Evaluation of Saskatchewan Designated Protected Lands as Ecological Benchmarks for Forest Management

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Overview

This report is organized as follows:

We describe the concept of ecological benchmarks and discuss applications to forest management in Saskatchewan.

We 1) describe the study area, and 2) broadly assess the designated lands of the RAN as ecological benchmarks based on spatial metrics related to ecological processes and intactness.

We 1) describe and provide rationale for the use of intrinsic patches and enduring features, 2) describe criteria for representation, 3) describe the analytical methods for assessing representation, and 4) evaluate the RAN based on representation criteria.

We provide a preliminary assessment of the adequacy of the RAN to act as benchmarks for the effects of forest management on woodland caribou.

We present a simple qualitative analysis of the forest age structures within the study region

BEACONs has developed a process-based approach for constructing system-level benchmarks *de nouveau* in the boreal region of Canada. Here, we describe the approach and provide examples of system-level benchmarks identified in Saskatchewan.

We describe next steps for expanding or enhancing analyses applied to date, including addressing hydrologic connectivity, and discuss approaches to benchmark construction.

Appendices A-G:

The report also includes several appendices that elaborate on the methods applied.

Appendix A: Building Regional Planning Units for the Delineation of Ecological Benchmarks (pages 80-83)

We describe how Regional Planning Units were constructed.

Appendix B: Evaluating Stratification Units (pages 84-97)

We assess the ability of intrinsic patch (IP) structures to discriminate between ecozones, regional planning units, and ecoregions.

Appendix C: Association of Intrinsic Patch and Enduring Feature Type (pages 98-101)

We explore the relationship between enduring feature attributes and terrestrial intrinsic patch composition.

Appendix D: Appendix D: Criteria for Selecting Landscapes (page 102)

We describe the criteria used to select landscapes for IP compositional analyses.

Appendix E: Development of Intrinsic Patch from Vegetation Inventories (pages 103-113)

We describe how intrinsic patches were constructed, including ArcView code.

Appendix F: Proportions of Intrinsic Patches Entirely Captured Within Enduring Feature Polygons (pages 114-116)

We explore some of the implications of the decision to not split intrinsic patches (IPs) across enduring feature (EF) polygons.

Appendix G: Simulations of Expected Forest Age Structure Under Alternative Fire Regimes and Sources of Measurement Error (pages 117-127)

We explore by simulation the kinds of time-since-fire distributions that would be expected under various assumptions about the natural and managed fire regimes. We also consider how various forms of measurement error would affect the age structure reported in forest inventory data.

Summary

Saskatchewan Environment desired an assessment of the adequacy of designated lands in Saskatchewan as ecological benchmarks for forestry practices evaluation. The BEACONs Research Group (Boreal Ecosystems Analysis for Conservation Networks) is undertaking an analysis of ecological benchmarks for boreal and taiga regions of Canada. A series of discussions indicated that there was excellent potential for collaboration on this project, the terms of which are specified in a Memorandum of Understanding between the parties concerned. The BEACONs Project is an independent, university-based research project, established in partnership with the Canadian Boreal Initiative, and working with a broad base of industry, First Nations, ENGO and government partners. The mandate of BEACONs is to develop a credible scientific framework for comprehensive conservation planning that includes both protected areas and lands managed for other values.

The BEACONs mandate is national in scope, but regional case studies have been identified as an important tool for refining concepts and incorporating geographic variation in biophysical attributes and management objectives. BEACONs has proposed a Conservation-Matrix Model (CMM) which represents a pro-active approach to land-use planning that combines the strengths of systematic planning for reserves with the systematic process of adaptive management of appropriate resource development. The CMM has four principle components: 1) ecological benchmarks, 2) additional reserve areas, 3) adaptive resource management areas, and 4) the conservation matrix, in which all components are embedded. Unlike the traditional framework for conservation, the Conservation-Matrix Model recognizes the critical contribution of the matrix to attain the goals of conservation and sustainable land use. The vision for implementation of the CMM is the establishment of a national conservation network comprised of ecological benchmarks (the anchors), additional reserves, and management areas that are embedded in a conservation matrix governed by the principles of adaptive resource management.

The Saskatchewan evaluation of ecological benchmarks for forest management activities complements the national mandate of BEACONs, affording BEACONs the opportunity to apply concepts for the identification of ecological benchmarks in conjunction with the objectives of the MOU between BEACONs and the Saskatchewan Government (see Box 1). This report addresses MOU objectives 1 and 2.

Box 1: Saskatchewan MOU Objectives

1) Definition of the role of and criteria for ecological benchmarks, with particular reference to forest management activities.
2) Evaluation of the adequacy of existing designated lands in Saskatchewan as ecological benchmarks for forest management activities and identification of gaps.
3) Analysis of potential ecological benchmarks in Saskatchewan and adjacent areas.
4) Assessment of options for expansion of designated lands in Saskatchewan to serve as ecological benchmarks, as appropriate.

In the context of ecologically sustainable forest management (i.e., maintenance of functioning ecosystems while permitting the appropriate development of forest resources), benchmarks serve as reference sites or controls for understanding both the natural dynamics of ecosystems, as well as their response to resource development activities.

As controls, ecological benchmarks should have the following characteristics:

- Composition and condition representative of the range of natural conditions of the region in which they are located.
- Sufficiently large to capture the full spatial extent of the biological and physical processes being monitored and shield the processes from the influence of activities outside of the benchmark.
- Absence of human activities (past, present, and future) that could affect the natural process(es) being monitored.

Consistent with these principles, in the context of Saskatchewan's Representative Areas Network, Saskatchewan Environment has expressed the view that:

> *One of the roles of the Representative Areas Network (RAN) is to serve as a "control" for forest management activities. As ecological benchmarks, representative areas provide the opportunity to monitor and compare the outcomes of forest management to the natural system and provide the basis for adaptive management. Ecological benchmarks must capture the range of forest ecosystems they are intended to represent and, to the degree feasible, be of sufficient spatial extent in relation to boreal forest disturbance events.*

The designated lands of the RAN consist of many classes of areas with varying levels of protection: National and Provincial Parks, Recreation Areas, Wilderness Areas, Natural Areas, Ecological Reserves, Game Preserves, Park Reserves, Provincial Forests, Provincial Historic Sites, Crown land, Representative Areas, Ducks Unlimited management areas, Wildlife Management Areas, and Wildlife Refuges. Generically, we refer to all designated lands in the RAN as **Representative Areas (RAs)**. The designated lands that fall within the study area (defined below) were included in all analyses, and collectively, are referred to as the **RAN** area. The **NONRAN** area refers to the forested landbase outside the RAN.

The study area for this project was broadly delineated by the commercial forest zone of Saskatchewan and adjacent areas of Manitoba and Alberta, intersected with the boreal plains and boreal shield ecozones. Within ecozones, additional ecological considerations were included in the boundary delineation by selecting water catchments that intersected the commercial forest zone. Finally, we coarsely stratified components of our assessment within the study area by Regional Planning Units (RPUs), defined by the intersection of major ocean drainage basins and ecozones. This stratification system is being applied by BEACONs across all boreal and taiga regions in Canada.

Ecological benchmarks can be considered across a range of spatial and temporal extents. At the highest level, system-level benchmarks (SLBs) serve as controls for monitoring a breadth of large-scale processes and are the ideal benchmarks for forest management. SLBs are representative of environmental variation, maintain viable populations of native species and intact ecological and evolutionary processes, including disturbance regimes, hydrological processes, nutrient cycles, and biotic interactions. SLBs are of sufficient size to experience the largest, anticipated natural disturbance (*e.g.,* fire) while maintaining internal recolonisation sources for species whose habitat is rendered unsuitable by such disturbances (e.g., Pickett and Thompson 1978[1]). Where possible, BEACONs is identifying candidate SLBs within each RPU in boreal Canada. However, recognizing the limitations that existing land-use patterns place on such sites, we have developed a conceptual framework that considers a sliding scale of benchmarks relative to ecological processes.

Based on a coarse assessment of areal extent, none of the existing representative areas (RAs) in Saskatchewan meets the size requirements for system-level benchmarks; however, the utility of smaller areas to act as ecological benchmarks for appropriately-scaled processes should not be discounted. Provincial and national parks and other designated lands, which are typically small relative to SLBs, present practical options for such benchmarks, due to historical restriction of human use and existing legal protection. Further, an evaluation of the intactness of the RAN based on the Global Forest Watch Canada (GFWC) Intact Forest Landscape indicates that there are opportunities to expand some of the more northern RAs to include adjacent, intact areas. Physical intactness is relevant to ecological benchmarks as a measure of the absence of many direct human influences and thus serves as a proxy for the intactness of biological and physical processes.

A more detailed assessment of the forest composition of RAs was conducted to evaluate the adequacy of the existing designated lands in Saskatchewan's RAN as ecological benchmarks for forest management activities based on representation criteria. We structured our evaluation as a 3-level hierarchical representation problem wherein patch and landscape scale attributes in and outside of the RAN were measured and compared within strata. We defined 3 hierarchical scales: intrinsic patch (IP), landscape, and regional planning units (strata) (Figure 1). *Intrinsic patches* (IPs) are the finest scale. We used a small number of broad forest or land cover classes derived from inventory data to delineate spatial units which are expected to remain invariant under natural disturbances, stand dynamics (e.g. natural regeneration and succession) and forest management. There are 5 intrinsic patch types: MIX, UPCON, BOG, RIPARIAN, and WATER (Table 1). IPs provide two fundamental types of measurements for assessing aspects of representation that are relevant to forest management: patch size and patch composition. *Landscapes* were defined by enduring feature boundaries and represent sampling units comprised of collections of IPs. *Regional Planning Units* (RPUs), described previously, are the highest level of stratification. There were 4 RPUs intersecting the study area (Figure 1). The representation analyses were restricted to the 2 largest RPUs (RPU 10 and 12) which roughly correspond to the boreal plain and boreal shield ecozones.

At the patch scale, we found that the RAN adequately represents IP MIX, and under-represents IP BOG, in both RPU 10 and 12. At the landscape scale, we found that the IP composition of UPCON, MIX, and BOG are different between RAN and NONRAN. In RPU 10, the RAN adequately represents landscapes jointly high in MIX and UPCON, but

[1] Pickett, S.T.A. and J.N. Thompson. 1978. Patch dynamics and the design of nature reserves. Biological Conservation 13:27-37.

poorly represents landscapes with high abundances of MIX (>70%) and landscapes jointly >33% BOG and UPCON. In RPU 12, the RAN landscapes are almost entirely > 33% UPCON. Representation of landscapes *jointly* > 33% BOG and UPCON are lacking in the RAN. However, landscapes that are *jointly* > 33% in MIX and UPCON are well represented. The significance of this is discussed relative to the habitat requirements of woodland caribou.

Figure 1: The three hierarchical scales used by BEACONs to evaluate the representativeness of Saskatchewan's RAN.

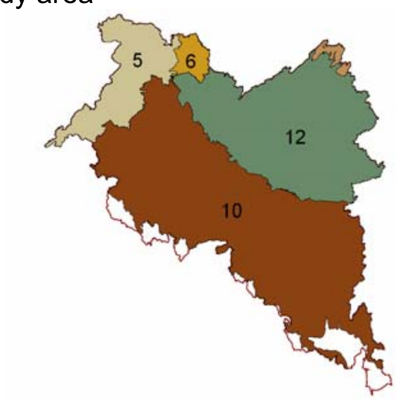
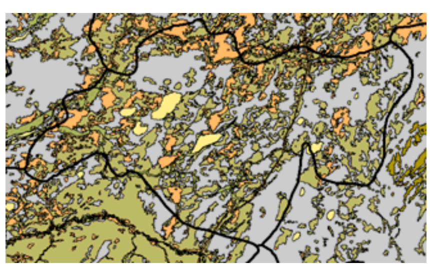
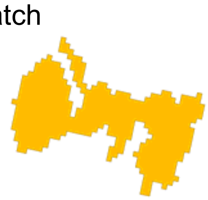
Scale	BEACONs Evaluation
Strata	Regional Planning Units defining the study area 
Landscape	Landscapes (black outline) containing Intrinsic Patches 
Patch	Intrinsic Patch 

Table 1: General description of Intrinsic Patch (IP) types

IP Type	Definition of IP
WATER	Open surface water (lakes, rivers > 20 m and ponds) or flooded lands
MIX	Leading aspen or white spruce mesic stands
UPCON	Upland conifer of leading black spruce and/or jack pine
BOG	1: Treed, open bog (peatland) 2: Lowland leading black spruce and/or larch stands
RIPARIAN	Area described as open grassland or shrub and/or areas that are grassland or shrub and poorly drained
OTHER	Rock, sand etc

The size distribution of IPs for UPCON, MIX, and BOG are characterized by many small IPs in both RPU 10 and 12, with large patches being rare on the landscape. The patch size distribution of the RAN and NONRAN are different. In RPU 10, the RAN under-represents larger patches of BOG and RIPARIAN; however, larger patches of UPCON and MIX are well represented. The RAN under-represents large patch sizes for all patch types (UPCON, MIX, BOG, and RIPARIAN) in RPU 12.

We evaluated representation with respect to the distribution of "large" patches (patches >500 ha) because there are many reasons why a single large patch may have greater ecological potential and conservation importance than a collection of smaller patches with the same total area. In addition, the range of possible patch-level management treatments increases with patch size, and a key function of large IPs is to maintain a natural range of patch states within the RAN through time. For the present study, we suggest that large IPs should be present in the RAN at least in proportion to their availability with the managed forest. We also evaluated the representation of large, intact patches. Large patches were considered intact if they were entirely contained within one of the blocks of intact forest delineated by Global Forest Watch Canada (Lee et al. 2003[2]). In RPU 10, large intact patches of UPCON and BOG are under-represented in the RAN, while large intact patches of MIX are well represented. For MIX, we propose two possible interpretations: 1) the RAN are over-representing large intact MIX IP types or 2) large MIX IP types have already been fragmented in NONRAN landscape, thus the RAN may be representing what *was* there prior to industrial activity. In RPU 12, the proportion of large intact UPCON and BOG is greater in the RAN, and MIX is lower. Overall, large intact patches of UPCON and MIX in the RAN and NONRAN are similar in size. However, the number of large intact patches is very small. RPU 12 is dotted with lakes and rivers that tend to break up an otherwise contiguous landscape of UPCON and BOG IP types.

Another approach to assess the potential of RAs as benchmarks is based on predetermined questions and associated processes specific to forest management. For example, as proof-of-concept only, we evaluated the adequacy of the RAN to act as benchmarks for the effects of forest management on woodland caribou based on the landscape-scale intrinsic patch composition of areas of the critical caribou habitat areas delineated by Arsenault (2003; Fig. 2)[3].

Beyond the composition of intrinsic habitat patches, variation in forest age-structure between RAN and NONRAN lands is of obvious interest. We present a simple qualitative analysis of the forest age structures within the study region. Forest age structures are derived from the same inventory data sets used to delineate the Intrinsic Patches. Of the six IP types, only two (MIX and UPCON) are predominantly composed of forested mapped polygons of potentially determinant age. The main results of this analysis is that forest age structures within the RAN are reasonably representative of those outside the RAN, for both IP classes MIX and UPCON and in both the Boreal Plains and Boreal Shield ecozones. The main exception to this general finding is that the youngest age classes (0-20yr and 20-40yr), though rare overall, are less well represented. This applies

[2] Lee, P., D. Aksenov, L. Laestadius, R. Nogueron, and W. Smith. 2003. Canada's Large Intact Forest Landscapes. Global Forest Watch Canada, Edmonton, AB. 84pp.

[3] Arsenault, A.A. 2003. Status and conservation management framework for woodland caribou (Rangifer tarandus caribou) in Saskatchewan. Fish and Wildlife Technical Report 2003-3. 40pp.

especially to forest of type MIX within the Boreal Plains. Young forest of both IP classes is very under-represented within the Boreal Shield RAN areas.

We highlight several limitations in the analyses employed to date, as well as critical next steps in the identification of candidate benchmark areas through expansion or enhancement of the existing RAN. Hydrologic connectivity – the flow of energy and organisms through the hydrologic cycle - was not addressed, and aspects of the size and spatial configuration of terrestrial ecosystems components should be further explored. We present an approach to delineate benchmark areas. We demonstrate that it is possible to construct system-level benchmarks in the boreal plain and boreal shield ecozones in Saskatchewan including the potential to build system-level benchmarks from existing and proposed representative areas in all RPUs. We conclude at this stage that the ability of the existing RAN to act as benchmarks for forest management in Saskatchewan is constrained by considerations related to size, composition and structure. However, there appear to exist promising opportunities to enhance the system to address these limitations.

Chapter 1: Ecological Benchmarks

1.0 Definition and Role

The concept of ecological benchmarks addresses the need to establish reference sites or controls for detecting the impacts of human activity on ecosystem composition and function (i.e., the full suite of organisms and biological and physical processes inherent in natural systems)[4]. This need arises from recognition that our understanding of the dynamics of ecosystems is incomplete, and thus the outcomes of management decisions are uncertain. These uncertainties include direct and indirect effects of resource exploitation on target and non-target species, alteration of key ecological processes (e.g., natural disturbance and hydrological regimes, and predator-prey dynamics), and changes in ecosystem services (e.g., air and water quality) (Schmiegelow et al. *in review*). Establishment of ecological benchmarks, combined with application of adaptive resource management (Walters 1986), provides a powerful tool for learning and addressing the uncertainty inherent in the management of natural resources and maintenance of functioning ecosystems.

Ecological benchmarks, akin to ecological baselines in most ecological literature, have been defined in the context of their role as references for the detection of change in ecosystems in the presence of human activity (Sinclair 1998, Szaro et al. 1998, Wiersma 2005, and others[5]), reference sites for restoration (Hunter 1997), and as ‘classrooms’ for understanding ecosystem processes and the natural state and range of variation of biotic communities and ecosystems (Arcese and Sinclair 1997, Sinclair 1998, Landsberg and Crowley 2004). As such, ecological benchmarks are sites that experience little or no direct human impact at present and in the future (Sinclair 1998, Sinclair et al. 2002, Wiersma 2005) and that are sufficiently large to: 1) maintain and represent natural processes, habitats, and ecosystem dynamics (Sinclair 1998), 2) contain the historic complement of species (Wiersma 2005), 3) prevent the effects of human activity to penetrate the core of the benchmark (Arcese and Sinclair 1997), and 4) permit monitoring of indicators at an ecosystem scale (Schmiegelow et al. *in review*).

1.1 Ecological Benchmarks for Forest Management Activities

The concept of ecological benchmarks is directly applicable to forest management. If one goal of forest management is ecological sustainability (i.e., maintenance of functioning ecosystems, while permitting the development of forest resources), the role of benchmarks is to serve as classrooms and reference sites for understanding the dynamics of ecosystems and the response of ecosystems to development. In the southern boreal region of Canada, the development of forest resources is occurring rapidly over very large spatial extents. At the current rate of development, given incomplete knowledge, we must learn by doing (Nudds 1999), which is the foundation of adaptive resource management (Walters 1986). In adaptive resource management, ecological benchmarks are controls for management experiments. Without controls,

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[4]Processes are dynamic interactions that occur among and between biotic and abiotic components of the biosphere that act directly, indirectly, or in combination, to shape and form the ecosystem.

[5] Others include: Fule et al. (1997), Hunter (1997), McIntosh et al. (1997), Timoney et al. (1997), Dayton et al. (1998), Williams and James (1998), Millar and Woolfenden (1999)

sliding baselines of reduced naturalness and expectations can occur (Dayton et al. 1998). These have been observed in both terrestrial and marine systems (e.g., Sinclair 1998, Pitcher 2001, Baum and Myers 2004). Such sliding baselines can contribute to the failure to detect the degradation of ecosystems along slow temporal scales until there is a "sudden catastrophic collapse of the original system" (Sinclair 1998), or a loss of resilience to environmental change.

As controls, ecological benchmarks should have the following characteristics:

- The landscape composition and condition of ecological benchmarks should be representative of the range of natural conditions of the region. For example, home range sizes of mammals have been linked to productivity (Harestad and Bunnell 1979, Hundertmark 1997). In order to capture the natural variation of home range sizes, which influences movement rates and fitness, benchmark areas should represent sites of high, medium, and low productivity. The composition and spatial structure of different vegetation types and aquatic ecosystem components should also be adequately represented.

- Absence of human activity (past, present, and future) is advocated. However, human activity is permissible if the activity does not interfere with the process(es) being monitored (Sinclair 1998, Wiersma 2005). For example, recreational fishing or hunting has a low likelihood of interacting with the processes of seed dispersal or pollination. However, fishing and hunting could significantly affect fish and wild game population dynamics. An ecological benchmark (control) must therefore exclude the human activity (treatment) being monitored, as well as activities that could affect related processes.

- Ecological benchmarks must be large enough to capture the full spatial extent of the biological and physical processes being monitored and shield the processes from the influence of activities outside of the benchmark (Arcese and Sinclair 1997). Processes should be operating free of human intervention, recognising that exogenous influences such as air borne pollutants and climate change are unavoidable. Ecological benchmarks should be of sufficient size to experience the largest, anticipated natural disturbance (*e.g.,* fire), and still maintain internal recolonisation sources for species whose habitat is rendered unsuitable by such disturbances (Pickett and Thompson 1978). We refer to ecological benchmarks with these characteristics as system-level benchmarks. When the identification of a system-level benchmark is not possible, a range of benchmark sizes can be considered.

  The concept of a sliding scale for ecological benchmarks addresses variation in the spatial extent of ecological processes occurring in the region of interest, such that as the size of an ecological benchmark increases, the number of processes contained within the benchmark increases. If the spatial extent of a process exceeds the size of a benchmark, the benchmark cannot be considered a control for that process. For example, in Saskatchewan, the minimum mean annual home range size of a female caribou is 208 km^2 or 20,800 ha (Rettie and Messier 2001), and the circular home range[6] of bees is 314 ha. Based solely on size, Selenite Point Representative Area in central Saskatchewan (3,764 ha) is a potential benchmark for monitoring pollination by bees but is too small for monitoring the movement of female caribou relative to anthropogenic disturbances.

In summary, the following description captures the definition and role of ecological benchmarks for forest management, in the context of Saskatchewan's Representative Network:

> *One of the roles of the Representative Areas Network (RAN) is to serve as a "control" for forest management activities. As ecological benchmarks, representative areas provide the opportunity to monitor and compare the outcomes of forest management to the natural system and provide the basis for adaptive management. Ecological benchmarks must capture the range of forest ecosystems they are intended to represent and, to the degree feasible, be of sufficient spatial extent in relation to boreal forest disturbance events.*

[6] Eickwort and Ginsberg (1980) report the honey bees generally forage up to 1 km from the hive, but distances of up to 14 km have been recorded. The area reported is based on a foraging radius of 1 km. (Area of home range = π x foraging radius squared)

Chapter 2: Evaluating Designated Lands as Ecological Benchmarks for Forest Management

2.0 Introduction

One of the characteristics of ecological benchmarks is that a benchmark must be large enough to capture the full spatial extent of the biological and physical processes being monitored. A second characteristic is that the processes being monitored should be intact such that the processes are operating free of human disturbance. Here, we introduce the study area and evaluate the RAN based on the metrics of size and intactness.

2.1 Study Area

The study area (Figure 2.1) was restricted to the commercial forest zone of Saskatchewan, Manitoba, and Alberta and the boreal plains and boreal shield ecozones. The rationale was that we had detailed forest cover data for these regions. Within the ecozones, an ecologically-relevant boundary was established by selecting the Prairie Farm Rehabilitation Administration (PFRA) "Sub_4c" catchment boundaries that intersected the data for the commercial forest zone. Adjustments were made in some areas to include areas of known forestry expansion along the northern study area boundary. We excluded small portions of the southern commercial zone in Saskatchewan, Manitoba and Alberta because the portions fell within the prairie ecozone. The MOU with SEFS includes examining the possibility of establishing benchmarks across provincial jurisdictions. The spatial extent of the forest commercial zone of Manitoba and Alberta included was based on the overlap of one[7] or two whole “Sub_4c” catchment boundaries that extended into the adjacent provinces.

2.1.1 Study Area and Ecozones

The study area intersects two ecozones from the Ecological Land Classification of Canada (Marshall and Schut 1999): boreal plains and boreal shield (Figure 2.2).

[7] One catchment was used if its boundary was entirely or nearly entirely contained within the adjacent province.

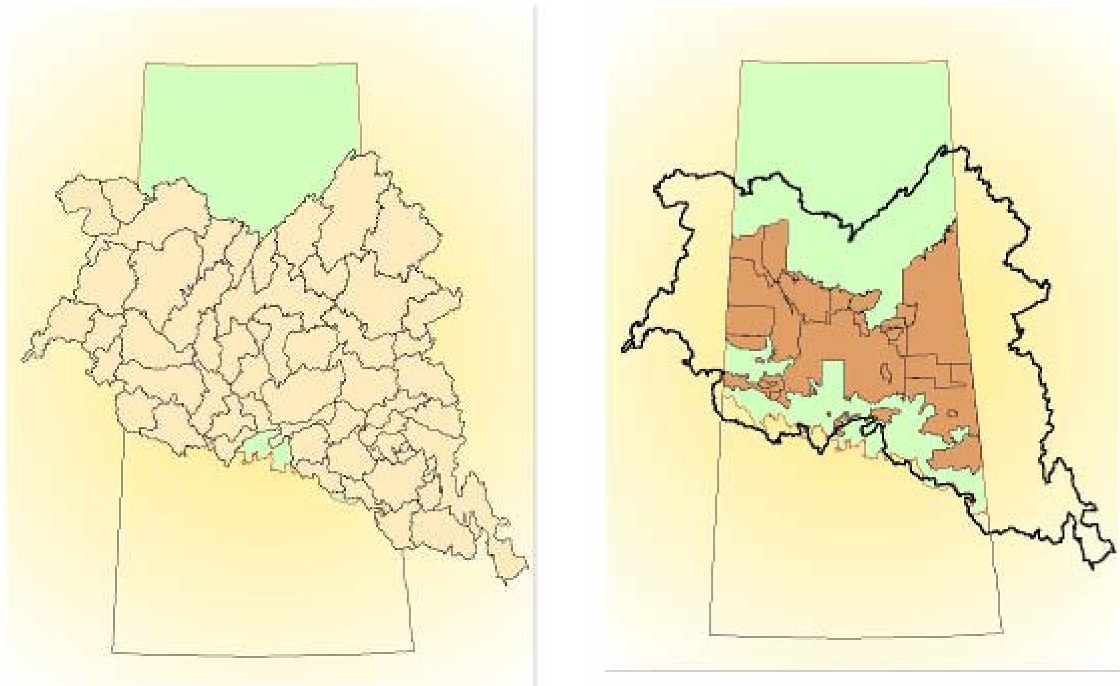


Figure 2.1: Study area for the Saskatchewan Case Study. The image on the left shows the PFRA "Sub_4c" catchment boundaries used to define the study area. The image on the right shows the outline of the study area in black. The brown represents the timber supply zones of the commercial forest zone of Saskatchewan (source: Global Forest Watch Canada).

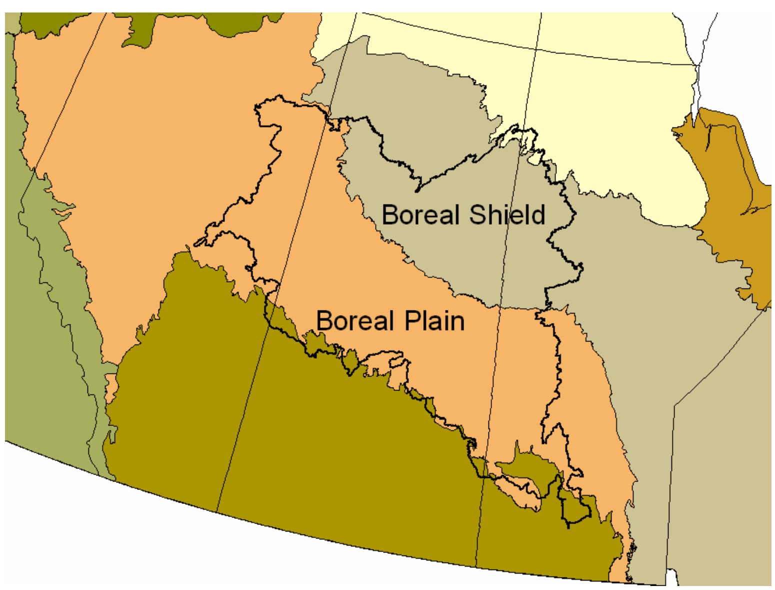


Figure 2.2: The study area intersects the boreal plain and boreal shield ecozones.

2.1.2 Study Area and Regional Planning Units

The study area intersects 4 regional planning units (RPU) (Figure 2.3): RPU 5, RPU 6, RPU 10, and RPU 12. Regional Planning Units stratify the boreal based on the intersection of Major Ocean Drainages and Ecozones. This stratification will be used in the analyses of representation of Representative Areas. A more detailed description of RPUs and an evaluation of RPUs (as well as ecozones and ecoregions) as stratification units can be found in Appendices A and B.

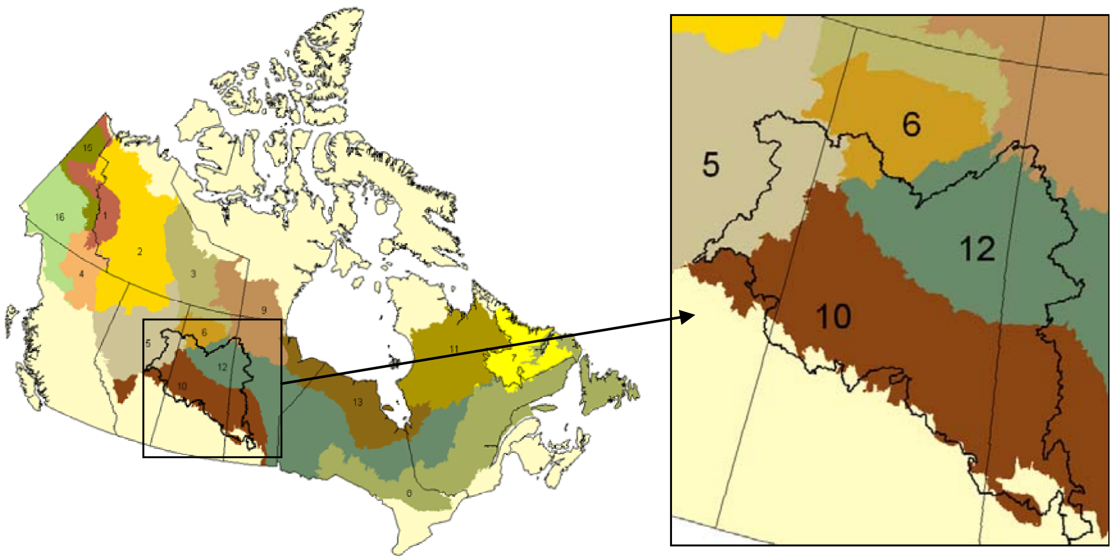


Figure 2.3: The study area intersects RPUs 5, 6, 10, and 12.

2.1.3 Representative Areas Network (RAN)

The designated lands of the RAN consist of many classes of areas with varying levels of protection: National and Provincial Parks, Recreation areas, Wilderness Area, Natural Areas, Ecological Reserve, Game Preserve, Park Reserve, Provincial Forest, Provincial Historic Site, Crown land, Representative Areas, Ducks Unlimited management areas, Wildlife Management Area, and Wildlife Refuge. Generically, we refer to all designated lands in the RAN as **Representative Areas (RAs)**. The designated lands that fall within the study area were included in all analyses, and collectively, are referred to as the **RAN** area. The **NONRAN** area refers to the forested landbase outside the RAN.

2.2 Sliding Scale for Ecological Benchmarks

System-level benchmarks (SLB), as described in Chapter 1, are ecological benchmarks of sufficient size to experience the largest, anticipated natural disturbance (*e.g.,* fire), and still maintain internal recolonisation sources for species whose habitat is rendered unsuitable by such disturbances. However, the spatial requirement for a SLB is not always achievable because of landscape condition or political constraints. Without diminishing the value of SLBs, the utility of smaller areas as ecological benchmarks for appropriate processes should not be discounted. Provincial and national parks and other designated lands, which are typically small relative to SLBs, present practical options for such benchmarks, due to historical restriction of human use and existing legal protection.

It is useful to consider a framework that allows for inclusion of a range of benchmark sizes in a sliding scale designed to capitalize on opportunities and achieve varying objectives. Such a sliding scale should be tied to the spatial extent of processes (see Table 2.1) relevant to the human activities being monitored. In other words, the processes selected for monitoring inform the size requirements for benchmarks. For example, Prince Albert National Park (PANP) is the largest RA (3957 km^2; Table 2.2) in Saskatchewan. Figure 2.4 illustrates the size of PANP relative to several large-scale processes operating in the region. PANP could potentially serve as a benchmark for any process that is smaller than the park, recognizing that landscape composition and spatial structure must also be considered. For processes operating at scales larger than PANP, there is not a suitable benchmark in the RAN. It is worthy of mention that while PANP is sufficient to capture the largest observed fire for the province (3017 km^2), the estimated maximum fire size (4251 km^2 boreal plains; Cumming and Mackey unpublished data) and associated landscape dynamics of the characteristic fire cycle far exceed the bounds of the park.

In Figure 2.5, we examine the size of RAs (Table 2.2) in relation to a selection of home range sizes for terrestrial and aquatic fauna residing in the boreal region of Saskatchewan. Home range sizes are not processes *per se* but do correspond to the spatial extent within which individuals move and reproduce. For wide-ranging animals, such as lynx, there are 7 RAs that could potentially contain the home range of a lynx (Figure 2.5). However, a benchmark for lynx should be able to support a viable population of the species. As a very simplistic illustration of this concept, based on the 50-500 rule[8] advanced in consideration of minimum viable populations (MVP), the largest RA (PANP) is not sufficient to support a MVP of lynx, and thus cannot act as an adequate control for activities that influence the population dynamics of this species. In general, for wide-ranging species, RAs in SK are too small to capture important aspects of life history that contribute to regional persistence. For more narrow-ranging species, such as the territorial ermine, areas as large as 200 km^2 (N=50) and 2500 km^2 (N=500) could be required to support MVPs; RAs of these spatial extents do exist. Whether the RAs have the landscape composition and spatial structure required to support the populations still needs to be addressed.

[8] This rule states that, in order to avoid inbreeding, a local population of 50 individuals is the minimum viable population size. For long term survival, at least 500 individuals are required so that a population will not lose genetic variability and can cope and evolve with changing environments (Soulé and Wilcox 1980).

Ideally, the potential of RAs as benchmarks would be assessed based on predetermined questions and associated processes specific to forest management. The next step in this type of evaluation is to identify these questions and processes. Examples of processes relevant to forest management include those related to soils (Chanasyk et al. 2003) and soil organisms (Addison 1996), aquatics (e.g., water quality and dissolved organic carbon; Carigan et al. 2000), stream flow (Buttle and Metcalfe 2000), natural disturbance (e.g., fire and climate change; Weber and Flannigan 1997), carbon and nitrogen dynamics (Peng et al. 2003), fish (Tonn et al. 2003, Ripley et al. 2005), invertebrates (Buddle et al. 2000), and pollination (Carter 2005), to name a few.

Table 2.1: Examples of processes and associated spatial and temporal extents for Saskatchewan and beyond.

Element	Block	Sub-unit	Spatial	Temporal	References
Natural Disturbance					
Fire	Saskatchewan	largest observed fire	3017 km^2		Canadian Large Fire Database
		estimated max fire size (boreal plains)	4251 km^2		Cumming and Mackey (unpublished data)
		estimated max fire size (boreal shield)	3518 km^2		
Blowdown	Saskatchewan		6373 ha (Blow12X-13X); 1461 ha (Hills Park)		Source: SEFS
Flooding	Saskatchewan		2,320 km^2		Bart Oegema, pers. comm.
Spruce Budworm	Saskatchewan		8088 km^2		Robert Moore, pers. comm.
Trees	conifer	white spruce	seed rain15-250m	seed production typically 25 yrs; maximum lifespan 350 yrs	de Groot et al. 2003; Stewart et al. 1998
	conifer	white spruce	seeds maximum disperal 100-300m (1) & 10-60m (2)	seeds annually; seed life 2 yrs (ref. #3)	1) Zasada (1971) as cited in Galipeau et al. (1997); 2) Zasada and Lovig 1983 as cited in Asselin at al 2001; 3) Nieustadt and Zasada (1990) as cited in Stewart et al. (1998)
	conifer	balsam fir	seed maximum disperal 160m (1) & 60-115m (2)		1) Sins et al. (1990) as cited in Galipeau et al. (1997); 2) Frank 1990 as cited in Asselin et al 2001
Mammals				Maximum lifespan	
Home range	Carnivore	small (weasel)	3.74 km^2	10 yrs	Jedrzejewski et al. 1995
		medium (lynx)	1400 km^2	27 yrs	Linnell et al 2001
		large (wolf)	692 km^2	16 yrs	Cook et al. 1999
	Herbivore	small (mouse)	36/ha density	8 yrs	Davidson and Morris 2001
		medium (snowshoe hare)	0.28 km^2	7 yrs (estimate)	Hodges 1999
		large (caribou)	280 km^2	20 yrs	Ferguson and Elkie 2004
		Beaver	4.3 ha		Naiman R.J & Johnston, 1990
	Omnivore	Red Squirrel	0.65ha; dispersal 1km		Larsen and Boutin 1994
Movement	Carnivore	small (weasel)	0.93 km		Jedrzejewski et al. 1995
		medium (lynx)	1100 km		Schwartz et al. 2002
		large (wolf)	87 km		Cook et al. 1999
	Herbivore	small (mouse)	1.98 km		Teferi and Millar 1993 , Rehmeier et al. 2004
		medium (snowshoe hare)	16 km		Gillis and Krebs 2000 Snowshoe dispersal
		large (caribou)	15.7 km		Ferguson and Elkie 2004

Table 2.1: continued

Element	Block	Sub-unit	Spatial	Temporal	References
Birds					
Territory	songbirds	red-eyed vireo	territory (0.11-0.6 ha)		Marshall and Cooper 2004
		black-throated green warbler	territory (2-2.6 ha)		Theresa Hannah, pers. comm.
Movement		warbler (hooded warbler)	movement up to 2.5 km when breeding		Norris and Stutchbury 2001
Fish				Maximum lifespan	
Home range	Piscivore	small (yellow perch, max 38cm)	0.5-2.2ha	10 yrs	Fish and Savitz 1983, Scott and Crossman 1998
		medium (walleye, brook trout and arctic char, max 106cm)		walleye (20yrs), brook trout (8yrs), arctic char (40 yrs)	Scott and Crossman 1998
		large (northern pike, lake trout and chinook salmon, max 147cm)	northern pike (28-52ha), lake trout (62-199ha)	Northern pike (26 yrs), lake trout (25 yrs +), chinook salmon (9 yrs)	Jepsen et al. 2001, Scott and Crossman 1998, Schmalz et al. 2002, Walch and Bergersen 1982
	Non-piscivore	small (fathead minnow, longnose dace, max 10cm)		fathead minnow (3 yrs), longnose dace (5 yrs)	Danylchuk and Tonn 2003, Scott and Crossman 1998
		medium (arctic grayling and white sucker, 63cm)		arctic grayling (12 yrs), white sucker (15 yrs)	Scott and Crossman 1998
		large (lake whitefish and lake sturgeon, max 226cm	lake sturgeon (1528ha)	lake whitefish (28 yrs), lake sturgeon (80 yrs)	Haxton 2003, Noakes et al. 1999, Scott and Crossman 1998
Movement	Piscivore	small (yellow perch, max 38cm)			Scott and Crossman 1998
		medium (walleye, brook trout and arctic char, max 106cm)	walleye (161km), arctic char (1690km)		Deccico 1992, DiStefano and Hiebert 2000, Rasmussen et al. 2002, Scott and Crossman 1998
		large (northern pike, lake trout and chinook salmon, max 147cm)	northern pike (16km), lake trout (161km), chinook salmon (1931km)		Ovidio and Philippart 2002, Rossel and MacOscar 2002, Scott and Crossman 1998
	Non-piscivore	small (fathead minnow, longnose dace, max 10cm)			Scott and Crossman 1998
		medium (arctic grayling and white sucker, 63cm)	arctic grayling (3.4km), white sucker (1km)		Brown et al. 2001, Hughes 2000, Reid et al. 2002, Scott and Crossman 1998
		large (lake whitefish and lake sturgeon, max 226cm	lake whitefish (241km), lake sturgeon (402km)		Anras et al. 1999, Auer 1996, *Knights et al. 2002* , Scott and Crossman 1998
Insects					
	Defoliator - Aspen	forest tent caterpillar	suggested that forest fragmentation exacerbates outbreaks;	outbreak every 10 yrs, lasting for 4-6 yrs	Roland 1993; Fleming 2000; Cerezke 1991
	Ddefoliation or mortality of White Spruce and Balsam Fir	spruce budworm	variable size of outbreaks; 150 000 ha infested in 1998 in SASK	outbreaks cycle 20-60 yrs, lasting 5-15 years; depends on forest age structure; mortality high in mature stands	Fleming 2000; Cerezke 1991
Home range	Pollinator	honey bees (Bombus sp)	314ha circular home range; movement <14km		Eickwort and Ginsberg (1980)
Water	Saskatchewan	Range of Sub Drainage Basins intersecting the boreal plains and boreal shield of Saskatchewan	7925-82719 km^2		Watershed Atlas of Canada
Peatlands	Boreal Region			8,000 years	

Table 2.2: Representative Areas greater than 10 km^2 in Saskatchewan in the study area

	Area (km2)		
Name	**Total**	**Lake**	**Terrestrial**
E.B. Campbell	10	1	9
Woody River - Spirit Lake	11	3	8
Waskwei River	13	0	13
Brockelbank Hill	13	0	13
Anglin Lake	15	3	12
Woody River - Woody Lake	16	4	12
Amisk Lake	17	2	15
Nisbet Trails	18	0	18
Sand Lakes Provincial Park	20	1	19
Greenbush River	25	0	25
Makwa Lake	26	4	22
Big Buffalo Beach	36	6	30
Selenite Point	38	9	29
Sturgeon-weir River	48	0	48
Fir River	49	0	49
Pasquia River	49	0	49
Halldorson Bay	68	15	53
Candle Lake - Minowukaw Beach	78	2	76
Caribou Flats	96	10	86
Bronson Forest	154	25	129
Clarence-Steepbank Lakes	163	6	157
Budd Lake	179	3	176
Primrose Lake Provincial Ecological Reserve	195	10	185
Greenwater Lake	209	15	193
Wildcat Hill	213	3	210
Whiteswan Lakes	307	48	259
Mari Lake	311	74	237
Jan Lake	354	139	216
Perry Lake	395	83	312
Narrow Hills	583	20	563
Wapawekka Hills	679	27	652
McCusker Lake	1394	84	1310
Meadow Lake	1680	354	1326
Seager Wheeler Lake	1772	60	1712
Clearwater River	2350	56	2294
Lac la Ronge	3202	1789	1414
Prince Albert National Park	3957	398	3558

Figure 2.4: The spatial extent of natural disturbances and large-scale process operating in Saskatchewan relative to Prince Alberta National Park, the largest park in Saskatchewan. Note: X-axis is not to scale.

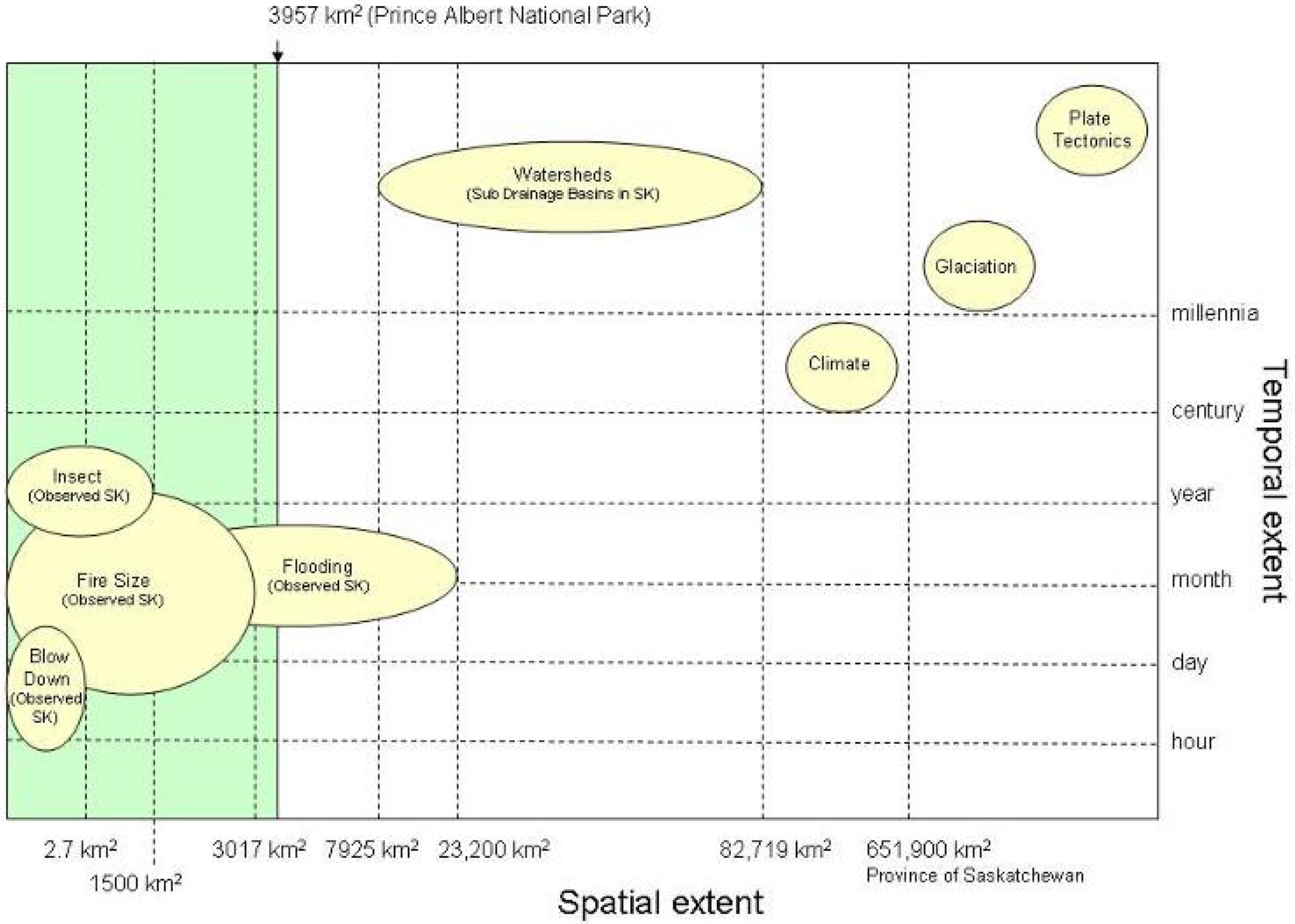
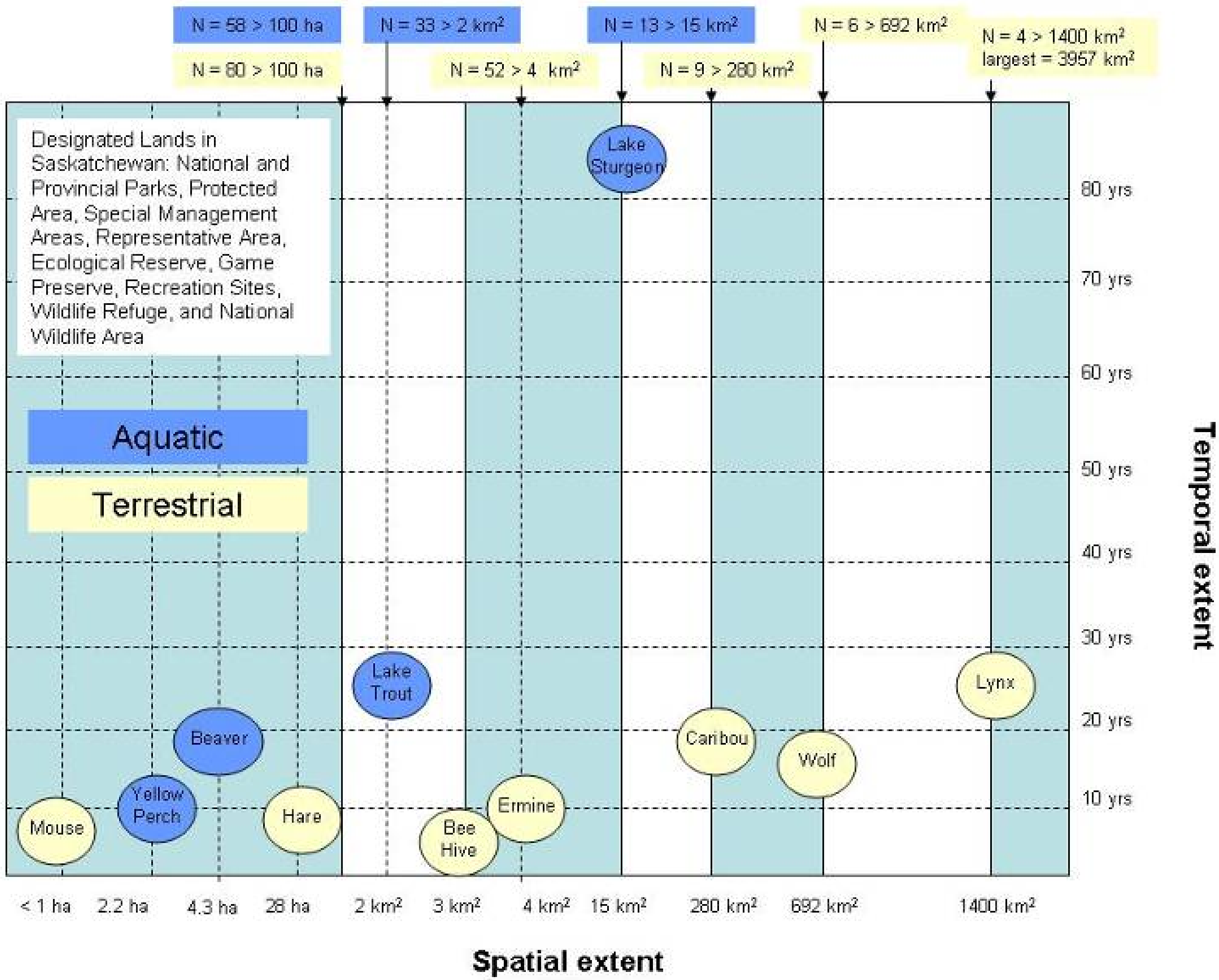


Figure 2.5: The spatial extent of home ranges and the number of RAs in the study area that capture the home ranges.



2.3 Intactness

Intactness is a measure of the absence of human activity and is a proxy for the intactness of biological and physical processes. The following assessment of the Representative Areas Network is based on Canada's Forest Landscape Fragments produced by Global Forest Watch Canada (Lee et al. 2006). Canada's Forest Landscape Fragments captures terrestrial and aquatic environments (e.g., lakes) and consists of contiguous blocks of intact landscape greater than 10,000 ha (Figure 2.6). This assessment is only a measure of the potential of a RA with respect to intactness because it ignores the composition and size structure of the landscape. Of the RAs in the study area greater than 10 km^2 (N = 38), 33 intersect the intact landscape (Table 2.3). Intact areas within the RAs range from a total area of 7 to 3430 km^2. It is encouraging that a number of the more northern representative areas intersect very large intact patches (Figure 2.6) into which the RAs could expand if intactness was a priority for ecological benchmarks for forest management.

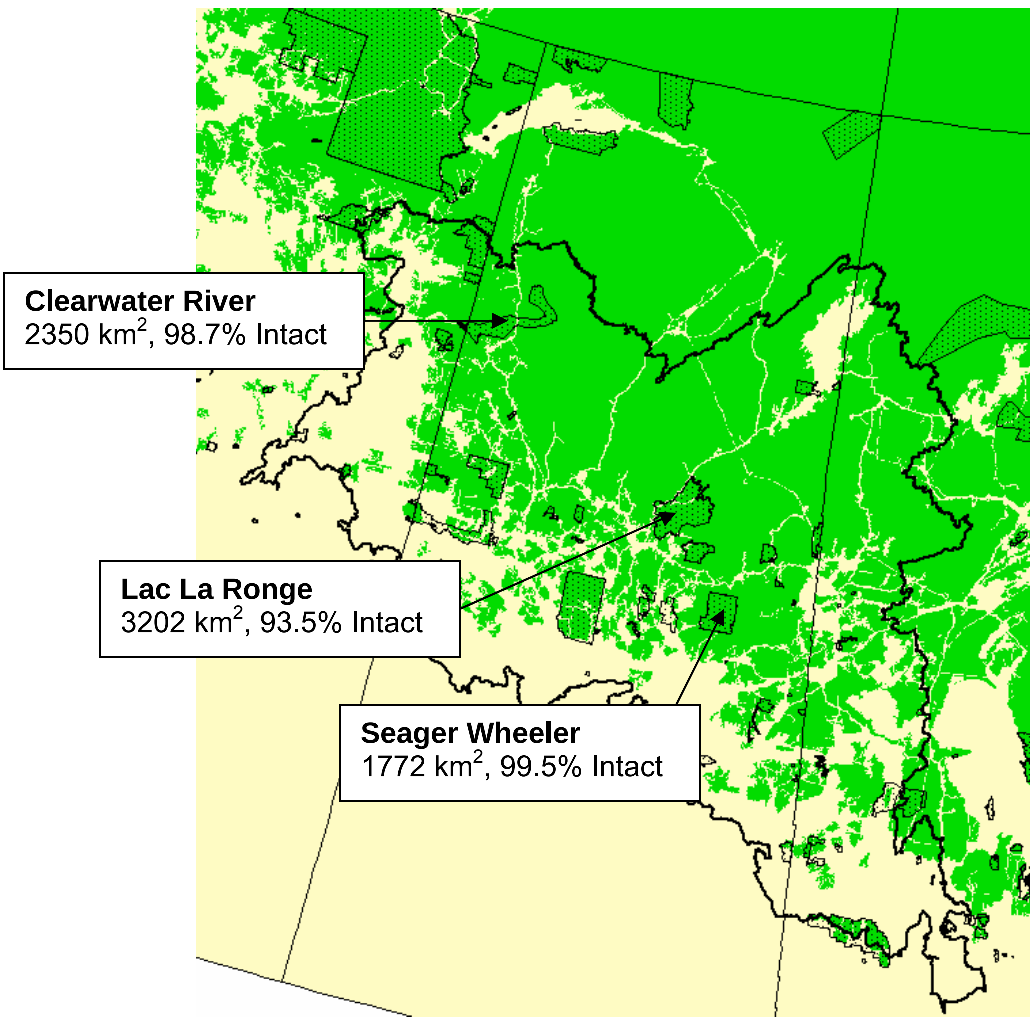


Figure 2.6: An overlay of the GFWC Canada's Forest Landscape Fragments (green) and Protected Areas > 10 km^2 (dotted) in and around the study area.

Table 2.3: The area and percent intactness of Saskatchewan Representative Areas (>10 km^2; N=38) in the study area. The protected areas are listed by % Intact from smallest to largest. Area Intact and % Intact are aquatic and terrestrial environments combined.

Name	Total Area (km^2)	Area Intact (km^2)	% Intact
Upper Armit River	10	0	0
E. B. Campbell	10	0	0
Nisbet Trails	18	0	0
Makwa Lake	26	0	0
Greenwater Lake	209	0	0
Narrow Hills	583	127	21.8
Fir River	49	14	29.3
Bronson Forest	154	52	33.4
Greenbush River	25	9	36.2
Brockelbank Hill	13	5	37.8
Woody River - Woody Lake	16	8	48.0
Meadow Lake	1680	854	50.9
Waskwei River	13	7	55.0
Anglin Lake	15	9	61.1
Whiteswan Lakes	307	192	62.6
Big Buffalo Beach	36	26	70.8
Selenite Point	38	30	79.8
Candle Lake - Minowukaw Beach	78	63	81.3
Perry Lake	395	321	81.3
Halldorson Bay	68	56	82.0
McCusker Lake	1394	1169	83.8
Prince Albert National Park	3957	3430	86.7
Caribou Flats	96	84	87.1
Mari Lake	311	278	89.3
Clarence-Steepbank Lakes	163	147	90.6
Jan Lake	354	326	92.1
Wapawekka Hills	679	630	92.8
Lac La Ronge	3202	2995	93.5
Woody River - Spirit Lake	11	10	93.8
Primrose Lake Provincial Ecological Reserve	195	186	95.6
Clearwater River	2350	2319	98.7
Seager Wheeler Lake	1772	1764	99.5
Sturgeon-weir River	48	48	99.9
Wildcat Hill	213	213	100
Budd Lake	179	179	100
Sand Lakes Provincial Park	20	20	100
Pasquia River	49	49	100
Amisk Lake	17	17	100

Chapter 3: Evaluating RAs for Forest Management: Representation of Intrinsic Patches

3.0 Introduction

The primary objective of this exercise is to evaluate the adequacy of the existing designated lands in Saskatchewan's RAN as ecological benchmarks for forest management activities. We structured our evaluation as a 3-level hierarchical representation problem wherein various patch and landscape scale attributes in and outside of the RAN are measured and compared within strata. Secondarily, we assess the validity of using enduring features and ecoregions for designing the RAN, both of which have been used prior to assess and build the existing and proposed RAs for Saskatchewan (Beveridge et al. 1998, Wright et al. 1998, Forest Ecosystems Branch 2003).

In order to assess the RAN, we stratified the study area into units that can be compared. We defined 3 hierarchical scales: intrinsic patch (IP), landscape, and regional planning units (RPUs) (Figure 3.1).

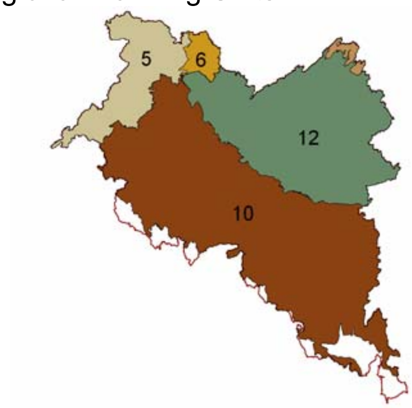
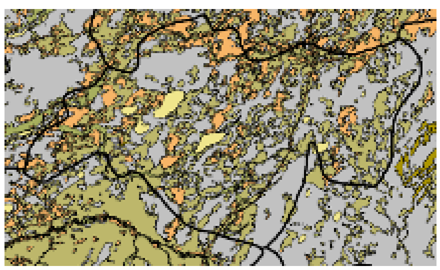
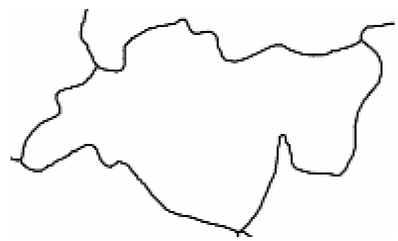
Intrinsic patches (IPs) are the finest scale. We use a small number of broad forest or landcover classes derived from inventory data to delineate spatial units which are expected to remain invariant under natural disturbances, stand dynamics (e.g. natural regeneration and succession) and forest management. Each IP is characterised by its class, area and spatial location. IPs provide two fundamental types of measurements for assessing aspects of representation that are relevant to forest management: patch size and patch composition. The representation of patch size can be assessed quantitatively by comparing between RAs and non-RAs within a stratum. However, patch compositions (or proportional areas of patch types) within RAs and non-RAs can only be qualitatively assessed. In order to assess patch composition in a quantitative manner, we must move up the hierarchy to the landscape scale.

Landscapes, in this context, are collections of patches for which patch compositions can be derived (e.g., 40% patch A, 55% patch B, 5% patch C). Based on the landscapes, we can derive a multivariate distribution of landscape compositions. These multivariate distributions can then be used to quantitatively assess the representation of patch composition in RAs and non-RAs within a stratum (e.g., boreal plains). The distributions can also be used to assess the ecological relevance of stratification units (e.g. regional planning units, ecozones, ecoregions, enduring features) for assessing representation.

In this section of the report, we will describe the elements and methods for the assessment of representation. We will

1. define Intrinsic Patches and motivate their use as the fundamental units for evaluating representation at patch and landscape scales,
2. explain choice of enduring features as landscape units, and
3. describe the criteria for representation

Figure 3.1: The three hierarchical scales used by BEACONs and SEFS to evaluate the representativeness of the Saskatchewan's RAN.

Scale	BEACONs Evaluation	SEFS Evaluations
Stratification Unit	Regional Planning Units 	Ecoregions 
Landscape	Enduring Features with Intrinsic Patches 	Enduring Features 
Patch	Intrinsic Patch 	Sample plot opportunities 

3.1 Intrinsic Patches

To motivate the concept of intrinsic patch structure and its application in the evaluation of benchmarks for forest management, we begin with the characteristic upland mesic sites of the boreal mixedwood region (sensu Kabzems et al. 1986) To a first approximation, these are the sites:

a) that can support populations of trembling aspen (*Populus tremuloides*) and/or white spruce (*Picea glauca*);
b) where either species can exclude the available alternatives (*Pinus banskiana*, *P. contorta*, *Picea mariana* and *Larix laricina*); and
c) where commercial forestry in the western boreal forest, including much of Saskatchewan's allocated lands, is concentrated

Regionally, forest management changes forest age-class structures and may also change the relative abundances of certain stand types or species combinations: "unmixing the mixedwood" (Cumming *et al.* 1994). At landscape levels, forest management reduces the abundance of old forest and changes the patch size structure in fairly predictable ways (Figure 3.2). Altering the size-class structure and spatial distribution of residual or dynamically managed stands of post-rotation age forest may be the most promising strategy to maintain old-forest dependent species on managed lands (other than an actual reduction in cut). If such strategies are foci of future management experiments, a system of benchmarks must function as controls for such experiments and whatever else may be required of it.

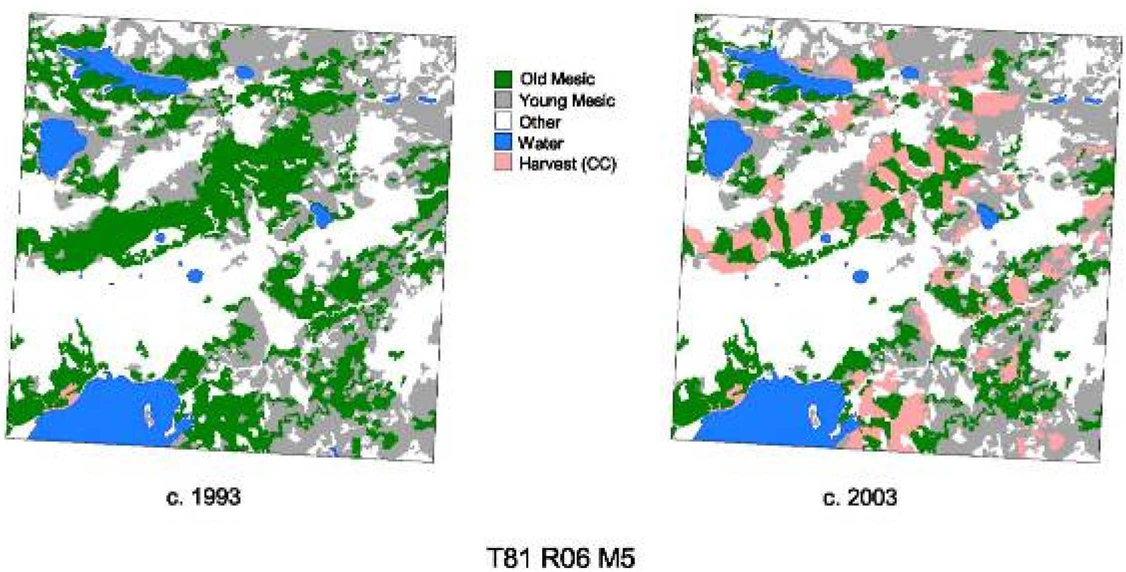


Figure 3.2: Classified forest inventory maps for a single landscape unit (a township, ~9700 ha) before (left panel) and after (right panel) the first pass of a conventional two-pass clearcut harvesting schedule. The pre-harvest landscape appears to have been in an essentially natural state. Upland mesic sites were identified from forest inventory attributes and stratified by age as old mesic (≥ 90yr) and young mesic (<90yr).

Before establishing criteria for benchmarks as controls for forest management, we must first decide: what are the treatment units? Harvest blocks are not distributed randomly over a forest estate or within landscapes or compartments. Harvest block layouts typically do not follow the boundaries of mapped inventory stands. Neither mapped stands nor cutblocks retain their vegetation attributes or spatial identity through time. Even within relatively homogeneous patches of old mesic forest, the distribution and abundance of some biotic indicators such as forest songbirds are sensitive to variation in forest structure at a scale of at least 500 m (Vernier et al. 2002). For these reasons, we do not consider that existing mapped stands, or individual cutblocks or leave areas, to be appropriate choices of treatment unit at the patch scale. Some larger and more enduring structures need to be identified. If we could look more closely at the mesic forests in Figure 3.2, we would find that the areas of old mesic forest (dark green) were composed of discrete patches of varying age (90-150+ years), some with canopies dominated by aspen, others by white spruce. Looking more closely still, beneath the canopy, we would see that some patches of younger mesic forest (grey) were on a "white spruce" trajectory while others were destined to remain deciduous for a very long time. Obviously, barring fire or harvesting, all of the younger mesic forest will become old eventually, probably within 30-60 years in these cases. The patches created by dissolving the boundaries between young and old mesic forest are the intrinsic mesic patches for the area shown in Figure 3.2. We argue that these are the appropriate (mesic) treatment units for management experiments.

Forest patches are intrinsic by virtue of their surficial geology, climatic conditions and other characteristics that facilitate or limit the establishment and survival of species. In the boreal mixedwood, dominant vegetation species (whether trees, shrubs, mosses or sedges) are distributed along gradients of soil moisture or drainage which are determined by parent material (Kabzems et al. 1986), slope position (Bridge and Johnson 2000) and climate (Hogg and Schwarz 1997, Johnstone and Chapin 2003). These "Permanently Operating Factors" can produce very small patches such as pine growing on a few square meters of sand hummock within a black spruce bog, intermediate sized patches such as the intrinsic mesic patches of Figure 3.2, or very large patches of contiguous wetlands areas. Within intrinsic patches, processes like succession, fire, seed dispersal and regeneration play out. The result of this play is visible as variation in the more refined attributes such as age and canopy species within an intrinsic patch. We refer to these and any other attributes measurable from digital forest inventories as the IP state. The stochastic nature of ecosystem processes means that a presently observed IP state is unlikely to persist for long or to ever be repeated (Cumming et al. 1996). Nevertheless, IPs retain their identity through meaningful time: bogs generally do not turn into aspen stands. We expect that internal spatial and temporal heterogeneity increases with intrinsic patch size (i.e., because large patches are less likely to be entirely affected by a single disturbance event, large patches have greater internal site-heterogeneity, etc.) Based on this premise, in order to capture the inherent variability of patch state in time and space, a benchmark should contain many intrinsic patches with a distribution of sizes and spatial contexts comparable to the managed landscapes for which the benchmark is a control. Intrinsic patches in this evaluation are the fundamental units of representation.

Obviously, upland mesic forest is not the only vegetation type directly affected by forest management. Moreover, forest harvesting may indirectly affect ecological processes in areas where no harvesting occurs. Therefore, all vegetation types within the managed lands are of conservation interest, and need to be represented within benchmarks. The arguments we developed for the existence of intrinsic mesic patches apply to other vegetation types as well (and also of course to permanently non-vegetated areas such as lakes or exposed rock).

For the Saskatchewan case study, we identified 5 intrinsic patch types based on Rettie et al. (1997) and the Saskatchewan Provincial Forest Inventory: WATER, MIX, UPCON, BOG, RIPARIAN and OTHER. The IP types are described in Table 3.1. In the report, MIX BOG, UPCON, and are often collectively referred to as the terrestrial subcomposition of IP types. For additional details on how IPs were constructed, refer to Appendix E.

Table 3.1: General description of Intrinsic Patch types (IPs) and relationship to stand types defined in Rettie et al. (1997)

IP Type	Definition of IP	Stand Types and Definitions (Rettie et al. 1997)
WATER	Open surface water (lakes, rivers > 20 m and ponds) or flooded lands	
MIX	Leading aspen or white spruce mesic stands	B: Stands of all ages and closure classes with the canopy dominated by white spruce frequently in combination with aspen and/or black spruce; stands with mixed canopies dominated by black spruce in combination with aspen and/or white spruce. D: Overstory is dominated by aspen of all ages, occasionally combined with white spruce or jack pine, canopy closure >65% (unless the stand was selectively logged for white spruce then the closure may be <45%)
UPCON	Upland conifer of leading black spruce and/or jack pine	A1: Jack Pine stands of <55% canopy cover and > 40-years-old C: Young (<40-years-old) jack pine canopy of all closure classes F: Mixed jack pine/black spruce canopies, pure black spruce stands of <55% cover and <90-years-old, jack pine stands of >40-years-old and >55% closure G: Pure black spruce stands of all ages and >55% canopy closure
BOG	1: Treed, open bog (peatland) 2: Lowland leading black spruce and/or larch stands	A2: Black spruce bog and may include tamarack in relatively open canopy of <45% closure E: Mature black spruce stands (>90-years-old) with canopy closure of <55%, includes mixed stands of black spruce and tamarack.
RIPARIAN	Area described as open grassland or shrub and/or areas that are grassland or shrub and poorly drained	
OTHER	Rock, sand etc	

3.2 Enduring features as landscape units

Figure 3.3 illustrates two classified ≈10,000 ha forest planning units in northeast Alberta, each with about 15% by area of old mesic forest. The distance between them is about 200 km, so most ecological processes acting on these units must be weakly coupled. In particular, their disturbance histories are effectively independent. Thus, it is a highly improbable event that both contain the same amount of forest in some age-class and cover-type. Similarities or differences in the present species composition and age-structure of the mesic forest (and other vegetation types) among such units is of importance for forest management planning, for assessing relative potentials to support populations of various habitat specialists, for sampling designs and similar immediate purposes. We argue that none of these considerations are central to an areal units' utility as a benchmark component. Rather, we propose that the fundamental landscape-scale characteristic is the intrinsic landscape patch structure (ILPS). An ILPS is determined by the proportional areas, size distributions and spatial arrangement within the landscape of each IP class. These structures are temporally stable if the individual IPs are as we assume. The ILPS influences the characteristics and relative importance of intermediate-scale ecological processes such as disturbance regimes (Cumming 2001, Krawchuk et al. in press) For example, of the two units in Figure 3.3, the one on the left has a larger proportion of mesic forest; other things being equal, it would be more likely to maintain a minimum target abundance of old mesic forest through time. The ILPS incorporates many of the factors that determine what biotic communities a landscape can potentially support. It follows that benchmarks for forest management should include a representative sample of landscape structures with sufficient replication to allow for intra-landscape and intra-patch dynamics.

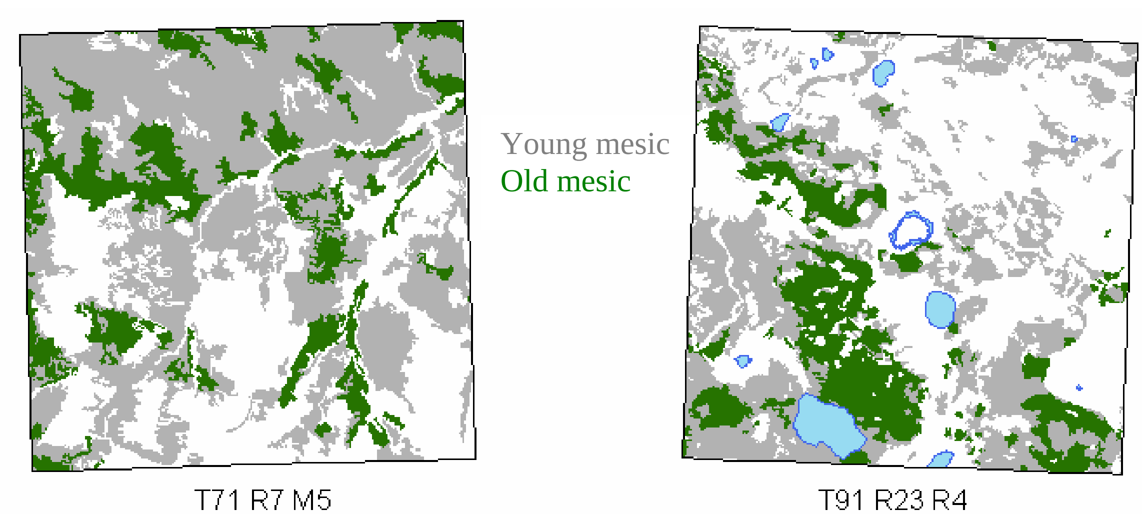


Figure 3.3: Two landscapes with similar amounts of old mesic forest (green) but quite different total amounts of mesic forest (green + grey).

Ideally, landscape boundaries should: correspond to processes intermediate to the characteristic scales of the IPs and RPUs; be congruent with IP boundaries; maximise between-landscape variance in ILPS attributes; and minimise the spatial-autocorrelation in ILPS attributes. Some of the challenges in defining such boundaries are illustrated in Figure 3.4. Obviously, no regular grid corresponds to natural landscapes. At scales of 10,000 ha, grids seriously distort IP structures by splitting patches across artificial boundaries. Natural landscapes in the boreal plains may be up to an order of magnitude larger (Cumming et al. 1996). It is not obvious how many distinct landscapes occur in Figure 3.4 (there seems to be more than one) nor how they could objectively be delineated. Delineation of landscape units is a problem outside the scope of this study. However, Enduring Features may approximately satisfy most of the criteria. They are intermediate in size between IPs and RPUs (we examine the congruence of EF and IP boundaries in Appendix C). They are nested within ecoregions and nearly so within RPUs. They were delineated from generalised maps of soil development, surficial geology and topography (the determinants of IPs), and therefore, should correspond to spatial variation in landform processes within RPUs. EFs larger than 5,000 ha (the minimum size used in our analysis) have median and mean sizes of 293 and 645 km^2, respectively and so may be large enough to represent statistically independent spatial units. For these reasons, **we adopted EFs as the landscape units for this study.**

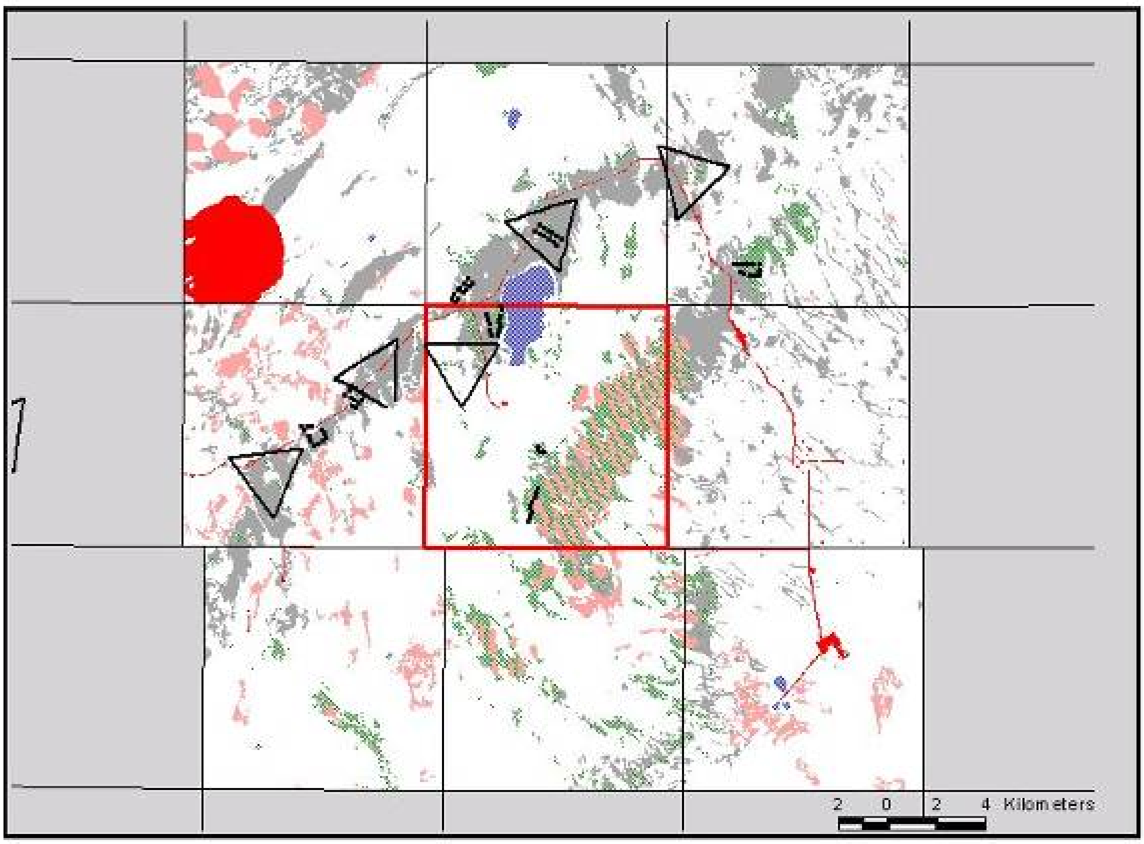


Figure 3.4: The intrinsic mesic patch structure for a 30x30 km region in NE Alberta. Old (90+yr) mesic patches are coloured green, young mesic grey, recent clearcuts (all formerly OM) pink, lakes in blue, roads and permanent clearings red, and other forest or naturally vegetated non-forested areas white. The township grid (~10,000ha) is overlaid.

3.3 Criteria for representation

3.3.1 Patch scale

There are many reasons why a single large patch may have greater ecological potential and conservation importance than a collection of smaller patches with the same total area. In addition, the range of possible patch-level management treatments increases with patch size (Figure 3.2). The mean and range of IP sizes varies over space (Figure 3.4) and, within limits, this variation is independent of the total local abundance of a given IP class. Because the RAN is composed of discrete spatial units that were not randomly constructed, their patch size structures are an important component of representation. One strict representation criterion is that size distributions of IPs within the RAN should be a random sample from the population within the entire strata (i.e., RPUs). We test for significant differences in size distributions by Kolmogorov-Smirnov (KS) tests. To qualitatively assess the importance of any differences, we report the mean sizes of RAN and non-RAN IPs. The size distributions are highly skewed (most patches are smaller than a few ha), long-tailed (with some very large patches) and right-truncated (sizes are finite and the largest patches are small relative to the size of the study region). Because of these characteristics, we have not yet determined a statistically valid and informative test for differences in the means.

We also evaluate representation with respect to the distribution of "large" patches. Although the appropriate threshold between large and small presumably varies among IP classes, we adopt a uniform value of 500 ha. Because the number of large patches is relatively low (less than 0.5% for terrestrial classes), KS and means tests on the full size distributions are uninformative with respect to representation of large patches. In addition to the presumed importance of patch size as such, a key function of large IPs is to maintain a natural range of patch states within the RAN through time. Therefore, it is not straightforward to specify quantitative representation criteria for large IPs. Detailed spatial simulation modelling may be needed to estimate how many large IPs are necessary to maintain variation within acceptable limits. For the present study, we suggest that large IPs should be present in the RAN at least in proportion to their availability with the managed forest. We compare the proportional abundances, mean sizes and proportional total areas of large IP classes for UPCON, MIX and BOG between the RAN and the target strata. We qualitatively assess representation of large IP size distributions with quantile-quantile plots.

The importance of intactness to ecological benchmarks has been discussed previously. We measure intactness at the patch scale by the frequency and total area of large intact IPs. Large IPs were considered intact if they were entirely contained within one of the blocks of intact forest delineated by Global Forest Watch Canada (Lee et al. 2003). We assess representation of this indicator by comparing the relative frequencies, mean sizes and total areas of large intact IPs between the RAN and the target strata, as above. Because sample sizes were small, we do not compare size distributions. We note that the distribution of large intact IPs outside the RAN is of independent interest. Such patches would be possible foci for systematic methods to enhance the RAN by expanding existing RAs or designating new ones.

3.3.2 Landscape scale

Within strata, we generated two samples of landscapes by intersecting Enduring Features with RAN elements (i.e., RAN landscapes and non-RAN landscapes). EFs that crossed a RAN boundary were partitioned into two landscapes, which we assumed to be independent. We assess representation at the landscape scale by comparing the distributions of intrinsic landscape patch structure (ILPS) attributes between the two samples. The three components of an ILPS (composition, size structure and configuration) are not independent. The proportional area of a given IP class is determined by the size-structure of that class. Although the converse is not true, the number of patches, mean patch size and similar statistics are likely to be strongly correlated with proportional area (Cumming, Wang and Schmiegelow, unpublished data). Landscape configuration metrics (at least for mesic patch types) are also strongly correlated with patch size structures (Cumming and Vernier 2002). Many theoretical and empirical studies suggest that composition is probably the most informative single component of an ILPS. Therefore, compositions were the primary landscape attributes used in this study. The log shifted landscape area

$$lnA = \log A/5000,$$

where A is landscape area in ha, is also used in some analyses.

Neglecting disturbed, anthropogenic or unclassified patches, a landscape *composition* is a 5-dimensional vector of the proportional areas of five IP classes:

$$C_5 = (UPCON, MIX, BOG, RIP, WATER)^{9}.$$

We refer to these as vectors of crude proportions. These vectors sum to 1, which complicates their statistical analysis. Aitchison (1986) developed methods for analysis of compositional data. See Aebischer et al. (1993) for a more accessible introduction in the context of habitat use/availability studies and Cumming (2001) for compositional analysis of classified forest inventory data.

The logratio transformation

$$S_4 = (\log MIX/UPCON, \ldots, \log WATER/UPCON)$$

yields 4-dimensional data which can be analysed within the conventional framework of multivariate normal distribution theory. However, given the relatively small sample sizes of RAN landscapes, multivariate analysis of the full compositions was problematic. Accordingly, we assessed representation using the 3-dimensional *terrestrial subcompositions*

$$C_3 = (UPCON, MIX, BOG) / (UPCON + MIX + BOG).$$

[9] To minimize notation, the names UPCON, MIX, etc. are used to refer to specific IP classes and also as variables for the proportional area of that IP class within a landscape or other areal unit. The meaning should always be clear from context.

In the restricted landscape samples used in our analyses, these three IP classes account for a mean of 89% of total landscape area. The logratio transformation

$$S_2 = (\log MIX/UPCON, \log BOG/UPCON)$$

yields 2-dimensional data. We assume approximate bivariate normality: $S_2 \sim N(\mu,\Sigma)$. In this framework, a strict criterion of landscape-scale representation would be formulated as the hypothesis that the subcompositions of RAN landscapes are a random sample from the population of landscapes within the strata. However, it is not clear how to properly formulate or test this hypothesis. Instead, we regard the RAN and non-RAN landscapes (as derived from the enduring features) as two independent samples of possible natural landscapes that could be delineated within the strata. Under this model, the criterion for representation is that the two sets of landscapes were drawn from the same population. We denote the samples of subcompositions from RAN and non-RAN landscapes by L1 and L0, respectively, and assume that $L0 \sim N(\mu_0,\Sigma_0)$ and $L1 \sim N(\mu_1,\Sigma_1)$. Then, our representation criterion implies that L0 and L1 are identically distributed, which is expressed as the joint null hypothesis

$$H0: \mu_0=\mu_1 \wedge \Sigma_0=\Sigma_1$$

and tested by standard methods (Aitchison 1986). If H0 is rejected, we test for significant differences in the sample covariance structures

$$H1a: \Sigma_0=\Sigma_1$$

and the sample means

$$H1b: \mu_0=\mu_1$$

These tests indicate in what respects the RAN is non-representative. In addition, we developed graphical methods to further quantify the nature of non-representation and qualitatively assess its ecological significance.

3.3.3 Stratification (Appendix B)

In this analysis, strata are the spatial units within which representation is assessed. Spatial stratification is appropriate when the indicators used to set representation criteria differ significantly across space. However, such global representation does not guarantee adequate representation within strata. We evaluated alternate stratification schemes for landscape-scale representation of IP composition, as defined above. Informally, at this scale, stratification may be indicated if each strata has an abundance of some landscape with characteristics that are rare in other strata. We assessed the utility of a hierarchy of alternate stratification schemes: 1) ecozones within the study area; 2) RPUs within ecozones, and 3) ecoregions within RPUs.

At each level, we tested for differences in landscape compositions between strata, using the bivariate compositions and hypothesis tests presented above. However, statistically significant differences in the distributions does not necessarily justify a stratification level. The differences should be ecological significant, in that each strata should contain a unique subpopulation of landscape characteristics. We used a multivariate classification method, linear discriminant analysis (LDA) to test for ecologically relevant differences between strata at each level. LDA constructs linear combinations of covariates that attempt to classify a sample into prior groups such that the group means are as far apart as possible. We conducted two LDAs at each level. The first used the terrestrial subcompositions (S_2), landscape size (lnA) and interaction terms as covariates. The second used the full compositions (S_4) and landscape size (lnA), with no interaction terms. If neither set of landscape attributes could reliably classify landscapes to their strata, we considered there to be no need to stratify at that level.

This evaluation can be found in Appendix B.

3.3.4 Relation Between IPs and EFs (Appendix C)

Our landscapes are derived from mapped enduring features which were delineated based on four layers of the Soil Landscapes of Canada (SLC). If unique combinations of SLC attributes correspond to unique intrinsic landscape patch structures, then our use of IP compositions may be superfluous. To evaluate the relationship between IP composition and SLC attributes, we stratified EFs by attributes of the two most variable SLC layers. For strata with sufficient samples of EFs, we conducted pair-wise tests on the compositional distributions between strata, tested for a significant mean effect by MANOVA (Johnson and Wichern 1992), and tested for the ability of landscape composition and size to discriminate between these strata by linear discriminant analysis. If SLC attributes uniquely determine landscape composition, then: 1) there should exist significant differences in compositional distributions between SLC strata; and 2) IP composition and landscape size should correctly classify landscapes according to their strata.

This evaluation can be found in Appendix C.

3.4 Analytical evaluation of the adequacy of existing designated lands in Saskatchewan as ecological benchmarks using IPs

In this section, we report the results of the statistical analyses evaluating the adequacy of existing designated lands in Saskatchewan's Representative Areas Network (RAN) to serve as ecological benchmarks for forest management activities. We conducted separate analyses for RPU 10 and RPU 12, which are essentially the boreal plain and boreal shield ecozones within the Saskatchewan study area (Figure 3.5), and assessed the representation of patch size structure and composition at the patch and landscape scales.

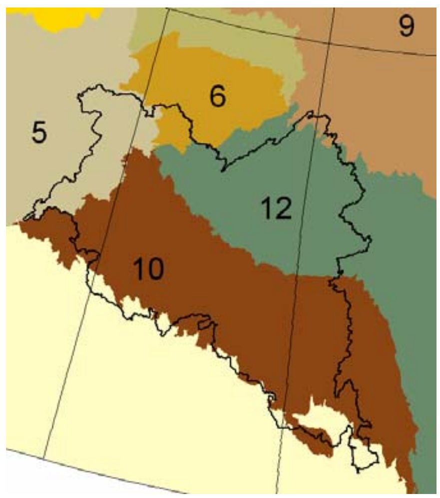


Figure 3.5: Regional Planning units in the study area.

Results: Regional Planning Unit 10

3.4.1 Patch Composition RPU 10

3.4.1.1 Intrinsic Patch Compositions – Patch Scale

The IP compositions of RAN and NONRAN are qualitatively compared using the relative total proportion of IP types in RAN and NONRAN. The total proportions were derived by summing across RAN and NONRAN landscapes (i.e., EFs). However, for this analysis, not all landscapes were included. In total, there were 150 landscapes in the RAN of total area[10] 884,076 ha. After applying our standard selection criteria (Appendix D), 32 RAN landscapes were retained for compositional analyses. The total area of the 32 RAN landscapes (778,982 ha) accounted for 88.1% of the entire RAN. There were 305 NONRAN landscapes with a total area of 9,876,400 ha. We retained 183 NONRAN landscapes for compositional analysis which accounted for 93.9% of the total NONRAN area. The mean sizes of RAN and NONRAN landscapes were 24,300 ha and 50,700 ha, respectively. NONRAN landscape units were significantly larger (t-test, p<0.001).

[10] The total area excludes unclassified and anthropogenic patches.

Based on the subset of landscapes retained for compositional analysis, the RAN contained higher proportions of UPCON and MIX and a lower proportion of BOG, relative to NONRAN (Table 3.2). This finding held true when all landscape units in the RAN (N=150) and NONRAN (N=305) were included in the analysis, except for UPCON which has similar proportions in RAN and NONRAN (Figure 3.6). Based on this qualitative comparison, the RAN has more MIX and less BOG than the NONRAN. For UPCON, the findings are not clear.

Table 3.2: Proportion of UPCON, MIX, and BOG within RAN and NONRAN in RPU 10 based on the subset of landscapes.

	N	UPCON	MIX	BOG
NON RAN	183	0.330	0.252	**0.418**
RAN	32	**0.403**	**0.347**	0.250

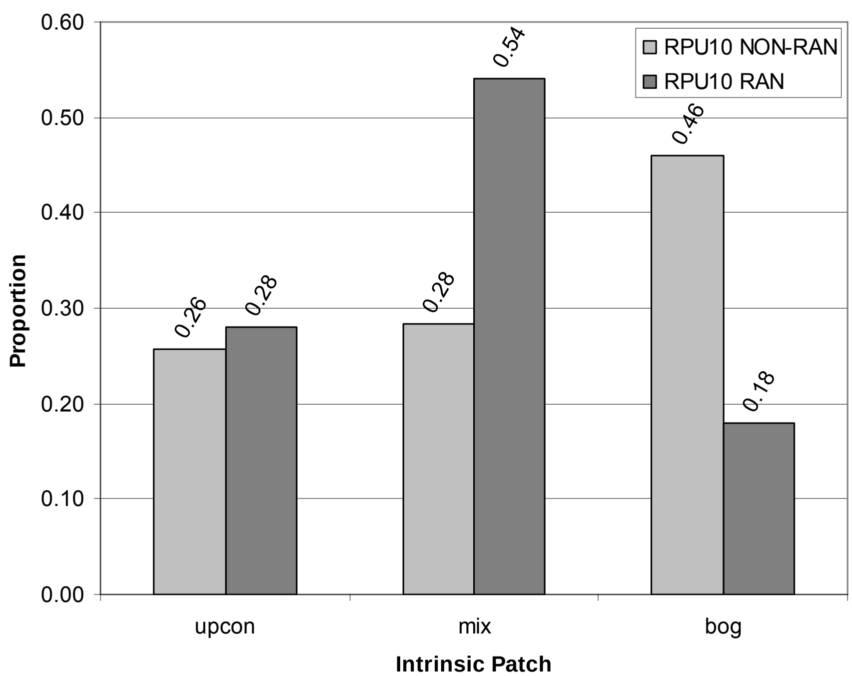


Figure 3.6: Proportion of UPCON, MIX, and BOG within RAN and NONRAN in RPU 10 based on the full set of landscapes.

3.4.1.2 Intrinsic Landscape Patch Composition – Landscape Scale

For this analysis, we derived multivariate distributions of IP terrestrial subcompositions (UPCON, MIX, and BOG) for RAN (N=32 landscapes) and NONRAN (N = 183 landscapes) at the landscape scale (Figure 3.7). For a tutorial on how to interpret figures like Figure 3.7, please refer to Appendix B, pages 68-69. The terrestrial subcompositions of RAN and NONRAN landscapes differ. The variance structures were the same, but the mean subcompositions for RAN (-0.150,-0.477) and NONRAN (-0.273,0.235) differed significantly (p=0.0035).

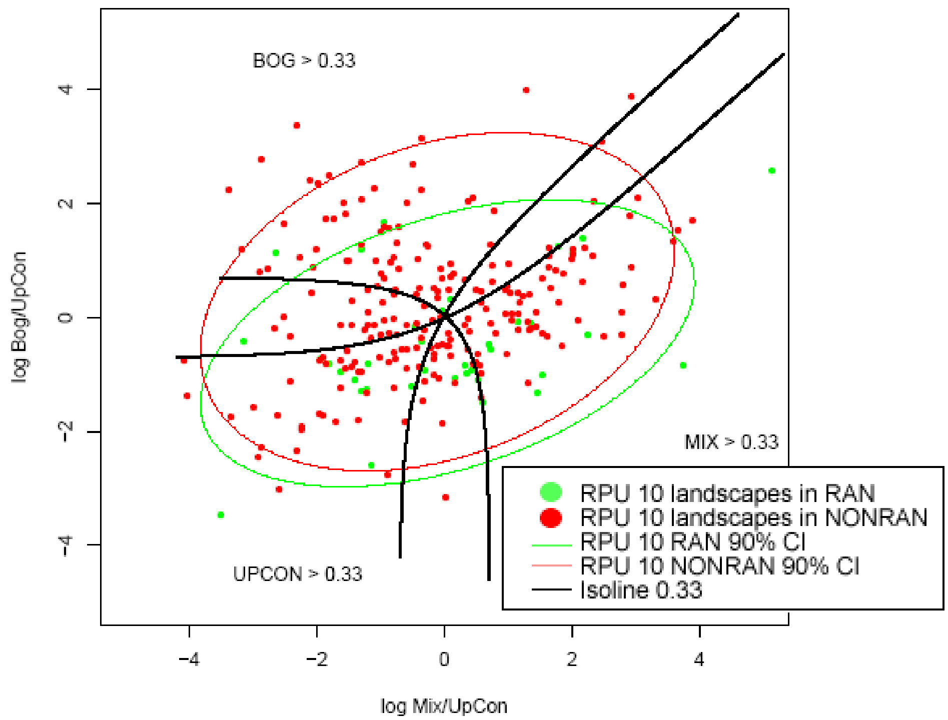


Figure 3.7: Terrestrial subcompositions for landscapes within RAN (N=32) and NONRAN (N=183) for RPU 10.

In Figure 3.7, there is a distinct shift in the RAN towards greater proportion of representation of MIX and UPCON. If the 0.33 isoline is used as a guide, the majority of RAN landscapes are greater than 33% MIX and UPCON. Interestingly, very few landscapes, in either RAN or NONRAN, are *jointly* greater than 33% MIX and BOG which indicates that this combination rarely occurs in RPU 10. Also of special note, very few RAN landscapes *jointly* have greater than 33% BOG and UPCON, while the NONRAN areas are quite abundant in these types of landscapes. Landscapes that are *jointly* high in MIX and UPCON are reasonably well represented in the RAN. However, landscapes with very high abundances of MIX > 70%, are less well represented in the RAN.

3.4.1.3 Landscape samples

For the intrinsic landscape patch compositional analysis, we use the 32 RAN and 183 NONRAN landscapes described earlier. However, because only 8/32 RAN landscapes were entirely contained within the RAN, we had to split the remaining 24 landscapes into RAN and NONRAN components, which could introduce bias. Although we could not perform the appropriate multivariate paired-Hotelling T-test, the compositional distributions of the two components did not appear to differ. Paired t-tests on the marginal distributions were not significant. We also compared the component log(MIX/UPCON) in RAN and NONRAN partitions by linear regression ($y = 0.27 + 0.97x$, $R^2=0.54$). The slope was not significantly different from 0, and the constant term did not differ from 0. We concluded that proportion of MIX, BOG and UPCON did not appear to differ amongst the split landscapes, and therefore, the splitting of landscapes across RAN boundaries will not introduce bias. Further, this leads us to conclude that the RAN boundaries were not delineated to capture specific UPCON, MIX or BOG configurations, at least at the landscape level (i.e., EF). RAN partitions were smaller (paired t-test (log area), $p<0.001$). The mean size of NONRAN partitions was 1,180 km^2 compared to 281 km^2 for RAN partitions (Figure 3.8).

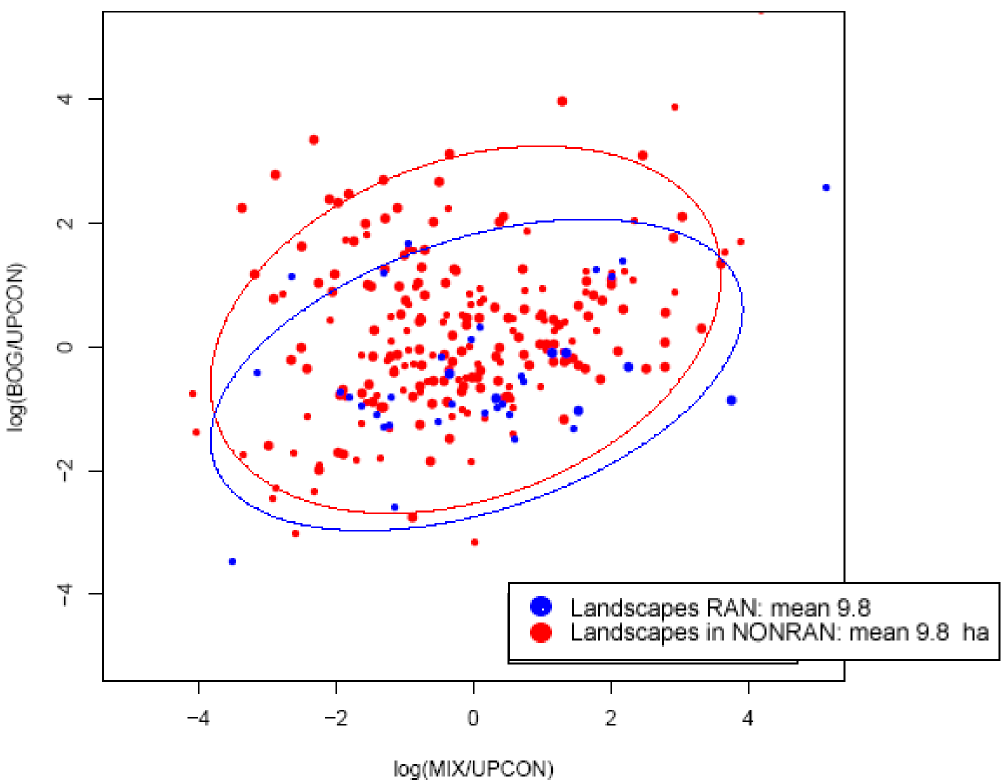


Figure 3.8: Terrestrial subcompositions for landscapes (i.e., EFs) within the study area stratified by RAN and NONRAN in RPU 10.

3.4.2 Intrinsic Patch Size Structure RPU 10

3.4.2.1 Intrinsic Patch Size – Patch Scale

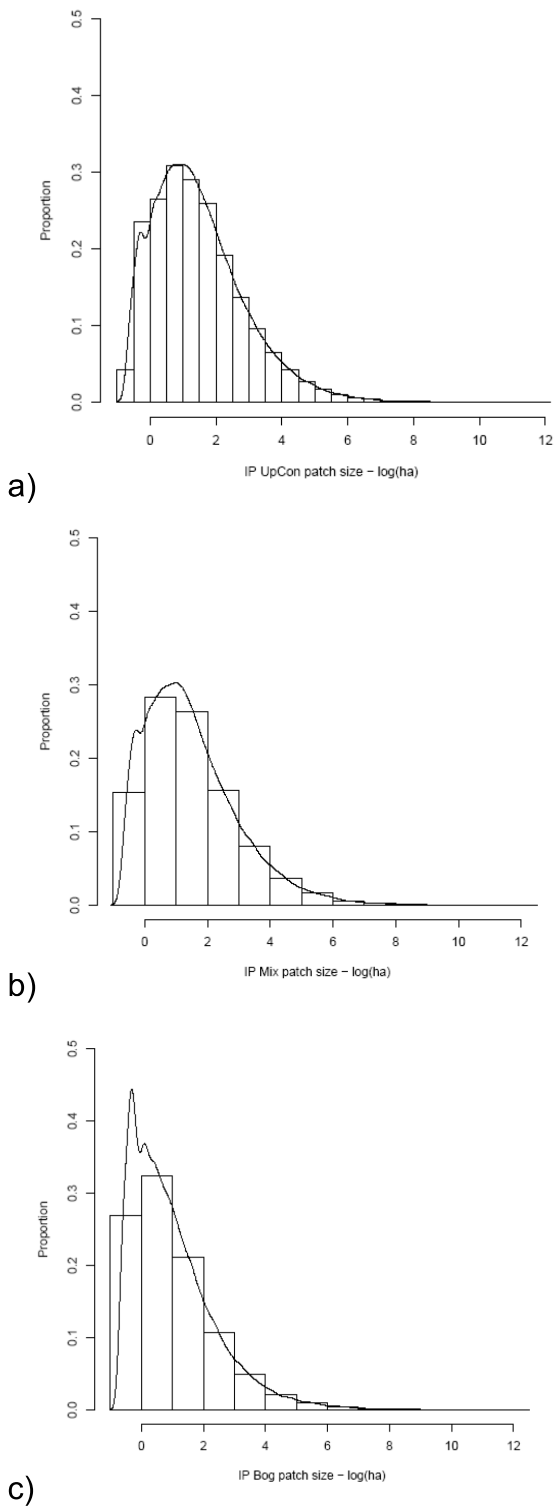
The distributions of intrinsic patch size structure for RAN and NONRAN were generated by pooling all patches that intersected the RAN and all patches that intersected NONRAN. The patch size (log area) distributions of the IP types UPCON, MIX, BOG, and RIPARIAN are significantly different (KS test, p-value < 0.0000) between RAN and NONRAN areas (Table 3.3). The mean patch sizes for UPCON, and MIX were larger in the RAN, while BOG and RIPARIAN were larger in the NONRAN (**Table 3.4**Table 3.4). Although the IPs are approximately log-normal, the histograms show that the distribution of all IP types are skewed to the right meaning that small IPs are more numerous than large ones (Figures 3.9 and 3.10). The interpretation is that the patch size distribution of the RAN is not a random sample from RPU 10. The RAN appears to under-represent larger patches of BOG and RIPARIAN and over-represent larger patches of UPCON and MIX.

Table 3.3: Kolmogorov-Smirnov test for differences in size distributions (log area) between RAN and NONRAN within RPU 10 for Intrinsic Patch types UPCON, MIX, BOG, and RIPARIAN.

IP Type	D statistic	p-value
UPCON	0.0295	< 0.0001
MIX	0.0357	< 0.0001
BOG	0.0457	< 0.0001
RIPARIAN	0.0504	< 0.0001

Table 3.4: Mean Intrinsic Patch size (ha) for NONRAN and RAN in RPU 10

	Mean Patch Size (ha)	
IP Type	NONRAN	RAN
UPCON	23.13	**27.80**
MIX	38.91	**73.62**
BOG	**43.39**	16.36
RIPARIAN	**9.39**	7.70



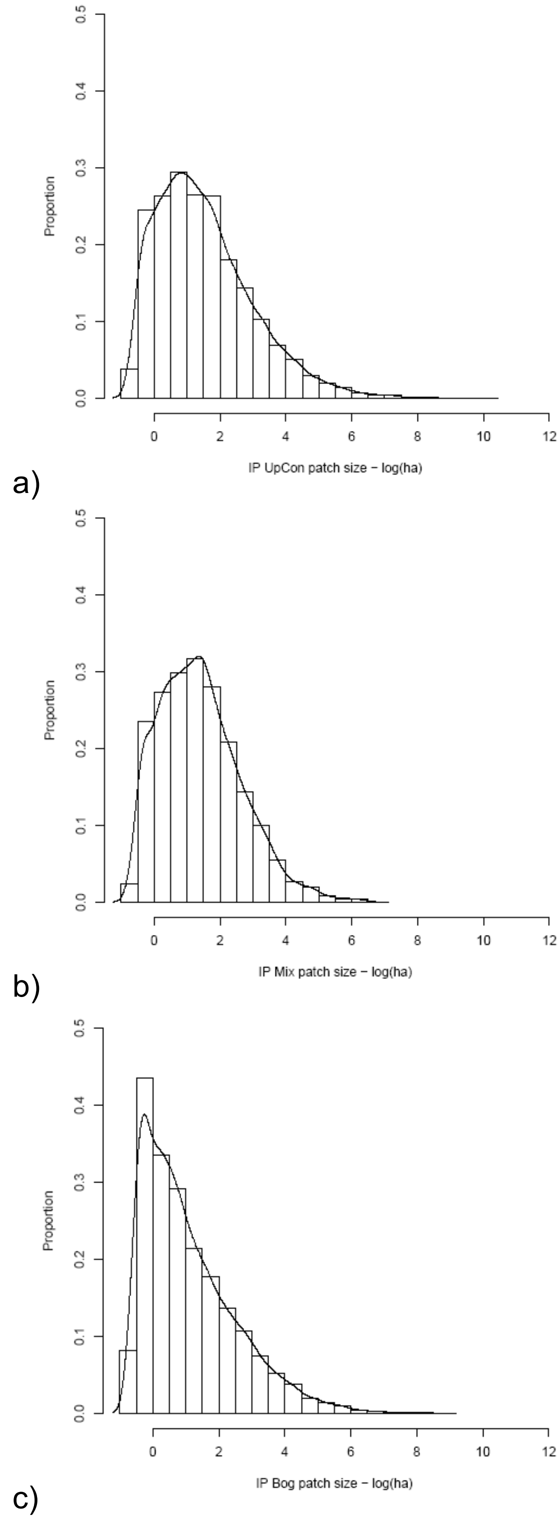


Figure 3.9: The **NONRAN** IP size distribution for a) UPCON, b) MIX and c) BOG within RPU 10.

Figure 3.10: The **RAN** IP size distribution for a) UPCON, b) MIX and c) BOG within RPU 10.

3.4.2.2 Representation of Large Intrinsic Patches

In order to formally evaluate the representation of large intrinsic patches within the RAN, we sought a parametric statistical model of their size distribution. Cumming and Vernier (2002) found that the log normal distribution was a useful model of the size distribution of mapped inventory stands within boreal mixedwood landscapes (~100km^2), where a small number of large patches contribute most of the total area for a given stand type. This pattern is typical of so-called "heavy-tailed" distributions, which have high frequencies of large values relative to, for example, the normal distribution. However, the log-normal model was not appropriate in this case, tending to markedly over-predict the frequency of very large patches; in a sense, the log-normal tail is too heavy. Preliminary investigations suggested that the log-gamma distribution was preferable. This distribution has been used to model physical processes such as peak water flows (Andrews et al. 2004) and has many applications in insurance risk analysis and actuarial problems where the frequency of extreme events is of critical importance. Identification of a correct statistical model of patch size is important for formal representation analysis, and as an objective criterion for future systematic reserve design efforts. Although as noted, the log-gamma appears preferable to the log-normal, the data suggest that a more complex model is required. We were unable to identify or apply such a model within the scope of the present study.

Table 3.5: Representation of large intrinsic patches (> 5 km^2) in RPU 10.

	NONRAN			RAN		
IP Type	nLP[1]	Area[2]	Mean[3]	nLP	Area	Mean
UPCON	660	12,597.23	**19.09**	70	1,179.18	16.85
MIX	620	20,060.69	32.36	84	3,096.48	**36.86**
BOG	741	41,696.12	**56.27**	41	545.78	13.31

[1] Total number of large intrinsic patches
[2] Total area of large intrinsic patches (km^2)
[3] Mean large patch size (km^2). The locations of the largest mean size (RAN or NONRAN) are indicated in bold face.

Informally, we compared the frequency, total area and mean size of large IP patches of class UPCON, MIX and BOG between the RAN and NONRAN for RPU 10 (Table 3.5). The proportion of total area and number of large UPCON and MIX patches within the RAN was about 10%, roughly commensurate with their abundance within RPU 10, and the mean sizes were approximately the same. However, there were comparatively few large BOG patches in the RAN (5.2%) accounting for less than 1.2% of the total area of large BOG IPs, and these patches were smaller on average compared to the NONRAN. For a qualitative, graphical analysis of representation, we use quantile-quantile plots to compare the empirical size distributions between the RAN and NONRAN areas for various IP types (Figure 3.1). These plot the sample IP sizes from the RAN against the approximated empirical quantiles from the NONRAN. Some variant of a log-gamma distribution was clearly indicated, so we use the scaled, shifted patch sizes $z=\log(x/500)$, where x is the large IP size in ha. If the sizes of the RAN and NONRAN patches describe a straight line with a slope of 1 and intercept 0, then we may assume that the two sizes are drawn from the same distribution, and the RAN may be considered representative with respect to the large patch sizes of a given IP class. The size distributions for UPCON and MIX IPs do approximate this straight line, although intermediate sizes of both classes are slightly more abundant, and the very largest patch sizes are slightly less abundant, than expected. Although these differences would likely be significant in a statistical sense, we conclude that the RAN is representative with respect to the distribution of large UPCON and MIX IPs. However, in the case of IP class BOG, patches larger than 800 ha (0.5 ≈ log 800/500) are markedly under-represented in the RAN, consistent with the relatively mean size of large BOG IPs within the RAN. Overall, we conclude that, in RPU 10, the abundance and size-structure of large MIX and UPCON IPs are well-represented within the RAN. Large BOG IPs are very poorly represented, under these criteria. Notably, the RAN contains only one BOG patch larger than 4000ha, although such patches are fairly abundant elsewhere in RPU 10.

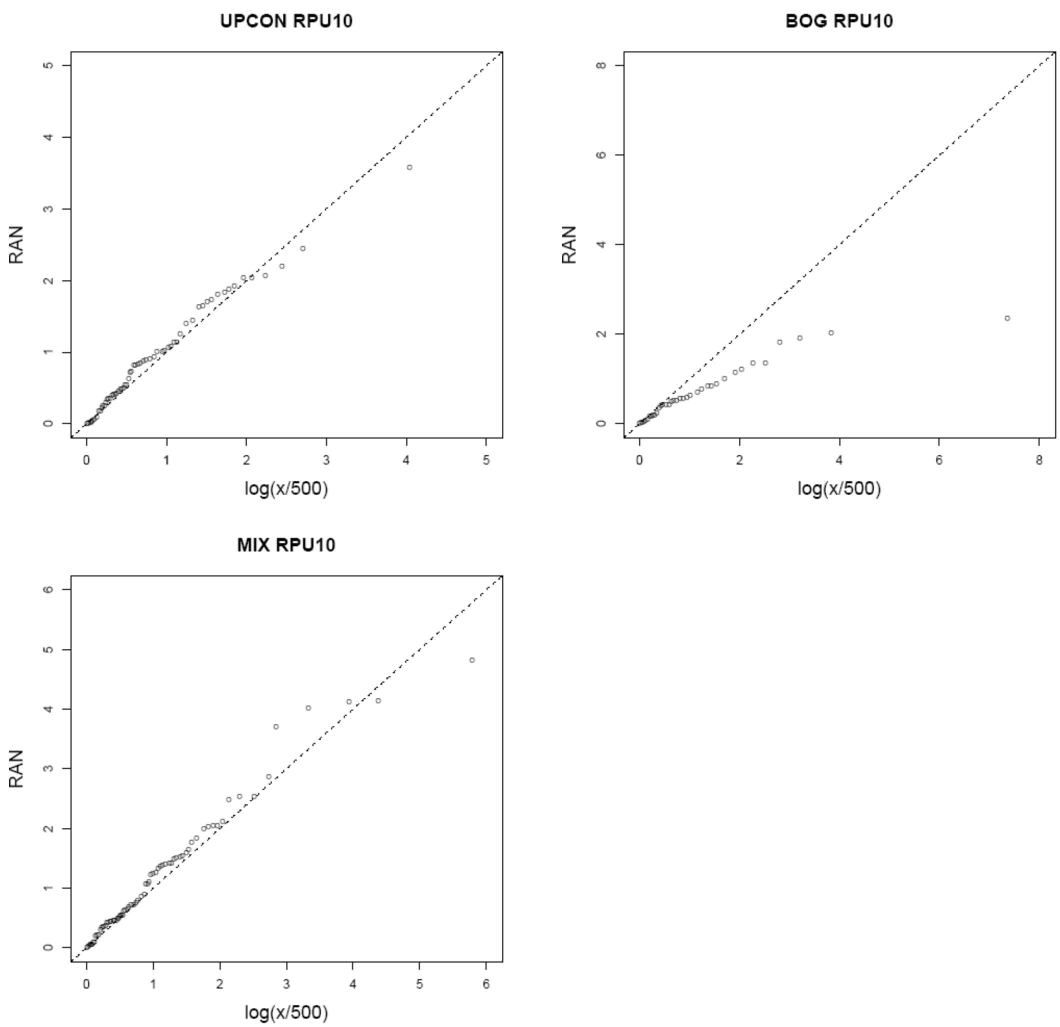


Figure 3.11: Quantile-quantile plots of log-shifted large IP sizes (>500ha) for IP classes UPCON, MIX, and BOG. The x-axes are the quantiles of the empirical NONRAN distributions, and the y-axes are the sample sizes from the RAN (further explanations are given in the text). The units of the x and y axes are the same, log(x/500), as indicated. However, the x axis is the empirical quantiles of LIP sizes in the NONRAN, whilst the y axis is the empirical distribution of actual LIP sizes in the RAN. The size distributions of large UPCON and MIX IPs are well-represented in the RAN, but the size distribution of large BOG IPs is very poorly represented.

3.4.2.3 Representation of Large Intact Patches

Based on the Global Forest Watch Canada Large Intact Landscape data layer, the proportion of 100% intact IPs > 5 km^2 (herein, Large Intact IPs) was greater in the RAN than in the RAN & NONRAN landscapes combined (Table 3.6). The mean size of Large Intact IPs of UPCON and BOG are larger in NONRAN areas, while the MIX is considerably larger in the RAN (Table 3.7). For MIX, we propose two possible interpretations of this result: 1) the RAN are over-representing large intact MIX IP types or, 2) large MIX IP types have already been fragmented in NONRAN landscape, thus the RAN may be representing what *was* there prior to industrial activity. Interestingly, although some logging activity must occur in UPCON in NONRAN areas, the mean patch size is larger in NONRAN. For Large Intact BOG IPs in the RAN, the mean size is smaller. This is likely related to the bias of the RAN towards low representation of Bog IP types.

Table 3.6: The proportion of large patches that are intact.

IP Type	NONRAN & RAN	RAN
UPCON	0.29	0.37
MIX	0.07	0.18
BOG	0.15	0.41

Table 3.7: Representation of large intact intrinsic patches (> 5 km^2) in RPU 10

	NONRAN			RAN		
IP Type	nBP[1]	Area[2]	Mean[3]	nBP	Area	Mean
UPCON	168	2332.41	**13.88**	26	286.83	11.03
MIX	27	234.00	8.67	15	289.85	**19.32**
BOG	96	1849.21	**19.26**	17	223.36	13.14

[1] nBP is total number of big patches
[2] Area is their total area (km^2)
[3] Mean is the mean big patch size (km^2)

Results: Regional Planning Unit 12

3.4.3 Patch Composition RPU 12

3.4.3.1 Intrinsic Patch Compositions – Patch Scale

The IP compositions of RAN and NONRAN are qualitatively compared using the relative total proportion of IP types in RAN and NONRAN. The total proportions were derived by summing across RAN and NONRAN landscapes (i.e., EFs). However, for this analysis, not all landscapes were included. There were 93 landscapes in the RAN of total area 185,353 ha. Only 8 were retained for compositional analysis, accounting for 94.6% of the total area (175,268 ha). There were 43 NONRAN landscapes retained for the comparison. The mean sizes of the RAN and NONRAN landscapes were 21,910 ha and 46,420 ha, respectively. NONRAN landscapes were significantly larger than RAN landscapes (t-test, $p<0.046$). Although small sample sizes limit the interpretability of these results, landscapes sampled by the RAN in RPU 12 do not appear to be significantly smaller than landscapes outside the RAN. The size distributions of EFs could not be distinguished (KS-test, $p=0.87$).

Overall, the RAN had significantly more MIX and less BOG than the NONRAN landscape (Table 3.8). The proportion of UPCON, MIX and BOG calculated from landscapes selected for compositional analysis agrees with the trend in composition using all the IPs occurring in the RAN and NONRAN landscapes (Figure 3.12).

Table 3.8: Proportion of UPCON, MIX and BOG within RAN and NONRAN landscapes in RPU 12.

	N	UPCON	MIX	BOG
NONRAN	43	0.473	0.181	**0.345**
RAN	8	**0.509**	**0.293**	0.197

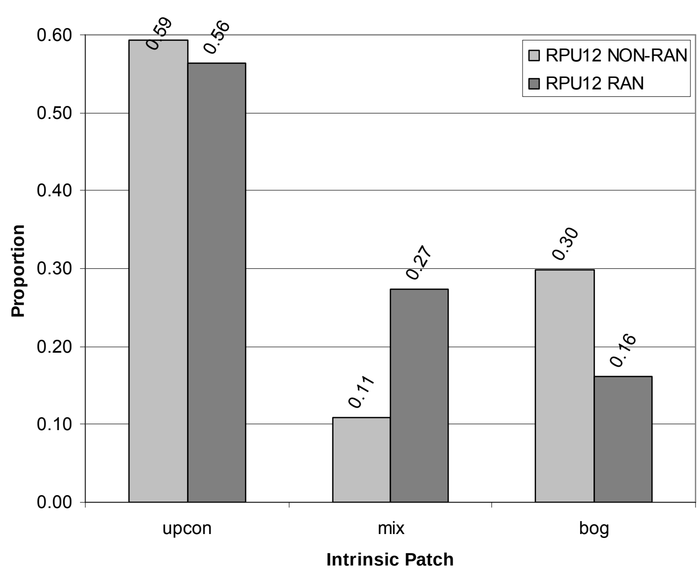


Figure 3.12 Proportion of Intrinsic Patch types within the study region stratified by RPU 12 a) Upcon, Mix, and Bog

3.4.3.2 Intrinsic Landscape Patch Composition – Landscape Scale

The multivariate distribution of landscape IP terrestrial subcompositions for RAN and NONRAN are illustrated in Figure 3.13. The terrestrial subcompositions of RAN and NONRAN landscapes differ. The variance structures were significantly different (p = 0.0065). The means for NONRAN (-0.959,-0.315) and RAN (-0.552,-0.949) were also significantly different (p = 0.002). In the RAN, the variance is smaller and the mean is shifted down and to the right towards over-representation of MIX IP. Although there appears to be more MIX > 33% in the RAN, it is likely an artefact of the lack of representation of the BOG. The RAN landscapes are almost entirely > 33% UPCON. Representation of landscapes *jointly* > 33% BOG and UPCON are lacking in the RAN. However, landscapes that are *jointly* > 33% in MIX and UPCON are well presented.

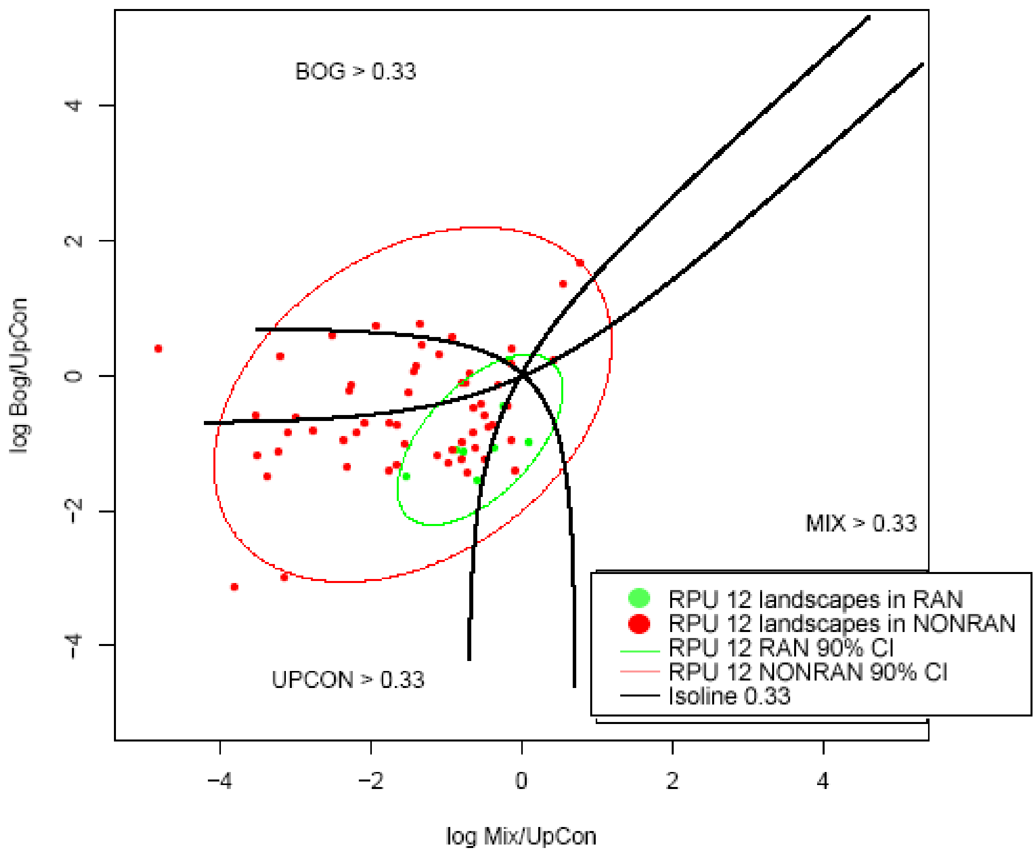


Figure 3.13: Terrestrial subcompositions for landscapes within RAN (N=8) and NONRAN (N=43) for RPU 12.

3.4.4 IP Size Structure RPU 12

3.4.4.1 Intrinsic Patch Size – Patch Scale

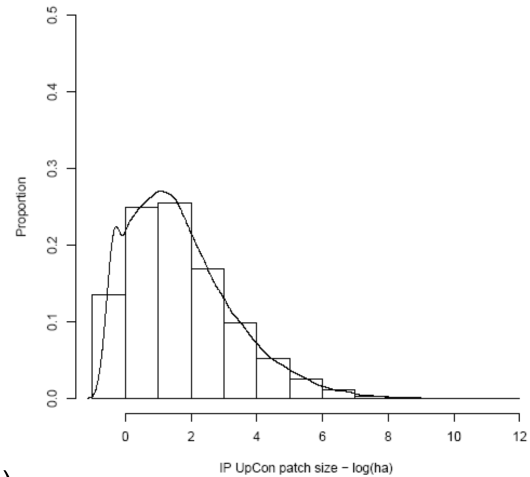
The distributions of intrinsic patch size structure for RAN and NONRAN were generated by pooling all patches that intersected the RAN and all patches that intersected NONRAN. The patch size (log-area) distribution of IPs UPCON, MIX, BOG and RIPARIAN were significantly different (K-S test, p-value < 0.0000) between RAN and NONRAN IPs (Table 3.9). Mean patch size for all patch types was smaller in the RAN (Table 3.10). Like RPU 10, the histograms show that the size distributions of all IP types are skewed to the right as the overall mean patch size decreases (Figures 3.14 and 3.15).

Table 3.9: Kolmogorov-Smirnov test for differences in size distributions (log area) between RAN and NONRAN within RPU12 for IP types UPCON, MIX, BOG, and RIPARIAN.

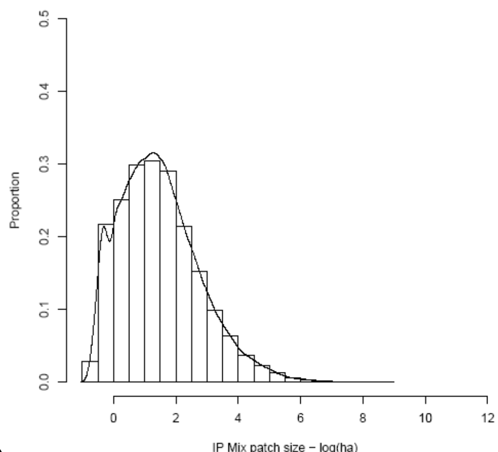
IP Type	D statistic	p-value
UPCON	0.1499	< 0.0001
MIX	0.0318	0.0007
BOG	0.0989	< 0.0001
RIPARIAN	0.0885	< 0.0001

Table 3.10: Mean Intrinsic Patch size (ha) for RPU 10, in NONRAN and RAN

	MEAN IP Size (ha)	
IP Type	NONRAN	RAN
UPCON	**46.20**	26.82
MIX	**12.72**	11.08
BOG	**14.12**	5.48
RIPARIAN	**4.99**	3.00



a)



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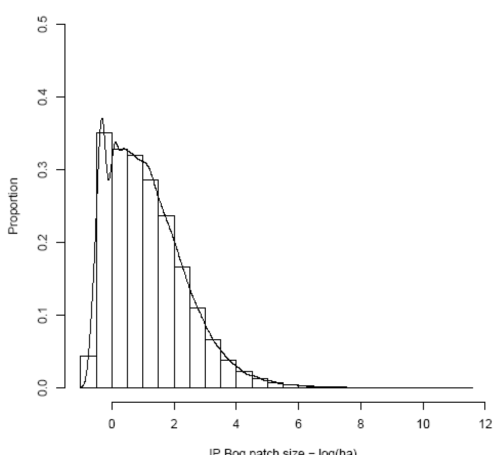

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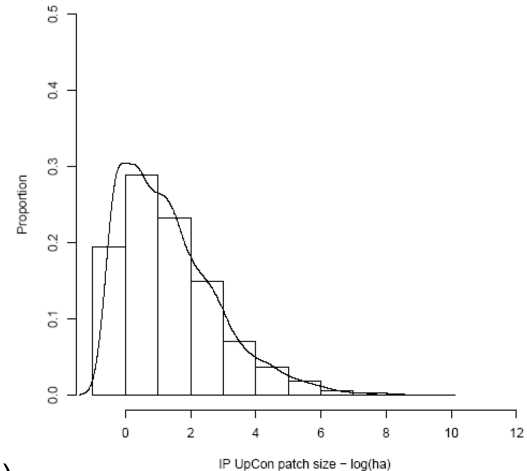
Figure 3.14: The **NONRAN** IP size distribution for a) UPCON, b) MIX and c) BOG within RPU 12.



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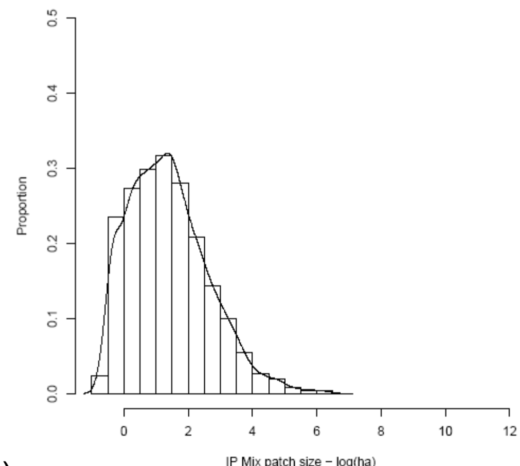

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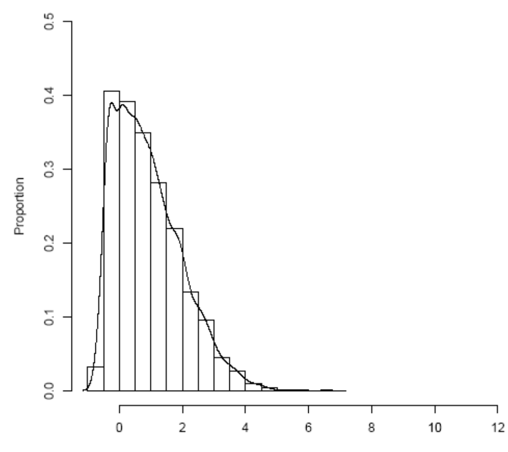

c)

Figure 3.15: The **RAN** IP size distribution for a) UPCON, b) MIX and c) BOG within RPU 12.

3.4.4.2 Representation of Large Intrinsic Patches

For large intrinsic patches, regardless of intactness, UPCON, MIX, and BOG mean patch size was larger in the NONRAN (Table 3.11). The distribution of large UPCON patches in the RAN is similar to the NONRAN (Figure 3.16). There is deviance towards lower representation of UPCON of extremely large UPCON patches. It is very difficult to assess representation of MIX and BOG types as the number of patches actually sampled was very low. MIX patches represented in the RAN are all much smaller than MIX patches in the NONRAN (Figure 3.16).

Table 3.11: Representation of large intrinsic patches (> 5km^2) in RPU 12

	NONRAN			**RAN**		
IP Type	nBP[1]	Area[2]	Mean[3]	nBP	Area	Mean
UPCON	599	13534.61	**22.60**	26	464.85	17.88
MIX	46	409.03	**8.89**	5	28.36	5.67
BOG	180	4669.36	**25.94**	2	14.41	7.21

[1] nBP is total number of big patches
[2] Area is their total area (km^2)
[3] Mean is the mean big patch size (km^2)

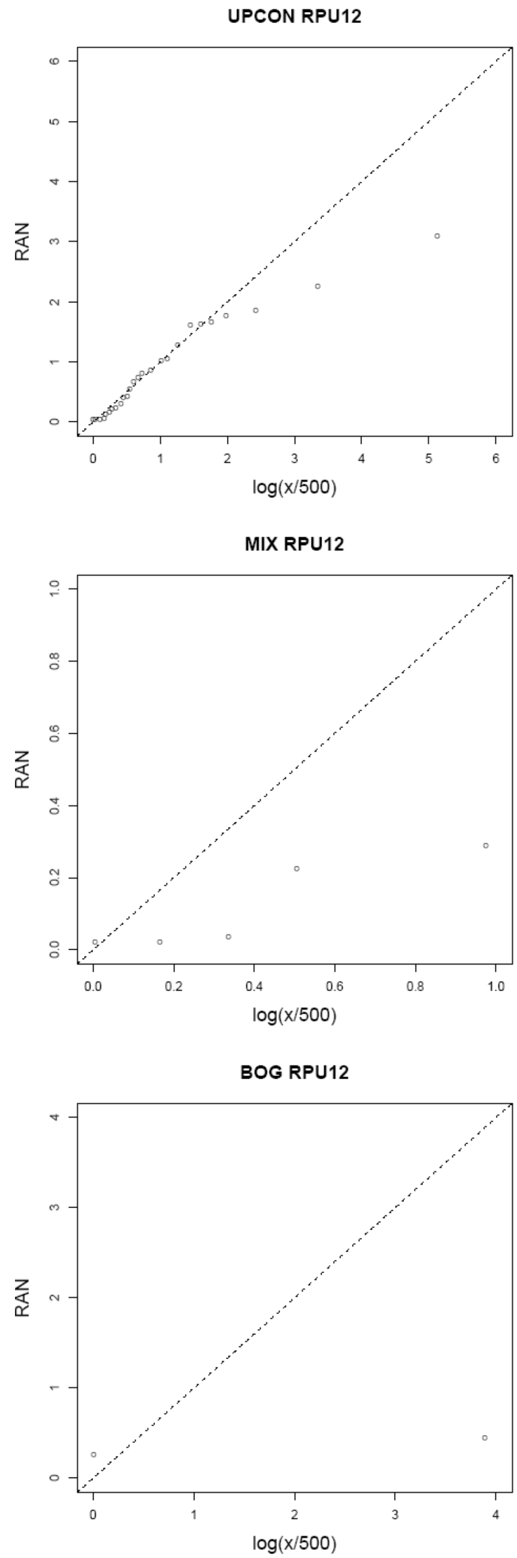

Figure 3.16: Expected distribution of the RAN IPs against the actual distribution (log-gamma) for IPs > 500-ha.

3.4.4.3 Representation of Large Intact Patches

Proportions of Large Intact UPCON and BOG were greater in the RAN than the RAN & NONRAN landscapes combined, and MIX which was lower in the RAN (Table 3.12). Overall, Large Intact Patches of UPCON and MIX in the RAN and NONRAN are similar in size (Table 3.13). However, the number of Large Intact Patches is very small. Likely, in the RPU 12 where IP patches are much smaller on average than in RPU 10, large intact patches > 5 km^2 may be unrealistic. RPU 12 is dotted with lakes and rivers that tend to break up an otherwise contiguous landscape of UPCON and BOG IP types.

Table 3.12: Proportion of large patches that are intact

IP Type	NONRAN & RAN	RAN
UPCON	0.24	0.46
MIX	0.59	0.40
BOG	0.19	1.00

Table 3.13: Representation of large intact intrinsic patches (> 5 km^2) in RPU 12

	NONRAN			RAN		
IP	nBP[1]	Area[2]	Mean[3]	nBP	Area	Mean
UPCON	129	2926.56	**22.69**	12	252.14	21.01
MIX	25	187.97	**7.52**	2	11.38	5.69
BOG	33	762.49	**23.11**	2	14.41	7.21

[1] nBP is total number of big patches
[2] Area is their total area (km^2)
[3] Mean is the mean big patch size (km^2)

3.6 Influence of Large Parks on Compositional Analysis

The number of landscapes (i.e., EFs) contained in the RAN was much smaller than in the NONRAN area (Table 3.14). Furthermore, several designated lands in the RAN such as Prince Albert National Park are comparatively more uniform in some IP types. We tested the hypothesis that large individual designated lands may drive some of the log-ratio compositional results. Figure 3.17 shows the spatial location of four large designated lands in RPU 10: 1) Prince Albert National Park, 2) Meadow Lake, 3) McCusker Lake and, 4) Narrow Hill. There is sufficient variation in IP composition within landscapes belonging to the same park such that no one park or group of parks is influencing the mean or covariance structure.

Table 3.14: The number of landscapes (i.e., EFs) within the larger designated lands of the RAN within RPUs 10 and 12.

RPU	Designated Land	Number of landscapes	Area (km^2)
RPU10	Prince Albert National Park	4	3818.1
RPU10	McCusker Lake	5	1024.8
RPU10	Meadow Lake	6	933.9
RPU10	Wapawekka Hills	3	537.2
RPU10	Narrow Hills	3	287.0
RPU10	Whiteswan Lakes	2	267.9
RPU10	Greenwater Lake	1	204.6
RPU10	Wildcat Hill	2	191.4
RPU10	Seager Wheeler Lake	2	151.9
RPU10	Candle Lake	1	103.6
RPU10	Caribou Flats	1	101.0
RPU10	Budd Lake	1	100.3
RPU10	Clearence-Steepbank Lakes	1	68.2
RPU12	Lac LaRonge	6	1485.3
RPU12	Jan Lake	1	157.5
RPU12	Mari Lake	1	109.9

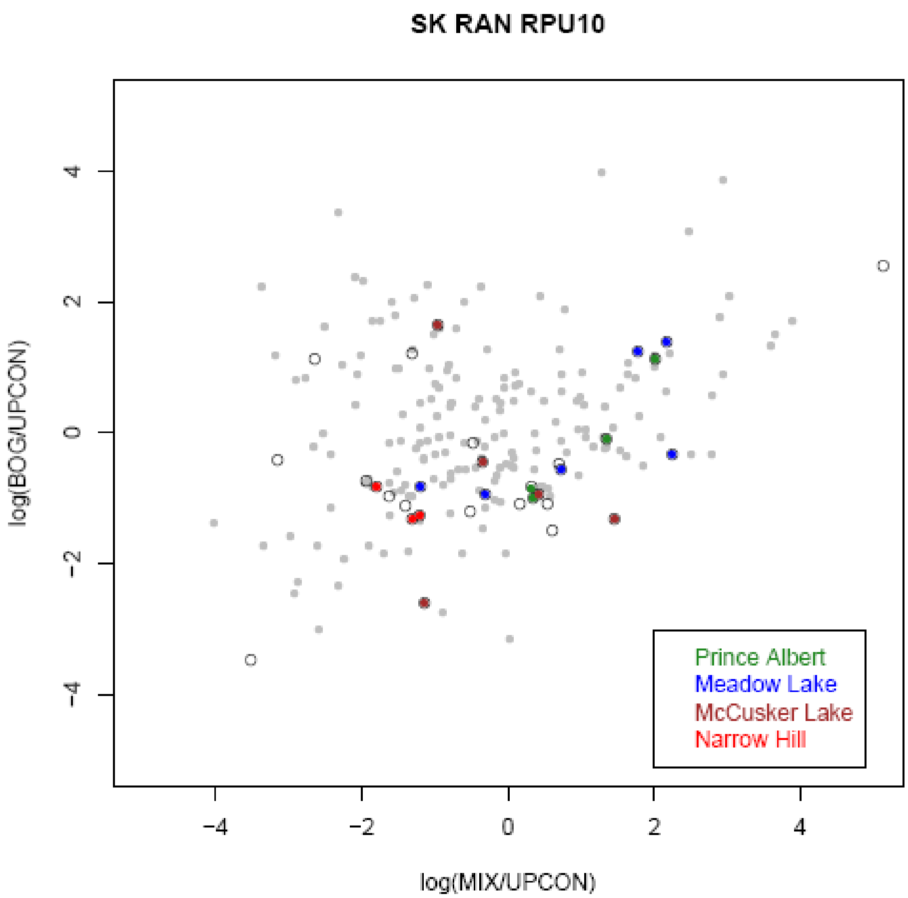

Figure 3.17: Parks (mean area = km^2) sampled in RPU 10.

3.7 Summary

Representation was assessed at the patch and landscape scale using metrics of IP composition and IP size. At the landscape scale, we evaluated representation based on the intrinsic landscape patch structure attribute of IP composition. For this evaluation, landscape units were defined by enduring feature boundaries. The representation of size structure of IPs was evaluated at the patch scale and was based on 1) the size distribution of IPs, 2) distribution of large IPs, and 3) the distribution of large intact IPs.

<u>IP Composition</u>

RPU 10 - At the patch scale, the RAN over-represents MIX and under-represents BOG. The findings for UPCON are not clear. At the landscape scale, we found that the terrestrial IP subcomposition (UPCON, MIX, and BOG) of RAN and NONRAN are different. The RAN adequately represents landscapes jointly high in MIX and UPCON, but poorly represents landscapes with high abundances of MIX (>70%) and landscapes jointly >33% BOG and UPCON.

RPU 12 – At the patch scale, the RAN over-represents MIX and under-represents BOG. The findings for UPCON are not clear. At the landscape scale, we found that the terrestrial IP subcomposition (UPCON, MIX, and BOG) of RAN and NONRAN are different. The RAN landscapes are almost entirely > 33% UPCON. Representation of landscapes *jointly* > 33% BOG and UPCON are lacking in the RAN. However, landscapes that are *jointly* > 33% in MIX and UPCON are well presented.

IP Size-Structure

RPU 10 - In RPU 10, the size distribution of IPs for UPCON, MIX, and BOG, are characterized by many small IPs with large patches being rare on the landscape. The patch size distribution of the RAN and NONRAN are different. The RAN under-represents larger patches of BOG and RIPARIAN and over-represents larger patches of UPCON and MIX. When we examined the large IPs (patches > 5 km^2) more closely, we found that the distribution of large patches of UPCON and MIX in the RAN and NONRAN are similar with a deviance towards lower representation of the extremely large patches in the RAN, and that large patches of BOG are under-represented in the RAN. When intactness is considered, large intact patches of UPCON and BOG are under-represented in the RAN, while large intact patches of MIX are over-represented. For MIX, we proposed two possible interpretations: 1) the RAN are over-representing large intact MIX IP types or, 2) large MIX IP types have already been fragmented in NONRAN landscape, thus the RAN may be representing what *was* there prior to industrial activity.

RPU 12 – In RPU 12, not unlike RPU 10, the size distribution of IPs for UPCON, MIX, and BOG, are characterized by many small IPs with large patches being rare on the landscape. The patch size distribution of the RAN and NONRAN are different. The RAN under-represents large patch sizes for all patch types (UPCON, MIX, BOG, and RIPARIAN). When we examined the distribution of large IPs (patches > 5 km^2) more closely, we found that the mean patch size for UPCON, MIX, and BOG is larger in the NONRAN. In other words, the RAN under-represents the larger patch sizes of UPCON, MIX, and BOG. The distribution of large UPCON patches in the RAN is similar to the NONRAN with a deviance towards lower representation of extremely large UPCON patches. It was difficult to assess representation of MIX and BOG types as the number of patches actually sampled was very low. However, MIX patches represented in the RAN are all much smaller than MIX patches in the NONRAN. When intactness is considered, the proportion of large intact UPCON and BOG is greater in the RAN, and MIX is lower in the RAN. Overall, large intact patches of UPCON and MIX in the RAN and NONRAN are similar in size. However, the number of large intact patches is very small. Likely, in the RPU 12 where IP patches are much smaller on average than in RPU 10, large intact patches > 5 km^2 may be unrealistic. RPU 12 is dotted with lakes and rivers that tend to break up an otherwise contiguous landscape of UPCON and BOG IP types.

Chapter 4: Ecological Benchmarks for Woodland Caribou

Special Note: This chapter should be interpreted as proof-of-concept only with regard to the application of the methods developed in this report. In this analysis, we defined critical caribou habitat based on Arsenault (2003). However, since Arsenault (2003), the definition of critical caribou habitat in Saskatchewan has been expanded. As a result, our analysis under-represents total caribou critical habitat which would affect our assessment of the representativeness of critical caribou habitat in the RAN. In addition, Al Arsenault (Provincial Wildlife Population Biologist, Saskatchewan Government) has recommended that the representation analysis of the RAN, with respect to caribou, be expanded to include: (1) upland-conifer stand age, which is critical in terms of lichen biomass, and (2) forest stands preferred by caribou. In RPU 10 (Boreal Plains), preferred habitat consists of larger, older (>50 yrs) conifer stands, and conifer-dominated stands, particularly jack pine dominated, especially if proximate to larger treed peatland complexes. In RPU 12 (Boreal Shield), preferred habitat consists of mosaics of jack pine ridges interspersed with peatlands as well as large expanses of conifer-dominated old-growth uplands.

4.0 Introduction

Woodland caribou ranges have been characterised as mosaics of wetland and upland conifer patches (Arsenault 2003). The scale and structure of caribou habitat mosaics has not been quantified to our knowledge, but landscapes with more than 2/3 by area of BOG, UPCON or BOG/UPCON mixtures are not abundant within the RAN. In this section, we evaluate the adequacy of the RAN to act as benchmarks for the effects of forest management on woodland caribou.

We first examine the landscape-scale composition of areas of the critical caribou habitat (CCH) areas delineated by Arsenault (2003; Fig. 2). Landscape samples were generated by intersecting the enduring feature (EF) and CCH coverages at IP resolution, where each IP was assigned to an EF and a CCH independently by majority area rules. We compared CCH and non-CCH landscapes within and between RPU10 and RPU12 by multivariate tests on the terrestrial subcompositions (S_2), and by graphical analysis. We also tested for significant between-CCH compositional variation by MANOVA. Finally, to evaluate the present representation of CCH areas within the RAN, we generated CCH landscapes as sampled by the RAN by intersecting the EF, CCH and RAN coverages at IP resolution.

4.1 CCH landscapes

The study area intersected 20 CCHs. Their total area (as estimated after the intersection procedure) was 32,029 km^2. Individual CCHs ranged in area from 66.1 to 5,391.2 km^2 with a mean of 1,601.5 km^2. The mapped CCHs contained or intersected 183 unique EFs, yielding 224 CCH landscapes. There were 33/183 (18%) EFs that intersected more than one mapped CCH (27 EFs intersected 2 CCHs, 4 intersected 3 CCHs, and 2 intersected 4 CCHs). Only 19/183 EFs (10.4%) were entirely contained within a CCH. Hence, CCH and EF boundaries are not congruent.

Our standard landscape-filtering step retained 101 CCH landscapes representing 18/20 mapped CCHs and 28,697.0 km^2 (89.6%) of the initial total CCH area. The mean and median areas of CCH landscapes were 284.1 and 175.6 km^2, respectively. The 208 non-CCH landscapes had mean and median areas of 445.2 and 240.2 km^2, respectively. The non-CCH landscapes were significantly larger (t-test, p=0.012). The 101 CCH landscapes represented 92 unique EFs. Of these, 58 (63%) were also represented in the non-CCH landscapes. That is, 58/92 EFs were partitioned into CCH and non-CCH landscapes (63%). The mean and median areas of the 58 CCH partitions were 323.5 and 186.9 km^2, respectively. The mean and median areas of the 58 non-CCH partitions were 950.1 and 335.4 km^2, respectively. The non-CCH partitions were significantly larger (paired t-test, p=0.002). In addition, the logratio log(MIX/UPCON) differed between partitions (paired t-test, p=0.005): the ratio of MIX to UPCON IP areas outside the CCHs was 1.5 times higher on average than within the CCHs. We conclude that the delineation of CCH areas was independent of the prior mapped EF boundaries. The CCH boundaries may reflect spatial variation in forest composition within EFs.

4.2 Compositional comparisons of CCH and non-CCH landscapes

4.2.1 RPU 10

RPU 10 included 18 mapped CCH areas, of which 15 were entirely contained within the RPU. There were 78 CCH landscapes from 69 unique EFs, with mean and median sizes of 297.8 and 185.1 km^2, respectively. The individual CCHs were represented by 1 to 11 landscapes, 12/18 CCHs were represented by at least 3 landscapes, and the top 6 CCHs accounted for 47/78 landscapes (59%). There were 165 non-CCH landscapes, each (by construction) from a unique EF.

The terrestrial subcompositions S_2 of the two landscape samples were not identically distributed ($p<0.001$; Figure 4.1). The covariance structures did not differ (p=0.172), but the means were not the same ($p<0.001$). The median non-CCH landscape is a nearly even mixture of the three IP classes (Table 4.1). The median CCH landscape is dominated by BOG and UPCON, with only 10% MIX.

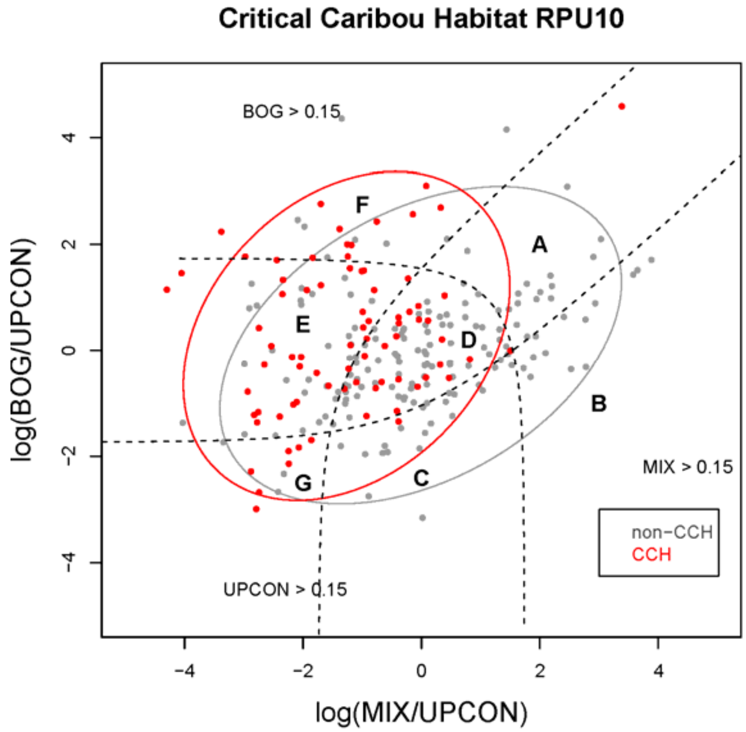

Figure 4.1: Logratio subcompositions of CCH landscapes (red) and non-CCH landscapes (grey) within RPU 10. Approximate 90% ellipsoids are shown in corresponding colours. The seven regions of subcomposition space delineated by the 0.15 isolines are described in Table 4.2.

Table 4.1: Mean landscape subcompositions and median crude proportions for non-CCH and CCH landscapes within RPU 10. The means correspond to the centroids of the ellipses in Figure 4.1.

		Median crude proportions		
	mean(S_2)	UPCON	MIX	BOG
Non CCH	(-0.013, 0.096)	0.324	0.320	0.356
CCH	(-1.265, 0.268)	0.386	0.109	0.505

To more precisely contrast the types of landscape within and outside mapped CCHs (Figure 4.1), we used the 0.15 compositional isolines to define 7 "regions" or groups of landscape compositions (Table 4.2). Of 13 landscapes dominated by MIX IPs (region B), none occur within mapped CCHs. Similarly, the CCHs contain essentially no landscapes dominated by MIX/BOG or MIX/UPCON mixtures (regions B and C), although both are well-represented within the RPU. Most (7/10) homogeneous UPCON landscapes (region G) are found within the CCHs, as are 12/19 homogeneous BOG landscapes (region F). The vast majority of CCH landscapes fall within regions D and E. Region E landscapes are UPCON/BOG mixtures with less than 0.15 of MIX.

Landscapes in region D contain at least a moderate proportion of all three IP types, and several contain more than 1/3 of MIX.

Table 4.2: The boundaries of 7 regions of Figure 4.1 as determined by the 0.15 isolines, and the key habitat characteristics of landscapes within each region in terms of the proportional areas of IP classes UPCON, MIX and BOG (the UPCON=0.2 isoline gives slightly better discrimination)

Region	Isoline values			Landscape characteristics
	UPCON	MIX	BOG	
A	<0.15	>0.15	>0.15	MIX+BOG>0.85
B	<0.15	>0.15	<0.15	MIX>0.7
C	>0.15	>0.15	<0.15	MIX+UPCON>0.85
D	>0.15	>0.15	>0.15	maximum heterogeneity
E	>0.15	<0.15	>0.15	UPCON+BOG>0.85
F	<0.15	<0.15	>0.15	BOG>0.7
G	>0.15	<0.15	<0.15	UPCON>0.7

In summary, the predominant terrestrial IP classes within mapped CCHs are UPCON and BOG. However, at the landscape scale, the composition of these habitat areas is highly variable. They contain some landscapes of nearly homogenous UPCON and BOG and many landscapes with relatively high proportions of MIX. We also note that RPU 10 contains many landscapes in regions E and (especially) D that appear to be potentially suitable components of caribou habitat, but which are not included in the mapped CCHs. This may be related to the spatial context of particular landscape units, or to other attributes not considered in this analysis.

4.2.2 RPU 12

RPU 12 included portions of 3 mapped CCH areas, all of which overlapped with RPU 10. There were 23 CCH landscapes from 23 unique EFs, with mean and median areas of 297.8 and 185.1 km^2, respectively. There were 38 non-CCH landscapes.

The terrestrial subcompositions S_2 of the non-CCH and CCH landscapes were not identically distributed (p=0.020), but the magnitude of the differences is much less than in RPU 10. CCH landscapes tended to have slightly higher proportions of UPCON and slightly less MIX, relative to non-CCH landscapes (Table 4.3), but the mean compositions did not differ (p=0.318). Their covariance structures were significantly different (p=0.013). CCH landscapes exhibit greater variation along the log(MIX/UPCON) axis (Figure 4.2). This may be due to a few outliers, possibly representing incompletely sampled or partitioned EFs. Overall, the sub-compositional differences between CCH and non-CCH landscapes within RPU 12 appear to be marginal.

Table 4.3: Mean landscape subcompositions and median crude proportions for non-CCH and CCH landscapes within RPU 12. The means correspond to the centroids of the ellipses in Figure 4.2.

		Median crude proportions		
	mean(S_2)	UPCON	MIX	BOG
Non CHH	(-0.829, -0.370)	0.470	0.205	0.325
CCH	(-1.209, -0.545)	0.532	0.159	0.307

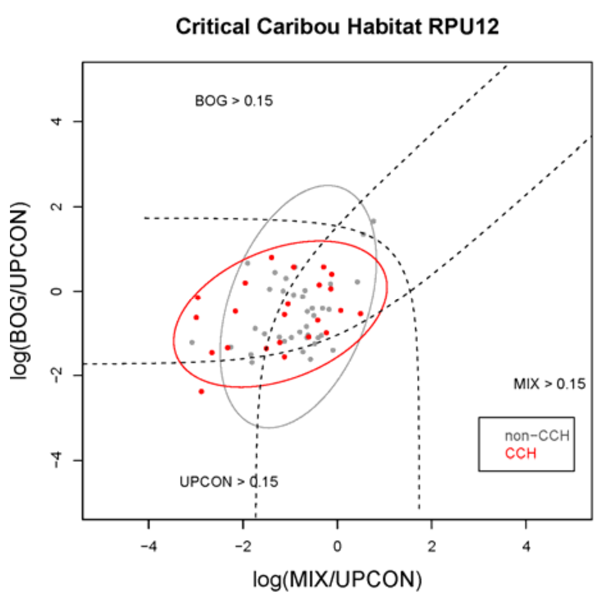

Figure 4.2: Logratio subcompositions of CCH landscapes (red) and non-CCH landscapes (grey) within RPU 12. Approximate 90% ellipsoids are shown in

corresponding colours. The seven regions of subcomposition space delineated by the 0.15 isolines are described in Table 4.2.

4.2.3 Landscape variation between RPUs and mapped CCHs

The compositions of CCH landscapes differed between RPUs 10 and 12 (p=0.001), with respect to both the means (p=0.001) and covariance structures (p=0.006). We attribute the differences simply to the larger range of landscape compositions available within RPU 10, both within and outside of mapped CCH areas.

The three CCH regions (our codes 15, 3 and 4) represented in RPU12 were only partially contained within that RPU (the proportional areas were 0.80, 0.48 and 0.62, respectively). Therefore, to examine compositional variation within and between mapped CCH areas, we used complete CCHs unstratified by RPU. The 4 complete CCHs with 10 or more landscapes accounted for 53/101 (52.5%) of all CCH landscapes. There is considerable variation in landscape subcomposition within and between mapped CCHs (Figure 4.3). There was a significant group effect (MANOVA, p=0.000017). Informally, between group differences in the mean subcomposition accounted for about 50% of the generalized variance among the four CCHs. Essentially, the same results were obtained when the top 6 CCHs with 77/101 landscapes (76.2%) were evaluated, or if analysis was restricted to RPU10. Although CCH landscapes can generally be characterised as UPCON/BOG mixtures, mapped CCHs are in fact a multi-scaled mosaic, with compositional variation at the scales of IPs within landscapes, landscapes within CCHs, and CCHs within the study region.

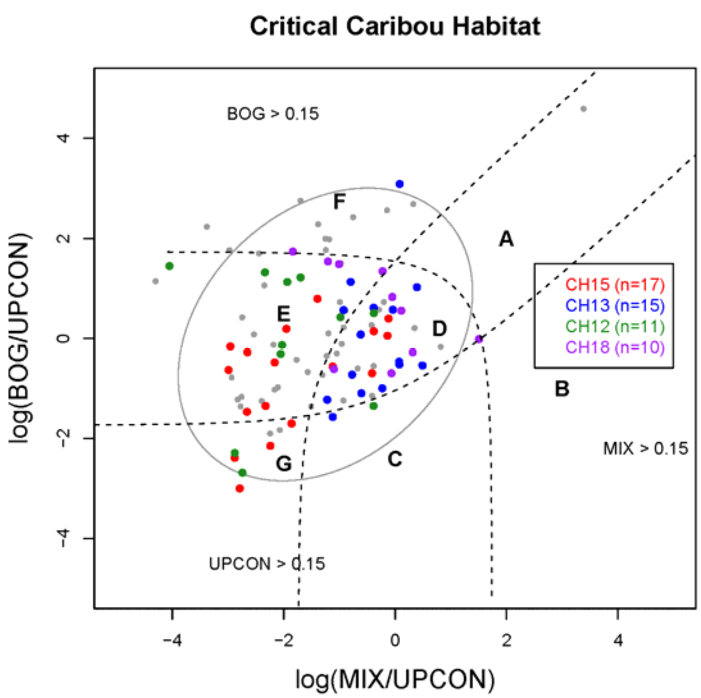

Figure 4.3: Logratio subcompositions of CCH landscapes for RPUs 10 and 12 combined, stratified by mapped CCH area. The seven regions of subcomposition space delineated by the 0.15 isolines are described in Table 4.2.

4.3 CCHs within RANs

Only about 6% of CCH area was included in the RANs.

4.3.1 RPU 10

Based on our area estimates, the RAN includes only about 2.0% of mapped areas of critical caribou habitat within RPU 10. The RAN sampled only 5 CCH landscapes, all of which were smaller than the median CCH landscape size (Figure 4.4). We did not formally test the two distributions because of the small sample size.

The RAN does contain a number of landscapes that fall within regions D and E of Figure 4.1, where most CCH landscapes are also found. However, most of these are either close to the isoline boundary between regions D and C or are smaller than the median CCH landscape size. Only 2 RAN landscapes are clearly in the interior of regions D and E and are larger than the median size. Even neglecting other factors (e.g. landscape spatial context, relevant landscape attributes other than terrestrial subcomposition), areas of critical caribou habitat within RPU 10, mapped or otherwise, are very poorly represented by RAN.

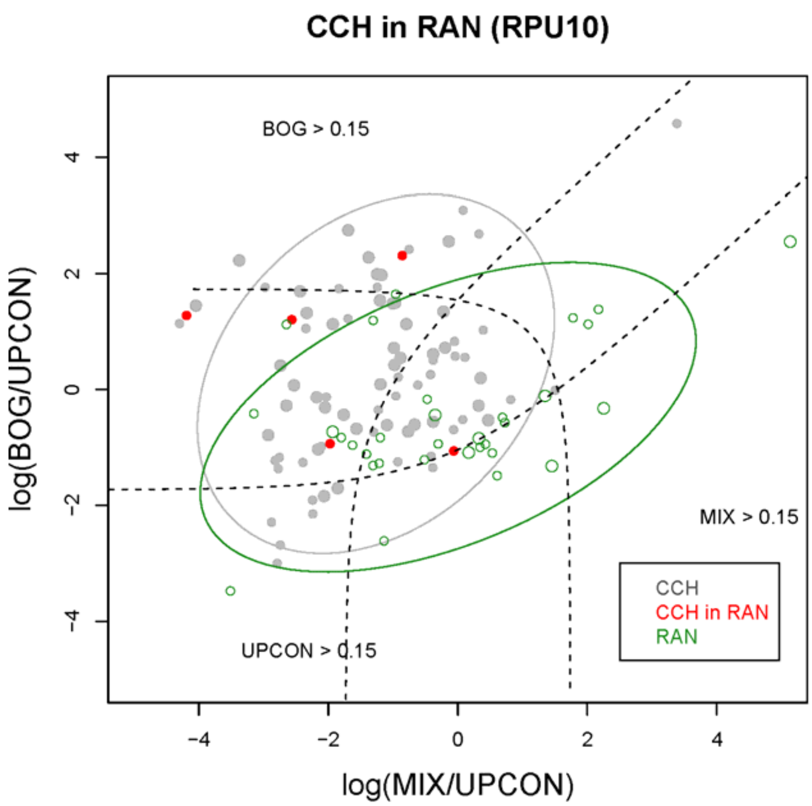

Figure 4.4: CCH landscapes within the RAN, RPU 10. All CCH landscapes are marked in grey, CCH landscapes or sublandscapes within the RAN are coloured red, and all RAN landscapes are coloured green. The point sizes indicate if the landscape is smaller or larger than the median CCH landscape area in RPU 10.

4.3.2 RPU12

The RAN includes about 13.6% of the mapped CCH areas within RPU 12. The RAN sampled only 4 CCH landscapes, all of which were below the median size of CCH landscapes within RPU 12 (Figure 4.5). The RAN contains three additional landscapes above the median size that fall within the 90% ellipsoid of RPU 12 CCH landscapes. However, the RAN landscapes sample only a portion of the distribution of CCH landscape compositions within RPU 12. In particular, the RAN contains no landscapes from region E (Figure 4.1, Table 4.2).

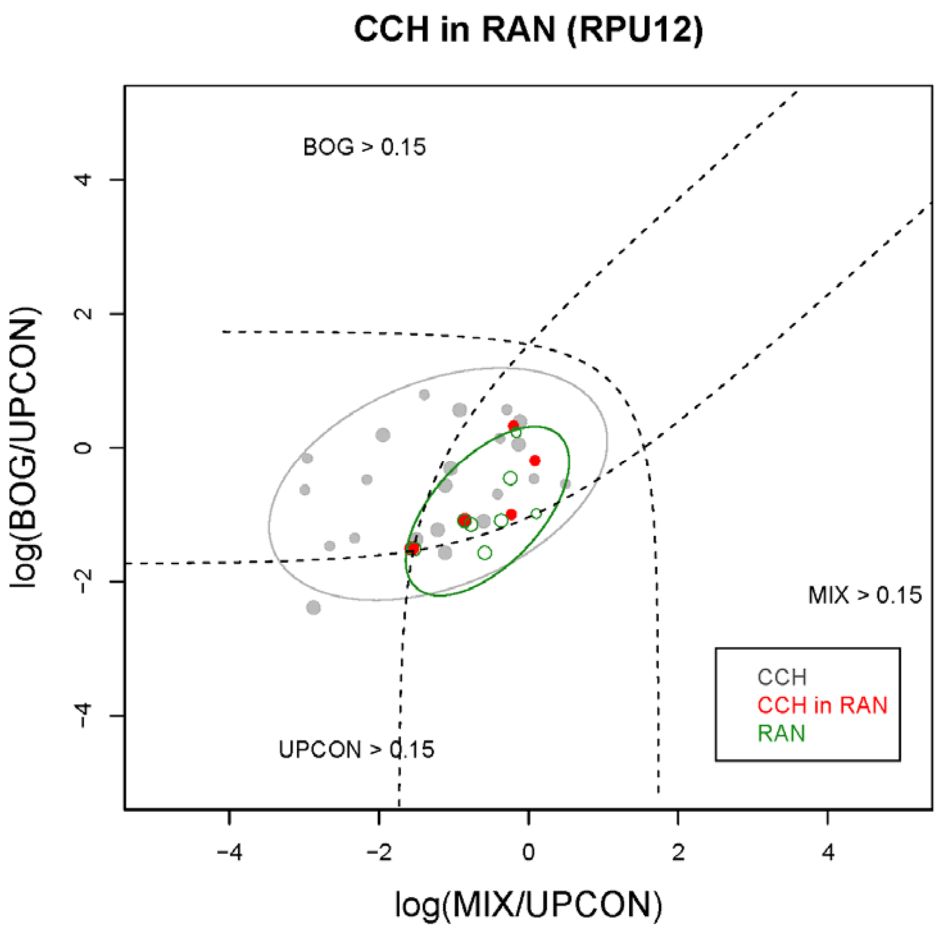

Figure 4.5: CCH landscapes within the RAN, RPU 12. All CCH landscapes are marked in grey, CCH landscapes or sublandscapes within the RAN are coloured red, and all RAN landscapes are coloured green. The point sizes indicate if the landscape is smaller or larger than the median CCH landscape area in RPU 12.

4.4 Summary

The areas of critical caribou habitat delineated by Arsenault (2003) are poorly represented within the RAN with respect to the total area of contiguous habitat within specific CCHs, the size of individual fragments of CCH landscapes that are sampled and, especially within RPU10, landscape composition. In RPU10, CCHs are composed of a variety of distinct landscape types that are absent or poorly represented within the RAN. We note that caribou range contractions along the southern fringes of the forested zone since 1950 (Arsenault 2003; Figure 1). The degree to which forest management (independent of other land-use changes) may have contributed to this contraction, or how it may contribute to future contractions is not well understood. Our findings indicate that the existing RAN cannot serve as a benchmark for forest management practices as they may affect woodland caribou. Within the boreal plains, at the landscape scale, a large proportion of mapped critical caribou habitat is within landscapes dominated by BOG, UPCON or BOG/UPCON mixtures. As some of these landscapes may be of relatively little commercial value, they may present opportunities to expand the RAN to incorporate benchmarks for woodland caribou.

Chapter 5: Age-Structure Analysis

5.0 Introduction

The evaluations of representation of forest attributes (Chapter 3) and of caribou habitat (Chapter 4) were in terms of the compositions of intrinsic habitat patches. Intrinsic patches (IPs) are defined from forest inventory attributes by aggregating adjacent mapped stands into larger areas that are expected to be temporally stable. That is, the definition of IP specifically seeks to abstract away from features of habitat that change over time, such as apparent age, canopy height, or (within limits) species composition. As such, IP analyses are not intended to reveal anything about the effects of past or future disturbances, such as fire or forest harvesting. However, in considering representative areas as potential benchmarks for forest management, variation in forest age-structure between the RAN and managed lands are obviously of interest. Therefore, in this Chapter, we present a simple qualitative analysis of the forest age structures within the study region.

The main comparison is between forest age structures within the Representative Areas Network (RAN) and outside the network (Non-RAN). Forest age structures are derived from the same inventory data sets used to delineate the Intrinsic Patches. Of the six IP types, only two (MIX and UPCON) are predominantly composed of forested mapped polygons of potentially determinant age.

Within the broader comparison of RAN and Non-RAN age structures, our analysis is stratified on two main axes: by ecozone and IP class. As the boreal plains and boreal shield ecozones have different fire regimes and management histories, one would expect differences in their age structures. Accordingly, we report results for the two ecozones separately. Within ecozones, IP classes MIX and UPCON are also known or expected to have different susceptibility to fire and have also historically been subject to different rates of harvest. Thus, the age distributions of MIX and UPCON may be expected to differ within and between RAN and Non-RAN areas. Accordingly, we present age structures for the two IP classes separately and in aggregate as an estimate of overall forest age structure.

5.1 Methods

Stratification of forest polygons by ecozone, IP-class and RAN/Non-RAN had already been accomplished to support the main analysis of this study (e.g. Chapter 3). The Saskatchewan forest cover attribute data were stored within a tiled structure. For this application, tiles that crossed either ecozone or RAN area boundaries were split. The resulting set of original or split tiles were assigned to one of four main categories: 1) Boreal Shield, RAN, 2) Boreal Shield, Non-RAN, 3) Boreal Plains, RAN, and 4) Boreal Plains, Non-RAN. In some cases, Boreal Plains, RAN was sub-categorized into a) all areas, b) all areas except Prince Albert National Park (PANP), and c) Prince Albert National Park only.

Age structures were determined by aggregating ages derived from the recorded forest inventory attributes. The inventory attributes used to determine age and age-structures were AREA (stand area in ha), YOO (year of stand origin), DIST (disturbance modifier code), YDR (disturbance year), HGT (height class code), and for PANP, COND (stand condition code). The complicating factors were: 1) not all polygons had a recorded YOO attribute; and 2) attribute sets or ranges varied among inventories.

Where recorded, YOO and YDR followed the conventions of Table 5.1, except in PANP. A significant area of forest stands was not assigned a YOO attribute but had DIST attributes indicating that they had been burned or harvested. For disturbed stands, the YDR attribute value, if assigned and non-zero, was taken as equivalent to YOO. Stand ages were calculated by subtracting the assigned year of origin from the reference year 2000. The inventory for PANP did not specify a YOO or YDR attribute, but it was possible to infer an age from the stand condition attribute (COND; Table 5.2). Note that for PANP, the oldest age class was 80+yr.

Table 5.1: Codes and interpretations for YOO (year of origin) attribute outside of PANP.

YOO Code	YEAR RANGE	Assigned Year of Origin
84	1836 - 1845	1840
85	1846 - 1855	1850
86	1856 - 1865	1860
87	1866 - 1875	1870
88	1876 - 1885	1880
89	1886 - 1895	1890
90	1896 - 1905	1900
91	1906 - 1915	1910
92	1916 - 1925	1920
93	1926 - 1935	1930
94	1936 - 1945	1940
95	1946 - 1955	1950
96	1956 - 1965	1960
97	1966 - 1975	1970
98	1976 - 1985	1980
99	1986 - 1995	1990

Some stands had neither a YOO nor DRY attribute. For many of these, it was possible to impute an age based on the height class attribute (Table 5.3). We used the ages for stands having a YOO to calculate height-class mean ages for each IP class and ecozone (Table 5.4). The proportion of total age-able stands whose ages were imputed from stand height varied from 1 to 14% depending on IP class and ecozone.

We calculated the mean age for each intrinsic patch type within the RAN and Non-RAN for the boreal shield and boreal plains (**Error! Reference source not found.**). Age-structures were represented graphically by binning stands into 20yr age classes and plotting proportional areas as bar-plots. Plots were prepared for each ecozone. For the boreal plains, we prepared two plots, one excluding PANP and one including PANP. For the case including PANP, the oldest age class included all stands 80yr and older and age-class bins were based on Table 5.2; stands in COND classes 5 and 5a were assigned to the 80yr+ age bin. Otherwise, the maximum age class was 120+yr. Within each of the three cases, we present 4 plots: 1) all forested areas including both IP classes MIX and UPCON; 2) forested areas and the proportions of disturbed or burned areas; 3) IP class MIX; and 4) IP class UPCON. Each plot contrasts the age-structures for RAN and Non-RAN stands.

Table 5.2: Description COND attribute classes for Prince Albert National Park inventory and interpretation of stand ages based on class levels

COND Class	Description	Year Range	Assigned Age of Origin
1	forest land restocked following disturbance	10	5
2	young stands	10-30	20
3	semi-mature stands	30-60	45
4	mature stands	60-80	70
5	mature stands	> 80	100
1A	forest land not restocked following disturbance	10	5
2A	stands similar in height to class 2 but with retarded growth due to site or overstocking	10-30	20
3A	stands similar in height to class 3 but with retarded growth due to site or overstocking	30-60	45
5A	overmature stands showing signs of decadence	> 80	100

Table 5.3: Saskatchewan Forest Inventory, database structure for Height (HGT) class code

Code	Height Class
5	2.5M < HGT < = 7.5M
10	7.5M < HGT < = 12.5M
15	12.5M < HGT < = 17.5M
20	17.5M < HGT < = 22.5M
25	22.5M < HGT

Table 5.4: Age estimates for MIX and UPCON Intrinsic Patch stands within the boreal shield and plains ecozones within five height classes. Age is estimated from year of origin (YOO) and year of disturbance (DYR) using year 2000 as the reference year. In brackets is the number of stands used to estimate age

Code	2.5 < 7.5m	7.5 < 12.5m	12.5 < 17.5m	17.5 < 22.5m	> 22.5m
Plains-MIX	29.70 (20775)	59.13 (47684)	74.26 (101843)	99.89 (107170)	114.24 (30194)
Plains-UPCON	44.49 (31422)	82.21 (157138)	90.97 (136505)	105.77 (34328)	119.84 (547)
Shield-MIX	42.04 (3560)	64.30 (26941)	86.33 (25293)	117.47 (11272)	124.88 (281)
Shield-UPCON	51.71 (10795)	75.72 (53878)	101.82 (59775)	123.90 (19582)	113.87 (31)

Table 5.5: Mean age estimates for Saskatchewan Forest Inventory and Prince Albert National Park. The group-percentage of stands that had an age estimate is shown in parenthesis.

Code	Plains-RAN	Plains-NonRAN	Shield-RAN	Shield-NonRAN	Prince Albert
MIX	79.07 (100.00%)	74.95 (99.99%)	94.14 (100.00%)	75.54 (99.99%)	54.37 (100.00%)
UPCON	80.77 (99.87%)	80.97 (99.67%)	95.74 (79.48%)*	86.42 (81.49%)*	42.05 (100.00%)

The full spatial data set, computer scripts and data tables used to generate this report are retained by BEACONs and will be made available to the Government of Saskatchewan on request.

5.2 Results

We assigned observed or imputed age to a large majority of the forest area (Table 5.5). The only significant areas lacking ages were UPCON stands in the boreal shield. This IP class included up to 30% by area of stand mapped as “Treed Rock” which was considered to be UPCON based on their possible suitability as caribou foraging habitat. In the boreal plains, there was little difference between group-mean ages of MIX or UPCON forest in or out of the RAN (Table 5.5). The apparent exception of PANP is attributable to the upper age class being 80yr. The highest mean age occurred on boreal shield within the RAN, where stands were about 15yr older than in Non-RAN areas. The reasons are unclear.

5.2.1 The Boreal Plains

In the boreal plains, excluding PANP, the total forest age structures and the proportion of recently disturbed areas did not differ markedly between the RAN and Non-RAN areas (Figure 5.1, top two panels). In particular, the total areas of disturbed and young forest (≤ 40yr; the youngest two age-classes) were roughly equal. Within IP classes, differences in age structure between the RAN and Non-RAN were slightly more pronounced. Age-structural differences were most pronounced within stands of class MIX. Stands of youngest age-class (<20yr) accounted for roughly 11% by area of all aged stands outside the RAN but only about 4% within the RAN. The proportional area of the 20-40yr age class was roughly 5% outside the RAN and 2% within the RAN. Thus, it could be said that these two age classes of mixedwood forest are somewhat under-represented within the RAN in the boreal plains. This probably reflects recent harvesting of this stand type. The 40-60 and 60-80yr age classes were somewhat over-represented within the RAN. There was little age-structural difference among UPCON forest. Age structures including PANP (Figure 5.2) highlighted the relative lack of young MIX forest within the RAN; the youngest age class (0-10yr) was relatively unrepresented. The results were otherwise very similar to those of Figure 5.1.

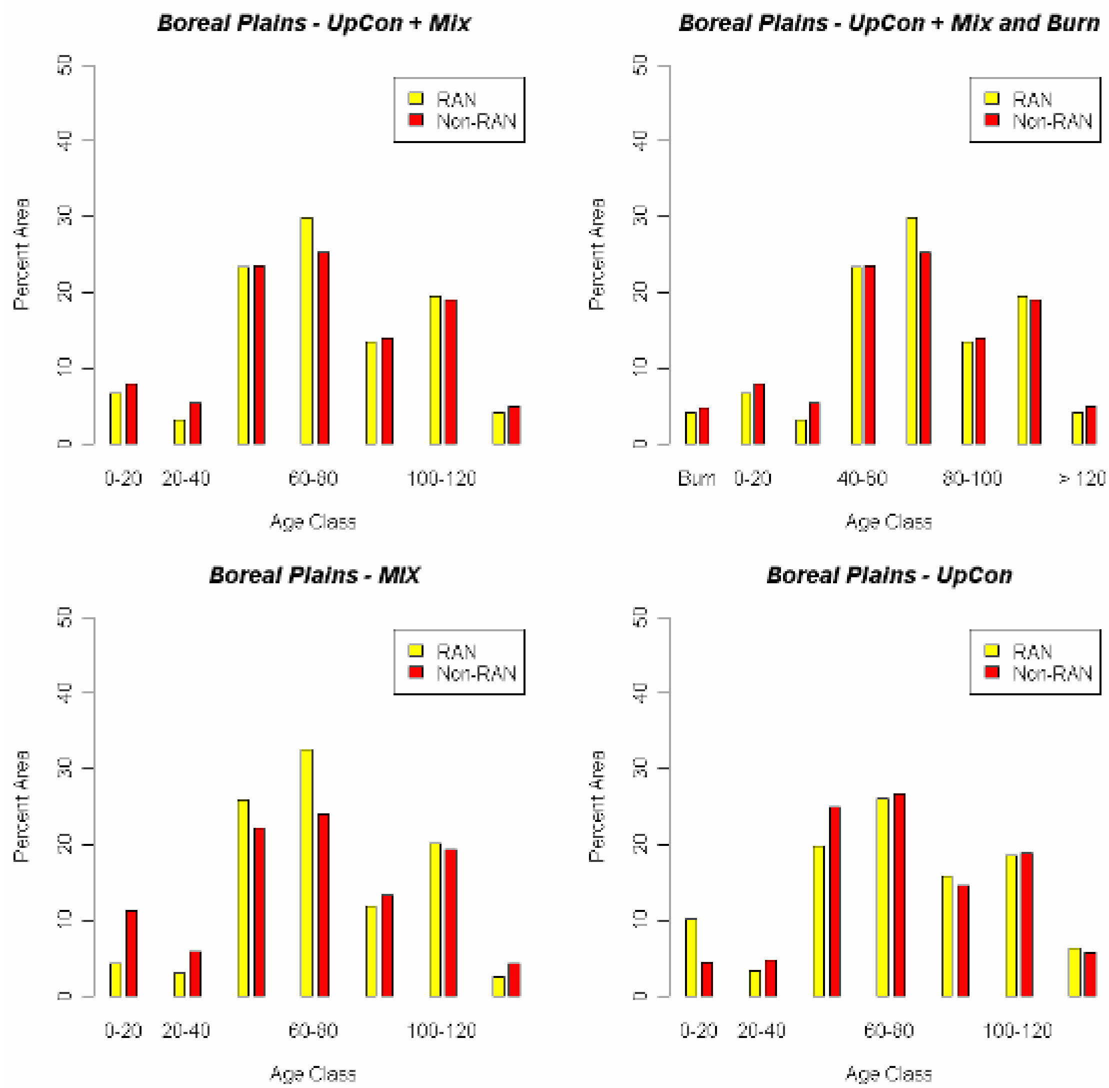

Figure 5.1: Estimated forest age structures within the boreal plains, **excluding** Prince Albert National Park. Bar plots contrast the proportional class areas within and outside the RAN for: total forested areas (upper left), total forested area and all recent disturbances (upper right), forest in IP class MIX (lower left), and forest in IP class UPCON (lower right). Category "Burn" includes the proportional areas of all recent or un-dateable harvested and burned stands.

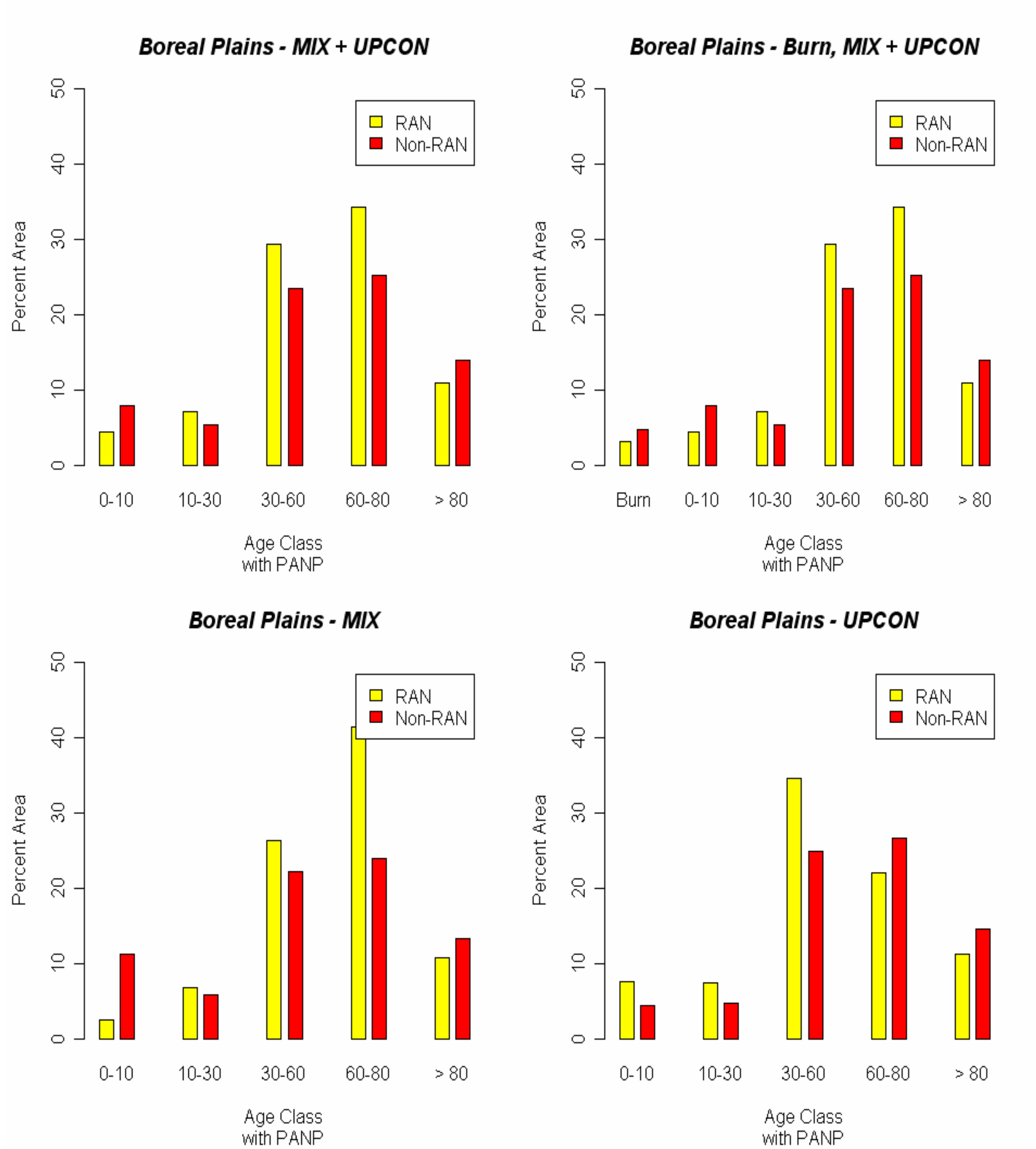

Figure 5.2: Estimated forest age structures within the boreal plains, **including** Prince Albert National Park. Bar plots contrast the proportional class areas within and outside the RAN for: total forested areas (upper left), total forested area and all recent disturbances (upper right), forest in IP class MIX (lower left), and forest in IP class UPCON (lower right). Note that the maximum age class is 80+ yr, as PANP inventory did not record any greater ages. Category “Burn” includes the proportional areas of all recent or un-dateable harvested and burned stands.

5.2.2 The Boreal Shield

There was considerable difference in age structures between RAN and Non-RAN areas on the boreal shield (Figure 5.3) for forest areas in general and among IP classes. Differences appeared more pronounced than on the boreal plains. Representation of the oldest age class (120+yr) was equitable, being roughly 10-15% for both MIX and UPCON forest within RAN and Non-RAN areas. We note that, compared to the boreal shield, this older age class seems relatively under-represented within the entire boreal plains. Probably, the most ecologically significant aspect of non-representativeness is the near-absence within the RAN of forest in the 0-20yr and 20-40yr age classes, whether MIX or UPCON IP types. Recently disturbed areas, by contrast, are well-represented (Figure 5.3, upper right panel).

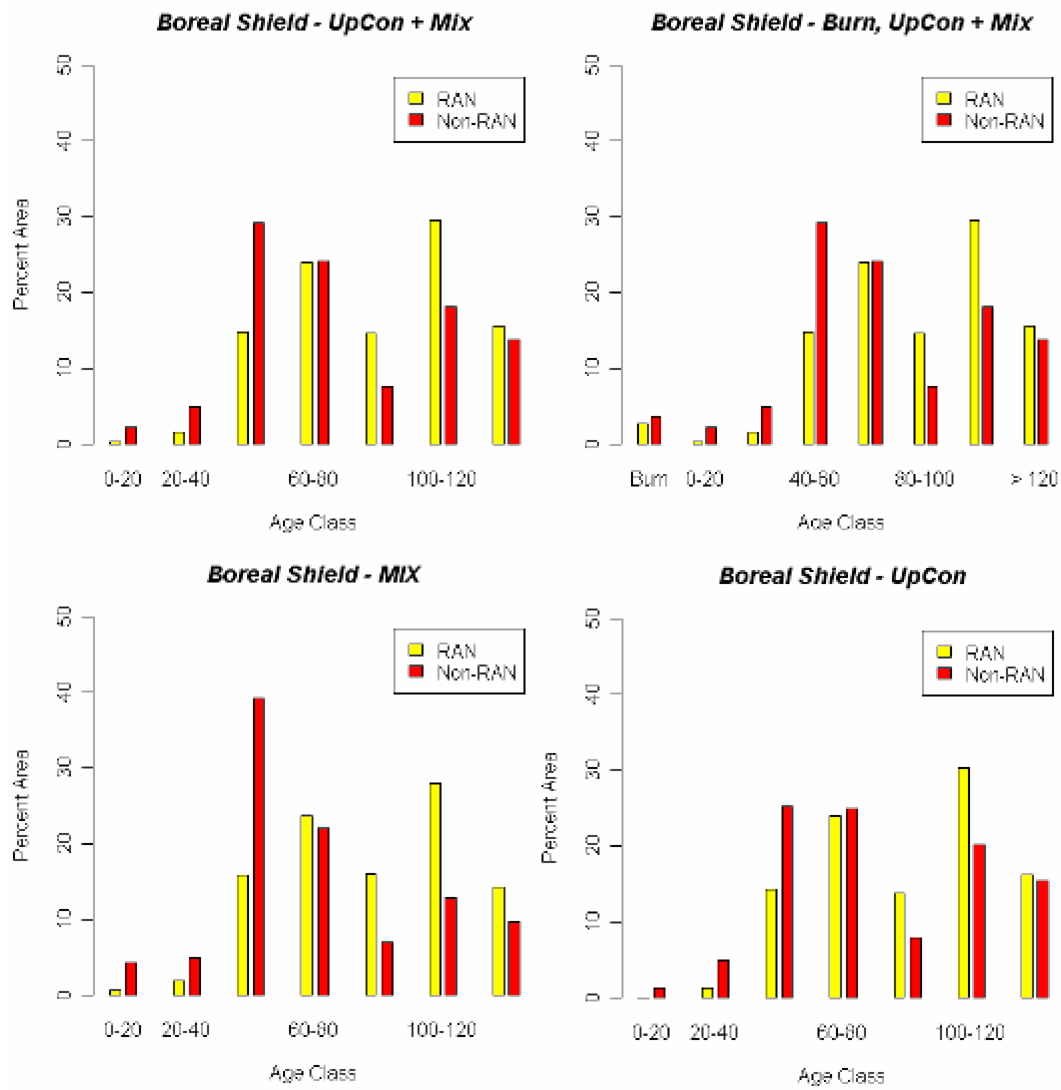

Figure 5.3: Estimated forest age structures within the boreal shield. Bar plots contrast the proportional class areas within and outside the RAN for: total forested areas (upper left), total forested area and all recent disturbances (upper right), forest in IP class MIX (lower left), and forest in IP class UPCON (lower right). Category "Burn" includes the proportional areas of all recent or un-dateable harvested and burned stands.

5.3 Conclusion

The main results of this analysis is that forest age structures within the RAN are reasonably representative of those outside the RAN, for both IP classes MIX and UPCON and in both the boreal plains and boreal shield ecozones. The main exception to this general finding is that the youngest age classes (0-20yr and 20-40yr), though rare overall, are less well represented. This applies especially to forest of type MIX within the boreal plains. Young forest of both IP classes is very under-represented within the boreal shield RAN areas.

Although the absolute areas involved are small compared to total forest area, it may be that as harvesting activity proceeds in the managed forest and the forest in the RAN ages, the RANs will have limited ability to function as controls for forest harvesting; in that it will be difficult to compare ecological responses on young anthropogenic and fire-origin stands because comparable areas of fire-origin forest will not exist within the RAN. This potential problem could be addressed by augmenting the existing RAN with areas of recently burned forest, while taking steps to ensure that such forest was periodically recruited into the network. Placing recently harvested areas into the RAN would not in itself provide adequate controls unless the desired contrast was between young and old forest as such, and not between the different treatments of fire and harvesting.

We recommend that further studies be conducted with a view to: 1) specifying in detail the nature and size of control areas required in terms of size, age, spatial distribution and initial condition (e.g., canopy species composition), with particular emphasis on retaining un-salvaged burned areas; and 2) designing an active management strategy to maintain the desired distribution and abundance of control areas over time. We suggest that different strategies may be required in the boreal plains and boreal shield to account for differences in their climatic and site conditions and in the level of past and current harvest rates.

Chapter 6: Independent Identification of System-Level Benchmarks in Saskatchewan

BEACONs has developed a process-based approach for constructing system-level benchmarks *de nouveau* in the boreal region of Canada. System-level benchmarks are of sufficient size to experience the largest, anticipated natural disturbance (*e.g.,* fire), and still maintain internal recolonisation sources (Pickett and Thompson 1978) (see Chapter 1). Here, we describe the approach and provide examples of system-level benchmarks identified in Saskatchewan. Also, we describe the Conservation Matrix Model, the scientific framework guiding BEACONs. It is useful to consider the role and design of benchmarks in the context of this model.

6.1 Conservation Matrix Model

The Conservation Matrix Model (CMM) proposes a scientific framework for proactive conservation planning in boreal regions of Canada. The CMM is comprised of 4 principle components: ecological benchmark areas, additional reserves, active management areas, and the larger conservation matrix within which the former three elements are embedded, and to which they contribute (Figure 1). **Ecological benchmarks** are the anchors of a comprehensive reserve network and serve as reference sites or controls for understanding both the natural dynamics of ecosystems as well as their response to human activities. **Additional reserves** capture values that may not be well represented within benchmark areas, such as identified special elements (e.g., early-season open water for migrating waterbirds, areas of cultural significance, rare species occurrences), and may include existing and new protected areas that do not fulfill benchmark criteria. **Active management areas** are sites of relatively intense human activity, such as forestry, mining, or oil and gas exploration. These areas are managed under the principles of Adaptive Resource Management, such that management activities are treated as experiments designed to identify truly sustainable practices. The **conservation matrix** is the supportive landscape within which less intense human activities are carefully managed so as not to erode other values.

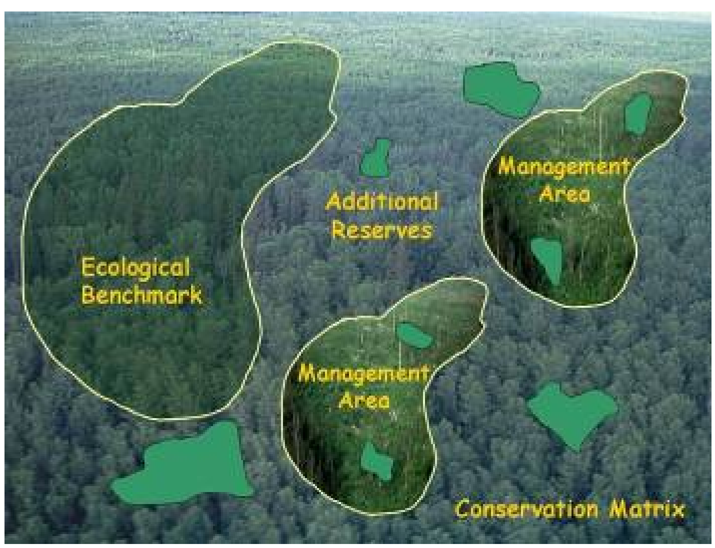

Figure 6.1: Conservation Matrix Model

6.2 Identification of System-Level Benchmarks in Saskatchewan

To identify system-level benchmarks, we developed tools for constructing and ranking candidate areas based on biophysical criteria. The benchmarks presented here are a subsample from a national analysis and do not represent a comprehensive assessment of the potential for system-level benchmarks in Saskatchewan. Rather, they illustrate the concepts and approach.

6.2.1 Study Area

The study area for this exercise included Regional Planning Units (RPUs) 10.1 and 12.1 (Figure 2). These RPUs vary slightly from the RPUs described in Chapter 2 because we have since incorporated our work on fire regionalization (Cumming *et al. unpublished*) into the delineation of RPUs. RPU 10.1 and 12.1 nevertheless remain largely comprised of the Boreal Plains and Boreal Shield ecozones, respectively.

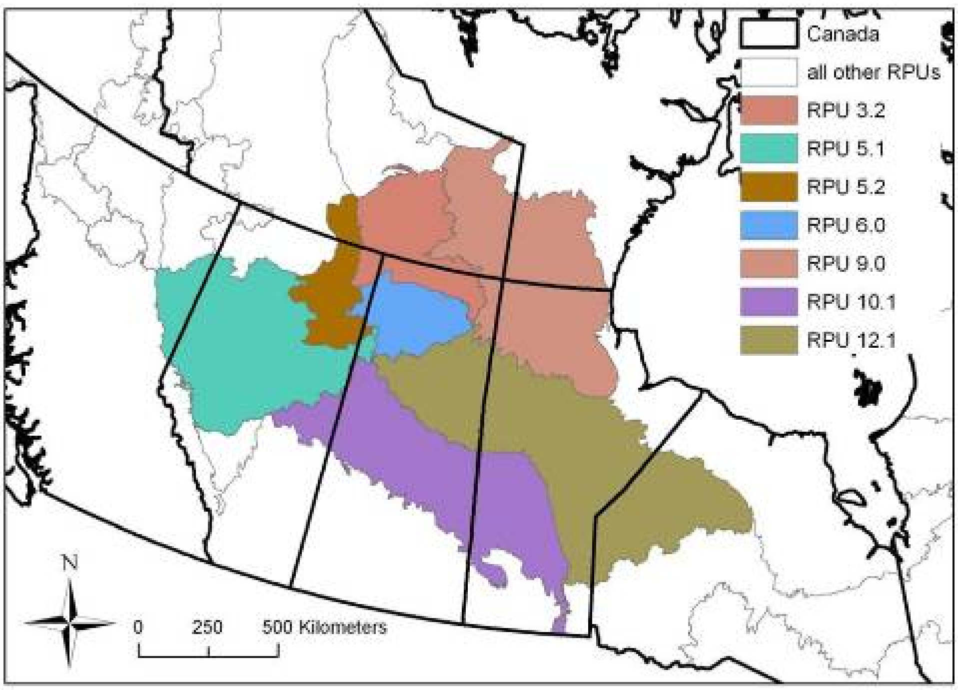

Figure 6.2: Regional Planning Units intersecting Saskatchewan.

6.2.1 Methods

Benchmark Builder - The Benchmark Builder is a computer automated algorithm for the construction of ecological benchmarks. Benchmarks are grown by aggregating small (~100 km^2) drainage areas, called "catchments," based on criteria for: (1) **intactness**, a measure of the absence of human industrial activity (Lee et al. 2006) and a proxy for the intactness of biological and physical processes; (2) **hydrologic connectivity**, as a measure of the integrity of aquatic systems; and (3) **area**, a measure of the resilience of the system to disturbance. Using catchments as building blocks, the Benchmark Builder aggregates neighbouring catchments along stream networks to incorporate hydrologic connectivity into benchmark construction. It attempts to capture headwaters within benchmarks by adding upstream catchments first. The Benchmark Builder only uses catchments that exceed a user-defined intactness threshold (*i.e.*, catchment-intactness), and tracks the overall area-weighted intactness of each benchmark. To incorporate resilience to disturbance, the benchmark area is defined in relation to the local fire regime, with the intent to maintain internal recolonisation sources after major fire events.

For this exercise, we defined catchment-intactness and area-weighted intactness of benchmarks as 80% and 95%, respectively. The 95% area-weighted intactness for benchmarks was derived from an area-weighted intactness analysis of IUCN category Ia and Ib protected areas in boreal Canada (BEACONs *unpublished*). The 80% catchment-intactness threshold was based on a literature review of disturbance and detectable change in aquatic processes. The area requirement for benchmarks was 3 times a locally-derived estimated maximum fire size. The estimated maximum fire size was derived from our novel work of fire regions in the boreal (Cumming and Mackey *unpublished*). The 3 times multiplier was derived from Leroux *et al.* (2007) based on simulation experiments.

Ranker - The benchmark Ranker uses statistical dissimilarity metrics to rank benchmarks according to their ability to represent a set of biophysical criteria. Representation is measured by comparing the distributions of biophysical criteria within one or more benchmarks and the associated regional planning unit. Any number of continuous or discrete distributions can be incorporated. At present, we use four: climate moisture index (CMI), gross primary productivity (GPP), lake-edge density, and remote-sensed land cover. More detailed information on these attributes and our reasons for selecting them is available. Benchmarks are chosen to match, as closely as possible, the marginal distributions of these metrics over the entire RPU. Dissimilarity metrics are calculated for each distribution, and then scaled and aggregated into a univariate index of multivariate representation. This index is used to rank the benchmarks in order of representation with respect to the distributions of indicators in the target RPU. This approach is derived from an earlier study of landscape scale representation in terms of joint forest stand age and size-class structures within a portion of the boreal plains ecozone (Cumming, Burton and Klinkenberg 1996).

Analyses - To demonstrate the Builder and Ranker, we only use benchmarks from RPU 10.1 and 12.1 that were constructed in Saskatchewan. For ranking, we compare the distribution of biophysical criteria within benchmarks to distributions for the entire RPU. Ideally, we would restrict the RPU distributions to the regions of RPUs intersecting Saskatchewan.

6.2.2 Results - Benchmark Construction and Ranking

Boreal Plains (RPU 10.1) – Only a small region of the boreal plains, east-central Saskatchewan, was able to support our stringent criteria for system-level benchmarks. In this region, we identified and ranked 6 benchmarks. All benchmarks intersect the Seager Wheeler Lake Representative Area (Figure 3). The benchmarks range in size from 4,033 to 5,574 km^2.

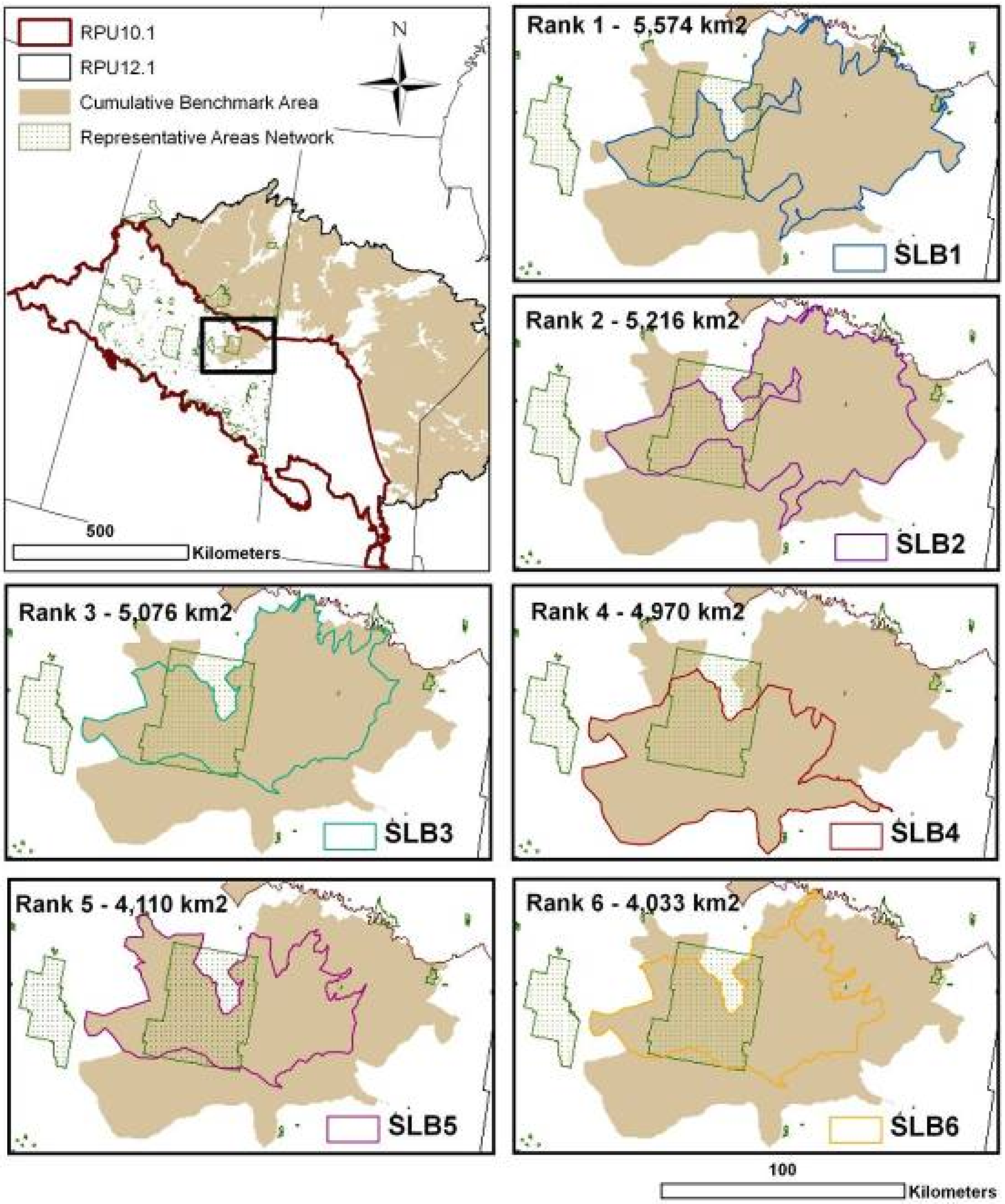

Figure 6.3: Six system-level benchmarks produced by the Benchmarks Builder for 80% catchment-intactness and 3x multiplier. The benchmarks were ranked using the Ranker.

Boreal Shield (RPU 12.1) – Figure 4 shows the cumulative area of RPU 12.1 captured by candidate system-level benchmarks. Benchmarks were identified across RPU 12.1 (N= 287) with 43 benchmarks completely within Saskatchewan. We ranked the benchmarks in Saskatchewan and present the 12 top-ranked. The benchmarks range in size from 9,112 to 16,488 km^2.

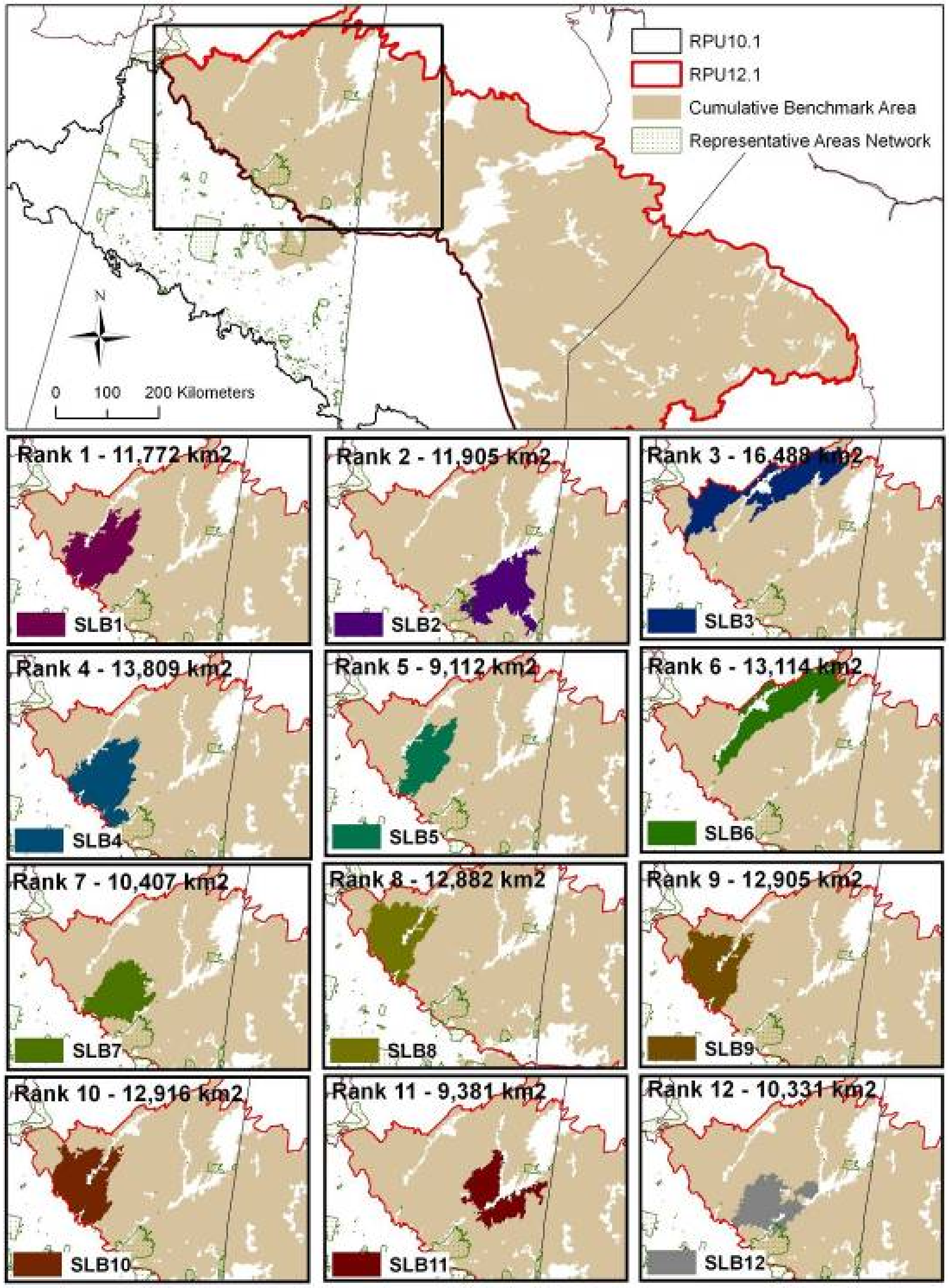

Figure 6.4: The 12 top-ranked system-level benchmarks in Saskatchewan produced by the Benchmark Builder for 80% catchment-intactness and 3x multiplier.

6.3 Discussion

It is possible to construct independently-derived system-level benchmarks in Saskatchewan, in both RPU 10.1 and 12.1. In RPU 10.1, based on the criteria used, options for system-level benchmarks are restricted to a small region in east-central Saskatchewan. This is due to extensive human development in the boreal plains and fragmented patches that are too small for system-level benchmarks (Figure 5). The benchmarks identified intersect the Seager Wheeler Lake Representative area. The representative area could be expanded to serve as a system-level benchmark. Presently, we are working on modifying the Builder to grow benchmarks from multiple-catchment seeds (*i.e.*, existing reserves). Once the Builder is modified, it may also be possible to grow system-level benchmarks from exiting reserves in the boreal shield, such as Lac La Ronge Provincial Park. RPU 12.1 is highly intact (Figure 5). As a result, there is considerable opportunity to establish benchmarks across provincial boundaries, including the potential to coordinate efforts with Manitoba.

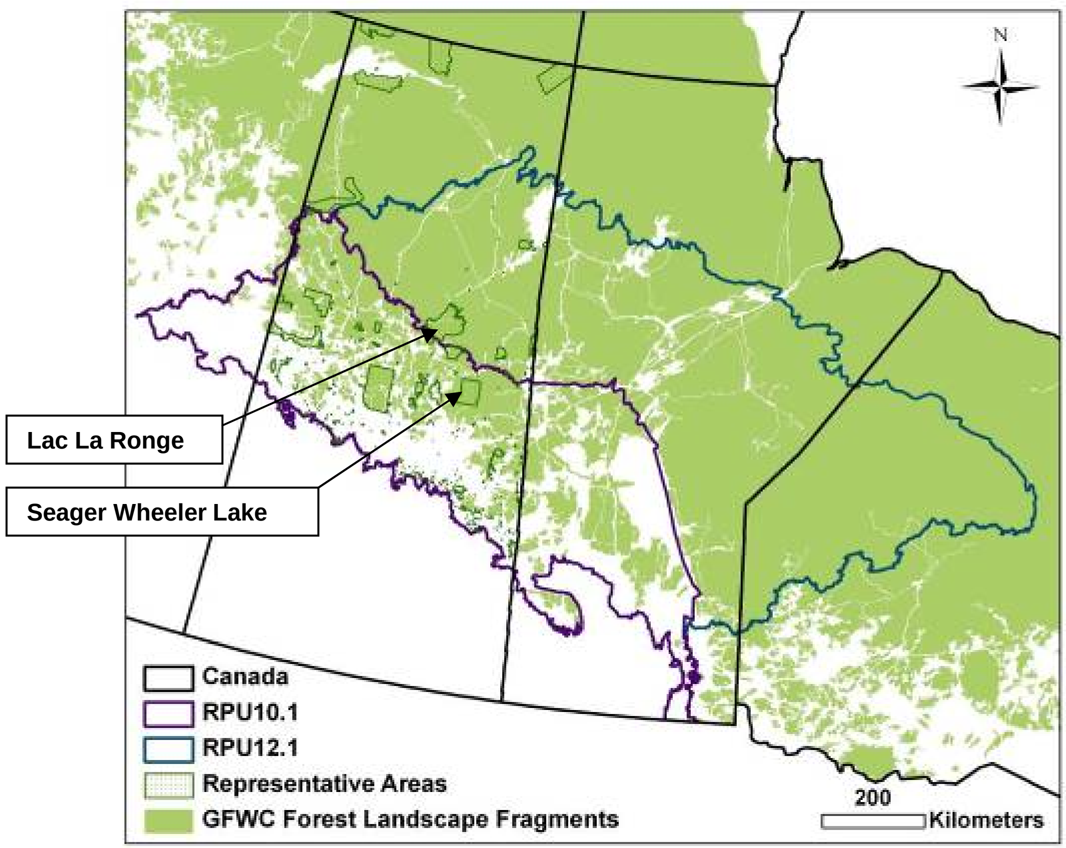

Figure 6.5: Green represents areas of intact forest landscape fragments as described by Lee et al. (2006).

The multiplier used in this analysis was based on simulation experiments in the Northwest Territories using CONSERV, a dynamic simulation model for determining the area of minimum dynamic reserves (Leroux *et al.* 2007). CONSERV can be parameterized to derive multipliers specific to Saskatchewan. This would require a suite of vegetation succession rules and fire parameters for Saskatchewan. If a smaller multiplier is identified for Saskatchewan's boreal plains, there may be improved opportunity for system-level benchmarks in this region.

The benchmarks were ranked based on 4 biophysical criteria. Another ranking criterion to consider is the hydrologic connectedness of the benchmark to the surrounding landscape (*i.e.*, the number of streams flowing into and out of a benchmark). Benchmarks with fewer inflow and outflow streams are less at risk from exogenous hydrologic disturbance. Alterations in the flow of surface and subsurface waters, and the input of pollutants and introduced species, for example, all have the potential to influence the integrity of aquatic and terrestrial ecosystems in reserves (Freeman *et al.* 2007, Wipfli *et al.* 2007, Winter 2007, Pringle 2001, Pringle 1997). Ideally, the establishment of a benchmark would include special management of all headwaters flowing into the benchmark. The transmission of disturbance upstream from downstream sources should also be carefully monitored.

We restricted this analysis to regions of Saskatchewan intersecting RPUs 10.1 and 12.1, and to only a subsample of the benchmarks constructed in RPU 12.1. Based on our national analysis of boreal system-level benchmarks, it is possible to construct system-level benchmarks in all RPUs intersecting Saskatchewan (Figure 6). There is potential to build system-level benchmarks from existing and proposed representative areas in all RPUs.

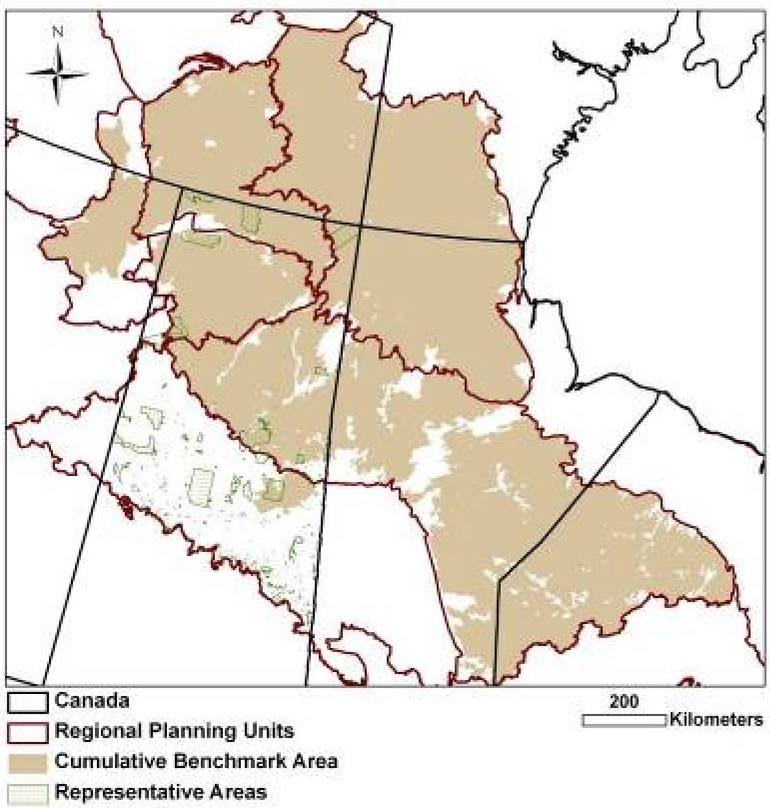

Figure 6.6: RPUs intersecting Saskatchewan and the cumulative area of all system-level benchmarks identified in these RPUs.

`Chapter 7: Next Steps

This project included four major components, two of which are dealt with in detail in this report: defining the role and criteria for ecological benchmarks in the context of forest management and evaluating the adequacy of the existing RAN in Saskatchewan relative to this objective. The final two components: identifying potential benchmark areas in Saskatchewan and adjacent areas and opportunities for expansion of the RAN to better serve this role, are yet to be completed. We here address several aspects of the analyses to date which could be enhanced or refined, and outline a strategy for undertaking the final components of this work.

7.1 Intrinsic Patches

We quantified intrinsic patches as a coarse-scale measure of forest composition, and to some extent spatial structure, but there are limitations to the IP approach, some of which could be addressed through further analyses. The degree to which this is warranted depends on higher-level objectives. We highlight several areas for consideration.

1) The intrinsic patch classification of mixed stands of pine and aspen is problematic. This stand type becomes increasingly common northwards into the shield. There is literature (Venier and Pearce 2005.) to suggest that such stands have some of the characteristics of MIX intrinsic patches with regard to avifauna, for example; however, these stands are not invariant MIX over time. The potential significance of this requires further consideration.

2) We may need to refine the IP classification for some purposes. For example, intrinsic patch types BOG and UPCON are both comprised of more than one stand type (see 3. 1, Chapter 3), some of which are important for caribou. When evaluating the representation of critical habitat for caribou, the IP classification should be refined so that only relevant stand types are included in the analysis.

3) We were not able to classify recently burned areas. Most forest inventories, including Saskatchewan and Alberta, remove attributes of burned forested stands but leave burned BOG alone. This means that most of the unclassified burned areas were either MIX or UPCON. It should be possible to develop GIS procedures to classify most burns with reasonable accuracy. We also assumed that all recent cuts had been MIX; however, this may not be true for some remote areas of Saskatchewan and Manitoba. Addressing this would require more information from SEFS and other agencies, in conjunction with additional GIS analysis, to refine the classification of burns and recent cuts.

4) The representation analysis should be extended to deal with multivariate size-class structures, not just compositions. In other words, it should consider the joint size-distributions of MIX, BOG and UPCON patches within a landscape unit, not just the proportional areas of each type. Also, the three-way analysis should be expanded to include more than three IP types (e.g. RIPARIAN and WATER).

5) The IP analysis should be extended to deal with size and connectivity issues. Existing software designed for constructing reserves with specified characteristics could be utilized.

7.2 Hydrologic Connectivity

Our evaluation did not address hydrologic connectivity, an important component of ecological integrity. Hydrologic connectivity encompasses "water-mediated transfer of matter, energy, and/or organisms within or between elements of the hydrologic cycle" (Pringle 2001). In order for RAs to serve as benchmarks for hydrologic processes, the influence of anthropogenic alterations or inputs into the hydrologic cycle of the RA must be minimized or ideally, absent. Human disturbances can affect hydrologic connectivity as well as associated aquatic and terrestrial biomes through both upstream and downstream influences. Pringle (2001) notes that much of a landscape's surface configuration can be attributed to its drainage network of rivers that form a predictable structural pattern affecting watershed geochemistry, topography, climate, and vegetation. Thus, hydrologic connectivity should be carefully managed within and outside of benchmarks through consideration of both surface and sub-surface flows. At present, we are only able to address surface flow because of the challenges of modelling and mapping subsurface flow patterns, although emerging research (*e.g.*, Devito *et al.* 2005) holds promise for the latter. A next step in the benchmark evaluation of RAs is to assess hydrological connectivity.

Here, we briefly describe an approach for such an assessment that could also be used in benchmark construction. Considering Prince Albert National Park as an example, one can use the stream flow network and catchments[11] to determine the headwater regions connected to the park. Figure 7.1 illustrates the flow direction of streams in the park. Focusing on the top-right corner of the park, we traced the flow of the stream network to the headwaters of a third-order watershed and identified all catchments contributing to the surface flow of the watershed. The result is the light, medium and dark green area in Figure 7.2. In this example, the headwaters of the watershed lie outside of the park. Perturbations to these headwaters could affect aspects of the hydrologic cycle within the park and thus compromise the hydrologic benchmark potential of the park. By adding headwaters to the park, or areas of special management, the influences on the hydrologic cycle within the park would be captured and managed such that the hydrologic connectivity and integrity of the benchmark would be maximized.

[11] The catchment dataset is a polygon coverage representing an approximate catchment area for each individual river segment and lake in the drainage network.

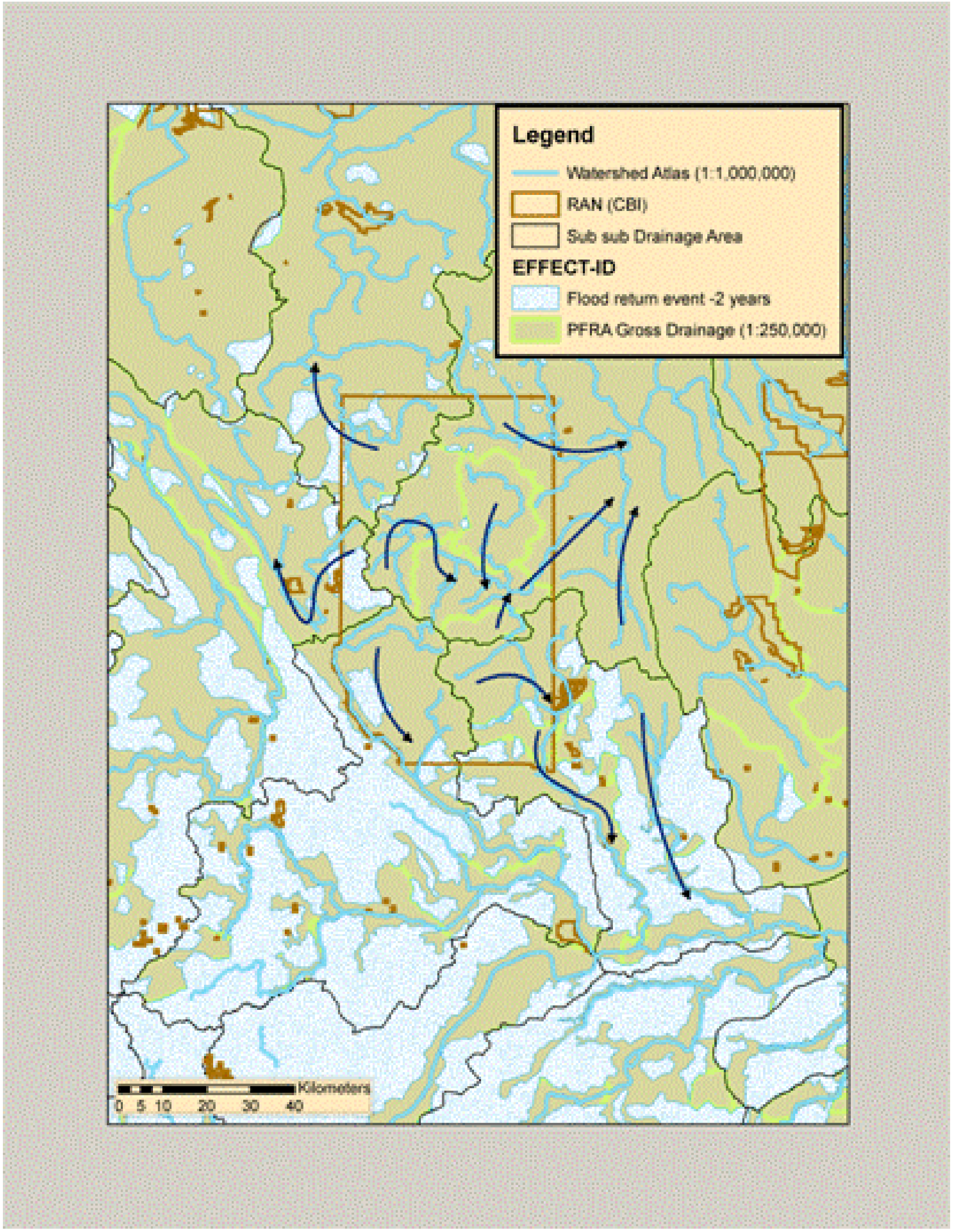

Figure 7.1: Stream flow direction in Prince Albert National Park.

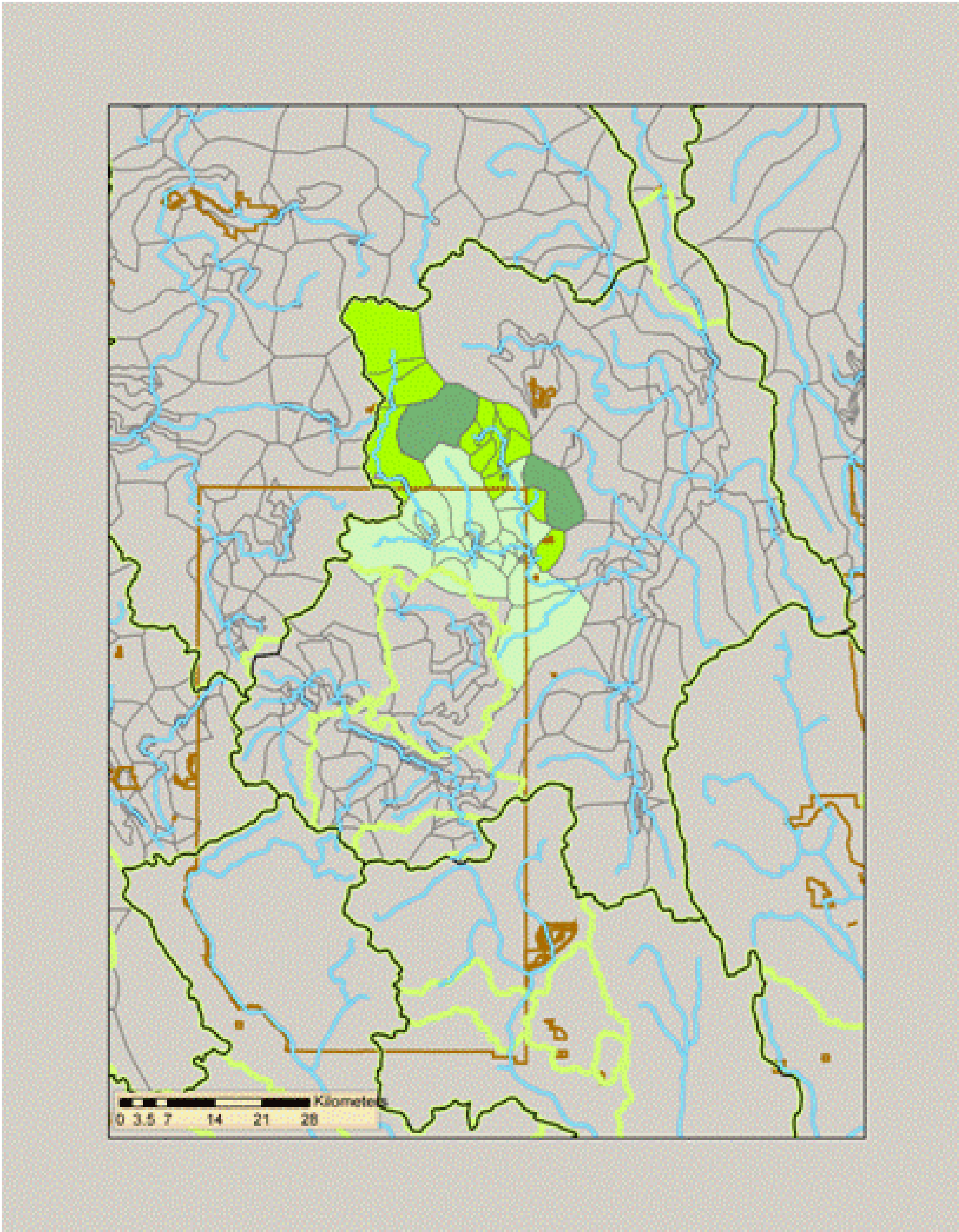

Figure 7.2: The light, medium, and dark green areas, at the top-right corner of Prince Albert National Park, are the headwaters of a third-order watershed.

7.3 Northern Saskatchewan

Our evaluation to date is restricted to the central and southern portion of the boreal region in Saskatchewan, due to the paucity of spatial land cover data for the north. To expand components of the analysis north, development of methods for deriving intrinsic patch structure from old non-spatial inventory and remotely-sensed data must be developed. Coarser-scale analyses using alternative environmental variables derived from remotely-sensed data are also possible.

7.4 Benchmark Construction

The priority for next steps is to identify candidate benchmark areas from existing RAs and additional lands within and adjacent to the study area. Two categories of benchmarks are considered here: system-level benchmarks (SLBs) and less comprehensive benchmarks that conform to the sliding scale introduced in Chapter 2. Based on the preliminary evaluation in Chapter 2, none of the existing RAs meet the criteria of a system-level benchmark, due primarily to size limitations.

System-level benchmarks serve as controls for monitoring a breadth of large-scale processes and addressing many questions related to sustainable forest management. For this reason, SLBs are the ideal category of benchmarks for forest management. A method for identifying SLBs was outlined in Chapter 6. This includes exploring opportunities to expand existing RAs.

Each of the RAs may have some utility as a benchmark, but a benchmark for what? As stated in Chapter 1,

> *One of the roles of the Representative Areas Network (RAN) is to serve as a "control" for forest management activities. As ecological benchmarks, representative areas provide the opportunity to monitor and compare the outcomes of forest management to the natural system and provide the basis for adaptive management. Ecological benchmarks must capture the range of forest ecosystems they are intended to represent and, to the degree feasible, be of sufficient spatial extent in relation to boreal forest disturbance events.*

Based on the description above, a system-level benchmark would be of sufficient spatial extent relative to forest disturbance events such as fire. RAs can serve as benchmarks (i.e., controls) for adaptive management, but the utility of a RA as a benchmark is dependent on its size, composition, and spatial context in relation to the process(es) to be monitored. Beyond expanding the RAN such that the RAs collectively capture the range of forest ecosystems (e.g., representation of IP size structure and composition), regional benchmarks for adaptive management require identification of the questions and processes relevant to forest management in Saskatchewan. The example of benchmarks for caribou in Chapter 4 illustrates how RAs can be evaluated for specific monitoring objectives. While we emphasize the value of system-level benchmarks from a broader ecosystem perspective, the utility of existing RAs may be evaluated with less stringent criteria to meet narrower, specific objectives. A helpful next step would be for forest managers in Saskatchewan to identify the questions and processes they consider to be of high priority.

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Appendix A: Building Regional Planning Units for the Delineation of Ecological Benchmarks

Conservation planning is facilitated by dividing large "***Planning Areas***" into homogenous "***Regional Planning Units***". The "***Regional Planning Units***" enable finer spatial representation of natural variability and promote dispersion of land-use options. Accordingly, the strata to be used for landscape classification should reflect the characteristics that are intended to be represented by the planning process.

The Saskatchewan Representative Areas Network is intended to delineate areas that represent the range of ecological systems across the Province. Therefore, to define "***Regional Planning Units***", we used ecological characteristics to define the strata used for landscape classification. Moreover, because the ecosystems reflect the underlying ecological processes, we used ecological processes as the basis for delineating "***Regional Planning Units***".

To decide which ecological processes to use to delineate "***Regional Planning Units***", we used first principles of ecological processes that indicate that larger, slower ecological processes generally have greater influence on smaller, faster ecological processes. The reverse, however, rarely is true.

Ocean Drainage Areas

"***Ocean Drainage Areas***" are the areas of Canada that drain all precipitation or groundwater into a particular ocean. The boundary of a drainage basin is the ridge beyond which water flows in the opposite direction (Natural Resources Canada 2004). There are five "***Ocean Drainage Areas***" in Canada (Figure 1).

Figure 1: Ocean drainage areas of Canada.

(source:http://atlas.gc.ca/site/english/maps/freshwater/distribution/drainage).

Natural Resources Canada (2004) describes the five “***Ocean Drainage Areas***” as follows:

1. The **Pacific Ocean drainage area** drains the area west of the Rocky Mountains. The Fraser, Yukon and Columbia rivers are the largest rivers draining this region. It is separated from all other drainage areas by the **continental divide**. This is defined as the north-south line along the western Cordillera separating rivers flowing ultimately into the Pacific Ocean from those flowing into other oceans.

2. The **Arctic Ocean drainage area** is the area flowing directly into the Arctic Ocean or into the channels of the Arctic Islands. Hudson, James and Ungava bays are considered to be part of the Arctic Ocean, but for most purposes their drainage area is usually considered as a separate entity. The Mackenzie River dominates the Arctic Ocean drainage area.

3. The **Hudson Bay drainage area** is a huge area that captures about 30% of total Canadian runoff. Many of its river systems such as the Nelson and Churchill River (of Manitoba) drain eastwards from the continental divide to Hudson Bay. As well, many large rivers drain from the south and east into Hudson Bay or James Bay. The extensive area of drainage into Ungava Bay is also considered to be part of the Hudson Bay drainage area.

4. The **Atlantic Ocean drainage area** is dominated by the Great Lakes-St. Lawrence system but there are other significant drainage basins such as those of the Churchill River (of Labrador) and the Saint John River in New Brunswick.

5. **Gulf of Mexico drainage area**: a small portion of southern Alberta and Saskatchewan drains south into the Mississippi system which ultimately drains into the Gulf of Mexico.

   (Natural Resources Canada. 2004. Drainage patterns of Canada. Source: http://atlas.gc.ca/site/english/maps/freshwater/distribution/drainage/1)

These “***Ocean Drainage Areas***” delineate ecological patterns that reflect the effects of ecological processes such as plate tectonics and glaciation. These ecological processes function at the largest spatial extents and at the slowest time scales that are relevant to the way that humans interact with ecological systems (Holling 1986). The time and spatial continuum of ecological processes that Holling (1986) presents is relevant to the stratification of the “***Planning Area***” because larger, slower processes generally influence smaller, faster processes and underlie the ecological patterns that result. For example, the formation of mountains through plate tectonics will influence the weather; however, the weather does not influence how plate tectonics proceed.

(Holling, C.S. 1986. The resilience of terrestrial ecosystems; local surprise and global change. In: W.C. Clark and R.E. Munn (eds.). Sustainable Development of the Biosphere. Cambridge University Press, Cambridge, U.K. Chap. 10: 292-317.)

The “***Ocean Drainage Areas***” are distinct hydrological units because water flows remain within each drainage area. However, “***Ocean Drainage Areas***” link aquatic and terrestrial ecosystems because the drainage basin also defines the area of land that directs the flow of precipitation and ground water to a particular ocean. Drainage basins also are referred to as catchment basins or watersheds. Catchment basins are areas that draining to a lake, stream or measuring site (USGS 1978 in ECOMAP 1995). A watershed is the land enclosed by a continuous hydrologic-surface drainage divide and lying upslope from a specified point, in this case, upslope from an ocean.

<u>EcoZones</u>

"***Ecozones***" as defined by the National Ecologic Framework for Canada (Agriculture and Agri-Food Canada 1999) are generalized ecological units that are characterized by similar abiotic and biotic factors. There are seven "***Ecozones***" in the boreal region of Canada (Figure 2).

Figure 2: Ecozones of the boreal region of Canada

(source: http://sis.agr.gc.ca/cansis/nsdb/ecostrat/intro.html#ecological%20land%20classification)

"***Ecozones***" are part of a classification system that represents the cumulative effect of ecological processes such as soil formation, nutrient cycling, photosynthesis, and population dynamics. These ecological processes function at spatial and temporal scales that are the next order of magnitude smaller and faster than the ecological processes of plate tectonics and glaciation define the "***Ocean Drainage Areas***" (Holling 1986). "***Ecozones***" reflect the ecological patterns that distinguish broad regions of Canada.

Regional Planning Units

"***Regional Planning Units***" are delineated by the intersection of the "***Ocean Drainage Areas***" and the "***Ecozones***" (Figure 3). "***Regional Planning Units***" are ecosystems with similar patterns of hydrological flow and terrestrial composition at a spatial extent that is relevant for the planning of process-based system-level ecological benchmarks.

Figure 3: Regional Planning Units

Appendix B: Evaluating Stratification Units

Here, we assess the ability of intrinsic patch (IP) structures to discriminate between components of alternate stratifications of the study region (ecozones, regional planning units, and ecoregions). This is relevant to the evaluation of Saskatchewan's RAN as benchmarks and to the identification of candidate benchmark areas for forest management. Stratification for these purposes is intended to partition the planning region into ecologically relevant units to facilitate the spatial representation of the natural variability of ecological systems. We employ qualitative comparisons and univariate and multivariate inferential statistics in this assessment to determine if IPs can discriminate between or among:

1. Boreal Plain and Boreal Shield
2. RPU 5 and RPU 10 within the Boreal Plain
3. Ecoregions 139, 148, and 149 within RPU 10

Boreal Plains and Boreal Shield

The study area intersects 2 ecozones from the Ecological Land Classification of Canada (Marshall and Schut 1999): the boreal plain and boreal shield (Figure 1). Ecozones were defined based on geologic, landform, soil, vegetative, climatic, wildlife, water, and human factors (Marshall and Schut 1999). If IPs are ecologically meaningful with respect to ecozones, we should be able to discriminate the ecozones based on the composition and size structure of IPs.

Figure 1: Ecozones in the study area.

<u>Ecozone compositions</u>

The composition of IPs between ecozones is qualitatively compared by examining the relative total proportion of intrinsic patch types in the each ecozone (Figures 2 and 3). Based on the full composition of IP types (Figure 2), the boreal plain has a greater proportion of MIX, BOG and RIPARIAN patch types, while the boreal shield has a greater proportion of UPCON and WATER.

The relative proportions of the terrestrial subcomponents (UPCON, MIX, and BOG) are described in Figure 3. In the boreal shield, the proportion of UPCON is approximately 2 times greater than in the boreal plains. The proportion of MIX and BOG in the boreal plains are approximately 2 and 1.5 fold greater than in the boreal shield.

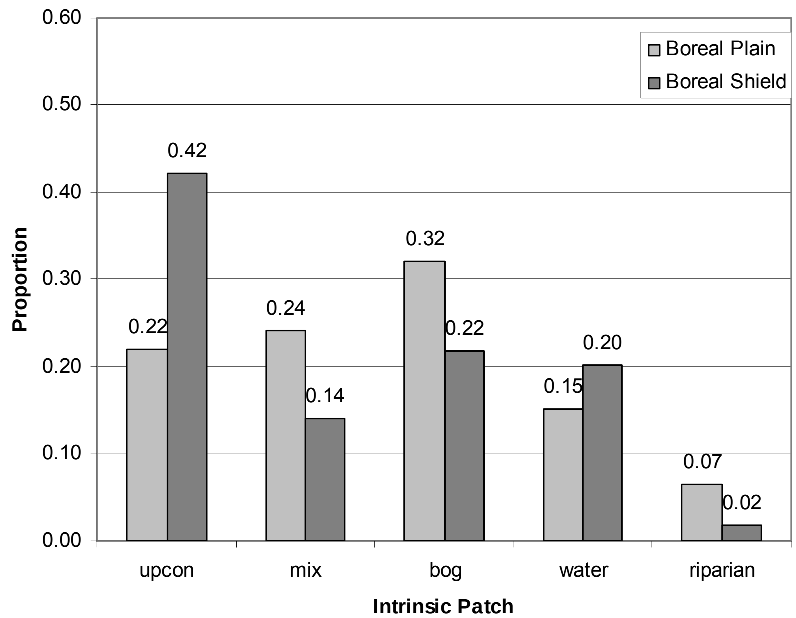

Figure 2: Proportion of Intrinsic Patch types (UPCON, MIX, BOG, WATER, and RIPARIAN) within the study region stratified by Ecozone.

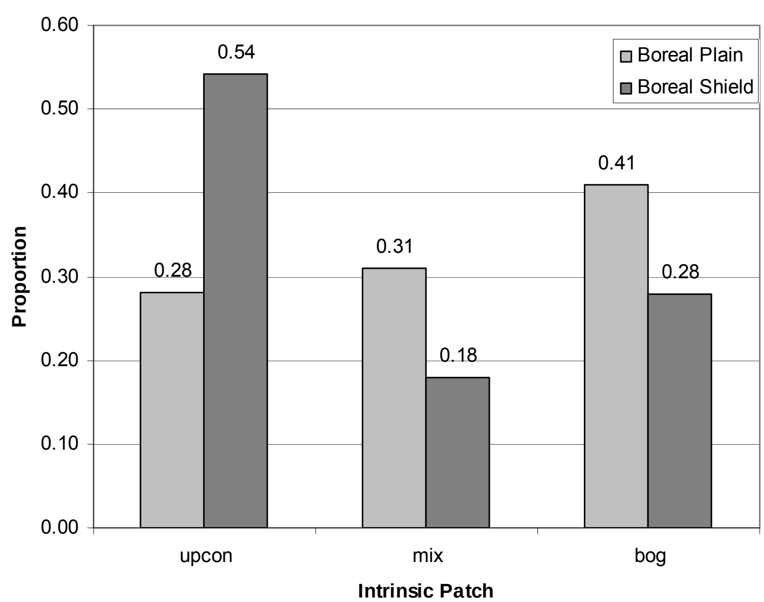

Figure 3: Proportion of Intrinsic Patch types (UPCON, MIX, and BOG) within the study region stratified by Ecozone.

Multivariate Comparison of the IP Terrestrial Subcomposition

Using enduring features to aggregate intrinsic patches into landscape units, we are able to derive a multivariate distribution of landscape IP subcompositions (UPCON, MIX, and BOG) for the boreal shield and boreal plain.[12] These 3-dimensional multivariate distributions can be illustrated in 2-dimensions (Figure 4) by plotting logratios with a common denominator on the Y- and X-axis, BOG/UPCON and MIX/UPCON, respectively. In Figure 4, the green points are landscape units (i.e., enduring features) located in the boreal plain. The red points are landscape units in the boreal shield. Each landscape unit is comprised of some combination of proportions of BOG, MIX, and UPCON which sum to 1.

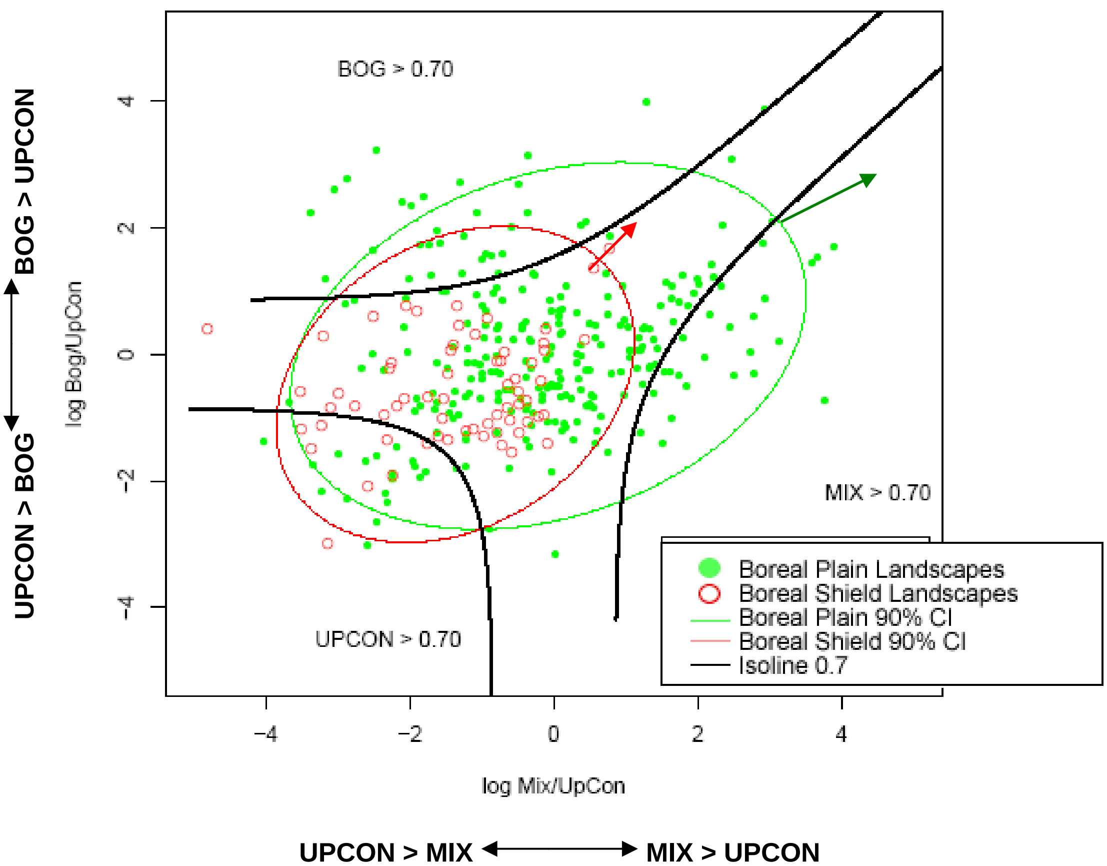

Figure 4: Terrestrial subcompositions (UPCON, BOG, and MIX) for landscape units (i.e., enduring features) within the study region stratified by Ecozone with, a) 90% confidence interval ellipsoids, and b) isolines for proportion of MIX, BOG, and UPCON

[12] Within the study region, the boreal shield and boreal plain intersected 99 and 421 landscape units, respectively. Of these, 66 and 271 were retained for analysis, accounting for 89.6% of the total area.

The axes of Figure 4 are natural log scales:

log(x), where x = the log ratio of proportions of BOG (or MIX)/UPCON.

At the Y-axis value of 0, the proportion of BOG equals UPCON (*i.e.*, BOG/UPCON = 0.3/0.3 = 1, such that log(1) = 0). For Y-values > 0, the proportion of BOG is greater than UPCON. For Y-values < 0, the proportion of UPCON is less than BOG. The same principle applies to the X-axis. Interpreting the axes together, the interception point of (0,0) occurs when BOG, UPCON and MIX are of equal proportion in the landscape unit (i.e., BOG = UPCON = MIX = 0.33).

Isolines (Figure 4) are guideposts for interpretation and can illustrate meaningful transitions in the data. In Figure 4, the isolines represent proportions of 0.70. The upper most isoline represents all points on the graph were the proportion of BOG is 0.70. Above the isoline, the proportion of BOG is > 0.70. Below the isoline, the proportion of BOG is < 0.70. Isolines for UPCON and MIX (the lower left and right, respectively) can be interpreted in the same fashion.

The ellipses represent the 90% confidence intervals centred on the mean of the distribution of the proportion of IP types in the landscape units within each ecozone. The shape and direction of the ellipse describes the distribution for each ecozone. The ellipse of the boreal plain (green) is much larger and captures a greater variety of landscape unit compositions. The boreal shield is almost entirely captured within the boreal plain (green). This indicates that the two ecozones are very similar with regard to landscape units dominated by BOG and UPCON. However, the boreal plain (green) is markedly different from the boreal shield (red) in that it has considerably more landscape units comprised of MIX proportions >0.33.

Despite the spatial overlap of the distributions, the two distributions are distinct (p<0.001). Both their covariance structures (p=0.002) and means differ (p<0.001). The mean compositions of the boreal shield and boreal plain are (-1.371, -0.479) and (-0.082, 0.129), respectively. These two points are the centroids of the ellipses in Figure 4. The corresponding landscape mean crude proportions of UPCON, MIX and BOG are shown in Table 1. This analysis confirms that the boreal shield landscapes contain significantly higher proportions of UPCON and significantly lower proportions of MIX, relative to the boreal plains.

Table 1: The landscape mean crude proportions of UPCON, MIX, and BOG for the boreal shield and boreal plain. Note: These values differ from the absolute proportions (Figure 3) because the means are not area-weighted).

Ecozone	**Intrinsic Patch Type**		
	UPCON	MIX	BOG
Boreal shield	0.534	0.136	0.331
Boreal plains	0.327	0.301	0.372

Discriminating the Boreal Plain and Boreal Shield

Using linear discriminant analysis, the terrestrial subcomposition covariates distinguish the two Ecozones with classification accuracies for the boreal shield and boreal plain of 18.2% and 95.9%, respectively. Using the full composition (i.e., all 5 IP types), the overall classification accuracy for the boreal shield and boreal plain improves to 56.7% and 94.1%, respectively. The interpretation is that each ecozone contains one or more groups of landscapes with characteristic structures (as measured by IP compositions) that are poorly represented in the other.

Comparison of Intrinsic Patch Size

K-S tests show that the distribution of patch sizes for IP classes (UPCON, MIX and BOG) are significantly different ($p<0.05$) between ecozones (Table 1), such that the mean patch size of UPCON is 2-fold greater in the boreal shield while MIX and BOG are more than 2-fold greater on the boreal plain.

Table 2: Kolmogorov-Smirnov test for the Boreal Shield and Boreal Plain for Intrinsic Patch types (UPCON, MIX, and BOG) and the mean patch size for patches >0.5 ha

Intrinsic Patch Type	**K-S test**		**Mean Patch Size (ha) (for patch sizes > 0.5 ha)**	
	D statistic	p-value	Boreal shield	Boreal plains
UPCON	0.065	0.000	48.9	24.1
MIX	0.064	0.000	12.8	39.7
BOG	0.064	0.000	16.6	38.0

Regional Planning Units (RPU 5 and 10)

There are 4 regional planning units that intersect the study area (Figure 5). Because of data limitations, the comparison of intrinsic patch composition and size between RPUs was restricted to the regions of RPU 5 and 10 in the study area. RPU 5 and 10 form the boreal plain in the study area.

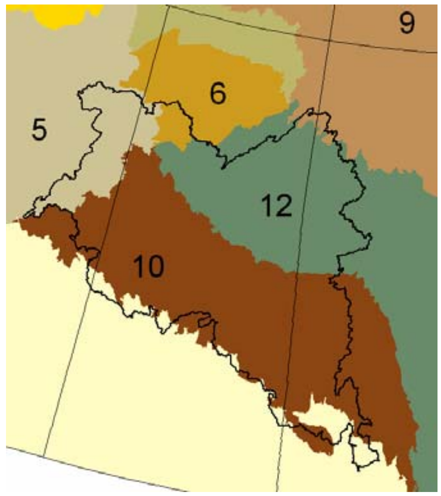

Figure 5: RPU 5 and 10 in the study area.

<u>RPU Compositions</u>

The relative total proportion of IP types in the each ecozone is qualitatively compared in Figures 6 and 7. Based on the full composition of IP types (Figure 6), RPU 10 is wetter than RPU 5 with greater proportions of BOG and Water patch types accounting for 0.51 of the planning unit. RPU 5 is drier with greater proportions of UPCON and MIX accounting for 0.64 of the planning unit.

The relative proportions of the terrestrial subcomponents (UPCON, MIX, and BOG) are described in Figure 7. RPU 5 and 10 have similar proportions of MIX. RPU 5 and 10 are characterized by greater proportions of UPCON and BOG, respectively.

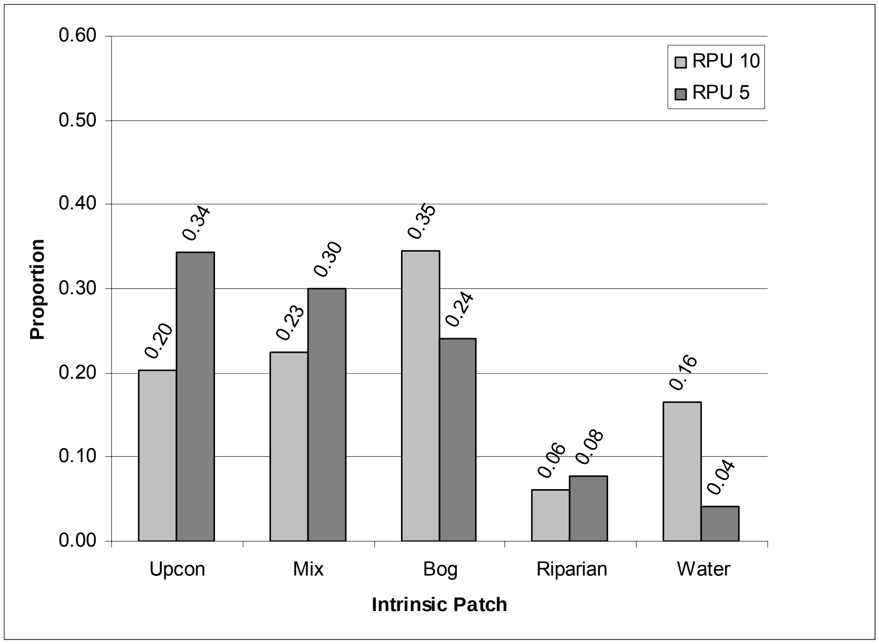

Figure 6: Proportion of Intrinsic Patch types (UPCON, MIX, BOG, WATER, and RIPARIAN) within the study region stratified by Regional Planning Unit.

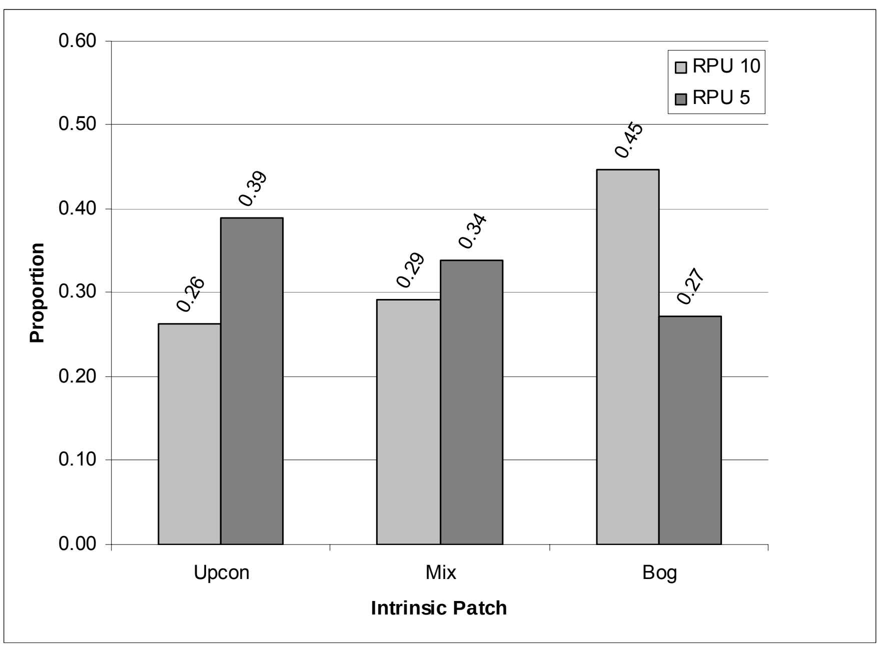

Figure 7: Proportion of Intrinsic Patch types (UPCON, MIX, and BOG) within the study region stratified by Regional Planning Unit.

Multivariate Comparison of the IP Terrestrial Subcomposition

The multivariate distribution of landscape IP terrestrial subcompositions for RPU 5 (N=31 landscape units) and RPU 10 (N=240 landscape units) are illustrated in Figure 8. The distributions of subcompositions for the two RPUs were not identical (p=0.009). The mean compositions for RPU 5 (-0.154,-0.490) and RPU 10 (-0.073,0.209) differ (p=0.004) The covariance structures do not differ (p=0.058). The means, expressed as crude proportions in Table 3, indicate that landscapes in RPU5 contain significantly higher proportions of IP classes UPCON and MIX, relative to landscapes in RPU 10. Notably, few RPU 5 landscapes contain more than a 1/3 BOG (Figure 8).

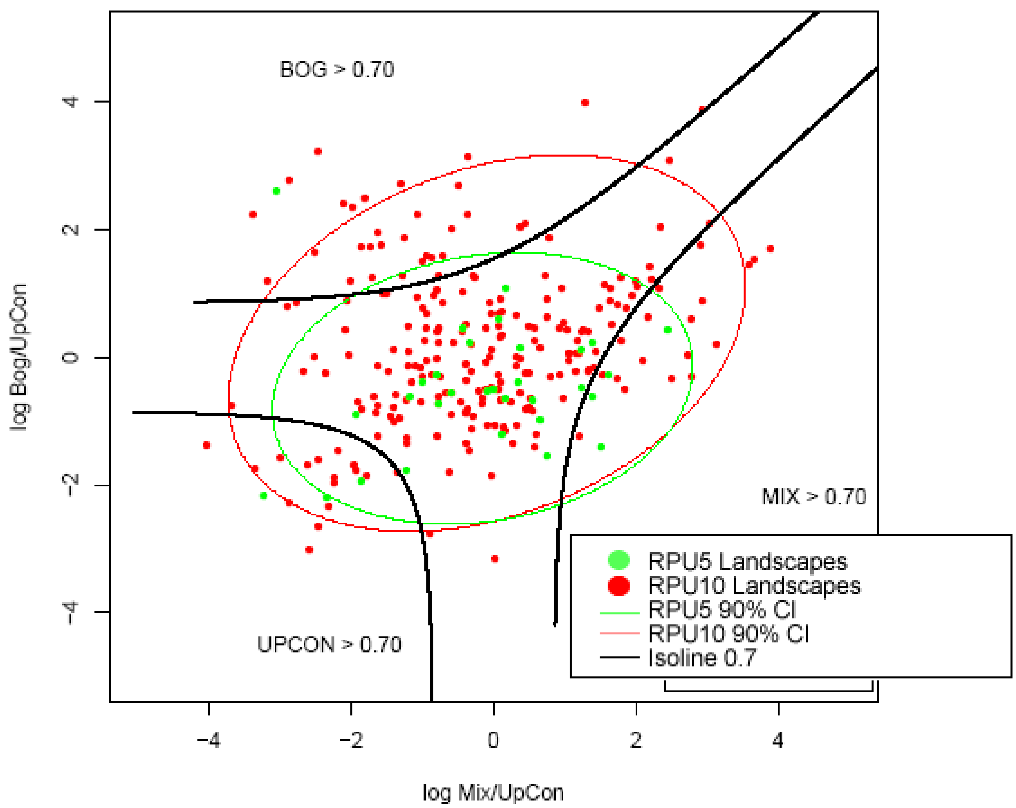

Figure 8: Terrestrial subcompositions for landscapes (i.e., enduring features) within the study area stratified by Regional Planning Units 5 and 10 with 90% confidence interval ellipsoids and isolines for proportion of MIX, BOG, and UPCON.

Table 3: The landscape mean crude proportions of UPCON, MIX, and BOG for RPU 5 and 10. Note: These values differ from the absolute proportions (Figure 7) because the means are not area-weighted).

RPU	Intrinsic Patch Type		
	UPCON	MIX	BOG
RPU 5	0.405	0.347	0.248
RPU 10	0.316	0.294	0.390

Discriminating RPUs 5 and 10

Using linear discriminant analysis, the terrestrial subcomposition covariates distinguish the two RPUs with classification accuracies for RPU 5 and RPU 10 of 0.0% and 99.6%, respectively. Using the full composition, the classification accuracy for RPU 5 and RPU 10 improves slightly to 6.5% and 99.6%, respectively. Interpretation is that although the landscapes in the two RPUs are not drawn from the same population, RPU 5 contains no characteristic group of landscapes not well represented in RPU 10. However, RPU 10 contains many landscape types not found in RPU 5 as shown in Figure 8.

Comparison of Intrinsic Patch Size

K-S tests show that the distribution of patch sizes for IP classes are statistically different ($p<0.05$) between RPU 5 and 10 for UPCON, MIX, and BOG (Table 4), such that the mean patch sizes of MIX and BOG are larger in RPU 10, and the mean patch size of UPCON is marginally larger in RPU 5.

Table 4: Kolmogorov-Smirnov test for RPU 5 and 10 for Intrinsic Patch types UPCON, MIX, and BOG, and the mean patch size for patches >0.5 ha

Intrinsic Patch Type	**K-S test**		**Mean Patch Size (ha) (for patch sizes > 0.5 ha)**	
	D statistic	p-value	RPU 5	RPU 10
UPCON	0.07862	0.00000	26.32	23.44
MIX	0.10361	0.00000	33.06	41.05
BOG	0.07535	0.00000	23.16	41.33

Ecoregions

The comparison of intrinsic patch composition and size amongst ecoregions was restricted to the ecoregions within RPU 10 (Figure 9). There were too few ecoregions for comparisons in the remaining RPUs. There are 3 major ecoregions that fall within RPU 10: 139, 148, and 149 (Figure 9).

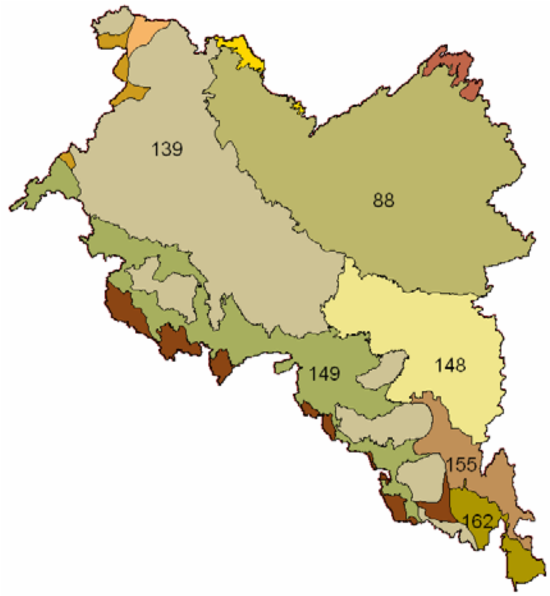

Figure 9: Ecoregions in the study area.

Ecoregion Composition

The relative total proportion of intrinsic patch types in ecoregions 139, 148, and 149 is qualitatively compared in Figures 10 and 11. Based on the full composition of IP types (Figure 10), the ecoregions have similar proportions of RIPARIAN. Ecoregion 139 is characterized by similar proportions UPCON, MIX and BOG with much lesser amounts of RIPARIAN and WATER. Ecoregion 148 is very wet with a high proportion of BOG and WATER accounting for 0.70, and terrestrial patches (UPCON and MIX) account for only 0.22. Ecoregion 149 is much drier with UPCON and MIX accounting for 0.63.

The relative proportions of the terrestrial subcomponents are described in Figure 11. Ecoregion 139 has 1/3 more UPCON than Ecoregion 148 and 149. Ecoregion 149 has 2-4 times more MIX than 139 and 148, respectively. Ecoregion 148 has approximately 2-3 times more of its area composed of BOG.

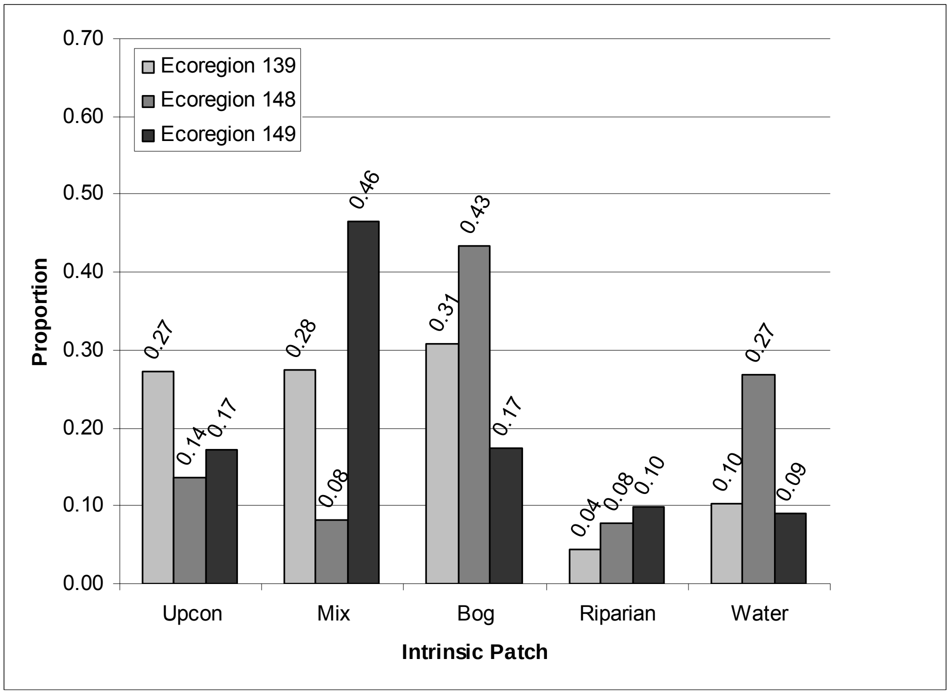

Figure 10: Proportion of Intrinsic Patch types (UPCON, MIX, BOG, WATER, and RIPARIAN) within the study region stratified by ecoregion.

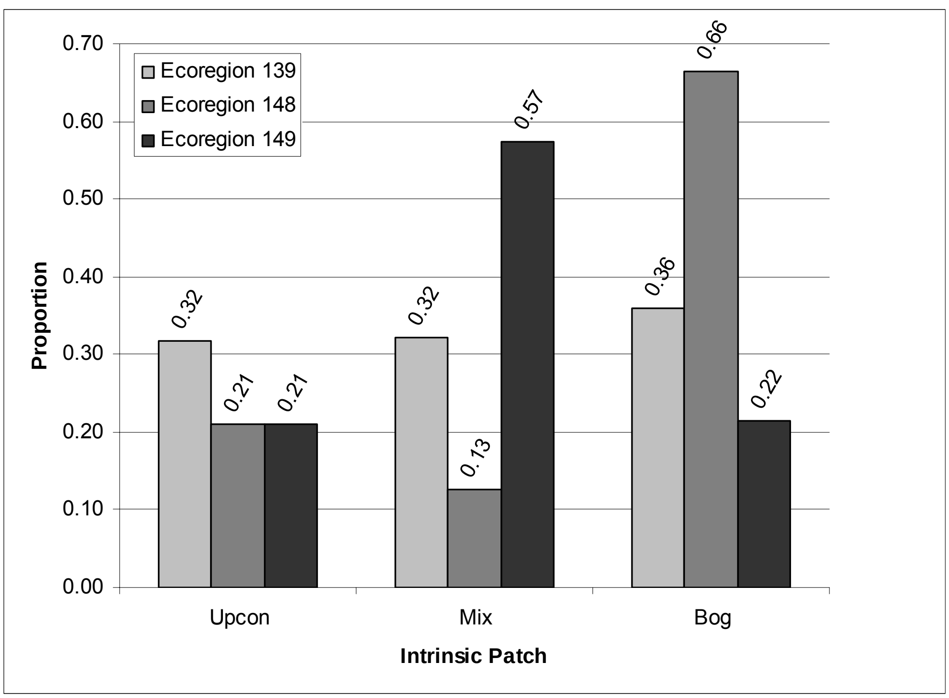

Figure 11: Proportion of Intrinsic Patch types (UPCON, MIX, and BOG) within the study region stratified by ecoregion.

Multivariate Comparison of the IP Terrestrial Subcomposition

The multivariate distribution of landscape IP terrestrial subcompositions for ecoregions 139 (N=145 landscape units), 148 (N=52 landscape units), and 149 (N=30 landscape units) are illustrated in Figure 12. Ecoregion 139 (red) is almost entirely contained within ecoregions 148 or 149. The landscapes of ecoregion 149 are predominately MIX. Ecoregion 148 has landscapes with higher proportions of BOG. All three distributions are bivariate normal. The mean subcompositions for ecoregions 139, 148, and 149 were (-0.500,0.051), (-0.090,0.695), and (1.429, 0.009), respectively. The mean subcompositions of ecoregions 139 and 148 did not differ (p=0.079). Both differed from ecoregion 149 with respect to covariances and means (all tests $p < 0.001$). The means as crude proportions are given in Table 5.

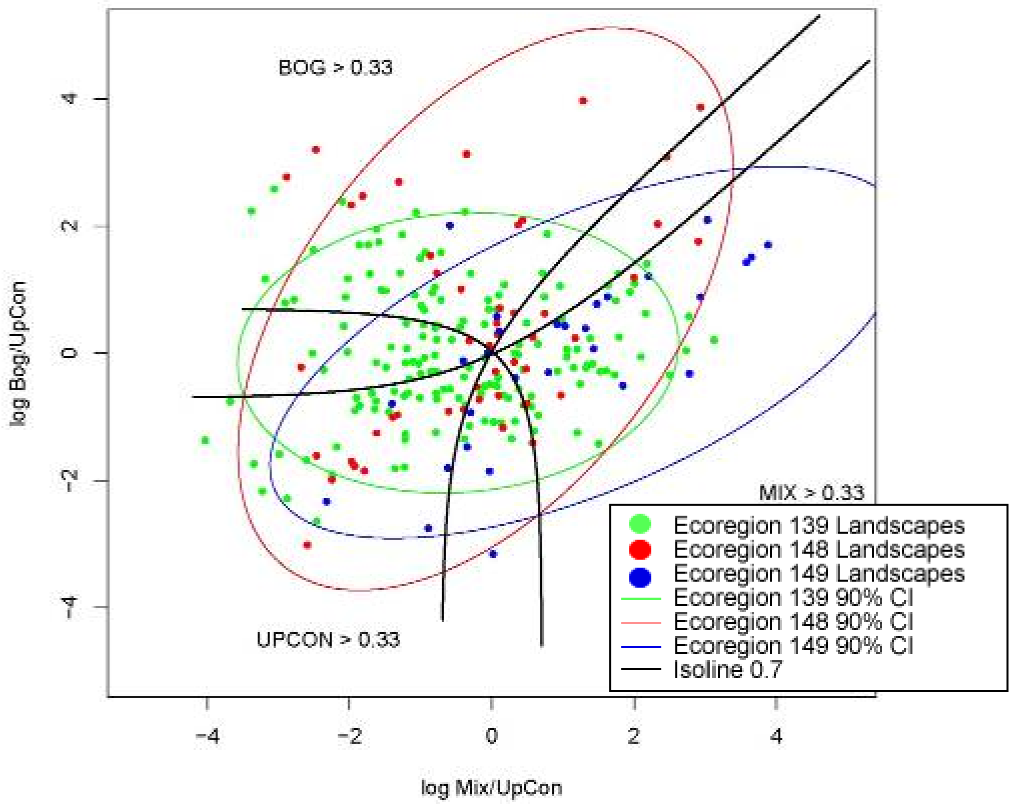

Figure 12: Terrestrial subcompositions for ecoregions within RPU 10 with 90% confidence interval ellipsoids for proportions of MIX, BOG, and UPCON.

Table 5: The landscape mean crude proportions of UPCON, MIX, and BOG for ecoregions 139, 148, and 149. Note: These values differ from the absolute proportions (Figure 7) because the means are not area-weighted.

RPU	Intrinsic Patch Type		
	UPCON	MIX	BOG
139	0.376	0.228	0.396
148	0.255	0.233	0.511
149	0.162	0.675	0.163

Discriminating Ecoregions

Using linear discriminant analysis, terrestrial subcomposition covariates had classification accuracies for ecoregions 139, 148, and 149 of 92.4%, 15.4%, and 33.3 %, respectively. Using the full composition, the classification accuracy of ecoregions 139, 148, and 149 were 96.6%, 15.4%, and 20.0%, respectively. Most landscapes within ecoregions 138 and 149 were classified as belonging to ecoregion 139.

Comparison of Intrinsic Patch Size

K-S tests and T-tests show that the size distributions of the terrestrial intrinsic patch types are significantly different ($p<0.05$) when compared between ecoregions (Tables 6-8). For patch types UPCON and BOG, the mean patch size from largest to smallest is ecoregions 148 > 139 > 149. For MIX, the largest mean patch size is the most southern ecoregion, 149 > 139 > 148.

Table 6: Kolmogorov-Smirnov test for differences in size distributions between ecoregions 139 and 148 for Intrinsic Patch types UPCON, MIX, and BOG, and the mean patch size for patches >0.5 ha

Intrinsic Patch Type	**K-S test**		**Mean Patch Size (ha) (for patch sizes > 0.5 ha)**	
	D statistic	p-value	Ecoregion 139	Ecoregion 148
UPCON	0.10792	0.00000	22.65	**31.86**
MIX	0.11237	0.00000	**40.86**	30.12
BOG	0.04818	0.00000	26.58	**152.60**

Table 7: Kolmogorov-Smirnov test for differences in size distributions between ecoregions 139 and 149 for Intrinsic Patch types UPCON, MIX, and BOG, and the mean patch size for patches >0.5 ha

Intrinsic Patch Type	**K-S test**		**Mean Patch Size (ha) (for patch sizes > 0.5 ha)**	
	D statistic	p-value	Ecoregion 139	Ecoregion 149
UPCON	0.04146	0.00000	**22.65**	17.00
MIX	0.03748	0.00000	40.86	**41.11**
BOG	0.02790	0.00000	**26.58**	14.16

Table 8: Kolmogorov-Smirnov test for differences in size distributions between ecoregions 148 and 149 for Intrinsic Patch types UPCON, MIX, and BOG, and the mean patch size for patches >0.5 ha

Intrinsic Patch Type	**K-S test**		**Mean Patch Size (ha) (for patch sizes > 0.5 ha)**	
	D statistic	p-value	Ecoregion 148	Ecoregion 149
UPCON	0.14207	0.00000	**31.86**	17.00
MIX	0.14177	0.00000	30.12	**41.11**
BOG	0.06217	0.00000	**152.60**	14.16

Summary and Discussion

The findings can be summarized and interpreted as follows,

1. Based on the qualitative comparison of IP compositions between stratification units, Ecozones had the greatest difference in IP proportions. Based on this finding, the boreal shield and boreal plains are more different from each other than RPU 5 is from RPU 10. Ecoregion compositions are highly variable within RPUs.

2. The multivariate distributions of IP subcompositions and compositions were statistically different for Ecozones and RPUs, but not for Ecoregions. Based on this finding, Ecozones and RPUs, but not Ecoregions, describe unique populations or compositions that are rare everywhere else. The similar findings for Ecozones and RPUs were expected because the RPUs are basically the boreal plain and boreal shield in Saskatchewan.

3. The size distributions of the terrestrial IP types are significantly different between/among Ecozones, RPUs, and Ecoregions. In other words, the strata are capturing differences in IP size structure.

4. For Ecozones and RPUs, the full composition of IPs did a better job of distinguishing the stratification units. Neither the subcomposition nor full composition of intrinsic patch types could distinguish the ecoregions. Therefore, Ecoregions are not suitable for stratification. Overall, Ecozones had the best classification accuracy. Because RPU 5 and 10 are basically the boreal shield and boreal plains in Saskatchewan, one explanation for the poorer discriminatory ability of the IP covariates for RPU 5 and 10 is that only a small region of RPU 5 was used in this analysis. The region of RPU 5 used in the analysis borders RPU 10 where one would expect the greatest similarity between the two RPUs. If a larger proportion of RPU 5 had been included in the analysis, we predict that the classification accuracy would improve which justifies our use of RPUs.

References

Marshall, I. B. and P.H. Schut. 1999. A National Ecological Framework for Canada. Ecosystems Science Directorate, Environment Canada and Research Branch, Agriculture and Agri-Food Canada. pp http://sis.agr.gc.ca/cansis/nsdb/ecostrat/intro.html

Appendix C: Association of Intrinsic Patch and Enduring Feature Type

Here, we explore the relationships between the Enduring Feature attributes and the terrestrial IP subcompositions (UPCON, MIX, BOG).

Enduring features were derived from two sources: 1) NTS 1:250,000 source-tiled version of the Soil Landscape Units (v2.1) (Eco_SK as derived by Saskatchewan, Wright and Beveridge 1998), 2) 1:1,000,000 Soil Landscape Units (v2.2). For both sources, enduring features (EFs) were defined using four Soil Landscapes of Canada (SLC) attribute layers: soil development[13] (DEV), local surface form[14] (LOC), parent material mode of deposition[15] (PMD) and slope[16] (SLO). EFs for Saskatchewan were defined using Eco_SK and Alberta and Manitoba were defined using SLC2.2.

We used 271 EFs within the Boreal Plain that were selected by our standard criteria; thus, all were at least 5,000 ha in size with limited proportions of water, disturbed, anthropic or unclassifed areas. We selected by ecozone rather than RPU because no EFs are split across ecozone boundaries.

The 271 EFs included 86 distinct combinations of the four SLC attributes, of which 40 occurred only once. The median occurrence frequency was 2.0, the mean was 3.151 and the maximum was 29. Of the four attributes, DEV was the most variable.

There are 36 classes of DEV in the Soil Landscapes of Canada[17], 19 of which occur in Saskatchewan[18]. The three most abundant DEV classes were F (Gray Luvisolic, n=102), Y (Mesisol, n=54) and M (Eutric Brunisol, n=45) (Table 1).

Table 1: Description of the three most abundant DEV clases in Saskatchewan

DEV Class	Description[19]
F	Gray Luvisolic - Soils occurring in moderately cool climates and developed under deciduous and coniferous forest cover
Y	Mesisol - Soils that are saturated for most of the year.
M	Eutric Brunisol – Brunisol soils "occur under a wide variety of climatic and vegetative conditions."

[13] "Soil development is how soils were formed through various factors like climate, soil organisms, the nature of the parent material, the topography of an area, and time." (SERM 1999)
[14] "Physical landscape features such as eskers and potholes." (SERM 1999)
[15] "This relates to the method by which material such as soil, gravel or rocks was deposited (i.e. wind, water, glacial melt water)." (SERM 1999)
[16] "Slope refers to the steepness or grade of the surface terrain." (SERM 1999)
[17] http://sis.agr.gc.ca/cansis/nsdb/slc/v2.0/component/devel.html
[18] Classes A-H, J-M, O, P, R, T, U, X, and Y
[19] For full definition visit http://sis.agr.gc.ca/cansis/glossary/index.html

DEV classes, F, Y, and M, accounted for 201/271 (74%) of EFs in the sample. The group subcompositions of F, Y, and M are shown in Figure 1. The distributions of F and M were bivariate normal. There was some evidence of nonnormality within Y. Pairwise tests showed the three distributions to be all distinct (p<0.001), as Figure 1 suggests. The group mean subcompositions, expressed as crude proportions, are given in Table 2.

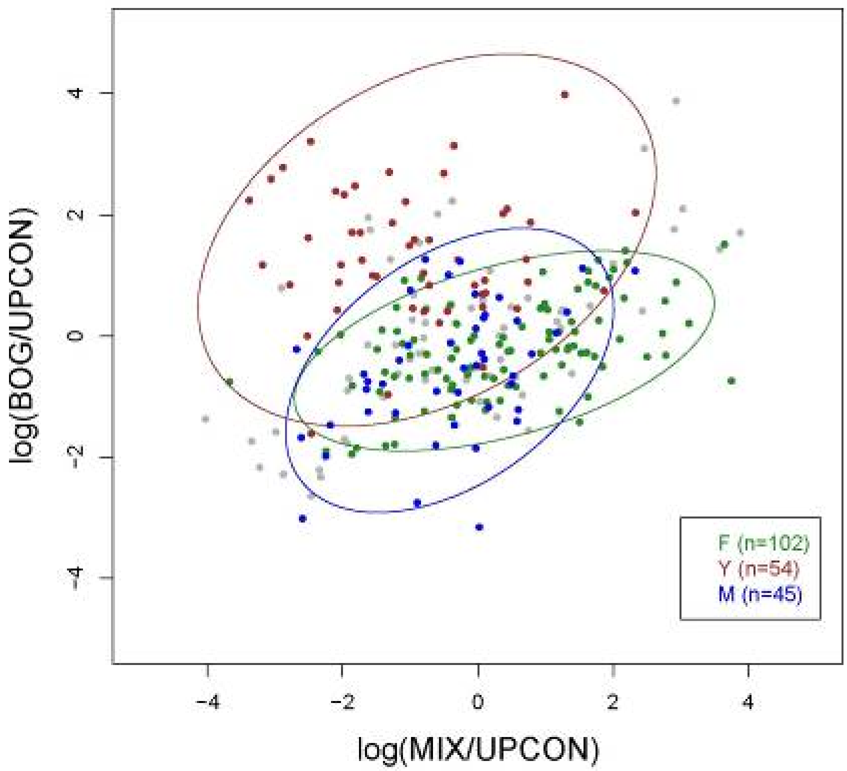

Figure 1: The intrinsic patch subcompositions (BOG, UPCON, MIX) of DEV classes F, Y, and M with 90% confidence intervals.

Table 2: The group mean subcomposition of the DEV Class distributions depicted in Figure 1.

	Intrinsic Patch Type		
DEV Class	UPCON	MIX	BOG
F	0.307	0.453	0.240
Y	0.158	0.075	0.767
M	0.449	0.296	0.254

The EFs in DEV class Y are preponderantly BOG as expected (Figure 1). The compositions of groups F and M differ mainly in the relative abundances of UPCON and MIX, with MIX predominating in group F. However, the 90% confidence intervals overlap considerably, especially between groups F and M. This suggests that the latter two groups are poorly distinguished by IP subcomposition. We assessed this using linear discriminant analysis with covariates (log MIX/UPCON, log BOG/UPCON, lnA). The correct classification rates for groups F, Y and M were 0.63, 0.88 and 0.11, respectively. The respective predictive accuracies were 0.64, 0.80 and 0.33. As expected, group M is not distinguished from F. Groups F and Y are much better separated, but their between group error rates range from 0.04 to 0.28.

Among SLC attribute pairs (comprised of combinations of DEV, LOC, PMD, and SLO), the most variable appears to be DEV and LOC. Accordingly, we examined variation in IP subcomposition within DEV class F in relation to LOC (local surface form). The three most abundant LOC attribute values were K (Knoll and Kettle, n=29), D (Dissected, n=26) and H (Hummocky, n=21), accounting for 76/102 (75%) of EFs within DEV class F. Their IP sub-compositions are shown in Figure 2; the 90% CIs overlap almost completely. All three distributions were bivariate-normal. The distributions of LOC classes K and D did not differ. The distributions of K and H were not identical (p=0.046) due to a significant difference in their means (p=0.028). The distributions D and H were not identical (p=0.026); their means were the same but their covariance structures differed, as suggested in Figure 2. The group mean subcompositions, expressed as crude proportions, are given in Table 3.

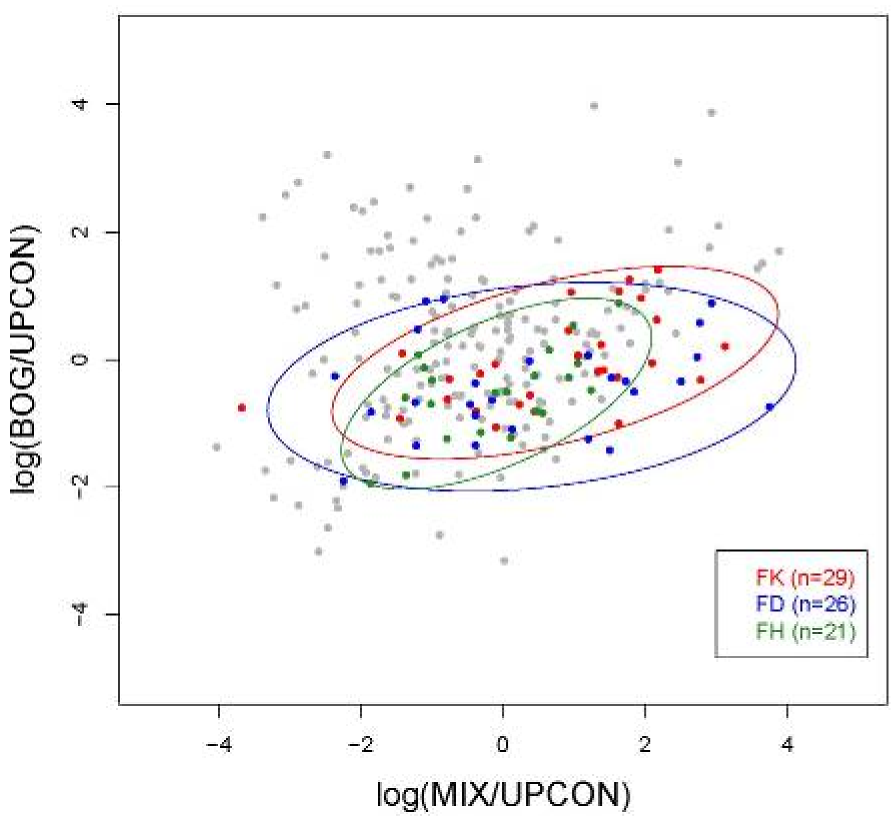

Figure 2: The intrinsic patch subcompositions (BOG, UPCON, MIX) of DEV classes F, Y, and M with 90% confidence intervals.

Table 3: The group mean subcomposition within DEV class F in relation to local surface form (LOC) attributes K, D, and H as depicted in Figure 2.

	Intrinsic Patch Type		
LOC Attribute	UPCON	MIX	BOG
K	0.248	0.516	0.236
D	0.318	0.475	0.207
H	0.400	0.365	0.234

In groups K and D, the predominant IP class is MIX. Group H differs in that MIX is less abundant while the proportions of MIX and UPCON are nearly the same. Linear discriminant analysis prediction accuracies ranged from 0.38 to 0.45 and classification accuracies ranged from 0.31 to 0.59 (for group K). In other words, the ability of IP subcompositions and EF area to discriminate between the dominant subgroups of soil development class F is little better than random.

Although much further work is needed, these preliminary results indicate that enduring features do not correspond to well-defined clusters of landscape structural characteristics, as measured by IP subcompositions. We emphasize that it does not follow that EF attributes have no distinct ecological meaning. For example, additional covariates, including but not limited to more refined IP classification and measures of within-EF IP size structure, might yield better discrimination and better concordance between measured vegetation patch structure and SLC classes. However, we believe these results show: 1) the use of EFs as such, independent of variation in their intrinsic patch structures and other attributes, as fundamental spatial units of representation requires further empirical support; and 2) in comparison to SLC attributes, IP composition provides a more sensitive measure of spatial variation in landscape structure attributes that are relevant to forest management.

References

Wright. R.A., E. Beveridge, G. Freif, and O. Naelapea. 1998. Procedure for identifying candidate RAs in the forest, based on ecological criteria. Saskatchewan Environment and Resources Management.

Saskatchewan Environment and Resource Management (SERM). 1999. Amisk Lake Representative Area: Concept Management Plan. http://www.se.gov.sk.ca/ecosystem/sran/

Appendix D: Criteria for Selecting Landscapes

Landscapes were used in the composition analysis if they met the following five criteria:

1. The total area of the landscape was ≥ 5,000 ha. Analysis using landscapes smaller than 5,000 ha would yield inaccurate estimates of composition. The total area was calculated using all IP types.
2. The proportion of unclassified area was ≤ 0.25 the total area.
3. The proportion of IP type Anthropogenic (see Appendix A) was ≤ 0.10 the total area.
4. The proportion of IP type Water was ≤ 0.30 the total area.
5. The proportion of IP type Burn was ≤ 0.40 the total area.

For the composition analysis IP types ROCK and SAND were added to IP UPCON. Cut-off values for inclusion of landscapes into the sample were decided by considering the tradeoff between loss of sample size and total area of the selected landscapes. No IPs were split between RAN and NONRAN. IPs were assigned to landscapes by a majority area rule.

Appendix E: Development of Intrinsic Patch from Vegetation Inventories

The following 4 steps were taken to classify the landscape into intrinsic patches.

Step 1: Data collection

Unless otherwise stated all data was stored in ArcInfo Coverages using an Albers projection defined for Saskatchewan with a NAD83 datum. All datum conversions used the NTv2 74 adjustment for ArcInfo, with the exception of inventory data which was converted using NTv1.

Forest inventory maps for Saskatchewan were provided by Saskatchewan Environment. Al-Pac provided Alberta Vegetation Inventory (AVI) for the portion of the study area in Alberta. Forest inventory for Manitoba was downloaded from the Manitoba Land Initiative Website http://web2.gov.mb.ca/mli/forestry/index.html.

Step 2: Classified the landscape into IPs based on forest inventories

The conversion of the different forest inventories to IP classes varied slightly. A detailed description of the ArcView programming code to convert the stand types is given in below.

Step 3: Pre-processing IPs

The study area was divided into 4 subareas. For each subarea, the polygon coverages were converted into a 30 m x 30 m grid of IP classes. The grids were then joined to form a continuous grid within each subarea. The continuous grid was then converted back to a polygon coverage. As a result of the polygon-to-grid-to-polygon process some polygons were distorted which affected the estimated area of individual IPs. Table 1 shows the average percent change in patch area for 6 categories of IP patch size. Another consequence of the polygon-to-grid-to-polygon conversion was the creation of small and isolated artefact polygons ≤ 0.5 ha[20]. Prior to analysis, we eliminated the artefact polygons. With the exception of the RIPARIAN IPs, the elimination of the artefact polygons had a negligible effect on total area and proportion estimates. For total area of UPCON, MIX, BOG, and RIPARIAN, the percentage of artefact polygons was 1.50, 1.54, 2.09, and 7.61%, respectively. Riparian zones are characterized by narrow bands along lakes and rivers. Likely, these zones require a raster resolution finer than 30 m. We considered the loss of resolution in estimating RIPARIAN area and inclusion of small RIPARIAN patches acceptable.

[20] Note: 1) the elimination of polygons ≤ 0.5 ha was after dominance calculation and 2) the minimum mapping unit of most 1:20,000 inventories is ~ 0.5 ha.

Table 1: Percent change in patch size for 5 size categories of IP patches for initial polygon area and final polygon area after grid conversion. * 1st and 3rd quartiles for IP size distribution.

Patch size (ha)	Average change in Area (%)
<= 2	11.45
> 2 and <= 5	4.00
> 5 and <= 10	2.39
> 10 and <= 1000	1.20
> 1000	0.12
> 3 and < 14*	2.59

Step 4: Intersected IPs and data layers

IPs were intersected with 9 data layers

1) Saskatchewan's Enduring Features (Wright and Beveridge 1998) 1:250,000
2) Enduring Features from the Soil Landscapes of Canada Version 2.1, 1:1,000,000
3) Ocean Drainage Areas (Watershed Survey of Canada), 1:10,000,000
4) Saskatchewan Timber Supply Areas, 1:250,000
5) Global Forest Watch for Forest Tenures in Alberta and Manitoba, 1:250,000 (Lee et al. 2003)
6) Caribou Critical Habitat, 1:1,000,000 (Arsenault 2003)
7) GFW Canada's Large Intact Forest Landscapes, 1:1,000,000
8) Representative's Area Network for Saskatchewan, Alberta and Manitoba[21]

If an IP patch crossed more than one polygon from an intersected coverage, the dominant polygon (by area), was designated to the IP patch. Therefore, in theory, the IP could cross multiple polygons, but the IP would only be attributed to the dominant polygon. Overall, very few occurrences of dominance were attributed to polygons where the proportion of IP was < 60%. In Appendix F, we explore some of the implications of the decision to not split IPs across polygons.

References

Arsenault, A.A. 2003. Status and conservation management framework for woodland caribou (Rangifer tarandus caribou) in Saskatchewan. Fish and Wildlife Technical Report 2003-3. 40pp.

Lee, P., D. Aksenov, L. Laestadius, R. Nogueron, and W. Smith. 2003. Canada's Large Intact Forest Landscapes. Global Forest Watch Canada, Edmonton, AB. 84pp.

[21] Assembled by BEACONs from provincially-held inventories.

Wright. R.A., E. Beveridge, G. Freif, and O. Naelapea. 1998. Procedure for identifying candidate RAs in the forest, based on ecological criteria. Saskatchewan Environment and Resources Management.

ArcView Programming Code to Convert Forest Inventories to IPs

***Note: Consult the inventory manuals as you read through this document.**

Saskatchewan

The script for classifying forest inventory in SK into intrinsic patches closely follows the classification scheme that was put forth by Rettie *et al.* (1997). There are seven stands types identified in the paper:

A1 - Jack Pine, bog cranberry, reindeer lichen
A2 - Black Spruce, bog cranberry, reindeer lichen
B - White Spruce
C - Jack Pine/Black Spruce, blueberry, pine <40 years
D - Trembling Aspen
E - Black Spruce, Labrador tea, <55% cover, >90 years
F - Black Spruce/Jack Pine, Labrador tea, black spruce <90 years
G - Black Spruce, Labrador tea, moss, >55% cover

Nadele did the previous work of reprojecting all of the layers to a standard projection system and creating a data library. Using the Habitat classifier developed for the Remote Areas Project as a template, I added new code and altered the script to recognize the SK data structure. The Saskatchewan Forest Inventory Data Dictionary and database structure document was an invaluable tool during this process. The fields of interest are as follows:

SP10, SP11, D, YOO, MLEVEL, DIST, NP

SP10, SP11: Note that some of the species codes that appear in SP10 and SP11 are different from those found in AB. Some of the problems that I experienced with the script were due incorrect references to the species for example, using SB instead of BS in the code.

YOO: Year of stand origin was used in the script to determine if a stand was less than 40 years, greater than 40 years, less than 90 years, and greater than 90 years.

Less than 40: <=96
Greater than 40: > 96
Less than 90: <=91
Greater than 90: >91

MLEVEL: The various management levels were assigned to different intrinsic patch groupings.

ANR, PLA, SIL, SPM, SPO, SPS, STD = Mix
SDA = Anthropogenic
EXP, FG, OP = Unknown[22]

DIST: All disturbance types corresponded except for one. Clearcuts were assigned to Mix and Burn-Over was assigned to Unknown.
SCO, WCO, OCO, SPC, WPC, OPC = Mix
BO = Unknown

We investigated the possibility of classifying the burns using texture and drainage information but the results were not that accurate. Burns were considered to be unknown because we could determine what type of vegetation would be regenerating.

NP: Many of these numerical codes were placed into a specific intrinsic patch category. The problem with the NP codes is that detailed information does not exist for the polygons and so we can't verify that are 100% correct when we associate NP codes with intrinsic patches.

3100, 3300, 3900 = Bog
3200 = Upcon
3400, 3800 = Rock
3500, 3600 = Brush
3700, 4000 = Anthro
5100, 5200, 5210, 5220 = Water
9000 = No Data

Decision Rules
The decision rules can be seen in the ArcView code that was written to reclassify the forest inventory. I will detail some of the more important lines in the code and the rationale behind the rules.

<u>Mixed</u>

- Predominant species #1 is BS and predominant species #2 is White Spruce or Balsam Fir or Aspen then Mixed. This follows Rettie's classification under **Type B.**
- Predominant Species is White Spruce follows Rettie's classification under **Type B.**
- Predominant Species is deciduous follows Rettie's classification under **Type D.**

[22] The Unknown category is for stand types that we don't know how to classify as well as burned areas. Some areas with ambiguous management activities are also found under this heading. Unknown is probably a bad name as it does not necessarily mean NO DATA.

```
elseif ((t.ReturnValue(disturb,r) <> ""))then
        t.SetValue(Class,r,MIX)

 elseif ((t.ReturnValue(mod1,r) ="ANR") or (t.ReturnValue(mod1,r)="PLA") or
(t.ReturnValue(mod1,r)="SIL") or (t.ReturnValue(mod1,r)="SPM") or (t.ReturnValue(mod1,r)="SPO") or
(t.ReturnValue(mod1,r)="SPS") or (t.ReturnValue(mod1,r)="STD")) then
        t.SetValue(Class,r,MIX)
```

<u>Upcon</u>

- Pure Black Spruce with density greater than 55% follows Rettie's **Type G.**
- Pure Black Spruce with density less than 55% and less than 90 years follows Rettie's **Type F.**
- Black Spruce/Jack Pine and Jack Pine Black Spruce mixes follow Rettie's **Types C and F.**
- Jack Pine falls into various Rettie types. The script has the ability to distinguish the different types (see below) but for this reclassification we did not need that kind of detail. This statement is important if the various types of Upcon are to be further broken down.

```
elseif (sp1="JP") then
        if ((t.ReturnValue(origin,r) <= 96)) then
          if ((t.ReturnValue(crowncls,r)="A") or (t.ReturnValue(crowncls,r)="B")) then
            t.SetValue(Class,r,UPCON) 'type A1
          else
            t.SetValue(Class,r,UPCON) 'type F with crowncls C,D
          end
        else
          t.SetValue(Class,r,UPCON) 'type C  >96 and A,B, C, D
        end
```

<u>Bog</u>

- Rettie's classification considers black spruce with density less than 55% to be a bog of **Type A2. Type E** has the added criteria of being greater than 90 years old.
- Tamarack of all densities and ages should always be a bog.

```
elseif ((sp1="BS") or (sp1="TL")) then
        if ((t.ReturnValue(crowncls,r)="A") or (t.ReturnValue(crowncls,r)="B")) then
          if ((t.ReturnValue(origin,r) <= 91)) then
            t.SetValue(Class,r,BOG) 'type E     A,B and Age <=91
          else
            t.SetValue(Class,r,BOG) 'type A2    A,B Age >91
          end
       else
         t.SetValue(Class,r,BOG) 'type E special  C,D density
       end
```

- Pure black spruce stands with greater than 55% density have already been removed in the Upcon portion of the script so only Tamarack is affected by the final "else" statement in this excerpt from the script.

<u>Unknown</u>

```
elseif (t.ReturnValue(disturb,r)="BO") then
         t.SetValue(Class,r,UNKNOWN) 'Burn-over, disturbance type

elseif ((t.ReturnValue(mod1,r)="OP") or (t.ReturnValue(mod1,r)="FG") or
(t.ReturnValue(mod1,r)="EXP")) then
        t.SetValue(Class,r,UNKNOWN) 'Unmanaged or Unknown
```

Prince Albert National Park

Prince Albert has a forest inventory that is quite different from the rest of the province. Despite that, the results of the reclassification were extremely close to the surrounding areas. I continued to follow Rettie's classification scheme here.

A1 - Jack Pine, bog cranberry, reindeer lichen
A2 - Black Spruce, bog cranberry, reindeer lichen
B - White Spruce
C - Jack Pine/Black Spruce, blueberry, pine <40 years
D - Trembling Aspen
E - Black Spruce, Labrador tea, <55% cover, >90 years
F - Black Spruce/Jack Pine, Labrador tea, black spruce <90 years
G - Black Spruce, Labrador tea, moss, >55% cover

The fields of interest are as follows:

C1DENS, C1COND, C1SPEC, C2SPEC, C3SPEC, C4SPEC, C5SPEC, SA1

C1DENS: Class 1 and 2 are considered as <55% to fit with Rettie's classification
Class 1 = 0-30% cover
Class 2 = 30-60% cover
Class 3 = >60% cover

C1COND: This was the best indicator of age present in the dataset. The maximum age in this dataset was 80 years old. This differs a little from Rettie's classification but the same age rules were applied only using 80 years old instead of 90. Classes with the suffix "A" were considered to be the same as the classes without a suffix.

Class 1 = 10 years
Class 2 = 10-30 years
Class 3 = 30-60 years
Class 4 = 60-80 years
Class 5 = >80 years

C1SPEC: Contains species information. Codes for a mixed stand contain a combination of the various species' codes. For example, Pg is White Spruce and Pt is Aspen. A mixed white spruce aspen stand appears as PGPT . Consult the ArcView script for a complete list of all the codes found in the dataset.

U1, U2, M1, M2 = Brush
C = Anthropogenic
FL = Water

Decision Rules
The decision rules can be seen in the ArcView code that was written to reclassify the forest inventory. I will detail some of the more important lines in the code and the rationale behind the rules.

Mixed

- There were many combinations of species that I determined to be mixed. Aspen/Jack Pine stands were considered to be in this group.
- The groupings follow the same conventions as Rettie. Any stands with deciduous or white spruce leading were placed in the mixed category. Black Spruce stands with white spruce or aspen were also considered mixed.

Upcon

- One change that exists in the PANP script that differs with Rettie is that JackPine/White Spruce mixes were considered as Upcon.
- Pure black spruce with greater than 55% cover
- Pure black spruce with less than 30% cover and less than 80 years old follows Rettie's **Type F** but not perfectly. Modifications because of the different cutoffs for density and age has to be made.

```
   elseif ((sp1="PM") and (t.ReturnValue(crowncls,r)="3")) then
  t.SetValue(Class,r,UPCON) 'type G

elseif ((sp1="PM") and (t.ReturnValue(crowncls,r)="1") and (t.ReturnValue(origin,r)<>"5")) then
   t.SetValue(Class,r,UPCON) 'type F

elseif ((sp1="PMPB") or (sp1="PBPM") or (sp1="PGPB") or (sp1="PBPG"))  then
   t.SetValue(Class,r,UPCON) 'type F

elseif (sp1="PB") then
```

- Jack Pine stands were classified according to Rettie except in some places where density and age values had to be modified because of the different cutoffs used in PANP. This statement is important if the various types of Upcon are to be further broken down.

```
if ((t.ReturnValue(origin,r)<>"1") or (t.ReturnValue(origin,r)<>"2")) then
        if ((t.ReturnValue(crowncls,r)="1") or (t.ReturnValue(crowncls,r)="2")) then
        t.SetValue(Class,r,UPCON) 'type A1
        else
        t.SetValue(Class,r,UPCON) 'type F with crowncls 3
        end
else
    t.SetValue(Class,r,UPCON) 'type C  condition/age <3 and closure 1,2,3
end
```

Bog

- Black spruce stands were classified according to Rettie except in some places where density and age values had to be modified because of the different cutoffs used in PANP. This statement is important if the various types of Bog are to be further broken down.
- All Black spruce/Larch mixes and pure Larch stands are considered as bog.

```
elseif (sp1="PM") then
       if ((t.ReturnValue(crowncls,r)="1") or (t.ReturnValue(crowncls,r)="2")) then
         if ((t.ReturnValue(origin,r)="5")) then
           t.SetValue(Class,r,BOG) 'type E     open canopy and age >80years
         else
           t.SetValue(Class,r,BOG) 'type A2    open canopy and age <80 years
         end
       else
         t.SetValue(Class,r,BOG) 'type E special  closed canopy
       end


    elseif ((sp1="PMLL") or (sp1 = "LL") or (sp1 = "LLPM")) then
           t.SetValue(Class,r,BOG) 'type E
```

Manitoba

Manitoba's forest inventory is freely available and fairly extensive. However, for most areas in the province age information is lacking for the forest polygons. This caused me to have to change the decision rules a little for this province. Another important observation about this dataset is that I used a numerical code for classification as opposed to the species codes. This was done because of how the data was organized and due to time constraints. The numerical code had to parsed before classification so this was an extra step that had to be performed solely for MB.

Original field **Covertype** was parsed into **Subtype, Site, Crwcls,** and **Cutcls.** Fields **Subtype, Site, Crwcls,** and **Cutcls** were concatenated to form the field **NFcode.** The field **NFcode** is only relevant when **Subtype** = 99.

Split Script

This script parses the Covertype field in a number of new fields that will used to reclassify the Manitoba forest inventory. The main fields that will be creates are Subtype, Crown Class, Cutting Class, Site, NFcode, Spec1 and Scpec2.

```
fldCoverType = t.FindField("CoverType")
fldSpecies = t.FindField("Species")
fldSite = t.FindField("Site")
fldCutcls = t.FindField("Cutcls")
fldCrowncls = t.FindField("Crowncls")

find2 = t.FindField("Subtype")
find3 = t.FindField("Site")
find4 = t.FindField("Cutcls")
```

```
  find5 = t.FindField("Crowncls")
  find6 = t.FindField("Spec1")
  find7 = t.FindField("NFcode")
  find8 = t.FindField("Spec2")

  find12 = t.FindField("Sp1per")
  find13 = t.FindField("Sp2per")

  theCalc2 = t.ReturnValue(fldCoverType,r).Middle(0,2).AsString
  theCalc3 = t.ReturnValue(fldCoverType,r).Middle(2,1).AsString
  theCalc4 = t.ReturnValue(fldCoverType,r).Middle(3,1).AsString
  theCalc5 = t.ReturnValue(fldCoverType,r).Middle(4,1).AsString

  'GET SPECIES CODES

  theCalc6 = t.ReturnValue(fldSpecies,r).Middle(0,2).AsString
  theCalc7 = t.ReturnValue(fldSpecies,r).Middle(3,2).AsString 'Get code for secondary species

  'GET SPECIES PERCENTAGES
  theCalc11 = t.ReturnValue(fldSpecies,r).Middle(2,1).AsString
  theCalc12 = t.ReturnValue(fldSpecies,r).Middle(5,1).AsString 'Get percentage for secondary species

'FILL FIELDS WILL CALCULATED VALUES
  t.setValue(find2,r,theCalc2)
  t.setValue(find3,r,theCalc3)
  t.setValue(find4,r,theCalc4)
  t.setValue(find5,r,theCalc5)
  t.setValue(find6,r,theCalc6)
  t.setValue(find8,r,theCalc7)
  t.setValue(find12,r,theCalc11)
  t.setValue(find13,r,theCalc12)
  t.setvalue(find7,r,t.ReturnValue(fldSite,r)+t.ReturnValue(fldCutcls,r)+t.ReturnValue(fldCrowncls,r))
```

The fields of interest for the reclassification are as follows:

SUBTYPE, SITE, CRWCLS, NFCODE, SPEC1 and SPEC2

Subtype: Codes from 1-99 that represent different stand types. Consult the Manitoba inventory manual for a detailed description of each

Site: The site description was only used in conjunction with decision rules having to do with bogs. If the leading species was black spruce or tamarack and the site class was 2, then the stand was classed as a bog.

Crown Class:

0 = 0-20% crown closure
2 = 21-50%
3 = 51-70%
4 = >71%

NFcode: Three digit codes that described particular land types or land uses. Consult the Manitoba inventory manual for a complete list of the NFcodes.

Decision Rules

The decision rules can be seen in the ArcView code that was written to reclassify the forest inventory. I will detail some of the more important lines in the code and the rationale behind the rules.

Mixed

- In the case below we used decision rules to place a stand in either Mix or Bog. Subtype 55 is black spruce leading with balsam fir. Crown closure and site class help to determine if the stand is a bog or mixedwood. We assumed that if a stand had an open canopy (following Rettie <55% cover) and the site class was 2 than the stand was a bog.

```
elseif ((t.ReturnValue(sp1fld,r)="55")) then
 if (((t.ReturnValue(crown,r)="3") or (t.ReturnValue(crown,r)="4")) and (t.ReturnValue(sitecls,r)<>"2"))
then
        t.SetValue(Class,r,MIX)
      else
        t.SetValue(Class,r,BOG) ' crowncls 1,2
      end
```

- Subtype 51 is white spruce leading with balsam fir and jack pine. We verified if the secondary species was balsam fir and if so we classified the stand as a mixed otherwise the stand type was Upcon. The mixed class is consistent with Rettie's **Type B.**

```
   elseif ((t.ReturnValue(sp1fld,r)="51")) then
      if (t.ReturnValue(sp2fld,r)="BF") then
        t.SetValue(Class,r,MIX)
      else
        t.SetValue(Class,r,UPCON) ' crowncls 1,2
      end
```

Upcon

- In the script lines below we used decision rules to place a stand type in either Upcon or Bog. Subtypes 13, 17, 53, 55 and 57 are pure black spruce and black spruce leading with balsam fir and eastern cedar respectively. Here, crown closure and site class help to determine if the stand is a bog or upcon. We assumed that if a stand had an open canopy (following Rettie <55% cover) and the site class was 2 than the stand was a bog.

```
elseif ((t.ReturnValue(sp1fld,r)="13") or (t.ReturnValue(sp1fld,r)="17") or (t.ReturnValue(sp1fld,r)="53")
or (t.ReturnValue(sp1fld,r)="57")) then
      if (((t.ReturnValue(crown,r)="3") or (t.ReturnValue(crown,r)="4")) and
(t.ReturnValue(sitecls,r)<>"2")) then
        t.SetValue(Class,r,UPCON)
      else
        t.SetValue(Class,r,BOG) 'crowncls 1,2
```

```
end
```

Unknown

- There were some subtypes that we didn't have enough knowledge to classify. These included Eastern Cedar, Hackberry, and Hop Hornbeam.
- There were some codes found in the dataset that were not described in the inventory manual.
- In the Manitoba script, NO DATA values were also put in the Unknown category. This is a potential mistake.

```
elseif ((t.ReturnValue(sp1fld,r)="36") or (t.ReturnValue(sp1fld,r)="37") or (t.ReturnValue(sp1fld,r)="76")
or (t.ReturnValue(sp1fld,r)="77")) then
 t.SetValue(Class,r,UNKNOWN) 'Cedar

elseif ((t.ReturnValue(sp1fld,r)="9C") or (t.ReturnValue(sp1fld,r)="9D") or
(t.ReturnValue(sp1fld,r)="O4")) then
 t.SetValue(Class,r,UNKNOWN) 'Hackberry and Hop Hornbeam

elseif ((t.ReturnValue(nat_non,r)="814") or (t.ReturnValue(nat_non,r)="833") or
(t.ReturnValue(nat_non,r)="834")) then
        t.SetValue(Class,r,UNKNOWN)

elseif ((t.ReturnValue(sp1fld,r)="") and (t.ReturnValue(nat_non,r)="")) then
        t.SetValue(Class,r,UNKNOWN)
```

APPENDIX F: Proportions of Intrinsic Patches Entirely Captured Within Enduring Feature Polygons

Here, we explore some of the implications of the decision to not split intrinsic patches (IPs) across enduring feature (EF) polygons.

If an IP intersects more than one EF, then the dominant EF, by area, was designated to the IP. Therefore, in theory, the IP could cross multiple EF polygons, but the IP will only be attributed to one dominant EF. No matter how low the percentage was for the calculated dominance (i.e., 30% of the IP area), it was still considered dominant for one EF polygon. Overall, very few occurrences of dominance was attributed to polygons where the proportion of IP was < 60%. Likely, multiple possible dominant polygons in the data layer would be problematic for IPs that are very large.

We analysed the concordance between delineated IPs and mapped EF boundaries using patches >1ha of classes UPCON, MIX and BOG, within 271 EFs from the boreal plain as selected for the preceding analysis.

IPs were binned into discrete size classes on a logarithmic scale (1,2,5,10,20,50,... ha). For each size class we calculated:

pArea - the proportion of correctly assigned area,
p100 - the proportion of IPs entirely contained within their assigned EF,
p75 - the proportion of IPs at least 75 within their assigned EF and,
Cum - the cumulative proportion of correctly assigned areas within the size class and all smaller size classes.

The proportion of correctly assign areas for IP classes UPCON, MIX and BOG were 0.91, 0.79 and 0.74, respectively (Figure 1). That is, about 21% of the total area of class MIX is assigned to the wrong EF. The proportion p100 was low for all three IP classes except for the smallest size classes. The majority of IPs (larger than 500 ha) intersect more than one EF. Almost all IPs (larger than 10,000 ha) intersect more than one EF. The numbers of such large patches are few (19, 45 and 37 for UPCON, MIX and BOG respectively), but they account for a significant proportion of the total class area (0.126, 0.357 and 0.423, respectively).

Intersections of less than 100% could be attributable to mapping errors or to the differing spatial resolutions of the underlying data sets. However, the proportion of IPs with at least 75% assignment accuracy decreased markedly for sizes above 1,000 or 2,000 ha, especially for classes MIX and BOG. This corresponds to circular errors of at least 1 km which are not attributable to registration or resolution issues between the 1:250,000 scale Eco_SK EFs[23] and 1:20,000 vegetation inventory mapping (IP). Thus, we conclude that the mapped boundaries of EF are not entirely congruent with the landscape structures delineated by our intrinsic patches.

[23] NTS 1:250,000 source-tiled version of the Soil Landscape Units (v2.1) (Eco_SK as derived by Saskatchewan, Wright and Beveridge 1998)

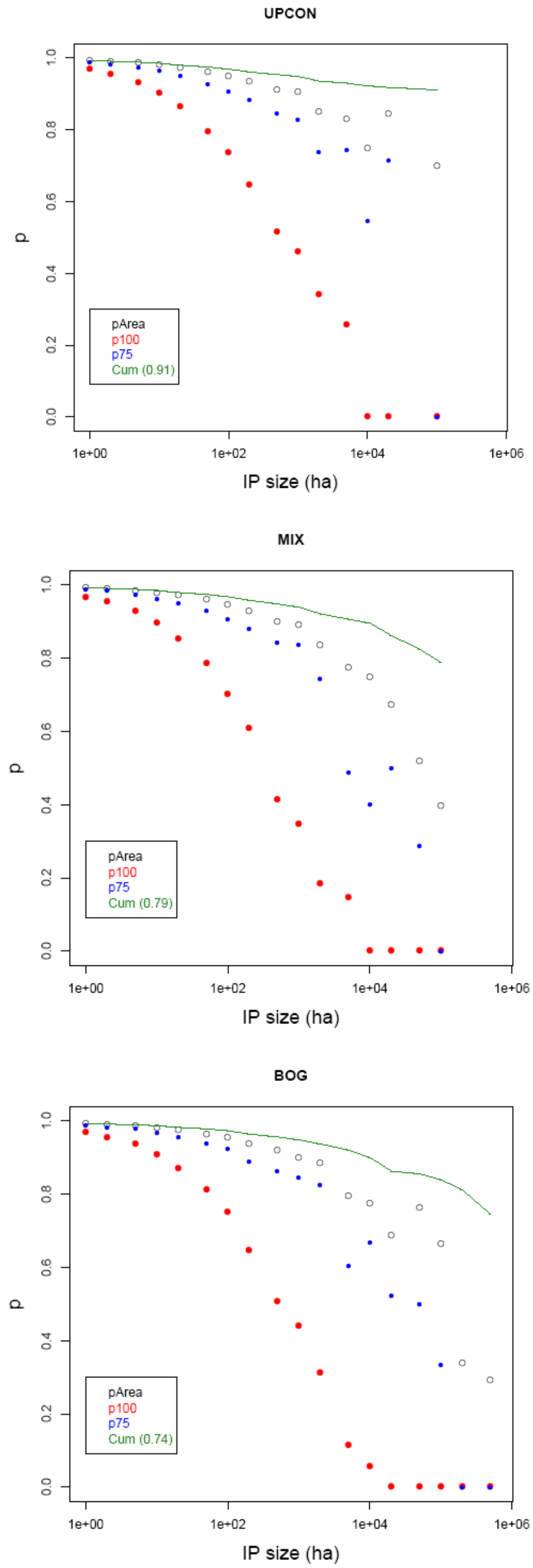

Figure 1: IP to EF spatial errors: rates and magnitudes by size class for UPCON, MIX and BOG.

The potentially most serious classification errors are when the assigned EF accounts for less than 50% of total IP area, indicating that the IP intersects at least three EFs. The proportional areas of such cases are 0.006, 0.166 and 0.204 for classes UPCON, MIX and BOG, respectively. Thus, a non-trivial proportion of the total area of MIX and BOG is reallocated among EFs by our assignment strategy. The mean proportions of correctly assigned areas were well above 0.50 for all but the largest MIX and BOG IPs. Among IPs of 10,000ha or larger, there are only 12 MIX and 6 BOG IPs of this kind. Thus, although the absolute errors may be fairly high in some cases, only a small proportion of EFs are likely to be affected.

The net effect is this: the land cover compositions we derive for EFs are not precise measurements of the IP abundances within the mapped EF boundaries. Rather, they should be regarded as samples from areas of approximately the correct size, defined by effectively random perturbations of the true EF boundary (to account for the assignment of IPs by majority or plurality of area.) Although these relationships should be explored in more detail, we think that the EF compositions used in these analyses are adequate to act as independent units by which to measure spatial variation in IP composition across the study area.

References

Wright. R.A., E. Beveridge, G. Freif, and O. Naelapea. 1998. Procedure for identifying candidate RAs in the forest, based on ecological criteria. Saskatchewan Environment and Resources Management.

Appendix G: Simulated Mean Forest Age-Structures

Introduction

In this appendix, we present a preliminary simulation study of expected forest age structures under alternate hypotheses about the natural disturbance regime, the effects of fire suppression, stand dynamics and measurement error in the forest inventory process. The purposes of this study were to identify

1. factors having the largest effect on age structures, actual and as observed in forest inventory data;
2. research needs to provide an empirical basis for inferring fire-regime parameters from inventory data.

Preparation of a fully referenced document was beyond the scope of the enabling contract. All the empirical claims or references to scientific results made herein can be supported by the primary literature; for issues or questions about specific claims, readers may contact the author[24]. The simulation model and graphics were produced in R 2.4.1. The main model function (NewAs), and the functions used to generate the simulations and graphical output (e.g. Fig1, Fig2, etc.) appear at the end of this Appendix.

Methods

We developed a simple age-structured simulation model to represent dynamics of a generic forest estate under natural and managed fire regimes. The model uses 1yr age classes and a 1yr time step. The forest is homogeneous except with respect to age. In the simplest formulation, in every time step, the annual area burned is distributed along age classes and regenerates at age 1 while unburned forest ages by 1 yr. In any such model, there must be a maximum age which acts as an accumulator age class. Here, the accumulator class includes forest 600yr and older. The total area of this oldest age class is negligible (<0.25%) under any of the scenarios considered here, so our results are not sensitive to the choice of maximum age. The model and parameters explored here are conceived to apply to conditions typical of mixedwood forests on the boreal plains.

Natural fire regime

We model two aspects of fire regime: mean fire return interval and age-dependent hazard. Over the period of historical record, mean annual area rates on the boreal plains have been less than 0.5%. However, some researchers claim the "natural" rate of burn characteristic of these forests is as high as 2% per year. We evaluate two alternate values of the point-level fire return interval *b*, 50yr and 100yr. This value is the mean interval between consecutive burns at a given location. Under the classic Van Wagner model of age and spatial independence, this implies mean annual burn rates of 2% and 1% respectively. In the model, the indicated proportion of each age class is considered to burn in very year, which produces at equilibrium an exponential age structure. Under

---

[24] S.G. Cumming, Département des sciences du bois et de la forêt, Université Laval, stevec@sbf.ulaval.ca.

age-independent hazard, this proportion is constant for all age classes, and hence the mean annual area burned is constant over all years. Many other model parameters are interpreted relative to the parameter *b*.

There is considerable evidence that recently burned boreal forest is relatively less susceptible to burning again for a period of several decades. There is also evidence that hazard of burning increases after 80 to 100yr, associated with successional changes from deciduous to conifer dominance, or to senescence of mature aspen stands leading to higher densities of fine fuels due to increased abundance of grasses. The standard, though likely inadequate, model for age dependence in forest fire regimes is the Weibull distribution. This is governed by a shape parameter α which determines how hazard changes with age, and a scale parameter. For $\alpha > 1$, hazard increases with age. The model allows the user to specify the value of α, and then adjusts the Weibull scale parameter so that the expected fire return interval is equal to *b.* It then calculates a vector of the age dependent hazard rates. For $\alpha = 1$ (the default), hazard is constant and the distribution reduces to an exponential. Here, we contrast $\alpha = 1$ (age independence) and $\alpha = 2$ which corresponds to a moderate degree of age-dependence. For $\alpha \neq 1$, although the age-dependent hazard is constant over time, the total annual area burned varies depending on the age structure.

The main contrast considered in simulations reported here is between high and low fire return intervals (*b*=50 and 100yr) and between age independent burning ($\alpha = 1$) and moderate age dependency ($\alpha = 2$).

Fire management

It has been shown that fire suppression by initial attack has reduced the proportion of large (>200ha) fires in the boreal plains of Alberta. There is evidence of two change points, one in 1971 and the other in 1983, corresponding to changes in fire management policy or behaviour. Each may have reduced mean burn rates by a factor of approximately 2. We assume a further factor of $2^{0.5}$ applied for the decade of the 1960s, prior to which there was no fire suppression. The mean effect over the period 1960 to 2000 is thus approximately 2.85. We assume further that the preceding conclusions apply also to the Saskatchewan boreal plains. We simulate the effect of fire suppression on present age structures by running the model to equilibrium under specified fire regime parameters (*b* and α) and then for another 40yr with hazard rates reduced by a factor of 2.85.

Forest Inventory effects

Binned age classes

Forest inventories assign origin times to mapped stands, in classes of 10yr, 20yr or wider. The (assigned origin) age structure observed in forest inventory data is not the same as the actual (time-since-fire) age structure of the inventoried forest. One difference is the binning. Fires do not occur in 20yr chunks, and the apparent age structure therefore depends to some extent on the choice of reference year. Perhaps more importantly, after a certain age is reached, even large age differences between stands are difficult or impossible to detect in aerial photographs or even by standard field measurements. This is one reason why most inventories have a maximum or oldest age class: in Saskatchewan, this generally lumps all apparently old forest into the 120yr+

category, even though substantial areas of forest much older than 120yr may be expected to exist. We report simulated age structures according to these conventions, binned as 20yr age classes up to 120yr, and 120yr+ class for older forest. This simulates the (stand origin) age structure that would be observed were an inventory to be conducted immediately after the end of a model run. This binning procedure is employed for all simulation runs. We refer henceforth to age-of-origin and time-since-fire age classes as the "observed" and "actual" age class structures, respectively.

Stand collapse and apparent regeneration age

Stands can undergo marked changes in structure in the absence of stand-replacing fire. If these changes also cause a change in the apparent stand age, e.g. under inventory photo-interpretation standards, that would be another source of discrepancy between observed and actual age class structures. In the boreal mixedwood, it is known that old aspen stands sometimes "collapse" in the sense that mortality reduces the density of the original (fire-origin) canopy cohort below detectable (or map-able) levels. The rate at which this occurs is unknown. The fate of post-collapse stands is also unknown in any quantitative sense. In at least some cases, based on analysis of consecutive inventories, collapsed stands appear to photo interpreters as if they were 40-80yr old fire-origin stands. It is also possible that they reemerge as newly regenerating stands, an assumption often embedded in forest management plans. Other possible dynamics, such as degradation to persistent grass cover until reset by a severe fire, are not considered here. We assume that stand collapse begins to occur only after 120yr. We explore two parameters: the rate of collapse of old stands (p), and their apparent age post-collapse (R). The number of factors now under evaluation is too large for easy comparison of results. We therefore evaluate levels of p and R within a single scenario of natural fire regime (b=100, α=2) and assuming recent fire suppression management, which we consider the most plausible of the 8 combinations of these three factors. We contrast low and high rates of collapse (p=0.01 and 0.05, respectively) and two levels of apparent regeneration age (R = 1 and 60yr).

Regeneration delays: the "brush effect"

Recently burned stands are often not included as forest land because they can not accurately be typed from aerial photography (and it seems that the inventory process seldom makes use of historical information such as fire maps or previous inventories). As a result, younger age classes may be under-represented in forest inventories. Some researchers in Alberta have suggested that this bias compromises inferences about ecological process based on forest inventories. To explore this, we added a brush stage to the model. Brush behaves like young forest, but is not counted in the inventory until a regeneration period of B years has passed. We contrasted two levels of B of 1 and 20yr, under the base scenario (b=100; α=2; 40yr of fire suppression). To explore the effects of other processes, we included a high rate of stand senescence (p = 0.05) at two regeneration ages (R=1 and 60yr).

Results

The main contrasts between high and low natural burn rates and age-dependent hazard are shown in Figure 1. The mean age structures under age independent hazard (Fig. 1a) produce a marked overabundance of the oldest or youngest age classes, respectively, compared to what is actually seen in the Saskatchewan inventories (Chapter 5) or in any published boreal forest inventory age structures that we know of. Intermediate burn rate (e.g. b=75 yr) would clearly produce U-shaped age structures which likewise are never observed. Moderate age-dependent hazards, such that probability of burning increases with stand age, are clearly inconsistent with a high rate of burn (Fig. 1b). However, at b=100yr, they do produce a more balanced age structure that resembles actual data, although the preponderance of forest in the first three age classes (0-60yr) is not found in any existing inventory we have seen. We conclude that parameter set b=100yr and α=2 results in the most plausible age structure of the four; the other three combinations can be rejected from further consideration. However, these two processes are probably not sufficient to explain the observed inventory age structures.

Adding the hypothesized effect of 40yr of fire suppression does produce more realistic-looking age structures for some values of the two main parameters, b and α (Fig. 2). In this case, two contradictory combinations of parameters lead to age structures that are somewhat similar to the observed inventories. These are high burn rates and age independent hazard (Fig 2a, b=50) and low rate and age-dependent hazard (Fig 2b, b=100). Both produce the peak in abundance at intermediate ages as seen in Saskatchewan forest (Chapter 5) and almost everywhere else in the Canadian boreal. The other two combinations predict marked excesses of either young or old forest. We note that the two more plausible competing explanations are contradictory yet probably could not be distinguished by any amount of age structure data without recourse to other information, e.g. independent evidence of fire suppression effectiveness.

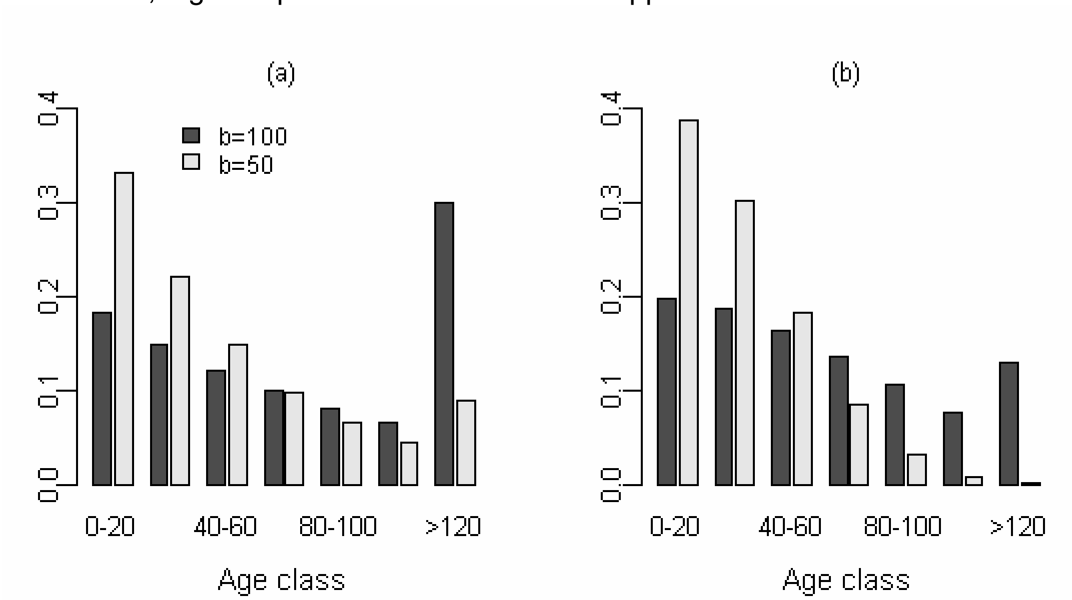

Figure 1: Contrast in simulated observed forest age structures under high (100yr) and low (50yr) point fire return intervals with (a) age independent burning and (b) moderate age dependence such that the hazard of burning increases with stand age.

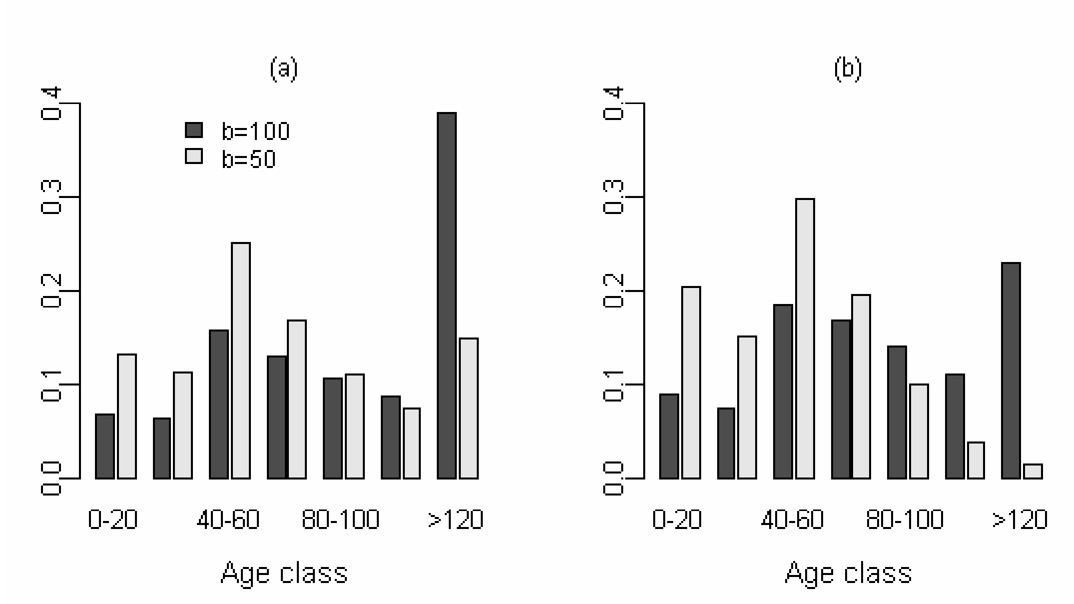

Figure 2: Contrast in simulated observed forest age structures under high (100yr) and low (50yr) point fire return intervals with (a) age independent burning and (b) moderate age dependence such that the hazard of burning increases with stand age; in each case the simulations as for Figure 1 were followed by 40yr of fire suppression with hazard rates reduced by a factor of 2.5.

The differences between the six stand senescence scenarios (Fig 3) would be difficult to detect except in the relative abundances of the oldest age class. The most important parameter is p, the rate of stand collapse. High rates p=0.05 markedly reduce the expected abundance of old forest. The effects of differing regeneration age (R=1 or R=60yr, contrasted in Fig 3a and Fig 3b) are surprisingly minor. At p=0.05, 50% of 120yr stands should collapse within 13yr, a prediction easily testable by comparing consecutive inventories or by a short-term monitoring program: we strongly recommend that one or both of these studies be initiated in the Saskatchewan boreal plains. We note that the main effects of p and R are confounded with variation in fire suppression and age-dependent burning in other scenarios (not shown).

The effects of a brush stage, composed with the previous factors, are shown in Figure 4. The main effect (B=20yr) is contrasted in each panel with the results of a corresponding run from Figure 3 with no brush stage (B=1yr). The only pronounced effect is on the relative abundance of forest in the 20-40yr age class. The effect is not detectable without the 40yr transient caused by the imposition of fire management; more generally, given the model structure, there would be no brush stage unless the forest age structure was not in equilibrium. The location of the predominant effect (here in the second age class) depends on the length of the brush stage, and the magnitude of the departure from an equilibrium age structure. We have not fully explored all the interactions between model components, but it seems possible that the presence of a brush stage could confound the ability to discern the effects of age-specific hazard and fire suppression (Fig.2b). Again, we note that the presence of such a brush stage could be detected, and compensated for, if the forest inventory process attempted to estimate the pre-fire composition of recently burned areas.

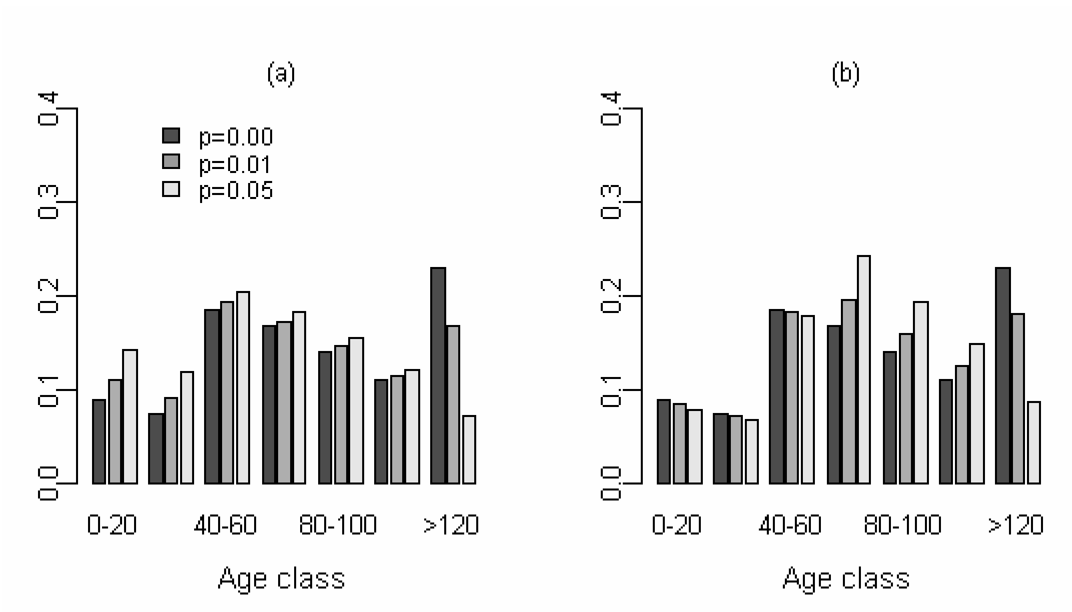

Figure 3: Simulated structures resultant from varying stand collapse rates and two levels of apparent regeneration age: *R*=1 (a) and *R*=60yr (b). Other model parameters are as for the low burn rate (*b*=100yr) scenario with age dependent fire hazard and fire suppression (Fig. 2b).

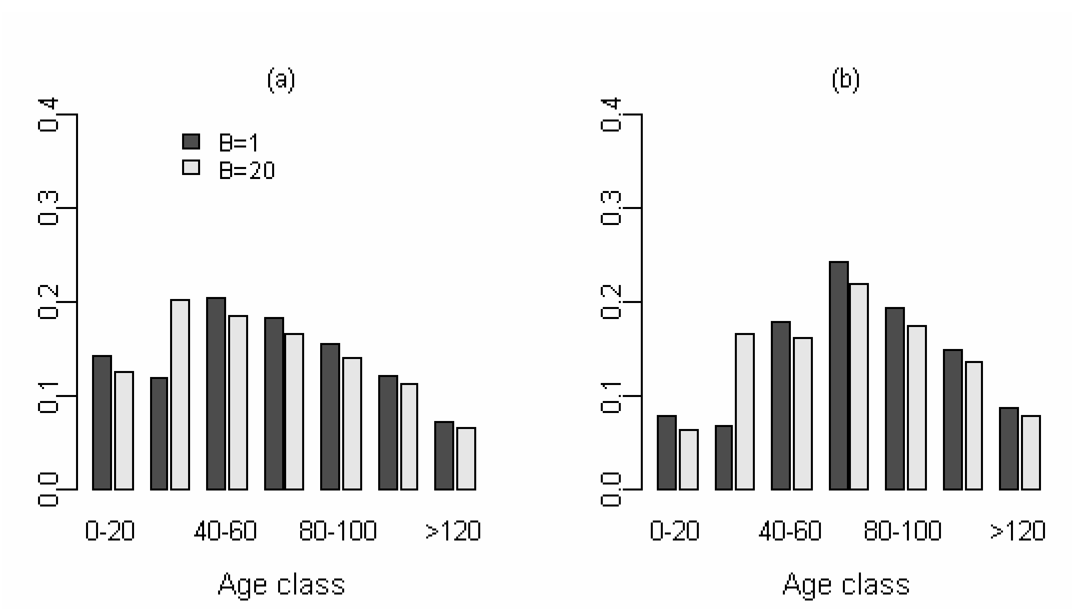

Figure 4: Effects of a 20yr brush stage on observed forest age structures given low burn rate (*b*=100yr), age dependent hazard (α=2), and 40yr of fire suppression, combined with high rate of stand senescence (*p*=0.05) and post-collapse regeneration ages of (a) *R*=1 and (b) *R*=60yr.

Conclusion and Recommendations

This very simple model nevertheless has enough structure to explore some meaningful questions and in considerably more detail than can be fully reported here. We have attached the R-code that implements the model and the R package itself is freely downloadable from www.r-project.org. The recipients can easily explore other scenarios and modify the model as desired. Note that the model does not simulate inter-annual variation in area burned. As is well known, this variation is very high in the boreal forest. As a result, forest age structures vary widely depending on the time of observation, *even if the underlying processes remain constant*. However, the equilibrium age structure may be considered as the expected value or the mean of a large sample of independent age structure observations. Although equilibrium boreal forests are never observed and probably do not exist at any spatial scale, comparison of equilibrium age structures simulated under alternate assumptions is still informative. For one thing, if alternate suites of hypotheses generate similar equilibrium age structures, it may be impossible to distinguish these hypotheses using a single observed age structure, e.g. from a snapshot estimated in a forest inventory.

Age structures aggregated from forest inventory data are often used to make inferences about ecological processes and to provide guidance for forest management. Our results show that these interpretations must be made with caution, because they necessarily depend on assumptions about many poorly quantified processes. A given age structure does not uniquely determine the processes that generated it. One consequence of this is that ***the Van Wagner "roll back method" can not estimate the average rate of disturbance from forest inventory data,*** unless all other factors, including stand senescence and inventory aging errors are accounted for. Successional transitions between forest types of different fire susceptibility (e.g. leading aspen and white spruce mixedwood) must also be considered[25]. To our knowledge, these factors have never been considered in inventory age structure analysis.

On the basis of the present analysis, and of past studies, we conclude that the best explanation of the current observed forest age structures in the western boreal plains is: a pre-management mean rate of burn of at most 1% per year; moderate age-dependent fire hazard; and biased representation of older forest in inventory that results from a combination of height-based origin estimates and failure to account for stand dynamic processes including senescence and succession.

We reiterate two recommendations based on these results. The government of Saskatchewan should consider improving the forest inventory process so that historical information (e.g. on pre-fire forest composition and apparent forest structure of the same patch within consecutive inventories) is considered. Studies should be initiated to determine the rate of stand senescence or collapse, and the fate of stands post-collapse, so that histories can be reflected in forest inventories as these accumulate over time. We have already proposed, in a separate document, a more thorough study of the natural range of variation in forest age structures, based on spatial simulation and a more detailed accounting of ecological processes.

---

[25] This could be accomplished by a simple generalisation of the current model to include two or more related populations of forest types, although the stand transition probabilities would need to be estimated.

```
NewAs<-function(b=100,alpha=1,   #rate and shape pars for Weibull
         fs=0, db=3,      #fire management: run for fs years with
                                #     hazard reduced by factor of db
         MaxAge=1000,p=1,  #collapse model
             RegenAge=1,
         Bage=1,InvL=0,   #Bage: if >1, length of brush stage
                          #      for regenerating burns;
         Amax=600,N=500)  #Run parameters; max age to track, N years
{
  #Weibull hazard function;
  #b is expected survival time; equiv to exponential when alpha=1
  wl<-function(b,alpha){
     lambda<-(b/gamma(1+(1/alpha)))^alpha
     1/lambda
  }
  hazard<-function(x,b,alpha){
     lambda<-wl(b,alpha)
     h<-x^(alpha-1)
     alpha * lambda * h
  }
  InitAge<-function(x,b,alpha){
     N<-length(x)
     x0<-x-1
     lambda<-wl(b,alpha)
     a <- 1 - exp(-lambda * (x^alpha))
     a <- a - (1-exp(-lambda * (x0^alpha)))
     a[N] <- a[N] + (1-sum(a))
     a
  }
   x<-1:Amax
   a<-InitAge(x,b,alpha)
   h<-hazard(x,b,alpha)

   brush<-rep(0,Bage)
   MeanBrush<-0
   burn<-rep(0,Amax)
   collapse<-ifelse(x<MaxAge,0,p)

   i<-0
   while(i<N+fs){
     if (i==N){h<-h/db}        # Apply one time fire suppression effect
     burn<-a*h
     a0<-a-burn
     old<-a0[Amax]
     a0[Amax-1]<-a0[Amax-1]+old
     if (Bage > 1) {
       bb<-brush * h[1]    #assume brush burns like young forest
       brush<-brush-bb
       a<-c(brush[Bage],a0[1:Amax-1])
       brush<-c(sum(burn)+sum(bb),brush[1:Bage-1])
       MeanBrush<-MeanBrush+sum(brush)
     }
     else {
        a<-c(sum(burn),a0[1:Amax-1])
     }
     a0<-a*collapse
     a<-a-a0
     a[RegenAge]<-a[RegenAge]+sum(a0)
     i<-i+1
   }
   Err<-ifelse(seq(1,Amax)< 21,InvL,0)
   a0<-a*Err
   a<-a-a0
   a<-a/sum(a)    #Normalise to simulated area of inventory.
   a
}

bin20<-function(a){ #function to bin the NewAs() output for plotting.
    c(sum(a[1:20]),sum(a[21:40]),sum(a[41:60]),sum(a[61:80]),
      sum(a[81:100]),sum(a[101:120]),sum(a[121:length(a)]))
}
```

```
Fig1<-function(a=2){
  def.par<-par(no.readonly=TRUE)
  par(cex.axis=0.8,cex.main=0.85,cex.lab=0.9,
      font.main=1,
      mgp=c(2,0.5,0),
      mar=c(3,2,2,2)+0.1
     )
  nf <- layout(matrix(c(1,2), 1, 2, byrow=TRUE), respect=TRUE)
  xn<-c("0-20","20-40","40-60","60-80","80-100","100-120",">120")

  foo<-NewAs(N=500,alpha=1,b=100)
  bar<-NewAs(N=500,alpha=1,b=50)
  barplot(t(cbind(bin20(foo),bin20(bar))),beside=TRUE,
          ylim=c(0,0.4),
          names=xn,
          cex.names=0.8,
          space=c(0.2,1),
          xlab=c("Age class"),
          main=c("(a)")
         )
  legend(5,0.4,c("b=100","b=50"),bty="n",fill=c("grey30","grey90"),cex=0.75)

  foo<-NewAs(N=500,alpha=a,b=100)
  bar<-NewAs(N=500,alpha=a,b=50)
  barplot(t(cbind(bin20(foo),bin20(bar))),beside=TRUE,
          ylim=c(0,0.4),
          names=xn,
          cex.names=0.8,
          space=c(0.2,1),
          xlab=c("Age class"),
          main=c("(b)")
         )
  par(def.par)
}
Fig2<-function(a=2){
  def.par<-par(no.readonly=TRUE)
  par(cex.axis=0.8,cex.main=0.85,cex.lab=0.9,
      font.main=1,
      mgp=c(2,0.5,0),
      mar=c(3,2,2,2)+0.1
     )
  nf <- layout(matrix(c(1,2), 1, 2, byrow=TRUE), respect=TRUE)
  xn<-c("0-20","20-40","40-60","60-80","80-100","100-120",">120")

  db<-(2^0.5 + 2 + 8)/4
  foo<-NewAs(N=500,alpha=1,fs=40,db=db,b=100)
  bar<-NewAs(N=500,alpha=1,fs=40,db=db,b=50)
  barplot(t(cbind(bin20(foo),bin20(bar))),beside=TRUE,
          ylim=c(0,0.4),
          names=xn,
          cex.names=0.8,
          space=c(0.2,1),
          xlab=c("Age class"),
          main=c("(a)")
         )
  legend(5,0.4,c("b=100","b=50"),bty="n",fill=c("grey30","grey90"),cex=0.75)

  foo<-NewAs(N=500,alpha=a,fs=40,db=db,b=100)
  bar<-NewAs(N=500,alpha=a,fs=40,db=db,b=50)
  barplot(t(cbind(bin20(foo),bin20(bar))),beside=TRUE,
          ylim=c(0,0.4),
          names=xn,
          cex.names=0.8,
          space=c(0.2,1),
          xlab=c("Age class"),
          main=c("(b)")
         )

  par(def.par)
}
```

```
Fig3<-function(a=2,b=100,fs=40){
  def.par<-par(no.readonly=TRUE)
  par(cex.axis=0.8,cex.main=0.85,cex.lab=0.9,
      font.main=1,
      mgp=c(2,0.5,0),
      mar=c(3,2,2,2)+0.1
     )
  nf <- layout(matrix(c(1,2), 1, 2, byrow=TRUE), respect=TRUE)
  xn<-c("0-20","20-40","40-60","60-80","80-100","100-120",">120")

  db<-(2^0.5 + 2 + 8)/4
  foo<-NewAs(N=500,alpha=a,fs=fs,db=db,b=b,MaxAge=120,p=0.00,RegenAge=1)
  bar<-NewAs(N=500,alpha=a,fs=fs,db=db,b=b,MaxAge=120,p=0.01,RegenAge=1)
  baz<-NewAs(N=500,alpha=a,fs=fs,db=db,b=b,MaxAge=120,p=0.05,RegenAge=1)
  barplot(t(cbind(bin20(foo),bin20(bar),bin20(baz))),beside=TRUE,
          ylim=c(0,0.4),
          names=xn,
          cex.names=0.8,
          space=c(0.2,1),
          xlab=c("Age class"),
          main=c("(a)")
         )
  legend(5,0.4,c("p=0.00","p=0.01","p=0.05"),bty="n",fill=c("grey30","grey60","grey90"),ce
  x=0.75)

  foo<-NewAs(N=500,alpha=a,fs=fs,db=db,b=b,MaxAge=120,p=0.00,RegenAge=60)
  bar<-NewAs(N=500,alpha=a,fs=fs,db=db,b=b,MaxAge=120,p=0.01,RegenAge=60)
  baz<-NewAs(N=500,alpha=a,fs=fs,db=db,b=b,MaxAge=120,p=0.05,RegenAge=60)
  barplot(t(cbind(bin20(foo),bin20(bar),bin20(baz))),beside=TRUE,
          ylim=c(0,0.4),
          names=xn,
          cex.names=0.8,
          space=c(0.2,1),
          xlab=c("Age class"),
          main=c("(b)")
         )
  par(def.par)
}
Fig4<-function(a=2,fs=40,B=20){
  def.par<-par(no.readonly=TRUE)
  par(cex.axis=0.8,cex.main=0.85,cex.lab=0.9,
      font.main=1,
      mgp=c(2,0.5,0),
      mar=c(3,2,2,2)+0.1
     )
  nf <- layout(matrix(c(1,2), 1, 2, byrow=TRUE), respect=TRUE)
  xn<-c("0-20","20-40","40-60","60-80","80-100","100-120",">120")

  db<-(2^0.5 + 2 + 8)/4
  p<-0.05
  r<-1
  foo<-NewAs(N=500,alpha=a,fs=fs,db=db,b=100,MaxAge=120,p=p,RegenAge=r,Bage=1)
  bar<-NewAs(N=500,alpha=a,fs=fs,db=db,b=100,MaxAge=120,p=p,RegenAge=r,Bage=B)
  barplot(t(cbind(bin20(foo),bin20(bar))),beside=TRUE,
          ylim=c(0,0.4),
          names=xn,
          cex.names=0.8,
          space=c(0.2,1),
          xlab=c("Age class"),
          main=c("(a)")
         )
  legend(5,0.4,c("B=1","B=20"),bty="n",fill=c("grey30","grey90"),cex=0.75)

  p<-0.05
  r<-60
  foo<-NewAs(N=500,alpha=a,fs=fs,db=db,b=100,MaxAge=120,p=p,RegenAge=r,Bage=1)
  bar<-NewAs(N=500,alpha=a,fs=fs,db=db,b=100,MaxAge=120,p=p,RegenAge=r,Bage=B)
  barplot(t(cbind(bin20(foo),bin20(bar))),beside=TRUE,
          ylim=c(0,0.4),
          names=xn,
          cex.names=0.8,
```

```
          space=c(0.2,1),
          xlab=c("Age class"),
          main=c("(b)")
         )

  par(def.par)
}
```