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Ecological impacts of shortening fire return intervals on boreal peatlands and transition zones using integrated in situ field sampling and lidar approaches

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Abstract

Aridity associated with rising air temperatures in northern latitudes is expected to contribute to increased frequency of wildland fires. Here, we examined regenerating vegetation following short return interval (SRI) fire (56 years post-fire) compared to long return interval (LRI) fire (>80 years post-fire) in boreal peatlands and their adjacent transitional areas. The objectives of this study were to quantify if differences exist between (1) peatland and transitional soil characteristics in LRI versus SRI areas and (2) regenerating vegetation species, structural characteristics and diversity. We also determined if patterns of vegetation structural characteristics observed using field data also occur across the broader landscape using airborne lidar data. The Utikuma Region Study Area (URSA) is located in central Alberta, Canada. Here, 19 peatlands were sampled, coincident with an airborne lidar survey of the broader region, where 120 peatlands in short and long fire return intervals were identified. We found that SRI transitional areas had significantly deeper organic soil deposits than those found in LRI ($p < 0.0001$). Proportions of regenerating species differed significantly between peatlands and transitional areas in SRI versus LRI, where greater proportion of coniferous species were observed in LRI. Deciduous transitional–upland species and taller post-fire vegetation heights were more commonly found SRI peatlands compared with LRI. This suggest that fires with SRIs in this region may result in enhanced deciduous succession, which may transition boreal peatlands into ecosystems that have some characteristics of transitional and upland forests.

KEYWORDS
aridity, ecosystem succession, fire, lidar, regeneration, remote sensing, return interval, wetlands

1 | INTRODUCTION

Within the western region of the Boreal Plains Ecozone, Canada, peatlands exist at the dry limit of peatland formation, where potential evapotranspiration is equal to or slightly exceeds precipitation inputs during most years (Devito et al., 2016; Yu, 2012). Peatlands in this region are responsive to relatively small shifts in water balance and are, therefore, especially sensitive to climatic changes. Enhanced warming and drying for this region over the next 100 years (Flannigan et al., 1998; Schneider et al., 2009; Thompson et al., 2017) will continue to exacerbate overall water loss within these sensitive peatland ecosystems (Thompson et al., 2017), increasing the potential for greater fire susceptibility (Flannigan et al., 2013; Kettridge et al., 2019; Thompson et al., 2017).

Boreal peatlands are composed of thick layers of insulating moss vegetation, which extend to depths greater than 0.4 m due to gradual processes of carbon sequestration, biomass accumulation and slow rates of decomposition (Benscoter et al., 2011; Kuhry, 1994). The largely moist characteristics of peatlands make them an important refuge from wildland fire, increasing the years between fire compared with surrounding upland forests (Turetsky et al., 2004; Whitman et al., 2019; Zoltai et al., 1998). The period between fires is described as a 'fire return interval'. Estimates of fire return intervals within boreal peatlands typically range from between 100 and 130 years (Turetsky et al., 2004; Wieder et al., 2009), though some peatlands may not burn for over 1000 years (Kuhry, 1994; Rowe, 1983; Zoltai et al., 1998).

Long return intervals between fires are necessary for re-establishment of some peatland vegetation species. For example, *Sphagnum* moss can take decades to re-establish to conditions prior to fire following longer fire return intervals (Benscoter & Vitt, 2008; Kuhry, 1994), while black spruce (*Picea mariana*) continue to increase in stem density for up to approximately 80–90 years post-fire (Sirois & Payette, 1989; Wieder et al., 2009). Peatlands reach maturity at approximately 100–125 years, with successional vegetation community changes becoming less variable over time (Benscoter & Vitt, 2008).

Under normal climatic conditions, post-fire hydro-ecological feedbacks maintain peatland resilience to fire through the maintenance of moisture conditions (reviewed in Nelson et al., 2021). Moisture resilience characteristics that are important for peatland maintenance and reducing the impacts from wildfire include (1) hydrological connectivity to groundwater within peatlands, which may also be expressed as larger peatland complexes. Connectivity and persistent groundwater discharge into peatlands result in greater stability of the depth to water table and soil moisture regimes (Lukenbach, Devito, et al., 2015). (2) High proportional coverage of *Sphagnum* moss regeneration in the years following fire enhance moisture retention during dry periods. *Sphagnum* moss spp. accumulate into hummocks over time, increasing the variability of microtopography (Thompson & Waddington, 2013). (3) Post-fire regeneration of black spruce trees conserve peat moisture due to lower stomatal conductance (compared with deciduous species) and lower overall rates of evapotranspiration (Dang et al., 1997; Whitman et al., 2019). (4) Moss water repellency also conserves water by limiting capillary flow to the surface, thus reducing surface evaporation (Kettridge et al., 2014, 2017; Moore et al., 2017). Changes in the moisture characteristics of peatlands combined with increased climatic drying can increase fire risk, thereby predisposing them to more intense fires and greater burn severity (overall loss of biomass) in the future (Flannigan et al., 2013; Helbig et al., 2020; Kettridge et al., 2019).

In an alternative scenario, increased fire frequency could increase the depth of burn along the margins or transitional areas of peatlands, thereby reducing the depth to water table associated with lowered elevation, resulting in pools of standing water (Hokanson et al., 2016; Lukenbach, Hokanson, et al., 2015). Flooding may reduce the establishment of deciduous species, resulting in acceptable environmental conditions for water-tolerant species. While several studies have examined the post-fire effects of 'normal' return-interval wildland fires on peatlands (up to ~150 years), relatively little is known about the implications of shortening return intervals on boreal peatlands.

During spring/summer of 1956 and 2011, the area north of Utikuma Lake (Alberta, Canada) was disturbed by wildland fire. The fire scars overlap in a 12-km^2 area (Alberta Wildfire, 2018), providing a unique opportunity to examine the impacts of a shortened return interval fire (55 years, herein 'short return interval' [SRI]) and longer return interval fire (burned in 2011, herein 'long return interval' [LRI] with no recorded fires in the last ~80 years), on peatlands and transition zones. Transition zones are defined as the area between peatlands and adjacent upland forests. These transitional areas often have soil profiles, water tables and vegetation species that exist within the characteristics of both peatlands and forested areas (Dimitrov et al., 2014; Lukenbach, Hokanson, et al., 2015; Yarrow & Marín, 2007). The overall aim of this study was to examine potential indicators of sensitivity of peatlands and transition zones to a shortened fire return interval by quantifying differences in post-fire soil characteristics and vegetation regeneration. Specific questions include the following: (a) Do differences in soil characteristics (bulk density, pH and soil depth) exist between LRI fire and SRI fire in peatlands? (b) Do these differences vary between peatlands and transitional areas? (c) Were there differences in the density and variability of regenerating vegetation species and height (representing growth rate), within LRI and SRI peatlands and transition zones? (d) If differences in rate of regeneration exist between LRI and SRI peatlands and transitional areas measured in the field, do we also observe similar trends across the broader landscape using lidar remote sensing? (e) Do we observe differences in peatland fragmentation, as represented by a perimeter to area ratio, in SRI peatlands, in comparison to LRI peatlands? To answer these questions, we pose a series of hypotheses followed by observations from the literature summarized in Table 1.

TABLE 1 Hypotheses aligning with objective questions (a–c, e) based on findings in the literature

Objective/question	Hypothesis	Evidence	Reference
(a) Do differences in soil characteristics exist between LRI fire and SRI fire in peatlands?	Ha—SRI peatland soils will have greater bulk density (upper layers) than LRI peatlands	Burned peat had greater surface soil bulk density than unburned peat due to burning of living upper aerobic layer (catotelm), revealing lower compacted anaerobic layer (acrotelm).	Thompson and Waddington (2013); Benscoter and Vitt (2008); Benscoter et al. (2011)
(b) Do differences vary between peatlands and transitional areas?	Ha—Transitional zones have higher bulk density than peatlands enhancing vulnerability to fire	Greater bulk density increases variations in depth to water table, especially in transitions	Lukenbach, Hokanson, et al. (2015); Hokanson et al. (2018); Nelson et al. (2021)
(c) Were there differences in density and variability of post-fire vegetation species and height in LRI/SRI peatlands/ transition zones?	Ha—Post-fire species in SRI areas will be more diverse than those of LRI areas; therefore, the variations in vegetation structural characteristics, especially height, will also be more variable	Short intervals in adjacent upland forest fires have fewer coniferous recruitment and greater broadleaf regeneration (e.g., aspen). Increasing severity of fires and drying cause a shift to broadleaf species	Whitman et al. (2019); Kettridge et al. (2015); Johnstone et al. (2010); Baltzer et al. (2021)
(e) Do differences in perimeter to area ratio occur between SRI and LRI peatlands?	Ha—SRI peatlands will have greater shape complexity (perimeter: area) compared with LRI peatlands	Complex edges predispose peatlands to greater susceptibility to wildfire, drier soil characteristics	Lukenbach, Hokanson, et al. (2015); Lukenbach et al. (2017); Hokanson et al. (2018)

Note: Alternative hypotheses are presented herein and are denoted as Ha.

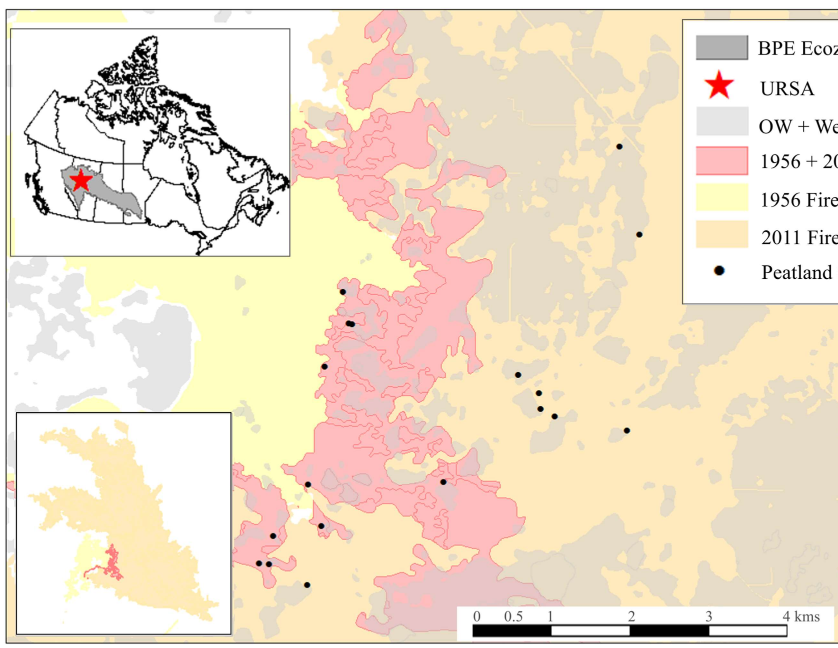

FIGURE 1 The study area is found within the Boreal Plains Ecozone (inset, top). Here, polygons indicate open water (OW) and wetlands (mostly peatlands) found within the areas burned by fire in 1956, 2011 (LRI) and in the area of SRI (1956 and reburned in 2011). Full extent of burn scar from 1956 and 2011 can be seen in bottom inset. Peatland sites represent areas of field data collections, including small plots of 1 m^2 along transects

2 | METHODS

2.1 | Study area

Post-fire peatlands are located in the Utikuma Region Study Area (URSA), about 400 km north of Edmonton, Alberta, Canada, within the Boreal Plain Ecozone (56.107°N, 115.561°W) (Figure 1). The climate in this region is sub-humid with cold, long winters and cool dry summers. Mean annual temperature and cumulative annual precipitation are 1°C and 483 mm, respectively. Annual total evapotranspiration averages 517–519 mm and is the dominant hydrological flux, much of which comes from the many shallow ponds in the study area (Devito et al., 2016; Woo & Winter, 1993). Dry periods are punctuated by 11- and 25-year high precipitation periodic cycles (Mwale et al., 2009). The combined influences of summer precipitation, peatland local topographic position, storage and groundwater/pond inputs/outputs during dry versus wet periods, respectively (Devito et al., 1997; Ferone & Devito, 2004; Smerdon et al., 2005), maintain general site-wide peatland moisture, despite high rates of evapotranspiration relative to precipitation (Smerdon et al., 2005).

The study area is characterized by low relief with three dominant glacio-derived geologic landforms, which influenced the formation and proportion of well-drained mineral upland forests and organic rich wetlands, typical of the Boreal Plains (Devito et al., 2017).

Peatlands are the dominant wetland class (bogs and fens) and represent 30–40% coverage in low-lying regions of coarse textured glacial fluvial outwash landforms. Of these, 20–30% are isolated or loosely connected peatlands on fine-textured hummock moraine landforms, while greater than 60% of peatlands occur over the lacustrine clay plains (Devito et al., 2016, 2017; Thompson et al., 2017). Vegetation within the region consists of predominantly mixed-wood upland forests and large expanses of wetland complexes. Upland forests are dominated by broadleaf deciduous trees, including aspen (*Populus tremuloides*) and balsam poplars (*Populus balsamifera*), as well as coniferous white spruce (*Picea glauca*) (Devito et al., 2016; Petrone et al., 2007). Wetlands are dominated by black spruce (*P. mariana*), bog birch (*Betula glandulosa*) and Labrador tea (*Rhododendron groenlandicum*) as well as *Sphagnum* moss species (Devito et al., 2016).

During early summer 1956, a large wildfire burned approximately 5700 ha within the study region (Alberta Wildfire, 2018), and 55 years later, in May 2011, the Utikuma Complex Wildfires burned approximately 120,000 ha (Alberta Wildfire, 2018; Devito et al., 2016) (Figure 1). The 1956 fire burned eastward over the fine-textured hummocky moraine, partially burning several kilometres into the adjacent peatland dominated clay plain. In contrast, the 2011 fire originated in peatland dominated clay plain and burned to the west, where the fire naturally extinguished after entering the hummocky moraine area. Figure 1 illustrates fire overlap (1956 + 2011, SRI), the LRI area of 2011, and the locations of field transects and a coincident lidar survey. Wetlands within the study area were identified using a wetland/upland classification of Chasmer et al. (2016) and were classified into wetland class (bog/fen) using manual interpretation and field validation. A large portion of this landscape is dominated by forested uplands, open water ponds/lakes, swamps and marshes (some of which can be seen in Figure 1), which were not analysed in this study.

2.2 | Field data collection

To quantifying the influence of SRI and LRI on vegetation and soil characteristics, 19 transects were installed in late July/early August, 2019, coincident with a lidar data collection. Each transect was located perpendicular to the approximate edge of individual peatlands, classified into bog or fen in situ using the Alberta Wetland Classification System. Nine transects were located in the LRI (burned in 2011) and 10 were located in the SRI (burned in 1956 and 2011). These were further divided into three bogs and five fens (LRI) and six bogs and four fens (SRI). Transects were approximately 30 m in length and included 1-m × 1-m micro-plots every other metre and 5-m radius tree plots at either end.

Transects extended across transition zones, towards the centres of peatlands. Transition zones were identified in situ by locating the edge of the peatland, sometimes with identification of the lag and transition from relatively consistent peat depth found within peatlands and gradually reduced peat depth towards uplands (measured using a steel rod). Transition zones also contained mixed peatland- and upland-dominant vegetation, with all factors (lag, vegetation and depth to mineral soil) visually assessed. The transition zone was included within the first 10 m from the visually approximated peatland edge towards the forested upland to the start (0 m) of the transect; thus, in each transect, 10 m consisted of 'transition to upland' while the remainder was peatland.

Start and end points of each transect were located using Global Navigation Satellite Systems (GNSS) and operated for 1 h to resolve transect locations to centimetre accuracy. A tape measure was placed along the transect to locate the position of 1-m × 1-m vegetation plots every 2 m along each transect with bearing recorded. To determine elevation, survey level measurements were recorded at 1-m intervals at the centre of each plot. Micro-plot vegetation measurements included dominant and sub-dominant vegetation overstory, understory and ground cover species, visual and photographic estimates of per cent cover, and understory vegetation height at three locations within each plot (averaged). Per cent cover was estimated for each species/cover type. Depth of peat was also measured at 2-m intervals along the transect by inserting a 2-m steel probe into the peat until it intercepted a mineral lens to a maximum of 1.5 m.

Along each transect, three soil samples were collected within the transition zone and three soil samples within the peatland zone (at the start and end of the transects) at depths of 0–10 cm and 10–20 cm (12 samples in total). Deeper samples were not considered due to the position of the water table. Soil samples were extracted using a steel 'cookie cutter' and a knife, to ensure that the peat/organic layer was not compressed in the extraction process. For the deeper layer, we cut a small area on one side of the sample so that the layer (10–20 cm) could be removed. Measurements were taken in situ for calculation of soil volume using a ruler. Upon return from the field, soil samples were weighed and placed into industrial grade-ovens. Samples were dried at a temperature of 105°C, for a minimum of 72 h or until dry weight remained consistent (maximum 8 days). Samples were weighed approximately every 12 h during this period and a final dry weight was measured and recorded. Dry weight and volume were then used to determine soil bulk density, following methods outlined in Gregorich and Carter (2007). Variations in bulk density can be indicators of intrinsic organic and moisture characteristics of the soil at the location of measurements. In addition to these measurements, dried samples were also used to measure pH using a 1:10 ratio of peat to deionized water via the Vernier Go Direct Tris-Compatible Flat pH sensor (Vernier, Oregon, USA). Soil pH controls many soil chemical and biological reactions, such as the availability of soil phosphorous and nitrogen, and is a determining factor in differentiating bogs and fens (Government of Alberta, 2015; Gregorich & Carter, 2007). These methods follow those described in Gregorich and Carter (2007).

2.3 | Lidar data collection and processing

An airborne lidar survey was acquired on the 29th of July 2019, using a Teledyne Optech Inc., (Ontario, Canada) Airborne Laser Terrain Mapper 'Titan' multi-spectral lidar. The Titan includes three laser emission wavelengths at 1550 nm (shortwave infrared, Channel 1), 1064 nm (near infrared, Channel 2) and 532 nm (green, Channel 3), though the multi-spectral capability of the sensor was not utilized for this study. Lidar data were used to compare differences in vegetation height associated with rate of growth in LRI and SRI peatlands since fire. This provides an opportunity to quantify spatial patterns of vegetation regeneration across a broader range of peatland characteristics than those sampled using field data.

Post-processing of lidar data included classification of ground and non-ground returns, quality control and removal of spurious points using TerraScan (TerraSolid Inc., Finland). Interpolation routines for deriving lidar metrics were performed using LasTools (Rapid Lasso GmbH, Germany) and Surfer (Golden Software, Inc., USA), and derivatives were gridded to a cell resolution of 2 m. A Digital Elevation Model (DEM) was derived using all ground-classified returns from Channel 2 (1064 nm) by generating a triangulated irregular network and rasterizing to a DEM (Las2DEM, Rapid Lasso GmbH, Germany). A canopy height model based on the difference between the digital surface model and the DEM was used to determine vegetation height since the fire across the broader region.

2.4 | Spatial expansion and fragmentation analysis

Wetlands versus uplands (and transitional zones between them) were identified using the high spatial resolution land cover classification map from Chasmer et al. (2016). Peatlands were then identified and classified manually into bogs and fens (with other wetland classes identified but not included here) by comparing with a lower resolution classification of Hird et al. (2017) and image interpretation of pre-fire satellite imagery from Google Earth. Automated classification was not used to reduce the potential for statistical uncertainty between classes. Transition zones were limited to a maximum of 10 m from peatland margins to remain consistent with field transects. To determine if differences exist between SRI and LRI fires in bogs and fens, $\sim$4000 random points were distributed across four main strata to reduce spatial autocorrelation within peatlands: SRI bogs and fens, and LRI bogs and fens. Strata were sampled 1000 times, with a minimum distance of 5 m between each point, used to extract lidar-derived wetland vegetation height. To determine if field observations were representative across the broader region, we used the lidar-derived canopy height model applied to peatlands and transitional zones in LRI and SRI areas. LRI was assessed in 60 peatlands (30 bogs and 30 fens) and SRI in 56 peatlands (30 bogs and 26 fens) across the broader region. Lidar provides greater understanding of the variability of post-fire vegetation regeneration associated with fire regime and environmental drivers.

Perimeter to area ratio (P:A) and peatland area were used to estimate complex shapes, possibly associated with fragmentation of peatlands within LRI and SRI areas. Larger (P:A) values represent greater peatland shape complexity, relative to area of each individual peatland.

2.5 | Statistical analysis

To quantify differences and test hypotheses (Table 1), numerous statistical tests were performed. All measured data variables (excluding depth of organic layer) were non-parametric, determined using a Shapiro–Wilk's normality test. Depth of organic matter (normally distributed) was compared using a parametric *T*-test and a Bartlett's test for analysing the normality of variance. Two-sample Kolmogorov–Smirnov tests were applied to remaining, non-normally distributed variables. Variances between the non-parametric sample populations were tested for significance of differences using the Fligner–Killeen test.

3 | RESULTS

3.1 | Do post-fire soils vary between LRI versus SRI?

A priori comparison with Landsat TM differenced Normalized Burn Ratio (dNBR) between the area of SRI (overlapping 1956 and 2011 fires) and LRI (2011 fire) within peatlands (not shown) did not illustrate significant differences in the severity of the fire between these two areas ($p = 0.12$). This indicates that pre-fire conditions and fire behaviour were not highly variable between the two areas. Shortening of the return interval had significant impacts on some aspects of the soil characteristics within peatlands and transitional areas. We found that depth of organic peat layers within peatlands were not significantly different between LRI and SRI (average = 67.4 cm, stdev ±20.7 cm, and 71.1 cm, stdev ±19.4 cm, respectively). Depth of peat within transitional areas was thinner than that in peatlands and became thicker as measurements approached the peatland centres. However, depth of organic layers within SRI transitional areas were significantly deeper ($p < 0.004$) than those in LRI (average = 50.8 cm, stdev ±23.8 cm [SRI], and 43.6 cm, stdev ±31.9 cm [LRI]; Figure 2).

We hypothesized that soil bulk density in SRI peatlands would be greater than peatlands that had experienced longer return intervals (Table 1). Bulk density increased with depth but was not significantly different between LRI and SRI peatlands (Figure 3a,b, $p = 0.15$, $n = 271$). Transition areas demonstrated significant variations in bulk density between LRI and SRI areas, with significantly greater ($p = 0.03$, $n = 166$) mineral proportions (average = 0.176 g cm^{-3}, stdev ±0.240 g cm^{-3}) than SRI transition zones (average = 0.122 g cm^{-3}, stdev ±0.138 cm^{-3}) (Figure 3a,b).

We found that, on average, pH within peatlands differed significantly depending on fire regime (LRI vs. SRI) ($p < 0.006$), particularly in the top 10 cm below ground surface (Figure 3c,d). All peatlands were acidic, though SRI peatlands had higher pH than LRI peatlands. Despite these differences, we found no significant differences in pH between bogs and fens. We also found that transitional areas pH was not significantly different between LRI and SRI sites.

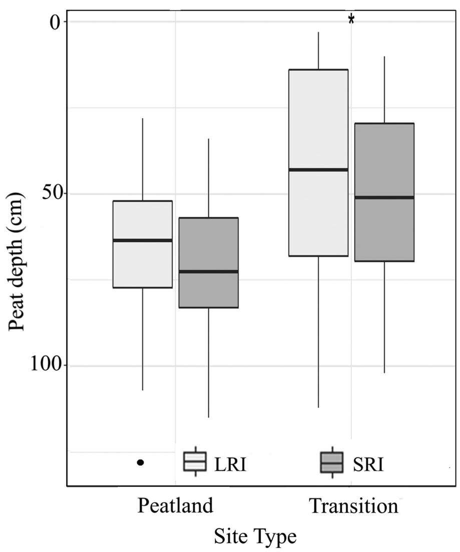

FIGURE 2 Depth of peat (cm) in LRI (long return interval) and SRI (short return interval) fires in peatlands and transition zones. Boxplots represent the interquartile range, where the centre line is the median of the data and whiskers are 1.5*IQR. The asterisk (*) signifies significant differences double burn and single burn transitions. *X*-axis is reversed to indicate depth of peat below ground surface (defined as 0 cm)

3.2 | Do variations in vegetation species occur following long versus short fire return intervals in situ?

Of the total number of field plots within peatlands ($n = 380$), 28% contained little to no vegetation regeneration. They often contained fully scorched (bare) ground, 7 years following fire, with 38% of plots within LRI and 21% of plots within SRI peatlands (sig. different, $p = 0.0003$) (Figure 4). In transitional areas, >99% plots had some vegetation regeneration, including ground surface moss cover, shrubby vegetation or trees (Figure 4). In peatlands, we also observed a greater proportion of scorched *Sphagnum* hummocks in LRI areas (46% of plots) compared with SRI (21% of plots), which had fewer regenerating and/or surviving *Sphagnum* species (Figure 4). Feather moss species experienced minimal regeneration, appearing only in a few of (2%) SRI plots.

Regenerating shrubs in peatlands following LRI fire included high proportions of peatland shrub species, whereas plots in SRI fire peatlands included high proportions of transitional and upland species. For example, 16% of LRI plots in peatlands contained bog birch, while 5% and 27% of plots in SRI peatlands and transitional areas, respectively, contained alder species. Similarly, higher proportions of willow species were also found in peatlands and transitional areas within SRI fire. No alders were found in LRI peatland plots

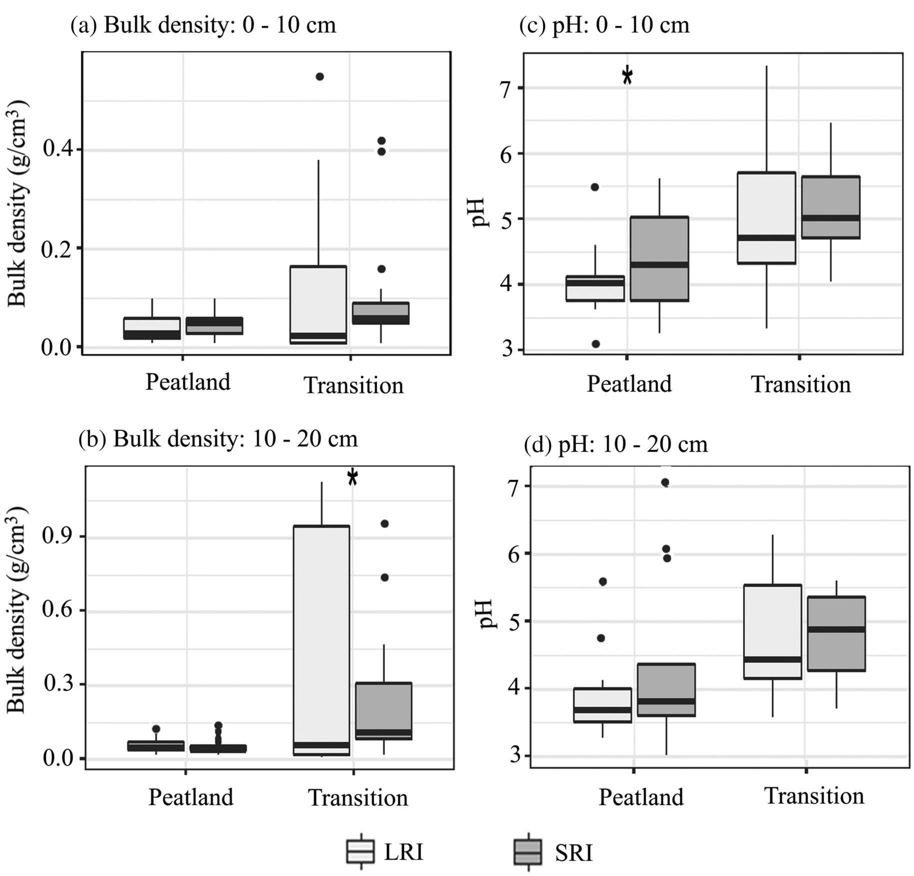

FIGURE 3 Bulk density of soil samples from peatlands and transition zones in LRI and SRI sites at a depth of 0–10 cm (a) and 10–20 cm (b). In the second column, soil pH from peatlands and transition zones in LRI and SRI sites at a depth of (c) 0–10 cm and (d) 10–20 cm

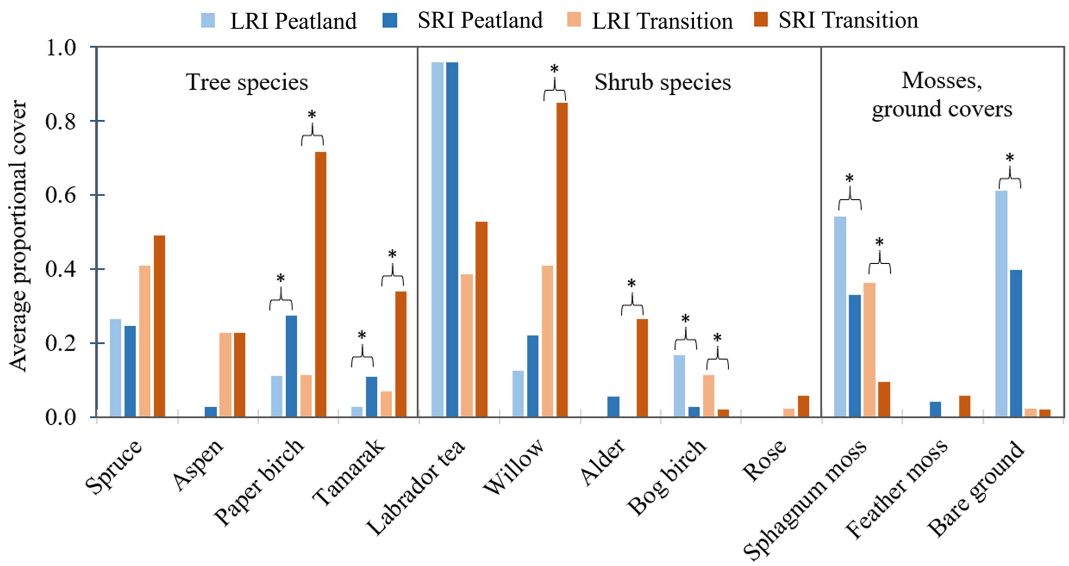

FIGURE 4 Bar plot of vegetation species distribution within LRI and SRI peatlands and LRI and SRI transitions determined from field measurements

($p = 0.05$ and $p < 0.0001$, respectively), which had fewer shrubs in general. With regard to tree species, LRI peatland plots, regardless of wetland class, had significantly higher proportions of black spruce ($p = 0.02$), while SRI plots had higher proportions of paper birch ($p = 0.0003$) and balsam poplar ($p = 0.01$), commonly found in upland forests.

3.3 | Do differences in measured vegetation height/cover occur in long versus short fire return intervals?

Variations in post-fire vegetation structural characteristics, such as height and canopy cover, were found between LRI and SRI areas, often corresponding to differences in regenerating vegetation species composition (e.g., conifer vs. broadleaf trees/shrubs; Figure 4). Here, we found that the average height of regenerating vegetation in SRI peatlands was much taller than in LRI peatlands ($p < 0.000$) (Figure 5). Vegetation heights were also more variable (indicated by height variance) in SRI peatlands when compared with LRI ($p < 0.0001$). This indicates greater complexity of vegetation structures in SRI peatlands (Figure 5), while canopy cover within peatland vegetation plots was greater in LRI (average 76% cover) compared with SRI (average = 53% cover) ($p < 0.0001$).

Measured regenerating vegetation height within transitional zones also varied. Shrubs were taller in transition zones adjacent to peatlands in SRI fire areas compared with LRI ($p = 0.004$) (Figure 5). Tree heights were taller in SRI sites but were not significantly taller (Figure 5). In transitional areas, heights were more variable in SRI areas than LRI ($p < 0.0001$), while LRI had higher canopy cover of trees ($p = 0.04$). There were no significant differences in shrub cover found in LRI and SRI.

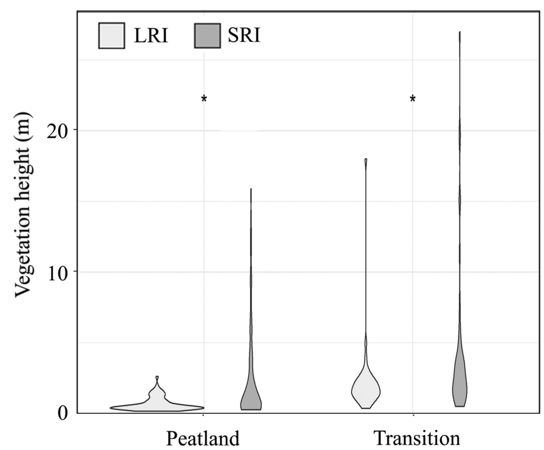

FIGURE 5 Violin plot of field measured heights, where maximum field height measured of the dominant tree or shrub in LRI and SRI peatlands and transition zones. Stretch in the y-direction indicates the range of measured heights and horizontal stretch in the x-direction represents per cent frequency

3.4 | Differentiating post-fire variations in vegetation heights along transects in bogs/fens using lidar

While differences in regenerating vegetation between bog and fen peatland classes may contribute to variations in vegetation structures across these classes, we found no significant differences between LRI bog/fen vegetation heights or SRI bog/fen vegetation heights. We suggest that (1) different sample sizes of bogs and fens between SRI and LRI areas did not contribute significantly to weighting of vegetation height within samples, and (2) differences observed when testing peatland types between fire regimes (LRI vs. SRI) were more likely to be the result of the fire regime than peatland type.

Trends observed in the lidar data, with regard to vegetation heights in peatlands, correspond with plot measurements, illustrating the efficacy of the use of lidar for quantifying post-fire vegetation growth. Within peatlands, both LRI bogs and fens had significantly shorter vegetation compared with SRI bogs and fens ($p < 0.0001$ and $p = 0.0001$, respectively). Further, bogs in SRI had more varied vegetation heights ($p < 0.0001$) compared with those in LRI burned area, while fens in LRI contained the shortest vegetation, overall compared with SRI fens ($p < 0.0001$) (Figure 6).

3.5 | Trends in post-fire vegetation regeneration in LRI and SRI exist across the broader region

Using lidar data acquired across the broader region, we observed slightly different results from our field/lidar-based measurements. Similarities with field measurements included greater proportion of shrubs/trees > 3 m in height in SRI peatlands (32% shrub/tree proportion) compared with LRI peatlands (29%, $p = 0.04$)). Also, vegetation heights in fens were taller in SRI compared with LRI ($p = 0.016$). While vegetation heights in bogs were taller in SRI compared with LRI fires ($p = 0.04$), similar to field measurements, these tended to be shorter overall than vegetation in fens.

Bogs located in SRI areas had significantly greater P:A, indicating that they have more complex shapes (average = 0.20, stdev ±0.12) than those found in LRI areas (average = 0.07, stdev ±0.05; $p < 0.000$) (Figure 7). Fens in SRI areas also had greater P:A (average = 0.10, stdev ±0.09) compared with those of LRI areas (average = 0.08, stdev ±0.09). Table 2 provides a summary overview of the comparisons described between measured bog and fen soil and vegetation characteristics within transects and vegetation structures and shape determined from lidar data and the land cover classification.

4 | DISCUSSION

4.1 | Soil characteristics of SRI and LRI fire in western boreal peatlands

Several studies have demonstrated the sensitivity of transitional zones, bulk density and variability in depth to water table to wildland fire and enhanced burn severity (e.g., Hokanson et al., 2016, 2018; Lukenbach, Hokanson, et al., 2015), while Wilkinson et al. (2018) illustrate the importance of lower surface yield on transition-area drying. Here, we found that peat depth was reduced in transitional zones compared with adjacent peatland (e.g., Lukenbach, Hokanson, et al., 2015; Mayner et al. (2018)). Peatlands in late spring/early summer SRI versus LRI fire had relatively similar depth of peat layer (Figure 2), while peat depths in transitional areas were significantly greater following SRI fire (Figure 2). These results were contrary to Whitman et al. (2019), who found that peat depths were greater in LRI fires compared with SRI fires that occurred in summer in both boreal forests and peatlands. Differences may be due to enhanced drying and disconnection from regional groundwater following evaporative drying in summer in peatlands that had not been burned in recent decades, compared with shallower depth to water table in peatlands that had burned recently (e.g., Kettridge et al., 2014). This may have resulted deeper organic soil layers, higher moisture conditions and overall lower fuel consumption. In our study, we found no significant differences in the depth of the peat layer between SRI versus LRI fires, where peatlands maintained deep organic soils (Figure 2).

We hypothesized that bulk densities would be greater in SRI compared with LRI peatlands, exposing layers of higher density peat within the acrotelm (Table 1). Contrary to expectation, we found that bulk density did not vary significantly between LRI and SRI peatlands.

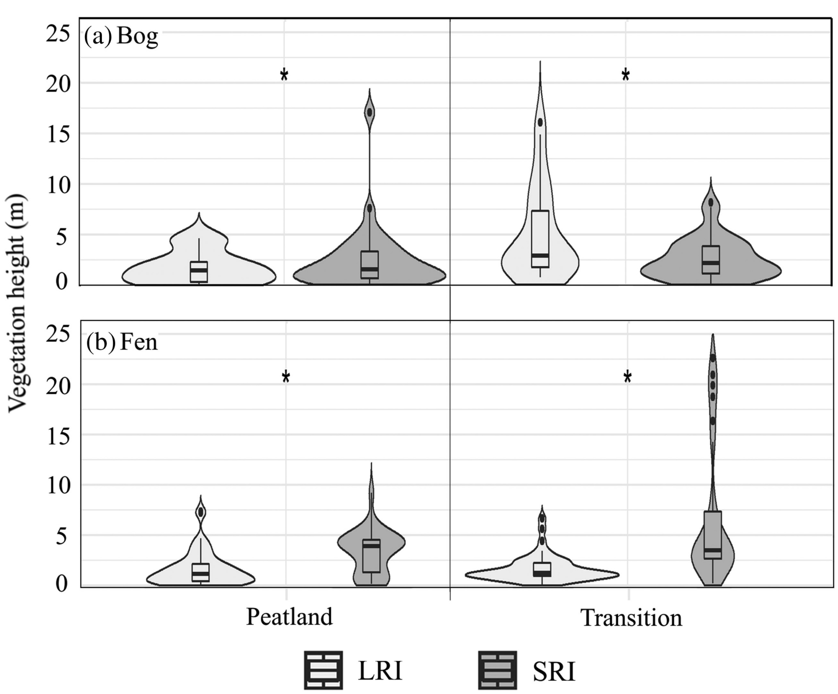

FIGURE 6 Lidar-derived height (m) of vegetation found within LRI and SRI measurement plots and per peatland class

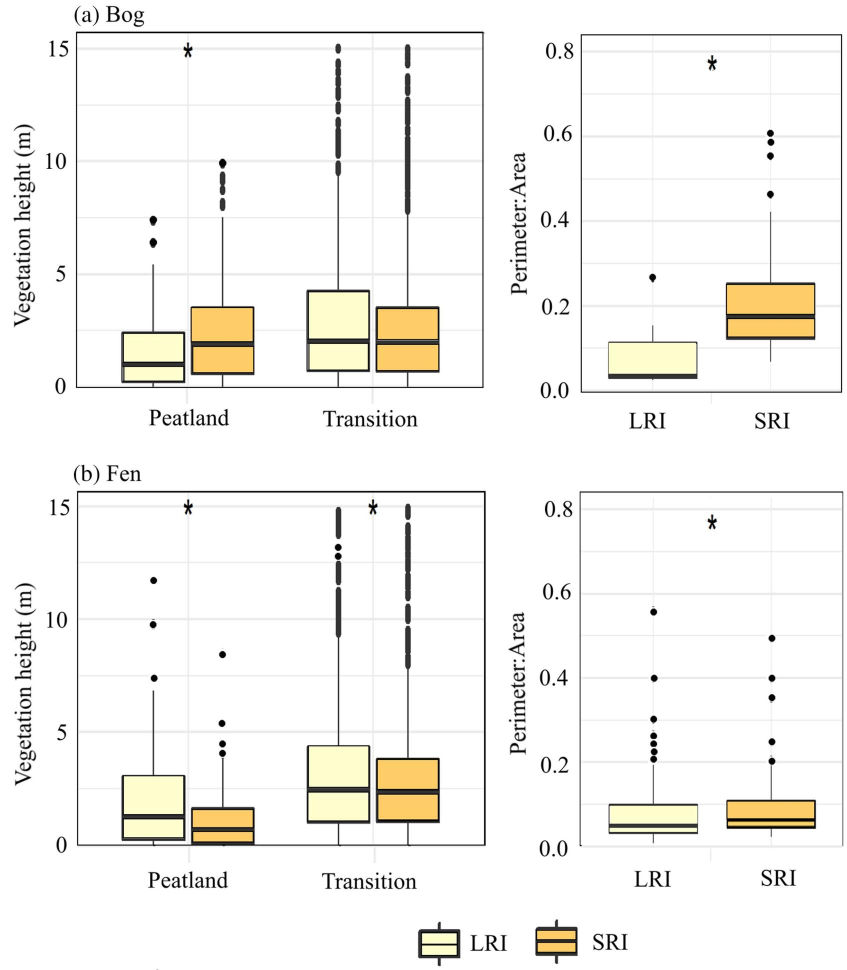

FIGURE 7 Vegetation height at the 95th percentile extracted from lidar data in LRI and SRI peatlands/transitions (left) and perimeter to area ratio (right) illustrating shape complexity of bogs and fens. Increasing P:A indicates increasing complexity, possibly associated with fragmentation or less 'rounded' shapes

TABLE 2 Summary of mean field measured bog, fen and transitional area characteristics for SRI and LRI fires average characteristics (±stdev)

	SRI fire				LRI fire			
Characteristics (units)	Bog	Bog transition	Fen	Fen transition	Bog	Bog transition	Fen	Fen transition
Soil bulk density (g cm^{-3})	0.05 (0.04)	0.12 (0.05)	0.04 (0.02)	0.16 (0.05)	0.06 (0.05)	0.40 (0.21)	0.03 (0.02)	0.19 (0.04)
Soil pH	4.23 (0.9)	5.07 (0.6)	4.52 (0.8)	5.04 (0.6)	3.76 (0.3)	5.43 (0.9)	4.02 (0.5)	4.69 (0.8)
Depth of peat (cm below ground)	68.6 (18.9)	50.7 (29.5)	58.5 (29.5)	38.1 (27.5)	41.8 (26.5)	15.9 (10.2)	68.0 (23.6)	61.3 (43.0)
Vegetation height (m)	2.75 (0.4)	3.37 (1.3)	1.85 (0.8)	5.74 (2.0)	1.71 (0.5)	2.90 (1.3)	0.73 (0.2)	1.77 (1.1)
Lidar vegetation height (m)	2.5 (2.1)	3.64 (1.1)	2.29 (1.6)	6.78 (2.6)	1.93 (1.3)	5.00 (1.7)	1.5 (0.9)	1.69 (1.2)
Lidar P:A	0.2 (0.12)	-	0.10 (0.09)	-	0.07 (0.05)	-	0.08 (0.09)	-

However, transition zones were characterized by higher mineral soil content, more densely packed organic materials and, therefore, higher bulk density (Table 2). Wilkinson et al. (2019) also found similar results, where greater differences exist between boreal peatland transition bulk densities (and further enhance fire) in coarse-grained glaciofluvial hydrogeological settings. Greater bulk densities and variability in depth to water table result in more severe fire within transition zones (Hokanson et al., 2018; Lukenbach, Hokanson, et al., 2015). Further, LRI transition zones had significantly greater mineral soil content, higher bulk densities and greater variability of bulk density than those found in the SRI areas (Figure 3; Table 2). These characteristics result in enhanced risk of surface fire within 10 years within LRI peatlands (e.g., Johnston et al., 2015).

4.2 | Differences in regenerating vegetation species and structural characteristics

We hypothesized that peatlands with SRI fire may experience a shift to upland broadleaf tree and shrub species in the years following fire

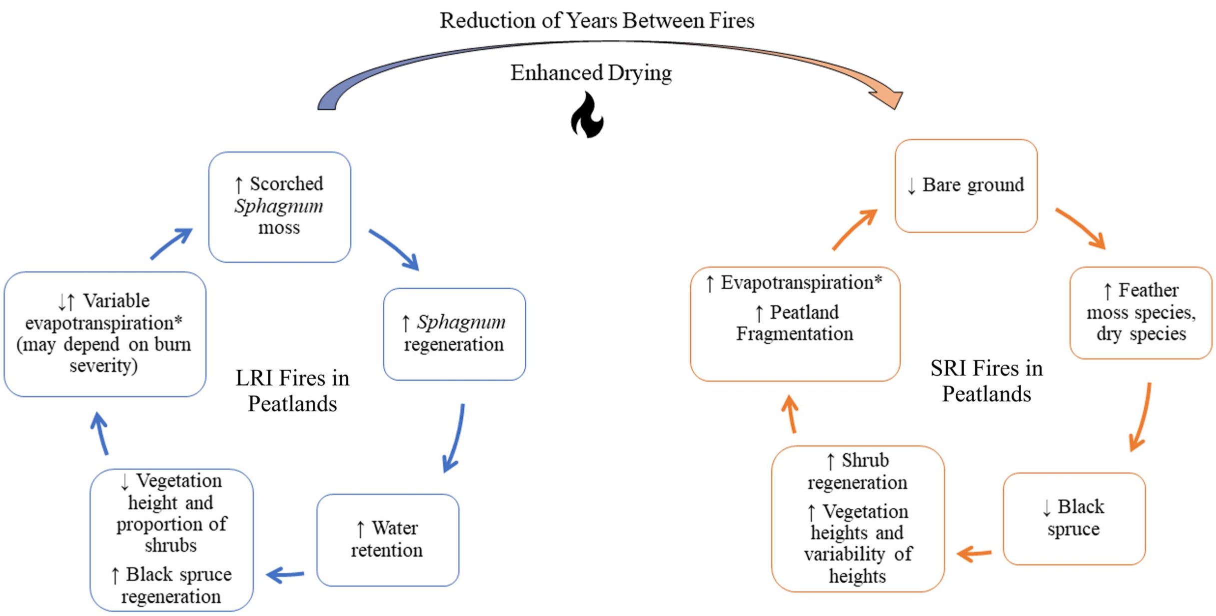

FIGURE 8 Feedback schematic based on the findings observed here using field and lidar methods. Notes marked with a * are predicted conclusions based on known literature. This schematic supports hypotheses in Kettridge et al. (2019) and Nelson et al. (2021) observed for (a) minimal sites and (b) within a review of the literature

(Table 1). Vegetation plots in LRI burned peatlands had significantly larger proportions of scorched, non-productive (bare) ground than that found in SRI (Figure 4). Large areas of scorched ground are not unusual in regenerating peatlands, as peatland species can take up to 20 years to re-establish to a significant seeding community (Benscoter & Vitt, 2008).

The composition of regenerating tree and shrub species differed significantly between fire regimes. Peatlands experiencing SRI or high intensity fire had higher proportions of broadleaf deciduous species and reduced black spruce (Baltzer et al., 2021; Johnstone et al., 2010; Whitman et al., 2019). The shift to broad-leaf species has significant implications for (1) the 'shrubificiation' or terrestrialization of peatlands, where increased deciduous canopy cover may enhance transpiration and peatland moisture deficits (Nicholas Kettridge et al., 2013), colonization of feathermoss species (Bisbee et al., 2001) and reduction in the overall area of peatlands (Thompson et al., 2017); and (2) changes to returning fuels including broadleaf establishment (which typically have highly variable foliar moisture content; Alexander, 2010) and feathermosses, which are more combustible than *Sphagnum* (Nelson et al., 2021; Shetler et al., 2008; Turetsky et al., 2010).

SRI burned peatlands had significantly taller shrubs than those found LRI peatlands, which further enhance biomass availability (Figure 6). This could have significant implications for future wildland fire in peatlands. For example, Van Altena et al. (2012) found that slow duration of combustion and highest fire temperatures (of five species examined) were found in boreal *Betula pubescens*. Greater density and taller birch species encroachment may similarly enhance peatland fires, especially in spring prior to leaf flush. Vegetation in SRI peatlands and transitional areas also had greater variability of vegetation height compared with LRI peatlands/transition areas (species composition hypothesis; Table 1). Greater aerodynamic roughness length of canopies increases evaporative losses through enhanced turbulence, thereby increasing stomatal conductance (e.g., Green et al., 2021).

Across the broader area of peatlands and transitional zones, shrubs and trees were taller and covered greater proportional areas in SRI versus LRI burned areas, determined from lidar data (Figure 7). Differences indicate enhanced productivity of early post-fire regeneration woody species following SRI fire. Similar results were also observed in Ward et al. (2014) who found that that fire increased productivity to up to 80 years post-fire. Expansion of the aerobic zone also changes the conditions for diverse upland shrub/tree growth by altering hydrology, peat mineralization, and nutrient cycling (Strack et al., 2006) and regeneration of bryophytes found in mineral uplands (Lukenbach et al., 2017).

4.3 | Peatland fragmentation and fire return interval

Peatlands with smaller areas and higher perimeter to area (P:A) ratio (greater length of edges) are more vulnerable to transition-area drying, disconnection from ground water and enhanced climatic drying (Hokanson et al., 2016, 2018; Wilkinson et al., 2018). We found that peatlands that have higher P:A are found predominantly in areas burned by SRI fire, especially in bogs (Figure 7). Kettridge et al. (2019) and Thompson and Waddington (2013) suggest a positive feedback, whereby shortening of the fire return interval results in progressively reduced peatland area associated with the combustion of peat moss in combination with shrub encroachment, resulting in increased P:A observed here. Fens tend to be connected to regional groundwater storage and are less sensitive to moisture deficits in the study area (Devito et al., 2012). The morphology of fens results in lower P:A ratio, which is not significantly different between SRI and LRI fires, and lower vegetation height compared with bogs. P:A ratio is also influenced by variability in regional geological gradients. Here, more expansive peatlands exist in fine-grained lacustrine clay plains, while smaller peatlands that are more susceptible to fire are found in heterogeneous fine-grained moraines (Devito et al., 2012) at the intersection of the SRI fire.

From this study (summarized in Figure 8), we find evidence to suggest that shortening of the return interval of fires in this region reduces the proportion of peatland vegetation species, while enhancing mixed deciduous species encroachment from uplands. This increases perimeter:area ratio, indicated by increased fragmentation. Longer edges, relative to peatland area, increase edge effects and susceptibility to fire, especially in areas of hummocky moraines where peatlands may be disconnected from groundwater.

5 | CONCLUSIONS

Our results suggest that, in this area, peatlands that had been burned by SRI fires tended to have taller vegetation (Figure 5) and greater prevalence of shrub species (Figure 4) compared with peatlands burned during LRI fire. These results corroborate findings presented in Baltzer et al. (2021) on black spruce resilience in the western boreal plains: Peatlands with SRI fires had greater establishment of boreal deciduous species. In this area, we find that soil bulk density rather than depth of peat corresponds with higher prevalence of upland deciduous species over black spruce. This provides evidence to suggest that SRI fires could reduce black spruce resilience and may transition boreal peatlands towards upland forest land covers over time. We found similar results associated with vegetation heights along transects using airborne lidar data, while across the broader region, bogs had significantly taller vegetation heights than fens (Figure 7).

The results of this study could have significant implications for drying in areas where climate-mediated changes result in a shortening of the fire return interval. Further, these findings may have implications in fuel development, especially where SRI fires in transitional areas have higher proportions of deciduous upland species. While deciduous species have higher foliar moisture content and are considered less flammable, variability in moisture content (Alexander, 2010) and dry shrubs in spring may enhance flammability. Over broader regions, Thompson et al. (2019) suggest that atmospheric drying, combined with peatland fragmentation (and enhanced forest connectivity), initiates the spread of large boreal fires, such as those that occurred in NWT in 2014.

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DATA AVAILABILITY STATEMENT

The data that support the findings of this study are available from the corresponding author upon reasonable request.

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