Received: 29 August 2023 | Revised: 20 January 2024 | Accepted: 28 January 2024

DOI: 10.1111/gcb.17185

RESEARCH ARTICLE

Global Change Biology WILEY

Shortening fire return interval predisposes west-central Canadian boreal peatlands to more rapid vegetation growth and transition to forest cover

Emily Ann Jones[1] | **Laura Elizabeth Chasmer**[1] | **Kevin John Devito**[2] | **Christopher Dennis Hopkinson**[1]

[1]Department of Geography and Environment, University of Lethbridge, Lethbridge, Alberta, Canada

[2]Department of Biological Sciences, University of Alberta, Edmonton, Alberta, Canada

Correspondence
Emily Ann Jones and Laura Elizabeth Chasmer, Department of Geography and Environment, University of Lethbridge, Lethbridge, AB, Canada T1K 3M4.
Email: emily.jones3@uleth.ca and laura.chasmer@uleth.ca

Funding information
Alberta Environment and Parks, Grant/Award Number: DMR#1907M20; Mitacs, Grant/Award Number: IT13390; Canada Foundation for Innovation, Grant/Award Number: 32426; Natural Sciences and Engineering Research Council of Canada, Grant/Award Number: 2017-04362 and NETGP 548629-19; Western Economic Diversification Canada, Grant/Award Number: 000015316

Abstract

Climate change in northern latitudes is increasing the vulnerability of peatlands and the riparian transition zones between peatlands and upland forests (referred to as ecotones) to greater frequency of wildland fires. We examined early post-fire vegetation regeneration following the 2011 Utikuma complex fire (central Alberta, Canada). This study examined 779 peatlands and adjacent ecotones, covering an area of ~182 km^2. Based on the known regional fire history, peatlands that burned in 2011 were stratified into either long return interval (LRI) fire regimes of >80 years (i.e., no recorded prior fire history) or short fire return interval (SRI) of 55 years (i.e., within the boundary of a documented severe fire in 1956). Data from six multitemporal airborne lidar surveys were used to quantify trajectories of vegetation change for 8 years prior to and 8 years following the 2011 fire. To date, no studies have quantified the impacts of post-fire regeneration following short versus long return interval fires across this broad range of peatlands with variable environmental and post-fire successional trajectories. We found that SRI peatlands demonstrated more rapid vascular and shrub growth rates, especially in peatland centers, than LRI peatlands. Bogs and fens burned in 1956, and with little vascular vegetation (classified as "open peatlands") prior to the 2011 fire, experienced the greatest changes. These peatlands tended to transition to vascular/shrub forms following the SRI fire, while open LRI peatlands were not significantly different from pre-fire conditions. The results of this study suggest the emergence of a positive feedback, where areas experiencing SRI fires in southern boreal peatlands are expected to transition to forested vegetation forms. Along fen edges and within bog centers, SRI fires are expected to reduce local peatland groundwater moisture-holding capacity and promote favorable conditions for increased fire frequency and severity in the future.

KEYWORDS
airborne lidar, change detection, climate change, remote sensing, wetland monitoring, wildland fire

1 | INTRODUCTION

The Canadian boreal biome spans an area of ~5.5 million km^2, more than half the area of Europe. The biome is characterized by a heterogeneous mix of upland forests, lakes, and wetlands with significant global importance for modulating the impacts of climate change (Bonan et al., 1992; Helbig, Waddington, Alekseychik, Amiro, Aurela, Barr, Black, Blanken, et al., 2020; Helbig, Waddington, Alekseychik, Amiro, Aurela, Barr, Black, Carey, et al., 2020), driving weather (Davis et al., 1997; Pennypacker & Wood, 2023), and provision of natural resources (Brandt et al., 2013). Peatlands, defined as wetlands with thick layers of dead, organic soils and mosses, comprise ~22% of the boreal biome (Thompson et al., 2016) and are distributed between upland forests and in localized topographically low-lying areas. The effects of the successional history of peatlands, which typically develop from the accumulation of organic material within shallow ponds via terrestrialization or in upland areas via paludification (Bauer et al., 2003), combined with cool, moist climate (Vitt, 2006), and growth of mosses, shrubs, and trees over time (Vitt, 2006) contribute to their ecological and climatological importance. As such, peatlands are considered climate change (Stralberg et al., 2020) and fire (Thompson, Parisien, et al., 2017) refugia largely due to the moisture retention of mosses and typical near-surface water table (Waddington et al., 2015). These characteristics limit the spread of fire (Turetsky et al., 2004) and act as a long-term atmospheric carbon sink due to the reduced decomposition of peat as a result of cold and anoxic conditions (Wang et al., 2015). The amount of carbon contained within peatlands is extensive, representing the Earth's largest terrestrial C store (Immirzi et al., 1992; Roulet, 2000; Turetsky et al., 2015).

However, the Canadian boreal biome, including peatlands, is changing rapidly (Gauthier et al., 2015). Climatic changes over recent decades have resulted in significant warming and drying of the boreal biome, especially in parts of western Canada (Price et al., 2013), and these changes are predicted to intensify over the next century (Helbig, Waddington, Alekseychik, Amiro, Aurela, Barr, Black, Blanken, et al., 2020; Helbig, Waddington, Alekseychik, Amiro, Aurela, Barr, Black, Carey, et al., 2020; Flannigan et al., 1998). Warmer air temperatures increase evaporative losses from peatlands to a greater extent than forests (Helbig, Waddington, Alekseychik, Amiro, Aurela, Barr, Black, Blanken, et al., 2020; Helbig, Waddington, Alekseychik, Amiro, Aurela, Barr, Black, Carey, et al., 2020), transitioning deep organic soils and mosses into dry fuels for wildland fire (Kettridge Turetsky et al., 2015; Nelson et al., 2021; Wilkinson et al., 2019; Wilkinson, Tekatch, et al., 2020; Wilkinson, Verkaik, et al., 2020). The potential for fire spread is further increased during prolonged warm, dry (severe fire weather) conditions, as determined by the Canadian Fire Weather Index (Thompson et al., 2019; Waddington et al., 2012). Upland forests are even more susceptible to rapid fire spread during these periods due to evapotranspiration, insect disturbances, and mortality (Frelich & Reich, 2010; Michaelian et al., 2011) with variations depending on species and local hydroecology (e.g., Pappas et al., 2018). This, combined with a lowering of the water table (Kettridge Turetsky et al., 2015; Wilkinson, Tekatch, et al., 2020; Wilkinson, Verkaik, et al., 2020), enhances fire spread and fire intensity (energy; Walker et al., 2020; Whitman et al., 2022) and may alter fire regimes (Whitman et al., 2019).

The boundary between peatlands and uplands, referred to here as the peatland ecotone (Figure 1), may be especially sensitive to wildland fire, including the potential to enhance fire spread into peatlands associated with changing moisture (Lieffers & Macdonald, 1990; Nelson et al., 2021; Wilkinson et al., 2018), afforestation (Waddington, Morris, Kettridge, Granath, Thompson, & Moore, 2015, Waddington, Morris, Kettridge, Granath, Thompson, Moore, & Sveriges, 2015; Weltzin et al., 2003), and post-fire hydro-ecological impacts (Jones et al., 2022). These include but are not limited to variable evaporative losses and water repellency feedbacks (Kettridge et al., 2014; Thompson et al., 2014), changes in hummock-hollow elevational variability (Benscoter & Vitt, 2008), and post-fire species recovery (Jones et al., 2022). Peatland ecotones can be characterized by a range of both terrestrial/hydrophobic and aquatic/hydrophilic vegetation species, including feather mosses (e.g., *Pleurozium schreberi*), *Rhododendron groenlandicum*, willows (e.g., *Salix borealis*), and low-density *Picea mariana* as these transition from uplands into peatlands along with reduced depth of organic soils and increasing depth to water table from peatland to upland forests (Dimitrov et al., 2014; Elmes et al., 2018; Mayner et al., 2018).

Peatland ecotones could exacerbate fire effects on peatlands by altering post-fire soil bulk density (Thompson & Waddington, 2013) and enhancing water table fluctuations (Lukenbach et al., 2017), increasing the potential for deep, smoldering fire (e.g., Hokanson et al., 2016). Furthermore, prolonged dry periods result in expansion of forest–peatland ecotones into peatlands, increasing terrestrial vegetation species, such as feather mosses, typically found in upland forests (Chasmer & Hopkinson, 2017; Holmgren et al., 2015; Lieffers & Macdonald, 1990; Mayner et al., 2018). These may increase the drying of peatland boundaries due to transpiration losses, enhancing shading, and feather moss expansion (Bisbee et al., 2001; Kettridge et al., 2013). Feather mosses and some *Sphagnum* species are especially prone to deep smoldering fires in peatlands (reviewed by Nelson et al., 2021) and especially along peatland ecotones (Elmes et al., 2018; Wilkinson et al., 2019).

Enhanced potential for fire spread into peatlands from surrounding upland forests via surface moisture reduction and vegetation changes along the ecotonal boundaries may alter current and future fire regimes within the Canadian boreal biome, with unknown impacts to boreal peatlands in the years following fire. Such changes are essential to quantify because these may indicate resilience or a shift to a new ecosystem state (e.g., Gibson et al., 2018). Boreal peatland fire return intervals (years between fires) typically range from 100 to 130 years (Turetsky et al., 2004; Wieder et al., 2009); however, over the last two decades, peatland fire return intervals have shortened, with some fires occurring within 20–100 years of a previous fire. While few studies have examined boreal peatland fire

FIGURE 1 Photographs taken during the 2019 field campaign of (a) an unburned peatland, looking toward the center of the peatland from approximately one-third distance (of the total width) into the peatland; (b) a burned peatland following the Utikuma complex fire in 2011, looking into the center of the peatland from about one-third distance toward the unburned upland forest on the other side of the peatland; (c) an unburned upland forest to peatland ecotone, looking from approximately the upland forest edge through the ecotone and into the peatland beyond; (d) a narrow, burned ecotone (represented by sedges in the foreground) with the burned peatland beyond the ecotone.

return intervals, Sutheimer et al. (2021) found that historic return intervals over the last several centuries for hemi-boreal peatlands south of the Canadian/US Great Lakes could occur as frequently as 7 and 31 years between low-intensity fires. Proximal drivers may also influence the propensity for shortening return intervals and post-fire effects (Turetsky et al., 2015). For example, Kettridge Turetsky et al. (2015), Whitman et al. (2019), and Grzesik et al. (2022) hypothesized a shift to more deciduous species, while Bourgeau-Chavez et al. (2020) show that bogs are more susceptible to greater burn severity. However, fens transition to greater susceptibility for burning when fire weather is severe.

Bogs receive moisture from precipitation, increasing bog hydrology coupling to fire weather conditions (Glaser et al., 1997). In peatland/upland ecotones, fluctuating hydraulic gradients (Bhatti et al., 2006; Goud et al., 2018) can enhance rapid temporal and spatial changes in vegetation species and structures associated with moisture retention at the peatland edge (Jones et al., 2022; Yarrow & Marín, 2007). Therefore, peatland shape and proximity to upland conifer versus deciduous forests are hypothesized to enhance the propagation of some species, including *Salix* spp. (Dimitrov et al., 2014; Mayner et al., 2018) into the ecotonal boundary during dry phases (e.g., Chasmer & Hopkinson, 2017). Fens, which are at an earlier successional phase to bogs within the southern and central Canadian boreal biome, are typically connected to groundwater discharge resulting in moderated water table fluctuations which may enhance resilience to fire in moderate fire weather scenarios (Bourgeau-Chavez et al., 2020). Therefore, the ratio of the peatland perimeter to area, where larger peatland areas relative to longer perimeter (which can enhance ecotonal edge effects) and adjacent species may alter the potential for a high-intensity fire, changes in fire regime, and post-fire effects (Mayner et al., 2018).

In this study, we consider the impacts of short versus typical fire return intervals (herein short return interval [SRI] fire and typical or long return interval [LRI]) on post-fire peatland and ecotone recovery within >700 peatlands using multi-temporal airborne lidar data in the Boreal Plains, Alberta Canada. We hypothesize that the recovery of peatlands with shortening fire return intervals will be directly related to hydrological functions as these vary with surficial geomorphological groups, peatland shape, and the potential for vegetation encroachment from proximal land covers (Ferone & Devito, 2004; Hokanson et al., 2018). To address this hypothesis, three questions are examined:

1. As shrub vegetation can be a proxy indicator of shifting hydrology, what are the rates of post-fire regeneration of peatland vegetation following SRI versus LRI fires within the broader geomorphological groups found across the study area?
2. Are peatlands that undergo shortening fire return intervals shifting toward upland vegetative structural characteristics compared with peatlands that experience longer return interval fires?
3. How do rates of change vary within and between peatlands burned by SRI versus LRI fire associated with environmental gradients (peatland class, shape, and proximity to adjacent upland forest species)?

This is the first study to consider the broad spatial and temporal range of factors influencing post-fire ecotonal and peatland regeneration following SRI versus LRI fires across hundreds of peatlands in western Canada. This provides a more complete assessment of potential future peatland and landscape evolution associated with changing feedback between climate, wildfire, and peatland eco-hydrology.

2 | MATERIALS AND METHODS

2.1 | Study area

The study was conducted in the Boreal Plains ecozone within the Utikuma Region Study Area (URSA) (56.06°N; −115.42°W), ~100 km North of Slave Lake, Alberta, Canada (Figure 2). The study region is characterized by low elevational relief with three spatially distinct geomorphological landforms: hummocky clay-rich glacial till moraine, fine-textured glacio-lacustrine plains, and coarse-textured glacial–fluvial and aeolian deposits (Devito et al., 2016; Hokanson et al., 2019; Thompson, Mendoza, et al., 2017). Within these landforms, exists a mosaic of forested uplands and wet lowlands connected via upland–wetland ecotones. Peatlands (bogs and fens) cover about 60% of the glacio-lacustrine plain area and 20%–30% of fine-textured hummock moraine landforms.

Wildland fires burned the area in 1956 and 2011 (Alberta Wildfire, 2018), with overlapping burn scars covering an area of about 12 km^2. Peatlands within this overlapping area, therefore, had a fire-return interval of 55 years (SRI fire), where the two fires occurred in nearly half of the time frame typical for this area (Benscoter, 2006; Benscoter & Vitt, 2008; Tarnocai et al., 2011; Turetsky et al., 2002). The area burned in 2011 is considered LRI and is more typical of the fire regime of this area, which has been monitored since the 1940s.

Upland forests include mostly aspen (*Populus tremuloides*) and balsam poplar (*Populus balsamifera*) deciduous trees, sometimes with mixed white spruce (*Picea glauca*) conifers (Devito et al., 2016; Petrone et al., 2007). Peatlands are dominated by black spruce (*P. mariana*), bog birch (*Betula glandulosa*), and Labrador tea (*R. groenlandicum*), with *Sphagnum* moss species (Devito et al., 2016; Jones et al., 2022). Post-fire ecotones include alder species (*Alnus viridis* and *Alnus crispa* (Ait.) Pursh.), paper birch (*Betula paperifera*), and willow species (*Salix* spp.) (Figure 1, Jones et al., 2022).

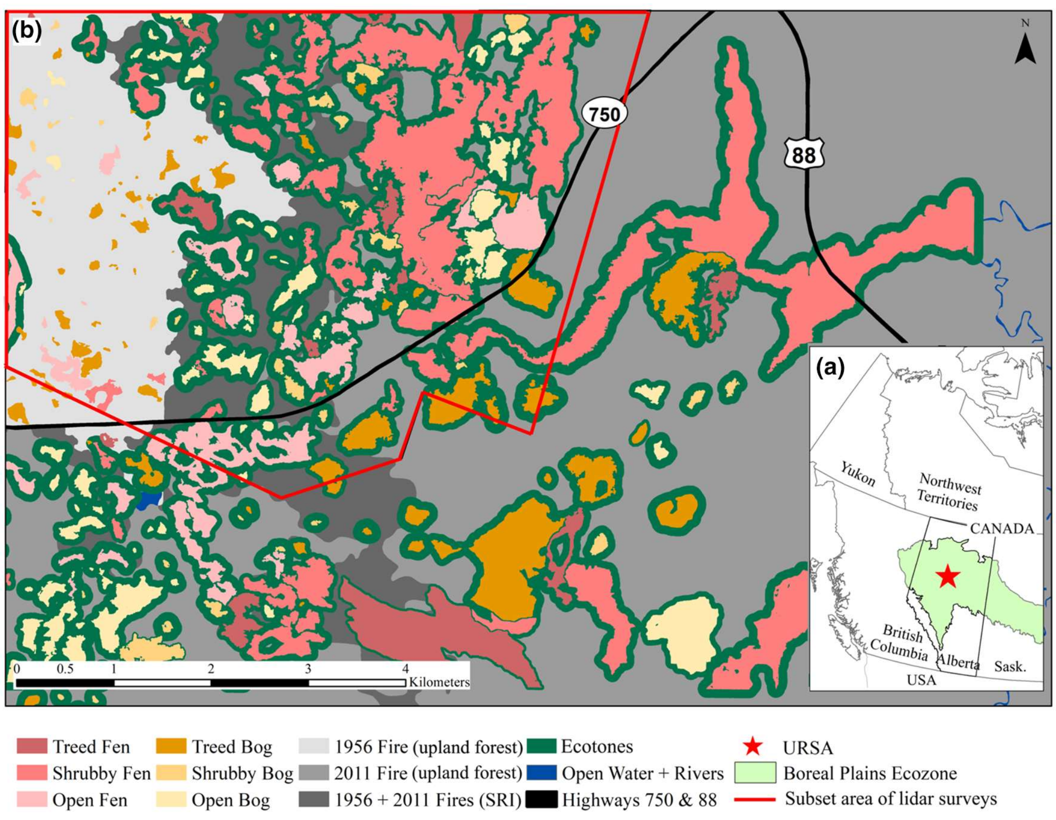

FIGURE 2 (a) The Utikuma Region Study Area (URSA) is located within the Boreal Plains Ecozone approximately 100 km north of Slave Lake, Alberta, Canada (inset). (b) Polygons indicate peatland class and form found within the areas burned by fire in 1956 only, 2011 only (LRI), and in the overlap of both fires (SRI). Ecotones exist along the perimeter between peatlands and upland forests (illustrated in dark green), where upland forests occur in gray areas. Wetland classification was modified from Chasmer et al. (2016) to include wetland class (bogs and fens, but also marshes, swamps and shallow open water; not included in this study) and form (predominantly open, with sedges and mosses, shrubby, or treed; Chasmer et al., 2024).

2.2 | Lidar data and processing

Six airborne lidar surveys were collected over 17 years (Table 1). Lidar data were collected pre-fire in the summer of 2002 (Hopkinson et al., 2005) and 2008 (Chasmer et al., 2016; Montgomery et al., 2019), and post-fire in July to early August 2011, 2016, 2018, and 2019 (methods described in Hopkinson et al., 2013; data available via Hopkinson et al., 2024). A post-processed kinematic survey was collected using a Global Navigation Satellite System (GNSS) receiver along Highways 88 and 750 (Figure 2) to calibrate and align all lidar point clouds to a common elevation control.

Post-processing of lidar data included classification of ground and non-ground returns, quality control, and removal of isolated points using TerraScan (TerraSolid Inc., Finland). While lidar data provide an opportunity to characterize the 3D structure of vegetation and underlying ground surface elevation, we focused on canopy height models (CHM) to determine canopy height changes in the years following fire. This reduces the potential for uncertainty associated with the frequency and distribution of laser returns due to survey configuration (Hopkinson et al., 2006, 2016; Næsset, 2009). A digital elevation model (DEM) was created from ground-classified returns at 1 m pixel resolution and was used to normalize vegetation height derived from a digital surface model (DSM) relative to the DEM. The DSM includes canopy and ground elevation, calculated using the maximum height of lidar returns within 1 m^2 cells from the canopy. These were then resampled to 5 m^2 by taking the average maximum height (Næsset, 2004; Wulder et al., 2012). CHMs were produced for each year of lidar data collection to determine changes in vegetation height. To remove the influence of remaining burned/standing tree stems from the CHM, these were segmented within each post-fire CHM dataset by removing cells taller than the regenerating vegetation.

2.3 | Peatland and ecotone classification and identification

Peatlands were identified using an enhanced wetland classification map modified from Chasmer et al. (2016) to include class (bog and fen) and form (open/sedge/moss, shrub, and treed) peatlands (Figure 3; classification available via Chasmer et al., 2024). Peatland class and form were identified in 2008 (pre-fire) remotely sensed data using a hillshade model from the 2008 lidar DEM, the CHM, and high-resolution optical SPOT imagery. Two hundred and twenty-one fens and 558 bogs were classified, and 30 were verified for classification accuracy from site visits, with 81% of peatlands correctly classified. About 18% of the 1956 burned area, 31% in the 2011 LRI fire, and 36% in SRI (burned both in 1956 and 2011) fire areas were classified as peatlands (Figure 2).

Ecotone boundaries are challenging to identify and map using optical remotely sensed data due to vegetation species and structural similarities between forested uplands and peatlands. To address this problem, ecotones were classified using a systematic "criteria-based" spatial analysis approach based on the topographic position of the elevation (using the DEM) and canopy height characteristics. Peatland maximum, mean elevation, and vegetation height were determined based on all cells for each peatland using the 2008 (pre-fire) lidar dataset. To determine the area of ecotones, 5 m incremental rings (or buffers) were created, extending from each peatland edge as elevation increased, transitioning from peatlands to uplands. The maximum and mean elevation and canopy height were calculated for each 5 m buffer ring to a maximum distance of 100 m (to reduce the potential to overlap with other peatlands). These were then differenced from those calculated for the peatland and that of the next closest buffer ring to the peatland (Figure 3). Based on this iterative process, a decision criterion was used to identify: (i) the maximum difference between elevation/canopy height from that found within the peatland, and (ii) the distance at which the differences between zones were minimized, indicating that elevation had become "flatter," as is expected when transitioning from ecotone to upland forest boundary (Figure 3; R Core Team, 2021). Based on this method, an adaptive transition distance was defined for each peatland on a per (5 m × 5 m) pixel basis (ecotonal classification available via Chasmer et al., 2024). This method assumes that the greatest changes occur along the peatland margins where fluctuating water tables and the combination of organic and mineral soil textures create a dynamic nutrient-rich environment (Hokanson et al., 2016; Jones et al., 2022; Lukenbach et al., 2015).

2.4 | Quantifying vegetation change using time-series lidar data

In this study, mapped vegetation height changes across the peatland and return interval strata were then determined between years by grouping and subtracting the vegetation CHM changes into the early post-fire period (from 2011 to 2016), the latter period (from 2016 to 2019), and the entire early post-fire period (from 2011 to 2019), each divided by the number of years to estimate a rate of growth per year (Hopkinson et al., 2006, 2016; Næsset, 2009). To determine the area proportion of vegetation change per peatland, pixel difference >0.15 m was included (as 0.15 m is greater than the vertical error of the point cloud elevation data; Andersen et al., 2006; Falkowski et al., 2008; Hodgson & Bresnahan, 2004). Changes and areas of vascular vegetation, especially in areas of expansion and reduction of shrub and tree heights within strata, were also examined by considering entire peatland and ecotonal areas. Here, the areas of vegetation >0.5 m in height between 2008 (pre-fire) and 2019 (8 years post-fire) were compared within each SRI versus LRI burned areas, bogs, and fens, and within ecotones. To determine the distribution of vegetation height characteristics (R Core Team, 2021), ~6000 points were

TABLE 1 Specifications of lidar data acquisitions (all systems by Teledyne Optech, Canada).

Year	Lidar systems	Flight altitude (m above ground level)	Repetition rate (kHz)	Scanner frequency (Hz)	Scan half angle (°)	Average point density (points per m^2)	Wavelength (nm)
2002	ALTM 2050	<1200	50	36	±16	2.2	1064
2008	ALTM 3100	1400	70	22	±25	1.8	1064
2011	ALTM 3100	1800	50	23	±23	1.3	1064
2016	Titan	1100	225	32	±32	3.1	532, 1064, 1550
2018	Titan	1000	300	32	±25	4.4	532, 1064, 1550
2019	Titan	1000	300	32	±25	5.9	532, 1064, 1550

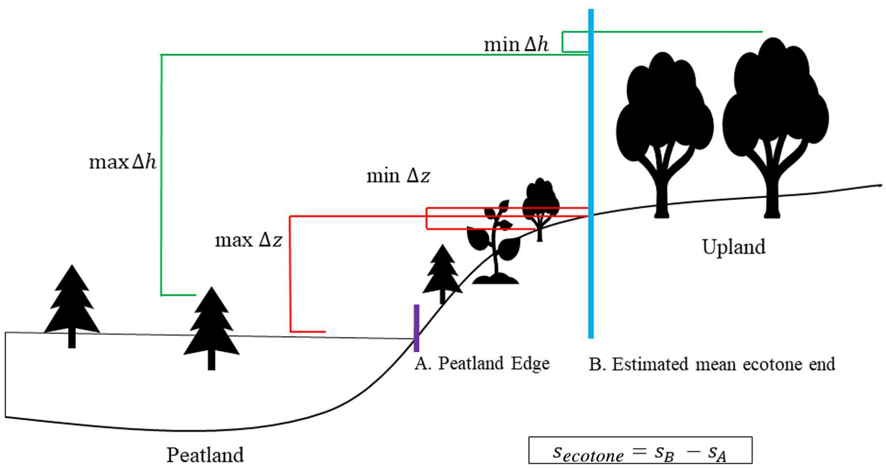

FIGURE 3 Schematic of ecotone identification using lidar data. Iterative methods estimate the length (s) of the ecotones ($s_{ecotone}$, point B) for each individual peatland, based on elevation differences (Δz) and vegetation height difference (Δh) derived from pre-fire (2008) airborne lidar data.

randomly distributed (with a minimum distance of 5 m) across the study area within each stratum. These quantify the range of variability of vegetation heights for each peatland strata: SRI and LRI bog, fen areas, and their adjacent ecotones.

2.5 | Proximity analysis of environmental gradients compared with post-fire vegetation change

Post-fire vegetation regeneration may be influenced by the eco-hydrological characteristics of peatlands, broadly estimated based on peatland class, shape and morphology, and adjacent vegetation species, which can influence succession and life history. The results are introduced by describing variability in vegetation growth associated with underlying surficial geomorphology (fine-textured glacio-lacustrine plains and hummocky moraine; Devito et al., 2016; Paulen et al., 2006). The influences of environmental gradients were then examined by regressing vegetation change within SRI and LRI burned areas to peatland shape and proximity to upland forest vegetation types. To determine peatland shape, the perimeter to area (P:A) ratio was determined from the area and perimeter of each pre-fire classified peatland, where lower P:A ratios indicate compact/round shapes and higher ratios indicate more complex shapes. Regression analysis was also used to determine if relationships exist between surrounding dominant upland forest types (deciduous, mixed wood, or coniferous) within the surrounding uplands, also based on pre-fire classifications from Chasmer et al. (2016) and Bourgeau-Chavez et al. (2017).

2.6 | Statistical analysis

Descriptive and statistical tests were performed to quantify differences between LRI and SRI fires in peatland classes/forms and ecotonal areas (R Core Team, 2021). All measured data variables were tested for normality using a Shapiro–Wilk test. Two-sample Kolmogorov–Smirnov tests and Mann–Whitney U-tests were applied to non-parametric variables to describe the entire data distribution, while normally distributed variables were tested using independent t-tests. To determine the impact of wetland form on vegetation growth, data were transformed logarithmically and then tested using a 2×3 factorial design with ANOVA. Differences in significance in variances for non-parametric variables were tested using the Fligner–Killeen test. Regression analysis was used to determine the relationship between peatland change in vegetation height, environmental drivers, and proximal landcover influences.

3 | RESULTS

3.1 | Spatial rates of vegetation change in peatlands and ecotones following SRI versus LRI fire

In this study, peatlands largely intercepted two geomorphic features: fine-textured hummocky moraine and fine-textured glacio-lacustrine plains. Peatlands in the LRI burn area were relatively evenly distributed among these geomorphological landform classes, with approximately 46% of peatlands in fine-hummocky moraine and 47% within the fine-clay plains, with the remaining area as coarse glacial–fluvial deposits. Peatlands within SRI burn areas were primarily found in fine-hummocky moraine (77%), with 23% in fine plains. Despite differences in geomorphological characteristics, there were no significant differences in vegetation heights in areas burned by SRI or LRI fires within these two geomorphological conditions (SRI $p=.06$; LRI $p=.08$; >.05 significance). This indicates that in the early stage of post-fire regeneration, the geomorphological landform did not greatly impact vegetation changes in this region. Vegetation heights in adjacent ecotonal areas demonstrated similar patterns, with even lower differences between fine hummocky moraines and fine plains (SRI $p=.11$; LRI $p=.23$).

Assuming similarity between geomorphological groups, annual rates of vascular vegetation growth were greater in ecotones than in peatlands. These changes were also greater following SRI fire compared with LRI fire (Figure 4). Figure 4 illustrates change in canopy height between lidar surveys collected in 2011 and 2016, 2016–2019, and across all years within peatlands and ecotones. In Figure 4, rates of vegetation growth proliferated in the early years, post-fire, especially in SRI ecotones, where average growth rates of trees and shrubs, including *Salix* (spp.) and *Alnus* (spp.), were 1.45 m year^{-1} (standard deviation [SD] = 0.41 m). Rates of growth were slightly reduced within ecotones following LRI fire (average = 1.35 m year^{-1}; SD = 0.47 m) but were not significantly different from SRI fire ($p=.19$). Growth rates were much lower

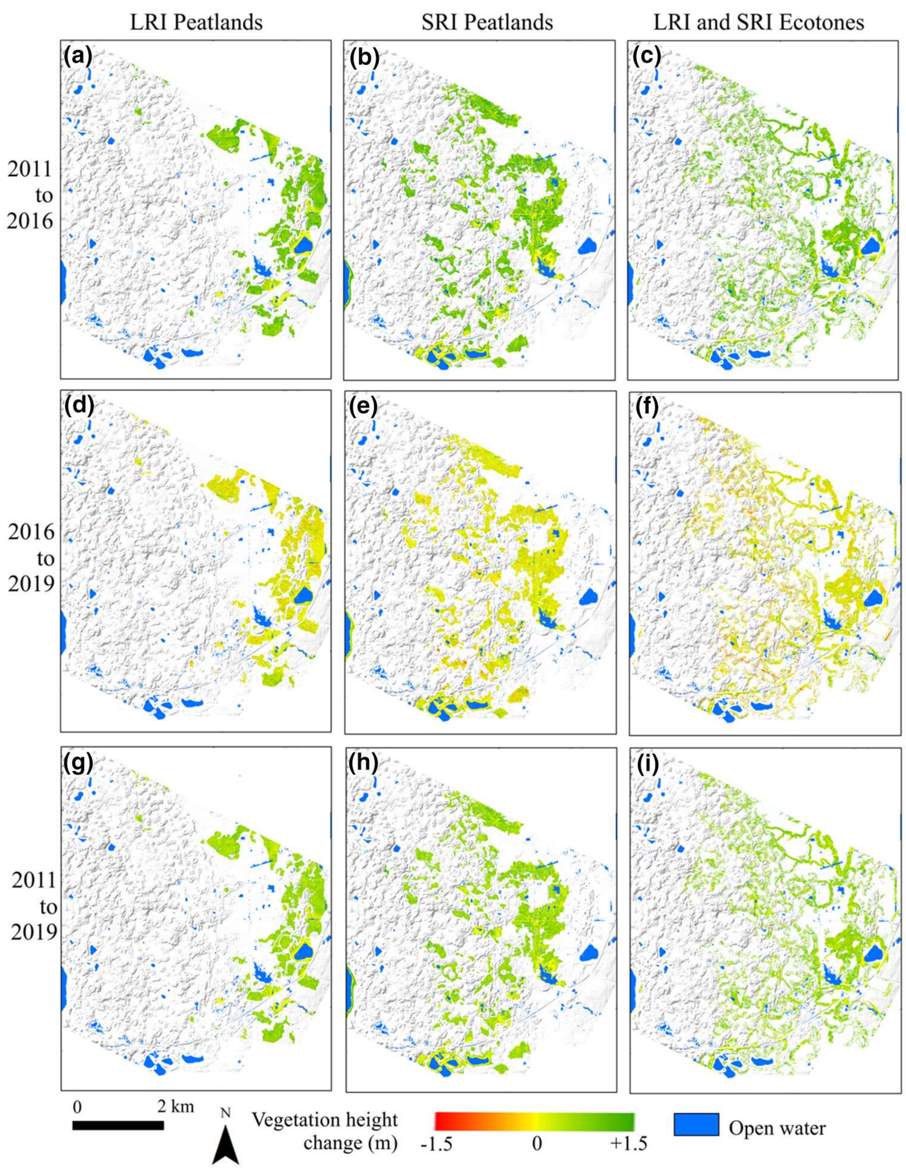

FIGURE 4 Annual vegetation height changes (>0.15 m) within peatlands and ecotone areas following the 2011 fire in LRI- and SRI-burned areas, where (a–c) represent annual change in height in the early years following fire (up to 5 years), (d–f) represent later early stage growth from years 5–8 as vascular vegetation growth slows, and (g–i) represent annual change since the fire to 2019. Background light gray-scale image is a hillshade model from the lidar DEM, which indicates topographic variability indicated by shadows.

in peatlands, though peatland post-fire vegetation growth rates were not greatly different following SRI and LRI fire ($p = .30$). The average early post-fire growth rate was 0.74 m year^{-1} (SD = 0.2 m) in peatlands following SRI fire and 0.70 m year^{-1} (SD = 0.28 m). Peatland edges also experienced the greatest growth of post-fire vegetation, with peatland centers often experiencing lower rates of growth of shrubby vegetation (with the exception of *R. groenlandicum* and mosses). Dominant species regeneration along peatland margins included *R. groenlandicum* and some *Salix* (spp.) observed at the site in 2016 (Schneider et al., 2016).

Between 2016 and 2019, vegetation growth rates reduced substantially within peatlands, especially within ecotones following SRI and LRI fires (Figure 4). In some ecotones, height reductions occurred, which may have been associated with wet summers in 2018 and 2019 and possible mortality of some shrub species, which has been observed in peatlands and ecotones in other studies (e.g., Schneider et al., 2016). During this period, the average annual rate of growth reduced to 0.27 and 0.32 m year^{-1} within peatlands and ecotones, respectively, following SRI fire where there was a greater decline in growth rate in peatlands and ecotones compared with LRI fire (reduction of 8%, on average). Despite slowing of the growth rate in the latter early post-fire period, vascular vegetation, including *P. tremuloides*, *Betula papyrifera*, *Picea* (spp.), and *Alnus* (spp.), proliferated within ecotones with some expansion into peatlands, especially following SRI fire (e.g. Figure 1).

3.2 | Pre-fire trends and post-fire encroachment of vascular vegetation into peatlands

Prior to the 2011 Utikuma Complex fire, forested areas burned in 1956 (SRI), and those that were not burned in the recent past (LRI) were characterized as mid- to late-stage growth. Vascular vegetation heights within peatlands (Figure 5) burned in 1956, and those that had not been burned in the recent (monitored) past were similar in 2002 and 2008 prior to the 2011 fire.

Over the 8 years following fire, total annual vascular vegetation growth rates were significantly greater in bogs compared with fens and greater following SRI fire. Here, on average, bogs following SRI fire experienced 11% greater height growth than those following LRI fire ($p = .012$), including the reduced growth rates in 2018 and 2019 (Figure 5). Differences in post-fire vegetation growth following SRI versus. LRI fire in fens were not as large (4% on average greater following SRI fire, $p = .023$). Therefore, bogs were more susceptible to rapid post-fire growth of shrub and tree species than fens, especially with shortening fire return intervals.

Growth rates of vegetation within peatlands following SRI fire may have also been due to encroachment of vascular vegetation from ecotones. Average vascular vegetation heights (>0.5 m) were about 8.97 m adjacent to bogs and ~1.5 m taller (on average) than fen ecotones by 2018 following SRI fire (Figure 5). These were also taller (on average) than ecotonal vegetation growth following LRI fire. However, while bogs experienced the greatest rate of vegetation growth, especially following SRI fire, the proportional area coverage of vegetation changes within peatlands at 8 years post-fire indicates changing vegetation succession (Figure 6).

Within the study area, bogs cover a much smaller area than fens and have larger ecotonal boundaries relative to fens. Here, we observed a shift to reduced shrub/tree cover within previously (pre-fire) treed/shrub bogs following SRI fire (from 44% of bog area, pre-fire to 24% shrub/tree cover, post-fire), while pre-fire open bogs with few shrubs/trees increased in shrub/tree cover from 4% (pre-fire) to 25% (post-fire; Figure 6a). These also had significantly greater vegetation heights (average = 0.30 m year^{-1}, $p = .003$) than open bogs burned in LRI fire. Bogs following LRI fire were largely resilient and did not experience significant shifts from pre-fire to post-fire shrub/tree vegetation cover. Similarly, there was a shift to post-fire encroachment of trees and shrubs into ecotones that were largely shrub/tree-free prior to fire. This was also found adjacent to fens, with greater encroachment of trees/shrubs into open post-fire fen ecotones where the average growth rate of trees and shrubs was 0.28 m year^{-1} and significantly greater ($p = .0004$) than shrub/treed fens following fire. Post-fire areas of regenerating shrubs and trees within fens were similar (within 3%) following LRI fire in previously shrub/treed and open fens (Figure 6b). However, a proportional increase in the area of trees/shrubs from 37% to 45% following SRI fire in fens illustrated differences in post-fire regeneration within fens following SRI fire.

3.3 | Influence of shape and proximal species on post-fire vegetation change

Peatland shape and area may influence the potential for post-fire vegetation encroachment and rates of growth, especially in peatlands that are small or have greater edge complexity determined by longer perimeters. These are prone to greater vegetation changes associated with ecotonal and proximal forest boundaries (e.g., observed in temperate forests in Morreale et al., 2021) and variable wetting and drying conditions within the peatland margin (lag) (Hokanson et al., 2018). Here, the range of peatland perimeter to area ratio (shape complexity determined per peatland) was greater in LRI-burned peatlands ($P{:}A > 0$ to ~4) than in SRI ($P{:}A > 0$ to <2). Peatland shape complexity was weakly positively correlated with annual average percent vegetation growth following LRI fire ($R^2 = .35$, $p < .0001$) (Figure 7a). In LRI-burned peatlands, vegetation height increased to a threshold of about 60% proportional cover of taller (vascular, shrub, seedling) vegetation in peatlands with increasingly complex shapes (Figure 7a, $R^2 = .45$, $p < .0001$). Following SRI fire, there was no relationship between the complexity of peatland shapes (higher perimeter to area ratio) and post-fire percent annual vegetation growth ($R^2 = .04$, $p = .73$, Figure 7b).

Adjacent deciduous and mixed wood forests (especially those producing aspen suckers) could also result in *P. tremuloides* expansion into peatlands. However, peatlands adjacent to other peatlands

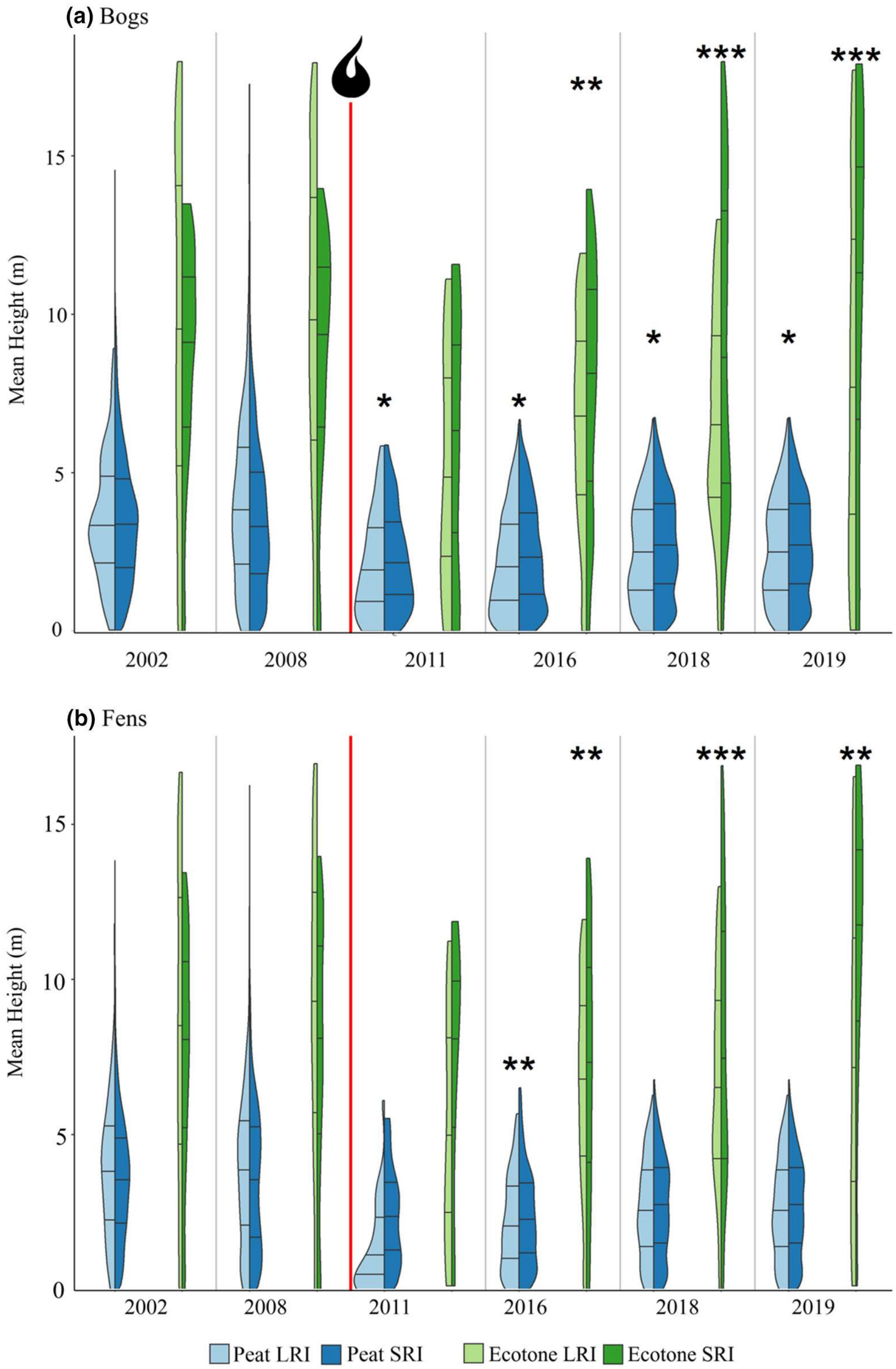

FIGURE 5 Average height and split distributions of height percentiles in peatlands ("Peat") and ecotones in (a) bogs and (b) fens determined from ~6000 randomly distributed points within the strata of the broader study area. The vertical redline indicates the fire event in 2011, which occurred prior to lidar data collection in 2011. The split violin plots represent on the left, the distribution of the LRI strata, and the right the SRI strata. Horizontal lines indicate the 25th, median, and 75th percentiles. Significance between SRI and LRI per year (i.e., peat LRI vs. peat SRI per year) is indicated at the following confidence levels: 95% (*), 99% (**), and 99.9% (***).

and not surrounded by high proportional areas of upland forests tended to have the greatest area of post-fire vascular vegetation growth (Figure 8). Counter to expectations, vascular vegetation growth rates were lower in peatlands burned following both LRI and SRI fires that were adjacent to larger areas of mixed and deciduous upland forests, $r^2 = .48$, .35, respectively ($p < .0001$, Figure 8a). Peatlands adjacent to conifer forests experienced positive changes in vegetation heights. Post-SRI fire vascular vegetation growth increases in peatlands surrounded by more than 50% upland conifer forests (Figure 8b).

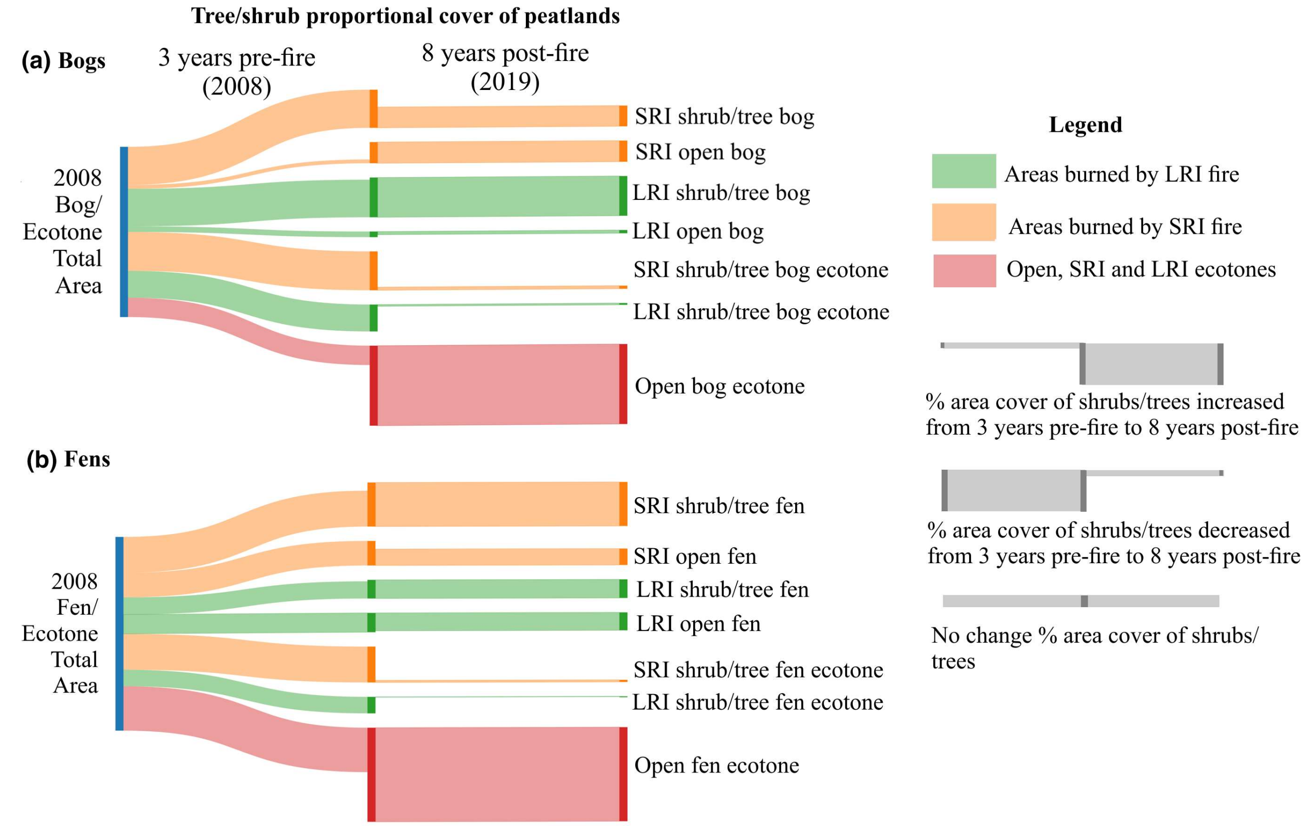

FIGURE 6 Sankey diagram of proportional area coverage of tree/shrub vegetation by total area by land cover type within SRI and LRI in 2019 in (a) bogs and bog ecotones; and (b) fens and fen ecotones compared with 2008, 3 years pre-fire area vegetation conditions. Proportions are subdivided within each peatland class and adjacent ecotones.

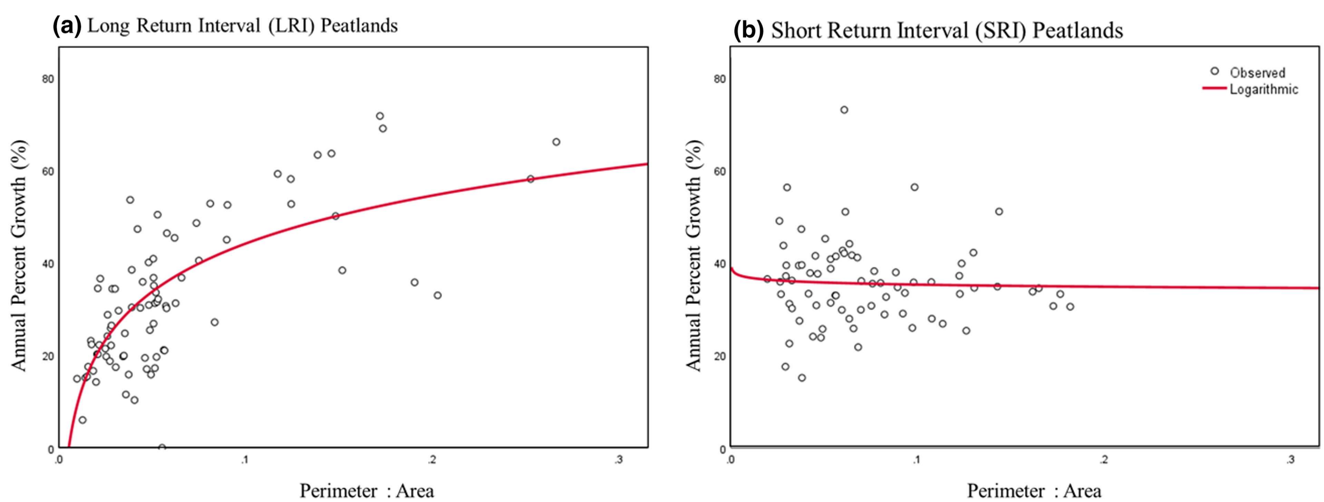

FIGURE 7 Nonlinear regressions, comparing perimeter:area ratios with annual percent growth between (a) LRI fire and (b) SRI fire within peatlands.

4 | DISCUSSION

4.1 | Fire regime impacts early post-fire vegetation succession within peatlands

This study aimed to assess whether shortening fire return intervals impacts vegetation growth trajectories relative to LRI fires, which are more typical of past fire regimes. While a few studies have quantified the impacts of SRI versus LRI fires on peatlands, this study used time-series airborne lidar data to quantify the spatial distribution and drivers of vegetation height change in the early post-fire period across a range of environmental conditions. This is important for the assessment of peatland vegetation succession following wildland fire, the maintenance of peatlands as critically important

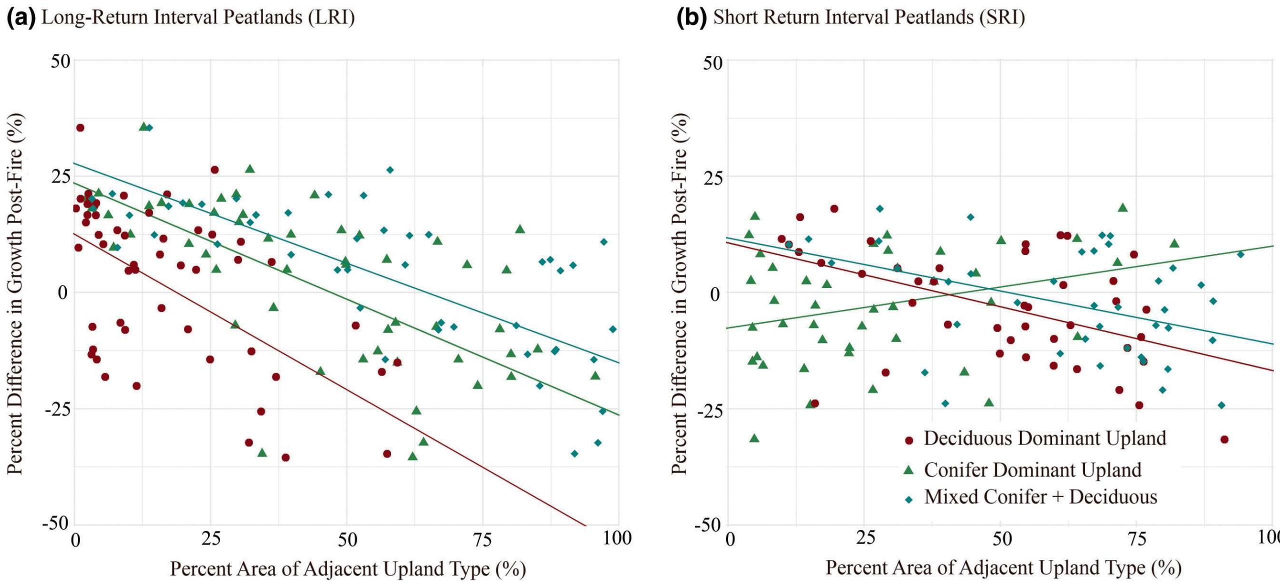

FIGURE 8 Linear regression models of the differences between percent growth compared with the percent area of surrounding upland type in (a) LRI-burned peatlands and (b) SRI-burned peatlands. Vegetation changes for the period between 2016 and 2019 were compared as a proportion of the rate of growth of shrubs and trees within peatlands between 2002 and 2008. Each data point represents the average of a single peatland.

sources of water for surrounding ecosystems (Bridgham et al., 2008; Waddington et al., 2015), and as fire-reducing land covers (Turetsky & Louis, 2006).

Accelerated vegetation growth and species shifts are predicted to enhance fuel connectivity with forest upland (Thompson, Parisien, et al., 2017). Here, SRI enhanced vascular vegetation growth rates, resulting in taller vegetation in bogs, fens, and adjacent ecotones compared with LRI fire (Figures 4 and 5). Accelerated growth occurred primarily in shrub and deciduous species within open peatlands, such as aspen (*P. tremuloides*), alders (*Alnus* spp.), willows (*salix* spp.), and Labrador tea (*R. groenlandicum*; Figures 4 and 6). While some of these species are commonly observed regrowing post-fire (Benscoter & Vitt, 2008), the increased occurrence of fire within open areas may have increased available nutrients such as nitrogen (Shenoy et al., 2013), resulting in faster regrowth and greater species diversity than that found in LRI-burned peatlands. Shifting fire regimes influence nutrient and moisture availability (Thompson, Parisien, et al., 2017), which may be associated with vegetation growth/mortality observed as a proxy indicator for environmental processes, including energy and mass exchanges. Deciduous species have greater demands on water resources, which could enhance transpiration losses in peatlands with greater regeneration of deciduous shrubs and trees instead of conifers (Johnston et al., 2015; Johnstone et al., 2004). While carbon losses from peatland burning contribute to atmospheric C emissions, Mack et al. (2021) demonstrate that rapid re-establishment of deciduous species may offset short-term C losses from fire. Since deciduous stands typically have lower flammability, resulting in lower severity fires, mixed conifer, and deciduous species could slow fire spread into peatlands (Parisien et al., 2011), though they also enhance the drying of peat fuel (Malhotra et al., 2020; Thompson et al., 2019). Within the first 5 years following the fire, Schiks et al. (2016), Jones et al. (2022), and Guêné-Nanchen et al. (2022) observed post-fire proliferation of short-statured shrubs, such as Labrador tea (*R. groenlandicum*) in a sample of measured peatlands.

While early rapid regeneration of deciduous species following fire is not uncommon (Johnstone et al., 2004), growth rates of post-fire vegetation within bogs and fens and between fire intervals were different, which could indicate vulnerability to long-term change. Bogs demonstrated a greater response to SRI fire regimes than fens, with increased rate of vascular vegetation growth (Figure 5), especially in open bogs (Figure 6a). One potential explanation is that fens are hydrologically connected to their surroundings and have more continuous inputs of ground/surface water (Goodbrand et al., 2019; Kuhry et al., 1993). Sherwood et al. (2013) and Kettridge Turetsky et al. (2015) observed greater evapotranspiration rates from fens following fire, resulting in enhanced shrub/vascular vegetation productivity. Bourgeau-Chavez et al. (2020) also observed lower burn severity in fens, which, when combined with greater evaporative potential, produces conditions favorable for the encroachment of deciduous species (Grzesik et al., 2022; Jones et al., 2022; Langdon et al., 2020; Lukenbach et al., 2017; Moore et al., 2022). However, trait plasticity and adaptability of shrubs to varied environmental conditions, including hydrology, require further assessment (Dabros et al., 2022; Moore et al., 2022; Turetsky et al., 2008).

In comparison, domed bogs maintain drier peat soil conditions than fens, making them more susceptible to deep smoldering fire (Bourgeau-Chavez et al., 2020; Price & Maloney, 1994). Here, some bogs expanded immediately following fire due to vegetation mortality and loss of peat substrate, particularly within ecotones. However, these areas also experienced significant post-fire vegetation encroachment (Figures 4 and 6a). Deep smoldering into peat layers results in the development of a burn moat and water movement/flooding of the moat, particularly during wet periods (Ingram et al., 2019; Lukenbach et al., 2017), which can result in vegetation mortality, especially along peatland margins (Figures 4 and 8). Furthermore, reduced elevation of the peat surface (and especially within hollows) maintains the high moisture characteristics of peatlands and reduces recolonization of upland species (Ott et al., 2006; Granath et al., 2010; Lukenbach et al., 2017). For example, bogs experienced high vegetation growth rates in the early years following both SRI and LRI fire. Rates of growth were significantly reduced between 2016–2018 and 2019 (Figures 4c,d and 5a) due to high early summer precipitation and peatland flooding, potentially maintaining the bog areas over time (Ingram et al., 2019; Lukenbach et al., 2016; Waddington et al., 2015).

Geomorphology, peatland shape, and proximity to upland forest types have low to moderate influence on post-fire vegetation changes. Geomorphology plays an important role in peatland hydrology (Devito et al., 2016; Kuhry & Turunen, 2006), and post-fire regeneration is likely to be influenced by the potential for flooding (in hydrologically disconnected SRI-burned bogs) or evaporative losses (fens). Increasingly complex peatland shapes also correlate with greater rates of vegetation growth, especially following LRI fire (Figure 7a). Finally, mixed-wood forests had a greater influence on both bog and fen vegetation growth. However, this declined as the proportional area of forests surrounding peatlands increases (Figure 8b). This is somewhat counterintuitive, given the potential for the expansion of aspen suckers into peatlands; however, the first 8 years represent only the early phase of post-fire regeneration, and the influence of the remaining seedbed in peatlands following fire may yet manifest (Ellison & Bedford, 1995; Guêné-Nanchen et al., 2022; Langdon et al., 2020).

4.2 | Ecotones indicate that the greatest changes are yet to come

Ecotones are at greatest risk for fluctuations in water table (Dimitrov et al., 2014), flooding (Lukenbach et al., 2017; Ingram et al., 2019), and deeper burns (Hokanson et al., 2016; Morison et al., 2021; Lukenbach et al., 2017; Wilkinson, Tekatch, et al., 2020; Wilkinson, Verkaik, et al., 2020) resulting in greater environmental extremes and more rapid changes that could either enhance vascular vegetation growth or induce mortality especially following SRI fires. Here, peatland ecotones and adjacent peatlands demonstrated significantly greater changes in both rates of vegetation growth and decline (Figure 4) as well as the proportional area of vascular vegetation within peatlands and ecotones (Figure 5) following SRI as opposed to LRI fires. More dynamic responses to changing fire regimes may enhance variations in moisture conditions between fires (Hokanson et al., 2016). If conditions increase in the periodicity and extent of drying, this could further exacerbate the potential for more intense and unpredictable fires (Kettridge Lukenbach et al., 2019; Turetsky et al., 2011). Additionally, burning of ecotones redistributes soil nutrients (Binkley & Fisher, 2019), further enhancing shrub and tree encroachment (Nelson et al., 2021; Shi et al., 2022). Given their proximity to peatlands, expanding ecotonal areas could introduce new seed beds into the peatlands and reduce overall peatland area while enhancing the shape complexity of peatlands (Figure 7; Langdon et al., 2020). Here, we find that ecotones are more responsive to changing conditions (Bhatti et al., 2006; Goud et al., 2018) (Figures 4 and 5) and are greater indicators of the impacts of climate change on ecosystems as these are often the first areas to change, indicated by changing vegetation (Neilson, 1993; Noble, 1993; Peteet, 2000). Therefore, rapid shifts in post-fire vegetation growth will likely affect peatland function for many years to come.

5 | CONCLUSIONS AND FUTURE RESEARCH

Peatlands and adjacent ecotones burned by SRI fires tend to have the greatest rates of vegetation change (growth and mortality) compared with peatlands burned during an LRI fire. However, peatland/upland ecotones demonstrated the greatest changes in post-fire vegetation change. While we expected ecotones to be more responsive to changing conditions (Bhatti et al., 2006; Goud et al., 2018), the rate at which vegetation is regenerating and, in many areas, expanding into peatlands is a potential concern as these processes increase forest fuel connectivity and propensity for burning (Senici et al., 2015; Thompson et al., 2019).

Additionally, shrubification of peatland centers increases transpiration and enhances drying (Moore et al., 2022), putting these peatlands at greater risk for fire due to the loss of peat moss through subsequent burns and increased ladder fuels (Shetler et al., 2008; Walker et al., 2020), thereby diminishing the traditional function of peatlands as fire refugia. We also observed the opposite response associated with vegetation mortality (height reduction) along ecotonal margins following a period of increased precipitation. The potential for enhanced flooding, especially within bog ecotones following SRI fire, could enhance peatland maintenance where hydraulic gradients are favorable.

This study is important because it presents early post-fire vascular vegetation recovery across hundreds of peatlands following SRI and LRI fires. With increasing frequency of fire in peatlands and the associated climate warming positive feedback (Thompson, Mendoza, et al., 2017), understanding the peatland responses to shortened fire return intervals is vital. Even within our study area, fire return intervals are shortening. In May 2023, another large fire burned across peatlands affected by the Utikuma Complex fire in 2011, shortening the fire return interval within the peatland complex to 12 years (Alberta, 2023). Relatively little has been examined regarding the impacts of increased fire frequency on peatlands, and as such, there are many opportunities for future research. We suggest future research focus on (1) determining the impact of burn severity between repeat fires on peatland vegetation and soil development; (2) monitoring long-term impacts on peatlands following SRI fires to better determine the trajectory of peatland and ecotone development under conditions of climate-mediated drying; and (3) large-area analyses of the state of peatlands and their changing hydrological conditions as fire refugia across the boreal biome.

AUTHOR CONTRIBUTIONS

Emily Ann Jones: Conceptualization; formal analysis; investigation; methodology; validation; visualization; writing – original draft; writing – review and editing. **Laura Elizabeth Chasmer:** Conceptualization; data curation; funding acquisition; investigation; methodology; project administration; resources; software; supervision; validation; visualization; writing – original draft; writing – review and editing. **Kevin John Devito:** Conceptualization; investigation; methodology; resources; validation; writing – review and editing. **Christopher Dennis Hopkinson:** Conceptualization; data curation; funding acquisition; investigation; methodology; project administration; resources; software; supervision; writing – review and editing.

ACKNOWLEDGMENTS

Funding for this project was supported by NSERC-Discovery Grants (Natural Sciences and Engineering Research Council of Canada) to Chasmer (#2017-04492) and Hopkinson (#2017-04362) and NSERC Canada Wildfire SPG-N sub-grant to Chasmer. Airborne and field survey equipment was funded by Western Economic Diversification Canada (#000015316) and Canada Foundation for Innovation (#32426) grants to Hopkinson. The 2002 lidar data collection was supported by a NSERC Collaborative Research and Development Grant (HEAD2) to Devito. The 2008 lidar data were provided by Alberta Environment and Parks under agreement DMR#1907M20. Jones was supported through a Mitacs Accelerate project (IT13390) in partnership with Ducks Unlimited Canada. We would also like to acknowledge Carolyn Forsyth and ArtisInn for providing lodgings during the summer of 2019. Thank you to Dr. Stewart Rood for helpful early discussion and the anonymous reviewers who helped us improve the manuscript.

CONFLICT OF INTEREST STATEMENT

We confirm that there is no conflict of interest.

DATA AVAILABILITY STATEMENT

The data that support the findings of this study are openly available in Canadian Federated Research Data Repository. 10.20383/103.0888 and 10.20383/103.0892 at Home|FRDR-DFDR.

ORCID

Emily Ann Jones https://orcid.org/0000-0001-8675-3369
Laura Elizabeth Chasmer https://orcid.org/0000-0002-8062-1530

REFERENCES

Alberta, G. O. (2023). *Alberta wildfire status*. https://www.arcgis.com/apps/dashboards/3ffcc2d0ef3e4e0999b0cf8b636defa3

Alberta Wildfire. (2018). *Spatial wildfire data*. http://wildfire.alberta.ca/resources/historical-data/spatial-wildfire-data.aspx

Andersen, H.-E., Reutebuch, S. E., & McGaughey, R. J. (2006). A rigorous assessment of tree height measurements obtained using airborne lidar and conventional field methods. *Canadian Journal of Remote Sensing*, *32*(5), 355–366.

Bauer, I. E., Gignac, L. D., & Vitt, D. H. (2003). Development of a peatland complex in boreal western Canada: Lateral site expansion and local variability in vegetation succession and long-term peat accumulation. *Canadian Journal of Botany*, *81*(8), 833–847.

Benscoter, B. W. (2006). Post-fire bryophyte establishment in a continental bog. *Journal of Vegetation Science*, *17*(5), 647–652.

Benscoter, B. W., & Vitt, D. H. (2008). Spatial patterns and temporal trajectories of the bog ground layer along a post-fire chronosequence. *Ecosystems*, *11*(7), 1054–1064.

Bhatti, J., Errington, R., Bauer, I., & Hurdle, P. (2006). Carbon stock trends along forested peatland margins in central Saskatchewan. *Canadian Journal of Soil Science*, *86*(Special Issue), 321–333.

Binkley, D., & Fisher, R. F. (2019). *Ecology and management of forest soils*. John Wiley & Sons.

Bisbee, K. E., Gower, S. T., Norman, J. M., & Nordheim, E. V. (2001). Environmental controls on ground cover species composition and productivity in a boreal black spruce forest. *Oecologia*, *129*(2), 261–270.

Bonan, G. B., Pollard, D., & Thompson, S. L. (1992). Effects of boreal forest vegetation on global climate. *Nature*, *359*(6397), 716–718.

Bourgeau-Chavez, L. L., Endres, S., Powell, R., Battaglia, M. J., Benscoter, B., Turetsky, M., Kasischke, E. C., & Banda, E. (2017). Mapping boreal peatland ecosystem types from multitemporal radar and optical satellite imagery. *Canadian Journal of Forest Research*, *47*(4), 545–559.

Bourgeau-Chavez, L. L., Grelik, S. L., Billmire, M., Jenkins, L. K., Kasischke, E. S., & Turetsky, M. R. (2020). Assessing boreal peat fire severity and vulnerability of peatlands to early season wildland fire. *Frontiers in Forests and Global Change*, *3*, 20.

Brandt, J. P., Flannigan, M., Maynard, D., Thompson, D. K. I., & Volney, W. (2013). An introduction to Canada's boreal zone: Ecosystem processes, health, sustainability, and environmental issues. *Environmental Reviews*, *21*(4), 207–226.

Bridgham, S. D., Pastor, J., Dewey, B., Weltzin, J. F., & Updegraff, K. (2008). Rapid carbon response of peatlands to climate change. *Ecology*, *89*(11), 3041–3048.

Chasmer, L., Chasmer, L., Hopkinson, C., Montgomery, J., & Petrone, R. (2016). A physically based terrain morphology and vegetation structural classification for wetlands of the Boreal Plains, Alberta, Canada. *Canadian Journal of Remote Sensing*, *42*(5), 521–540. https://doi.org/10.1080/07038992.2016.1196583

Chasmer, L., & Hopkinson, C. (2017). Threshold loss of discontinuous permafrost and landscape evolution. *Global Change Biology*, *23*(7), 2672–2686. https://doi.org/10.1111/gcb.13537

Chasmer, L., Jones, E., Devito, K., & Hopkinson, C. (2024). Utikuma region Alberta high resolution lidar-multispectral peatland and ecotone classification. Dataset available on the Canadian Federated Research Data Repository (FRDR). https://doi.org/10.20383/103.0892

Dabros, A., Higgins, K. L., Santala, K., & Aubin, I. (2022). Plant functional trait approach to assess the persistence of seismic line footprint in boreal peatlands of Alberta, Canada. *Forest Ecology and Management*, *503*, 119751.

Davis, K. J., Lenschow, D. H., Oncley, S. P., Kiemle, C., Ehret, G., Giez, A., & Mann, J. (1997). Role of entrainment in surface-atmosphere interactions over the boreal forest. *Journal of Geophysical Research: Atmospheres*, *102*(D24), 29219–29230.

Devito, K. J., Mendoza, C., Petrone, R. M., Kettridge, N., & Waddington, J. M. (2016). Utikuma region study area (URSA)–Part 1: Hydrogeological and ecohydrological studies (HEAD). *The Forestry Chronicle*, *92*(1), 57–61.

Dimitrov, D. D., Bhatti, J. S., & Grant, R. F. (2014). The transition zones (ecotone) between boreal forests and peatlands: Modelling water table along a transition zone between upland black spruce forest and poor forested fen in central Saskatchewan. *Ecological Modelling*, *274*, 57–70.

Ellison, A. M., & Bedford, B. L. (1995). Response of a wetland vascular plant community to disturbance: A simulation study: Ecological archives. *Ecological Applications*, *5*(1), 109–123.

Elmes, M. C., Thompson, D. K., Sherwood, J. H., & Price, J. S. (2018). Hydrometeorological conditions preceding wildfire, and the subsequent burning of a fen watershed in Fort McMurray, Alberta, Canada. *Natural Hazards and Earth System Sciences*, *18*(1), 157–170.

Falkowski, M. J., Smith, A. M., Gessler, P. E., Hudak, A. T., Vierling, L. A., & Evans, J. S. (2008). The influence of conifer forest canopy cover on the accuracy of two individual tree measurement algorithms using lidar data. *Canadian Journal of Remote Sensing*, *34*(supplement 2), S338–S350.

Ferone, J., & Devito, K. (2004). Shallow groundwater–surface water interactions in pond–peatland complexes along a Boreal Plains topographic gradient. *Journal of Hydrology*, *292*(1–4), 75–95.

Flannigan, M. D., Bergeron, Y., Engelmark, O., & Wotton, B. M. (1998). Future wildfire in circumboreal forests in relation to global warming. *Journal of Vegetation Science*, *9*(4), 469–476.

Frelich, L. E., & Reich, P. B. (2010). Will environmental changes reinforce the impact of global warming on the prairie–forest border of central North America? *Frontiers in Ecology and the Environment*, *8*(7), 371–378.

Gauthier, S., Bernier, P., Kuuluvainen, T., Shvidenko, A., & Schepaschenko, D. (2015). Boreal forest health and global change. *Science*, *349*(6250), 819–822.

Gibson, C. M., Chasmer, L. E., Thompson, D. K., Quinton, W. L., Flannigan, M. D., & Olefeldt, D. (2018). Wildfire as a major driver of recent permafrost thaw in boreal peatlands. *Nature Communications*, *9*(1), 3041.

Glaser, P. H., Siegel, D. I., Romanowicz, E. A., & Shen, Y. P. (1997). Regional linkages between raised bogs and the climate, groundwater, and landscape of north-western Minnesota. *Journal of Ecology*, *85*, 3–16.

Goodbrand, A., Westbrook, C. J., & van der Kamp, G. (2019). Hydrological functions of a peatland in a Boreal Plains catchment. *Hydrological Processes*, *33*(4), 562–574.

Goud, E. M., Watt, C., & Moore, T. R. (2018). Plant community composition along a peatland margin follows alternate successional pathways after hydrologic disturbance. *Acta Oecologica*, *91*, 65–72.

Granath, G., Strengbom, J., & Rydin, H. (2010). Rapid ecosystem shifts in peatlands: Linking plant physiology and succession. *Ecology*, *91*(10), 3047–3056.

Grzesik, E., Hollingsworth, T., Ruess, R., & Turetsky, M. (2022). Fuel loads and plant traits as community-level predictors of emergent properties of vulnerability and resilience to a changing fire regime in black spruce forests of boreal Alaska. *Journal of Geophysical Research: Biogeosciences*, *127*(3), e2021JG006696.

Guêné-Nanchen, M., LeBlanc, M.-C., & Rochefort, L. (2022). Post-fire peatland vegetation recovery: A case study in open rich fens of the Canadian boreal forest. *Botany*, *100*(5), 435–447.

Helbig, M., Waddington, J. M., Alekseychik, P., Amiro, B., Aurela, M., Barr, A. G., Black, T. A., Carey, S. K., Chen, J., Chi, J., Desai, A. R., Dunn, A., Euskirchen, E. S., Flanagan, L. B., Friborg, T., Garneau, M., Grelle, A., Harder, S., Heliasz, M., … Schulze, C. (2020). The biophysical climate mitigation potential of boreal peatlands during the growing season. *Environmental Research Letters*, *15*(10), 104004.

Helbig, M., Waddington, J. M., Alekseychik, P., Amiro, B. D., Aurela, M., Barr, A. G., Black, T. A., Blanken, P. D., Carey, S. K., Chen, J., Chi, J., Desai, A. R., Dunn, A., Euskirchen, E. S., Flanagan, L. B., Forbrich, I., Friborg, T., Grelle, A., Harder, S., … Zyrianov, V. (2020). Increasing contribution of peatlands to boreal evapotranspiration in a warming climate. *Nature Climate Change*, *10*(6), 555–560.

Hodgson, M. E., & Bresnahan, P. (2004). Accuracy of airborne lidar-derived elevation. *Photogrammetric Engineering & Remote Sensing*, *70*(3), 331–339.

Hokanson, K., Lukenbach, M., Devito, K., Kettridge, N., Petrone, R., & Waddington, J. (2016). Groundwater connectivity controls peat burn severity in the boreal plains. *Ecohydrology*, *9*(4), 574–584. https://doi.org/10.1002/eco.1657

Hokanson, K., Mendoza, C., & Devito, K. (2019). Interactions between regional climate, surficial geology, and topography: Characterizing shallow groundwater systems in subhumid, low-relief landscapes. *Water Resources Research*, *55*(1), 284–297.

Hokanson, K., Moore, P., Lukenbach, M., Devito, K., Kettridge, N., Petrone, R., & Waddington, J. (2018). A hydrogeological landscape framework to identify peatland wildfire smouldering hot spots. *Ecohydrology*, *11*(4), e1942.

Holmgren, M., Lin, C. Y., Murillo, J. E., Nieuwenhuis, A., Penninkhof, J., Sanders, N., & Vollebregt, M. E. (2015). Positive shrub–tree interactions facilitate woody encroachment in boreal peatlands. *Journal of Ecology*, *103*(1), 58–66. https://doi.org/10.1111/1365-2745.12331

Hopkinson, C., Chasmer, L., Barr, A. G., Kljun, N., Black, T. A., & McCaughey, J. H. (2016). Monitoring boreal forest biomass and carbon storage change by integrating airborne laser scanning, biometry and eddy covariance data. *Remote Sensing of Environment*, *181*, 82–95. https://doi.org/10.1016/j.rse.2016.04.010

Hopkinson, C., Chasmer, L., Colville, D., Fournier, R. A., Hall, R. J., Luther, J. E., & St-Onge, B. (2013). Moving toward consistent ALS monitoring of forest attributes across Canada. *Photogrammetric Engineering & Remote Sensing*, *79*(2), 159–173.

Hopkinson, C., Chasmer, L., Lim, K., Treitz, P., & Creed, I. (2006). Towards a universal lidar canopy height indicator. *Canadian Journal of Remote Sensing*, *32*(2), 139–152.

Hopkinson, C., Chasmer, L. E., Sass, G., Creed, I. F., Sitar, M., Kalbfleisch, W., & Treitz, P. (2005). Vegetation class dependent errors in lidar ground elevation and canopy height estimates in a boreal wetland environment. *Canadian Journal of Remote Sensing*, *31*(2), 191–206. https://doi.org/10.5589/m05-007

Hopkinson, C., Jones, E., Chasmer, L., & Devito, K. (2024). Utikuma region lidar canopy height model time series. Dataset available on the Canadian Federated Research Data Repository (FRDR). https://doi.org/10.20383/103.0888

Immirzi, C. P., Maltby, E., & Clymo, R. S. (1992). *The global status of peatlands and their role in carbon cycling*. N. p.

Ingram, R., Moore, P., Wilkinson, S., Petrone, R., & Waddington, J. (2019). Postfire soil carbon accumulation does not recover boreal peatland combustion loss in some hydrogeological settings. *Journal of Geophysical Research: Biogeosciences*, *124*(4), 775–788.

Johnston, D., Turetsky, M., Benscoter, B., & Wotton, B. (2015). Fuel load, structure, and potential fire behaviour in black spruce bogs. *Canadian Journal of Forest Research*, *45*(7), 888–899.

Johnstone, J. F., Chapin, F., III, Foote, J., Kemmett, S., Price, K., & Viereck, L. (2004). Decadal observations of tree regeneration following fire in boreal forests. *Canadian Journal of Forest Research*, *34*(2), 267–273.

Jones, E., Chasmer, L., Devito, K., Rood, S., & Hopkinson, C. (2022). Ecological impacts of shortening fire return intervals on boreal peatlands and transition zones using integrated in situ field sampling and lidar approaches. *Ecohydrology*, *15*(3), e2403.

Kettridge Lukenbach, M. C., Hokanson, K. J., Devito, K. J., Petrone, R. M., Mendoza, C. A., & Waddington, J. M. (2019). Severe wildfire exposes remnant peat carbon stocks to increased post-fire drying. *Scientific Reports*, *9*(1), 3727. https://doi.org/10.1038/s41598-019-40033-7

Kettridge, N., Humphrey, R., Smith, J., Lukenbach, M., Devito, K., Petrone, R., & Waddington, J. (2014). Burned and unburned peat water repellency: Implications for peatland evaporation following wildfire. *Journal of Hydrology*, *513*, 335–341.

Kettridge, N., Thompson, D., Bombonato, L., Turetsky, M., Benscoter, B., & Waddington, J. M. (2013). The ecohydrology of forested peatlands: Simulating the effects of tree shading on moss evaporation and species composition. *Journal of Geophysical Research: Biogeosciences*, *118*(2), 422–435.

Kettridge Turetsky, M., Sherwood, J., Thompson, D., Miller, C., Benscoter, B., & Waddington, J. (2015). Moderate drop in water table increases peatland vulnerability to post-fire regime shift. *Scientific Reports*, *5*, 8063.

Kuhry, P., Nicholson, B. J., Gignac, L. D., Vitt, D. H., & Bayley, S. E. (1993). Development of Sphagnum-dominated peatlands in boreal continental Canada. *Canadian Journal of Botany*, *71*(1), 10–22.

Kuhry, P., & Turunen, J. (2006). The postglacial development of boreal and subarctic peatlands. In *Boreal peatland ecosystems* (pp. 25–46). Springer.

Langdon, S. F., Dovciak, M., & Leopold, D. J. (2020). Tree encroachment varies by plant community in a large boreal peatland complex in the boreal-temperate ecotone of northeastern USA. *Wetlands*, *40*(6), 2499–2511.

Lieffers, V. J., & Macdonald, S. E. (1990). Growth and foliar nutrient status of black spruce and tamarack in relation to depth of water table in some Alberta peatlands. *Canadian Journal of Forest Research*, *20*(6), 805–809.

Lukenbach, M., Devito, K., Kettridge, N., Petrone, R., & Waddington, J. (2016). Burn severity alters peatland moss water availability: Implications for post-fire recovery. *Ecohydrology*, *9*(2), 341–353. https://doi.org/10.1002/eco.1639

Lukenbach, M., Hokanson, K. J., Devito, K., Kettridge, N., Petrone, R., Mendoza, C., & Waddington, J. M. (2017). Post-fire ecohydrological conditions at peatland margins in different hydrogeological settings of the boreal plain. *Journal of Hydrology*, *548*, 741–753.

Lukenbach, M. C., Hokanson, K. J., Moore, P. A., Devito, K. J., Kettridge, N., Thompson, D. K., & Waddington, J. M. (2015). Hydrological controls on deep burning in a northern forested peatland. *Hydrological Processes*, *29*(18), 4114–4124.

Mack, M. C., Walker, X. J., Johnstone, J. F., Alexander, H. D., Melvin, A. M., Jean, M., & Miller, S. N. (2021). Carbon loss from boreal forest wildfires offset by increased dominance of deciduous trees. *Science*, *372*(6539), 280–283.

Malhotra, A., Brice, D. J., Childs, J., Graham, J. D., Hobbie, E. A., Vander Stel, H., & Iversen, C. M. (2020). Peatland warming strongly increases fine-root growth. *Proceedings of the National Academy of Sciences of the United States of America*, *117*(30), 17627–17634.

Mayner, K. M., Moore, P. A., Wilkinson, S. L., Petrone, R. M., & Waddington, J. M. (2018). Delineating boreal plains bog margin ecotones across hydrogeological settings for wildfire risk management. *Wetlands Ecology and Management*, *26*(6), 1037–1046.

Michaelian, M., Hogg, E. H., Hall, R. J., & Arsenault, E. (2011). Massive mortality of aspen following severe drought along the southern edge of the Canadian boreal forest. *Global Change Biology*, *17*(6), 2084–2094. https://doi.org/10.1111/j.1365-2486.2010.02357.x

Montgomery, J., Brisco, B., Chasmer, L., Devito, K., Cobbaert, D., & Hopkinson, C. (2019). SAR and LiDAR temporal data fusion approaches to boreal wetland ecosystem monitoring. *Remote Sensing*, *11*(2), 161.

Moore, P. A., Pypker, T. G., Hribljan, J. A., Chimner, R. A., & Waddington, J. M. (2022). Examining the peatland shrubification-evapotranspiration feedback following multi-decadal water table manipulation. *Hydrological Processes*, *36*(11), e14719.

Morison, M., van Beest, C., Macrae, M., Nwaishi, F., & Petrone, R. (2021). Deeper burning in a boreal fen peatland 1-year post-wildfire accelerates recovery trajectory of carbon dioxide uptake. *Ecohydrology*, *14*(3), e2277.

Morreale, L. L., Thompson, J. R., Tang, X., Reinmann, A. B., & Hutyra, L. R. (2021). Elevated growth and biomass along temperate forest edges. *Nature Communications*, *12*(1), 7181.

Næsset, E. (2004). Practical large-scale forest stand inventory using a small-footprint airborne scanning laser. *Scandinavian Journal of Forest Research*, *19*(2), 164–179.

Næsset, E. (2009). Effects of different sensors, flying altitudes, and pulse repetition frequencies on forest canopy metrics and biophysical stand properties derived from small-footprint airborne laser data. *Remote Sensing of Environment*, *113*(1), 148–159.

Neilson, R. P. (1993). Transient ecotone response to climatic change: Some conceptual and modelling approaches. *Ecological Applications*, *3*(3), 385–395.

Nelson, K., Thompson, D., Hopkinson, C., Petrone, R., & Chasmer, L. (2021). Peatland-fire interactions: A review of wildland fire feedbacks and interactions in Canadian boreal peatlands. *Science of the Total Environment*, *769*, 145212.

Noble, I. R. (1993). A model of the responses of ecotones to climate change. *Ecological Applications*, *3*(3), 396–403.

Ott, L., Mann, P., & Van Cleve, K. (2006). Successional processes in the Alaskan boreal forest. In F. Stuart Chapin, M. W. Oswood, K. van Cleve, L. A. Viereck, & D. L. Verbyla (Eds.), *Alaska's changing boreal forest* (p. 100). Oxford University.

Pappas, C., Matheny, A. M., Baltzer, J. L., Barr, A. G., Black, T. A., Bohrer, G., & Sonnentag, O. (2018). Boreal tree hydrodynamics: Asynchronous, diverging, yet complementary. *Tree Physiology*, *38*(7), 953–964.

Parisien, M.-A., Parks, S. A., Miller, C., Krawchuk, M. A., Heathcott, M., & Moritz, M. A. (2011). Contributions of ignitions, fuels, and weather to the spatial patterns of burn probability of a boreal landscape. *Ecosystems*, *14*(7), 1141–1155.

Paulen, R. C., Pawlowicz, J. G., & Fenton, M. M. (2006). *Surficial geology of the Utikuma area*. Alberta Geological Survey. https://ags.aer.ca/publication/dig-2004-0001

Pennypacker, S., & Wood, R. (2023). Grounding our understanding of the impacts of boreal forest expansion on shallow cumulus clouds with a simple modelling framework. *Journal of Hydrometeorology*, *24*, 2333–2349.

Peteet, D. (2000). Sensitivity and rapidity of vegetational response to abrupt climate change. *Proceedings of the National Academy of Sciences of the United States of America*, *97*(4), 1359–1361.

Petrone, R., Silins, U., & Devito, K. (2007). Dynamics of evapotranspiration from a riparian pond complex in the Western boreal forest, Alberta, Canada. *Hydrological Processes*, *21*(11), 1391–1401. https://doi.org/10.1002/hyp.6298

Price, D. T., Alfaro, R., Brown, K., Flannigan, M., Fleming, R. A., Hogg, E., Girardin, M. P., Lakusta, T., Johnston, M., McKenney, D. W., Pedlar, J. H., Stratton, T., Sturrock, R. N., Thompson, I. D., Trofymow, J. A., & Venier, L. A. (2013). Anticipating the consequences of climate change for Canada's boreal forest ecosystems. *Environmental Reviews*, *21*(4), 322–365.

Price, J., & Maloney, D. (1994). Hydrology of a patterned bog-fen complex in southeastern Labrador, Canada. *Hydrology Research*, *25*(5), 313–330.

R Core Team. (2021). *R: A language and environment for statistical computing*. R Foundation for Statistical Computing. https://www.R-project.org/

Roulet, N. T. (2000). Peatlands, carbon storage, greenhouse gases, and the Kyoto protocol: Prospects and significance for Canada. *Wetlands*, *20*(4), 605–615.

Schiks, T. J., Wotton, B. M., Turetsky, M. R., & Benscoter, B. (2016). Variation in fuel structure of boreal fens. *Canadian Journal of Forest Research*, *46*(5), 683–695.

Schneider, R. R., Devito, K., Kettridge, N., & Bayne, E. (2016). Moving beyond bioclimatic envelope models: Integrating upland forest and peatland processes to predict ecosystem transitions under climate change in the western Canadian boreal plain. *Ecohydrology*, *9*(6), 899–908.

Senici, D., Chen, H. Y., Bergeron, Y., & Ali, A. A. (2015). The effects of forest fuel connectivity on spatiotemporal dynamics of Holocene fire regimes in the central boreal forest of North America. *Journal of Quaternary Science*, *30*(4), 365–375. https://doi.org/10.1002/jqs.2790

Shenoy, A., Kielland, K., & Johnstone, J. F. (2013). Effects of fire severity on plant nutrient uptake reinforce alternate pathways of succession in boreal forests. *Plant Ecology*, *214*, 587–596.

Sherwood, J. H., Kettridge, N., Thompson, D. K., Morris, P. J., Silins, U., & Waddington, J. M. (2013). Effect of drainage and wildfire on peat hydrophysical properties. *Hydrological Processes*, *27*(13), 1866–1874. https://doi.org/10.1002/hyp.9820

Shetler, G., Turetsky, M. R., Kane, E., & Kasischke, E. (2008). Sphagnum mosses limit total carbon consumption during fire in Alaskan black spruce forests. *Canadian Journal of Forest Research*, *38*(8), 2328–2336.

Shi, F.-X., Chen, H.-M., Wang, X.-W., & Mao, R. (2022). Alder encroachment alters subsoil organic carbon pool and chemical structure in a boreal peatland of Northeast China. *Science of the Total Environment*, *850*, 157849.

Stralberg, D., Arseneault, D., Baltzer, J. L., Barber, Q. E., Bayne, E. M., Boulanger, Y., Brown, C. D., Cooke, H. A., Devito, K., Edwards, J., Estevo, C. A., Flynn, N., Frelich, L. E., Hogg, E. H., Johnston, M., Logan, T., Matsuoka, S. M., Moore, P., Morelli, T. L., … Whitman, E. (2020). Climate-change refugia in boreal North America: What, where, and for how long? *Frontiers in Ecology and the Environment*, *18*(5), 261–270.

Sutheimer, C. M., Meunier, J., Hotchkiss, S. C., Rebitzke, E., & Radeloff, V. C. (2021). Historical fire regimes of North American hemiboreal peatlands. *Forest Ecology and Management*, *498*, 119561.

Tarnocai, C., Kettles, I., & Lacelle, B. (2011). *Peatlands of Canada. Geological Survey of Canada, Open File 6551* (p. 10). Natural Resources Canada.

Thompson, C., Mendoza, C. A., & Devito, K. J. (2017). Potential influence of climate change on ecosystems within the Boreal Plains of Alberta. *Hydrological Processes*, *31*(11), 2110–2124. https://doi.org/10.1002/hyp.11183

Thompson, D. K., Benscoter, B. W., & Waddington, J. M. (2014). Water balance of a burned and unburned forested boreal peatland. *Hydrological Processes*, *28*(24), 5954–5964.

Thompson, D. K., Parisien, M.-A., Morin, J., Millard, K., Larsen, C. P., & Simpson, B. (2017). Fuel accumulation in a high-frequency boreal wildfire regime: From wetland to upland. *Canadian Journal of Forest Research*, *47*(7), 957–964.

Thompson, D. K., Simpson, B. N., & Beaudoin, A. (2016). Using forest structure to predict the distribution of treed boreal peatlands in Canada. *Forest Ecology and Management*, *372*, 19–27. https://www-sciencedirect-com.ezproxy.uleth.ca/science/article/pii/S0378112716301463

Thompson, D. K., Simpson, B. N., Whitman, E., Barber, Q. E., & Parisien, M.-A. (2019). Peatland hydrological dynamics as a driver of landscape connectivity and fire activity in the boreal plain of Canada. *Forests*, *10*(7), 534.

Thompson, D. K., & Waddington, J. M. (2013). Peat properties and water retention in boreal forested peatlands subject to wildfire. *Water Resources Research*, *49*(6), 3651–3658. https://doi.org/10.1002/wrcr.20278

Turetsky, M., Donahue, W., & Benscoter, B. (2011). Experimental drying intensifies burning and carbon losses in a northern peatland. *Nature Communications*, *2*(1), 514.

Turetsky, M., Wieder, K., Halsey, L., & Vitt, D. (2002). Current disturbance and the diminishing peatland carbon sink. *Geophysical Research Letters*, *29*(11), 21-21–21-24.

Turetsky, M. R., Amiro, B. D., Bosch, E., & Bhatti, J. S. (2004). Historical burn area in western Canadian peatlands and its relationship to fire weather indices. *Global Biogeochemical Cycles*, *18*, GB4014. https://doi.org/10.1029/2004GB002222

Turetsky, M. R., Benscoter, B., Page, S., Rein, G., Van Der Werf, G. R., & Watts, A. (2015). Global vulnerability of peatlands to fire and carbon loss. *Nature Geoscience*, *8*(1), 11–14.

Turetsky, M. R., Crow, S. E., Evans, R. J., Vitt, D. H., & Wieder, R. K. (2008). Trade-offs in resource allocation among moss species control decomposition in boreal peatlands. *Journal of Ecology*, *96*, 1297–1305.

Turetsky, M. R., & Louis, V. L. S. (2006). Disturbance in boreal peatlands. In R. K. Wieder & D. H. Vitt (Eds.), *Boreal peatland ecosystems* (pp. 359–379). Springer.

Vitt, D. H. (2006). Functional characteristics and indicators of boreal peatlands. In R. K. Wieder & D. H. Vitt (Eds.), *Boreal peatland ecosystems* (pp. 9–24). Springer.

Waddington, J., Morris, P., Kettridge, N., Granath, G., Thompson, D., & Moore, P. (2015). Hydrological feedbacks in northern peatlands. *Ecohydrology*, *8*(1), 113–127. https://doi.org/10.1002/eco.1493

Waddington, J., Thompson, D., Wotton, M., Quinton, W., Flannigan, M., Benscoter, B., Baisley, S. A., & Turetsky, M. (2012). Examining the utility of the Canadian Forest fire Weather Index System in boreal peatlands. *Canadian Journal of Forest Research*, *42*(1), 47–58.

Walker, X., Rogers, B., Veraverbeke, S., Johnstone, J., Baltzer, J., Barrett, K., … Dieleman, C. (2020). Fuel availability not fire weather controls boreal wildfire severity and carbon emissions. *Nature Climate Change*, *10*, 1–7.

Wang, H., Richardson, C. J., & Ho, M. (2015). Dual controls on carbon loss during drought in peatlands. *Nature Climate Change*, *5*(6), 584–587.

Weltzin, J. F., Bridgham, S. D., Pastor, J., Chen, J., & Harth, C. (2003). Potential effects of warming and drying on peatland plant community composition. *Global Change Biology*, *9*(2), 141–151. https://doi.org/10.1046/j.1365-2486.2003.00571.x

Whitman, E., Parisien, M.-A., Thompson, D. K., & Flannigan, M. D. (2019). Short-interval wildfire and drought overwhelm boreal forest resilience. *Scientific Reports*, *9*(1), 1–12.

Whitman, E., Parks, S. A., Holsinger, L. M., & Parisien, M.-A. (2022). Climate-induced fire regime amplification in Alberta, Canada. *Environmental Research Letters*, *17*(5), 055003.

Wieder, R. K., Scott, K. D., Kamminga, K., Vile, M. A., Vitt, D. H., Bone, T., Xu, B., Benscoter, B. W., & Bhatti, J. S. (2009). Postfire carbon balance in boreal bogs of Alberta, Canada. *Global Change Biology*, *15*(1), 63–81. https://doi.org/10.1111/j.1365-2486.2008.01756.x

Wilkinson, S., Moore, P., Flannigan, M., Wotton, B., & Waddington, J. (2018). Did enhanced afforestation cause high severity peat burn in the Fort McMurray Horse River wildfire? *Environmental Research Letters*, *13*(1), 014018.

Wilkinson, S., Tekatch, A., Markle, C., Moore, P., & Waddington, J. (2020). Shallow peat is most vulnerable to high peat burn severity during wildfire. *Environmental Research Letters*, *15*(10), 104032.

Wilkinson, S. L., Moore, P. A., & Waddington, J. M. (2019). Assessing drivers of cross-scale variability in peat smoldering combustion vulnerability in forested boreal peatlands. *Frontiers in Forests and Global Change*, *2*, 84.

Wilkinson, S. L., Verkaik, G. J., Moore, P. A., & Waddington, J. M. (2020). Threshold peat burn severity breaks evaporation-limiting feedback. *Ecohydrology*, *13*(1), e2168.

Wulder, M. A., White, J. C., Bater, C. W., Coops, N. C., Hopkinson, C., & Chen, G. (2012). Lidar plots—A new large-area data collection option: Context, concepts, and case study. *Canadian Journal of Remote Sensing*, *38*(5), 600–618.

Yarrow, M. M., & Marín, V. H. (2007). Toward conceptual cohesiveness: A historical analysis of the theory and utility of ecological boundaries and transition zones. *Ecosystems*, *10*(3), 462–476.

How to cite this article: Jones, E. A., Chasmer, L. E., Devito, K. J., & Hopkinson, C. D. (2024). Shortening fire return interval predisposes west-central Canadian boreal peatlands to more rapid vegetation growth and transition to forest cover. *Global Change Biology*, *30*, e17185. https://doi.org/10.1111/gcb.17185