Assessing Morphological Change in Canadian Boreal Forests

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Abstract
Boreal forest cover change occurs in Canada primarily due to fire, a process that is predicted to experience a regime modification due to a changing climate. While fire frequency and area burned are relatively easily measured and tracked, we seek to understand whether the morphological structure of fires has also been changing through time or whether differences are detectable among Canadian Provinces and Territories due to jurisdictional or geographic differences. We use jurisdictions as proxies for differing forest management policies and geographic position. This study compares morphological segmentation patterns of annual boreal forest cover change from 2001 to 2014 across the entire Canadian boreal biome. We implement a bootstrapping of join-count results that were computed for each morphological element type and use the means and variability within ANOVA and Levene's tests for assessing statistically significant differences among our groups (years and jurisdictions). Overall, the morphology of forest disturbance patterns within the Canadian boreal biome was not found to be trending in any specific way, though there were isolated differences detected. We highlight those specific combinations that are particularly interesting within the context of the research questions posed. Our approach is conservative, as to not produce an alarmist response; since we focus on means, and disturbances are likely to emphasize extremes, thus only substantial regime modifications will produce statistically significant results. Interestingly, even with projected increases to fire intensity and area burned, the morphological structure of fire remains relatively stable.

1. Introduction

Over 30% of Earth's terrestrial area is covered by forests (4 billion ha), where forests are defined as areas larger than 0.05 to 1.00 ha with 10 to 30% coverage by plants taller than 5 m at maturity (Neeff et al., 2006 and FAO, 2010). In the northern hemisphere, boreal forests form the dominant forested biome, creating a circumpolar swath that is perturbed by natural and human activities to produce spatially and temporally varying landscapes (Taylor et al., 2013). These forests are naturally disturbed by wildfires, pests, diseases, and wind or ice; however, wildfires pose the greatest natural vector of disturbance in this biome. Human driven forest harvesting represents the greatest non-natural disturbance in this biome, since the vast stocks of relatively uniform forest stands, make the delivery of specific species mixtures to mills straightforward. Hence, the boreal is subject to cyclic perturbations that naturally and anthropogenically alter the forest mosaic. When we consider the reality of a changing climatic regime in the northern hemisphere (Lucash et al., 2017 and Brecka et al., 2018), additional influences on both local weather and broader fire regimes are poised to influence the boreal biome even further. Understanding these changes requires an examination and comparison of the spatial patterns that they imprint on the landscape, whether for conservation and species diversity studies (Balmford et al., 2003) or to link ecological processes with observable spatial patterns (Turner et al., 1999, 2005, 2010 and Fu et al., 2011), or to assess whether changes are being influenced by a changing climate (Brecka et al., 2018) and whether changes are jurisdictionally pronounced due to differing management practices and guidelines (de Groot et al., 2013).

Disturbances in boreal forests produce remarkable and rapid changes to local landscape patterns and functions (Forman and Godron, 1986), and can alter the structure of an ecosystem, and even effect succession, species compositions, or the variability of these attributes (Taylor et al., 2013). With time however, these systems adapt or recover and stabilize to form a mosaic of forest stands at various ages and seral stages. The quantification and characterization of disturbance regimes has focused on return intervals, disturbance intensity, undisturbed residual content, extent, size class distributions, and aspects of spatial pattern (Morgan et al., 2001, Pickett and White, 2005, Cui and Perera, 2008 and Hanes et al., 2019). While spatial patterns have been quantified by hundreds of landscape metrics (Baker and Cai, 1992, Riitters et al., 1995, Haines-Young and Chopping, 1996, Uuemaa et al., 2009 and Kupfer, 2012), and specifically for Canadian forest fires (e.g., Parisien et al., 2006), these metrics pose inherent problems and difficulties for comparison (Remmel and Csillag, 2003 and Remmel and Fortin, 2013), particularly across scales (Kedron et al., 2018), but also to tease out impacts of composition and configuration (Remmel, 2009 and Riitters, 2018).

Given that fire disturbance pressure varies annually due to many interconnected variables related to temperature, precipitation, fuel availability and its state (Bonan, 1989 and Podur and Martell, 2009), we aimed to devise a method that would not identify differences simply because there were more or fewer fires, but to test whether the morphological structure of disturbances is changing through time. We hypothesize that trending changes in disturbance morphology could result from a changing climate or fire regime.

We also test whether such differences exist among political jurisdictions, potentially implying policy and management differences. Therefore, we focus not on landscape metrics *per se*, but on measuring, intuitive, tangible and mappable fire disturbance morphologies (Vogt et al., 2007 and Soille and Vogt, 2009) of boreal forests in this paper. Specifically, we assess whether disturbance morphologies differ through annual time-steps or vary jurisdictionally among Canadian provinces and territories over a 14-year period (2001-2014). Morphological segmentation focuses on the form, shape, and connectivity of image classes; it is a theoretical framework for analyzing the spatial structure of a landscape.

Fires during the study period affected an average of 23,238 km^2 annually but ranged between 6,264 km^2 and 45,633 km^2, (Canadian Council of Forest Ministers, 2019) values that have nearly doubled in the past two decades (Burton et al., 2008). The effects of fire are varied, from assisting plant regeneration (Stocks et al., 2002) to increasing local plant diversity (Ruokolainen and Salo, 2006), and influencing vegetation succession and composition or influencing carbon sequestration and release in the context of climate change (Flannigan et al., 2000). Regardless of the increasing trend in discrete fires and area burned (Canadian Council of Forest Ministers, 2019), we want to address whether the spatial structure of these fires has been changing (i.e., to understand whether the structural mechanism of fire is changing).

This study quantifies the spatial morphology of Canadian boreal forest fire disturbances using a standardized morphological approach and statistically tests for significant differences across space and through time to answer: (1) does the morphology of boreal fire disturbances in Canada differ through time? and (2) does the morphology of boreal fire disturbances in Canada differ among provinces and/or territories?

2. Study Area

The study area includes the boreal biome in Canada (Figure 1), and is dominated by typical boreal forest species (Strahler and Archibold, 2011), wetlands, and lakes. This biome provides a wide variety of ecosystem services, habitats, food sources, natural resources, recreational space, and boasts extensive cultural values (Brandt et al., 2013) and is widely seen as a carbon sink (Pohjola et al., 2003). The boreal biome is anything but static and aside from being perturbed by fire, harvesting, insects, and disease, the landscape mosaic is the result of these and other biophysical processes that continually sculpt this region at multiple spatial and temporal scales (Bonan, 1989 and Bonan and Shugart, 1989).

3. Methods

The methods are summarized by the flowchart provided in Figure 2. We downloaded 40 of the total 504 tiles of forest change data (>600,000 Landsat scenes) from the Global Forest Change repository (earthenginepartners.appspot.com), covering an area larger in extent than Canada. Each tile (10° × 10°) represents consistently processed results of time-series analyses to provide annual maps of forest change at approximately 30 m spatial resolution (at the equator) in 8-bit format (Hansen et al., 2013). The attribution in each cell is either a 0 (no forest loss) or an integer indicating the year of forest loss. We retained only the data pertaining to the year of forest loss (2001 through 2014) and then clipped those tiles to the boundaries of the boreal forest biome polygons that are defined in the openly available data provided by The Nature Conservancy (maps.tnc.org) to eliminate forest changes occurring beyond the boreal.

We further clipped this result to each of the Canadian administrative political jurisdictions (10 Provinces and 3 Territories) as obtained from the Global Administrative Areas repository (www.gdam.org), and retained 9 of them that have boreal forests within them. These 14 years × 9 jurisdictions resulted in 126 disturbance maps.

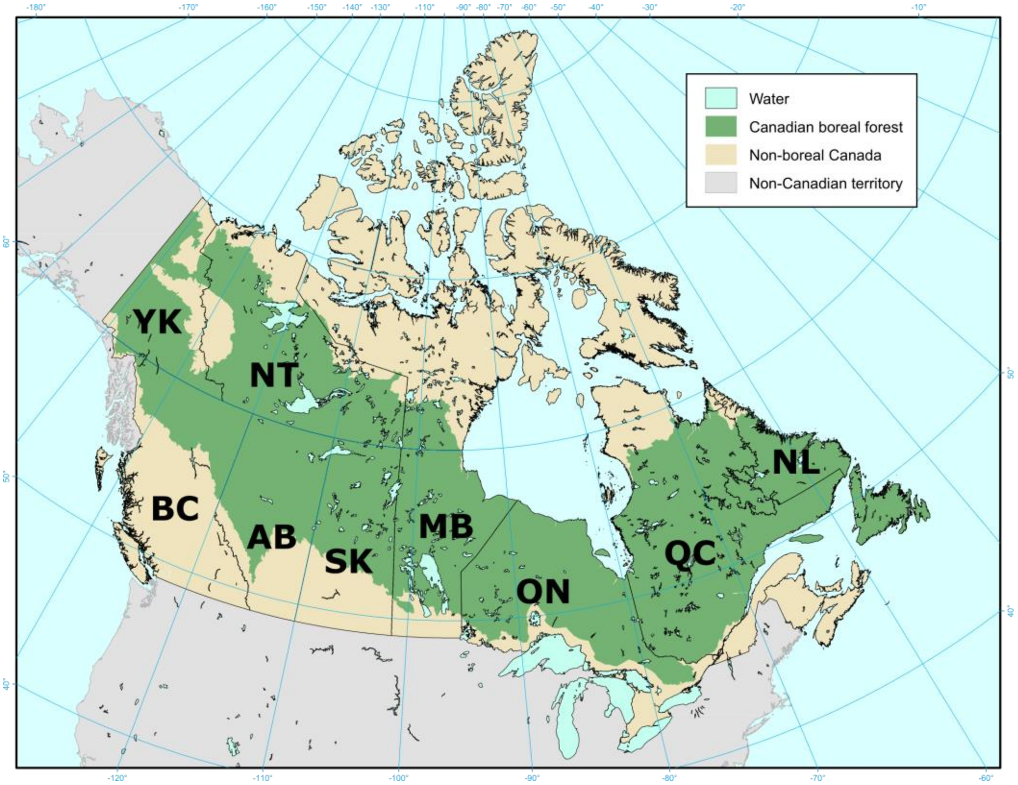

Figure 1: Map of the boreal biome in Canada with Provincial and Territorial boundaries. Jurisdictional codes are provided for those that experience substantial fire disturbances and are examined in this study. YK = Yukon, NT = Northwest Territories, BC = British Columbia, AB = Alberta, SK = Saskatchewan, MB = Manitoba, ON = Ontario, QC = Quebec, and NL = Newfoundland and Labrador. Data sources: The Nature Conservancy and the Global Administrative Areas Repository

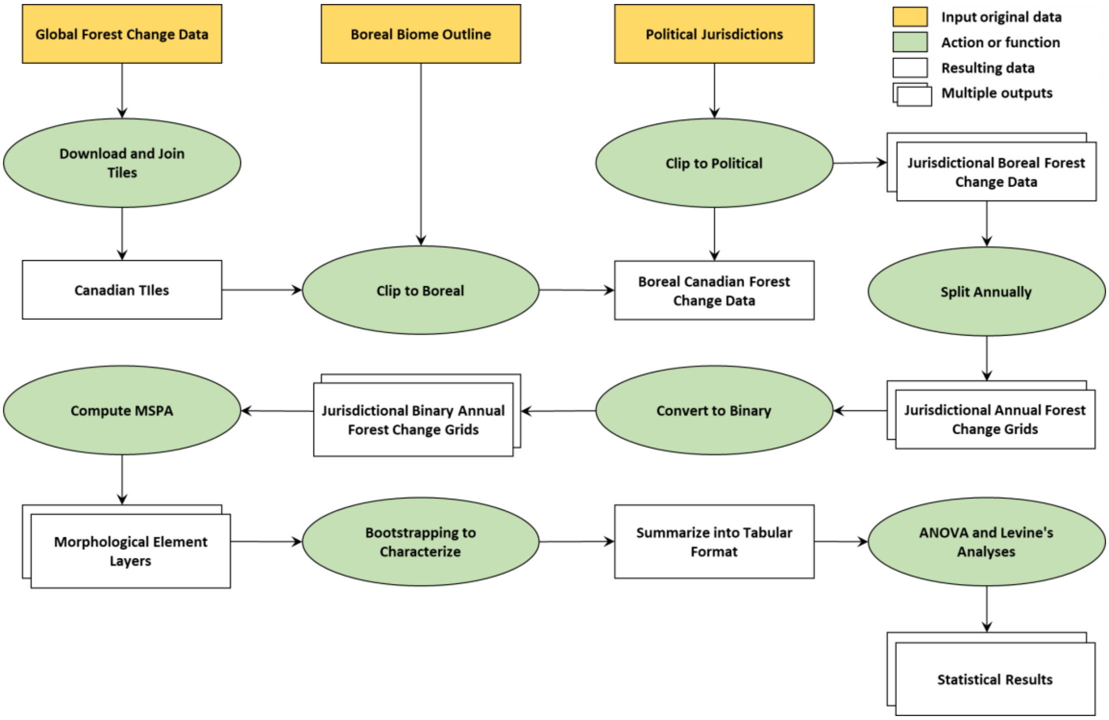

Figure 2: Flow chart of methods implemented

The individual files were converted to binary (disturbed/non-disturbed) and processed by the Morphological Spatial Pattern Analysis (MSPA) software tool to conduct morphological segmentations (Riitters et al., 2009 and Vogt and Riitters, 2017) of individual provinces and territories in each year and produce both tabular and spatial results. The MSPA results assign a mutually exclusive morphological class (i.e., background, core, islet, loop, bridge, perforation, edge, or branch – see Figure 3) to each pixel (Soille and Vogt, 2009). We parameterized MSPA to use 8-neighbour (Queen's case) *connectivity* (Sawada, 1999) and a default minimum of a 1-pixel *edge-width* for defining edges and perforations. We turned the *intext* parameter off to avoid explicitly encoding nested structures (e.g., cores contained within perforations) and thereby avoided the added complexity that these relatively rare cases present and that would make comparisons among our study's factors nearly impossible due to low replicate numbers. Each output map is further split to produce binary maps, one for each of the 7 non-background morphological elements (126 disturbance maps × 7 morphological elements = 882 binary MSPA maps). Each binary MSPA map was subjected to a join-count analysis (Upton and Fingleton, 1985) using the *spdep* package (Bivand and Wong, 2018) in R (R Core Team, 2018) to characterize both the observed and expected frequencies of like-morphological elements being spatial neighbours (Figure 4). Since a global join-count measure would not provide insight to the variability of the morphology within the jurisdiction and year observed, we opted to draw random sub-samples from each output using a bootstrapping approach (Garcia et al., 2008). These bootstrapped samples permitted the computation of distribution parameters (i.e., mean and variance) for each morphological element within each jurisdiction and year combination. Bootstrapping was conducted on $n = 500$ randomly placed windows of $s = 200^2$ pixels within each image; the number (n) and size (s) of sub-samples were selected based on prior sensitivity and stability testing by the authors. Join-count statistics are computed for the rook's case, where adjacency is determined by like-morphological elements sharing a pixel edge (Sawada, 1999), not just a corner, and assuming a torus structure.

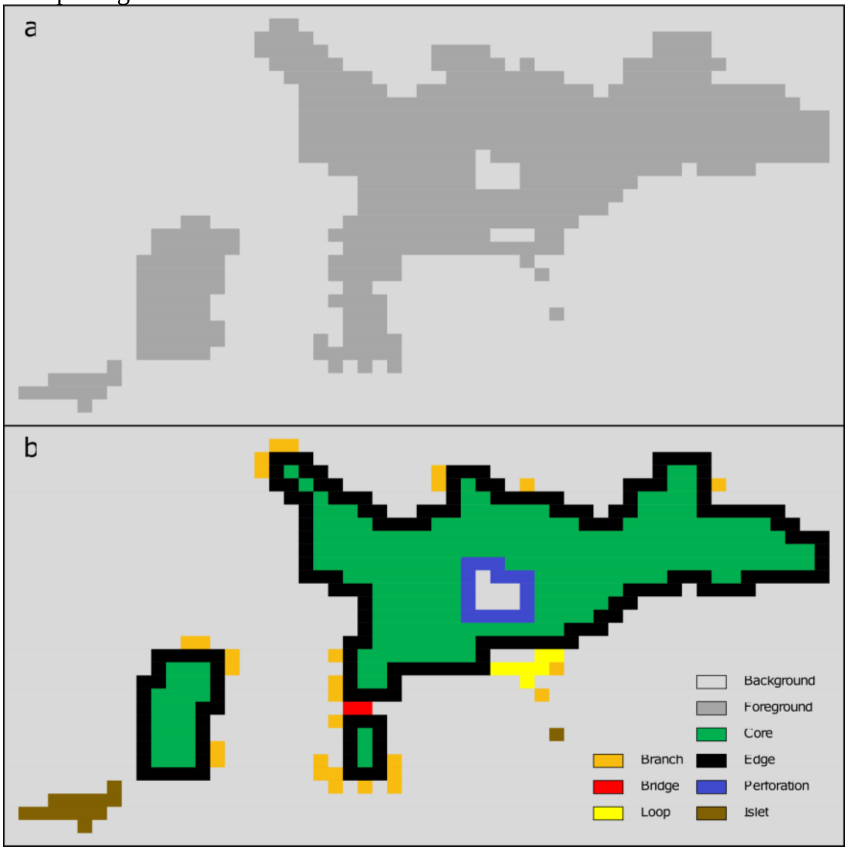

Figure 3: A subset of a binary forest disturbance map (a) that is processed by MSPA (b) showing that the foreground is not partitioned into the 7 morphological elements and a background (non-disturbed) land cover class

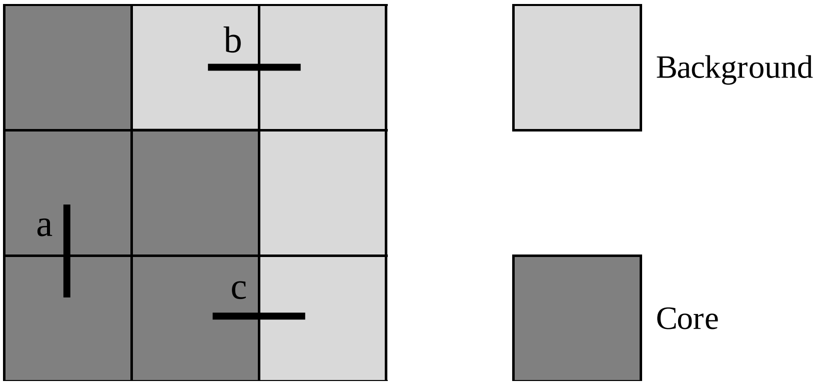

Figure 4: Examples of the three types of joins possible in join-count analysis of a binary map. Assuming that we have a focal morphological element type (i.e., Core) and the Background joins can be (a) Core–Core, (b) Background–Background, or (c) Core–Background

A torus structure is where geographic space is considered continuous due to an implied wrapping of the extents back to the opposing sides of the map in an effort to minimize edge effects. Join-count outcomes, in addition to the observed and expected frequencies, are coupled with variances, and *z*-values for assessing the significance of morphological structures. Join-count analyses produce three sets of results for each of the morphological elements measured. For example, the Core morphological element would produce join-count results for Core–Core, Core–Background, and Background–Background spatial configuration combinations of neighbouring structures (Figure 5). Our analysis focuses only on the like-joins, in this case the Core–Core joins (the focal morphological element), because these indicate the predominance of a morphological element to make a contiguous imprint on the landscape. By focusing on the contiguity of a single morphological element, we consistently track a clear and definable state through time and space, rather than the numerous possible neighbouring structures possible among 7 morphological elements and the background class. The cells identified as Background are scrutinized when the other morphological elements are analyzed, as the Background becomes some other focal foreground morphological element class.

Two versions of ANOVA tests were conducted. The first tested whether the mean number of each like-morphological join type (e.g., Core–Core joins) differ among the years of study, the second tested whether the mean number of each like-morphological join type differ geographically among Canadian jurisdictions within each study year. These tests were paired with corresponding Levene's tests to determine whether the variability of the like-joins for each morphological class differed among the identical groupings.

We examined both the statistical results (the threshold of significance was set at $p < 0.05$) and visualized the data by plotting corresponding boxplots to reveal some interesting observations and conditions. For each significant test, Tukey HSD post hoc tests were performed to identify the group (or groups) that differed significantly with respect to the focal morphological element. Of the thousands of output plots and statistical results, in this paper we curate and present the most informative results.

4. Results

The morphological element Core is most analogous to the tangible notion of the fire footprint, but more precisely it excludes all edges and areas that do not have sufficient width as to contain core area. The test among years for the mean number of Core–Core joins was significant ($F_{13,15091} = 13.38$, $p < 0.05$) as was the corresponding Levene's test ($F_{13,15091} = 33.64$, $p < 0.05$). Post hoc tests identified significantly higher means and variances in 2004, 2005 and 2013. Further investigation (see Figure 6) showed that Core–Core joins were significantly elevated for Manitoba in 2013 ($F_{13,6902} = 117.6$, $p < 0.05$), and also more highly variable ($F_{13,6902} = 296.9$, $p < 0.05$). This result confirmed a data anomaly related to cloud contamination and thus bad data which was confirmed with a visual observation of the original data. For context, Figure 7 provides the frequency of Canadian forest fires and the total area burned in each year. During the study period, while there is variability, there is a decreasing trend in the total number of fires, but an increasing trend in the area burned. With changes to the Core, we would likewise expect Edge to be affected, since they contain the Core and form the interface between the fire disturbed areas from the undisturbed landscape matrix.

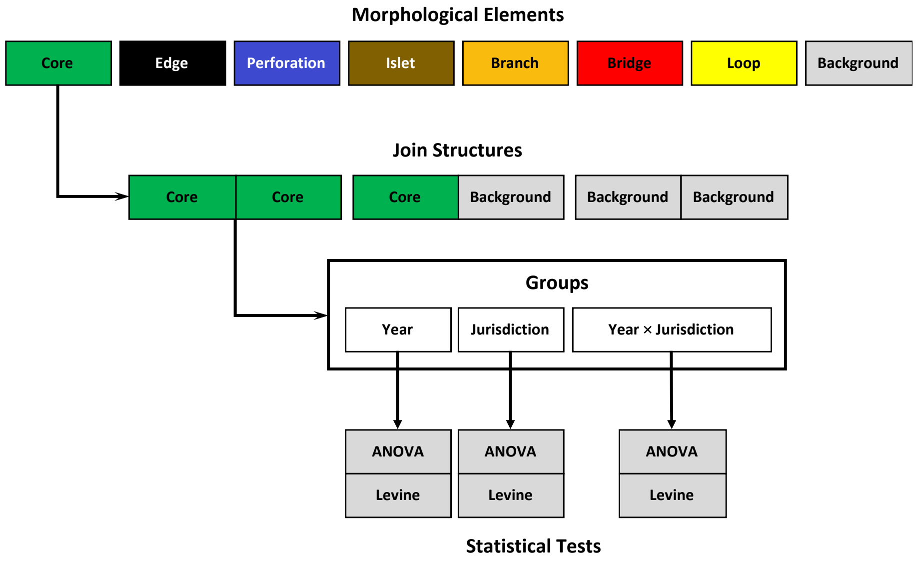

Figure 5: Analytical structure, demonstrated for Core, but replicated for all morphological elements except Background

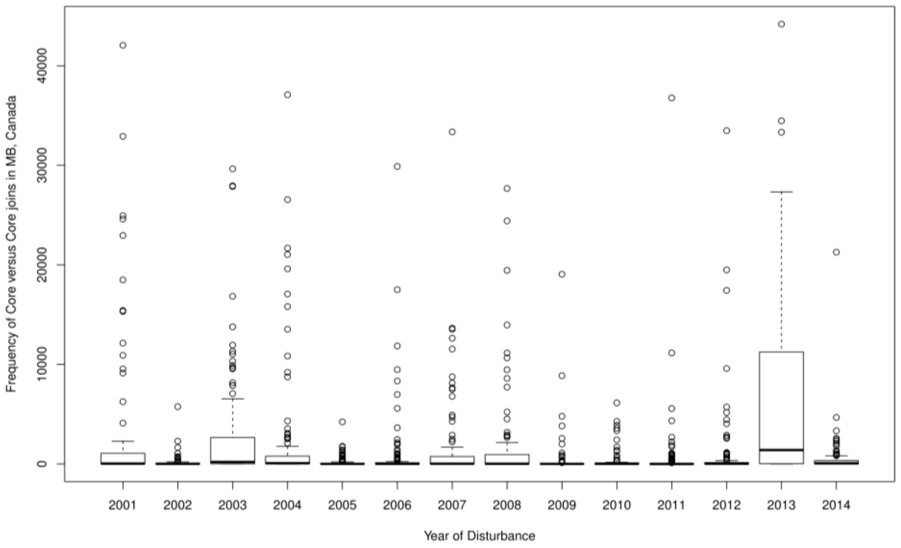

Figure 6: Core–Core joins in Manitoba between 2001 and 2014

Corresponding tests for Edge–Edge joins were also significant for means ($F_{13,30546} = 9.474$, $p < 0.05$) and variances ($F_{13,30546} = 26.08$, $p < 0.05$). Results from post hoc tests identify the same years as being significantly different, but also include 2009 and 2014 in some pairwise comparisons.

While Edge measures the outer margin of disturbances, perforations account for internal holes, indicating the presence of waterbodies or patches of residual vegetation. A similar test among mean Perforation–Perforation joins was significant ($F_{13,4688} = 2.596$, $p < 0.05$) as was the test among variances ($F_{13,4688} = 6.858$, $p < 0.05$). Post hoc analyses indicate that mean Perforation–Perforation joins for 2013 are significantly higher than 2003, 2009, 2011, 2012, and 2014, but that the consistent differences observed for Core and Edge morphological elements is not evident. Corresponding post hoc analyses for variances differed dramatically, indicating that the variability of the 2003 and 2013 are significantly higher than in other years. Thus, there is some variability in Core, Edge, and Perforation contiguity within the study period, but the differences are not consistent, nor do they provide a definitive and observable trend. When fires burn, they tend to produce spot fires ignited by sparks ejected from the main fire; these spot fires can remain small and produce morphological islets.

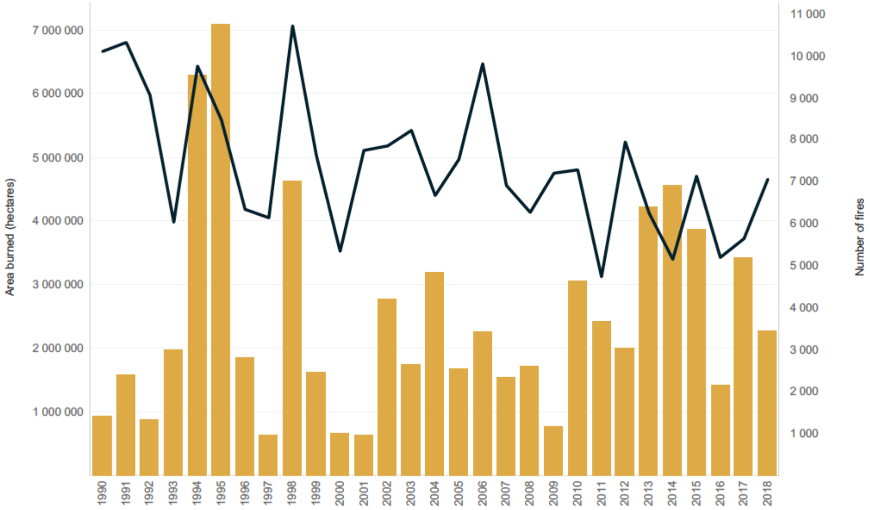

Figure 7: Fire frequency (line) and area burned (bar) in Canada (Modified from: Canadian Council of Forest Ministers, 2019). The red box identifies the period of our study

Our analysis identified 2002 as having statistically higher mean Islet–Islet joins than in any other year in the study, specifically in Ontario and Quebec ($F_{8,4426} = 15.64$, $p < 0.05$). We also identify that Islet–Islet joins among the Canadian provinces and territories differ significantly in 2005 ($F_{8,4360} = 18.96$, $p < 0.05$). Post hoc analyses indicate that Islet–Islet occurrences for Northwest Territories and Yukon tend to be significantly higher than in other provinces/territories in that year. The corresponding Levene's test was also significant ($F_{8,4360} = 50.33$, $p < 0.05$).

An ANOVA test among provinces and territories was significant ($F_{8,300} = 3.057$, $p < 0.05$) in 2011 for the mean number of Perforation–Perforation joins. Post hoc analyses identify Yukon as having a consistently higher frequency than any other province/territory. Our corresponding Levene's test was also significant and identically isolated Yukon's higher variability in Perforation–Perforation joins ($F_{8,300} = 8.164$, $p < 0.05$) relative to all other jurisdictions. Taking a closer look at Yukon, we tested for significant differences in mean Perforation–Perforation joins among years, and this was significant ($F_{8,609} = 1.745$, $p = 0.048$) as was the Levene's test for differences among variances ($F_{13,609} = 5.187$, $p < 0.05$). Post hoc tests support the outlier status of 2011.

A one-way ANOVA testing whether the average numbers of Core–Core joins differ among the years 2001 through 2014 in the Northwest Territories was significant ($F_{8,1457} = 3.984$, $p < 0.05$). Post hoc analyses indicate that Core–Core occurrences in the Northwest Territories for 2013 are significantly higher than most other years. The corresponding Levene's test was also significant ($F_{13,1457} = 10.62$, $p < 0.05$) with post hoc results identifying 2013 as being significantly higher in most years. We present test results for Bridge–Bridge and Loop–Loop joins compared among jurisdictions in 2014 (identified as an anomalous year). ANOVA ($F_{8,667} = 17.96$, $p < 0.05$) and Levine's ($F_{8,667} = 37.99$, $p < 0.05$) tests for Bridge–Bridge were both significant; post hoc results identify Saskatchewan and Northwest Territories as having more and increasingly variable Bridge–Bridge joins. Similarly, the ANOVA ($F_{8,694} = 15.84$, $p < 0.05$) and Levine's ($F_{8,694} = 25.97$, $p < 0.05$) tests for Loop–Loop variability were significant; post hoc results reflect highly similar results from the ANOVA mean's comparison.

5. Discussion

If a fire regime were to experience a statistically significant climate-induced change, then we expect that fire frequency would exhibit an increasing trend with larger areas being burned, but that these changes would also become increasingly variable. National reporting by the Canadian Council of Forest Ministers (2019) does show an uptick in area burned, but with a decreasing trend in fire frequencies (Figure 4). This may be accounted for by the coalescence of multiple smaller fires into fewer larger ones, but the actual mechanism is not yet fully understood or known. Simultaneously, the morphological structures of fires are expected to adjust to the realities of a new climate state, flux, or normal, to express the process of forest landscape burning. By segmenting disturbance maps into their morphological structural elements, we were able to explore the spatial structures of fire disturbances in terms of what elements comprise a fire event's footprint and to track how those change or differ thorough time and/or geographic space. In order to fully comprehend the results of the join-count analyses, this discussion briefly contextualizes how differing numbers of joins in a specific morphological class can influence the size, shape, and complexity of disturbances.

Islets represent small or sinuous disturbances that do not contain edge or core class types. Given the spatial resolution of this data and the minimum forest harvest size in Canada, these can represent either erroneous classifications, but more likely spot fires. As the number of Islet–Islet joins increase, the size of these linear and sinuous disturbance patches increases without gaining width. Generally, high numbers of islet joins represent elongated patches or spider-like configurations. Islets can amount to substantial areas of burn, that due to their compact structure and generally small extent, may not be mapped or tracked using conventional means, or affiliated with nearby larger fire footprints, and thus affect accounting for carbon budgets and landscape alteration tracking. We flag these as important elements in identifying potential fire behaviour and as a signal of potential wind behaviour during burning (Fernandez-Pello, 2017).

Perforations present an interesting landscape morphological class, since waterbodies and rock outcrops are the most common land cover types that cause their formation. More specifically, under the assumption that outcrops and waterbodies can generally be considered stable on a landscape, thus their likelihood of forming edges or perforations is consistent through time. Thus, dramatic changes to the presence of perforations on a landscape are likely the result of residual patches (Perera et al., 2004, Araya et al., 2015, 2016a, 2016b), those areas that escaped burning for any number of reasons within a fire footprint. As fire regimes change, the potential to impact these features is also possible. Thus, our measurements form good base-lines from which future deviations can be tested.

Core pixels are perhaps the most intuitive morphological class to think about. Three aspects of the disturbances can possibly be explained by looking at the number of Core–Core joins: size, compactness (or inversely the linearity), and complexity. As the size of the patches increase, so do the number of Core–Core joins; thus, an elevated number of Core–Core joins can be interpreted as expressing larger disturbed patches. The second characteristics of disturbances can be explained by the number of joins is their compactness/linearity. Two patches comprising the same number of pixels, will have a lower Core–Core count for an elongated or linear patch than a compact one. Similarly, a lower Core–Core count can be an indicator of higher complexity, since pixels will be less compact and able to extend in many directions. While increased core joins indicate larger patches, the shape of those patches is not reflected by this measure beyond knowing that there is substantive width to the additional area, otherwise core would not exist.

Bridge and Loop morphological classes act as connectors. As the number of their like-joins increase, either longer linear connections are present or there are simply more of them (since these connectors cannot have widths that would introduce core and edge classes). Thus, with an increased number of Bridge–Bridge joins, cores may be further apart but connected or begin to form hub-and-spoke type patterns where multiple cores are connected by corridors. Similarly, as Loop–Loop joins increase, either higher numbers of corridors that connect to a common core patch increase, or more likely, those loops get longer and somewhat wider (but without introducing Core and Edge classes). Together, these cases illustrate increasing complexity and could result from topographic constraints, unevenness in moisture distribution, or chaotic wind patterns.

Quantification and comparison of the forest cover change over vast geographic extents and long periods of time is a challenging task. We wished to study temporal morphological pattern changes and spatial differences in forest disturbance within the Canadian boreal biome from 2001 to 2014 and among provincial and territorial jurisdictions. The use of join-count statistics enabled characterization of the composition and configuration of the spatial patterns on binary maps, where disturbances were not mapped as objects but by independent morphological structural elements. Bootstrap resampling produced empirical distributions that facilitated the comparisons of the join-count analysis outcomes among the factor groups: (1) spatially groupings (i.e., Canadian provinces and territories) and (2) temporal groups (i.e., years 2001 through 2014). To statistically test the effect of spatial and temporal groupings, ANOVA and Levene’s tests were used to compare means and variances of join-count outcomes for each of the morphological classes respectively. Significant results were further investigated with post hoc tests.

While the number of comparisons made was immense, we present here some of the key conclusions of our search of statistically significant differences and summarize them appropriately. In the analysis of Core–Core and Edge–Edge joins, there were clear annual differences that separated 2004, 2005, and 2013 from the other years. Clearly the core and edge classes are related and when patch sized increase, both generally increase but not always at equal rates. With increased core and edge in these years, perforations only appeared to be significantly different in 2013; thus, there is some subtle difference between 2013 and 2004-2005. Here, the former year, 2013, was year with a high area burned in Canada, while the latter two years had lower area burned but had more fires. Detailed looks at Manitoba, Northwest Territories, and Yukon really highlight the differences in Core–Core joins in 2013 versus all other years; again, likely the effect of this being an active fire season. Interestingly, Yukon expressed significantly more perforations in 2011 which was actually a very inactive fire year, while 2004 did not register but experienced many fires with a high area burned.

Islets showed significant differences for Ontario and Quebec versus all other jurisdictions in 2002, while in 2005 the Northwest Territories and Yukon differed from all other jurisdictions. The 2002 situation could be explained by active an active fire season and the likelihood of spot fires, but the 2005 case is odd given the low fire activity. Connector joins (Bridge and Loop) seem to really separate nicely in 2014, putting Northwest Territories and Saskatchewan into a common group versus all other jurisdictions.

5.1 Future Considerations

The results of this lengthy and multi-faceted analysis are not consistent through time or geography, meaning that no obvious trends are observed. While this is counter to our expectation given the reality of a changing climate, the period may be too short to fully capture the effect of a gradually shifting climatic regime or the natural variability of boreal disturbances in this region. Furthermore, extreme weather is likely to more dramatically influence frequency and total burned area (or intensity) which does not necessarily mean a difference in morphological structure. We take this to mean that while forest disturbances may be becoming more prevalent and intense, the mean structural characteristics have remained relatively stable. However, we now have a baseline to which annual data can be added and continually tested and we can dig deeper into individual jurisdiction-year combinations to tease out possible causes for the patterns observed. Future work that explores a longer period and potentially examines temperature and precipitation covariates may yield insights to the varying patterns observed. Similarly, an extension of this study beyond Canada, to include the vastness of the entire boreal biome is something that we are considering.

6. Conclusions

There were two research questions that this study attempted to answer. The first question was whether the spatial and temporal morphology of forest disturbance pattern within the boreal biome of Canada differ through time. The answer to this question is yes; the spatial and temporal morphology of forest disturbance pattern within the boreal biome of does Canada differ through time and these differences manifest themselves in various provinces/territories and morphological classes, but not as a trend. The second question was whether the spatial morphologies of forest disturbance patterns in the boreal biome of Canada differ among provinces and/or territories. The answer to this question is also yes; the spatial morphologies of forest disturbance patterns in the boreal biome of Canada does differ among provinces and/or territories and these differences manifest themselves in various years and morphological classes, these also do not present a clear trend.

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References

Araya, Y. H., Remmel, T. K. and Perera, A. H., 2015, Residual Vegetation Patches within Natural Boreal Wildfires: Characterizing by Pattern Metrics, Land Cover Expectations and Proximity to Firebreak Features. *Geomatica*, Vol. 69(4): 327-338.

Araya, Y. H., Remmel, T. K. and Perera, A. H., 2016a, Spatially Explicit Prediction of Residual Vegetation Patch Occurrence within Boreal Wildfires. *International Journal of Geoinformatics*, 12(3): 1-15.

Araya, Y. H., Remmel, T. K. and Perera, A. H., 2016b, What Governs the Presence of Residual Vegetation in Boreal Wildfires?. *Journal of Geogrpahical Systems*, Vol. 18(2): 159-181.

Baker, W. L. and Cai, Y. M., 1992, The r.le-programs for Multiscale Analysis of Landscape Structure Using the GRASS Geographical Information-System. *Landscape Ecology*, Vol. 7(4): 291-302.

Balmford, A., Green, R. E. and Jenkins, M., 2003, Measuring the Changing State of Nature. *Trends in Ecology and Evolution*, Vol. 18(7): 326-330. DOI: 10.1016/S0169-5347(03)00067-3.

Bivand, R. S. and Wong, D. W. S., 2018, Comparing Implementations of Global and Local Indicators of Spatial Association. *TEST*, Vol. 27(3): 716-748. DOI: 10.1007/s11749-018-0599-x.

Bonan, G. B., 1989, Environmental Factors and Ecological Processes Controlling Vegetation Patterns in Boreal Forests. *Landscape Ecology*, Vol. 3(2): 111-130. DOI: 10.1007/BF00131174.

Bonan, G. B. and Shugart, H. H., 1989, Environmental Factors and Ecological Processes in Boreal Forests. Annual Review of Ecology and Systematics, Vol. 20, 1-28.

Brandt, J. P., Flannigan, M. D., Maynard DG, Thompson, I. D. and Volney, W. J. A., 2013, An Introduction to Canada's Boreal Zone: Ecosystem Processes, Health, Sustainability and Environmental Issues. *Environmental Reviews*, Vol. 21(4), 207-226. DOI: 10.1139/er-2013-0040.

Brecka, A. F. J., Shahi, C. and Chen, H. Y. H., 2018, Climate Change Impacts on Boreal Forest Timber Supply. *Forest Policy and Economics*, Vol. 92: 11-21. DOI: 10.1016/j.forpol.201-8.03.010.

Burton, P. J., Parisien, M. A., Hicke, J. A., Hall, J. A. and Freeburn, J. T., 2008, Large Fires as Agents of Ecological Diversity in the North American Boreal Forest. *International Journal of Wildland Fire*, Vol. 17(6), 754-767.

Canadian Council of Forest Ministers, 2019, National Forestry Database. Available at: http://nfdp.ccfm.org/en/index.php (accessed 19 September 2019).

Cui, W. and Perera, A. H., 2008, What Do We Know about Forest Fire Size Distribution and Why is this Knowledge Useful for Forest Management?. *International Journal of Wildland Fire*, Vol. 17(2), 234-244.

de Groot, W. J., Cantin, A. S., Flannigan, M. D., Soja, A. J., Gowman, L. M. and Newbery, A., 2013, A Comparison of Canadian and Russian Boreal Forest Fire Regimes. *Forest Ecology and Management*, Vol. 294, 23-34. DOI: 10.1016/j.foreco.2012.07.033.

FAO, 2010, Global Forest Resources Assessment 2010 - Main Report - FAO Forestry Paper 163. Available at: http://www.fao.org/3/i1757e/i1-757e00.htm (accessed 17 September 2019).

Fernandez-Pello, A. C., 2017, Wildland Fire Spot Ignition by Sparks and Firebrands. *Fire Safety Journal*, Vol. 91, 2-10. DOI: 10.1016/j.fir-esaf.2017.04.040.

Flannigan, M. D., Stocks, B. J. and Wotton, B. M., 2000, Climate Change and Forest Fires. *Science of the Total Environment*, Vol. 262(3): 221-229.

Forman, R. T. T. and Godron, M., 1986, *Landscape Ecology*. New York, NY: John Wiley & Sons.

Fu, B. J., Liang, D. and Lu, N., 2011, Landscape Ecology: Coupling of Pattern, Process and Scale. *Chinese Geographical Science*, Vol. 21(4), 385-391.

Garcia, T., Braun, J., Bryce, R. and Tymstra, C., 2008, Smoothing and Bootstrapping the Prometheus Fire Growth Model. *Environmetrics*, Vol. 19(8), 836-848.

Haines-Young, R. and Chopping, M., 1996, Quantifying Landscape Structure: A Review of Landscape Indices and their Application to Forested Landscapes. *Progress in Physical Geography*, Vol. 20(4): 418-445.

Hanes, C. C., Wang, X., Jain, P., Parisien, M. A., Little, J. M. and Flannigan, M. D., 2019, Fire-regime changes in Canada over the Last Half Century. *Canadian Journal of Forest Research*, Vol. 49(3), 256-269. DOI: 10.1139/cjfr-2018-0293.

Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., Thau, D., Stehman, S. V., Goetz, S. J., Loveland, T. R., Kommareddy, A., Egorov, A., Chini, L., Justice, C. O. and Townshend, J. R. T., 2013, High-Resolution Global Maps of 21st-Century Forest Cover Change. Science, Vol. 342(6160), 850-853. DOI: 10.1126/science.1244693.

Kedron, P. J., Frazier, A. E., Ovando-Montejo, G. A. and Wang, J., 2018, Surface Metrics for Landscape Ecology: A Comparison of Landscape Models Across Ecoregions and Scales. *Landscape Ecology*, Vol. 33(9), 1489-1504. DOI: 10.1007/s10980-018-0685-1.

Kupfer, J. A., 2012, Landscape Ecology and Biogeography: Rethinking Landscape Metrics in a Post-FRAGSTATS Landscape. *Progress in Physical Geography: Earth and Environment*, Vol. 36(3), 400-420. DOI: 10.1177/0309133-312439594.

Lucash, M. S., Scheller, R. M. J., Gustafson, E. and Sturtevant, B. R., 2017, Spatial Resilience of Forested Landscapes under Climate Change and Management. *Landscape Ecology*, Vol. 32(5), 953-969. DOI: 10.1007/s10980-017-0501-3.

Morgan, P., Hardy, C. C., Swetnam, T. W., Rollins, M. G. and Long, D. G., 2001, Mapping Fire Regimes across Time and Space: Understanding Coarse and Fine-Scale Fire Patterns. *International Journal of Wildland Fire*, Vol. 10(3-4), 329-342.

Neeff, T., von Luepke, H. and Schoene, D., 2006, Choosing a Forest Definition for the Clean Development Mechanism. *Forests and Climate Change Working Paper*, Vol. 4, 1-18, http://www.fao.org/forestry/11280-03f2112412-b94f8ca5f9797c7558e9bc.pdf.

Parisien, M. A., Peters, V. S., Wang, Y. H., Little, J. M., Bosch, E. M. and Stock, B. J., 2006, Spatial Patterns of Forest Fires in Canada, 1980 - 1999. *International Journal of Wildland Fire*, Vol. 15(3), 361-374. DOI: 10.1071/WF06009.

Perera, A. H., Buse, L. J. and Weber, M. G., 2004, *Emulating Natural Forest Landscape Disturbances: Concepts and Applications*, Edition, New York: Columbia University Press.

Pickett, S. T. A. and White, P. S., 2005, *The Ecology of Natural Disturbance and Patch Dynamics*, Edition, Nachdr. Orlando: Academic Press.

Podur, J. J. and Martell, D. L., 2009, The Influence of Weather and Fuel Type on the Fuel Composition of the Area Burned by Forest Fires in Ontario, 1996-2006. *Ecological Applications*, Vol. 19(5), 1246-1252.

Pohjola, J., Kerkela, L. and Makipaa, R., 2003, Credited Forest Carbon Sinks: How the Cost Reduction is allocated among Countries and Sectors. *Climate Policy*, Vol. 3(4): 445-461.

R Core Team, 2018, R: A Language and Environment for Statistical Computing. Vienna, Austria: R Foundation for Statistical Computing. Available at: https://www.R-project.org/.

Remmel, T. K., 2009, Investigating Global and Local Categorical Map Configuration Comparisons Based on Coincidence Matrices. *Geographical Analysis*, Vol. 41(1), 113-126.

Remmel, T. K. and Csillag, F., 2003, When are Two Landscape Pattern Indices Significantly Different? *Journal of Geographical Systems*, Vol. 5(4), 331-351.

Remmel, T. K. and Fortin, M. J., 2013, Categorical, Class-Focused Map Patterns: Characterization and Comparison. *Landscape Ecology*, Vol. 28(8), 1587–1599.

Riitters, K., 2018, Pattern Metrics for a Transdisciplinary Landscape Ecology. Landscape Ecology, Vol. 34, 2057-2063. DOI: 10.1007/s10980-018-0755-4.

Riitters, K., Vogt, P., Soille, P. and Estreguil, C. 2009, Landscape Patterns from Mathematical Morphology on Maps with Contagion. *Landscape Ecology*, Vol. 24: 699-709.

Riitters, K. H., O'Neill, R. V., Hunsaker, C. T., Wickham, J. D., Yankee, D. H., Timmins, S. P., Jones, K. B., and Jackson, B. L., 1995, A Factor Analysis of Landscape Pattern and Structure Metrics. *Landscape Ecology*, Vol. 10(1), 23-39.

Ruokolainen, L. and Salo, K. 2006, The Succession of Boreal Forest Vegetation During Ten Years after Slash-Burning in Koli National Park, *Annales Botanici Fennici*. Vol. 43(5), 363-378.

Sawada, M., 1999, ROOKCASE: An Excel 97/2000 Visual Basic (VB) add-in for Exploring Global and Local Spatial Autocorrelation. *Bulletin of the Ecological Society of America*, Vol. 80(4): 231-234.

Soille, P. and Vogt, P., 2009, Morphological Segmentation of Binary Patterns. *Pattern Recognition Letters*, Vol. 30(4), 456-459. DOI: 10.1016/j.patrec.2008.10.015.

Stocks, B. J., Mason, J. A., Todd, J. B., Bosch, E. M., Wotton, B. M., Amiro, B. D., Flannigan, M. D., Hirsch, K. G., Logan, K. A., Martell, D. L. and Skinner, W. R., 2002, Large Forest Fires in Canada, 1959-1997. *Journal of Geophysical Research*, Vol. 108(D1): 8149. DOI: 10.1029/2001JD000484.

Strahler, A. H. and Archibold, O. W., 2011, *Physical Geography: Science and Systems of the Human Environment*. 5 edition. Wiley.

Taylor, A. R., Hart, T. and Chen, H. Y. H., 2013, Tree Community Structural Development in Young Boreal Forests: A Comparison of Fire and Harvesting Disturbance. *Forest Ecology and Management*, Vol. 310, 19-26.

Turner, M. G., 2005, Landscape Ecology: What is the State of the Science?. *Annual Review of Ecology, Evolution, and Systematics*, Vol. 36, 319-344.

Turner, M. G., 2010, Disturbance and Landscape Dynamics in a Changing World. *Ecology*, Vol. 91(10), 2833-2849. DOI: 10.1890/10-0097.1.

Turner, M. G., Gardner, R. H. and O'Neill, R. V., 1999, Landscape Ecology in Theory and Practice: Pattern and Process. New York: Springer-Verlag.

Upton, G. J. G. and Fingleton, B., 1985, *Spatial Data Analysis by Example*. Toronto: John Wiley & Sons Inc.

Uuemaa, E., Antrop, M., Roosaare, J., Marja, R. and Mander, Ü., 2009, Landscape Metrics and Indices: An Overview of their Use in Landscape Research. Living Reviews in Landscape Research, Vol. 3(1), 1-28.

Vogt, P. and Riitters, K., 2017, GuidosToolbox: Universal Digital Image Object Analysis. European Journal of Remote Sensing, Vol. 50(1): 352-361. DOI: 10.1080/22797254.2-017.1330650.

Vogt, P., Riitters, K. H., Estreguil, C., Kozak, J. and Wade, T. G., 2007, Mapping Spatial Patterns with Morphological Image Processing. *Landscape Ecology*, Vol. 22(2), 171-177. DOI: 10.1007/s10980-006-9013-2.