Forest Ecology and Management 473 (2020) 118283

Contents lists available at ScienceDirect

Forest Ecology and Management

journal homepage: www.elsevier.com/locate/foreco

How much boreal lake shoreline is burned by wildfire? Implications for emulating natural disturbance in riparian forest management

Md.S. Newaz[2], Robert W. Mackereth[1], Azim U. Mallik*, Darren McCormick[1]

Department of Biology, Lakehead University, Thunder Bay, Ontario P7B 5E1, Canada

ARTICLE INFO

Keywords:
Shoreline buffer
ArcGIS, Ontario
Best management practice
Riparian buffer
END-based forest management
Wildfire

ABSTRACT

Understanding of post-fire residual vegetation patterns is important when shoreline vegetation management aims to emulate natural disturbance (END) patterns. To assess the impacts of fire, lake and lakeshed sizes on the burning pattern of shorelines and lakesheds, we quantified the burned shorelines and post-fire residual vegetation patterns in the lakesheds of 123 lakes of Ontario affected by 26 wildfires between 2005 and 2007. We used ArcGIS and Ontario's Enhanced Forest Resources Inventory (eFRI) GIS data to digitize burn patterns. The lake catchments for all lakes were delineated using ArcGIS via lake, river, and elevation data from the Integrated Hydrology geodatabase (MNRF, 2016). The shorelines of fire impacted lakes were generated from the eFRI polygon feature classes, and the polylines were then split according to the digitized burn pattern polygons by running a geometric intersection with these data. The results of this study show that the percentages of burned shorelines and lakesheds are positively correlated with the size of fire and negatively correlated with the sizes of lake and lakeshed. However, irrespective of the size of fire, lake or lakeshed, shorelines are not left completely unburned, which is contrary to existing practice of retaining fixed-width shoreline buffers. It may imply that under END based management, forest harvesting can be possible up to the shorelines in some areas of the landscapes that are left unharvested under a fixed-width riparian buffer management system. However, areas of strong hydrological connectivity between land and water serve as biogeochemical control points and require protection from disturbance during forest management planning and operations. We suggest that GIS-based models developed based on the hydrological and topographical features associated with the unburned shorelines and lakesheds might be useful to predict shoreline residual forest pattern and facilitate END based shoreline forest management.

1. Introduction

The numerous streams, rivers and lakes in the boreal forest have riparian zones of high structural and functional complexity (Sokal et al., 2010). Riparian zones are the interface between land and water, with ecosystem characteristics and biotic communities distinct from both and are naturally adapted to variable environmental conditions (Naiman and Décamps, 1997; Lamb and Mallik, 2003). These are areas of reciprocal influences between aquatic and terrestrial components with varying dimensions depending on geomorphology particularly topography (Richardson et al., 2005; Kuglerová et al., 2015). The ecological importance of riparian areas greatly exceeds their areal extent on the landscape as they support a wide range of plant, animal and microbial communities (Clary and Medlin, 1993). The complex and dynamic structural and functional diversity of the riparian ecosystem provides many important ecological services such as control of surface run-off and soil erosion, stabilization of stream banks and prevention of sedimentation, maintenance of high water quality, habitat for invertebrate communities and travel and migration corridors to animals (Naiman and Décamps, 1997; NRC, 2002; Gundersen et al., 2010; Kuglerová et al., 2014). Although riparian ecosystems are shaped by disturbances, natural (predominantly wildfire) and anthropogenic (mainly forest harvesting) disturbances have considerable differences in their influence on the resilience and underlying dynamics of riparian

* Corresponding author.
E-mail address: amallik@lakeheadu.ca (A.U. Mallik).
[1] Centre for Northern Forest Ecosystem Research, Ontario Ministry of Natural Resources and Forestry, 103–421 James Street S., Thunder Bay, Ontario P7E 2V6, Canada.
[2] Current address: Centre for Northern Forest Ecosystem Research, Ontario Ministry of Natural Resources and Forestry, 103–421 James Street S., Thunder Bay, Ontario P7E 2V6, Canada.

https://doi.org/10.1016/j.foreco.2020.118283
Received 12 March 2020; Received in revised form 16 May 2020; Accepted 26 May 2020

forests (McRae et al., 2001). Clearcut harvesting leaves more deciduous trees compared to severe wildfire leading to divergence in species composition between harvested and burned sites (Kemball, 2002; Simon and Schwab, 2005). Deeper and more heterogeneous edges and better conditions for seed germination and survival facilitate more abundant regeneration in burned sites compared to harvested sites (Kemball, 2002; Harper et al., 2004; Weyenberg et al., 2004). Furthermore, burned sites take less time to return to the pre-disturbance state of overstory species composition compared to clearcut sites (Ehnes, 1998).

Forest harvesting along lakes and stream shorelines affects the biophysical environment of the lakes (Kreutzweiser et al., 2009). Best management practices were introduced to mitigate the adverse impacts of forest harvesting in riparian zones by keeping unharvested riparian buffers (Thorell and Götmark, 2005; Trenholm et al., 2013). Shoreline forest management in Canada is mainly focused on mitigating potentially negative impacts to aquatic ecosystems by establishing fixed-width buffers at the land–water interface. Fixed-width intact, mature forest in shoreline areas is operationally easy to implement and effective in mitigating harvest induced hydrological and biogeochemical changes to some extent (Kreutzweiser et al., 2013; Sweeney and Newbold, 2014; Laudon et al., 2016). However, it cannot accommodate the natural range of variability in riparian forest composition and function and fails to maintain the diversity of the ecosystem that is historically achieved from natural disturbance such as wildfires (Naylor et al., 2012). Fixed-width buffers are based on the assumption that riparian patterns and processes are homogenous across the riparian area (Richardson et al., 2012), although riparian functions, biodiversity, hydrology and biogeochemistry vary considerably at small spatial scales (Kuglerová et al., 2014; Kuglerová et al., 2017; Leach et al., 2017). In fact, full protection of the waterbodies cannot be achieved through fixed width buffer zones which do not consider the hydrological and biogeochemical heterogeneity in riparian patterns and processes (Dosskey et al., 2012). It also appears to be contrary to an END based forest management strategy because it creates unnatural, linear patterns of old-growth forests along streams and around lakes (Buttle, 2002). Fixed-width buffers are also inefficient in terms of cost-benefit analysis (Tiwari et al., 2016). Hence many ecologists (Qiu et al., 2009; Sweeney and Newbold, 2014; Moussaoui et al., 2016) proposed site-specific and targeted ecological function based variable width buffer zones as an optimum solution for protecting riparian ecosystem structure and functions. Improved environmental protection can be provided to both surface water biogeochemistry and biodiversity by adapting site-specific hydrological condition-based buffer zones (Kuglerová et al., 2014). It has been suggested that forest landscapes created by hydrologically adapted buffers are more similar in composition and structure to landscapes created by natural disturbance such as wildfires than fixed-width buffers (Crow and Perera, 2004).

Forest fire is the dominant natural disturbance that strongly influences the structure and dynamics of boreal forests (Shenoy et al., 2011; Chambers et al., 2016; Whitman et al., 2018). Periodic wildfires significantly affect the understory vegetation and create succession patterns resulting in a mosaic of age classes and communities (Whitman et al., 2018). Forest fires often leave single or large groups of living trees within the burned areas (Bergeron et al., 2001) and these residuals serve as source pools or transitionary refuges, contribute as seed sources to recolonize the burned areas and conserve native biodiversity. The effects of forest fires can extend up to the edge of water often leaving the residual mature riparian forests intact in areas of wet soils or groundwater discharge, creating patchy riparian forests with stands of early successional regeneration interspaced with patches of older trees (Kreutzweiser et al., 2012). Since current forest management in Ontario operates under the END based management paradigm (OMNR, 2010; OMNR, 2014) and a reasonable range of natural variation in pattern and function may be achieved through careful harvesting in riparian buffers instead of fixed-width, no-harvest buffers (Naylor et al., 2012), models developed based on post-fire residual forest patterns might be useful in redesigning riparian buffers to mimic natural disturbance patterns. This may achieve the objectives of maintaining ecosystem integrity and establish a balanced ecological and economic trade-off between varying riparian buffer retention based on topography and hydrology.

Post-fire residual forest patterns in the boreal forest of Ontario are not yet well documented. The limited research done to date is mostly associated with the riparian zones of running water courses such as streams and rivers. The shoreline riparian forests of the more than 250,000 lakes in Ontario are rarely studied. To redesign riparian buffers around boreal forest lakes to better emulate natural disturbance patterns, forest managers should have a suite of disturbance pattern metrics representative of the wildfires along water edges to assist forest management planning. Hence, it is important to describe and quantify the post-fire residual forest patterns along boreal forest lakes and their underlying topographical and hydrological features. The objectives of this research were to (i) quantify post-fire shoreline residual vegetation patterns across the boreal forests in Ontario; and (ii) assess the impacts of fire, lake and lakeshed sizes on shoreline and lakeshed burning patterns.

2. Materials and methods

2.1. Study area

Our study is based on 26 natural forest fires which occurred between 2005 and 2007 within the boreal and Great Lakes-St. Lawrence forests, across three ecoregions (Lake Abitibi, Lake Nipigon and Pigeon River ecoregions) of the Ontario Shield Ecozone covering an area of approximately 246,000 km^2 (Fig. 1). The study area stretched across four watershed regions of Ontario, namely North West (NW), Far North Central 1 (FNC1), North Central (NC) and Far North East (FNE). The climate of the area is cold and moist with long cold winters and short warm summers. However, temperature, precipitation and humidity vary widely. Mean daily temperatures in January and July are -15° and 17 °C, respectively, being more moderate in the southern areas and around the Great Lakes. Mean annual precipitation ranges from 500 mm in the west to 850 mm in the east (Crins et al., 2009). The area is comprised of Precambrian bedrock with diverse surficial geology and substrates. However, a significant portion of the area has exposed bedrock. Topography of the area varies greatly depending on local bedrock and surficial deposits, which makes surface drainage patterns very complex. Rivers and lakes are abundant in most parts of the area. Since the drainage divide between the Great Lakes system and Hudson Bay is closer to the Great Lakes, most of this area of the province drains towards the north. Local variations in climate and soil result in local variations in runoff ratios (Baldwin et al., 2011). The presence of numerous wetlands, lakes and streams and interrupted drainage systems lead to irregular patterns in the forest of the area.

Most of the timber management in Ontario takes place in the boreal and Great Lakes-St. Lawrence forests. Boreal forests of the central and northern parts of the study area are adapted to a cold climate and frequent fires and are predominantly composed of black spruce (*Picea mariana* (Mill.) B.S.P.), balsam fir (*Abies balsamea* (L.) Mill.), jack pine (*Pinus banksiana* Lamb.), tamarack (*Larix laricina* (Du Roi) K. Koch) along with the hardwood species, white birch (*Betula papyrifera* Marsh.) and poplars (*Populus* spp.). The southern part is dominated by mixed and deciduous forest of tolerant hardwood species like sugar maple (*Acer saccharum* marsh.), yellow birch (*Betula alleghaniensis* Britt.) and American beech (*Fagus grandifolia* Ehrh.) (Crins et al., 2009). Fire is the dominant natural disturbance in the area. However, the frequency, intensity and burn size vary depending on the prevailing weather conditions, local topographic and hydrologic features and predominant vegetation (Thompson, 2000).

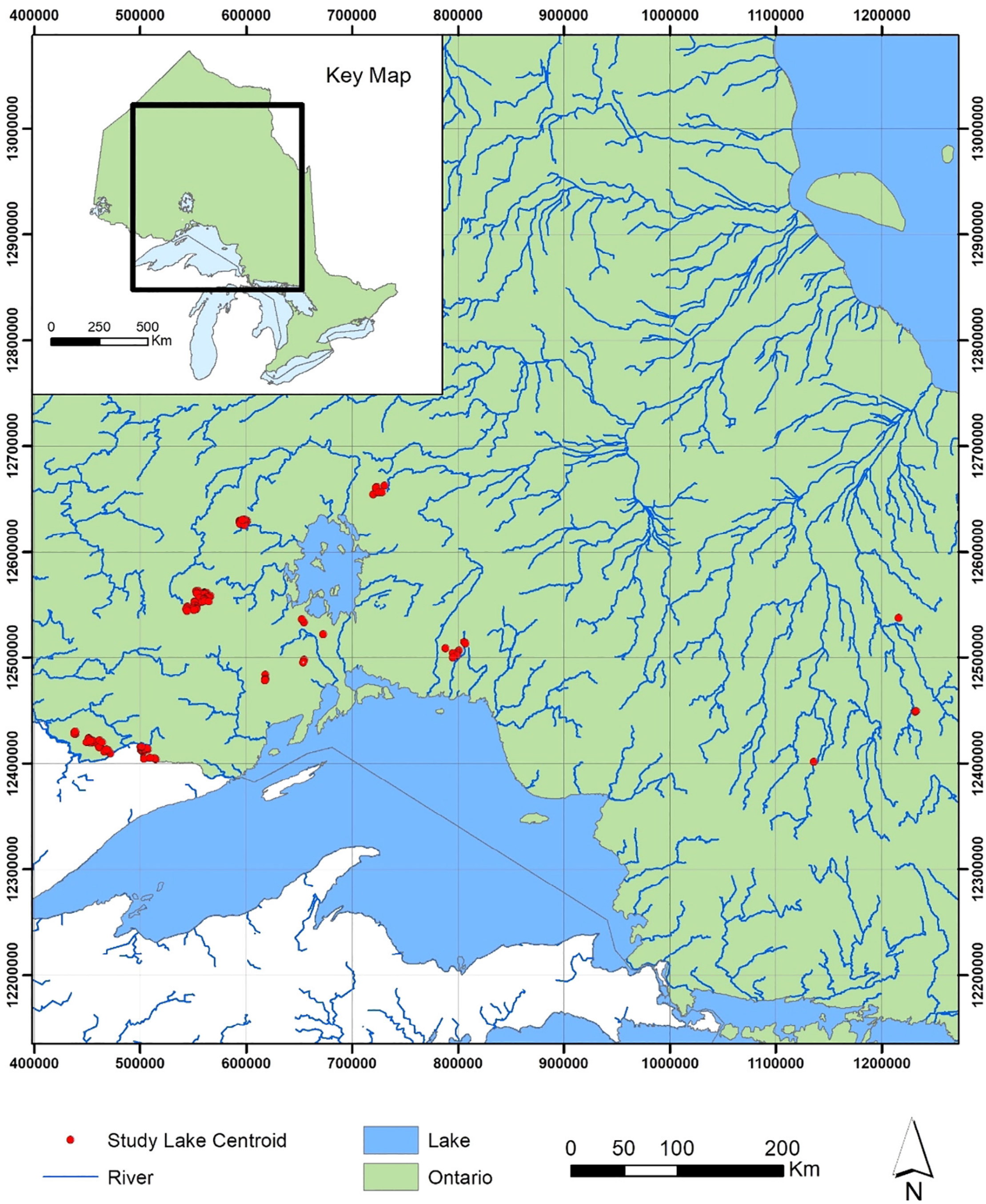

Fig. 1. Map of the study area showing the centroids of the study lakes.

2.2. Data

We used ArcGIS and Ontario's Enhanced Forest Resources Inventory (eFRI) GIS data (polygons and orthorectified, digital 2D aerial imagery (40 cm resolution, 4-band multi-spectral, 16-bit) to digitize burn patterns associated with fires that intersected lakesheds within the study area. Lakesheds were defined as the total watershed area upstream of the lake outflow point excluding the watershed area of all upstream lakes ≥ 5 ha surface area (i.e., the terrestrial portion of the watershed contributing directly to the lake via surface and subsurface flow not passing through another lake). The lake catchments for all lakes were delineated using ArcGIS via lake, river, and elevation data from the Integrated Hydrology geodatabase (MNRF, 2016). Aerial imagery collection was organized by Forest Management Units (FMUs) between 2006 and 2009. Natural fires, > 40 ha, that burned within two years of image collection were selected for this study. To capture burn patterns within the extent of the lakeshed both burned and unburned residual polygons were digitized. Polygons adjacent to lakes were closed to the perimeters of lake polygons represented in the eFRI GIS data. The shorelines of lakes impacted by fire were generated from the eFRI polygon feature classes, and these polylines were split according to the digitized burn pattern polygons by running a geometric intersection with the data. Using the eFRI imagery, the split shorelines were classified into six disturbance classes: (1) burned forest, (2) unburned forest, (3) burned organic soil, (4) unburned organic soil, (5) burned shallow soil, and (6) exposed bedrock. However, in the analyses, shoreline burned forest, burned organic soil and burned shallow soil were combined and referred to as burned shoreline while unburned forest and unburned organic soil were combined and called unburned shoreline. Each of the continuous burned or unburned split shoreline

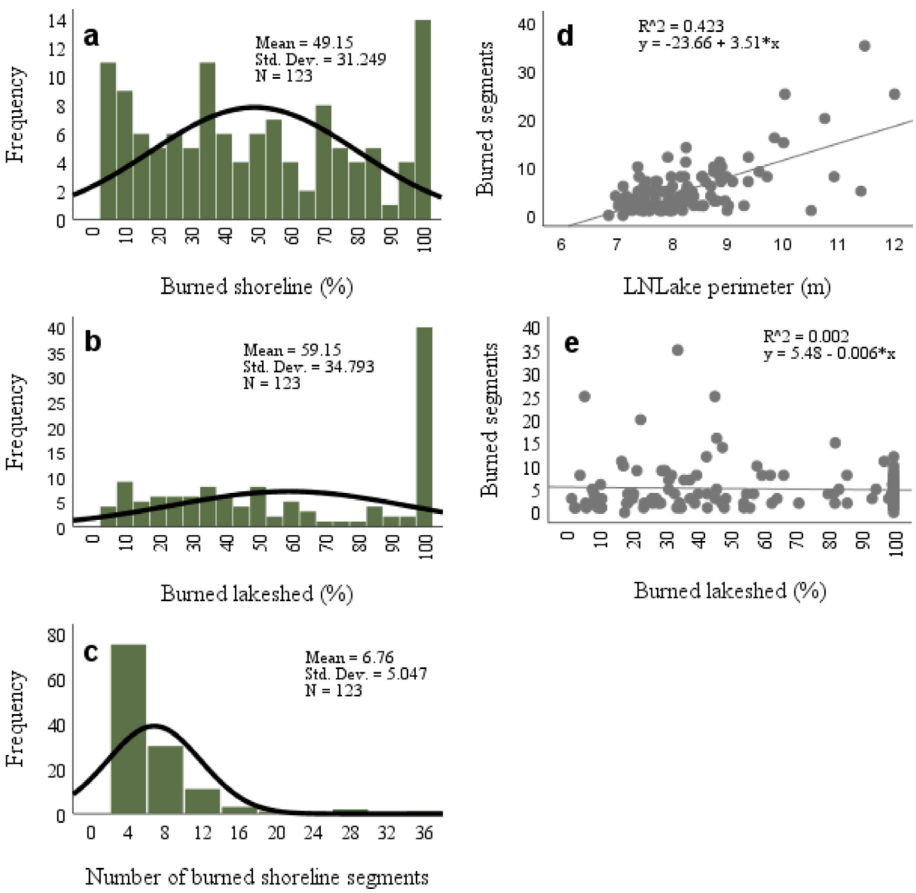

Fig. 2. Frequency distribution of burned shorelines, burned lakeshed areas and the number of burned shoreline segments among the fire affected study lakes (a, b & c). Numbers on X-axis represent the upper limits of distribution BINs. Figures d and e represent the relationship of the number of burned shoreline segments with lake perimeter and the percentage of burned lakeshed areas, respectively.

was referred to as a shoreline segment. Residual forest is defined as an assemblage of trees partially (at least 30% alive) or completely unaffected by fire at the time aerial photograph was taken. Isolated residual trees were not included in the study.

We studied a total of 26 wildfires which occurred between 2005 and 2007 and ranged in size from 58 ha to 13623 ha, originating mostly from lightning. To summarize the broad range of fire sizes which were irregularly distributed we classified the fires into three size classes which balanced the number of fires in each class: small (< 2000 ha), medium (2000 to 8000 ha) and large (> 8000 ha). We examined the shorelines and lakeshed areas of 123 fire-affected lakes ranging from 5.12 ha to 3273.59 ha. As with fires, lake and lakeshed size covered a wide range and was irregularly distributed. We divided lakes into four size classes along a logarithmic scale which balanced lake number in each class: small (up to10 ha), medium (> 10 to 100 ha), large (> 100 to 1000 ha) and very large (> 1000 ha). The lakeshed areas ranged from 0.1566 km^2 to 104.8203 km^2 and were divided into four size classes based on the shape of the frequency distribution: i) class 1 (up to 3.3 km^2), ii) class 2 (> 3.3 to 6.6 km^2), iii) class 3 (> 6.6 to10 km^2), and iv) class 4 (> 10 km^2). Finally, the percentage of burned lakeshed area in relation to total area of the lakeshed was divided evenly into five burn classes (BC); i) BC1 (0–20%), ii) BC2 (20.1–40%), iii) BC3 (40.1–60%), iv) BC4 (60.1–80%) and v) BC5 (> 80%). To evaluate how well actual burned areas on eFRI imagery compared with mapped fire areas on provincial landcover maps we digitized all the residual forest islands in a subset of 31 lakesheds and determined the actual burned area by subtracting the residual forest area from the total surface area of the respective lakeshed. We then compared the actual burned areas and mapped fire areas of these lakesheds and found no significant difference between the two (Kruskal-Wallis, $p = 0.281$). Therefore, in the subsequent analyses we used mapped fire area within a lakeshed as the burned lakeshed area of the respective lakeshed. In the analyses we used percentage values rather than absolute values to standardize disturbance measurements across a large range of lake, lakeshed and disturbance sizes, to ensure the distribution of disturbance measurements met the assumptions of the statistical tests and account for the influence of spatial scale on the patterns observed.

2.3. Statistical analyses

We determined the relationship between burned shoreline/lakeshed areas and fire, lake and lakeshed sizes using regression analyses. We also ran nonparametric Kruskal-Wallis tests to assess the effects of fire, lake and lakeshed sizes on burn area of shorelines and lakesheds. Fire, lake and lakeshed sizes were used as independent variables and percent burned shorelines and lakeshed areas as the response variables.

3. Results

3.1. Burned shorelines and lakesheds

The percentage of burned shorelines ranged from 0 to 100 among the fire affected lakes. Shorelines of two lakes were not burned at all whereas shorelines of 8 lakes were completely burned. The number of burned shoreline segments varied from 2 to 35, with minimum and maximum lengths of 9.98 and 5318.65 m, respectively. The percentage of burned lakeshed area ranged from 1.54 to 100. A significant number of lakesheds were completely burned. The distribution of burned shorelines, burned lakesheds and the number of burned shoreline segments is illustrated in Fig. 2(a, b & c, respectively). We found that the number of burned shoreline segments is significantly correlated ($p < 0.001$) with the length of the lake perimeter (Fig. 2d). However, we did not observe any relationship ($p = 0.639$) between the percentage of burned lakeshed area and the number of burned shoreline

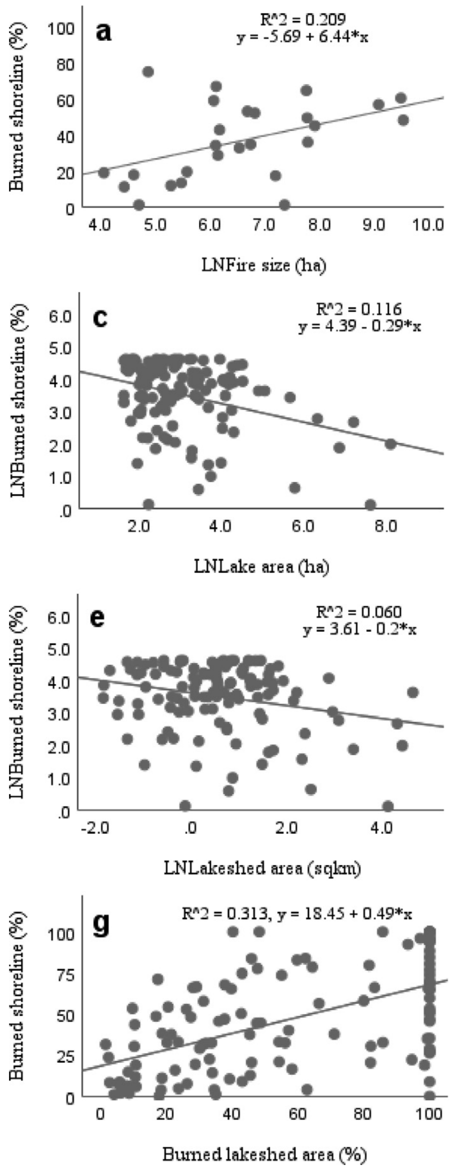

Fig. 3. Relationship between fire, lake and watershed sizes and burned shorelines and lakeshed areas. In the regression analysis the percentages of burned shorelines and burned lakeshed areas were used as the response variable and the fire, lake and lakeshed sizes as the predictor variables. Last panel (g) shows the relationship between burned lakeshed area and burned shoreline.

segments (Fig. 2e).

3.2. Fire size

There was a positive correlation between fire size and the percentage of burned shoreline ($r^2 = 0.209$, $p = 0.019$; Fig. 3a) as well as a positive relationship between fire size and the percent of a lakeshed area burned ($r^2 = 0.538$, $p < 0.001$; Fig. 3b). The average percentage of shoreline burned in small, medium and large fires was 33.68, 54.74 and 56.11%, respectively (Fig. 4a) with a difference that was not statistically significant (Kruskal-Wallis, $p = 0.078$). However, we observed significant differences in the percentages of burned lakeshed areas among small, medium and large fires ($p = 0.002$). In small fires 33.66% of the total lakeshed areas were burned while 63.78 and 80.43% of the total lakeshed areas were burned in medium and large fires, respectively (Fig. 4b).

3.3. Lake area

The percentage of shoreline burned was negatively correlated with the area of the lake ($r^2 = 0.116$, $p < 0.001$; Fig. 3c). There was also a significant negative relationship between the percentage of lakeshed area burned and the area of the lake ($r^2 = 0.085$, $p = 0.001$; Fig. 3d). We found a significant difference in burned shoreline among the lake size classes (Kruskal Wallis, $p = 0.021$) with small lakes having an average of 50.66% of the total shoreline burned while only 7.56% of the total shorelines were burned along very large lakes (Fig. 4c). Burned lakeshed area was also significantly different among lake size classes (Kruskal Wallis; $p = 0.016$) with 66.49% lakeshed areas of small lakes being burned on average compared to 14.53% for very large lakes (Fig. 4d).

3.4. Lakeshed area

The percentage of shoreline burned was significantly lower in larger lakesheds ($r^2 = 0.060$, $p = 0.007$; Fig. 3e). Similarly, larger lakesheds

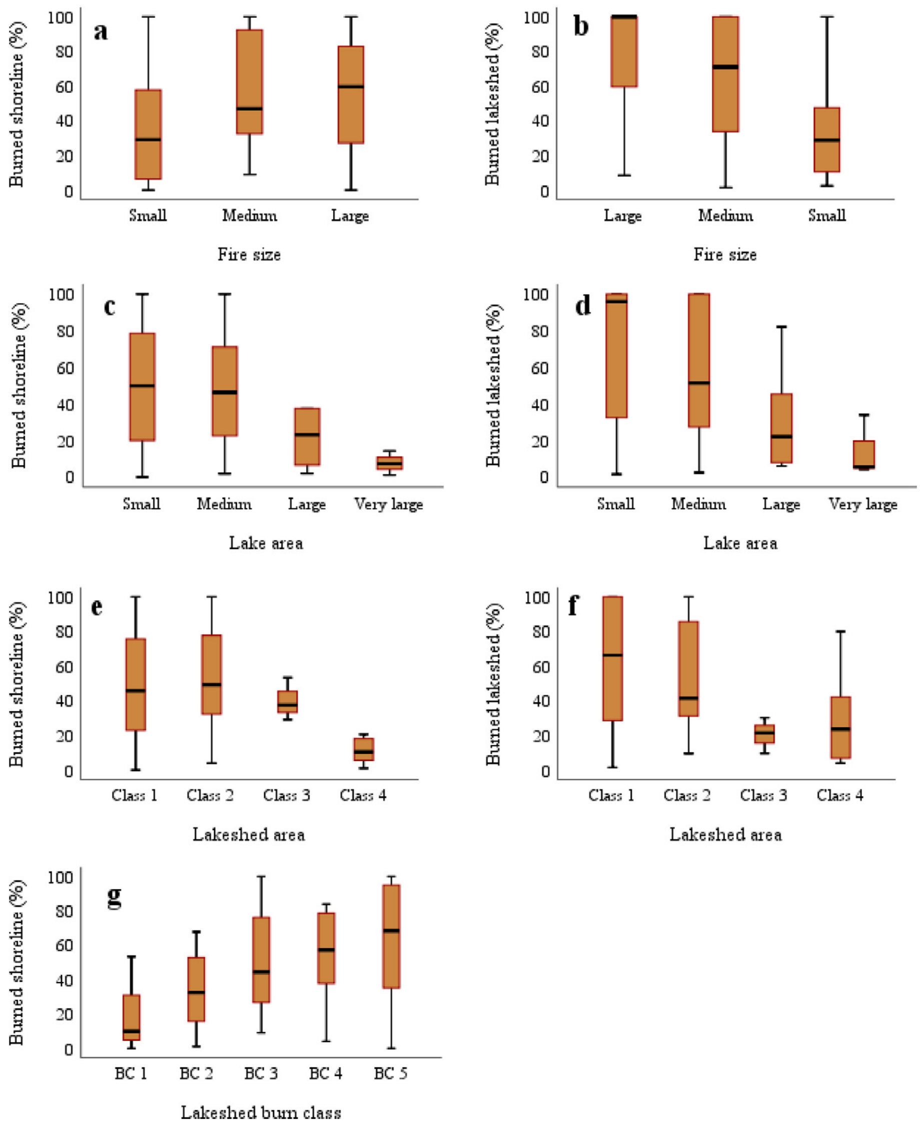

Fig. 4. Effects of fire, lake and lakeshed sizes on the burning of shoreline and lakeshed areas as well as the effects of lakeshed burning on the burning of shoreline. Y-axis represent percentages of burned shorelines and lakesheds and X-axis represent fire, lake and lakeshed sizes/classes. X-axis of the lowest panel represents the lakeshed burn classes.

had a significantly lower percentage of the lakeshed area burned than smaller lakesheds (r^2 = 0.061, p = 0.006; Fig. 3f). There were significant differences in the percentage of shoreline burned among the lakeshed size classes (Kruskal-Wallis, p = 0.006) with large (class 4) lakesheds having an average of 16.22% of their shoreline burned while smaller lakesheds (class 1 and 2) had 49.63 and 51.85%, respectively, of their shorelines burned (Fig. 4e). The percentage of lakeshed area burned also differed significantly among the lakeshed size classes (Kruskal-Wallis; p = 0.006). The lowest average lakeshed area burned was 20.44% in lakeshed class 3 and the highest was 63.32% in lakeshed class 1 (Fig. 4f).

3.5. Relationship between burned lakeshed area and shoreline burning

There was a significant positive relationship between the area of a lakeshed burned and the percentage of shoreline burned (r^2 = 0.313, p = < 0.001) (Fig. 3g). The percentage of burned shoreline differed significantly among the lakeshed burn classes (Kruskal Wallis, p < 0.001); the average shoreline burned in lakeshed burn class 1, 2, 3, 4 and 5 were 18.81, 34.64, 50.28, 53.14 and 65.23%, respectively (Fig. 4g).

3.6. Bi-factor interactions

We conducted some exploratory analyses to evaluate the interactions among the independent factors. The interactive effects of fire and lake sizes showed a clear trend of decreasing shoreline burning with increasing lake sizes in case of small fires. Since lakes of large and very large sizes were missing in the medium sized fires, no trend in shoreline burned could be identified for medium fires. However, in case of large fires the percentage of shoreline burned was highest for the medium sized lakes (60.50%) and lowest for the very large lakes (14.32%). The percentage of shoreline burned was largest (60.50%) for medium sized lakes within large fires and least (4.19%) for very large lakes within small fires (Fig. 5a). The burning of lakesheds showed a similar trend; burned lakeshed area was lower for lakes with larger areas for both small and large fires. The maximum burned area was observed in the lakesheds of small lakes having large fires (86.30%) and the minimum was in the lakesheds of very large lakes having small fires (4.81%) (Fig. 5b).

Fire and lakeshed sizes showed no clear interactive effects on either the percentage of shoreline or the area of lakeshed burned. Lakeshed class 2 that experienced large fires had the maximum percentage

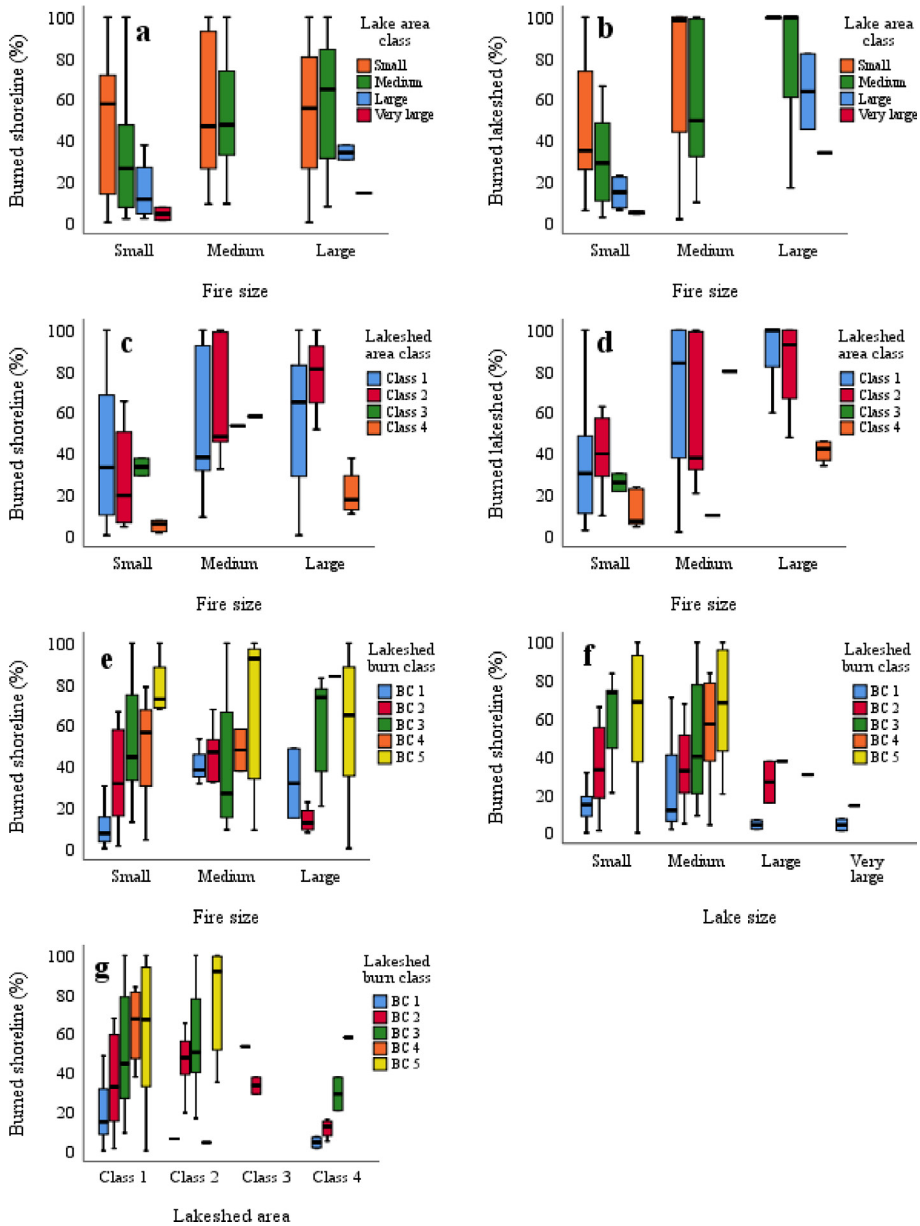

Fig. 5. Bi-factor interaction effects of fire, lake and lakeshed sizes on the burning pattern of shoreline and lakeshed areas (upper 4 panels) and the interactive effects of burned lakeshed and fire, lake and lakeshed sizes on shoreline burning (lower 3 panels).

(78.45%) of burned shorelines while lakeshed class 4 with small fires had the minimum (6.23%) (Fig. 5c). Within small fires, burned lakeshed areas were similar for lakeshed class 1 and 2 and were lower with increasing lakeshed areas. However, in lakeshed classes 1 and 2 the percentage of burned lakeshed area was higher with increasing fire sizes (Fig. 5d).

The lack of data limited the interpretation of the interactive effects of lake and lakeshed sizes on the burning pattern of shorelines and lakeshed areas. We observed a positive relationship between the percentages of burned lakeshed areas and burned shorelines irrespective of fire, lake and lakeshed sizes; the percentages of burned shorelines generally increased with increased burned lakesheds (Fig. 5e, f and g).

4. Discussion

Post-fire residual forest patterns are governed by complex interactions of biogeographic factors such as pre-fire vegetation, fuel load and distribution, topography, hydrology and wind speed and direction (Ryan, 2002; Cuesta et al., 2009; Madoui et al., 2009). Fuel moisture and duff moisture affect the burning pattern and formation of residuals (Perera et al., 2009). Since approaching fires modify the fuel and duff moisture gradually by the preheating effect of the fire fronts, burning pattern and post-fire residuals may be affected by fire size. Although larger fires have a greater probability of encountering fuel breaks or topographic barriers and are expected to have more residuals, we found that burning of shoreline and lakeshed areas increased with increasing fire size (Fig. 3a and b). This might be due to the longer burning period of larger fires which allows them to modify the fuel moisture through preheating and change of wind direction that might blow the fire back to the residuals that were previously left unburned.

Post-fire residuals are commonly associated with moist areas like wetlands and open water bodies and are concentrated within 100 m of wetlands (Arseneault, 2001; Araya et al., 2016). We found a higher percentage of burned shorelines and lakeshed area around small lakes (< 10 ha) compared to larger lakes (> 100 ha). There is likely a positive correlation between lake area and lakeshed area but the negative effect is slightly stronger for lake size, suggesting a 'lake' effect. Less burning around larger lakes can be explained by their enhanced ability to reduce fire intensity by modifying microclimate (Perera and Buse, 2014). Nielsen et al. (2016) also suggested that the size of water bodies has a significant influence on wildfires and large lakes have stronger controls over wildfires than smaller lakes. Larger lakes hold greater potential to decrease surrounding air temperature and increase relative humidity, which might slow the drying of adjacent areas and contribute to greater post-fire residuals.

Our analysis revealed a significantly higher percentage of shoreline and lakeshed areas burned in smaller lakesheds than in larger lakesheds. Fire effects and formation of post-fire residual patches are usually influenced by topography, pre-fire vegetation characteristics, wind variation and hydrologic conditions such as wetlands and low-lying areas (Arseneault, 2001; Ryan, 2002; Madoui et al., 2009). Since large lakesheds are likely have more variation in topographic and hydrologic conditions compared to small lakesheds, there would be more shoreline areas in larger lakesheds that are less likely to burn compared to smaller lakesheds. Larger lakesheds may also have more topographic barriers to the spread of fires as well as having more fire breaks (areas of bare soil or rock, within the lakeshed), which in turn may result in more residual forests after fires and consequently less burned watershed area.

Our results show that unlike fixed-width shoreline buffers, forest fires do not leave compact residual mature forests around lakes and in some cases, shorelines are burned right to the water edge, consistent with the conclusions of Buttle (2002). Although the primary objective of retaining a fixed-width buffer is to reduce solute and sediment fluxes from clearcuts to receiving lakes, several studies concluded that harvesting can be done close to the margin of lakes without any adverse impacts on lake water quality or lake ecosystems (Steedman, 2000; Steedman and Kushneriuk, 2000). Our results also indicate that END patterns may require harvesting to the lake margins in some areas, which can contribute to attaining natural vegetation patterns and additional timber harvesting.

5. Forest management implications

Our results show that the extent to which boreal lake watersheds are disturbed by wildfires varies widely over space and time. Similarly, the amount of shoreline forests consumed by fire, and the amount and distribution of residual forests within these areas, is highly variable. Many factors influence how forests burn; in our study larger and possibly more intense fires resulted in higher levels of watershed and shoreline disturbance. Our results indicate that in order to best emulate natural watershed disturbance patterns, managers should plan to maintain a range of harvest levels within watersheds across the landscape rather than considering a single disturbance threshold. Forest harvesting to the shoreline, along with planned residual areas, also represents a more natural disturbance pattern than the use of fixed-width buffers. However, Erdozain et al. (2020) recently showed that riparian zones contain areas of strong hydrological connectivity between land and water, and that these areas are sensitive to forest management. These 'biogeochemical control points' must be protected from forest management disturbances by harvesting equipment, road building etc. through careful planning and operations, including retaining residual forest patches in these areas. Therefore, harvesting in shoreline areas requires careful planning and operations to avoid harm to aquatic systems and protect the shoreline biogeochemical control points. The next step is to develop GIS-based models, which include hydrologic and topographic features, to predict shoreline areas that are more likely to burn or remain as residual forest in order to inform END based shoreline forest management.

CRediT authorship contribution statement

Conceptualization, Methodology and Research supervision: Rob Mackereth and Azim Mallik, Data curation and analysis: Shah Newaz and Darren McCormick, Writing- Original draft preparation: Shah Newaz, Reviewing and Editing: Azim Mallik, Rob Mackereth and Shah Newaz.

Declaration of Competing Interest

The authors declare no conflict of interest.

Acknowledgments

This research was supported by an NSERC (Natural Science nd Engineering Research Councile of Canada) Engage grant (#EGP-536369-18) awarded to A.U.M. and a Knowledge Transfer and Tool Development (KTTD) grant (#3A-2018) from Forestry Future Trust Finds of Ontario awarded to R.W.M and A.U.M.

References

Arseneault, D., 2001. Impact of fire behavior on post-fire forest development in a homogeneous boreal landscape. Can. J. For. Res. 31, 1367–1374.
Araya, Y.H., Remmel, T.K., Perera, A.H., 2016. What governs the presence of residual vegetation in boreal wildfires? J. Geogr. Syst. 18, 159–181.
Baldwin, D.J.B., Desloges, J.R., Band, L.E., 2011. Physical Geography of Ontario. In; Perera, A.H., Euler, D.L. Thompson, I.D. (Eds.) Ecology of a managed terrestrial landscape: patterns and processes of forest landscapes in Ontario. UBC Press. 346 pp.
Bergeron, Y., Gauthier, S., Kafka, V., Lefort, P., Lesieur, D., 2001. Natural fire frequency for the eastern Canadian boreal forest: consequences for sustainable forestry. Can. J. For. Res. 31, 384–391.
Buttle, J.M., 2002. Rethinking the donut: the case for hydrologically relevant buffer zones. Hydrol. Process. 16, 3093–3096.
Chambers, M.E., Fornwalt, P.J., Malone, S.L., Battaglia, M.A., 2016. Patterns of conifer regeneration following high severity wildfire in ponderosa pine-dominated forests of the Colorado Front Range. For. Ecol. Manag. 378, 57–67.
Clary, W.P., Medlin, D.E., 1993. Vegetation, nesting bird, and small mammal characteristics – Wet Creek, Idaho. USDA General Tech. Rep., INT-293.
Crins, W.J., Gray, P.A., Uhlig, P.W.C., Wester, M.C., 2009. The ecosystems of Ontario, part I: Ecozones and ecoregions. Ontario Ministry of Natural Resources, Peterborough, Ontario, Inventory, Monitoring and Assessment, SIB TER IMA TR- 01, 71pp.
Crow, T.R., Perera, A.H., 2004. Emulating natural landscape disturbance in forest management - An introduction. Landsc. Ecol. 19, 231–233.
Cuesta, R.M., Garcia, M., Retana, J., 2009. Factors influencing the formation of unburned forest islands within the perimeter of a large forest fire. For. Ecol. Manag. 258, 71–80.
Dosskey, M.G., Bentrup, G., Schoeneberger, M., 2012. A role for agroforestry in forest restoration in the lower Mississippi alluvial valley. J. For. 110, 48–55.
Ehnes, J., 1998. The influence of site conditions, age and disturbance by wildfire or winter logging on the species composition of naturally regenerating boreal plant communities and some implications for community resilience. Ph.D. thesis. University of Manitoba, Winnipeg, Manitoba.
Erdozain, E., Emilson, C.E., Kreutzweiser, D.P., Kidd, K.A., Mykytczuk, N., Sibley, P.K., 2020. Forest management influences the effects of streamside wet areas on stream ecosystems. Appl. Ecol. https://doi.org/10.1002/eap.2080.
Gundersen, P., Laurén, A., Finér, L., Ring, E., Koivusalo, H., Sætersdal, M., Weslien, J.O., Sigurdsson, B.D., Högbom, I., Laine, J., Hansen, K., 2010. Environmental services provided from riparian forests in the Nordic countries. Ambio 39, 555–566.
Harper, K.A., Lesieur, D., Bergeron, Y., Drapeau, P., 2004. Forest structure and composition at young fire and cut edges in black spruce boreal forest. Can. J. For. Res. 34, 289–302.
Kemball, K.J., 2002. Post disturbance dynamics of aspen mixed-woods: A comparison among fire, logging, and budworm outbreak. M. Sc. thesis, Lakehead University, Ontario.
Kreutzweiser, D.P., Capell, S.S., Good, K.P., Holmes, S.B., 2009. Sediment deposition in streams adjacent to upland clearcuts and partially harvested riparian buffers in boreal forest catchments. For. Ecol. Manag. 258, 1578–1585.
Kreutzweiser, D.P., Sibley, P.K., Richardson, J.S., Gordon, A.M., 2012. Introduction and a theoretical basis for using disturbance by forest management activities to sustain aquatic ecosystems. Freshw. Sci. 31, 224–231.
Kreutzweiser, D.P., Beall, F., Webster, K., Thompson, D., Creed, I., 2013. Impacts and prognosis of natural resources development on aquatic biodiversity in Canada's boreal zone. Environ. Rev. 21, 227–259.
Kuglerová, L., Ågren, A., Jansson, R., Laudon, H., 2014. Towards optimizing riparian buffer zones: Ecological and biogeochemical implications for forest management. For. Ecol. Manag. 334, 74–84.
Kuglerová, L., Jansson, R., Sponseller, R.A., Laudon, H., Malm-Renofalt, B., 2015. Local and regional processes determine plant species richness in a river-network

metacommunity. Ecology 96, 381–391.
Kuglerová, L., Hasselquist, E.M., Richardson, J.S., Sponseller, R.A., Kreutzweiser, D.P., Laudon, H., 2017. Management perspectives on *Aqua incognita*: Connectivity and cumulative effects of small natural and artificial streams in boreal forests. Hydrol. Process. 31, 4238–4244.
Lamb, E.G., Mallik, A.U., 2003. Plant species traits across a riparian zone/forest ecotone. J. veg. Sci. 14, 853–858.
Laudon, H., Kuglerová, L., Sponseller, R.A., Futter, M., Nordin, A., Bishop, K., Lundmark, T., Egnell, G., Ågren, A.M., 2016. The role of biogeochemical hotspots, landscape heterogeneity, and hydrological connectivity for minimizing forestry effects on water quality. Ambio 45, 152–162.
Leach, J.A., Lidberg, W., Kuglerová, L., Peralta-Tapia, A., Ågren, A., Laudon, H., 2017. Evaluating topography-based predictions of shallow lateral groundwater discharge zones for a boreal lake-stream system. Water Resour. Res. 53, 5420–5437.
Madoui, A., Leduc, A., Gauthier, S., Bergeron, Y., 2009. Spatial pattern analyses of post-fire residual stands in the black spruce boreal forest of western Quebec. Int. J. Wildland Fire 19, 1110–1126.
McRae, D.J., Duchesne, L.C., Freedman, B., Lynham, T.J., Woodley, S., 2001. Comparisons between wildfire and forest harvesting and their implications in forest management. Environ. Rev. 9, 223–260.
MNRF, 2016. Ontario Integrated Hydrology Data. Metadata record, Mapping and Information Resources Branch, Corporate Management and Information Division.
Moussaoui, L., Fenton, N.J., Leduc, A., Bergeron, Y., 2016. Can retention harvest maintain natural structural complexity? A comparison of post-harvest and post-fire residual patches in boreal forest. Forests 7, 243.
Naiman, R.J., Décamps, H., 1997. The ecology of interfaces: Riparian zones. Annu. Rev. Ecol. Syst. 28, 621–658.
Naylor, B.J., Mackereth, R.W., Kreutzweiser, D.P., Sibley, P.K., 2012. Merging END concepts with protection of fish habitat and water quality in new direction for riparian forests in Ontario: A case study of science guiding policy and practice. Freshw. Sci. 31, 248–257.
Nielsen, S.E., DeLancey, E.R., Reinhardt, K., Parisien, M.A., 2016. Effects of lakes on wildfire activity in the boreal forests of Saskatchewan, Canada. Forests 7 (265), 1–18.
NRC (National Research Council), 2002. Riparian areas: functions and strategies for management. National Academic Press, Washington, D.C., USA.
OMNR. 2010. Forest management guide for conserving biodiversity at the stand and site scales. Toronto, Queen's Printer for Ontario, pp. 211.
OMNR. 2014. Forest management guide for boreal landscapes. Toronto, Queen's Printer for Ontario, pp. 204.
Perera, A.H., Buse, L.J., 2014. Ecology of Wildfire Residuals in Boreal Forests. Wiley Blackwell, Hoboken, pp. 18–80.
Perera, A.H., Remmel, T.K., Buse, L.J., Ouellete, M.R., 2009. An assessment of residual patches in boreal fires, in relation to Ontario's policy directions for emulating natural forest disturbance. Ontario Ministry of Natural Resources, Ontario Forest Research Institute, Sault Ste. Marie, Ontario. Forest research report no. 169.
Qiu, Z., Hall, C., Hale, K., 2009. Evaluation of cost-effectiveness of conservation buffer placement strategies in a river basin. J. Soil Water Conserv. 64, 293–302.
Richardson, J.S., Naiman, R.J., Swanson, F.J., Hibbs, D.E., 2005. Riparian communities associated with Pacific Northwest headwater streams: Assemblages, processes, and uniqueness. J. Am. Water Resour. As. 41, 935–947.
Richardson, J.S., Naiman, R.J., Bisson, P.A., 2012. How did fixed-width buffers become standard practice for protecting freshwaters and their riparian areas from forest harvest practices? Freshw. Sci. 31, 232–238.
Ryan, K.C., 2002. Dynamic interactions between forest structure and fire behavior in boreal ecosystems. Silva Fenn. 36, 13–39.
Shenoy, A., Johnstone, J.F., Kasischke, E.S., Kielland, K., 2011. Persistent effects of fire severity on early successional forests in interior Alaska. For. Ecol. Manag. 261, 381–390.
Simon, N.P.P., Schwab, F.E., 2005. The response of conifer and broadleaved trees and shrubs to wildfire and clearcut logging in the boreal forests of central Labrador. North. J. Appl. For. 22, 35–41.
Sokal, M.A., Hall, R.I., Wolfe, B.B., 2010. The role of flooding on inter-annual and seasonal variability of lake water chemistry, phytoplankton diatom communities and macrophyte biomass in the Slave River Delta (Northwest Territories, Canada). Ecohydrol. 3, 41–54.
Steedman, R.J., 2000. Effects of experimental clearcut logging on water quality in three small boreal forest lake trout (*Salvelinus namaycush*) lakes. Can. J. Fish. Aquat. Sci. 57 (S2), 92–96.
Steedman, R.J., Kushneriuk, R.S., 2000. Effects of experimental clearcut logging on thermal stratification, dissolved oxygen, and lake trout (*Salvelinus namaycush*) habitat volume in three small boreal forest lakes. Can. J. Fish. Aquat. Sci. 57 (S2), 82–91.
Sweeney, B.W., Newbold, J.D., 2014. Streamside forest buffer width needed to protect stream water quality, habitat, and organisms: A literature review. J. Am. Water Resour. As. 50, 560–584.
Thompson, I.D., 2000. Forest Vegetation of Ontario: factors influencing landscape change. In: Perera, A.H., Euler, D.L., Thompson, I.D. (Eds.), Ecology of a Managed Terrestrial Landscape: Patterns and Processes of Forest Landscapes in Ontario. UBC Press, Vancouver, British Columbia, pp. 336.
Thorell, M., Götmark, F., 2005. Reinforcement capacity of potential buffer zones: Forest structure and conservation values around forest reserves in southern Sweden. For. Ecol. Manag. 212, 333–345.
Tiwari, T., Lundström, J., Kuglerová, L., Laudon, H., Öhman, K., Ågren, A.M., 2016. Cost of riparian buffer zones: A comparison of hydrologically adapted site-specific riparian buffers with traditional fixed widths. Water Resour. Res. 52, 1–20.
Trenholm, R., Lantz, V., Martinez-Espineira, R., Little, S., 2013. Cost-benefit analysis of riparian protection in an eastern Canadian watershed. J. Environ. Manag. 116, 81–94.
Weyenberg, S.A., Frelich, L.E., Reich, P.B., 2004. Logging versus fire: How does disturbance type influence the abundance of *Pinus strobus* regeneration? Silva Fenn. 38, 179–194.
Whitman, E., Parisien, M.-A., Thompson, D.K., Flannigan, M.D., 2018. Topoedaphic and forest controls on post-fire vegetation assemblies are modified by fire history and burn severity in the northwestern Canadian boreal forest. Forests 9, 151.