WILDFIRE REFUGIA WITHIN A BOREAL SHIELD PEATLAND AND ROCK BARRENS LANDSCAPE: IDENTIFICATION, DRIVERS, AND ECOHYDROLOGICAL INDICATORS

WILDFIRE REFUGIA WITHIN A BOREAL SHIELD PEATLAND AND ROCK BARRENS LANDSCAPE: IDENTIFICATION, DRIVERS, AND ECOHYDROLOGICAL INDICATORS

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TITLE: Wildfire refugia within a Boreal Shield peatland and rock barrens landscape: Identification, drivers, and ecohydrological indicators

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Lay Abstract

Areas which remain unburned, or burn at a low severity during a wildfire, are referred to as fire refugia by scientists and conservationists for their role in providing habitat to plants and animals following a fire and promoting the regeneration of the burned landscape. Here, we use modelling and field survey methods to examine the biological and physical controls of fire refugia occurrence in an Ontario Boreal Shield landscape. We find that large, deep peatlands and wetlands in bedrock depressions on this landscape are more likely to act as fire refugia, and that confirmed peatland fire refugia have distinct vegetation communities and more stable water tables when compared to other peatlands and wetlands on this landscape. These insights into fire refugia occurrence in the Ontario Boreal Shield will assist in the detection of potential refugia for the targeting of conservation and management strategies to help protect these ecologically important areas.

Abstract

Fire refugia, defined as unburned, functionally intact patches of habitat within a fire footprint, play an important role in post-fire recovery and landscape resilience to fires. Increased fire activity in the Canadian boreal forest due to climate change highlights the need to properly identify and manage wildfire refugia to protect the natural resilience of boreal ecosystems. While previous fire refugia research has focused on western Canada, we present the first characterization of fire refugia, with a focus on peatland fire refugia, in Ontario. We use remotely sensed multispectral imagery and stereo-derived DEM data from the 2018 Parry Sound 33 wildfire in the Ontario Boreal Shield to determine the primary drivers of fire refugia formation on this landscape, and to develop a model to predict the occurrence of potential fire refugia based on these drivers. We found that the Normalized Difference Moisture Index (NDMI) and the Topographic Position Index (TPI, 200m radius neighbourhood) had the strongest control on wildfire refugia probability in the model, with a combined relative influence of 63.8%. Additionally, wildfire refugia tended to form in peat-filled depressions, valleys, and forested areas within the study area, whereas drier, open rock barrens were most susceptible to fire. Overall, the model had a high predictive accuracy, with a cross-validated AUC of 0.88, and a sensitivity of 81.2%. We conclude that local scale topography and simple flow accumulation models can act as a powerful tool in predicting fire refugia occurrence in this landscape.

In the second part of this study, we examined the in-situ indicators of peatland fire refugia occurrence. We conducted vegetation surveys at eight peatland fire refugia and eight reference sites representative of the range of wetland types found on this landscape. We found that the peatland fire refugia had a significantly different understorey vegetation composition when compared to the reference sites. Environmental factors within the peatland fire refugia which significantly influenced this separation included median peat depth, pH, and specific conductance (SpC); where peatland fire refugia were deeper and had a lower pH and SpC when compared to the reference sites. While no vascular indicator species were identified within the peatland fire refugia, there were two bryophyte indicator species: *Sphagnum rubellum* and *Sphagnum magellanicum* which were significantly associated with the peatland fire refugia. We conclude that understorey vegetation composition, indicator species presence, peat depth, pH and SpC could be useful when distinguishing peatlands with a high refugia probability, however, further research is needed to understand how this may vary geographically and in response to top-down controls, such as fire weather. Overall, the preliminary characterization of fire refugia in the Ontario Boreal Shield will provide a basis for the identification and mapping of fire refugia within this ecozone for applications in conservation, restoration, and fire and land management.

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category. P-Values were adjusted for multiple testing following the Benjamini and Hochberg (1995) procedure.

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List of Abbreviations and Symbols

°	Degrees
°C	Degrees Celsius
AUC	Area under the curve
BA	Basal area
CI	Convergence Index
COOP	Central Ontario Orthophotography Project
cv	Cross-validated
DBH	Diameter at breast height
DEM	Digital Elevation Model
df	Degrees of freedom
dNBR	Differenced Normalized Burn Ratio
ELC	Ecological Land Classification
EST	Eastern Standard Time
FWI	Fire Weather Index
GBM	Gradient boosted regression model
GPS	Global Positioning System
IndVal	Indicator Value
IQR	Interquartile Range
NBR	Normalized Burn Ratio
NDMI	Normalized Difference Moisture Index
NIR	Near Infrared
NMDS	Non-metric multidimensional scaling

PS33	Parry Sound 33
PVC	Polyvinyl chloride
RdNBR	Relative Difference Normalized Burn Ratio
ROC	Receiver-operating characteristic
SD	Standard deviation
SpC	Specific conductance
SWI	SAGA Wetness Index
SWIR	Shortwave Infrared
TCA	Total Catchment Area
TPI	Topographic Position Index
TWI	Topographic Wetness Index
WT	Water table
WTD	Water table depth

Declaration of Academic Achievement

This thesis was written in "sandwich thesis" format, where all written material was prepared solely by this author. Chapters one and four provide a general introduction and conclusion, respectively. Chapters two and three constitute the main body of the thesis and are prepared as individual manuscripts. Dr. Mike Waddington and Dr. Sophie Wilkinson contributed substantially to the research design and direction. Dr. Paul Moore was responsible for the initial processing and calibration of the raw water level data. Dr. Chantel Markle was responsible for the acquisition of the Sentinel-2 data and assisted with statistics and research design, along with Dr. Colin McCarter. Also, Gracie Crafts, Maia Moore, Alexandra Clark, Emma Sherwood, Hope Freeman, Sarah Wiebe, Taylor North, Alexander Furukawa, Kyra Simone, and Amanda Qian assisted with the installation of groundwater wells, water level monitoring, vegetation and tree surveys, elevation surveys, and pH and SpC monitoring.

Chapter 1: General Introduction

While wildfires can burn areas a few hundred to thousands of hectares in size, some areas within these wildfire footprints remain unburned. These areas, called fire refugia, are of particular interest to fire managers and conservationists since they play an important role in biodiversity conservation and ecological resilience to wildfire (Meddens et al., 2018; Coop et al., 2019; Riva et al., 2020). During and shortly after a wildfire, refugia act as oases of intact habitat for surviving animal populations and new migrants (Banks et al., 2011). Plants which are not fire-adapted, and thus do not have propagules that can survive fire, also rely on refugia as sources for regeneration of the surrounding burned matrix (Wills et al., 2018; Landesmann & Morales, 2018). As such, developing an understanding of the characteristics and occurrence of fire refugia in multiple geographic locations and climates will aid in the identification and targeted conservation of these critical landscape elements.

Due to the prevalence of fire in dry, continental climates such as those found in Australia and western North America, many fire refugia case studies are concentrated in these areas (Brennan et al., 2011; Collins et al., 2019; Blomdahl et al., 2019; Krawchuk et al., 2016). Comparatively fewer studies have examined fire refugia occurrence and function in northern humid or lake-mediated climates (Nielsen et al., 2016; Barbé et al., 2017). Nevertheless, Stralberg et al. (2020) proposed that wet environments, such as boreal peatlands, may have a greater potential to act as climate-change or fire refugia because they exhibit 'ecological inertia' in response to rapid global change, due to multiple negative ecohydrological feedbacks (Waddington et al., 2015). Moreover, the topographic (*e.g.* relief, surficial geology) and hydrological (*e.g.* groundwater connectivity, precipitation inputs) factors that support cooler, wetter microclimates in these peatlands, buffer the impacts of climate-related disturbances such as drought and wildfire (Stralberg et al., 2020). This is of particular interest because wildfire is the most prevalent disturbance in northern peatlands, representing over 97% of disturbances by area (Turetsky et al., 2002). Wildfire frequency, area and severity within boreal peatlands are expected to increase due to climate-mediated drying and anthropogenic disturbances, such as draining and harvesting (Gillet et al., 2004; Kasischke & Turetsky, 2006; Wilkinson et al., 2018). Given that northern peatlands store approximately one third of Earth's soil carbon (Gorham, 1991), an increase in peat fire frequency and severity threatens the function of peatlands as carbon sinks (Turetsky et al., 2015) and global climate regulators by rapidly releasing this stored carbon into the atmosphere (Zoltai et al., 1998; Turetsky et al., 2011; Turetsky et al., 2015). At a broader scale, the value of peatlands as carbon sinks intersects with their potential function as fire refugia. Peatlands and wetlands are more likely to act as fire refugia when compared to uplands, increasing the overall resilience of peatland-dominated landscapes to fire (Bourgeau-Chavez et al., 2020; Whitman et al., 2018; Whitman et al., 2019). Therefore, it is important to understand what controls the spread of wildfire in these landscapes in order to inform fire management techniques and guide peatland fire refugia conservation.

Past studies have shown that the presence of fire refugia can act to increase ecological resilience at local to regional scales, by providing refuge for flora and fauna in the short-term (Williams et al., 2008; Steenvoorden et al., 2019), and subsequently acting as nuclei for the regeneration of the burnt landscape post-fire (Landesmann & Morales, 2018; Coop et al., 2019; Downing et al., 2019). Conservation and restoration of confirmed and potential fire refugia may aid in mitigating the impacts of increasingly frequent and severe wildfires with climate change. As discussed by Morelli et al. (2016), refugia need to first be identified and mapped, within the context of conservation or management goals, to be prioritized for conservation or adaptive management actions. Multiple studies have attempted to map potential fire refugia using predictive models trained on landscape-level topographic and ecohydrological variables (Camp et al., 1997; Krawchuk et al., 2016; Rogeau et al., 2018). However, most studies have focused on fire refugia occurrence in western North America, where drivers of refugia occurrence may differ from those in humid climates, and areas with lower topographic relief, such as the Boreal Shield.

1.1 Thesis Objectives

To address this research gap we examine a large (11,000+ ha) wildfire (2018 Parry Sound 33 (PS33)) in the eastern Georgian Bay rock barrens region of the eastern Boreal Shield to: i) estimate the proportion and size of fire refugia within the PS33 fire footprint using multispectral imagery, ii) create a gradient boosted regression model which can estimate the locations and likelihood of potential fire refugia using a suite of remotely sensed biophysical variables, iii) determine the relative influence of individual biophysical variables in controlling fire refugia probability, iv) characterize vegetation communities within peatland fire refugia and determine whether they differ significantly from wetlands in the surrounding unburnt landscape, and if so, which ecohydrological variables drive that separation, v) identify indicator species which show a high fidelity and specificity to peatland fire refugia, and vi) assess the maximum water table depth and water table drawdown rate during the longest summer rain-free period within peatland fire refugia and unburned reference sites. Within the context of this study, peatland fire refugia are defined as peatlands (with organic soils ≥ 0.4m deep) within the Parry Sound 33 (PS33) fire footprint which have a burn scar (*i.e.* charred/singed peat) coverage < 5% and no tree mortality due to fire. To address the first two objectives, we use remotely sensed Sentinel-2 L2A 20 m imagery to assess burn severity within the PS33 fire footprint and identify fire refugia. These fire refugia are used to train a gradient boosted regression model in R on a suite of biophysical variables calculated from the Central Ontario Orthophotography Project (COOP) 2016 stereo-derived 2m DEM (resampled to 20 m) (Ontario Ministry of Natural Resources, 2017), and pre-fire multispectral Sentinel-2 L2A 20 m imagery. To address the third and fourth objectives, general vegetation surveys were conducted in the summer (July-August) of 2021 using 10 x 1m^2 quadrats within eight peatland fire refugia, and eight unburned reference sites (outside the PS33 fire footprint). To address the fifth objective, 2” PVC wells were installed in the deepest open canopy section of eight peatland fire refugia and eight reference sites prior to the summer of 2021, and water levels were measured continuously during the 2021 growing season (May to October).

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Chapter 2: Bottom-up controls of fire refugia in a Boreal Shield landscape

2.0 Abstract

Fire refugia, defined as remnant patches of habitat which remain unburned following a fire, play an important role in post-fire recovery and the regeneration of the burned landscape. Previous work has shown that bottom-up controls, such as topography and vegetation, can be used to predict where fire refugia might occur on the landscape, in order to target conservation and management strategies. However, research to date has predominantly focused on western North America, in topographically complex, subhumid regions. Here, we present the first comprehensive study of the bottom-up controls of fire refugia in the eastern Boreal Shield in Ontario. We use a Gradient Boosted Regression Model (GBM), and remotely sensed data from the 2018 Parry Sound 33 wildfire in eastern Georgian Bay, Ontario, to assess the influence of seven variables on fire refugia probability: Normalized Difference Moisture Index (NDMI, a proxy for vegetation water content); Topographic Position Index (TPI; *i.e.* relative elevation) within a 200m radius; Euclidean distance to mapped water bodies; slope; Convergence Index (CI); and total catchment area (TCA). The most influential variables in the model were NDMI (43.2%) and TPI (20.6%), where soil-filled depressions with a high vegetation water content (*i.e.* NDMI) had a higher refugia probability compared to drier, sparsely-vegetated upland areas. Overall, the model had a high predictive accuracy, with a five-fold cross-validated AUC of 0.88 and a sensitivity of 81.2%. Our results show that simple, persistent biophysical drivers can influence where fire refugia occur on this landscape. This foundational knowledge provides a framework for the mapping and identification of potential fire refugia in Ontario to inform fire management, restoration and conservation strategies.

2.1 Introduction

Ontario Boreal Shield landscapes are a unique mosaic of open rock barrens, upland forests, organic soil-filled depressions and wetlands, and thus allow for the intersection of multiple important ecohydrological functions; including carbon sequestration - as runoff accumulation in bedrock depressions creates favourable conditions for peat formation - ; and habitat for flora and fauna, such as nesting habitat for turtle species-at-risk (SAR), provided by moss and lichen mats on the upland rock barrens (Markle et al., 2020a). The mosaic nature of these landscapes has also been linked to fire severity, where fire severity increases with increasing landscape position (lowland to upland) and with decreasing organic soil depth (Wilkinson et al., 2020). However, specific controls of fire refugia occurrence have yet to be identified in the Ontario Boreal Shield. Understanding these controls will help inform conservation and management strategies to help maintain the ecological resilience of these landscapes to wildfire.

The location of fire refugia within a burned landscape is dependent on both top-down and bottom-up controls (Krawchuk et al., 2016; Meigs et al., 2020; Mackey et al., 2021); where top-down controls include various aspects of fire weather, such as wind speed and direction, air temperature, relative humidity, and precipitation (Downing et al., 2021), and bottom-up controls include local and regional topography (*e.g.* relative elevation and slope), water table dynamics, and vegetation (Rogeau et al., 2018; Sommers & Flannigan, 2022). Top-down controls, such as shifts in wind direction during a fire, are stochastic (Meddens et al., 2018). Recent studies have shown that extreme fire weather conditions can override bottom-up controls to increase the stochasticity of fire refugia and climate change refugia occurrence and reduce refugium size (Tepley et al., 2017; Mackey et al., 2021). However, bottom-up controls tend to be persistent and easily measured, making them ideal for the mapping and subsequent prioritization of potential fire refugia for long-term conservation and management strategies (Rogeau et al., 2018; Meigs et al., 2020).

In recent decades, change detection using satellite imagery has become a common method for measuring fire severity (*e.g.* NBR: Key & Benson, 2006, RdNBR: Miller & Thode, 2007); where change detection involves the differencing of two or more consecutive images of the same location to detect changes in landscape features over time. The unique spectral signature of healthy vegetation displays a large near-infrared to red ratio, which can be used to detect the burn severity of previously vegetated areas in post-fire imagery. The Relative differenced Normalized Burn Ratio (RdNBR) is a change detection algorithm which has been used in several studies to detect fire refugia from multispectral imagery (Meigs & Krawchuk, 2018).

The detection and mapping of potential fire refugia is an important step within the conservation cycle outlined in Morelli et al. (2016) and supports the validation and prioritization of potential refugia for the implementation of further management actions. Fire refugia maps may also be intersected with maps of valued resources or potential climate change refugia in order to reveal areas with a disproportionately high conservation priority (*e.g.* highly resilient landscapes, or confirmed use of potential refugia by at-risk species); as has been done to assess fire refugia quality for the northern spotted owl (*Strix occidentalis caurina*) in the United States (Andrus et al., 2021). In particular, peatlands and wetlands, which are demonstrated to have a high refugia probability compared to drier upland habitats (Bourgeau-Chavez et al., 2020), provide habitat for at-risk turtle and snake species in the eastern Georgian Bay region of Ontario (Markle et al., 2020a; Markle et al., 2020b). Following the 2018 Parry Sound 33 (PS33) wildfire, large, mid-successional wetlands and beaver-impacted wetlands in this region experienced a relatively lower fire severity and may provide suitable conditions for the co-occurrence of overwintering snakes and turtles (Markle et al., in press). Thus, conserving and restoring highly resilient peatland ecosystems within this landscape could reinforce landscape resistance and resilience to wildfire while also ensuring the continued survival of at-risk reptiles.

The objective of our research is to predict fire refugia probability using a suite of biophysical remotely sensed input variables (*i.e.* relative elevation (TPI), distance to water bodies, slope, potential soil moisture (SWI), convergence index (CI), total catchment area (TCA), and fuel water content (NDMI)). Specifically, we will: (i) create a gradient boosted regression model which can estimate the locations and likelihood of potential fire refugia using a suite of remotely sensed biophysical variables, and (ii) determine the relative influence of individual biophysical variables (TPI, slope, NDMI, distance to water bodies, SWI, catchment area, and convergence index) in controlling fire refugia probability. We predict that depressions and large-interconnected valleys will have the highest refugia probabilities due to their deeper organic soils and greater water storage capacity to buffer the effects of drought and fire (Waddington et al., 2015; Wilkinson et al., 2020); specifically, (H1) an increase in refugia probability is related to characteristics associated with depressions: a decrease in (a) TPI, (b) slope and (c) CI, and an increase in (d) SWI and (e) catchment area (Rogeau et al., 2018; Meigs et al., 2020). We also hypothesize (H2) that areas with a greater pre-fire fuel water content (NDMI) will have a higher refugia probability as higher NDMI values are associated with wetlands, which were shown to be associated with fire refugia (Whitman et al., 2018), and higher fuel water contents increase the energy required for ignition and smouldering propagation (Van Wagner, 1972). Finally, we hypothesize (H3) that areas closer to water bodies will have a higher refugia probability due to water bodies acting as fuel breaks and supporting a cooler, wetter microclimate in surrounding areas (Nielsen et al., 2016). Understanding the relative influence of individual biophysical controls within the central

Ontario rock barrens will contribute to a broader understanding of fire refugia formation in less topographically diverse regions, and in humid, lake-mediated climates.

2.2 Methods

2.2.1 Study area characterization

Our study was conducted within the Parry Sound 33 (PS33) wildfire footprint, which burned over 11,000 ha within the eastern Georgian Bay rock barrens in 2018 (Figure 2-1). The eastern Georgian Bay rock barrens are located within the Grenville province in Ontario, at the southern edge of the Boreal Shield. Recent glaciation created a mosaic landscape with shallow peat-filled granitic bedrock depressions, uplands largely devoid of soil and covered in moss and lichen cushions, and upland forests dominated by Jack Pine (*Pinus banksiana*), White Pine (*Pinus strobus*), Paper Birch (*Betula papyrifera*), and Trembling Aspen (*Populus tremuloides*) growing in more developed soil patches. It has been theorized that frequent low-severity fire, and Indigenous cultural burning (Davidson-Hunt, 2003), has contributed to the persistent dynamism of this landscape, preventing the further succession of the uplands, and expansion of the peat outwards from the depressions (Pregitzer & Saunders, 2010; Markle et al., 2021). However, the humid climate (mean annual precipitation = 1005 mm, annual mean daily temperature = 4.7°C; Environment and Climate Change Canada, 1981-2010 climate normals for Monetville, ON, ~30km northeast of PS33), and modern-era fire management, resulted in a relatively long fire return interval in this area (pre-suppression: est. 100 years, post-suppression: est. 600 years; Ward et al., 2001), with no fires rivalling PS33 recorded in central Ontario in recent decades (Canadian Forest Service, 2022a). Extreme summer drought conditions in 2018 allowed the PS33 fire to spread rapidly, covering over 4,000 ha in a single day under high fire weather (FWI = 20-30; Canadian Forest Service, 2022b).

2.2.2 Relative Difference Normalized Burn Ratio (RdNBR)

The Relative Difference Normalized Burn Ratio (RdNBR) was described by Miller and Thode (2007) as a further improvement on the Normalized Burn Ratio (NBR) (Key and Benson, 2006). Both ratios detect vegetation burn severity based on the differences in the spectral signatures between healthy vegetation and barren land. The original NBR could be differenced (dNBR) between pre-fire and post-fire images to detect absolute vegetation burn severity. With the introduction of the RdNBR, changes in vegetation burn severity are relativized for initial vegetation density, such that vegetation burn severity is not underestimated in sparsely-vegetated areas. Due to the heterogeneous vegetation cover in the rock barrens landscape, the RdNBR was chosen as the ideal vegetation burn severity metric. The RdNBR is as follows:

$$NBR = \frac{(NIR - SWIR)}{(NIR + SWIR)} * 1000 \qquad [\text{Eq. } 1]$$

$$RdNBR = \frac{NBR_{Pre-Fire} - NBR_{Post-Fire}}{\sqrt{|NBR_{Pre-Fire}/1000|}} \qquad [\text{Eq. } 2]$$

Where NIR is near-infrared (Sentinel-2 band 8a), and SWIR is shortwave infrared (Sentinel-2 band 12). Values were scaled up by a factor of 1000 for ease of interpretation and comparability across studies (Key & Benson 2006; Miller & Thode, 2007).

The PS33 wildfire started on July 18, 2018, and was declared “under control” on August 23, 2018, indicating that suppression was sufficient to prevent further fire spread. The fire was fully extinguished on October 31, 2018. Sentinel-2 Level-1C 20 m resolution multispectral imagery was acquired for September 12, 2017 (pre-fire) and September 9, 2018 (post-spread). This imagery was atmospherically corrected from Top-of-Atmosphere reflectance to Bottom-Of-Atmosphere reflectance (Level-2A product) using the Sen2Cor v8.0 tool within the Sentinel Application

Package (SNAP). The RdNBR was calculated using the raster calculator in ArcGIS Pro 2.5.0 (Figure 2-2). To further calibrate for any changes between the images unrelated to the fire (*e.g.* phenological or climatic changes), an average unburned bias value was calculated for the land area within 5 km of the fire footprint edge; this value was subtracted from each pixel within the fire footprint as a calibration factor (Key, 2006). The RdNBR values within the fire footprint were normally distributed (although slightly left-skewed: skewness = -0.33) with long tails; hence, any extreme or anomalous values outside of the typically accepted range for RdNBR (1500 ≥ RdNBR ≥ -500; Miller & Thode, 2007; Miller et al., 2009) were removed, amounting to a removal of 0.3% of the original data.

2.2.3 Classification of a binary refugia layer

The RdNBR layer was classified into 20 m resolution refugia and non-refugia pixels in ArcGIS Pro 2.5.0 using a threshold of RdNBR ≤ 166 (Figure 2-3), following the methods of Meigs and Krawchuk (2018) who used this threshold in the western United States to identify forest fire refugia with a tree mortality <10%. Despite differences in the study area, such as a humid climate and low tree density in peat-filled depressions and rock barrens, the RdNBR ≤ 166 threshold had an overall classification accuracy of 90% when compared to field-based visual burn severity surveys within 10 m-radius circular plots at 50 haphazardly selected locations within the PS33 fire footprint (16/20 refugia, and 29/30 non-refugia were correctly classified). Therefore, we proceeded with the RdNBR threshold of ≤ 166.

2.2.4 Land cover classification

Land cover within the fire footprint was classified into five categories: rock, wetland, forest, developed and open water, using pixel-based supervised classification in ArcGIS Pro 2.5.0 trained on multispectral Sentinel-2 L2A 20 m resolution imagery from June 2018, with a roughly even training area (from image interpretation) in each class (~50,000 m^2) (Figure 2-4). Wetland classification accuracy was assessed in a small-scale field survey in 2020/2021 and was found to be 84% (21/25 cells correctly classified). The classification accuracy for the other cover classes was assessed visually at 25 haphazardly-selected pixels within each class using Sentinel-2 20 m resolution imagery and Esri ArcGIS Pro base map imagery (GeoEye-1 2 m resolution imagery): barren (64%; 16/25 cells correctly classified), water (96%; 24/25 cells correctly classified), forest (92%; 23/25 cells correctly classified), developed (48%; 12/25 cells correctly classified). This classified layer was used to determine whether the pre-fire land cover composition within the fire refugia differed from the pre-fire land cover composition within the fire footprint using a Chi-squared test.

2.2.5 Derivation of topographic variables using a Digital Elevation Model

The Central Ontario Orthophotography Project (COOP) digital elevation model (DEM) was derived from stereo imagery collected between May and June of 2016 and has a spatial resolution of 2 m. This product was first resampled to match the spatial resolution of the 20 m RdNBR layer, and was subsequently used to create raster datasets of the following variables: (H1a) Topographic Position Index (TPI) – representing relative elevation within a fixed-radius moving window, where lower elevations are predicted to have a higher refugia probability; (H1b) slope – where flatter slopes are expected to have a higher refugia probability; (H1c) convergence index – representing whether the topography surrounding a given cell is convergent or divergent upon that cell (Kiss, 2004), where convergent topography is expected to yield a higher refugia probability; (H1d) SAGA Wetness Index (SWI) – representing soil wetness due to runoff accumulation (Böhner et al., 2002; Böhner & Selige, 2006), where higher SWI values are predicted to correlate with a higher refugia probability; and (H1e) total catchment area (TCA) – the total upslope catchment area above a raster cell, where cells with a larger catchment area are predicted to have a higher refugia probability. The TPI was calculated in ArcMap 10.7.1. The Focal Statistics tool was used to calculate the mean elevation around each cell within an annulus window with a centre radius of 1 m, and an outer radius of 200 m. The 200 m radius neighbourhood was optimal for minimizing noise produced by small neighbourhood sizes, while providing enough spatial accuracy to detect soil-filled depressions in this landscape. The mean neighbourhood elevation was then subtracted from the elevation of each cell to calculate the TPI value for that cell. A negative TPI value indicates that a cell is lower than its surroundings and a positive TPI value indicates that a cell is higher than its surroundings. Slope was calculated within ArcMap 10.7.1 using the Slope tool and setting units to degrees. The SAGA Wetness Index (SWI) was calculated within the SAGA Wetness Index tool in SAGA v7.9.0. The SWI is a modified version of the Topographic Wetness Index (TWI), where TWI is designed to map flow direction and accumulation at a given cell based on upstream catchment slope and area. The SWI does not treat flow as a very thin film, and uses a modified catchment area which provides a better representation of flow accumulation in relatively flat grid cells (*e.g.* depressional wetlands and valley bottoms) (Böhner et al., 2002; Böhner & Selige, 2006). The Convergence Index (CI) function in the *starsExtra* R package was used to calculate the CI for each cell in the fire footprint based on the aspect of its eight surrounding cells. The CI is represented in degrees, with negative values corresponding to convergent flow (perfectly concave topography: -90°), and positive values corresponding to divergent flow (perfectly convex topography: 90°) (Kiss, 2004). The Total Catchment Area (TCA) was calculated using the Flow Accumulation (One-Step) tool in SAGA v7.9.0 with a Multiple Flow Direction (MFD) algorithm and sink filling following the Wang and Liu (2006) method.

2.2.6 Derivation of additional variables

Pre-fire fuel moisture content was assessed using the Normalized Difference Moisture Index (NDMI) for cloud-free, atmospherically corrected (L2A) Sentinel-2 20 m imagery collected on June 29, 2018 (19 days before the PS33 fire started). The NDMI ranges from 1 to –1, where values near 1 correspond to dense, healthy vegetation with a high water content, and values below zero correspond to dry, severely stressed vegetation, sparsely-vegetated areas, or barren areas. The equation used to calculate NDMI is as follows, where NIR is near-infrared (Sentinel-2 band 8a), and SWIR is shortwave infrared (Sentinel-2 band 11):

$$NDMI = \frac{NIR - SWIR}{NIR + SWIR} \qquad [\text{Eq. } 3]$$

The Euclidean distance of each cell centre within the fire footprint to the nearest mapped water body was calculated using the Euclidean Distance tool in ArcMap 10.7.1. The DMTI Spatial Inc. waterbodies layer (2019) was used as the input for the Euclidean Distance tool.

2.2.7 Gradient Boosted Regression modelling

We used a gradient boosted regression model (GBM) similar to models previously applied in western North America (Krawchuk et al., 2016; Rogeau et al., 2018; Meigs et al., 2020) to predict where fire refugia were more likely to occur within the PS33 fire footprint based on the calculated biophysical variables (*i.e.* NDMI, TPI, SWI, CI, TCA, slope, distance to water). GBM models are ensemble models which use boosting to reduce the error (residual deviance) of each model iteration in a sequential manner. This type of machine learning model was chosen to provide better comparison to the previous studies (Krawchuk et al., 2016; Rogeau et al., 2018; Meigs et al., 2020), and for its high performance at simulating complex, non-linear interactions.

All statistical analyses were conducted in R v3.6.1. Before being input into the model, the spatial resolution and spatial extent of each raster variable was resampled against the binary refugia layer (*i.e.* 20 m resolution, clipped to remove waterbodies and all areas outside of the PS33 footprint). All variable layers (including the binary refugia layer) were stacked. The values of each variable in the stack were extracted at each cell centroid and converted to a data frame. Due to the large size of the data set (>250,000 rows) stratified random sampling (by status: refugia/non-refugia) was conducted to reduce the amount of data to 30,000 rows with balanced data for refugia and non-refugia (*i.e.* 15,000 rows each); this data reduction was sufficient to cut processing times significantly while not substantially affecting the accuracy of model results. This data set was further split into a training and testing set (80% used for training, 20% used for testing) such that the accuracy of model predictions on the external test set could be further assessed following model training and cross-validation on the training set.

The *gbm.step* function in the *dismo* package in R v3.6.1 was used with the training dataset to model fire refugia occurrence probability on an optimum number of trees selected using 5-fold cross-validation. The learning rate, bag fraction, and tree complexity of the model were set to 0.02, 0.5, and 5, respectively, similar to previous studies (Krawchuk et al., 2016; Rogeau et al., 2018), but with a marginally higher learning rate to build a number of trees between 1000 and 3000 to minimize residual deviance and processing time while avoiding overfitting. The model started with 500 trees, and built an additional 50 trees at each iteration, until it minimized the residual deviance calculated by the cross-validation. The area under the receiver-operating characteristic curve (ROC-AUC) was used to assess the accuracy of the model. Receiver-operating characteristics (ROC) and area under the curve (AUC) analyses are commonly used for assessing the classification accuracy of machine learning models (Berry et al., 2015; Krawchuk et al., 2016;

Rogeau et al., 2018; Pham et al., 2020). Previous studies which use boosted regression tree (BRT) models for fire refugia prediction have produced AUC-ROC values near 0.7-0.9 (Cartwright et al., 2018; Rogeau et al., 2018), although, with notable losses in accuracy under high fire weather conditions (Krawchuk et al., 2016; Meigs et al., 2020); where an AUC of 1 would indicate that the model classifies the target variable (in this case, fire refugia) correctly 100% of the time, whereas an AUC of 0.5 would indicate that the model had a classification accuracy of 50% (Berry et al., 2015).

Finally, predictions of fire refugia probability were made on the full, unreduced dataset (256,835 rows), and exported as a raster layer. The predicted probabilities were classified into refugia/non-refugia using a 50% threshold (Rogeau et al., 2018), where cells with a refugia probability ≤ 50% were classified as non-refugia (0) and cells with a refugia probability > 50% were classified as refugia (1). Additionally, an accuracy assessment was conducted on the testing set by calculating the agreement between the actual and predicted refugia cells.

2.3 Results

2.3.1 General fire refugia characteristics

Fire refugia detected using the RdNBR data (with a threshold for fire refugia of RdNBR ≤ 166) from the Sentinel-2 L2A 20 m imagery covered a total area of 1288.3 ha, accounting for approximately 10% of the PS33 fire footprint. The majority of fire refugia were small, with a median size of 546 m^2 and a skewness of 15.4 (Figure 2-4). Approximately 5% of fire refugia were 1 ha or larger, with the largest detected refugium being 63.1 ha.

Wetlands were the most common land cover type represented within the fire refugia, accounting for 38.1% of fire refugia by area. Forested areas were the next most common land cover type at 33.6%. Compared to the entire fire footprint, the distribution of land cover types was significantly different within the fire refugia (χ^2 = 297.36, df = 4, p < 0.001). Wetlands were overrepresented within fire refugia (Refugia: 38.1%, Fire footprint: 32.6%) as well as forested areas (Refugia: 33.6%, Fire footprint: 20.2%), and open water (Refugia: 14.4%, Fire footprint: 6.4%) (Table 2-1).

2.3.2 Variable influence

The top influential variables in the model were the Normalized Difference Moisture Index (NDMI, a measure of vegetation water content), Topographic Position Index (TPI, a measure of relative elevation, within a 200 m radius circular window), slope, SAGA Wetness Index (SWI, a measure of runoff accumulation and soil wetness), and Euclidean distance to mapped water bodies, with relative influences of: 43.2%, 20.6%, 12.0%, 8.3% and 8.2%, respectively (Figure 2-5). Each of the subsequent variables in the model (TCA and CI) had a relative influence close to 4%. The most influential variable was fuel moisture content (*i.e.* NDMI), where a partial dependence analysis showed that increasing fuel moisture content (NDMI > 0.2) produced a higher refugia probability output when all other variables were held constant (Figure 2-6). Areas with a lower relative elevation (TPI < 0) also tended to have a higher refugia probability. In addition, poorly drained flat areas with a slope < 10° had higher refugia probability values, where refugia probability tended to decrease with increasing slope (although the data had relatively few cells (~0.3%) with slopes > 10°). Potential soil moisture and runoff accumulation in low-lying areas was estimated using SWI, where refugia probability increased for cells with a SWI > 7 (although a small decrease was observed for values below 7, likely owing to low data availability). Short Euclidean distances to water (< 100m) also generated a higher refugia probability.

2.3.3 Model accuracy metrics

The model had a minimum cross-validated residual deviance of 0.87 after model optimization on 2400 trees, and a five-fold cross-validated ROC-AUC accuracy metric of 0.88 (Figure 2-7). Using the model to predict on a separate test set of data not used for model training at any stage, the model had an overall accuracy of 80.3% based on the assumption that a refugia probability ≤ 50% corresponded to refugium absence (0) and a refugia probability > 50% corresponded to refugium presence (1) (Rogeau et al., 2018). Additionally, the model had a slightly higher accuracy when predicting refugium presence (sensitivity = 81.2%) compared to refugium absence (specificity = 79.3%).

2.3.4 Model predictions

The model output assigned a refugia probability >70% to 17.1% of the output cells, and a refugia probability <20% to 45.2% of the output cells (Figure 2-8). Of the cells with a high refugia probability (>70%; Rogeau et al., 2018), 41.5% were classified as wetlands, and 51.2% were classified as forests/densely vegetated areas. Comparing the predicted fire refugia to the actual fire refugia (*i.e.* the input fire refugia cells derived from thresholding the RdNBR data), 62.3% of cells were assigned a refugia probability >70%, and 19.0% were assigned a refugia probability of 50 – 70%. Contiguous areas with a refugia probability >70% ranged in size from 0.3 ha to 58.3 ha. A large portion of the model output (37.6%) was assigned an ambiguous refugia probability (20-70%).

2.4 Discussion

We developed a machine learning model which predicts the locations and probabilities of potential fire refugia from remotely sensed data with a reasonable accuracy (sensitivity = 81.2%).

While similar models exist for western North America (Rogeau et al., 2018; Meigs et al., 2020), these are the first model results for the biophysical drivers of fire refugia in the Ontario Boreal Shield, to our knowledge. Additionally, this model builds on early-stage research of fire refugia formation in peatland-dominated boreal landscapes (Whitman et al., 2018; Bourgeau-Chavez et al., 2020; Kuntzemann, 2021).

Our results supported our hypothesis that soil-filled bedrock depressions and interconnected valleys would have a higher probability of acting as fire refugia. Specifically, the model results provided evidence to support that refugia probability increased with: (H1a) decreasing TPI (below zero), where lower relative elevations accumulate runoff and act as cold air pools (Rogeau et al., 2018; Krawchuk et al., 2020); (H1b) increasing SWI, where areas with a higher potential soil wetness were less likely to burn (Krawchuk et al., 2016); (H1c) decreasing slope, where flat, poorly-drained areas, typically occurring in wetlands were less likely to burn, and have slower spread rates due to a lack of upslope heat transfer (Dupuy, 1995; Bradstock et al., 2010; Wood et al., 2011; Krawchuk et al., 2016); (H1d) decreasing CI, where CI values between 0 and -90° represent areas of convergent flow (*e.g.* streams and depressions) (Kiss, 2004; Krawchuck et al., 2016; Rogeau et al., 2018; Meigs et al., 2020); (H1e) increasing TCA, where higher runoff inputs into a raster cell help maintain a high water content and avoid drying and burning (Meigs et al., 2020). Altogether, these simple topographic characteristics had a combined relative influence of 48.5% on the model results, suggesting that simple topographic controls of runoff accumulation are strong predictors of refugia probability on this rock barrens landscape.

The granitic bedrock underlying this landscape creates a system with relatively simple runoff dynamics, where water runs off of the relatively barren uplands and collects in poorly-drained organic soil-filled depressions. Water residence times within these depressions depend primarily on evapotranspiration, and fill-and-spill dynamics – where water exceeding the storage capacity of the depression will spill over the lowest point on the depression edge (*i.e.* the "sill") (Spence & Woo, 2003). The depth, morphology, and position of these depressions within the catchment affects how quickly they dry out, as well as how much runoff they receive and their degree of connectivity to other soil-filled depressions. Organic soil depth in depressions on this landscape has previously been linked to burn severity following the PS33 fire (Wilkinson et al., 2020); where deeper organic soils were less susceptible to high burn severities due to greater storage capacities and stronger negative ecohydrological feedbacks in deeper organic soils to mitigate drying (Waddington et al., 2015; Wilkinson et al., 2020).

Due to the lack of topographic variability within this landscape (~32 m range in elevation within the PS33 footprint), typical drivers of fire refugia in topographically complex landscapes, such as poleward-facing aspect (Camp et al., 1997; Leonard et al., 2014; Rogeau & Armstrong, 2017), and increasing elevation (Dodson & Root, 2013; Rogeau & Armstrong, 2017) were not relevant at the landscape scale. It's also important to note that we did not observe rock outcrops and cliffs acting as large-scale fire breaks on this landscape as has been shown in previous studies (Clarke, 2002; Adie et al., 2017), likely due to the resolution of the model (20 m) and the fuel connectivity within these rock barrens supported by cryptogam mats and shrubs. Hence, small-scale changes in topographic position within this rock barrens landscape may have a disproportionate influence on refugia probability, compared to topographically complex regions.

Furthermore, our results supported our hypothesis that increased pre-fire fuel water content (*i.e.* NDMI) would correlate with an increased refugia probability (H2; Povak et al., 2020). Areas with a high NDMI tended to occur in peat-filled depressions and wetlands, and thus, this variable was also weakly correlated with TPI (r = -0.36). Hence, future iterations of this model could either exclude the NDMI and use topographic variables alone to predict locations of persistent fire refugia as has been done for western Canada (Rogeau et al., 2018), or use the NDMI alone as a rough predictor of fire refugia potential under current conditions. The NDMI has been used previously to detect hydrologic refugia (Cartwright, 2018), and although the transferability of this model to other types of refugia is not tested here, the strong hydrologic controls of fire refugia occurrence on this landscape may intersect with both hydrologic and climate change refugia drivers (Stralberg et al., 2020).

Finally, our results supported our hypothesis that areas closer to water bodies would have a higher refugia probability (H3), since water bodies act as physical barriers to fire spread (Senici et al., 2010; Nielsen et al., 2016; Rogeau et al., 2018; Sommers & Flannigan, 2022), and support cooler wetter microclimates in surrounding areas (Senici et al., 2010). While water bodies and peatlands have been shown here, and in previous studies (Nielsen et al., 2016; Kuntzemann, 2021), to enhance the fire refugia potential of their surroundings, it is possible that severe fire weather and high winds could overwhelm the capacity of water bodies to act as fire breaks, highlighting the need to consider the impact of fire weather conditions on model predictions.

While strong bottom-up controls may drive persistent fire refugia which survive over multiple fire events; stochastic, top-down controls, such as fire weather could impact the predictability of fire refugia by overriding bottom-up controls, or by driving the formation of ephemeral refugia, which form randomly during a single fire event. The degree to which fire weather influences the predictability of fire refugia based on bottom-up controls has not yet been assessed in Ontario but has been shown to be significant for extreme fire weather conditions in western North America (Collins et al., 2019; Meigs et al., 2020; Downing et al., 2021). With increasing fire activity and extreme fire weather days due to climate change (Wang et al., 2015), the accuracy of model predictions based on bottom-up controls may decrease as extreme fire weather conditions lead to high intensity fires which can overcome fire breaks or barriers to spread which exist under low to moderate fire weather conditions. Within the PS33 fire footprint, a large swath (>4,000 ha) burned on a day with high fire weather conditions (FWI = 20-30; Canadian Forest Service, 2022b). This section also had a relatively lower proportion of refugia compared to the rest of the fire footprint: 9.1% versus 11.7%, respectively. Further research should incorporate fire weather scenarios into fire refugia predictions for Ontario.

Drivers of fire refugia have been shown to vary between regions (Krawchuk et al., 2016; Nielsen et al., 2016; Whitman et al., 2018; Meigs et al., 2020) and between fires (Kolden et al., 2017; Downing et al., 2021). This study focused on a single wildfire in central Ontario. While this study lends insight into the drivers of fire refugia in a humid, peatland-dominated Boreal Shield landscape, there may be variability in the strength and relevance of these drivers between fires, due to top-down controls and stochasticity, and between landscapes, due to differences in climatic conditions, topographic relief, insolation, and vegetation characteristics. Thus, continued research on additional fires and landscapes within the Ontario Boreal Shield is necessary to corroborate and build on these results.

Along with a wider geographic variability, there may also be variability in the drivers of fire refugia at smaller scales. Microrefugia within individual landscape units (*e.g.* within a peatland) may have been overlooked due to the spatial resolution of the model (20 m). At a small scale, microrefugia still provide important habitat (Brennan et al., 2011; Zaitsev et al., 2014) and act as regeneration nuclei (Banks et al., 2017); for example, surviving bryophytes might expand and regenerate peatland microtopography and carbon uptake function following a fire, and intact cryptogam mats on the uplands may help attenuate runoff. However, the actual role of bryophyte microrefugia in regeneration and post-fire succession has not been clearly defined in previous studies (Hylander & Johnson, 2010; Barbé et al., 2017). Therefore, it is important that the role of microrefugia in post-fire succession be elucidated for peatland-dominated landscapes, and that fine-scale drivers of microrefugia occurrence are defined to fully characterize resilience at all scales and within different landscape units.

Finally, this model provides a general refugia probability based on remotely sensed data and should be corroborated with in-situ observations. While large-scale field surveys have yet to be conducted to confirm the model outputs, small-scale surveys of 50 haphazardly selected points in the PS33 fire footprint, suggest that model predictions are fairly accurate (21/23 field-identified refugia plots with a refugia probability >50%, 21/27 field-identified burned plots with a refugia probability <50%). However, the accuracy of model predictions will depend on the inherent error/noise and spatiotemporal resolution of the input data. Hence, this model should not be used alone in decision-making and conservation planning and should instead be used in tandem with in-situ field surveys and spatiotemporal data of conservation values. Future work should focus on understanding local-scale ecohydrological indicators which can be used to ground-truth potential fire refugia.

This study builds on past research on the PS33 fire showing that larger depressional wetlands with a greater complexity in land cover have also exhibited lower fire severities (Markle et al., in press). These complex wetlands facilitate the co-occurrence of species-at-risk reptiles in Ontario and therefore increase the importance of these areas for conservation (Markle et al., 2020c). Future work should couple results from this model with ecological data and in-situ observations to support conservation planning, restoration and fire management strategies which target depressional wetlands with a high fire refugia probability and a high habitat value for protection, restoration, and management.

2.5 Conclusion

Here we present a first-pass model to predict fire refugia probability in the Ontario Boreal Shield using remotely sensed data. Soil-filled depressions and valleys, identified using a suite of topographic and fuel moisture variables, had the greatest refugia probability. In particular, refugia probability increased with depression depth and catchment size. Pre-fire fuel moisture content was the strongest predictor, although high levels of fuel moisture were found primarily in depressions, hence, this variable could be substituted with more persistent topographic variables, such as relative elevation, in future model iterations. Overall, this preliminary model, trained on data from the 2018 Parry Sound 33 wildfire, had a high predictive accuracy (cross-validated AUC = 0.88, sensitivity = 81%).

These results build on past research in western North America (Krawchuk et al., 2016; Rogeau et al., 2018; Meigs et al., 2020) and may be useful for conservation prioritization in Ontario where potential fire refugia overlap with conservation values. Additionally, it may provide useful information for developing fire management strategies, where large fire refugia, which support post-fire recovery and the continued provision of ecosystem services, should be designated as values for fire management, which are actively managed and avoided during prescribed burns. Increased research, understanding, and active management of fire refugia will protect species-at-risk habitat and support connectivity and post-disturbance recovery, and enhance landscape resilience to wildfire to avoid ecological tipping points.

2.6 Figures

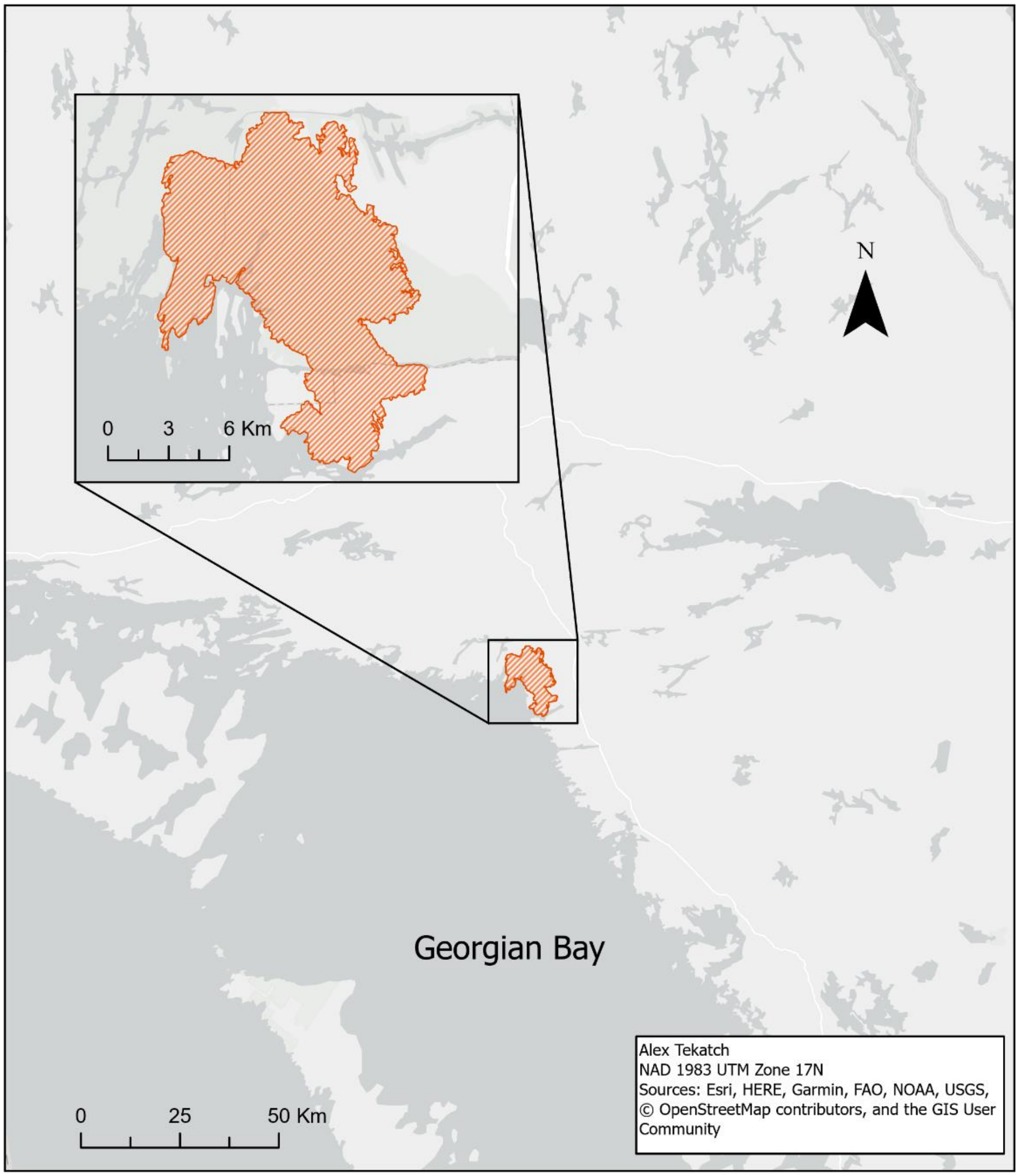

Figure 2-1: Map of the Parry Sound #33 (PS33) fire footprint (highlighted in orange) on the eastern shore of Georgian Bay, near French River, Ontario.

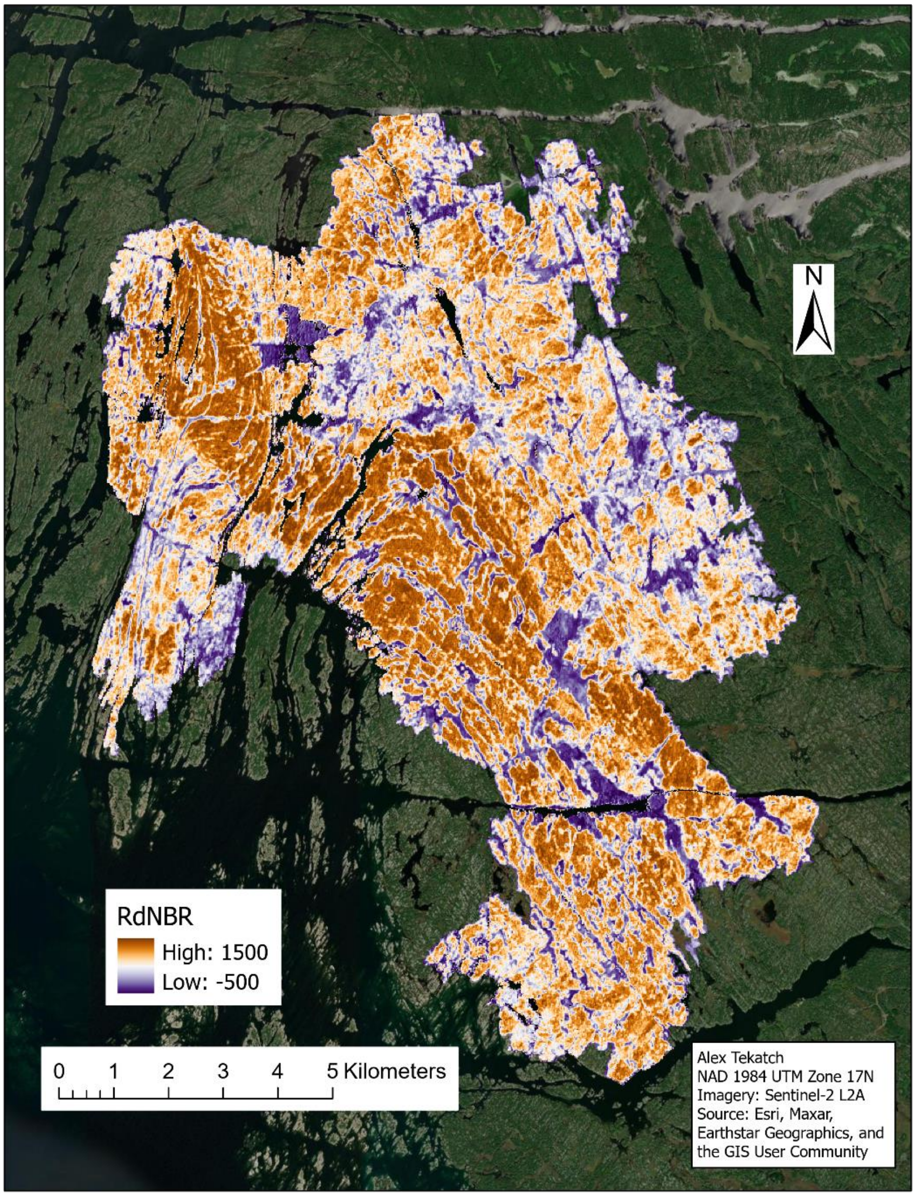

Figure 2-2: Map of Relative differenced Normalized Burn Ratio (RdNBR) values calculated from pre-fire (September 12, 2017) and post-fire (September 7, 2018) Sentinel-2 L2A 20 m resolution multispectral imagery. The raw RdNBR values were scaled up by 1000 for better interpretability. Darker orange colours represent higher burn severities, while darker purple colours represent areas which did not burn, or had enhanced vegetation density/greenness in the post-fire image.

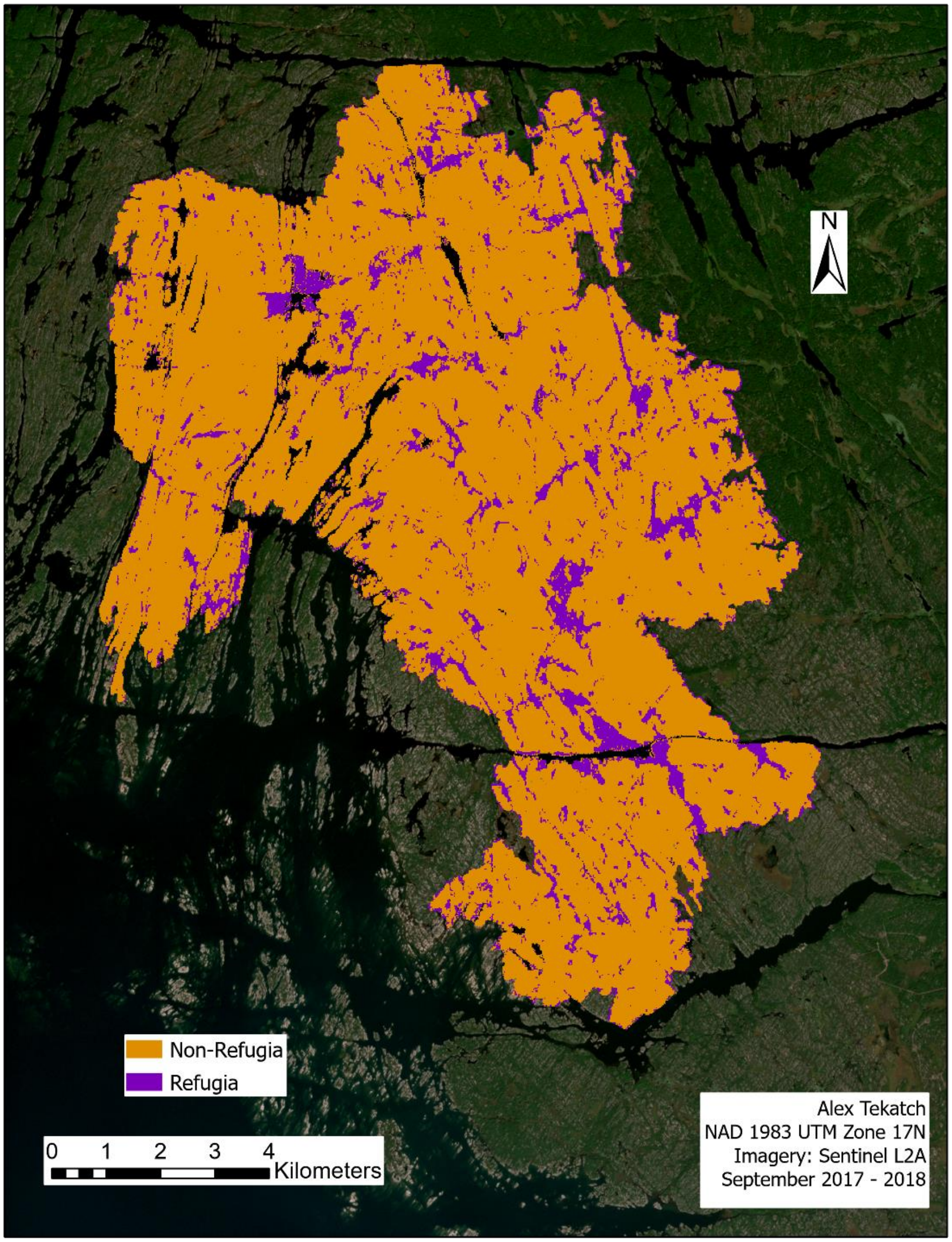

Figure 2-3: Map of fire refugia (purple) and non-refugia (orange) cells classified using the Relative differenced Normalized Burn Ratio (RdNBR) ≥ 166 threshold from Meigs & Krawchuk (2018). See Figure 2-1 for original RdNBR data.

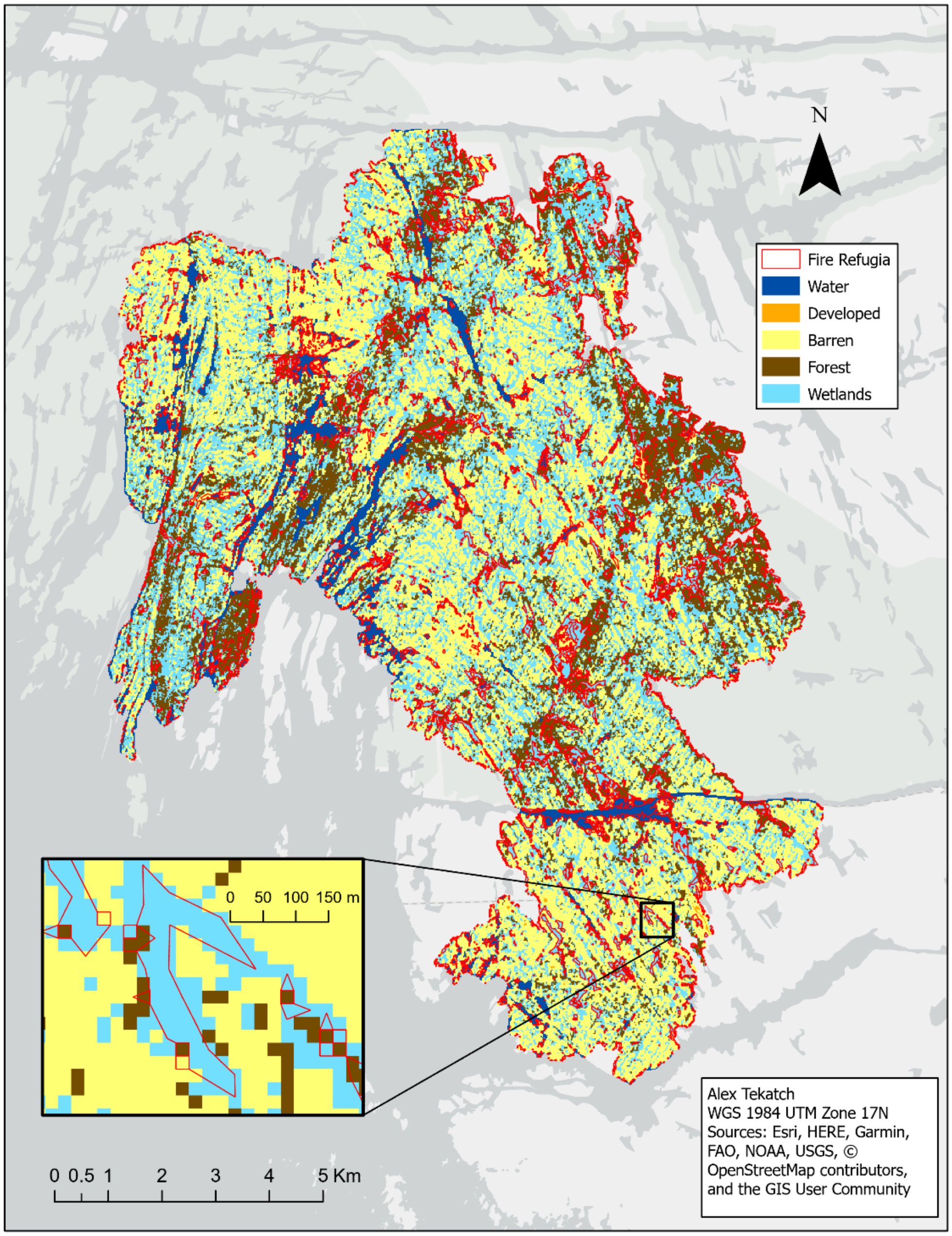

Figure 2-4: Supervised classification of land cover types within the PS33 fire footprint. Classified based on visual interpretation of Sentinel-2 20-m resolution imagery acquired in June 2018.

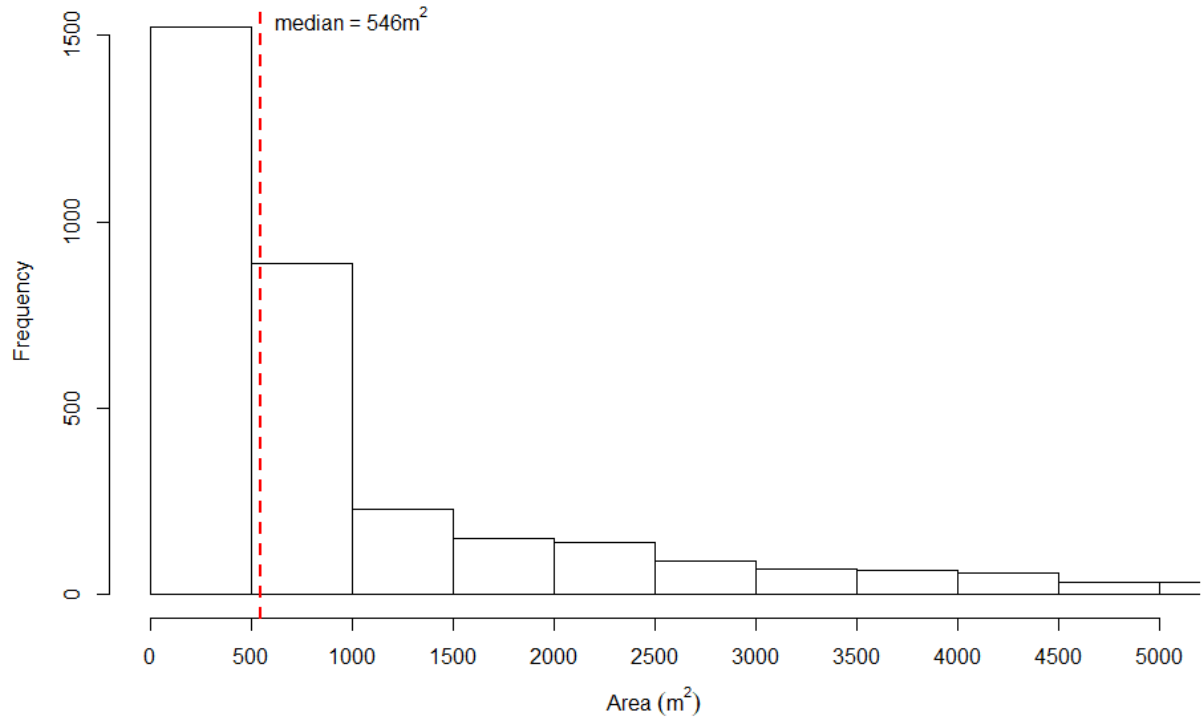

Figure 2-5: Frequency distribution of refugia sizes (area, m^2) within the PS33 fire footprint. The median area was 546 m^2, and the distribution has a skewness of 15.4.

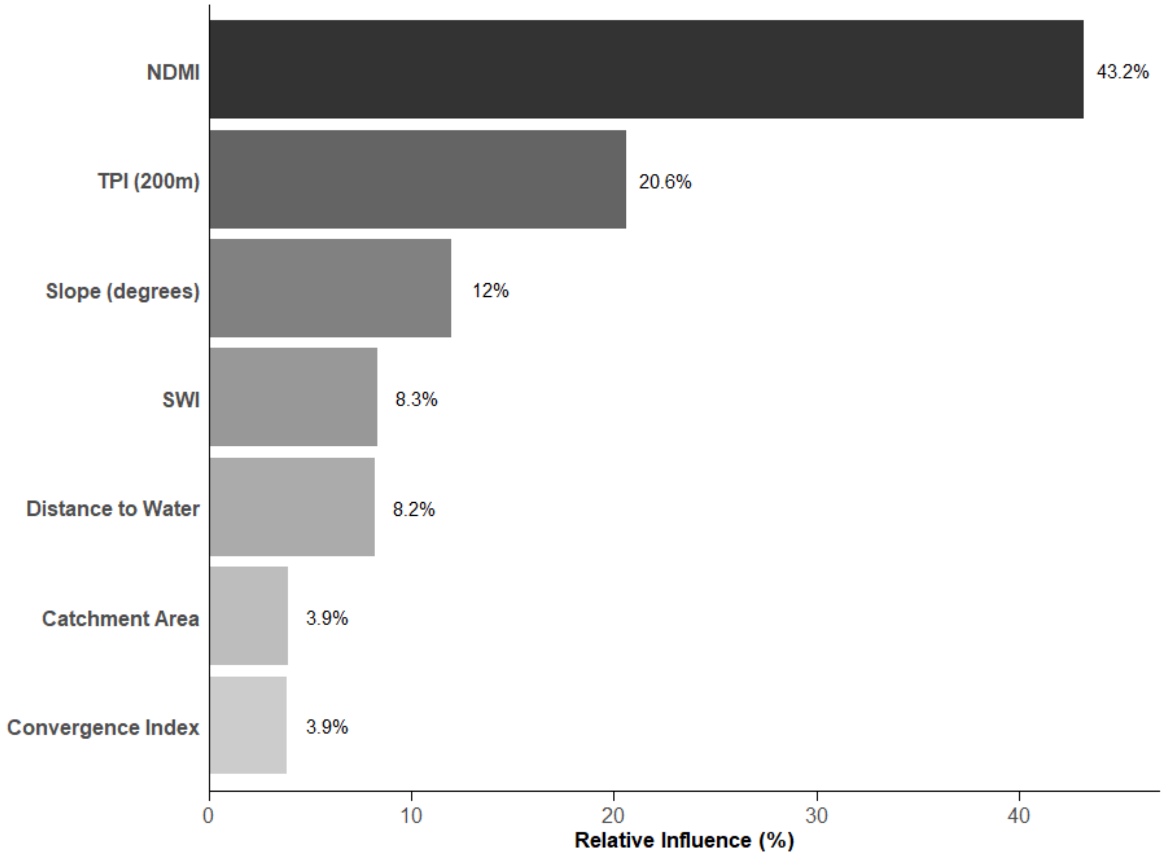

Figure 2-6: Relative influence (%) of each variable in the model (Normalized Difference Moisture Index (NDMI), Topographic Position Index (TPI) with a 200m radius circular neighbourhood, slope (°), SAGA Wetness Index (SWI), Euclidean distance to mapped water bodies (m), Catchment Area (m^2), and Convergence Index (°) on the output refugia probability.

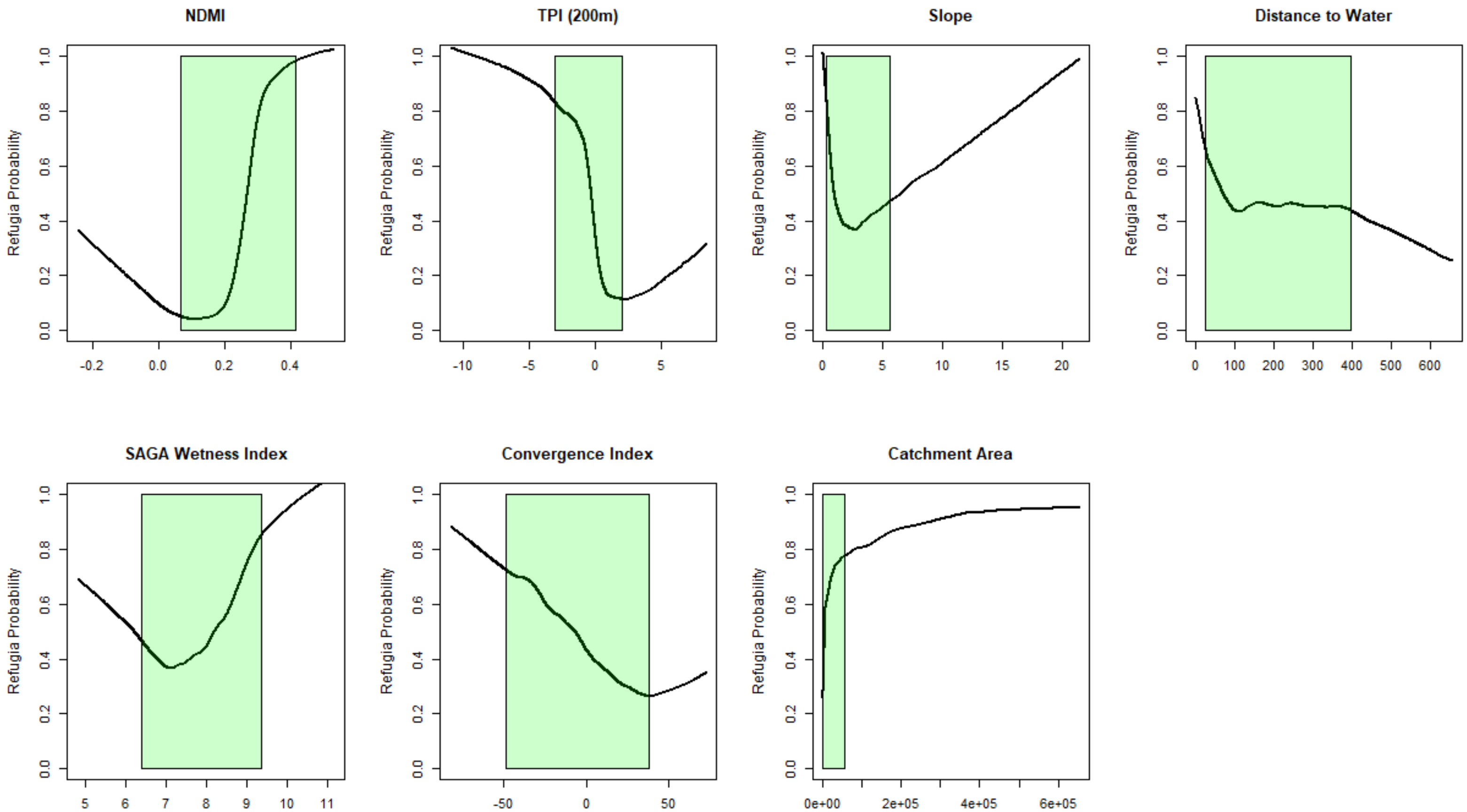

Figure 2-7: Partial dependence plots for modelled refugia probability. Each individual variable in the model is manipulated while the other variables are held constant, in order to independently assess its effect on refugia probability. The curves have been LOESS smoothed by a factor of 0.2. Green boxes show where 90% of the data occurs (5th percentile to 95th percentile).

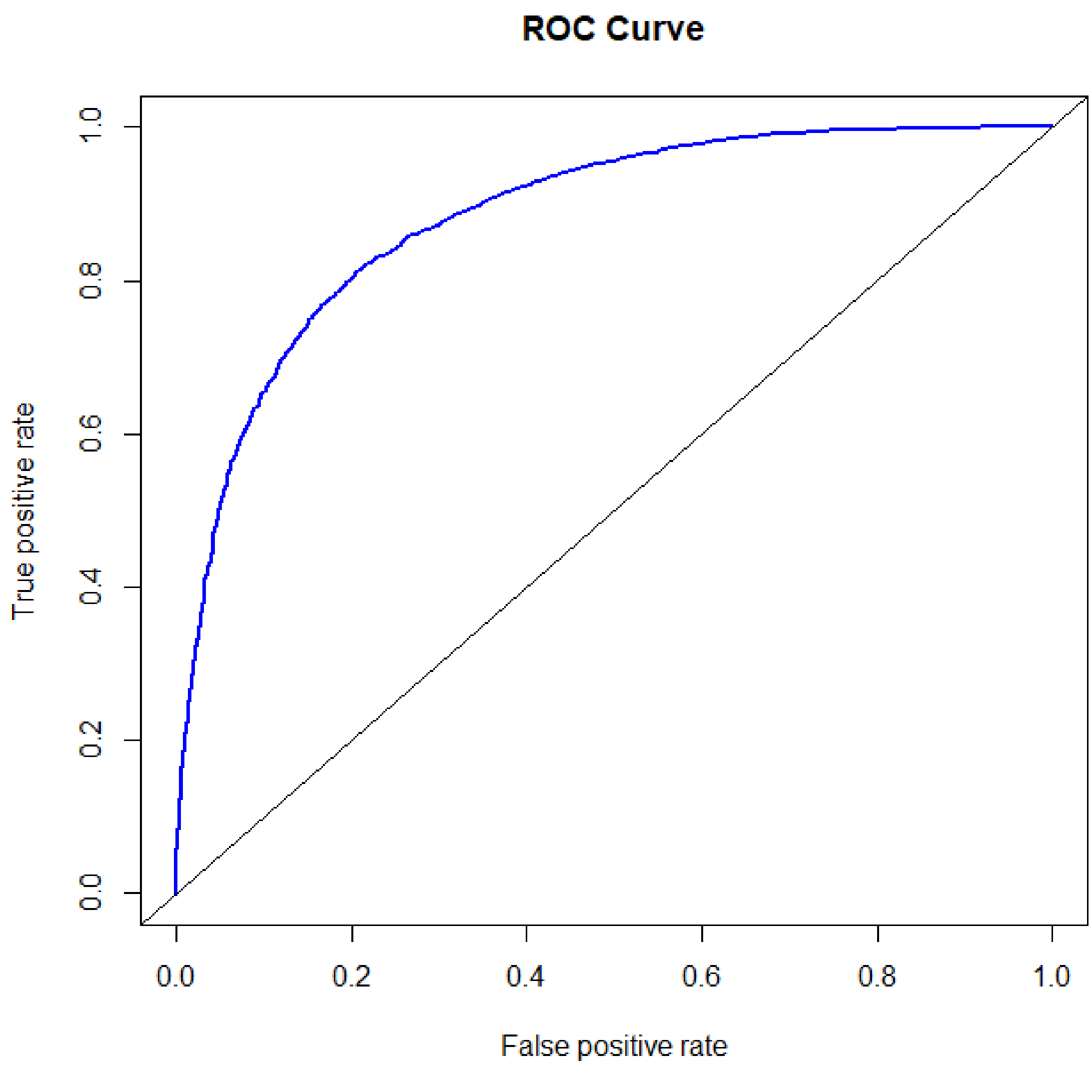

Figure 2-8: Receiver-Operating Characteristic (ROC) curve (blue line) for the Gradient Boosted Regression Model (GBM) results. The black line represents a model which makes completely random predictions, where the true positive rate (sensitivity) equals the false positive rate (specificity); the area under the black line is 0.5. The area under the ROC curve (AUC) is 0.88.

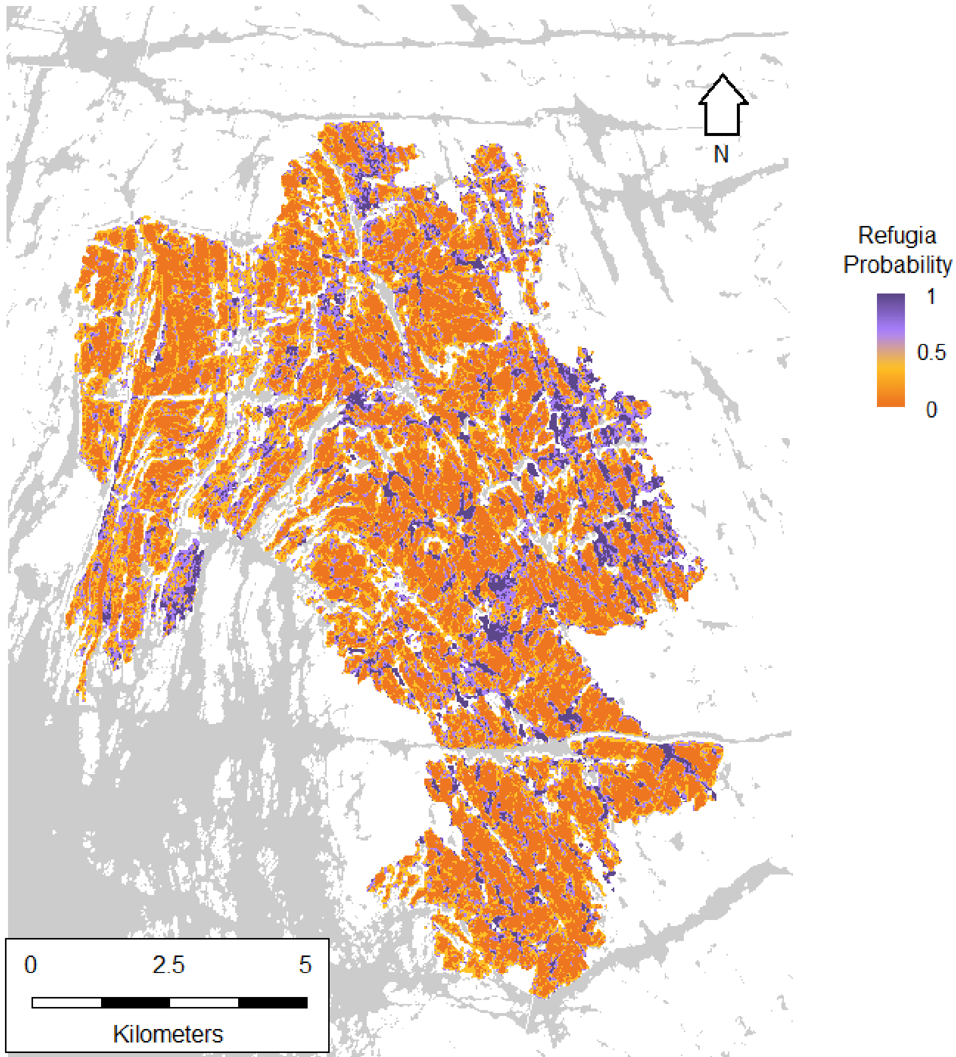

Figure 2-9: Modelled refugia probabilities for the PS33 fire footprint. High refugia probabilities (representing a high potential for a given cell to act as a refugium based on model input variables) are shown in dark purple, moderate refugia probabilities are shown in light purple, low refugia probabilities are shown in light orange, and very low refugia probabilities (high likelihood of burning) are shown in dark orange.

2.7 Tables

Table 2-1: Distribution of land cover types in the entire PS33 fire footprint compared to refugia only. The total area of classes in the fire footprint and the total area of classes in the fire refugia are given in square kilometers. Land cover types were assigned using supervised classification with training data created from the visual interpretation of multispectral imagery.

	Total Area (km^2)	**Open water (%)**	**Developed (%)**	**Barren (%)**	**Forested (%)**	**Wetland (%)**
Fire Footprint	122.7	6.4	0.05	40.8	20.2	32.6
Refugia Only	12.9	14.4	0.02	13.8	33.6	38.1

Table 2-2: Description of variables included in the model. The mean, median, standard deviation (SD), minimum (Min.) and maximum (Max.) value of each variable within the PS33 fire footprint is shown. The source data for calculating each variable is also given in the "Source" column, where DEM is the Central Ontario Orthophotography Project (COOP) 2016 stereo-derived digital elevation model (Ontario Ministry of Natural Resources, 2017). The DMTI Spatial Inc. water bodies layer (DMTI Spatial Inc., 2019), and Sentinel-2 L2A 20 m resolution multispectral imagery was also used.

	Units	**Description**	**Mean**	**Median**	**SD**	**Min.**	**Max.**	**Source**
Catchment Area	ha	Total contributing upslope area derived from flow accumulation	0.74	0.13	2.39	0.04	72.26	DEM
Convergence Index (CI)	°	Degree to which the aspects of the surrounding cells converge towards (+), or diverge from (-) the centre cell in a 3x3 cell window	-1.48	-1.57	25.45	-86.00	76.81	DEM
SAGA Wetness Index (SWI)	*N/A*	Index approximating soil wetness based on topographic controls. SWI uses a modified catchment area which does not treat flow as a very thin film	7.66	7.67	0.77	4.59	11.16	DEM
Topographic Position Index (TPI, 200m)	m	The elevation of a cell relative to a 200m radius circular neighbourhood surrounding that cell (negative values are lower than surroundings, positive values are higher than surroundings)	0.06	0.04	1.44	-11.08	9.02	DEM
Slope	°	Steepest slope between a given cell and its 8 neighbours in a 3x3 cell window	2.32	1.91	1.67	0.01	21.49	DEM
Distance to Water	m	Euclidean distance to the nearest mapped water body	165.23	134.61	121.11	0.01	663.10	DMTI Spatial Inc.
Normalized Difference Moisture Index (NDMI)	*N/A*	Normalized Difference Moisture Index (taken June 2018, pre-fire), increasing values (approaching 1) indicate dense, healthy vegetation not experiencing water stress	0.21	0.20	0.10	-0.36	0.53	Sentinel-2 L2A Imagery

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Chapter 3: Ecohydrological indicators of peatland fire refugia in a Boreal Shield landscape

3.0 Abstract

Peatlands are inherently resilient to disturbances, including wildfires and drought, due to multiple negative ecohydrological feedbacks which mitigate drying. This makes peatlands likely candidates to act as fire refugia – defined as unburned patches within a fire footprint which facilitate post-fire recovery and act as steppingstones for surviving flora and fauna. However, not all peatlands remain unburned following a fire. Here, we develop a suite of ecohydrological indicators for identifying potential peatland fire refugia. We examined the ecohydrological characteristics (*i.e.* water table dynamics, pH, specific conductance, understorey vegetation composition, and tree stand characteristics) of eight peatland fire refugia and eight unburned reference sites (outside the fire footprint) following the 2018 Parry Sound 33 wildfire in eastern Georgian Bay, Ontario. We found that the vascular and bryophyte understorey vegetation composition within the peatland fire refugia was significantly different from the reference sites ($p < 0.01$). While no clear evidence was provided of any vascular indicator species, *Sphagnum rubellum* (IndVal = 0.824, $p < 0.05$) and *Sphagnum magellanicum* (IndVal = 0.808, $p < 0.05$) may act as bryophyte indicator species for peatland fire refugia. Significant drivers of the difference in vascular vegetation composition were: i) median peat depth ($p < 0.01$), ii) maximum water table depth during the growing season ($p < 0.05$), and iii) pH ($p < 0.01$); where median peat depth was the only significant driver identified for the bryophyte composition. While the peatland fire refugia fell into the range of variability encompassed by the reference sites, they represented a unique subset of the reference sites, with a slower water table drawdown during the longest rain-free period of the summer 2021 growing season, and a generally shallower growing season maximum water table depth. These preliminary results within the Ontario Boreal Shield demonstrate that

ecohydrological characteristics may be useful in the in-situ separation of potential peatland fire refugia, as well as for the field confirmation of remote model-based classification of peatland fire refugia.

3.1 Introduction

The persistence of unburned, refugial habitat patches following a wildfire has important ecological implications, both in the short and long-term, for landscape recovery and resilience to wildfire. In the short-term, fire refugia act as safe havens for flora and fauna and can support their survival and migration within the post-fire landscape (Banks et al., 2011; Meddens et al., 2018). Over time, plants within fire refugia, and from outside the fire footprint, may contribute towards re-seeding the surrounding landscape and merge with resprouting plants, expanding available habitat for surviving fauna. In this way, fire refugia can act as nuclei for post-fire recovery, enhancing the overall resilience of a landscape to wildfire (Landesmann & Morales, 2018; Meddens et al., 2018). However, if the burned landscape crosses an ecological tipping point and shifts away from its historical function and range of ecological conditions, remaining fire refugia could exist as divergent remnant patches supporting unique communities (Meddens et al., 2018; Stralberg et al., 2020). Longer fire intervals in persistent fire refugia may shift the competitive dynamics between plant species and thus alter community structure compared to surrounding areas. For example, in the eastern Boreal Shield, black spruce (*Picea mariana*) stands, while generally adapted and resilient to fire, are susceptible to a state change to Jack pine (*Pinus banksiana*) dominance due to climate change-induced moisture deficits and increases in fire frequency (Baltzer et al., 2021). Additionally, studies have shown that understorey vegetation in fire refugia can diverge, in terms of their reproductive strategies, from areas where fire is more common; in the United States Pacific Northwest and Australia, resprouters and seed-banking species were found more frequently in historically burned sites, and obligate seeders were more prevalent in fire refugia (Clarke, 2002; Downing et al., 2019).

The timescale of mechanisms driving fire refugia occurrence can lead to the formation of either persistent or ephemeral fire refugia. Persistent fire refugia occur in the same location over multiple fire events and their occurrence is typically governed by strong deterministic factors such as convergent topography (Meddens et al., 2018; Rogeau et al., 2018; Downing et al., 2021). Whereas ephemeral fire refugia occur following a single fire event due to transient factors such as transient soil moisture conditions, or decreased fuel availability due to a recent burn (Meddens et al., 2018; Rogeau et al., 2018). Differences in the timescales of fire refugia persistence and ecosystem stability may affect community dynamics. Longer fire intervals within persistent fire refugia create favourable conditions for fire-sensitive species, and support late successional communities (Clarke, 2002; Rogeau et al., 2018; Blomdahl et al., 2019). The unique community dynamics found within persistent fire refugia can increase stand age and species diversity within a landscape, potentially increasing resilience to other disturbances such as disease and insect outbreaks (Rogeau et al., 2018; Krawchuk et al., 2020).

Persistent fire refugia can be categorized as terrain-mediated refugia, or ecosystem-protected refugia, based on the primary factors driving their formation (Stralberg et al., 2020). Ecosystem-protected refugia are highly resilient to disturbance by wildfire due to multiple strong, negative ecohydrological feedbacks (Stralberg et al., 2020). For example, in peatlands and peat deposits whereby peat deformation and decomposition, moss physiological stress thresholds and depth-dependent transmissivity feedbacks act to maintain a high water table (WT) and a high surface moisture content (Waddington et al., 2015), thereby limiting ignition and smouldering propagation (Wilkinson et al., 2019). In particular, peaty soils with stronger negative ecohydrological feedbacks, and a persistently high WT, such as in peat deposits deeper than 0.7m (Wilkinson et al., 2020), and peatlands with consistent groundwater inputs, such as fens (Bourgeau-Chavez et al., 2020), are more resistant and resilient to severe burning. However, further climate change, intensifying fire weather conditions, and anthropogenic impacts could reduce the resilience of peatlands (Balshi et al., 2009; Kettridge et al., 2015; Wilkinson et al., 2018) and persistent fire refugia (Mackey et al., 2021) to wildfires, leaving them vulnerable to a regime shift (Kettridge et al., 2015). While wildfires are currently the largest disturbance by area affecting northern peatlands (Turetsky et al., 2002), research on peatlands as fire refugia is limited (Stralberg et al., 2020; Bourgeau-Chavez et al., 2020, Kuntzemann, 2021), particularly within the eastern Boreal Shield (Barbé et al., 2017). However, the > 11,000 ha 2018 Parry Sound 33 (PS33) wildfire, occurring within a peatland-dominated Ontario Shield landscape has allowed for early remote sensing analyses of fire refugia occurrence in this ecozone. In Chapter 2, we identified the primary biophysical drivers of fire refugia following the PS33 wildfire. Using fire severity data from Chapter 2, Markle et al. (in press) further demonstrated that spatial heterogeneity in surface cover in mid-successional wetlands limits wildfire propagation and fire severity, while also providing suitable conditions for the co-occurrence of reptile species-at-risk. As such, there is a need to identify the ecohydrological indicators of potential wetland and peatland fire refugia to develop a comprehensive framework for prioritizing conservation and management strategies to these high-value areas.

This study characterizes the vegetation community composition, canopy structure and WT dynamics of eight Boreal Shield peatland wildfire refugia and compares these characteristics to eight Boreal Shield reference sites outside of the PS33 fire footprint which represent the range of wetland types found within the fire footprint. We hypothesize that: (H1) peatland fire refugia will have distinct vegetation communities when compared to a representative sample of wetland types on this landscape due to (H1a) unique associations of fire refugia with wetland type, specifically poor fens (*e.g*. evidence of consistent groundwater inputs, deep organic soils, open canopy, sedge-dominated) based on past studies indicating that open fens and large, complex wetlands have a higher tendency to act as refugia (Bourgeau-Chavez et al., 2020; Markle et al. 2022), and (H1b) a longer fire interval proxied by dominance of black spruce (*Picea mariana*) (Le Goff & Sirois, 2004; Boiffin & Munson, 2013; Baltzer et al., 2021); (H2) peatland fire refugia will have a slower WT drawdown rate during the longest summer rain-free period, and (H3) a shallower maximum WTD during the growing season when compared to a representative sample of wetland types on this landscape; where the latter hypotheses (H2 and H3) build on results from Wilkinson et al. (2020) showing that following the PS33 fire, less severely burned areas with deeper organic soils tended to have more resilient water table dynamics. Results from this study could aid in the in-situ classification and confirmation of potential fire refugia and could act as an additional tool for decision-making, conservation planning, and adaptive management to enhance the resilience of eastern Boreal Shield landscapes to fire.

3.2 Methods

3.2.1 Site selection

Peatland fire refugia sites for in-situ examination in this thesis were haphazardly selected by visual analysis of a 20 m resolution Relative difference Normalized Burn Ratio (RdNBR) map, in combination with COOP 0.2 m multispectral imagery and base map imagery in ArcGIS Pro 2.5.0 (Figure 2-1). Peatland refugia sites were visually selected as contiguous patches of relatively low RdNBR (< 300; Miller et al., 2009) in peatlands, which were accessible within 1 km of a road. In the autumn of 2020, and spring of 2021, a total of 22 suspected peatland fire refugia were visited and general surveys were conducted within a 15 m radius plot at each site to assess burn scar coverage, fire effects on vegetation cover/health, tree mortality due to fire, median peat depth, and dominant vegetation/ground cover classes, following a similar methodology to the Composite Burn Index developed by Key & Benson (2006). Of these 22 sites, 8 were selected for further investigation (groundwater monitoring and vegetation surveys) based on having a sufficient peat depth (> 0.4m) and canopy openness (100%) for well installation, a burn scar coverage <5%, and no tree mortality due to fire (*i.e.* no evidence of charring on dead trees).

Reference sites were selected from a subset of sites outside of the fire footprint which had been instrumented for previous experiments (Figure 2-1). The reference sites were located between 0.8 and 7 km away from the selected refugia sites and were haphazardly selected to represent the proportions of wetland types found within the fire footprint.

3.2.2 Vegetation surveys

Surveys of understorey vascular and non-vascular vegetation were conducted in eight peatland fire refugia, and eight reference sites outside of the fire footprint (0.2 - 2.2 km from the fire footprint edge) in July and August of 2021. The reference sites were chosen to represent the range of wetland types found within the burn scar using an Ecological Land Classification (ELC) map for the region derived from image classification in 2015 (AECOM, 2015). At each site, ten 1 m^2 quadrats were haphazardly placed to capture the variation in community composition and environmental conditions. Within each quadrat, the percent cover of each vascular plant species, and each non-vascular species/ground cover type, were separately assessed by visual estimation. The ratio of vascular plants to ground cover was also estimated. The total number of individuals, and the heights of five randomly selected individuals, were recorded for each vascular plant species. Finally, the GPS coordinates and peat depth were recorded at each quadrat, and a canopy photo was taken a breast height directly above the centre of the quadrat. If any plant could not be identified to species level in the field, a picture or voucher was taken for further identification in the laboratory. Vascular species vouchers were preserved using a standard plant press. Bryophyte species vouchers were stored in sealed plastic bags at 4°C prior to identification.

3.2.3 Vegetation analysis

To identify differences in vascular plant and bryophyte community composition between the eight surveyed peatland fire refugia and eight surveyed reference sites, a non-metric multidimensional scaling (NMDS) analysis, and all subsequent statistical analyses, were performed in R v3.6.1 using the *vegan* package. The NMDS analyses used a Bray-Curtis dissimilarity matrix with stem count data input for the vascular species analysis, and mean percent cover data input for the bryophyte species analysis. There were 20 random start configurations used for each NMDS analysis, and a stress threshold of <0.2 was used to assess the goodness of fit achieved by a given solution; a 2-dimensional analysis produced a stress value <0.2 without overcomplicating the interpretation, and was chosen as the optimum number of dimensions. The *adonis* function was used to conduct a permutational multivariate analysis of variance to identify whether there was a significant difference in the group centroid positions (*i.e.* between-group variation in vegetation composition). In addition, the *betadisper* function confirmed that the assumption of homogeneous multivariate dispersions (*i.e.* similar within-group variation in vegetation composition) in *adonis* was met (null-hypothesis of homogeneity not rejected: $p > 0.05$) to ensure that a significant result was solely due to a difference in the positions of the group centroids rather than a difference in group dispersions.

Environmental and intrinsic species drivers of vegetation community differences between sites were assessed using the *envfit* function in the R *vegan* package, with 9,999 permutations.

A multi-level pattern analysis was conducted using the *multipatt* function in the *indicspecies* package in R v3.6.1 to calculate the Indicator Value (IndVal; Dufrêne & Legendre, 1997) for each vascular and bryophyte species, respectively, and determine whether any may act as indicator species for either the peatland fire refugia, or the reference sites. Input community data matrices were calculated using stem count data for the vascular species, and percent cover data for the bryophytes. The IndVal index is a product of the specificity and fidelity of each species to a given site type (refugia or reference); where specificity is a ratio of the mean abundance of a given species in one site type, to the sum of the mean abundances of that species across all site types (*i.e.* the association between a species and a given site type), and fidelity is a ratio of the number of sites in a site type where a given species is present, to the total number of sites in that site type (*i.e.* the chance of finding a given species at a site, conditional on the site type) (Legendre, 2013). Significance testing to identify indicator species for each site type was conducted using a random permutation procedure, with 99,999 permutations, within the *multipatt* R function. The p-values produced by the *multipatt* package were corrected for multiple testing using the *p.adjust* R function with a Benjamini and Hochberg (1995) approach.

3.2.4 Tree surveys

Tree surveys were conducted within 10 m radius circular plots at three peatland fire refugia and three reference sites outside the PS33 fire footprint. Plots were centered around the PVC well at each site but were shifted further towards the centre of the peatland if they crossed the peatland or contiguous refugium edge. Within each plot, the tree species, height, diameter at breast height

(DBH), basal diameter, status (living, dead, dead due to fire), and GPS coordinates were recorded for each tree exceeding 1.3 m (the breast height of the individual conducting the survey). Due to the stunted growth of trees in these peatlands, as well as equipment limitations and dense vegetation obstructing the line-of-sight, tree height was measured directly using a measuring tape. An observer stood at a distance to assess the height of the measuring tape along the tree to improve the accuracy of the height estimation. Tree calipers were used to measure DBH and basal diameter.

3.2.5 Water level monitoring

To monitor water levels, 2” diameter PVC groundwater wells were installed in each of the eight selected peatland fire refugia. Wells were installed in the presumable deepest section of the peatland in an open location. All wells were installed by early May 2021, and all reference sites had wells installed far prior to the study period, with historical data available as early as 2017. A logging pressure transducer (Solinst Levelogger M5) was installed in each of the wells to track the water levels at 15-minute intervals throughout the 2021 growing season (May 1 to September 30). A barometric pressure logger (Solinst Barologger) was installed at a nearby site (0.6-3km away) to correct for local fluctuations in atmospheric pressure. Manual measurements of the water level and ground level with reference to the top of the PVC well casing were taken in order to calibrate the water level measurements to represent water table depth (WTD) below the surface. WT drawdown was calculated during the longest rain-free period of the growing season without data gaps: May 26 (07:30 EST) to June 5 (05:30 EST) (~9.94 days). The longest rain-free period was determined using precipitation data from a tipping bucket rain gauge located at a nearby site (0.6 - 7 km away from the refugia and reference sites; at the same location as the Solinst Barologger). The equation used to calculate WT drawdown is as follows:

$$WT_{drawdown} = \frac{WT_{initial} - WT_{final}}{T} \qquad [\text{Eq. } 1]$$

Where WT_{final} represents the final water table position (at the end of the rain-free period) below the surface in mm, $WT_{initial}$ represents the initial water table position (at the start of the rain-free period) below the surface in mm, and T is the time in days between the initial and final water table measurements.

The maximum WTD was calculated as the lowest WT position below the mean peat surface over the growing season (May to October). Referencing the WT to the mean peat surface accounted for variations in microtopography and elevation within the peatland that could not be captured at a single point (*i.e.* at the well). The mean peat surface was calculated by using a Leica DISTO S910 laser measuring tool to measure the relative elevations of 20 haphazardly selected points on the peat surface (distributed evenly between microform types and along the peatland centre-to-edge gradient), referenced to the peat surface at the well. The WTD below the peat surface was calculated at each point, assuming that the WT surface was level across the entire peatland. The WTD values across the peatland were then averaged to give a WTD value referenced to the mean peat surface.

3.2.6 pH and specific conductance

In May 2022, two pH and electrical conductivity (hereafter, specific conductance (SpC)) measurements were taken, approximately one week apart, at the eight refugia and eight reference sites described previously. The two rounds of measurements taken at each site were averaged to account for erroneous values. All measurements were taken using a YSI Professional 1030 pH/Conductivity Meter, which was calibrated using a 1413 μS/cm standard for SpC, and 4.00 and

7.00 pH standards. At each peatland site, pH and SpC were measured in the PVC well; for wells with loggers attached to metal wires, the wires were removed, and between 0.5 to 3 L of water was pumped from the well, depending on the initial water level in the well, and allowed to refill prior to the pH/SpC measurement, to avoid inaccurate measurements caused by rust buildup. Measurements were taken at a sufficient depth below the water level in the well to fully submerge the pH and SpC probes (approximately 0.1 to 0.5 m). The water temperature (°C) was also recorded for each measurement.

3.3 Results

3.3.1 Fire refugia and reference site preliminary characterization

The median area and peat depth of the eight peatland fire refugia were 2.7 ha and 1.7 m, respectively. The peatland fire refugia were mostly classed as fens (6/8), with the rest (2/8) classed as swamps (AECOM, 2015). Only 3/8 of the refugia had substantial tree cover, where black spruce (*Picea mariana*) and tamarack (*Larix laricina*) were the only observed tree species.

The median area and peat depth of the eight unburned reference sites were 0.6 ha and 0.5 m, respectively. The reference sites had a significantly shallower median peat depth when compared to the refugia (Figure 3-2; $U = 5$, $p < 0.01$). The wetland types of these reference sites were chosen to represent the range of wetland types within the PS33 fire footprint (PS33 footprint distribution: ~40% fen, ~35% swamp, ~5% bog, ~10% marsh, ~10% shallow water/other; Selected wetland reference sites: 4/8 fen, 2/8 swamp, 1/8 bog, 1/8 marsh). Half of the reference sites were treed (4/8), with the most common tree species being white pine (*Pinus strobus*), river alder (*Alnus* *incana*), Jack pine (*Pinus banksiana*), tamarack (*Larix laricina*), and black spruce (*Picea mariana*).

3.3.2 Species richness

Altogether, 38 vascular plant species and 10 bryophyte species (including 9 *Sphagnum spp.*) were identified in the peatland fire refugia. Within the reference sites, there were 60 identified vascular plant species and 13 bryophyte species. At the site level, the median species richness was not significantly different ($median_{refugia} = 16.5$, $median_{reference} = 16$, $p > 0.05$) between the peatland fire refugia and the reference sites.

3.3.3 NMDS analysis

A non-metric multidimensional scaling (NMDS) analysis (stress = 0.123) on vascular species count data revealed that vascular plant communities in peatland fire refugia were distinct from unburned reference sites ($F = 2.45$, $df = 1$, $p < 0.01$) (Figure 3-3). The species identified as the strongest drivers of vegetation community separation included: roundleaf sundew (*Drosera rotundifolia*; $r^2 = 0.72$, $p < 0.001$), bog cranberry (*Vaccinium oxycoccos*; $r^2 = 0.57$, $p < 0.01$), lowbush blueberry (*Vaccinium angustifolium*; $r^2 = 0.56$, $p < 0.01$), Fraser's Marsh St. John's-wort (*Triadenum fraseri*; $r^2 = 0.56$, $p < 0.01$), swamp candle (*Lysimachia terrestris*; $r^2 = 0.41$, $p < 0.01$), marsh cinquefoil (*Comarum palustre*; $r^2 = 0.39$, $p < 0.01$), and bluejoint grass (*Calamagrostis canadensis*; $r^2 = 0.36$, $p < 0.01$) (Figure 3-4). Aside from site classification (*i.e.* refugia vs. reference), additional environmental variables which significantly contributed to differences in vegetation composition included median peat depth ($r^2 = 0.54$, $p < 0.01$), pH ($r^2 = 0.57$, $p < 0.01$), and WTD_{max} ($r^2 = 0.43$, $p < 0.05$) (Figure 3-5).

An NMDS analysis (stress = 0.160) showed that bryophyte communities were also significantly different between refugia and reference sites ($F = 5.65$, $df = 1$, $p < 0.01$) (Figure 3-6). *Sphagnum rubellum* was identified as the species with the strongest influence on the separation between the sites ($r^2 = 0.68$, $p < 0.001$) (Figure 3-7). The only significant environmental driver of bryophyte community composition was median peat depth ($r^2 = 0.61$, $p < 0.01$) (Figure 3-8).

3.3.4 Indicator species analysis

An indicator species analysis revealed no significant associations ($p < 0.05$) of any vascular species with either the refugia or reference sites; although, there was weak evidence ($p \approx 0.06$) that bog laurel (*Kalmia polifolia*) may act as an indicator species for refugia (Table 3-1). There were 16 species found exclusively in refugia (*i.e.* species with a specificity of 1; Table 3-2), which, in order of decreasing fidelity, included white beak-sedge (*Rhynchospora alba*), Labrador tea (*Rhododendron groenlandicum*), bog aster (*Oclemena nemoralis*), and rose pogonia (*Pogonia ophioglossoides*). Conversely, 38 species were found exclusively in the reference sites, which, in order of decreasing fidelity, included white pine (*Pinus strobus*), three-way sedge (*Dulichium arundinaceum*), unbranched bur-reed (*Sparganium emersum*), winterberry holly (*Ilex verticillata*), and steeplebush (*Spiraea tomentosa*).

Significant bryophyte indicators within refugia included *Sphagnum rubellum* (IndVal = 0.824, $p = 0.028$) and *Sphagnum magellanicum* (IndVal = 0.808, $p = 0.028$) (Table 3-3). There were no significant indicator species for the reference sites. The only bryophyte species found exclusively in the refugia was *Sphagnum fuscum*, however, this species was rare and had a low fidelity (present in 2/8 refugia sites). In the reference sites, species with a specificity of 1 which were present in more than one site included *Dicranum scoparium* and *Amblystegium serpens*.

3.3.5 Tree characteristics and species composition

No significant differences were found in median tree density ($U = 6$, $p = 0.7$) or basal area ($U = 17837$, $p = 0.12$) between the three refugia sites and three reference sites surveyed. Tree density was slightly higher within the refugia, ranging between 0.11 - 0.42 trees m^{-2}, compared to reference sites, which ranged between 0.06 - 0.25 trees m^{-2}. However, only 3/8 refugia sites were substantially treed, compared to 4/8 reference sites. Additionally, trees in refugia were typically smaller than those in reference sites, with smaller median tree heights (refugia = 1.7 – 2.2 m; reference = 2 – 3.1 m), and smaller median basal areas (refugia = 8.0 - 17.3 cm^2; reference = 6.2 - 29.2 cm^2).

Only two tree species were identified within the three surveyed refugia sites: black spruce (*Picea mariana*) and tamarack (*Larix laricina*); where black spruce was dominant (representing 69% of surveyed trees). In contrast, there were seven tree species identified within the three surveyed reference sites, including black spruce and tamarack, as well as white pine (*Pinus strobus*), Jack pine (*Pinus banksiana*), paper birch (*Betula papyrifera*), river alder (*Alnus incana*), and trembling aspen (*Populus tremuloides*), of which tamarack was the most prevalent (61% of surveyed trees). There was a notably lower proportion of black spruce in the reference sites compared to the refugia: 4%, compared to 69%, respectively (Figure 3-9). The presence of Jack pine in the reference sites, and its relative absence in the refugia, should also be noted, as this species was observed to be prevalent (representing 47% of the total tree basal area) within the nearby burned sites during site-selection surveys.

3.3.6 Water chemistry

The refugia sites had a significantly lower median pH (median (refugia) = 5.11, median (reference) = 5.83, $U = 9$, $p < 0.01$) (Figure 3-10), and median SpC (median (refugia) = 35.5 µS cm^{-1}, median (reference) = 91.7 µS cm^{-1}, $U = 14$, $p < 0.05$) (Figure 3-11) when compared to the reference sites. Within the refugia, the pH ranged from 4.58 to 5.67 and the SpC ranged from 26.0 to 52.7 µS cm^{-1}; whereas in the reference sites the pH ranged from 5.02 up to 6.43, and SpC ranged from 24.1 to 170.8 µS cm^{-1}.

3.3.7 Water table dynamics

The maximum WTD within the refugia sites ranged from 0.17 to 0.32 m, and, on average, reached 0.27 ± 0.04 m during the growing season (Figure 3-12). In the reference sites, the maximum WTD ranged from 0.07 to 0.66 m, and, on average, was 0.32 ± 0.21 m (Figure 3-12). The median maximum WTD was not significantly different between the refugia and the reference sites ($U = 26$, $p = 0.57$) (Figure 3-13). However, the interquartile range (IQR) for the reference sites (IQR = 0.32m) was considerably larger than that observed in the refugia (IQR = 0.05 m) (Figure 3-13).

Within the refugia, WT drawdown rates ranged from 4.5 to 9.0 mm d^{-1} (median = 5.8 mm d^{-1}, IQR = 1.6 mm d^{-1}); while in the reference sites, WT drawdown rates ranged from 3.8 to 12.3 mm d^{-1} (median = 7.1 mm d^{-1}, IQR = 3.6 mm d^{-1}). The WT drawdown rate was not significantly different between the refugia and the reference sites ($U = 19$, $p = 0.34$) (Figure 3-14). The WT drawdown rate during the longest rain-free period was significantly correlated to the maximum growing season WTD (Adj. $R^2 = 0.661$, $p < 0.001$) (Figure 3-15), and weakly correlated to median peat depth (Adj. $R^2 = 0.184$, $p = 0.06$) in the refugia and reference sites (Figure 3-16).

3.4 Discussion

Here, we provide the first assessment of the in-situ ecohydrological characteristics of peatland fire refugia in the Ontario Boreal Shield. These characteristics, in conjunction with remotely sensed data and machine learning models (*e.g.* Chapter 2) may aid in the pre-fire identification of potential fire refugia for conservation and management prioritization (Morelli et al., 2016; Meddens et al., 2018), and provide insight into the drivers of fire refugia on this landscape to inform restoration goals geared towards increasing resilience to wildfire (Kolden et al., 2015).

While previous studies have examined the drivers of wetland and peatland fire refugia in other locations and contexts, such as wildfire vulnerability by wetland class (Whitman et al., 2018; Bourgeau-Chavez et al., 2020), surface cover complexity (Markle et al., in press), and peat depth (Wilkinson et al., 2020), we begin to tease out the ecohydrological indicators (specifically vegetation and WT dynamics) of potential peatland fire refugia on a peatland-dominated rock barrens landscape.

Our results support our hypothesis that vegetation communities within fire refugia can be distinguished from other peatlands on the landscape. This is primarily driven by wetland class and, potentially, a longer fire return interval. While it was assumed that the peatland fire refugia would be more fen-like (*i.e.* with greater groundwater inputs, signified by a higher SpC and pH), as past research has shown (Bourgeau-Chavez et al., 2020), the fire refugia selected for this study had notably more ombrotrophic conditions (lower SpC and pH) and bog-type vegetation (*e.g. Vaccinium oxycoccos, Sarracenia purpurea, Pogonia ophioglossoides*) when compared to a representative subset of wetland types on this landscape. It is possible that the significantly deeper peat measured within these fire refugia is linked to a larger water holding capacity and thus a greater resistance to drought and wildfire (Wilkinson et al., 2020). Slow water table drawdown rates and shallow maximum water table depths observed within the refugia during the summer dry period also provide evidence to support this. These conditions may have supported the persistence of these fire refugia through multiple fire events and thus promoted a longer successional trajectory for the vegetation communities in these refugia, compared to other peatlands on the landscape (Meddens et al., 2018). In particular, tree surveys revealed that black spruce (*Picea mariana*) and tamarack (*Larix laricina*) dominated the tree species composition within the refugia, while reference sites tended to be more varied with a larger proportion of Jack pine (*Pinus banksiana*). Although black spruce are generally fire-tolerant, a dynamic exists between black spruce and Jack pine in the eastern Boreal, where more frequent and severe fire regimes increase the rate of post-fire regeneration failure in black spruce, allowing for Jack pine dominance (Le Goff & Sirois, 2004; Boiffin & Munson, 2013; Baltzer et al., 2021). In general, black spruce are also associated with wetter, more poorly-drained sites, compared to Jack pine which occurs more often in well-drained upland sites (Le Goff & Sirois, 2004), making this species a potential indicator of persistent fire refugia on this landscape.

Although no significant vascular indicator species were identified in this study, there were two bryophyte indicator species which were significantly associated with the fire refugia sites: *Sphagnum magellanicum*, and *Sphagnum rubellum*. Both species tend to occur in bogs and poor fens (Gignac, 1987; Fraser et al., 2001; González et al., 2013), and are widespread throughout the study area; *Sphagnum magellanicum* has generalist habitat preferences (Gignac, 1987; Oke & Turetsky, 2020), and *Sphagnum rubellum* is often associated with low hummocks (Rydin &

McDonald, 1985). The reason for the association of these moss species with the fire refugia is unclear, but may have been driven by a difference in the range of wetland types within the reference sites when compared to the refugia sites; where the reference sites included swamps and marshes which are less likely to support peat-forming *Sphagnum spp.* (National Wetlands Working Group, 1997).

We found weak evidence to support our hypothesis that peatland fire refugia have a shallower maximum WTD during the summer growing season and a slower WT drawdown rate during the longest summer rain-free period when compared to a representative subsample of peatlands on the landscape. Specifically, the WT dynamics within the peatland fire refugia represented a unique subset of the range of variability observed within the reference sites, where the range of maximum WTD values in the refugia was shallower, and the refugia WT drawdown rates were slower, when compared to the reference sites. Given that the past fire history of the reference sites is unknown, and that they were selected to represent the range of wetland types on the landscape, we assume that they encompass the environmental conditions found in refugia, and that some reference sites may be potential refugia, should a fire burn through. The deep peat and shallow, stable water tables observed in the fire refugia builds on results from Wilkinson et al. (2020), showing that deeper peat is resistant to high burn severities due to strong, negative ecohydrological feedbacks supporting a high, stable water table (Waddington et al., 2015). However, it should be noted that, because the fire refugia were examined post-fire, increased runoff due to a lack of soil on the uplands could decrease the WTD relative to the pre-fire condition (Verkaik, 2021).

Overall, these results when combined with remote sensing and statistical modelling provide a comprehensive framework for the identification of peatlands with a high fire refugia potential.

In the context of peatland restoration and conservation planning, these results broaden our understanding of the peatland-scale indicators and drivers of resilience to wildfire. Specifically, peat-filled depressions on this Ontario Boreal Shield landscape with a median peat depth greater than 1.5 m, containing *Sphagnum magellanicum* and *Sphagnum rubellum*, and with black spruce and tamarack-dominated tree stands, may have a high fire refugia potential. However, we note the small scale of this study (eight peatland refugia and eight reference sites) and recommend further study to confirm these results in other locations with differing land cover types and climatic conditions. Additionally, results from this study should be corroborated for additional fires and different years, as there is likely stochasticity in fire refugia formation and responses to summer weather conditions that was not fully elucidated by this study. For example, 2021 had a wet growing season with 104 mm of precipitation in July (1.4 times the historical average; Environment and Climate Change Canada, 2020; Environment and Climate Change Canada, 2021), making it difficult to assess WT responses under deficit conditions.

Future work should link the ecohydrological indicators identified here with habitat values for species with a high conservation priority. Fire refugia which provide high quality habitat for at-risk species and have a high degree of connectivity provide habitat corridors and regeneration nuclei within the post-fire landscape, supporting the resilience of those species to wildfire. We also recommend the linkage of these results to other types of refugia, such as climate change refugia, where Stralberg et al. (2020) identified boreal peatlands as having a high potential to act as climate change refugia due to similar feedback mechanisms which govern resilience to disturbances such as wildfire and drought. Developing a full picture of the drivers and indicators of boreal peatland refugia potential will provide tremendous insight into how these peatlands respond to change, and how restoration and conservation strategies can be adjusted to support their resilience and continued functioning under future climate change and anthropogenic impacts.

3.5 Conclusion

Peatland fire refugia in the Ontario Boreal Shield had distinct vascular and bryophyte vegetation communities, displayed resilient water table dynamics (slower water table drawdown rate) and had a shallower maximum water table depth during the growing season, when compared to a representative sample of unburned wetland types. These results broaden our understanding of the ecohydrological indicators of peatland fire refugia and may be used in conjunction with remote sensing and modelling results (Chapter 2) to aid in the identification of potential fire refugia for conservation planning and fire management. Conceptually, these indicators appear to be associated with peatlands which are more resilient to disturbance overall. Hence, future research should expand these indicators to climate change and hydrological refugia to identify super-resilient peatlands with high conservation value.

3.6 Figures

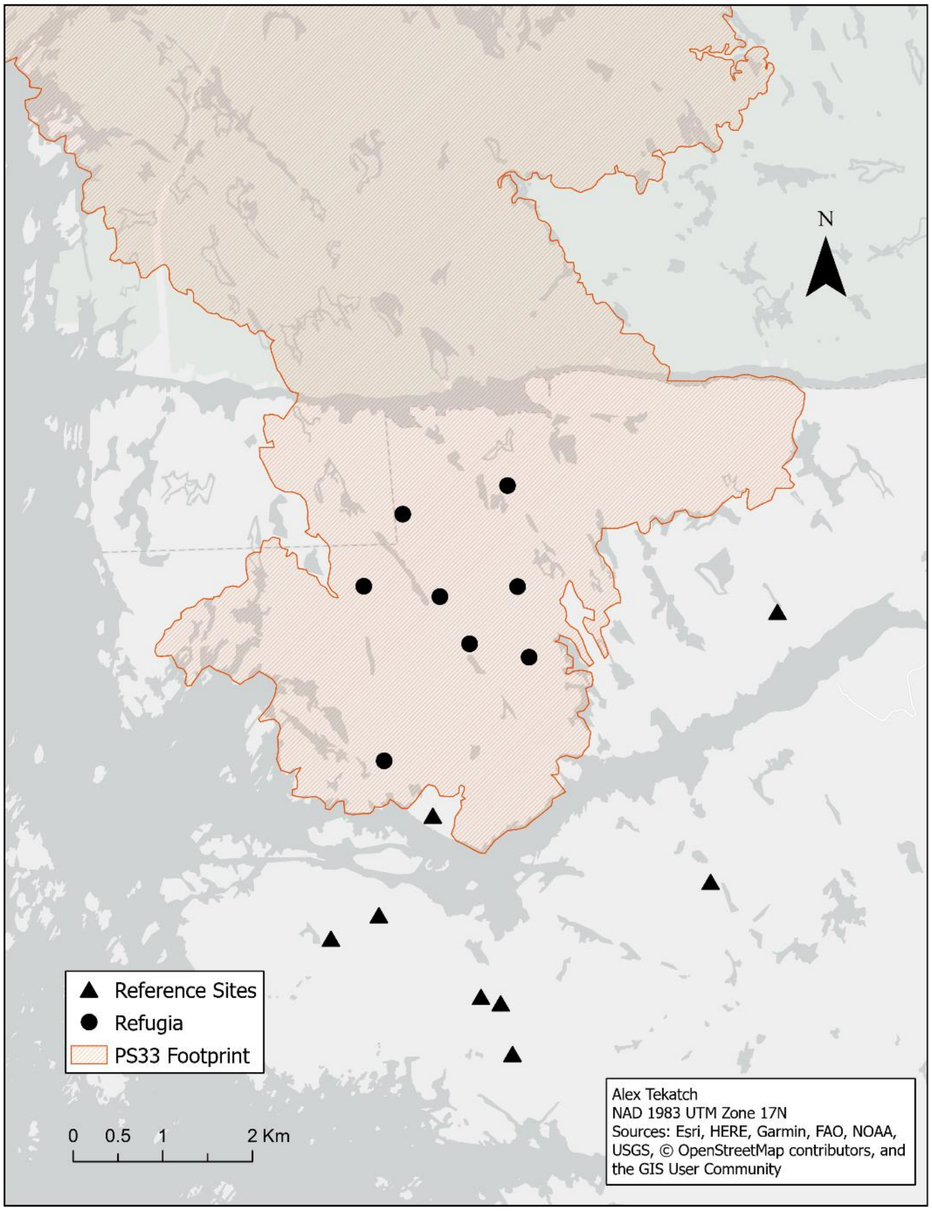

Figure 3-1: Map of peatland fire refugia (black circles) and reference sites (black triangles) examined in Chapter 3. The Parry Sound #33 (PS33) fire footprint is highlighted in orange, and dark grey areas represent water bodies.

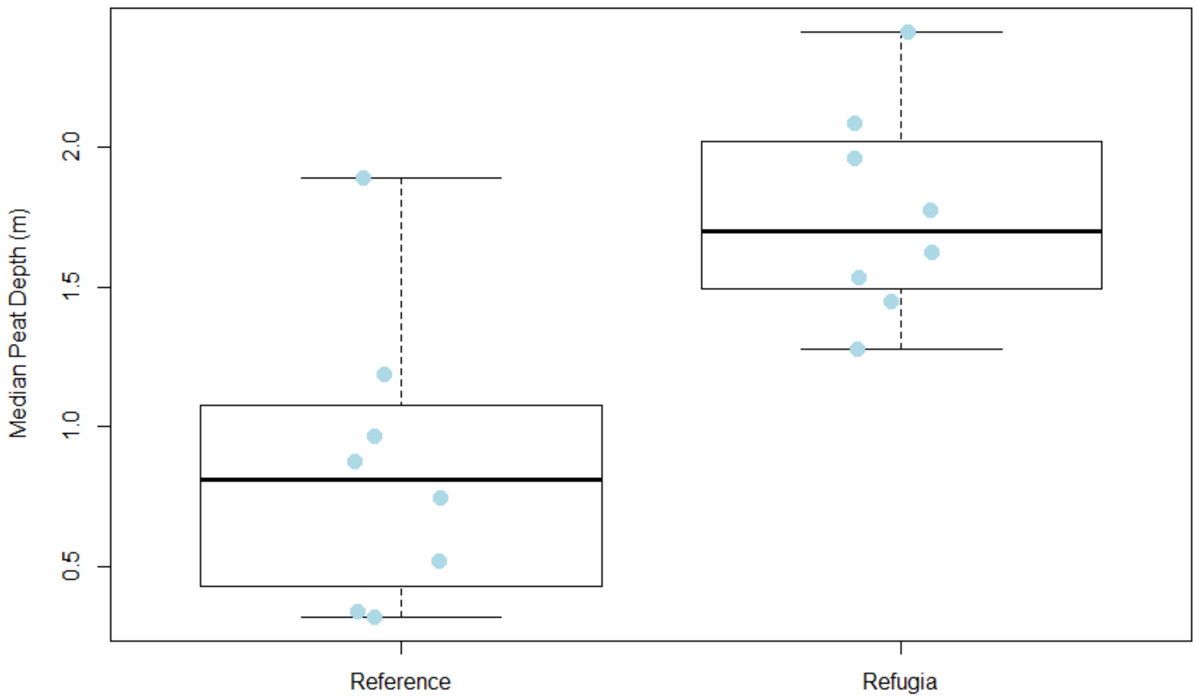

Figure 3-2: Boxplots showing the median peat depths for the reference sites and fire refugia. Individual points (light blue circles) represent the median peat depth at a given site. Median peat depths were significantly different between the reference sites and the refugia ($U = 5$, $p < 0.01$).

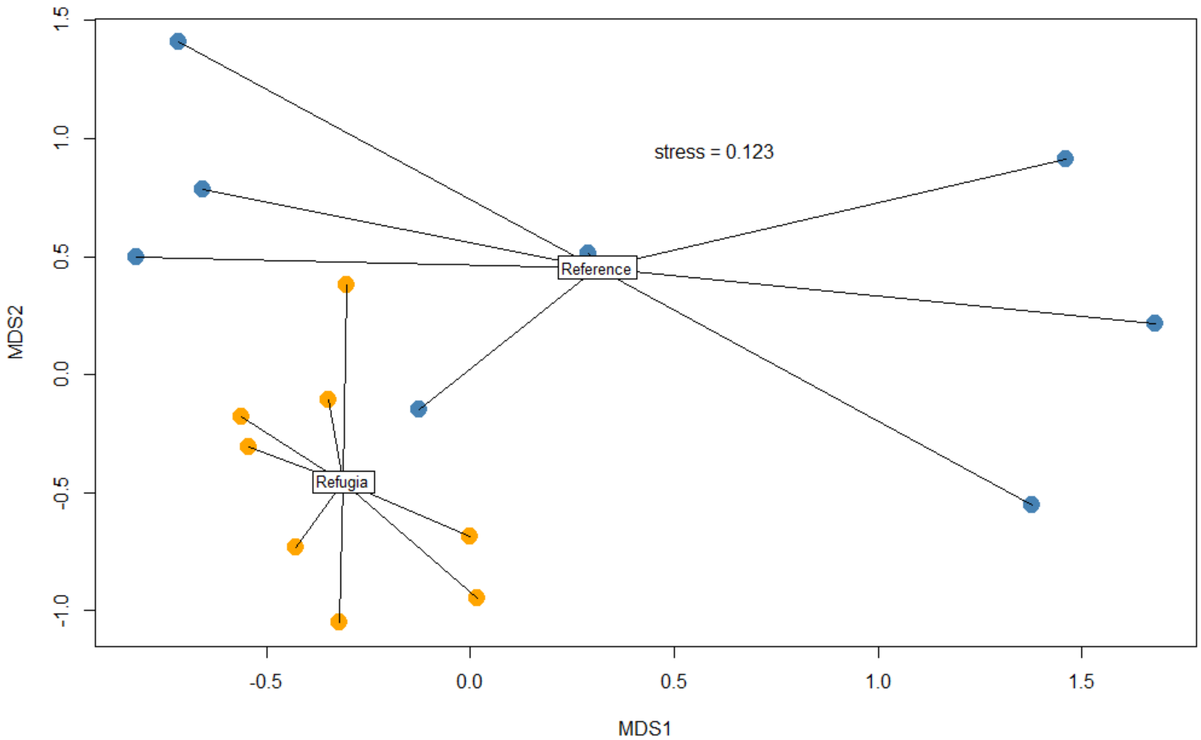

Figure 3-3: Non-metric multi-dimensional scaling (NMDS) plot of vascular understorey vegetation composition (aggregated to site-level) in eight peatland fire refugia (orange circles) and eight reference sites (blue circles). The peatland fire refugia had a significantly different vegetation composition compared to the reference sites ($F = 2.45$, $df = 1$, $p < 0.01$). The stress for the NMDS analysis was 0.123.

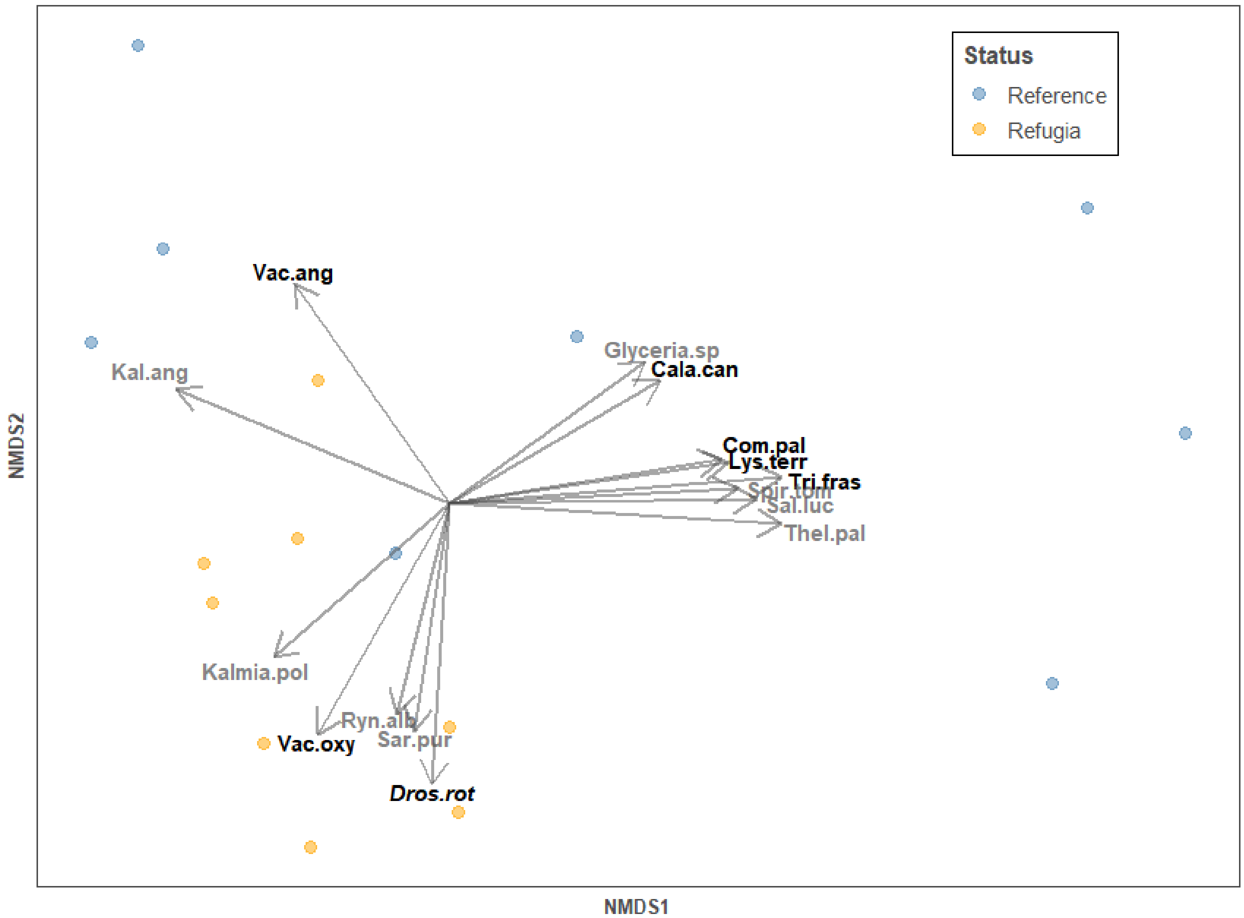

Figure 3-4: Intrinsic species plot for vascular understorey species in eight peatland fire refugia (orange circles), and eight reference sites (blue circles). All species shown have a significant influence on site ordination ($p < 0.05$). Species names shown in black indicate species with a highly significant influence on site ordination ($p < 0.01$: *Drosera rotundifolia* ("Dros.rot"; $r^2 = 0.72$, $p < 0.001$), *Vaccinium oxycoccos* ("Vac.oxy"), *Vaccinium angustifolium* ("Vac.ang"), *Triadenum fraseri* ("Tri.fras"), *Lysimachia terrestris* ("Lys.terr"), *Comarum palustre* ("Com.pal"), *Calamagrostis canadensis* ("Cala.can")).

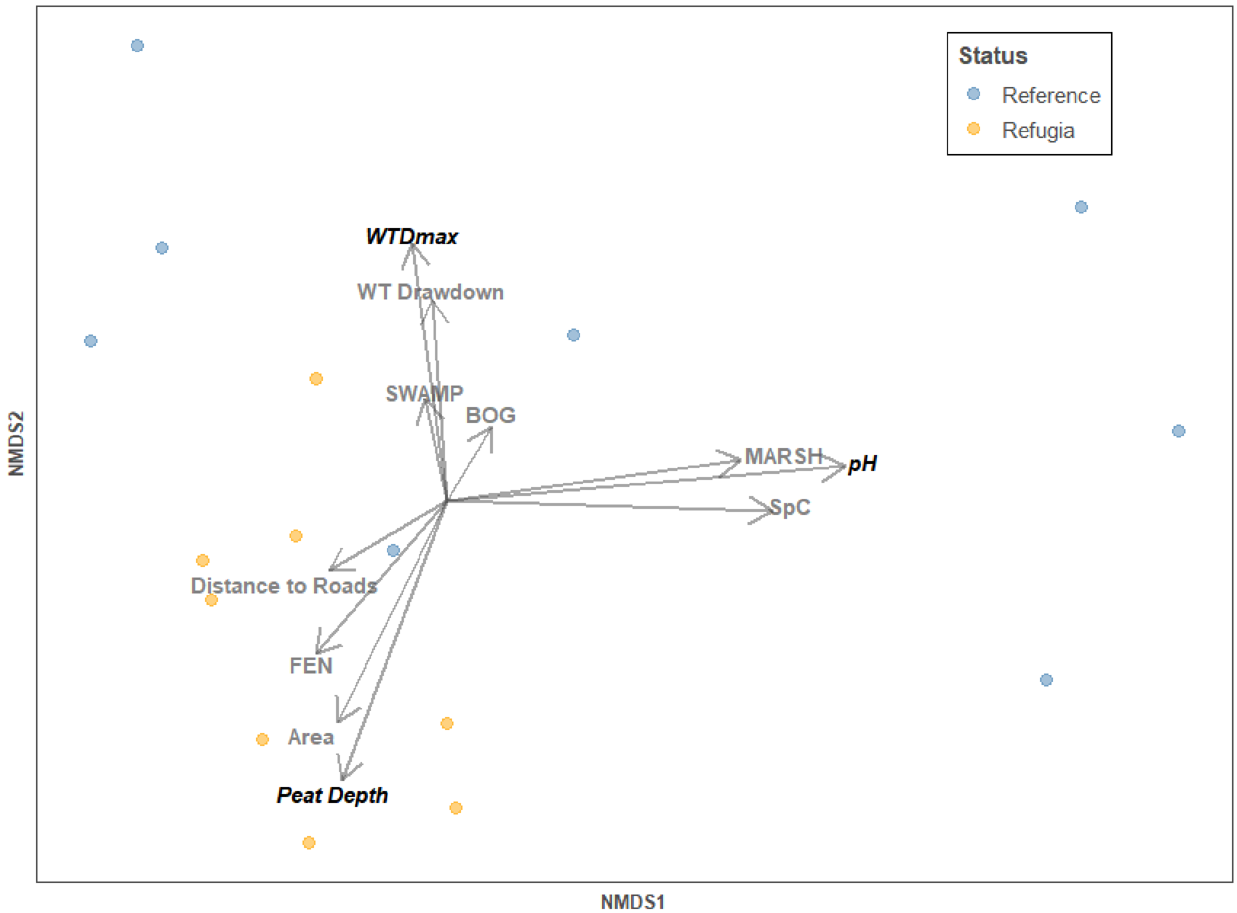

Figure 3-5: Environmental vectors influencing the vascular vegetation community composition of eight peatland fire refugia (orange circles), and eight reference sites (blue circles). Wetland class variables (SWAMP, BOG, MARSH, FEN) are binary (1 if a site belongs to the class, and 0 if not). WTD_{max} is the maximum water table depth during the 2021 growing season (May – October). Variables with a significant influence on site ordination are italicized: median peat depth (r^2 = 0.54, $p < 0.01$), pH (r^2 = 0.57, $p < 0.01$) and WTD_{max} (r^2 = 0.43, $p < 0.05$).

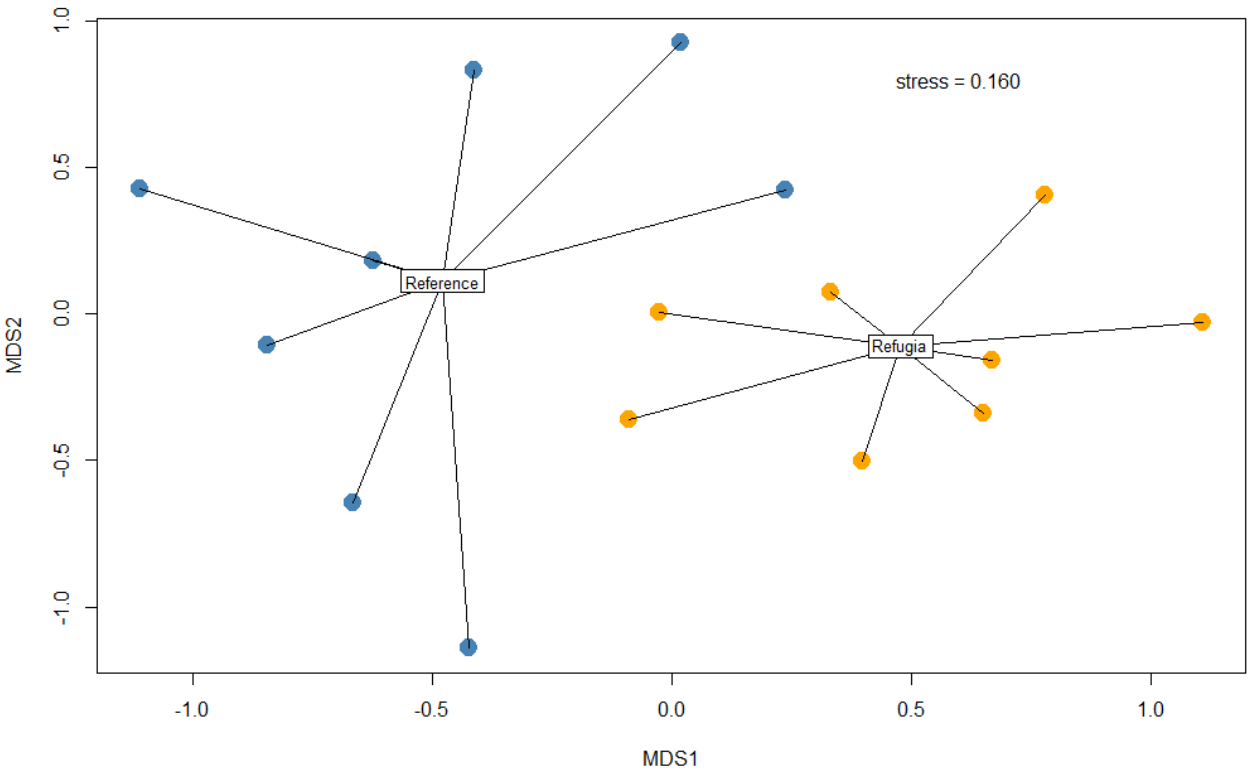

Figure 3-6: Non-metric multi-dimensional scaling (NMDS) plot of bryophyte vegetation composition (aggregated to site-level) in eight peatland fire refugia (orange circles) and eight reference sites (blue circles). The peatland fire refugia had a significantly different bryophyte composition compared to the reference sites ($F = 5.65$, $df = 1$, $p < 0.01$). The stress for the NMDS analysis was 0.160.

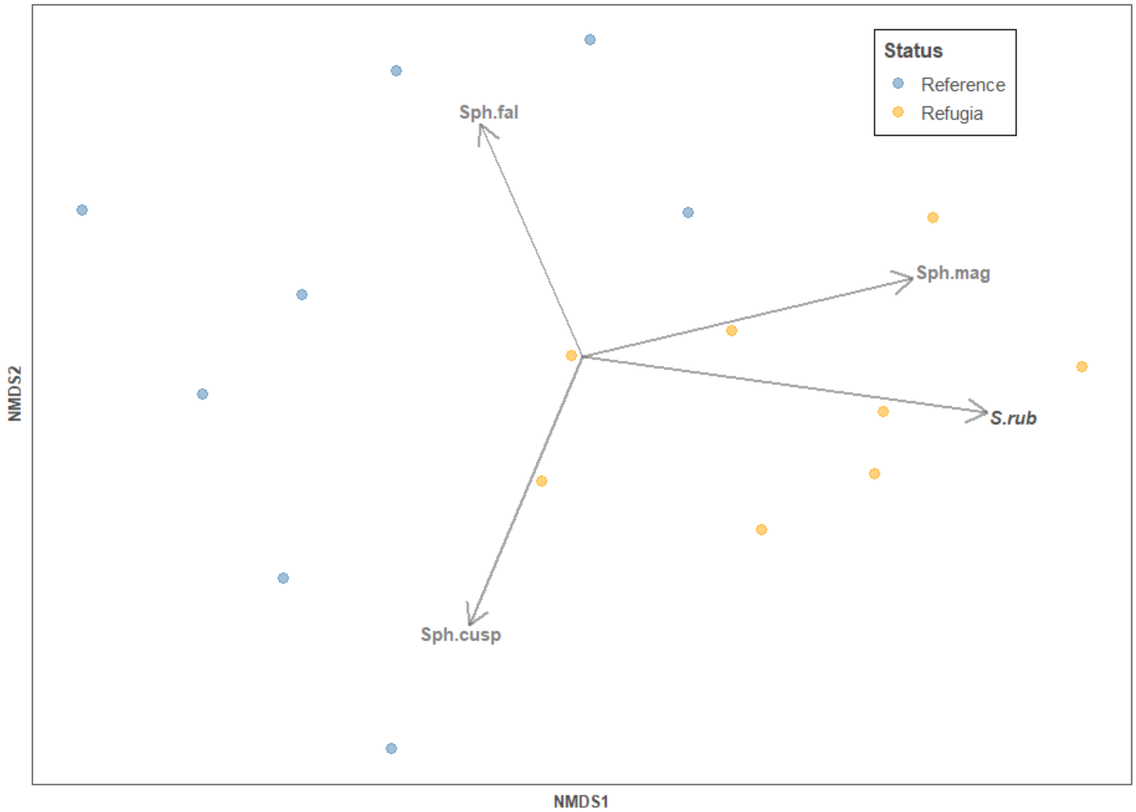

Figure 3-7: Intrinsic species plot for bryophyte species in eight peatland fire refugia (orange circles), and eight reference sites (blue circles). *Sphagnum rubellum* ("Sph.rub"), *Sphagnum fallax* ("Sph.fal"), *Sphagnum magellanicum* ("Sph.mag"), and *Sphagnum cuspidatum* ("Sph.cusp") were significantly correlated with site ordination on the plot (*Sphagnum rubellum*: $r^2 = 0.68$, $p < 0.001$; *Sphagnum fallax*: $r^2 = 0.41$, $p < 0.05$; *Sphagnum magellanicum*: $r^2 = 0.48$, $p < 0.05$; *Sphagnun cuspidatum*: $r^2 = 0.54$, $p < 0.01$).

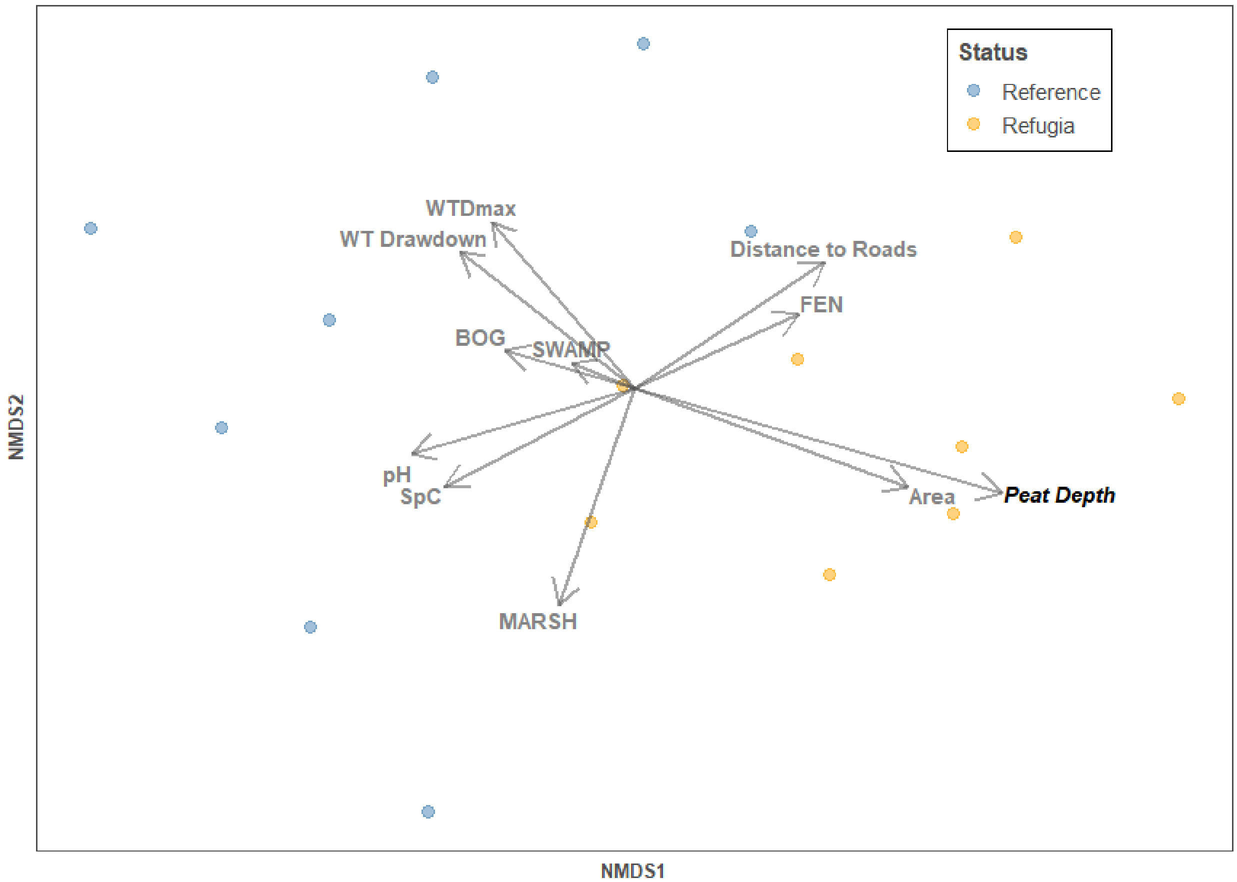

Figure 3-8: Environmental vectors influencing the bryophyte community composition of eight peatland fire refugia (orange circles), and eight reference sites (blue circles). Wetland class variables (SWAMP, BOG, MARSH, FEN) are binary (1 if a site belongs to the class, and 0 if not). WTDmax is the maximum water table depth during the 2021 growing season (May – October). Variables with a significant influence on site ordination are italicized: median peat depth (r^2 = 0.61, $p < 0.01$).

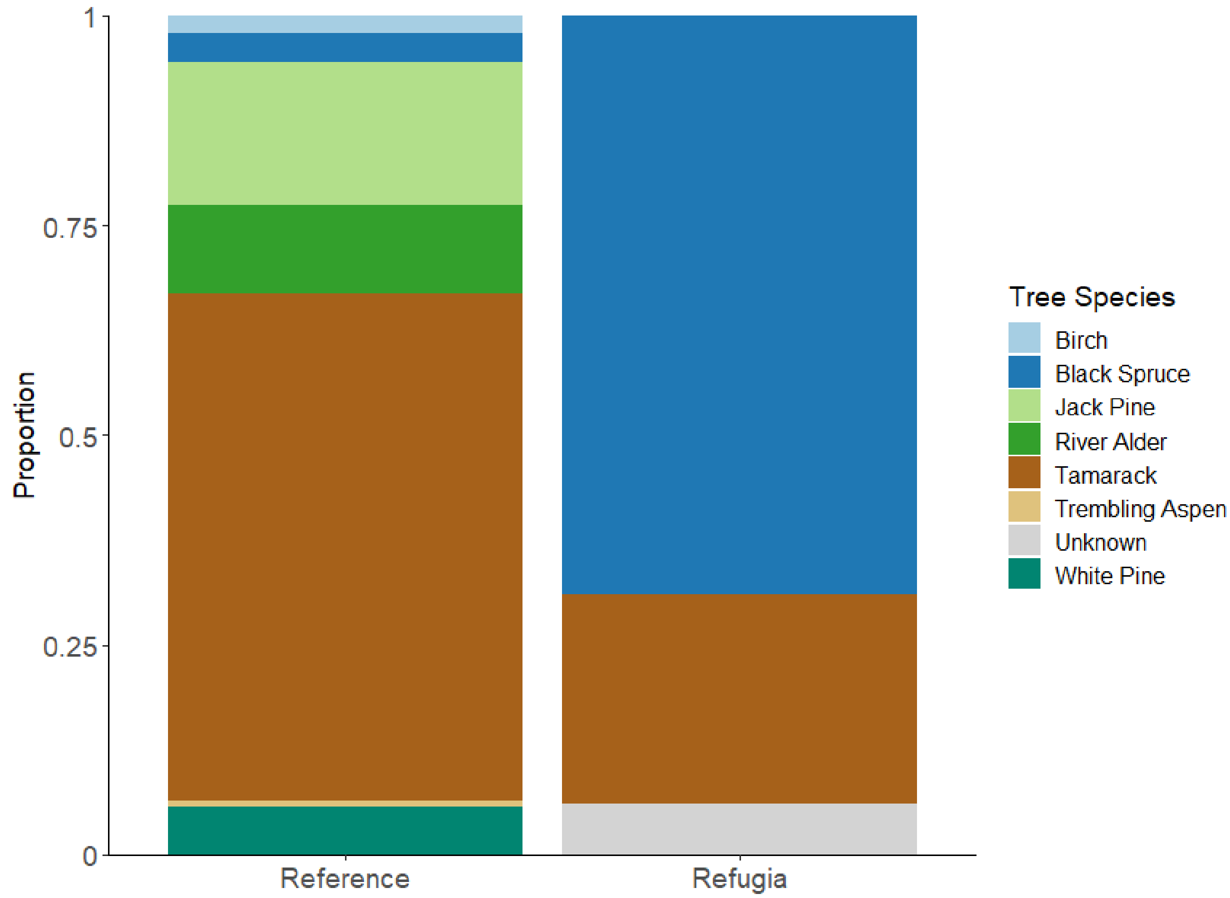

Figure 3-9: Proportional tree species composition for tree surveys conducted at three (lumped) peatland fire refugia, and three (lumped) reference sites outside the PS33 fire footprint.

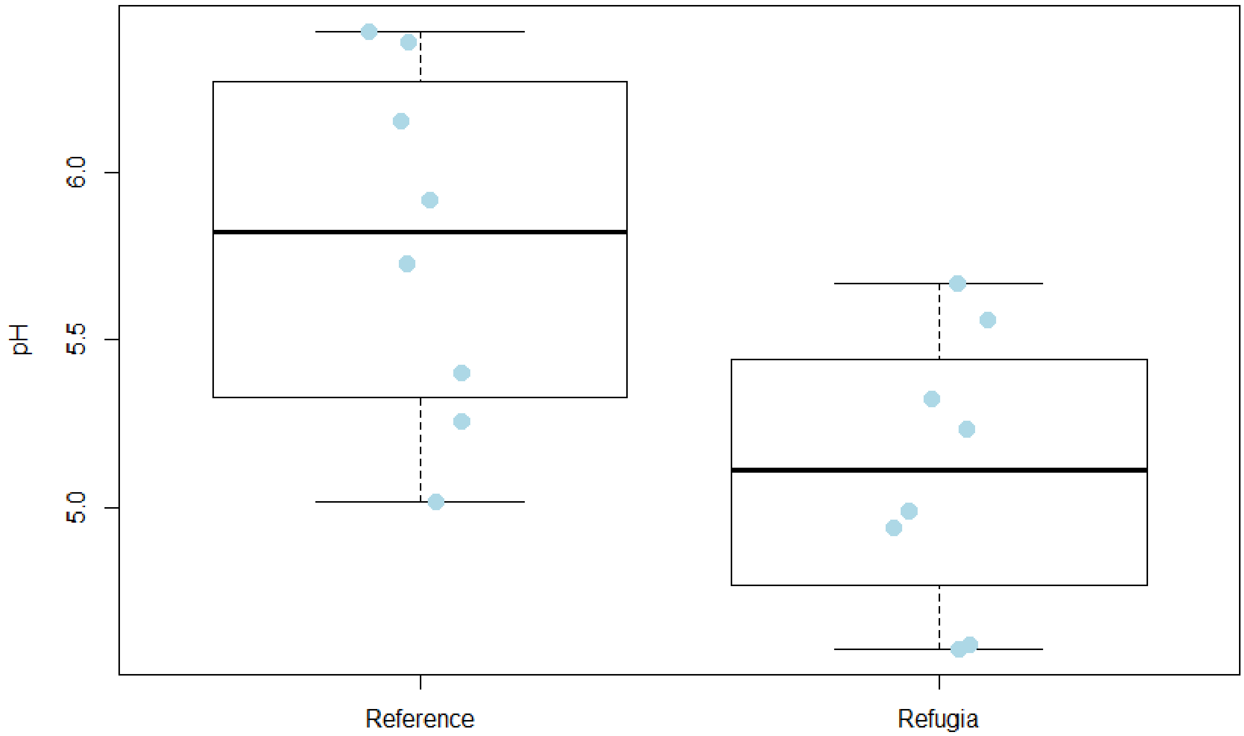

Figure 3-10: pH values measured within eight peatland fire refugia and eight reference sites in May 2022. Points (light blue circles) represent values at individual sites, where each value is an average of two measurements, taken approximately one week apart. The median pH is significantly lower in the refugia when compared to the reference sites ($U = 9$, $p < 0.01$).

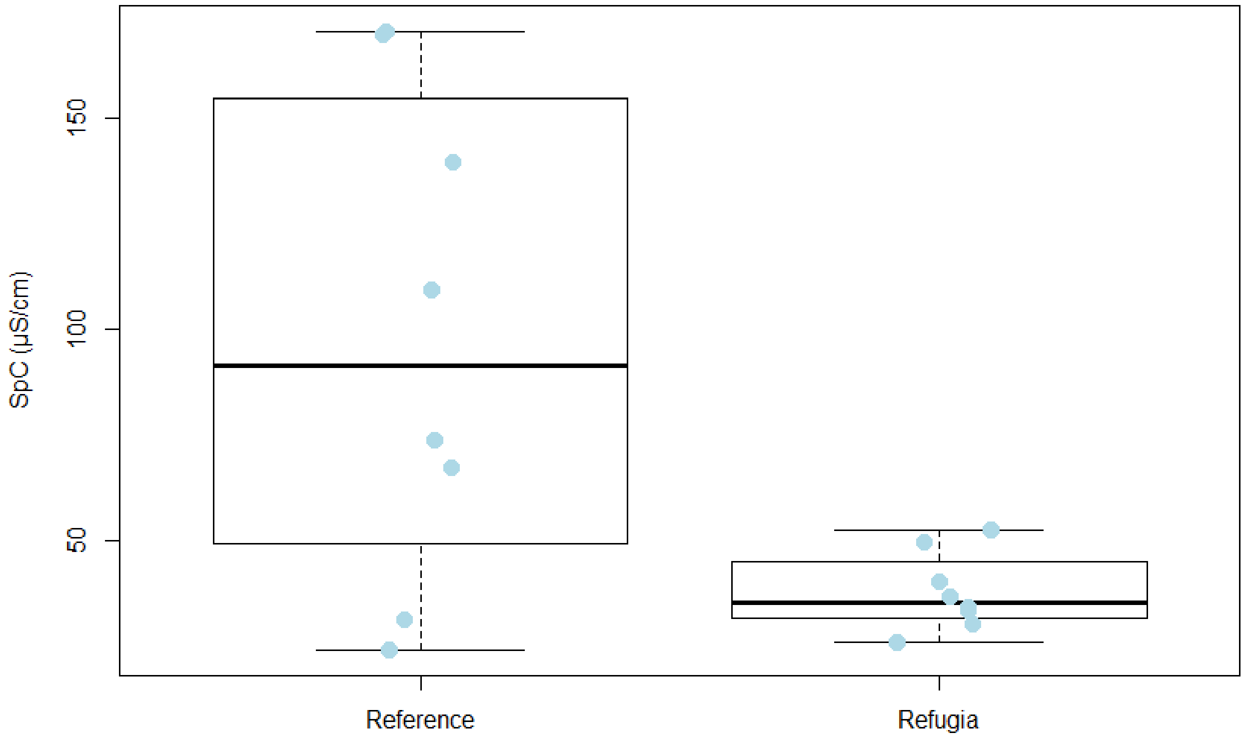

Figure 3-11: Specific Conductance (SpC; µS/cm) values measured within eight peatland fire refugia and eight reference sites in May 2022. Points (light blue circles) represent values at individual sites, where each value is an average of two measurements, taken approximately one week apart. The median SpC is significantly lower in the refugia when compared to the reference sites ($U = 14$, $p < 0.05$).

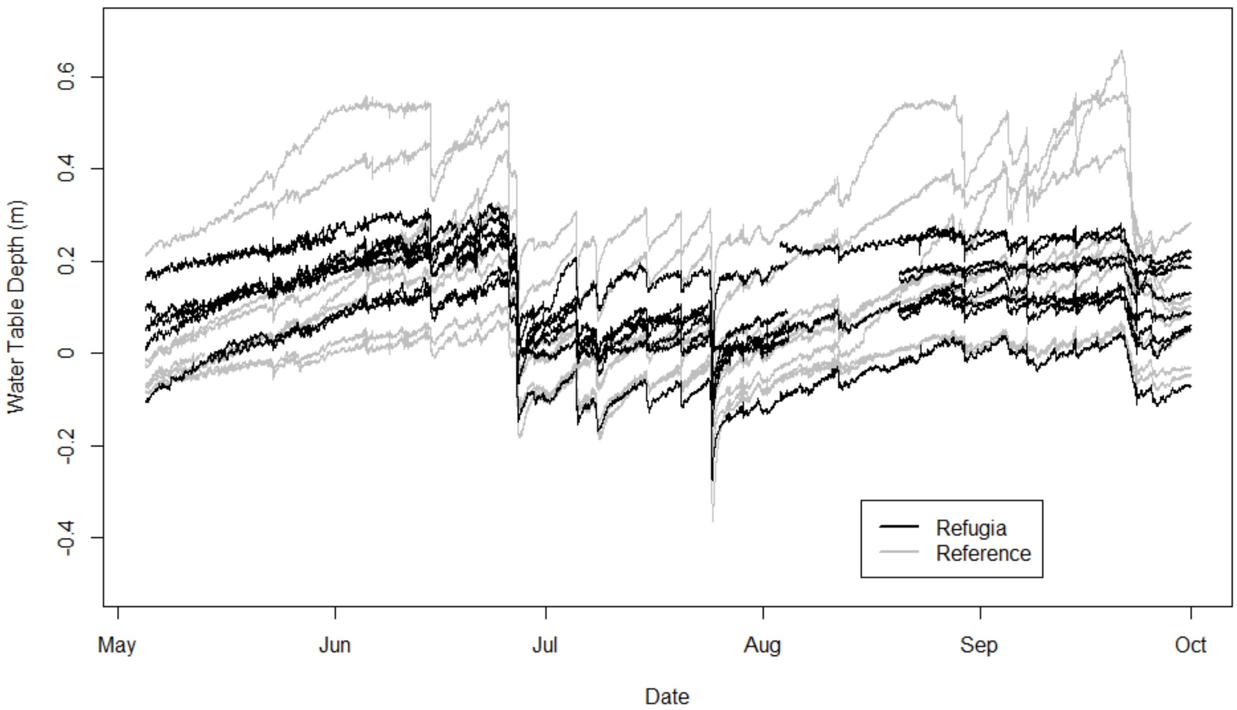

Figure 3-12: Average water table depth (m) referenced to the ground surface in eight peatland fire refugia (black lines) and eight reference sites (grey lines) during the 2021 growing season (May – October). Measurements were taken at 15-minute intervals using a logging pressure transducer. Negative values indicate water table positions above the ground surface, while positive values indicate water table positions below the ground surface.

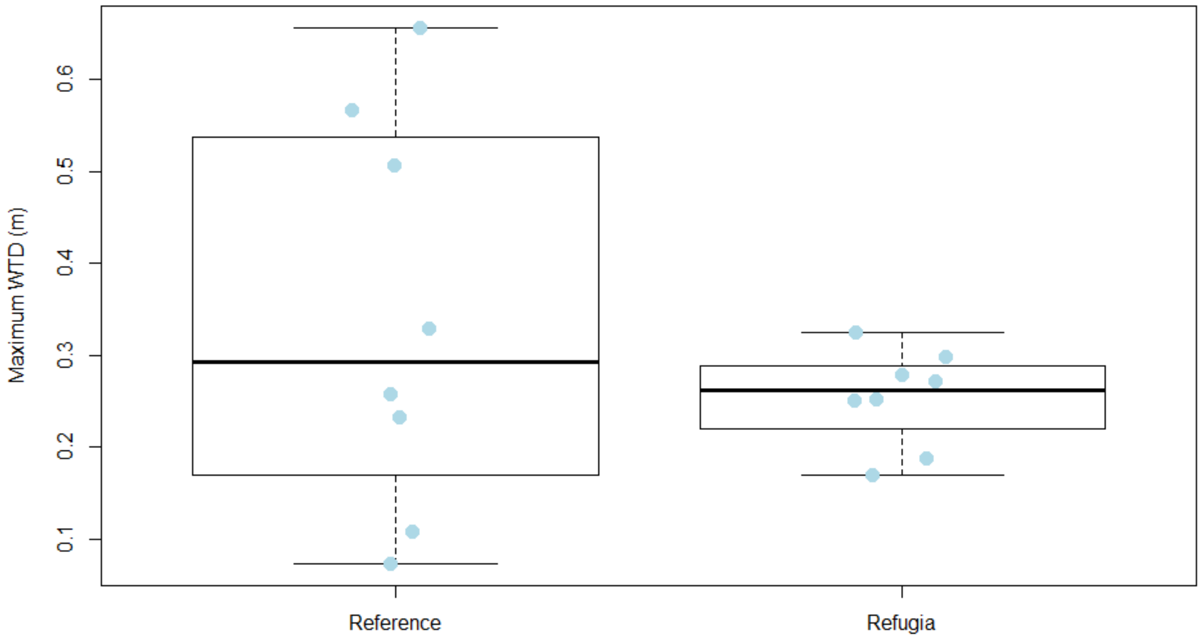

Figure 3-13: Maximum water table depths (WTD; m) observed in eight peatland fire refugia and eight reference sites during the 2021 growing season (May – October). The median maximum WTD was not significantly different between the refugia and reference sites (U = 26, p = 0.57).

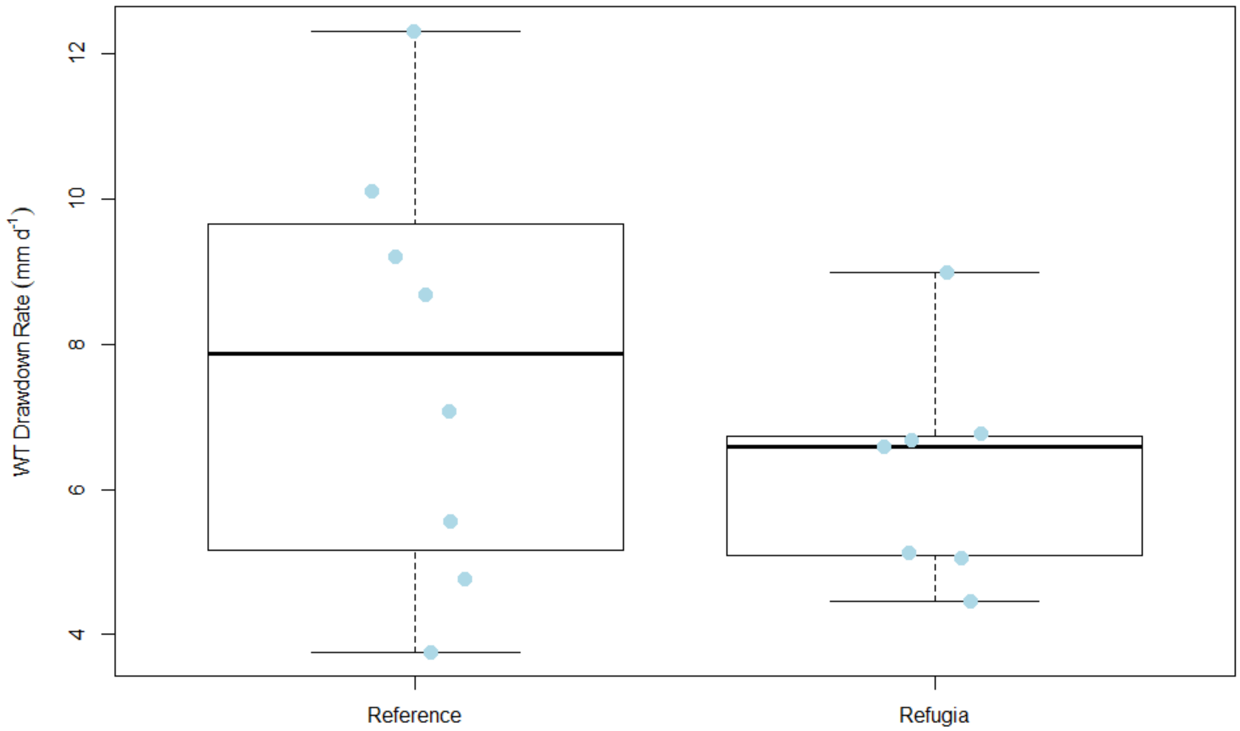

Figure 3-14: Water table (WT) drawdown rates (mm d^{-1}) observed in seven peatland fire refugia and eight reference sites during the longest rain-free period of the summer 2021 growing season (May 26 – June 5). The median WT drawdown rate was not significantly different between the refugia and reference sites ($U = 19$, $p = 0.34$).

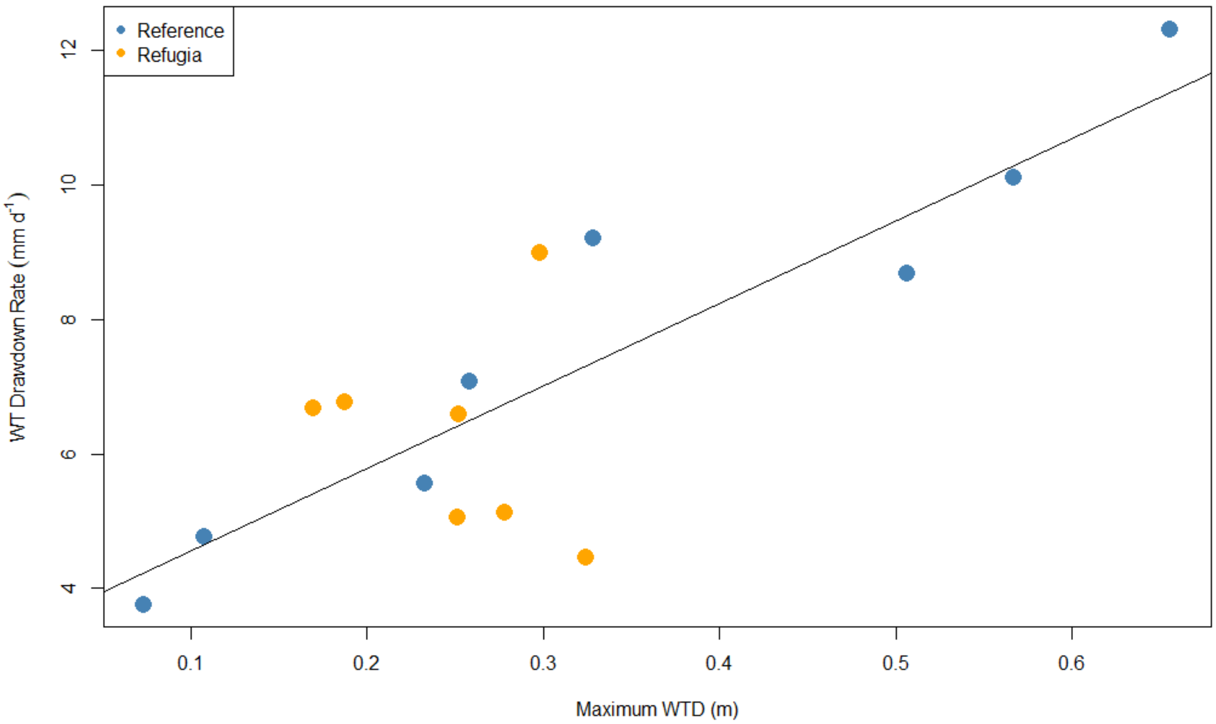

Figure 3-15: Regression (black line) between maximum WTD (m) and WT drawdown rate (mm d^{-1}) during the 2021 growing season at seven peatland fire refugia (orange circles) and eight reference sites (blue circles) (WT Drawdown = 12.22*WTD_{max} + 3.35, Adj. $R^2 = 0.661$, $p < 0.001$). Note that larger (positive) values of WT drawdown rate, indicate faster drawdown, while smaller values (closer to zero) indicate slower drawdown. Larger (positive) values of maximum WTD (m) indicate a WT position deeper below the peat surface.

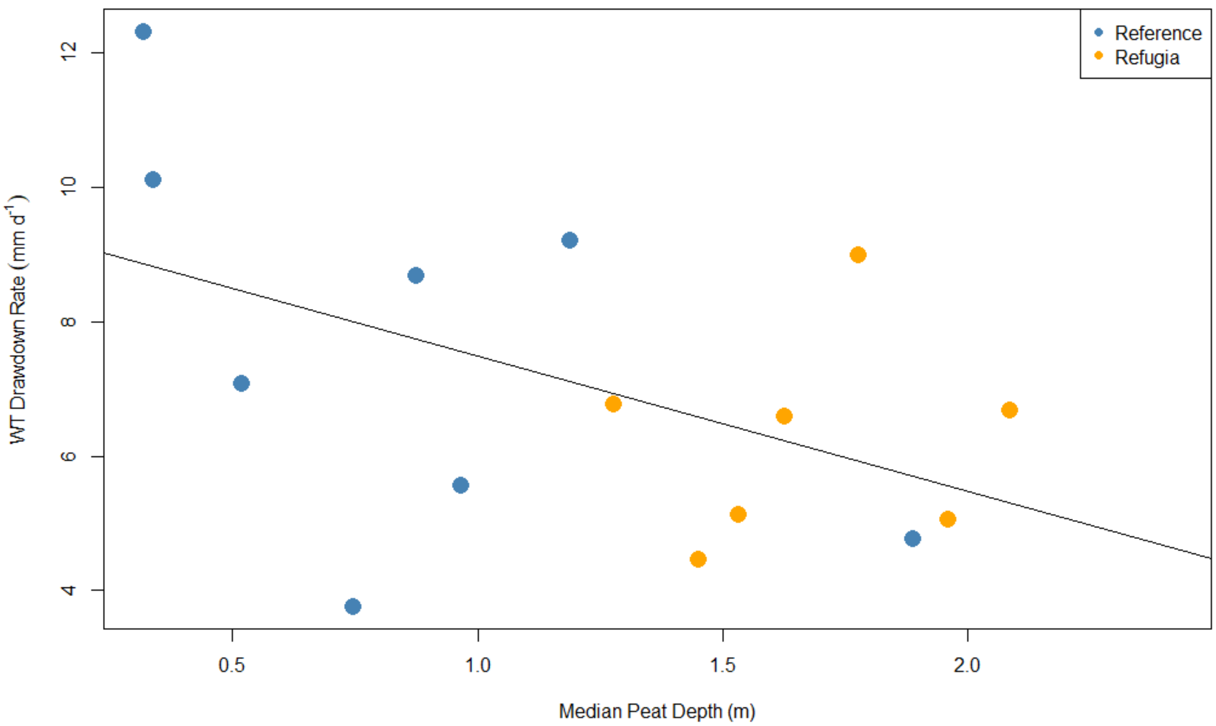

Figure 3-16: Regression (black line) between median peat depth (m) and WT drawdown rate (mm d-1) during the longest rain-free period (May 26 – June 5) of the 2021 growing season at seven peatland fire refugia (orange circles) and eight reference sites (blue circles) (WT Drawdown Rate= -2.01*Peat Depth + 9.50, Adj. R^2 = 0.184, p = 0.06). Note that greater (positive) values of WT drawdown rate, indicate faster drawdown, while smaller values (closer to zero) indicate slower drawdown.

3.7 Tables

Table 3-1: Indicator values (IndVal), specificity, and fidelity for vascular species in eight peatland fire refugia and eight reference sites. Values are ordered by descending IndVal, only species with an IndVal of 0.5 or greater were included for each category. P-Values were adjusted for multiple testing following the Benjamini and Hochberg (1995) procedure. The evidence category indicates the level of support for each species acting as an indicator for its respective category (*i.e.* refugia or reference), where an asterisk (*) indicates weak support ($0.05 < p < 0.1$), and a hyphen (-) indicates insufficient/no support ($p \geq 0.1$) (Muff et al., 2022).

	Indicator Value	Specificity	Fidelity	P-Value	Evidence
Refugia					
Kalmia polifolia	0.900	0.900	1.000	0.063	*
Vaccinium oxycoccos	0.815	0.931	0.875	0.118	-
Picea mariana	0.750	1.000	0.750	0.145	-
Drosera rotundifolia	0.707	0.808	0.875	0.265	-
Rhynchospora alba	0.625	1.000	0.625	0.265	-
Carex trisperma	0.617	0.823	0.750	0.424	-
Sarracenia purpurea	0.579	0.927	0.625	0.384	-
Rhododendron groenlandicum	0.500	1.000	0.500	0.424	-
Reference					
Pinus strobus	0.625	1.000	0.625	0.265	-
Dulichium arundinaceum	0.500	1.000	0.500	0.424	-

Table 3-2: Indicator values (IndVal), specificity, and fidelity for vascular species in eight peatland fire refugia and eight reference sites. Only species with a specificity of 1 were included for each category. P-Values were adjusted for multiple testing following the Benjamini and Hochberg (1995) procedure.

	Indicator Value	Specificity	Fidelity	P-Value
Refugia				
Picea mariana	0.750	1.000	0.750	0.145
Rhynchospora alba	0.625	1.000	0.625	0.265
Rhododendron groenlandicum	0.500	1.000	0.500	0.424
Oclemena nemoralis	0.375	1.000	0.375	0.778
Pogonia ophioglossoides	0.375	1.000	0.375	0.778
Reference				
Pinus strobus	0.625	1.000	0.625	0.625
Dulichium arundinaceum	0.500	1.000	0.500	0.424
Sparganium emersum	0.500	1.000	0.500	0.424
Ilex verticillata	0.375	1.000	0.375	0.778
Spiraea tomentosa	0.375	1.000	0.375	0.778

Table 3-3: Indicator values (IndVal), specificity, and fidelity for bryophyte species in eight peatland fire refugia and eight reference sites. Species with an IndVal of 0.25 or greater were included for each category. P-Values were adjusted for multiple testing following the Benjamini and Hochberg (1995) procedure. The evidence category indicates the level of support for each species acting as an indicator for its respective category (*i.e.* refugia or reference), where a double asterisk (**) indicates significant support ($p < 0.05$), and a hyphen (-) indicates insufficient/no support ($p \geq 0.1$) (Muff et al., 2022).

	Indicator Value	Specificity	Fidelity	P-Value	Evidence
Refugia					
Sphagnum rubellum	0.824	0.942	0.875	0.028	**
Sphagnum magellanicum	0.808	0.808	1.000	0.028	**
Sphagnum fuscum	0.250	1.000	0.250	0.603	-
Reference					
Polytrichum spp.	0.554	0.887	0.625	0.194	-
Sphagnum capillifolium	0.281	0.750	0.375	0.603	-
Dicranum scoparium	0.250	1.000	0.250	0.603	-
Amblystegium serpens	0.250	1.000	0.250	0.603	-

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Chapter 4: General Conclusion

Following a fire, unburned patches, called fire refugia, provide intact habitat in the short-term for flora and fauna (Riva et al., 2020), and act as post-fire regeneration nuclei over longer time scales (Hylander & Johnson, 2010; Downing et al., 2019), helping to kickstart post-fire recovery and enhancing the overall resilience of the landscape to wildfire. Wildfire activity and area burned are increasing in the North American boreal forest (Podur et al., 2002) due to an increasing wildland-urban interface and wildland-industry interface (Robinne et al., 2016; Wilkinson et al., 2021), as well as the influence of climate change on temperature (Gillet et al., 2004) and weather patterns, where blocking high pressure systems can lead to runaway drying and extreme fire risk (Marcias Fauria & Johnson, 2008). Hence, fire refugia will be important in preserving the ecological functioning of boreal landscapes under a changing fire regime. Within the boreal forest, peatlands have been indicated as having a high refugia potential (Stralberg et al., 2020) due to their inherent resilience to drought provided by multiple negative ecohydrological feedbacks (Waddington et al., 2015). In addition, previous studies have highlighted landscape position as a strong bottom-up control of fire refugia, where refugia are more frequently associated with wetlands and valley bottoms (Camp et al., 1997; Rogeau et al., 2018; Whitman et al., 2018) due to cold-air pooling (Wilkin et al., 2016), and runoff accumulation and groundwater inputs elevating fuel moisture in these areas compared to well-drained uplands (Holden & Jolly, 2011; Stralberg et al., 2020). Wetland classes with stable water table dynamics and sparse canopy cover (*e.g.* open fens) are disproportionately found in boreal fire refugia (Bourgeau-Chavez et al., 2020), and boreal peatlands were shown to lower fire severities in surrounding areas (Kuntzemann, 2021). However, not all peatlands avoid wildfire. In fact, wildfire is the dominant disturbance in northern peatlands, representing 97% of disturbances by area (Turetsky et al., 2002), threatening the functioning of these peatlands as carbon sinks. Thus, it is important to identify the drivers and dominant ecohydrological characteristics of fire refugia, with a focus on peatland fire refugia, in the North American boreal forest to inform conservation and fire management strategies, and set goals for restoring peatlands and peatland-dominated boreal landscapes to promote resilience to wildfire.

In Chapter 2, we present a first-pass model of the bottom-up controls of fire refugia in the Ontario Boreal Shield. We used multispectral Sentinel-2 imagery to calculate fire severity (RdNBR; Miller & Thode, 2007) and classify fire refugia and non-refugia within the 2018 Parry Sound 33 wildfire footprint, using standard thresholds (Meigs & Krawchuk, 2018). We trained a GBM model on a suite of biophysical variables (NDMI, TPI, SWI, CI, TCA, slope, Euclidean distance to water) derived from a DEM (Ontario Ministry of Natural Resources, 2017) and Sentinel-2 imagery. The GBM model had a high overall accuracy (cv AUC = 0.88) and a high sensitivity (81%). The NDMI and TPI had the greatest relative influence on refugia probability in the model (43% and 21%, respectively). Locations with a higher NDMI and lower TPI, which were associated with organic soil-filled depressions, had a higher refugia probability when compared to uplands (with a lower NDMI and higher TPI), corroborating results from studies on peatland-dominated landscapes in the western boreal (Whitman et al., 2018; Bourgeau-Chavez et al., 2020; Kuntzemann, 2021). Overall, we demonstrate the power of simple topographic characteristics for predicting fire refugia occurrence on this Ontario Shield rock barrens landscape, where fuel moisture, wildfire vulnerability, and refugia probability are governed by organic soil depth in depressions (Wilkinson et al., 2020), flow accumulation, and fill-and-spill runoff dynamics (Spence & Woo, 2003). Future work should assess the influence of fire weather on these bottom-up controls and compare the results to other fires and regions of the eastern Boreal.

Additional confirmation of results may also be achieved through in-situ surveys to evaluate the presence of ecohydrological indicators of fire refugia potential.

In Chapter 3, we conduct an initial assessment of the ecohydrological indicators of peatland fire refugia in the Ontario Boreal Shield. We surveyed the understorey vegetation composition, tree stand characteristics (tree height, DBH, BA, and species), and water table dynamics at eight peatland fire refugia and eight reference sites representative of the range of wetland types found within the fire footprint. We found that the understorey vascular vegetation and bryophyte community compositions within the peatland fire refugia differed significantly when compared to the reference sites, demonstrating that peatland fire refugia can be distinguished from other wetland types on this landscape based on their vegetation composition, allowing for the identification of potential fire refugia in advance of a wildfire. In particular, *Sphagnum rubellum* and *Sphagnum magellanicum* were identified as bryophyte indicator species, however, no vascular indicator species were identified. Vegetation composition differences were primarily driven by peat depth, groundwater pH, and maximum growing season WTD, where peatland fire refugia were deeper, and had a lower pH and a shallower maximum growing season WTD when compared to the reference sites. While not fully quantified in this study, the notable dominance of semi-serotinous black spruce (*Picea mariana*) suggests a longer fire interval in these peatland fire refugia compared to other wetlands and surrounding uplands, and is corroborated by past studies (Whitman et al., 2019; Stralberg et al., 2020; Baltzer et al., 2021) which note the presence of these species in areas which are less impacted by wildfire. Future work should determine the value of black spruce and other late-successional boreal tree species (*e.g.* white spruce (*Picea glauca*)) as reliable indicator species of potential peatland fire refugia in the North American boreal forest. Further study is also warranted to determine whether vegetation community composition can be used to distinguish potential fire refugia in other regions of the boreal forest and with larger sample sizes.

Overall, we provide the first ecohydrological characterization of fire refugia in the Ontario Boreal Shield. Simple biophysical factors, including fuel moisture and relative elevation, are strong predictors of fire refugia potential on this landscape, where a dichotomy exists between deep, peat-filled depressions with generally higher refugia probabilities, and thin soils on the rocky uplands which have a lower specific yield, experience rapid drying and have generally lower refugia probabilities; this builds on early evidence (Whitman et al., 2018; Bourgeau-Chavez et al., 2020; Kuntzemann, 2021) to support peatlands as not only fire refugia but climate change refugia as well, due to multiple negative ecohydrological feedbacks in deep peat deposits which make them inherently resilient to disturbance (Waddington et al., 2015; Stralberg et al., 2020). We present a statistical model for predicting fire refugia potential at large spatial scales using a suite of remotely sensed biophysical variables. This model may be useful for conservation planning and prioritizing fire management, with potential fire refugia treated as values for conservation and fire mitigation. At a smaller scale, and particularly at peatland sites with important ecological functionality and habitat value, we provide initial indicators of peatland fire refugia potential, including a high, stable water table, peat deeper than 2 m, the presence of *Sphagnum rubellum* and *Sphagnum magellanicum*, and a tree canopy (if present) dominated by black spruce (*Picea mariana*), and tamarack (*Larix laricina*). Together, these analyses provide an initial framework for the identification of peatland fire refugia in the Ontario Boreal Shield. Future work should focus on expanding these results to other areas, and further examining in-situ indicators of fire refugia potential which can be used to distinguish refugia using simple field-based methods.

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