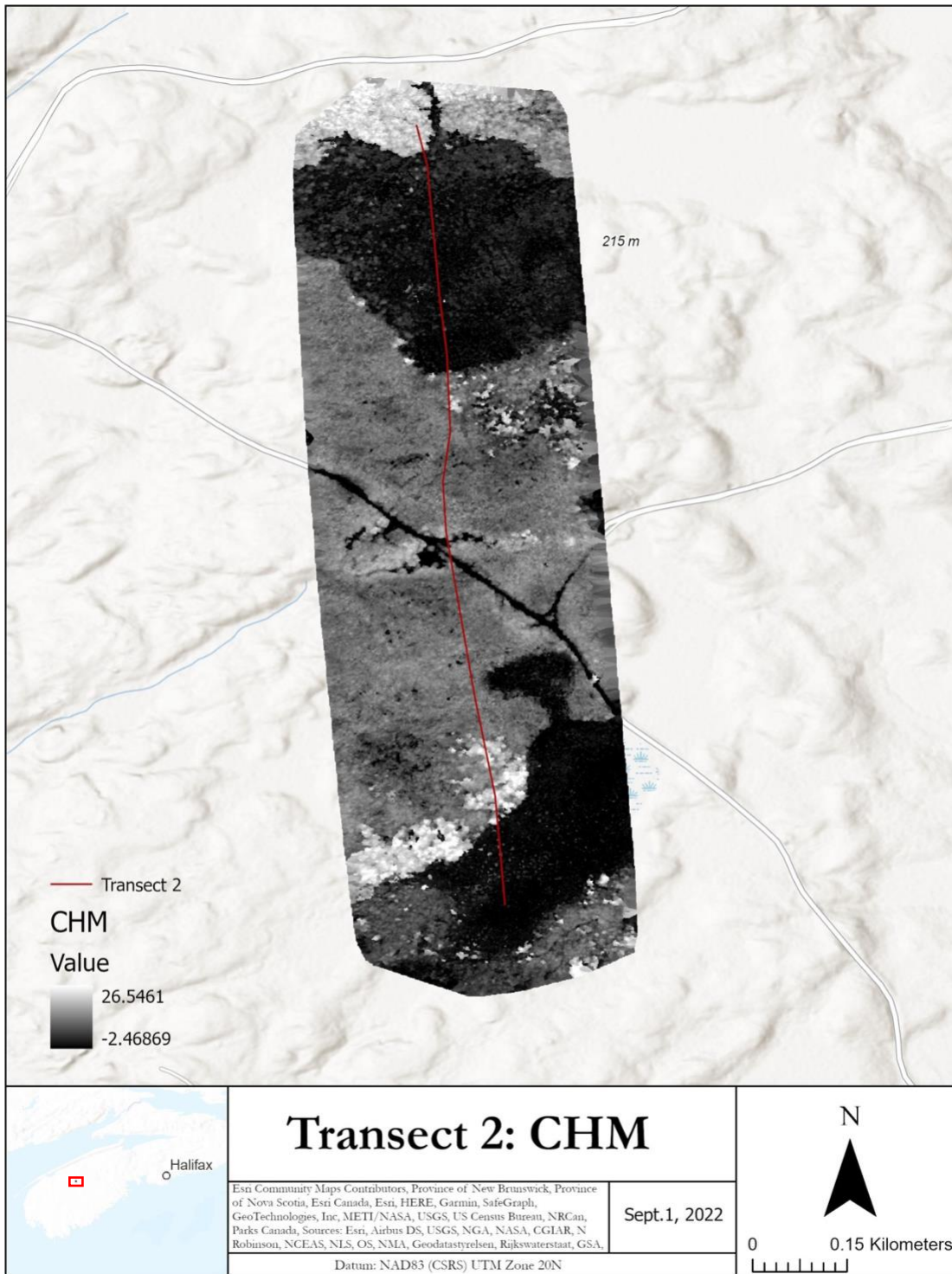
Appendix 3-2: Map with DSM of transect two, the harvested landscape in the Medway Lakes Wilderness Area.



Appendix 3-3: Map with CHM of transect one, the natural landscape in Kejimikujik National Park.



Appendix 3-4: Map with CHM of transect two, the harvested landscape in the Medway Lakes Wilderness Area.



Chapter 4: Conclusion

1. Results Summary

When examining the traditional field metrics for changes in structural diversity, tree structural diversity had more success in determining transitions than functional plant group diversity. Significant transitions occurred using functional plant diversity at anthropogenic edges but not at natural edges, and the natural landscape had fewer significant transitions than the anthropogenically disturbed landscape. Significant transitions occurred at all edge types when using tree structural diversity and were detected throughout planted and natural spruce dominant stands. Significant transitions were not obvious to determine at all forest edges between the two indices of structural diversity. The significant transitions detected by this analysis did not align at peaks in other variables outside of canopy cover in the anthropogenically disturbed landscape.

When examining structural diversity using alternate metrics to field sampling, no metric detected transitions at all edges. Significant boundaries were not located at edges in the natural landscape for the three metrics of structural diversity using photogrammetrically derived imagery. The three metrics were 1) variation in canopy height per quadrat using standardized height data (CHM), 2) raw height data (DSM) and 3) the concentration of points per quadrat from the densified point cloud (points per meter squared). The significant boundaries detected throughout the landscape were small and frequent across the landscape. On the harvested landscape, significant boundaries were more clustered and occurred near some edges but were inconsistent between different metrics. Overall, significant transitions were not obvious to determine at forest edges when comparing the indices of structural diversity between the metrics using photogrammetrically derived imagery and the field-based metrics.

2. Implications and future research avenues

Recent research has identified gaps in forest edge studies across different geographic locations and found that anthropogenic edges were more frequently studied than natural forest edges (Franklin et al., 2021). While in this study, no metric using drone imagery aligned exactly with field metrics, when using points per meter squared in the harvested landscape, there was mild success in detecting edges. When using field metrics, tree structural diversity had significant transitions at all edge types. These metrics of structural diversity could be used in the future to examine predictive habitat for boreal felt lichen (*E. pedicellatum*) by looking for areas with significantly higher density but a lower basal area of living trees and a lower crown closure (Power et al., 2018). Another species at risk for which these metrics of structural diversity could be beneficial in providing information for action is the Canada Warbler (*Cardellina canadensis*) which utilizes areas of higher structural complexity for breeding, such as forested wetlands and young post-harvest forests (Westwood et al., 2020).

Using these longer transects with contiguous quadrat sampling allows for the assessment of structural diversity and edge detection across multiple spatial scales (Brosofske et al., 1999; Saunders et al., 1998) and could be continued with further remote sensing methods. Thus, finding the right metric for comparison using remote sensing is crucial. While tree structural diversity was the most successful metric when compared to the indices using drone imagery, no metric using drone imagery aligned exactly with field metrics. Future studies should examine functional plant group diversities compatibility for comparison with LiDAR as it can provide a more robust point cloud by reaching the ground. As seen in alternate studies, aerial LiDAR has had more success in quantifying broader scale variation in structural diversity compared to terrestrial LiDAR, which was better at quantifying on a finer site-specific scale (LaRue et al.,

2020). Thus, a comparison of a variety of scales could be examined with different metrics of structural diversity and different forms of LiDAR.

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