Ecological Benchmarks to Support Landscape Conservation Design in the Northwest Boreal LCC Planning Region

Prepared by the BEACONs Project, University of Alberta – Kimberly Lisgo, Fiona Schmiegelow, Pierre Vernier, Marc Edwards, and Alberto Suarez-Esteban

December 2017

Citation: BEACONs. 2017. Ecological Benchmarks to Support Landscape Conservation Design in the Northwest Boreal LCC Planning Region. BEACONs Project, University of Alberta and Yukon College, Whitehorse YT. Available online: www.beaconsproject.ca/nwb.

Table of Contents

Introduction

To support the implementation of Landscape Conservation Design (U.S. Fish & Wildlife Service 2013), the Northwest Boreal Landscape Conservation Cooperative (NWBLCC) has adopted the Conservation Matrix Model (CMM) developed by the BEACONs Project. The CMM is a science-based framework for proactive conservation planning to facilitate biodiversity, conservation and sustainable use of natural resources across a spectrum of opportunities (Schmiegelow *et al.* 2014). It combines the strength of systematic planning for reserves with the systematic process of adaptive resource management, resulting in integrated conservation planning over large regions.

The CMM is a whole-landscape approach that acknowledges the contribution of all landscape elements to conservation, and involves explicit recognition of uncertainty about the outcomes of management decisions. As such, one of the key steps towards the implementation of the CMM is the identification of benchmark networks to support active adaptive management. Ecological benchmarks are reference sites or controls for understanding the response of biodiversity to human activity (Arcese and Sinclair 1997). This need arises from recognition that our understanding of the dynamics of ecosystems is incomplete, and that we must learn-by-doing through testing and monitoring of alternative management options.

This report details the methods and results of a NWBLCC-wide benchmark analysis (Figure 1), that included an evaluation of existing protected areas, to identify representative benchmark network options for adaptive management. The network options were evaluated and ranked based on fundamental benchmark properties, resilience to climate change, and the representation of habitat for priority focal species identified by the NWBLCC. The report concludes with guidelines on how to use the analysis output.

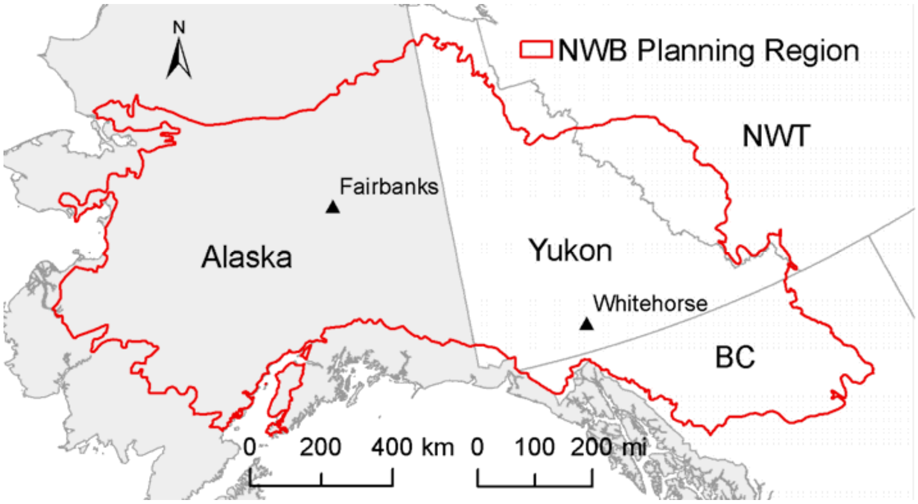

Figure 1. Benchmark network options to support the implementation of active adaptive management were identified across the NWBLCC planning region, which includes Alaska, Yukon Territory, Northwest Territories (NWT), and British Columbia (BC).

Basics of Benchmark Design

Benchmark design is central to implementation of the CMM. To serve as effective control areas for active adaptive management, ecological benchmarks are designed to:

- *Be **Intact**, with little to no human disturbance, so that ecological and evolutionary processes are operating without influence by human activity.*

Ecosystems with a high degree of integrity are often labeled as the baseline or benchmark condition (Sinclair 1998, Sinclair *et al.* 2002). This requires that processes operate free of human disturbance. With few exceptions, such as air borne pollutants and climate change, human disturbance has a quantifiable spatial footprint (*e.g.,* road, town site, clearcut). We identified intact landscapes for the NWB using the disturbance datasets described in Figure 2.

- *Be of **sufficient size** to capture large-scale ecological processes that shape landscape structure and maintain habitat types that are vulnerable to natural disturbance.*

Natural disturbances such as fire and insect outbreaks are examples of large-scale processes that play a significant role in shaping landscape structure and the adaptations of many organisms that inhabit the boreal (*e.g.*, Weber and Stocks 1998, Bond and Keeley 2005, Brandt *et al.* 2013). By capturing and maintaining natural disturbance regimes, benchmarks can support the natural function of processes operating at finer scales and the persistence of species. For species to persist, a benchmark must be of sufficient size to experience large, severe natural disturbance events and maintain internal recolonization sources or lifeboats of vulnerable habitats (*e.g*., flammable) for reliant species. By maintaining these lifeboats, the benchmark can continuously support effective monitoring of biodiversity and the implementation of adaptive management. To achieve size objectives, we used benchmark sizes defined by Minimum Dynamic Reserves (Figure 3), size estimates for reserves designed to represent natural disturbance and maintain vulnerable vegetation types (Leroux *et al.* 2007a).

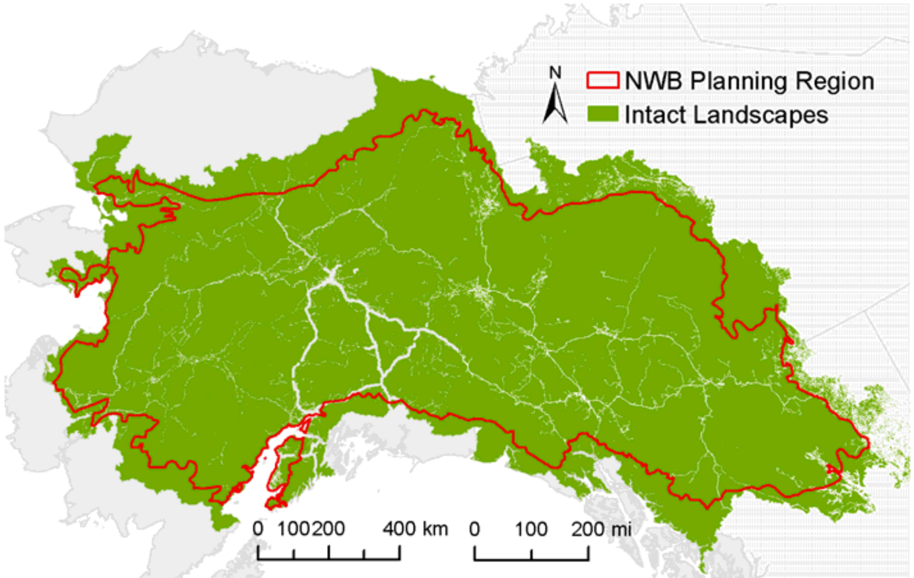

Figure 2. Intactness is often described based on the absence of a conspicuous human footprint (*e.g.*, Bryant *et al.* 1997, Sanderson *et al.* 2002). Two datasets were used to describe the landscape intactness of the NWBLCC planning region (red outline). In Alaska, intactness was quantified using the Landscape Condition of Alaska (Trammell and Aisu 2014). This dataset contains values ranging from 0 to 1, or low to high landscape condition, respectively, based on weighted measures of human impact. We identified intact landscapes for Alaska using values of 1, which this map displays. In Canada, we used Global Forest Watch Canada's Human Access dataset (Lee and Cheng 2014). Human access is defined as the combined land surface of anthropogenic disturbances caused mainly by industrial activities, which include roads, mines, clearcuts, wellsites, pipelines, transmission lines, and agricultural clearings. The dataset is a polygon-based layer that combines remotely-sensed data with ancillary datasets; prior to assembly, a 500-m buffer was applied to the data.

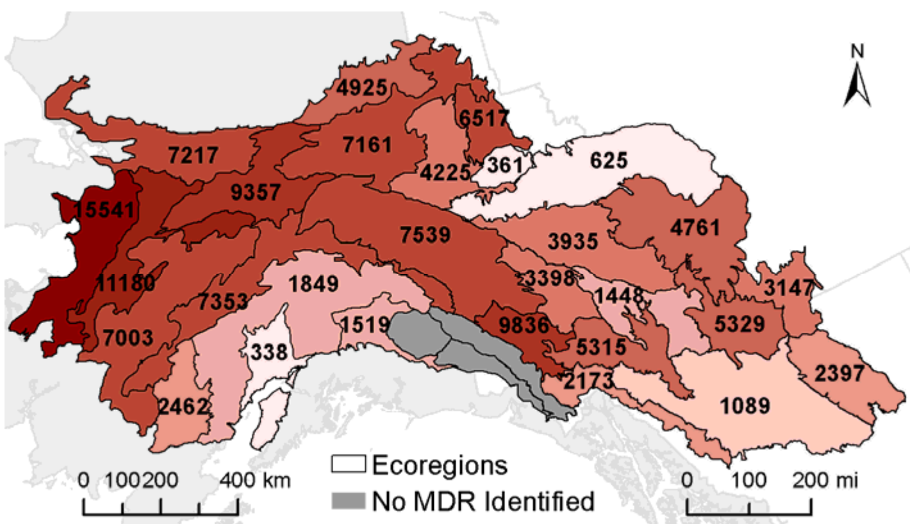

Figure 3. Minimum Dynamic Reserve sizes (MDR; km^2) were estimated for ecoregions (Figure 10) across the NWBLCC planning region based on characteristics of the local fire regime and the distribution of vulnerable vegetation types. MDR estimates range from 338 to 15,541 km^2 (130 to 6,000 mi^2). MDR estimates were not possible in three ecoregions (shaded grey) given too few fires; these ecoregions have large amounts of snow and ice. See the accompanying MDR Report (BEACONs 2017a) for analysis details.

• *Support* ***terrestrial and hydrologic connectivity*** *to facilitate the flow of nutrients and organisms that support the ecological and evolutionary processes essential for ecosystem function and integrity and species persistence.*

To integrate terrestrial and lateral and longitudinal hydrologic connectivity, benchmarks are assembled from catchments (Figure 4) along stream networks, using hydrology-based assembly rules embedded in the Benchmark Builder software (Figure 5). The rules prioritize the inclusion of headwaters. Protecting the long-term ecological integrity of aquatic and terrestrial ecosystems requires a foundation of intact and functional headwaters (Lowe and Likens 2005). Headwaters are the primary interface between upland and riparian areas and the overall stream network, and have several ecological functions which influence the structure, function, productivity, and biodiversity of downstream ecosystems (Gomi *et al.* 2002, Wipfli *et al.* 2007). To prevent threats to the ecological integrity of benchmarks from upstream disturbances, headwaters are necessary elements of benchmarks or should receive some form of special management.

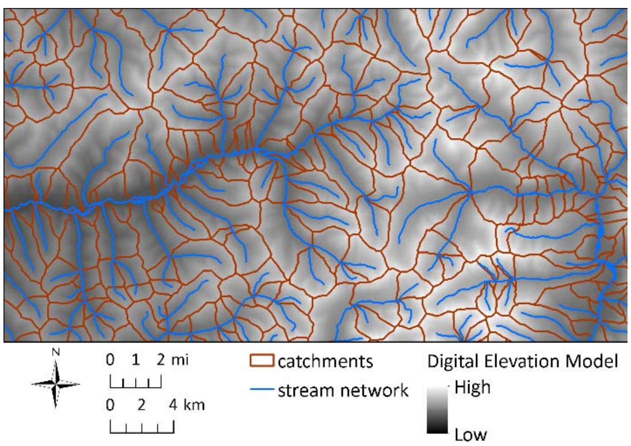

Figure 4. Catchments are approximate drainage areas for stream segments and capture lateral hydrologic connectivity while serving as the building blocks of ecological benchmarks. Catchments range in size from 0.5 to 277 km^2 (median = 2.2 km^2) or 0.2 to 107 mi^2 (median = 0.8 mi^2). The NWBLCC catchment dataset was created using the methods described in BEACONs (2015a) and the following stream networks and digital elevation models for Canada and Alaska: National Hydrological Network (NHN; 1:50,000 or better) Version 14 (NRCAN 2011-2014), Canada Digital Elevation Model Data (CDED; 1:50,000 or better; NRCAN 2001-2012), National Hydrology Dataset (NHD; 1:24,000; USGS 2014a), and National Elevation Dataset (NED; 1:24,000; USGS 2014b).

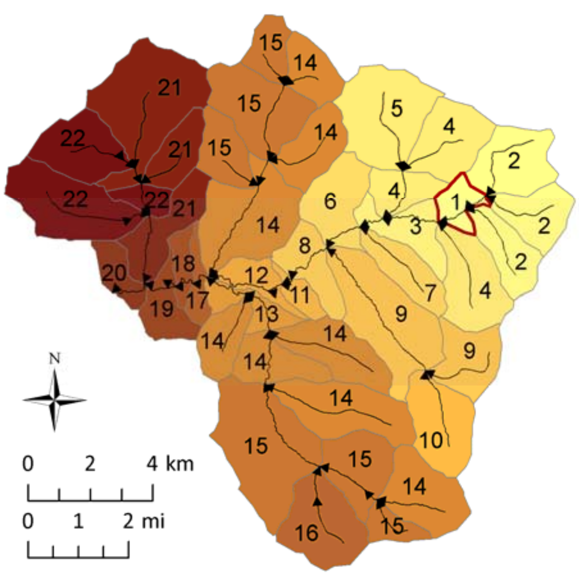

Figure 5. Benchmark Builder software assembles catchments along stream networks to a user-defined size (*e.g.*, MDR) and intactness (BEACONs 2016a). Starting from a seed catchment (red outline), Builder prioritizes growth in the upstream direction. This emphasizes inclusion of headwaters. Once all neighbouring upstream catchments are added, growth redirects downstream until upstream growth is possible. For the NWBLCC analysis, seed catchments were restricted to headwater catchments (Strahler Order 1 and 2) within the planning region (*i.e.*, ecoregion). Benchmarks for the NWBLCC were constructed using a minimum catchment-level intactness of 80%.

• *Be **representative** of environmental variation in the planning region.*

Potential benchmarks identified based on the three criteria above are assembled into representative benchmark networks using four biophysical indicators of environmental variation, which serve as surrogates for biodiversity: soil moisture (CMI), primary productivity (GPP), lake-edge density (LED), and land cover (Figure 6). We used two methods grounded in proportional representation to identify representative benchmark networks:

MDR-based Targets - This method requires the assignment of biophysical indicators to classes. As such, continuous indicators were re-classified to a standardized set of NWBLCC-wide classes (Table 1), allowing for ease of comparison across multiple scales. Representation was assessed by comparing the area of each indicator class within the benchmark network to MDR-based targets for the ecoregion.

Table 1. Reclassification of continuous indicators in Figure 6.

Indicator	Range of Values in NWBLCC	Class Width
CMI	-17 – 955 cm water/year	5; class 1 = -20 to -15
GPP	0 – 1.1 kg C/m^2/year	0.05; class 1 = 0 to 0.05
LED	0 – 0.7 km/km^2	0.025; class 1 = 0, class 2 = >0 to 0.025

MDR-based targets for each class are calculated by multiplying the MDR for the ecoregion by the proportion of that indicator class in the ecoregion. For example, if the indicator class makes up 10% of the ecoregion, the target would be 0.1 x MDR. For an MDR-based target to be achieved, it must be fully met within a single benchmark. Achieving targets for representation within a single benchmark minimizes the risk of losing environmental conditions associated with the biodiversity surrogate, to support a robust sampling design for effective and efficient monitoring. With this method, classes covering less than 5% of the ecoregion are considered rare and were not used to identify representative benchmark networks, but can be used in the selection process when multiple network options are available. The identification of benchmark networks began by evaluating all networks comprised of one benchmark. As required, the number of benchmarks was increased incrementally by one, until all indicator classes were represented. If a minimum of three benchmarks is required to meet all representation targets, for example, all possible combinations of three benchmarks were evaluated. Landscapes highly modified by humans were removed from the analysis, as defined by landcover classes *Cropland* and *Urban and builtup* (CEC 2013)

Dissimilarity Metrics (DM) - Once benchmark networks were identified using MDR-based targets, network options were ranked using dissimilarity metrics. Dissimilarity metrics compare the distribution of indicators within the candidate networks against the distribution within the planning region (*e.g.*, ecoregion): Kolmogorov-Smirnov for continuous indicators (Figure 7) and Bray-Curtis for categorical indicators (Figure 8). The indicator distributions are based on pixel-level values. Both dissimilarity metrics, range from 0 to 1, where 0 is most similar and 1 is most dissimilar. The closer the two distributions are to each other, the more representative the candidate network is to its reference area and the lower the value of the dissimilarity metric. While only four indicators were used in the NWBLCC analysis, the software can accommodate any number of continuous or categorical indicators, including species habitat distribution maps and climate-based indicators.

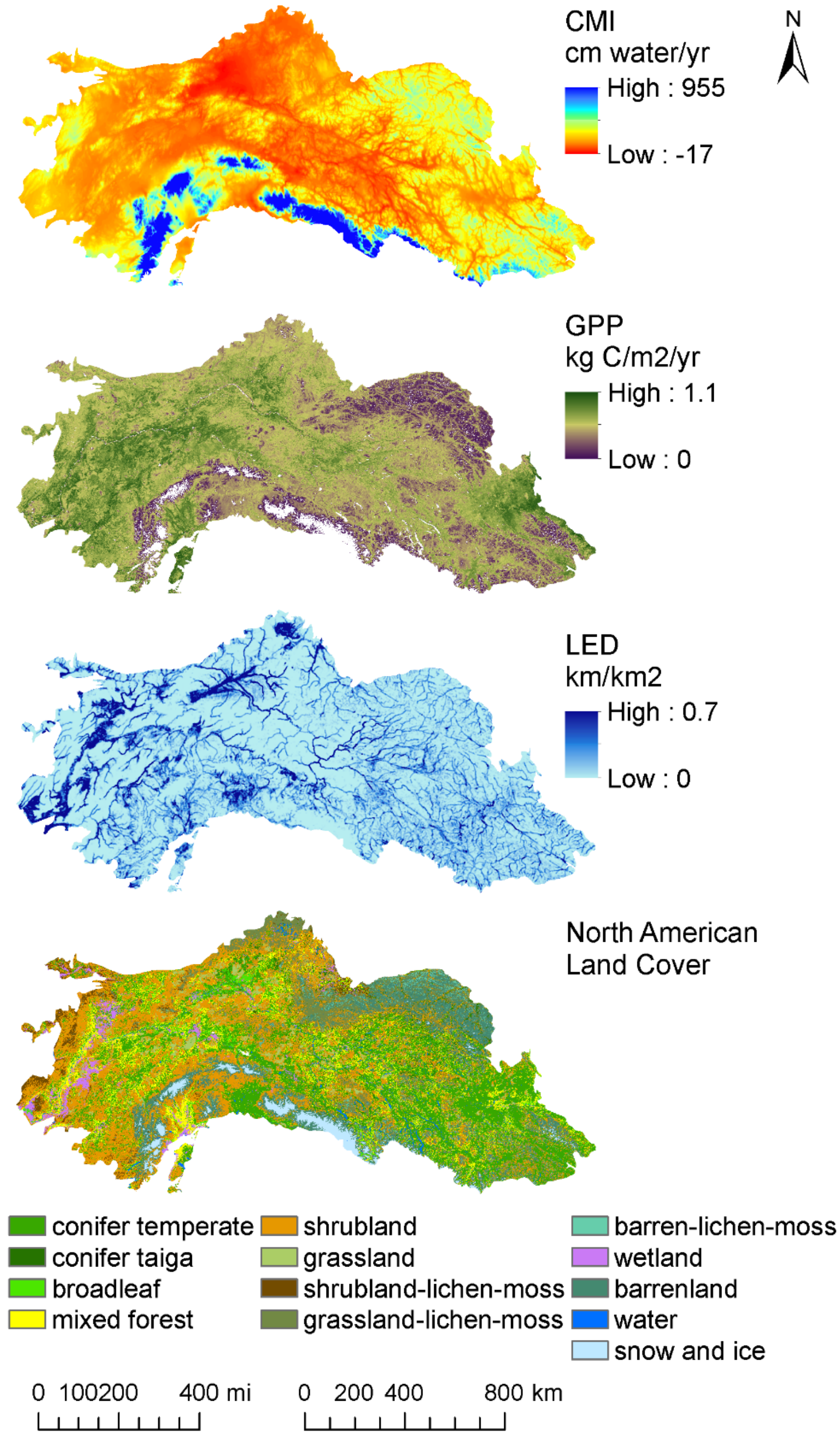

Figure 6. **Climate Moisture Index** (CMI 1961-1990; cm water/year) is a measure of water deficit or surplus in soil based on yearly average precipitation minus yearly potential evapotranspiration (Wang et al. 2016). Mean annual **Gross Primary Productivity** (GPP 2000-2014) is a measure of carbon absorbed by living plants or the amount of carbon absorbed during photosynthesis (kg C/m^2/year; Zhao and Running 2010). **Lake-Edge Density** (LED) characterizes the density of riparian habitat in km/km^2 (BEACONs 2015b). **North American Land Cover 2010** is based on 250-m MODIS satellite imagery and comprised of 19 cover types including forests, shrublands, grasslands, lichen, moss, wetlands, and non-vegetated areas (barren land, cropland, water, snow and ice; CEC 2013).

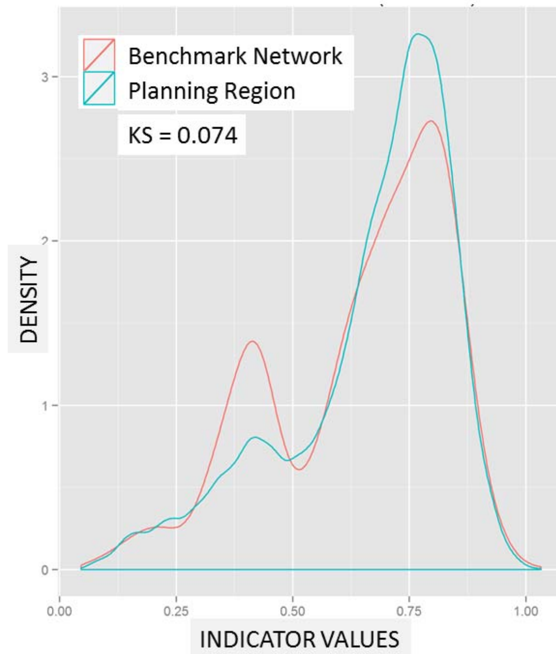

Figure 7. Density plots show the distribution of the indicator within the benchmark network (red) and within the planning region (blue). The Kolmogorov-Smirnov (KS) statistic describes the dissimilarity between these distributions, and ranges in value from 0 to 1, where 0 indicates perfect proportional representation within the benchmark network. Portions of the network distribution (red) that fall below the blue represent values for which proportional representation was not achieved.

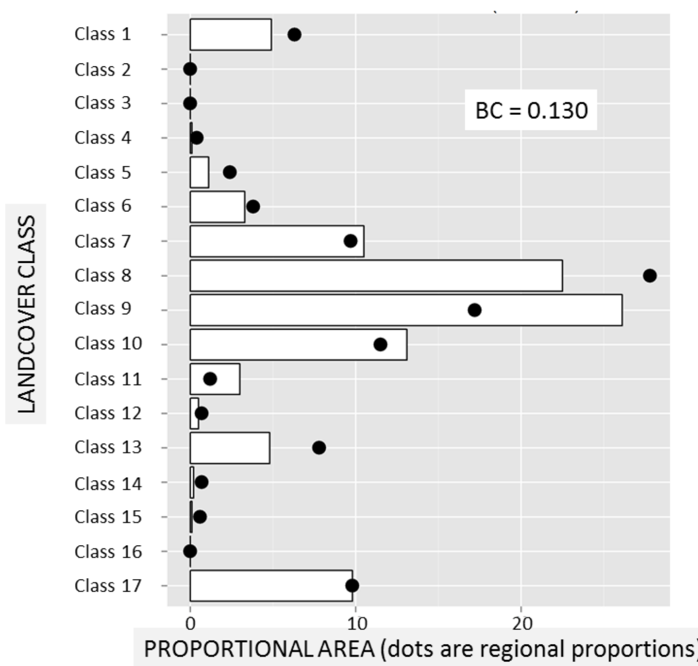

Figure 8. Barplots show the proportions of each indicator class (*i.e.*, land cover types) within the benchmark network (bars) and within the planning region (black dots). The Bray-Curtis (BC) statistic describes the dissimilarity between the bars and dots, and ranges in value from 0 to 1, where 0 indicates perfect proportional representation within the benchmark network. Bars that fall below the black dots indicate that proportional representation of that class was not achieved.

System- and Subsystem-Level Benchmarks

Benchmarks are designed along gradients of size and intactness (Figure 9). **System-level benchmarks** are the gold standard and are designed to be highly intact and sufficiently large to meet the size requirement of an MDR. When the identification of system benchmarks is not possible due to landscape condition, or implementation is constrained due to practical considerations, sub-system benchmarks that capture smaller processes or elements of larger processes (*e.g.*, migratory staging areas) may be identified. **Subsystem-level benchmarks** are smaller sites and/or sites impacted by humans that can still serve as benchmarks for the appropriate processes, but their utility is limited when compared to system benchmarks. For the NWBLCC analysis, system-level benchmarks were designed with a minimum catchment-level intactness of 80%. Subsystem-level benchmarks were defined as having a minimum catchment-level intactness of 80% and 80-99% MDR in size.

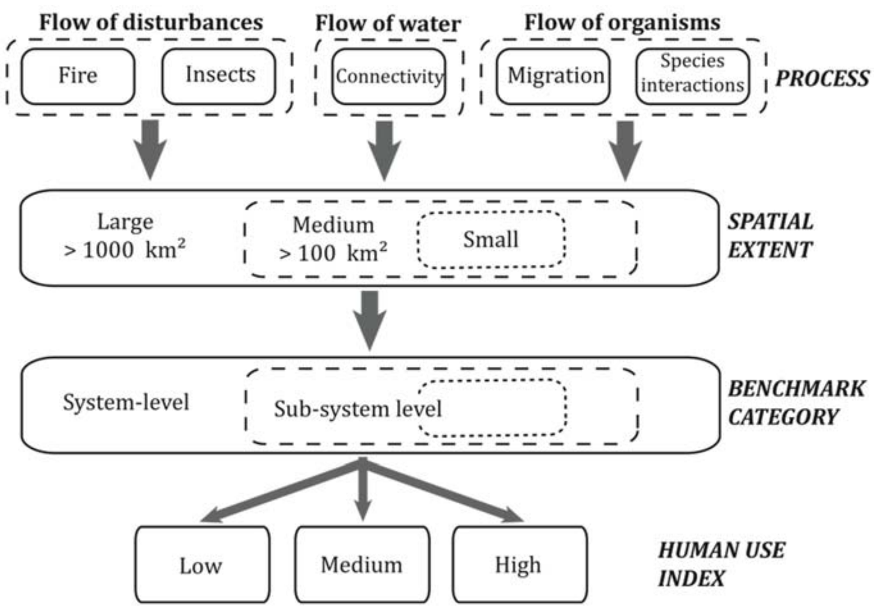

Figure 9. Benchmarks capture ecological processes at different spatial extents and levels of human activity. System-level benchmarks capture large endogenous processes, whereas subsystem-level benchmarks can be identified for monitoring smaller processes or elements of larger processes

Scale of Analysis – Stratifying the NWB Planning Region

The NWBLCC planning region was stratified into ecoregions to facilitate the design of ecological benchmark networks. Analyses at broader scales can be more efficient with respect to the spatial extent of benchmark networks; however, smaller regions better capture the range of environmental variability. Ecoregions are delineated based on distinctive regional ecological factors, including climate, physiography, vegetation, soil, water, and fauna, but do not account for the hydrology of the region.

Hydrologic connectivity has typically been overlooked in conservation assessments, to the detriment of biodiversity goals (Pringle 2001), and is increasingly recognized as a critical consideration in realizing conservation and resource management objectives for both terrestrial and freshwater biodiversity (*e.g.*, Pringle 2001, Abell *et al.* 2007, Hermoso *et al.* 2016). To address hydrology, terrestrial ecoregions (Figure 10) were blended with hydrology-based units (Figure 11), to eliminate artificial barriers to hydrologic connectivity (Figure 12). In addition to addressing barriers to hydrology, this pairing of units captures important information reflected in both the heterogeneity of terrestrial and aquatic communities. Further, both units are elements of hierarchical frameworks (*e.g.*, National Ecological Framework of Canada), thereby supporting analysis and planning at multiple spatial scales, if desired.

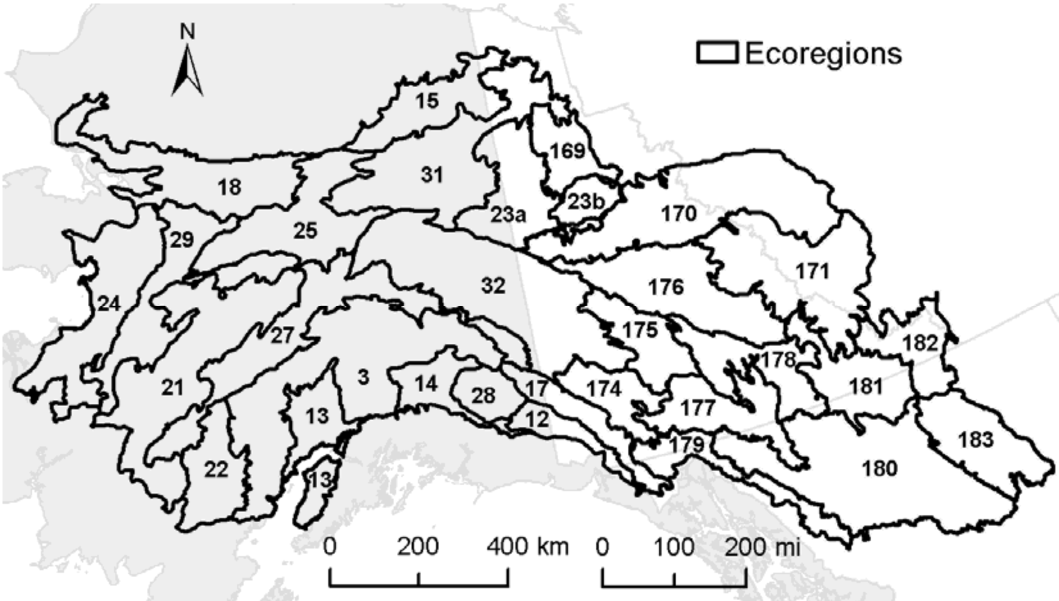

Figure 10. The NWBLCC planning region is comprised of 30 ecoregions defined by the Unified Ecoregions of Alaska (Nowacki *et al.* 2001) and the National Ecological Framework of Canada (Marshall *et al.* 1999). Ecoregion 23 was further stratified by ecodistricts to account for differences in vegetation cover during the estimation of MDRs.

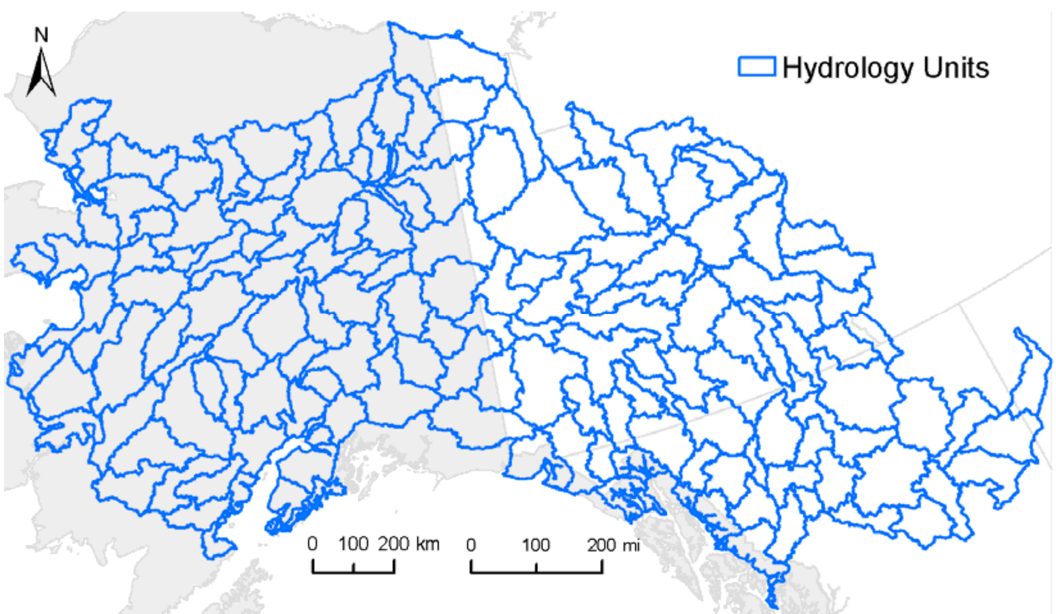

Figure 11. Canada's Fundamental Drainage Areas (FDAs; NRCAN 2003) and Alaska's Hydrologic Units 8-digits (HUC8; USGS 2014) were harmonized to create a continuous coverage of hydrology units for the NWBLCC planning region (BEACONs 2015c).

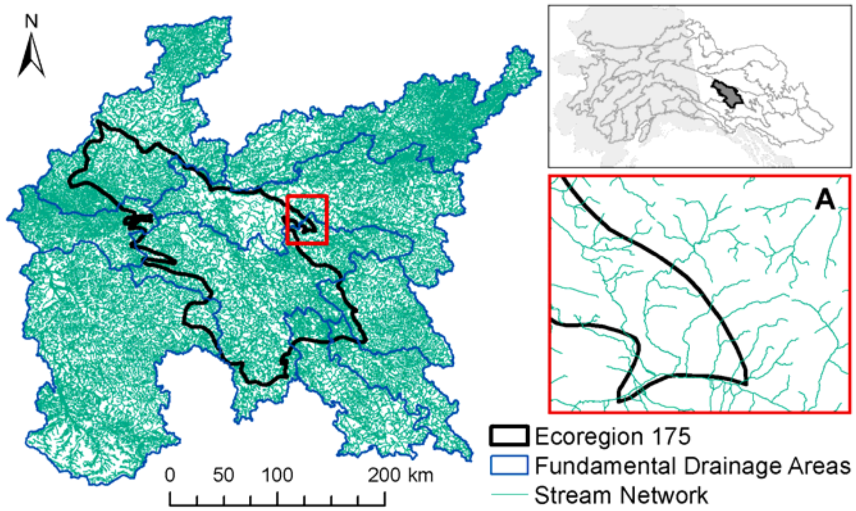

Figure 12. When designing benchmarks, ecoregions (Figure 10) were buffered by hydrology units (Figure 11) to remove the barrier to hydrology imposed by ecoregion boundaries. Inset A illustrates stream networks bisected by the ecoregion boundary.

Evaluation of Existing Protected Areas

Given their existing status, protected areas (PAs) are strong candidates for benchmarks (Figure 13). However, most PAs were not designed with that role in mind. In addition to identifying new benchmark areas, existing PAs were evaluated for their benchmark potential using the benchmark design elements described above (Figure 14). To qualify as a potential benchmark, the area of the PA benchmark intersecting the ecoregion must be ≥ 80% MDR, the size of a subsystem-level benchmark. For analysis, protected areas sharing a border were dissolved, and treated as a single protected area.

Assessment and Rank of Benchmark Network Options

Benchmarks can be ranked using metrics that describe the design criteria described above, as well as additional criteria. We ranked benchmark network options using the following:

- ***Fundamental benchmark properties -*** *Properties of benchmark structure can also be used to inform the selection of benchmarks from a suite of options. These properties include measures of internal and external vulnerability to human disturbances (e.g., upstream area and shape) and internal hydrologic connectivity.*

When designing and selecting ecological benchmarks, we must consider the potential for the quality and function of a benchmark to be impacted by human disturbances from not only within, but from beyond the benchmark boundary as well. Benchmarks are vulnerable to external influences via edge effects and the flow of surface and ground waters. Due to the extensive longitudinal connectivity of aquatic systems, particularly rivers, benchmarks are vulnerable to anthropogenic alterations or inputs from both upstream and downstream sources (Pringle 2001), including the loss of nutrients (*e.g.*, organic inputs from riparian areas), diversion of water, and pollutants. For benchmarks to best serve as controls, the benchmark should contain a well-connected hydrologic network and have little to no vulnerability to human disturbance via the stream network and edge effects. Given this objective, benchmark networks were ranked based on internal and external vulnerability to anthropogenic disturbance, as well as the hydrologic connectedness of the stream network within benchmarks.

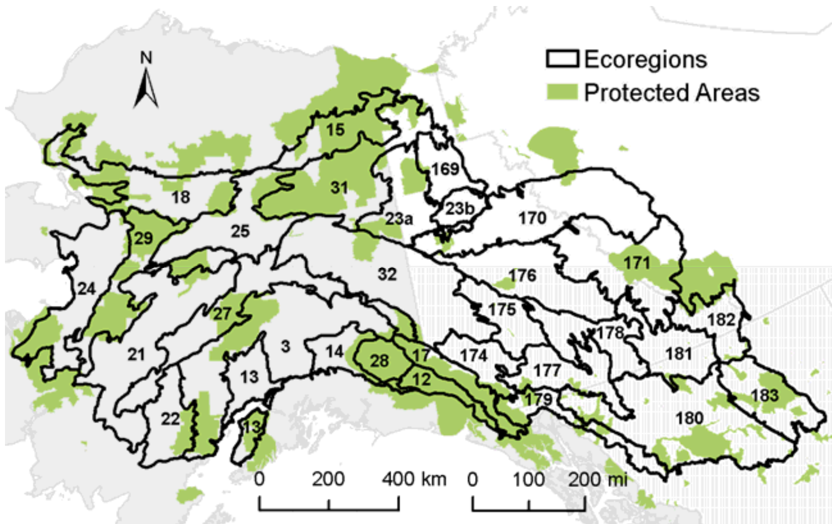

Figure 13. The existing protected areas dataset is a compilation of CARTS (CCEA 2014) and PAD-US v1.3 (USGS 2012). The benchmark potential of existing protected areas was restricted to those areas with a high degree of protection. In Alaska, this includes National Parks, Designated Wilderness Areas, Designated Wild Scenic Rivers, National Wildlife Refuges, and National Wilderness Areas. In addition to CARTS, Canada's protected areas included two proposed protected areas with government sponsorship in the Yukon, Kusawa and Asi Keyi Natural Environment Parks.

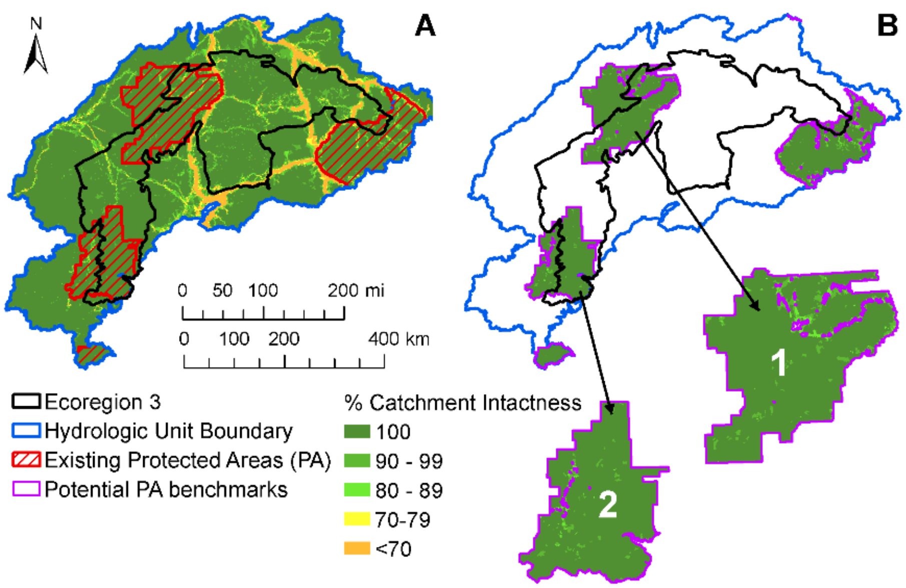

Figure 14. (A) To identify protected area (PA) benchmarks, PAs were clipped to the planning region, defined as the ecoregion plus intersecting hydrology units (HUC8 and/or FDA). (B) Next, patches of a specified catchment-level intactness (*e.g.*, ≥ 80%) were identified in the protected areas. The size of each patch was compared to the MDR for the ecoregion. In this example, three patches are ≥ MDR. To be included in the design of benchmark networks, the ecoregion portion of the PA benchmark must be ≥ 80% MDR in size. As such, only two of the three patches (labelled 1 and 2) have sufficient overlap to be included in further analyses.

Internal vulnerability addresses the presence of human disturbance (or low intact catchments) within the boundary of the benchmark network (*e.g.*, Figure 17B), and was measured as the proportion of area within the outer boundary of the network with low catchment intactness (*i.e.*, < 80%). Benchmarks with a lower proportion of disturbance would have lower internal vulnerability and thus rank higher.

External vulnerability was measured based on upstream area, upstream intactness, and benchmark shape. Streams upstream and downstream of a benchmark are potential sources of external vulnerability to outside influence mediated by water flows (Figure 15). For this analysis, we evaluated upstream vulnerability only; however, results for downstream vulnerability are available. Benchmark networks with low upstream area and high upstream intactness ranked higher. Irregular shapes should be avoided to minimize the intrusion of edge effects and vulnerability of the benchmark to outside influences. Shapes more closely resembling a circle were ranked higher (Figure 16). Shape was calculated at the benchmark-level. Because networks can be comprised of more than one benchmark, networks were ranked using the benchmark in the network that least resembled a circle.

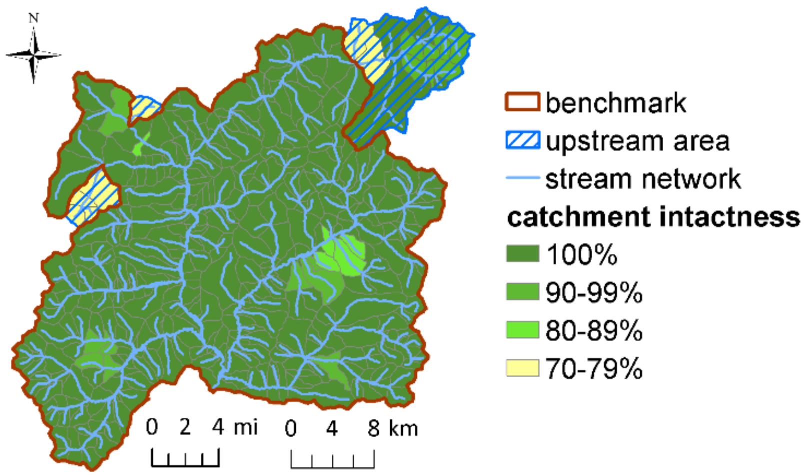

Figure 15. The vulnerability of benchmark networks to external sources via water flows was measured using upstream area and upstream intactness. Upstream area is the total area of catchments upstream. Upstream intactness is the mean area-weighted intactness of catchments upstream of the benchmark.

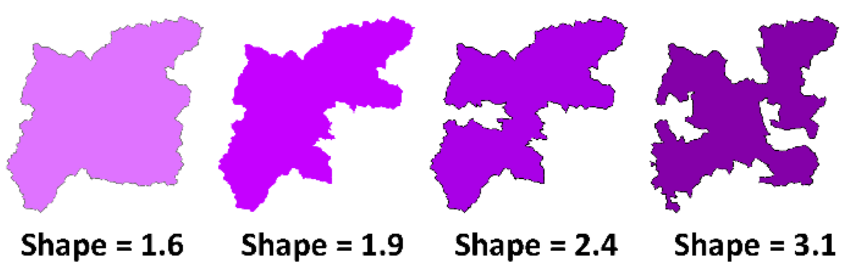

Figure 16. **Shape Index** describes the shape of the benchmark relative to a circle. Shape is measured with a standard edge/area ratio metric that measures the complexity of patch shape as the ratio of patch perimeter to that of a circular patch of equal area (McGarigal and Marks 1994). Thus SHAPE = 1 for a circular patch, and increases without bound as patch shape becomes increasingly irregular.

To measure the **internal hydrologic connectivity** of the stream network within benchmarks, we used the Dendritic Connectivity Index (DCI) which quantifies the "longitudinal connectivity of river networks based on the expected probability of an organism being able to move freely between two random points of the network" (Cote *et al.* 2009; Figure 17). The index ranges from 0 to 1 or low to high longitudinal connectivity, respectively. Reduced connectivity can occur when catchments with low intactness are ignored during benchmark construction, resulting in a fragmented stream network (Figure 17B). DCI was calculated at the benchmark-level. Because networks can be comprised of more than one benchmark, networks were ranked using the benchmark in the network with lowest DCI value.

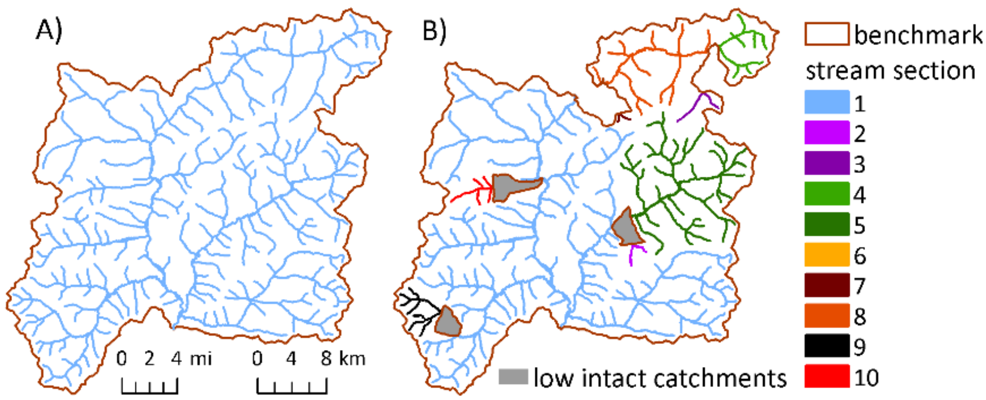

Figure 17**. Dendritic Connectivity Index (DCI)** measures the longitudinal connectivity of the stream network within a benchmark (Cote *et al.* 2009). The index ranges from 0 to 1 or low to high longitudinal connectivity, respectively. A) Benchmark A has a continuous stream network with DCI = 1. B). Relative to Benchmark A, Benchmark B illustrates how the presence of low intact catchments (grey), which are ignored during benchmark construction, can result in a fragmented stream network within the benchmark. The fragmented network is comprised of 10 sections with a DCI = 0.48. DCI was calculated using National Hydrological Network (NHN; 1:50,000 or better) Version 14 (NRCAN 2011-2014), and National Hydrology Dataset (NHD; 1:24,000; USGS 2014a). The fine-scale resolution of the stream network often results in multiple isolated (as opposed to fragmented) stream networks within a benchmark. As such, DCI was calculated as a mean length-weighted DCI for isolated stream networks.

The fundamental benchmark properties used to rank benchmark networks are summarized in Table 3.

- ***Climate change resilience*** *– Changes in patterns of environmental variation are expected under climate change. Benchmark networks can be ranked based on the ability to remain representative under predicted future climate, as well as refugia and colonization potential as species are forced to migrate with moving climates.*

The boreal is predicted to experience significant warming, and associated change, over the coming decades (Price *et al.* 2013, Chapin *et al.* 2014). The four measures of environmental variation used to design benchmark networks (Figure 5) are based on current climatic conditions and do not account for the influence of climate change on the ability of benchmark networks to remain representative of the planning region (*i.e.*, ecoregions). To address this, we ranked benchmark networks based on their ability to remain representative of two climate-projected (2041-2070) multivariate indicators of climatic conditions, which we refer to as Temperature and Precipitation Indicators given the explanatory power of temperature and precipitation variables in each indicator, respectively (Figure 18). Representation was measured using the Kolmogorov-Smirnov Dissimilarity Metric (Figure 8).

To assess the ability of benchmark networks to support biodiversity under climate change, we evaluated the potential for species to persist within, and colonize, benchmark networks, using forward and backward climate velocity (2041-2070; Figure 18), respectively. Networks were ranked based on the geometric means of forward and backward climate velocity (Loarie *et al.* 2009). For both datasets, higher climate velocities indicate greater vulnerability to species loss. Lower forward velocities indicate higher refugia potential for species, whereas lower backward velocities indicate higher colonization potential. As such, benchmark networks with lower velocities ranked higher.

The climate change properties used to rank benchmark networks are summarized in Table 4.

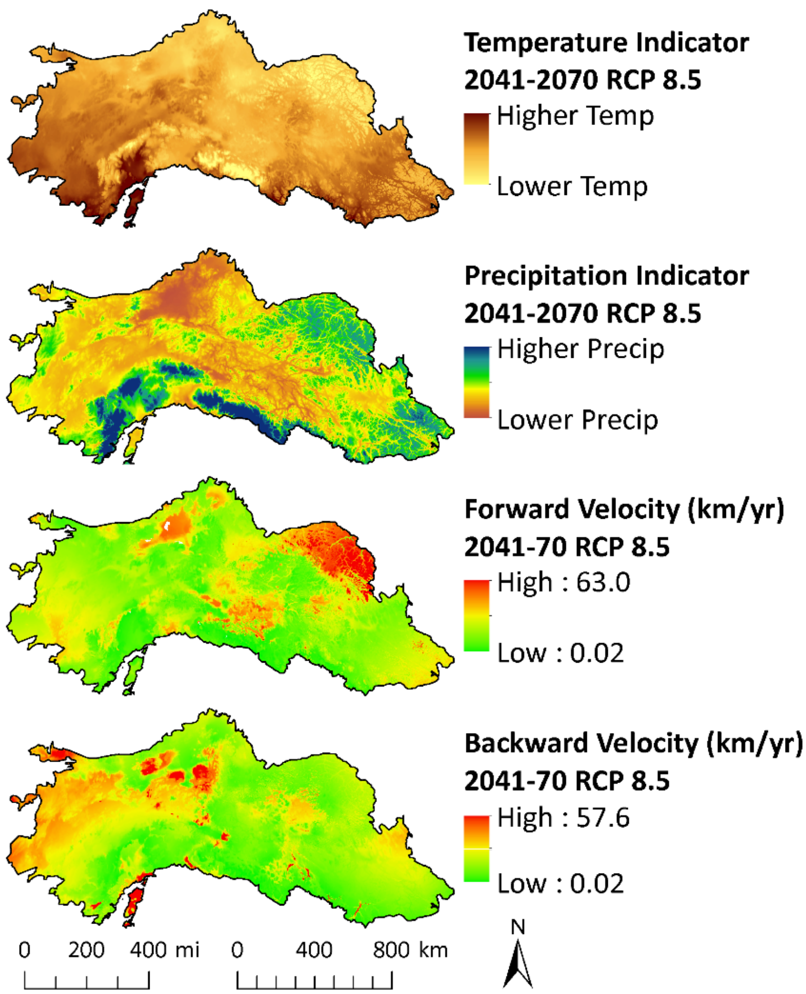

Figure 18. Benchmark networks were ranked using climate-projected datasets for the period 2041-2070 that were created using RCP 8.5, the Representative Concentration Pathway with the highest greenhouse gas emissions from IPCC (2014). **Temperature** and **Precipitation Indicators** are 1-km^2 resolution multivariate characterizations of climate based on Principal Components that are dominated by temperature and precipitation variables, respectively (AdaptWest Project 2015). **Forward and Backward Climate Velocity** (km/yr) are 1-km^2 resolution datasets, derived from the mean projection of 15 CMIP5 models (AdaptWest Project 2015). Forward climate velocity measures "the rate at which an organism in the current landscape has to migrate to maintain constant climate conditions" (Hamann *et al.* 2014). **Backward Climate Velocity** is the minimum rate of migration required by an organism to colonize a grid cell when moving from a cell of equivalent climate conditions (Hamann *et al.* 2014; AdaptWest Project 2015).

- ***Representation of priority focal species*** - *Benchmark networks can be ranked based on their ability to contribute towards conservation goals for focal species, including the protection of habitat and climate refugia.*

Up to this point, benchmark networks have been designed solely using coarse-filter indicators or surrogates for biodiversity (Figures 6 and 18). The introduction of fine-filter indicators, such as focal species, serves three objectives: (1) incorporate species that may not be well represented by coarse-scale indicators (*e.g*., rare, endangered, or migratory; Groves 2003), (2) validate the effectiveness of the coarse-filter indicators at representing biodiversity, and (3) consideration of species-specific conservation goals when selecting from benchmark network options, including species favoured for monitoring and adaptive management. With these objectives in mind, NWBLCC Partners identified a suite of priority focal species based on vulnerability to landscape change and social/cultural importance (Table 2) that would be used to rank benchmark network options, for a more robust selection process. To support this action, a comprehensive review of conservation/management plans and available data and models was undertaken (see Focal Species Report; BEACONs 2017b).

Benchmark networks contribute towards focal species goals and targets via the protection of habitat. Species data and models can be used to evaluate the amount of habitat protected within benchmarks using static spatial overlays and more sophisticated dynamic landscape simulation modelling (*e.g.*, Leroux *et al.* 2007b). We acquired habitat data, which supported the former only: current habitat, predicted future habitat, and climate refugia (Table 2, Figure 19). A thorough review of conservation/management plans did not reveal specific conservation targets, so benchmark networks were ranked based on maximizing the amount of habitat represented. The measures of focal species habitat used to rank benchmark networks are summarized in Table 5.

Table 2. Priority focal species selected by the NWBLCC Partners to support the ranking of benchmark networks, and the broad datasets types available for each species. See Focal Species Report for more details (BEACONs 2017b).

Focal Species	Current Habitat	Future Habitat	Climate Refugia
Caribou	X		
Sheep	X		
Moose	X		
Beaver	X		
Chinook Salmon	X		
Chum Salmon	X		
Waterfowl Guild[1]	X		
Rusty Blackbird	X	X	X
Old-Forest Bird Guild[2]	X	X	X

[1] Waterfowl guild is comprised of three species: Trumpeter Swan, Lesser Scaup, and White-Winged Scoter.
[2] Old-Forest Bird Guild is comprised of five species: Boreal Chickadee, Brown Creeper, Pine Grosbeak, White-Winged Crossbill, and Swainson's Thrush.

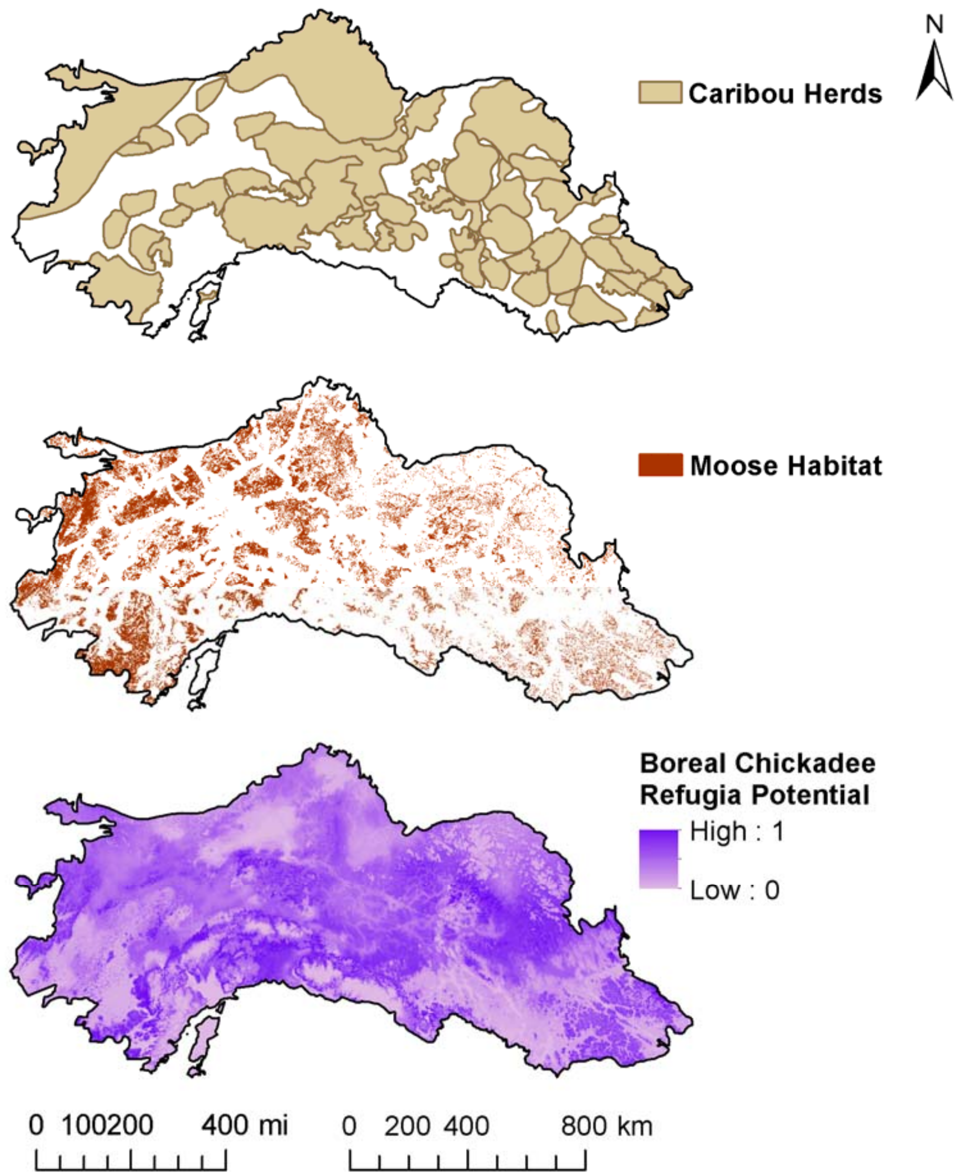

Figure 19. Benchmark networks were ranked based on the representation of focal species habitat. A variety of habitat-based datasets were used, including boundaries of **Caribou herds** (AK Dept. Fish and Game, YK Environment Yukon, BC Environment), M**oose Habitat** identified as intact areas of deciduous and shrub landcover classes (BEACONs 2016b; Figure 5), and **Boreal Chickadee Refugia Potential**, which ranges from 0 to 1, or low to high potential, respectively (Stralberg *et al.* in review; Box 1). For a list of all species datasets used to rank benchmark networks, see Focal Species Report (BEACONs 2017b).

Box 1. Refugia Case Study for NWBLCC

Diana Stralberg, University of Alberta

BEACONs' Conservation Matrix Model identifies alternative benchmark configurations that remain representative of landscape conditions over time, under a changing climate, but it does not guarantee persistence of all individual species, especially those with limited resources. Thus, for a suite of focal species, we used species distribution models and future climate projections to estimate future refugia potential, which was used as a post-hoc filter to rank benchmark networks.

Climate refugia may be defined as *in situ* or *ex situ*, with the former limited to areas of current climatic suitability for a species. *Ex situ* refugia may vary in proximity to a species' current distribution, with major implications for their conservation value. Thus, the concept of climate velocity (Loarie *et al.* 2009)—the speed at which an organisms must migrate to keep pace with climate change— is useful to compare and evaluate *ex situ* refugia. Climate velocity metrics have been used to identify species and ecosystems that are most vulnerable to future climate change, as indicated by high climate velocity (Loarie *et al.* 2009, Serra-Diaz *et al.* 2014, Barber *et al.* 2015). Using a nearest analog-based approach, both forward and backward velocity can be calculated, providing complementary information about patio-temporal responses to climate change (Hamann *et al.* 2014, Carroll *et al.* 2015). In particular, backward velocity calculations—and the corresponding distance traveled to reach a given future suitable climate—can be used to identify areas of high potential refugium value for a given time period and species (Stralberg *et al.* in review). Velocity-based refugia for a given species represent areas of future climatic suitability that are in close geographic proximity to currently occupied areas. That is, they represent the places where chances of rapid colonization (or persistence) in response to climate change is high.

For each of our six focal passerine bird species, we calculated an index of refugia potential based on backward velocity using an approach developed by Stralberg *et al.* (in review). Using species density projections for baseline and future (2041-2070) time periods (Stralberg *et al.* 2015), we converted density estimates within 4-km x 4-km grid cells for each species to binary estimates of its suitable core habitat, defined as areas where the species' predicted density exceeded its mean baseline predicted density within the boreal and sub-boreal model-building area. For each species *i* and for each GCM, we calculated the distance (d_{ij}) from each future (2041-2070) distribution pixel *j* to the nearest baseline distribution pixel, calculated its negative natural log (L_{ij}), and then converted it to a refugium index (R_{ij}) that was normalized via feature scaling to a common 0 to 1 scale. That is:

$$R_{ij} = \frac{L_{ij} - L_{min}}{(L_{max} - L_{min})}$$

In order to ensure the regional viability of refugia, we constrained projections to the ecoregion unit in question, requiring that relative velocity values were based on portions of the species' distributions contained within that ecoregion. Refugia values were then summed across each watershed unit and used as one of several ranking criteria for benchmark networks.

To illustrate the concept, mid-century refugia potential for Boreal Chickadee and Rusty Blackbird within the Northwest Boreal LCC planning region is portrayed in Figure 1.1. The wide-ranging Boreal Chickadee has higher refugia potential overall than the more range-restricted Rusty Blackbird. Both species have a greater refugia potential in ecoregion 32 than in ecoregion 21.

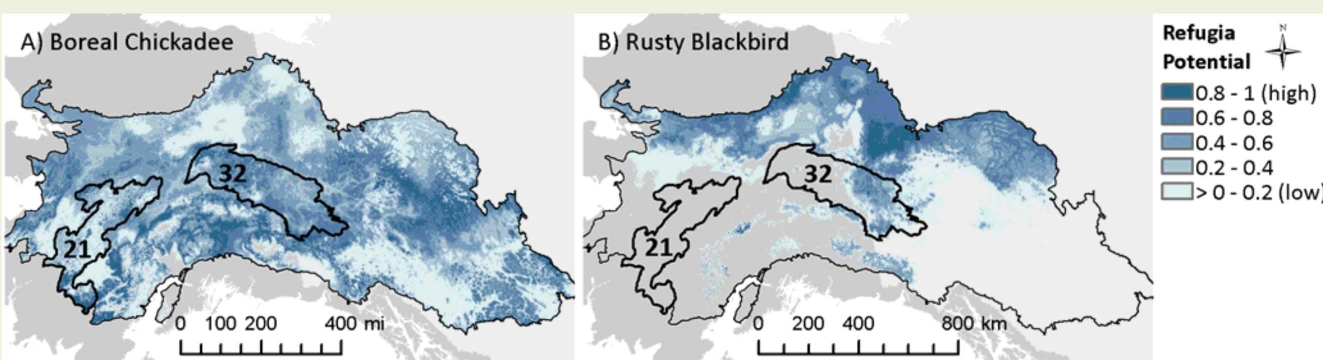

Figure 1.1. Refugia potential for A) Boreal Chickadee and B) Rusty Blackbird in the Northwest Boreal LCC planning region. For both species, the refugia potential of ecoregion 32 is greater than ecoregion 21.

Table 3. Summary of **Fundamental Benchmark Properties** used to rank benchmark networks. These properties are described in detail earlier in the document.

Fundamental Benchmark Property	Range of Values	Ranking
Internal Vulnerability – proportional area of low intact catchments	0 to 1	Lower values rank higher
External Vulnerability – upstream area	≥ 0 km^2	Lower values rank higher
External Vulnerability – intactness of upstream area	0 to 100%	Higher values rank higher
External Vulnerability – shape	≥ 1, where 1 = circle	Lower values rank higher
Internal Hydrologic Connectivity – longitudinal connectivity	DCI = 0 (low) to 1 (high) longitudinal connectivity	Higher values rank higher
Representation – Dissimilarity Metrics	KS[1] & BC[2] = 0 (high) to 1 (low) representation	Lower values rank higher

[1] Kolmogorov-Smirnov dissimilarity metric for continuous indicators
[2] Bray-Curtis dissimilarity metric for categorical indicators

Table 4. Summary of **Climate Change Resilience** measures used to rank benchmark networks. These properties are described in detail earlier in the document.

Climate Change Resilience	Range of Values	Ranking
Maintain Representation - temperature and precipitation indicators	KS[1] = 0 (high) to 1 (low) representation	Lower values rank higher
Refugia Potential – mean forward climate velocity	≥ 0 km^2	Lower values rank higher
Colonization Potential – mean backward climate velocity	≥ 0 km^2	Lower values rank higher

[1] Kolmogorov – Smirnov dissimilarity metric for continuous indicators

Table 5. Summary of **Focal Species Habitat** measures used to assess the representation and rank benchmark networks. These properties are described in detail earlier in the document.

Focal Species Habitat	Range of Values	Ranking
Area of habitat (All species)	≥ 0 km^2	Higher values rank higher
in situ and *ex situ* refugia potential (Old-Forest Birds)	0 (low) to 1 (high) potential	Higher values rank higher
Density of breeding pairs (Waterfowl)	# pairs per km^2 ≥ 0 km^2	Higher values rank higher

Weighted Ranks

Benchmark network options were ranked using a weighted-rank approach that is similar to the use of delta AIC and Akaike weights to rank statistical models (Burnham and Anderson 2002). Instead of using AIC values, we used the actual network values calculated for fundamental benchmark properties, climate change, and focal species.

The weighted rank (w_i) for a network is calculated as

$$w_i = \exp(-0.5\Delta_i) / \mathrm{sum}(\exp(-0.5\Delta i))$$

where $\Delta i = \text{network}_{max} - \text{network}_i$
when larger values rank higher (*e.g.*, amount of habitat)

OR

where $\Delta i = \text{network}_i - \text{network}_{min}$
when smaller values rank higher (*e.g.*, climate velocity).

Weights for all networks combined add up to 1, with larger weights ranking higher (Table 6). In comparison to a simple ranking system that treats the differences in ranks as being equal, differences amongst weighted ranks reflect size differences amongst the values being ranked. Because of the exp() components of the equation, a difference of ≥10 between the min and max values, from the suite of values to be ranked, creates small weighted ranks (< 1/10,000) that are difficult to interpret. To address this, values associated with areal measures (*i.e.*, km^2 or mi^2) were rescaled from 0 to 1 prior to ranking. In cases where all networks had the same value, the networks were given the same rank with the sum of weighted ranks equal to 1.

The output from the analysis is available such that users can apply an alternative ranking approach or use a subset of attributes/species for ranking benchmark networks if desired.

Table 6. Five networks are ranked based on three measures for Boreal Chickadee habitat: amount of current habitat (km^2), and mean *in situ* and *ex situ* refugia potential, which ranges from 0 to 1 or low to high potential, respectively. For all three habitat measures, larger values rank higher. A weighted rank (**WR**) was calculated for each attribute. An overall rank for the network was calculated as the mean WR. Network 2 is the top ranked network for Boreal Chickadee habitat with **Mean WR** = 0.217.

Network	**Habitat**		***In situ* Refugia Potential**		***Ex situ* Refugia Potential**		**Mean Weighted Rank**
	km²	**WR**	**mean**	**WR**	**mean**	**WR**	
1	4128	0.218	0.38	0.191	0.37	0.191	0.200 **3**
2	4416	0.230	0.57	0.210	0.57	0.211	0.217 **1**
3	3568	0.197	0.47	0.200	0.46	0.200	0.199 **4**
4	4016	0.214	0.59	0.212	0.57	0.211	0.210 **2**
5	1632	0.139	0.32	0.185	0.30	0.185	0.169 **5**

Benchmark Resilience to Natural Disturbance

Benchmarks are designed to be resilient to natural disturbance (*e.g.*, fire), such that internal recolonization sources for vegetation types vulnerable to natural disturbance are maintained within the benchmark at all times. Internal recolonization sources serve as life-boats for species that rely on vulnerable habitats, such as flammable vegetation types in the case of fire. Resilience to natural disturbance is addressed within the design process by using Minimum Dynamic Reserves to inform benchmark size (see MDR Report; BEACONs 2017a). However, MDRs are minimum estimates that may not be sufficient for all areas within the planning region, given variability in fire behaviour and the spatial arrangement of vegetation types. As such, a secondary testing of resilience through dynamic simulation modelling is recommended for all benchmark networks. Given the extent of the NWBLCC planning region, and the large number of network solutions, this step was not included in the analysis, but rather illustrated for Ecoregion 22 below. All benchmark networks identified as strong candidates should undergo this evaluation.

Ecoregion 22 Case Study – Testing the Resilience of Benchmarks to Natural Disturbance

We illustrate the secondary testing of resilience using benchmarks from five benchmark networks identified for Ecoregion 22 – Lime Hills (Figure 20). The methods and inputs used to evaluate the resilience of benchmarks to fire are the same as those used to identify the MDR for the ecoregion (see MDR Report). The resilience of vegetation communities to fire in the benchmarks of the five networks was evaluated using the dynamic landscape model CONSERV (BEACONs 2015d, Leroux *et al.* 2007a), which simulates vegetation succession and fire over large-spatial and temporal scales. For a benchmark to be resilient to fire, it must maintain <u>minimum amounts</u> of all flammable vegetation types for every year over a 250-year simulation, repeated 100 times. Multiple simulations are done to characterize the stochastic nature of this disturbance. The minimum amount required to be maintained in this evaluation was conservative at 1 km^2. Where specific area requirements for a species or monitoring program is known, these area requirements could, for example, be used to set the minimum. Five broad flammable vegetation types were identified for the simulation: black spruce, white spruce, deciduous north, deciduous south, and shrub tundra (Table 5). During the simulation, none of the flammable vegetation types dropped below 1 km^2, indicating that all benchmarks are resilient to fire based on the modest requirement of 1 km^2.

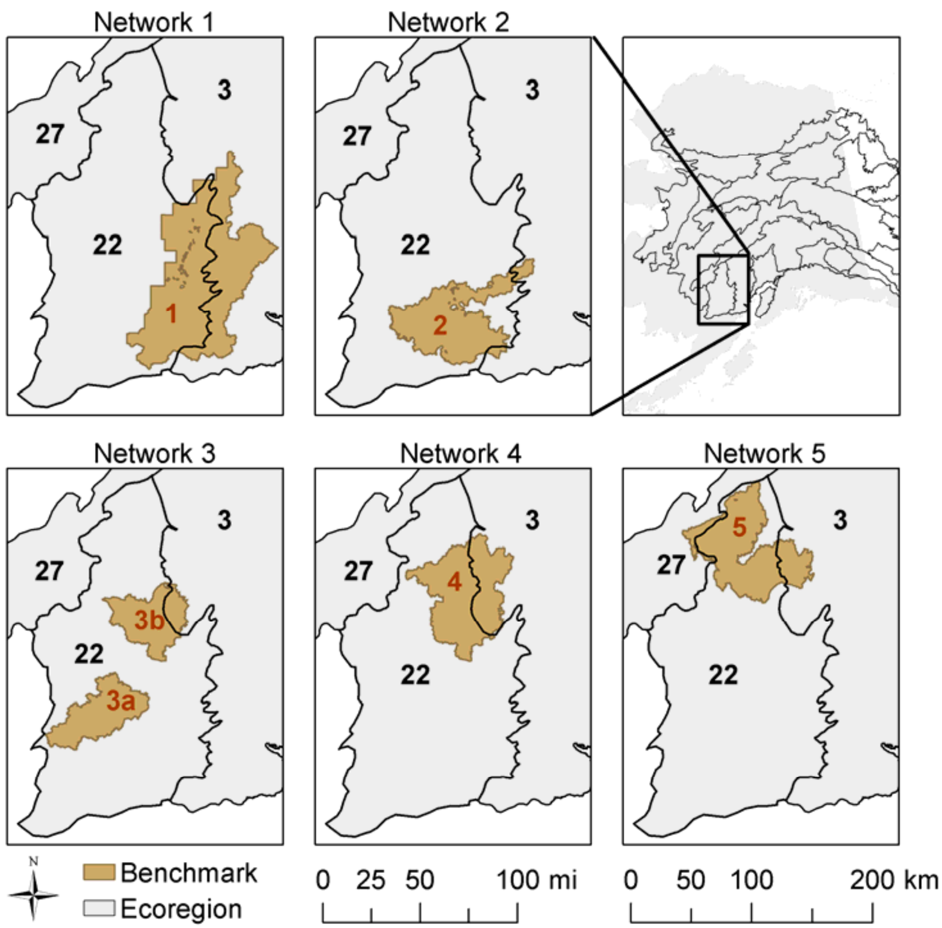

Figure 20. Locations of five benchmark networks in Ecoregion 22. Network 1 is an existing protected area. Networks 2, 4 and 5 are each comprised of two overlapping system-level benchmarks that were treated as a single benchmark for the simulation. All networks cross into a neighbouring ecoregion(s). The spatial extent of CONSERV simulations included all ecoregions intersecting the benchmarks tested, and ecoregion-specific model parameters were used. Benchmark labels correspond to ID in Table 4.

Table 5. The simulation tracked five flammable vegetation types: black spruce, white spruce, deciduous north, deciduous south, and shrub tundra. **Start** is the amount of each vegetation type in the benchmark at the start of the simulation. **Min** is the minimum amount of each vegetation type across all simulated years and replications (250 years x 100).

ID	Black spruce (km^2)		White spruce (km^2)		Deciduous North (km^2)		Deciduous South (km^2)		Shrub tundra (km^2)	
	start	min	start	min	start	min	start	min	start	min
1	531	483	787	567	1933	1254	1100	428	1038	1025
2	313	243	474	313	1582	871	770	259	587	549
3a	156	26	189	14	1204	343	618	96	276	252
3b	263	99	255	58	1053	346	596	73	124	112
4	344	74	534	113	1931	788	882	94	341	320
5	351	137	936	300	2063	1135	910	218	247	227

RESULTS

The largely intact landscape of the NWBLCC planning region supports the construction of system-level benchmarks across 90% of the region (Figure 21). Protected area system-level benchmarks were identified in 18 ecoregions, including ecoregions 12 and 28 which do not have MDR estimates (Table 6). Despite the lack of MDR estimates, ecoregions 12 and 28 have 100% protection, so are considered benchmarked. New system-level benchmarks were identified in all ecoregions, except for ecoregions without MDR estimates (12, 17, and 28; Figure 22). The number of new system-level benchmarks varied across ecoregions from 54 to 2,050. This variability reflects the range of ecoregion sizes and MDR estimates. New benchmarks often have a high degree of overlap and some solutions may differ by only a few catchments.

Representative benchmark networks comprised of system-level benchmarks were identified for all ecoregions (N = 10 to 174; Table 6), except for ecoregion 17. Representative protected area system-level benchmark networks were identified in 14 ecoregions, and protected area system-level benchmarks contributed to benchmark networks in an additional 4 ecoregions (Figure 22). Benchmark networks comprised of new benchmarks only were identified for all ecoregions, except for ecoregions without MDR estimates (12, 17, and 28).

To identify benchmarks for ecoregion 17, alternative methods are required to identify benchmark size, such as the needs of species and/or monitoring program. Ecoregion 17 has 88% protection, which includes all headwaters; further, all land cover classes have high protection (62-100%) except for shrub-lichen-moss which is relatively rare (6 km^2) with 8% of its area protected. As such, the potential for existing protected areas to serve as a system-level benchmark for ecoregion 17 is high.

Results for ecoregions are detailed in individual reports. These reports include the ranking of benchmark networks based on fundamental benchmark properties, climate change, and focal species. Ecoregion reports, and interactive maps and tables, can be accessed at www.beaconsproject.ca/nwb.

Table 6. Number of protected area (PA) and new system-level benchmarks and representative benchmark networks identified within each ecoregion. All system-level benchmarks were designed using a minimum catchment-level intactness of ≥ 80%. Representative benchmark networks were identified using MDR-based targets for climate moisture index, gross primary productivity, lake-edge density, and landcover. Networks were assembled from PA benchmarks only, a combination of PA and new benchmarks, and new benchmarks only. Ecoregions 12, 17, and 28 do not have MDR values. Despite this, PA benchmarks and benchmark networks were identified for ecoregions 12 and 28 because these ecoregions have 100% protection. Note ecoregion 23 was subdivided into 23a and 23b based on ecodistricts.

Ecoregion	Number of System-Level Benchmarks Identified		Number of Representative Benchmark Networks comprised of:			
	PA Benchmarks	New Benchmarks	PA Benchmarks	PA & New Benchmarks	New Benchmarks	Total
3	2	782	3	–	136	139
12	1	No MDR	1	–	–	1
13	2	653	1	–	18	19
14	1	278	1	–	29	30
15	1	288	1	–	25	26
17	No MDR	No MDR	–	–	–	–
18	1	523	–	25	23	48
21	–	428	–	–	70	70
22	1	282	1	–	57	58
23a	2	241	1	–	76	77
23b	–	421	–	–	24	24
24	–	149	–	–	41	41
25	–	121	–	–	115	115
27	1	321		29	142	171
28	1	No MDR	1	–	–	1
29	2	412	–	8	166	174
31	2	649	1	–	60	61
32	–	599	–	–	50	50
169	–	263	–	–	42	42
170	2	2,050	2	–	155	157
171	1	348	1	–	22	23
174	–	54	–	–	99	99
175	–	359	–	–	157	157
176	–	726	–	–	37	37
177	–	407	–	–	19	19
178	–	442	–	–	52	52
179	3	159	2	–	152	154
180	6	1,543	7	–	139	146
181	–	501	–	–	10	10
182	1	240	–	66	9	75
183	2	187	1	–	168	169

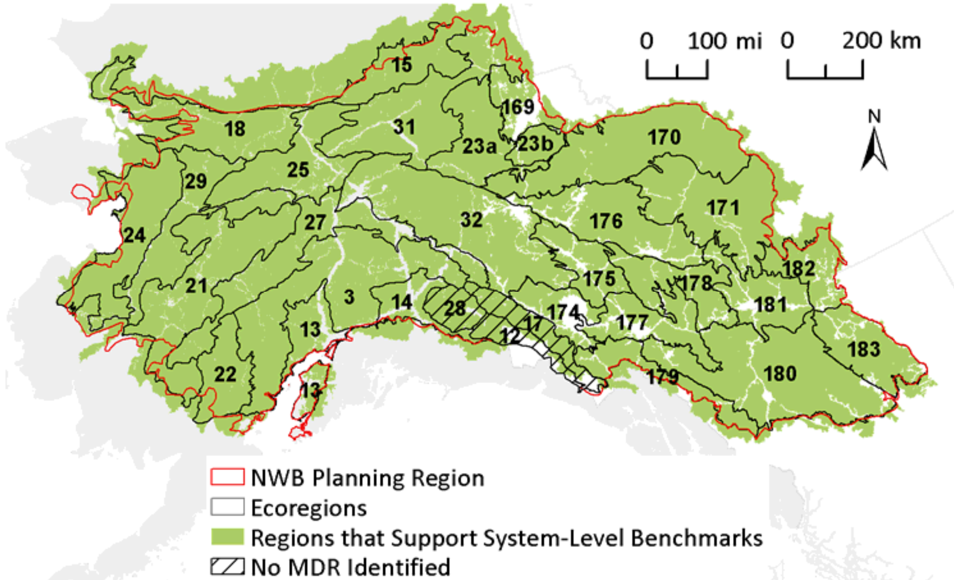

Figure 21. Regions that support the construction of system-level benchmarks using Minimum Dynamic Reserve estimates and a minimum catchment-level intactness of 80%. Regions extend beyond the NWBLCC planning region boundary and into ecoregions without MDR estimates (12, 17, and 28) due to the buffering of ecoregions by hydrology units (Figure 12).

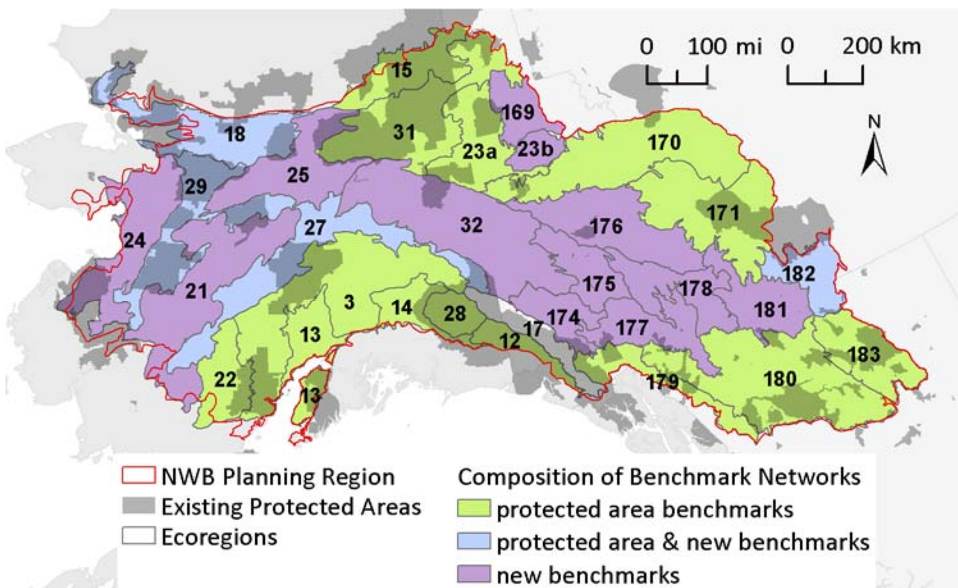

Figure 22. Representative system-level benchmark networks were identified for all ecoregions, except for ecoregion 17, where there is no MDR estimate. While MDR estimates do not exist for ecoregions 12 and 28, these ecoregions have 100% protection, so are benchmarked. Protected area (PA) system-level benchmarks solely comprise or contribute to representative benchmark networks in 18 ecoregions (green and blue, respectively). Networks designed solely from new benchmarks are required for 12 ecoregions (purple). While some protected areas may not meet the size and intactness criteria for a system-level benchmark, they may still contribute to portions of new benchmarks that spatially overlap some or all of the protected area. Regardless of the benchmark potential of existing protected areas, networks comprised solely of new benchmarks were identified for all ecoregions with MDR estimates (N=28). Note ecoregion 23 was subdivided into 23a and 23b based on ecodistricts.

GUIDELINES FOR USERS

Given the spatial extent of the analysis and the volume of solutions, refined benchmark design was not possible, and a more detailed examination of solutions is warranted. This section of the report provides guidance on how to use and refine benchmark network options, as well as additional values/attributes that can be used to further inform the selection of benchmark networks.

Flexible Approach

The process for designing benchmarks and representative benchmark networks is flexible to the use of alternative inputs and/or additional datasets. For example, while not an issue for the NWBLCC planning region, the greatest limitation to identifying benchmarks is most often low landscape intactness. For regions with extensive human disturbance, the use of regional datasets can improve benchmark potential, particularly datasets that provide a more detailed description of human disturbance that allow the user to explore solutions that ignore non-permanent disturbance types that can be targeted for reclamation (*e.g.*, cutblocks, seismic lines). Representative benchmark networks were identified using four biophysical indicators: Climate Moisture Index, Gross Primary Productivity, Lake-Edge Density, and MODIS-based land cover. Alternative and/or additional biophysical indicators can be used. For example, regional datasets are likely to better characterize the planning region, such as finer-scale forest inventory derived from air photos and ground plots rather than satellite-based land cover (*e.g.*, BC and Yukon vegetation and forest inventories). Measures of geodiversity such as land facets or enduring features could also be included (Beier *et al.* 2015).

Resilience to Natural Disturbance

Benchmarks for the NWBLCC were designed to be resilient to natural disturbance, namely fire. This resilience is addressed within the benchmark design process by using Minimum Dynamic Reserves to inform benchmark size (see MDR Report; BEACONs 2017a), and through secondary testing of benchmark networks with dynamic landscape simulation modelling. Given the magnitude of this study, the latter was not performed, but is recommended once candidate benchmark networks have been selected. With regards to the former, MDR estimates were identified to maintain minimum amounts of vulnerable vegetation types (*i.e*., 1 km^2), and as such, should be considered conservative. For a more robust estimate, the MDR estimates could be refined to maintain vegetation amounts based on the needs of a monitoring program and/or species. Further, the MDR estimates do not account for the interaction of multiple natural disturbance types (*e.g.*, fire, insects, disease, wind, *etc*.) operating on the landscape, and in some cases, may not be based on the largest-scale natural disturbance type in the ecoregion. For example, in northern British Columbia, ecoregion 181 has one of the smallest MDR estimates (1,089 km^2; Figure 3); however, mountain pine beetle (MPB) is a significant disturbance within this ecoregion, and its range is expected to expand north (BCMFLNRO 2017). Ecoregion 181 may require an MDR larger than the fire-based MDR used in this analysis to maintain resilience to both fire and MPB. When selecting from the benchmark networks identified by this analysis, a review of natural disturbances within the area of interest is recommended to ensure that other natural disturbance types are not erroneously ignored to the detriment of benchmark objectives. Lastly, MDRs were estimated using historic disturbance data. Climate change is anticipated to influence natural disturbance behavior in the boreal (Price *et al.* 2013, Gauthier *et al.* 2014). If the severity, frequency, and/or size of natural disturbances increase, the MDR estimates may be too small, or if the inverse is true, larger than required with regards to maintaining vulnerable vegetation types. Climate change predictions for the boreal region point to the former (Price *et al.* 2013, Gauthier *et al.* 2015). Given the uncertainty imposed by climate change and other natural disturbance types, repeated assessment of the benchmark network is advised to ensure that the network continues to function as an effective control for adaptive management. If the landscape is managed within a proactive framework, such as the Conservation Matrix Model, additional area can be added to the benchmark network should it be too small.

We were not able to identify benchmarks in three ecoregions (12, 17, and 28), where MDRs could not be estimated due to either too few or no fires. When that is the case, other disturbance types should be explored (*e.g.*, insects). When natural disturbances do not play a significant role in shaping the landscape, an alternative method must be used to derive benchmark size, such as species requirements and/or the needs of a monitoring program. Two of these ecoregions have 100% protection. The third ecoregion also has a high-level of protection (88%), and the existing protected areas are likely suitable benchmarks for the ecoregion.

Ranking Methods and Criteria

While we employed a weighted-ranks method for ranking benchmark network options, the data are available such that alternative methods can be used (*e.g.*, ordinal or dense rank). Users also have the option to subset the ranking attributes (*i.e.*, benchmark properties, climate change, and focal species), so that the rankings best compliment regional conservation goals.

Benchmark networks can be further ranked using regional datasets, other biophysical features (*e.g.*, carbon storage), heritage/cultural sites (Leroux *et al.* 2007b), and socio-economic drivers, such as mining claims. Avoiding conflict with existing and potential development will minimize risk to the ecological integrity of the benchmark network, as well as risk to the socio-economic interests of communities and local governments. Emerging datasets from the NWBLCC (*e.g.*, mining; Geist *et al.* 2017) will be helpful in this regard. Modelling tools exist for assessing the impact of candidate protected area networks on socio-economic values, such as timber supply analysis; however, the simplest approach is to overlay candidate networks with spatial layers representing existing and potential socio-economic values (Figure 23).

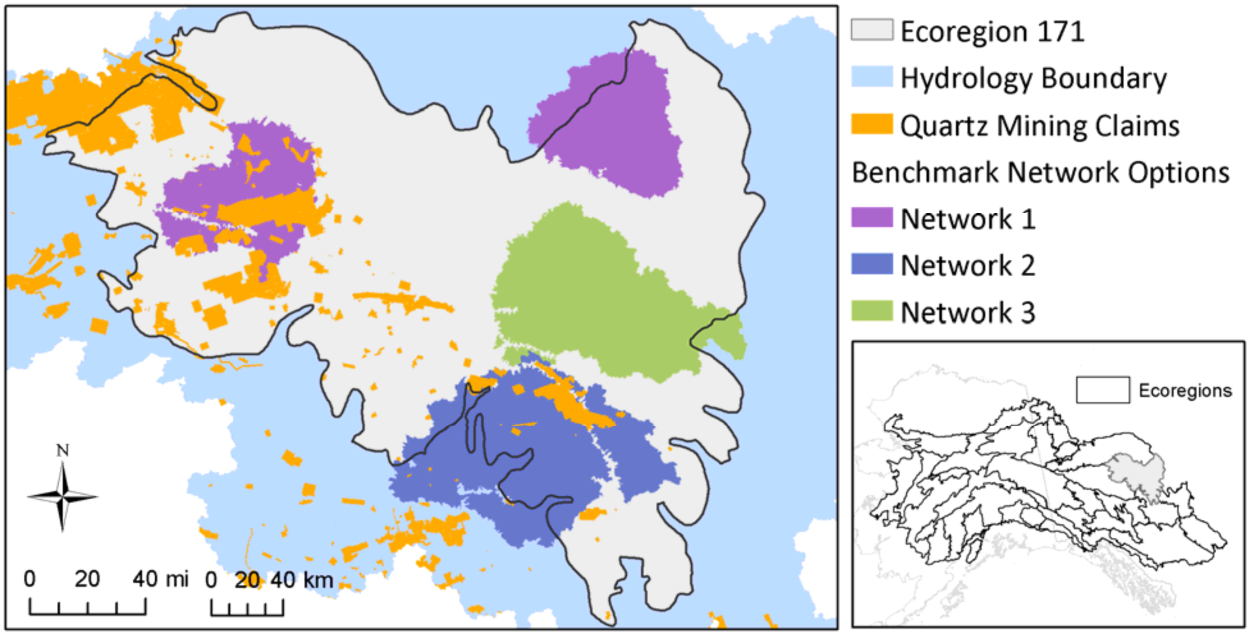

Figure 23. Benchmark networks can be ranked based on minimizing conflict with socio-economic interests, as such quartz mining (Yukon Government 2014). Based on spatial overlays, Network 3 has the least conflict (0 km^2), followed by Network 2 (650 km^2), and Network 1 (1,347 km^2).

Fundamental Benchmark Properties

Benchmarks are designed to be intact, integrate terrestrial and hydrologic connectivity, and capture headwaters. While these characteristics are addressed explicitly in the design process, the quality of a benchmark can be compromised by interruptions to the automated construction process of Benchmark Builder, either due to poor landscape condition or stoppage of construction once the size target has been met (*i.e.*, MDR), leaving the benchmark with incomplete headwaters. In the case of potential protected area (PAs) benchmarks, PAs are not guaranteed to have been designed with these attributes in mind, especially hydrology which has often been neglected in reserve design (Abell *et al.* 2007, Saunders *et al.* 2002). As a broad sweep analysis, a finer examination of solutions is warranted. It may be possible to improve benchmark characteristics through the refinement of benchmark boundaries either via the addition or exclusion of area. The latter is often possible for PA benchmarks because the size of many protected areas exceeds the MDR (Figure 24).

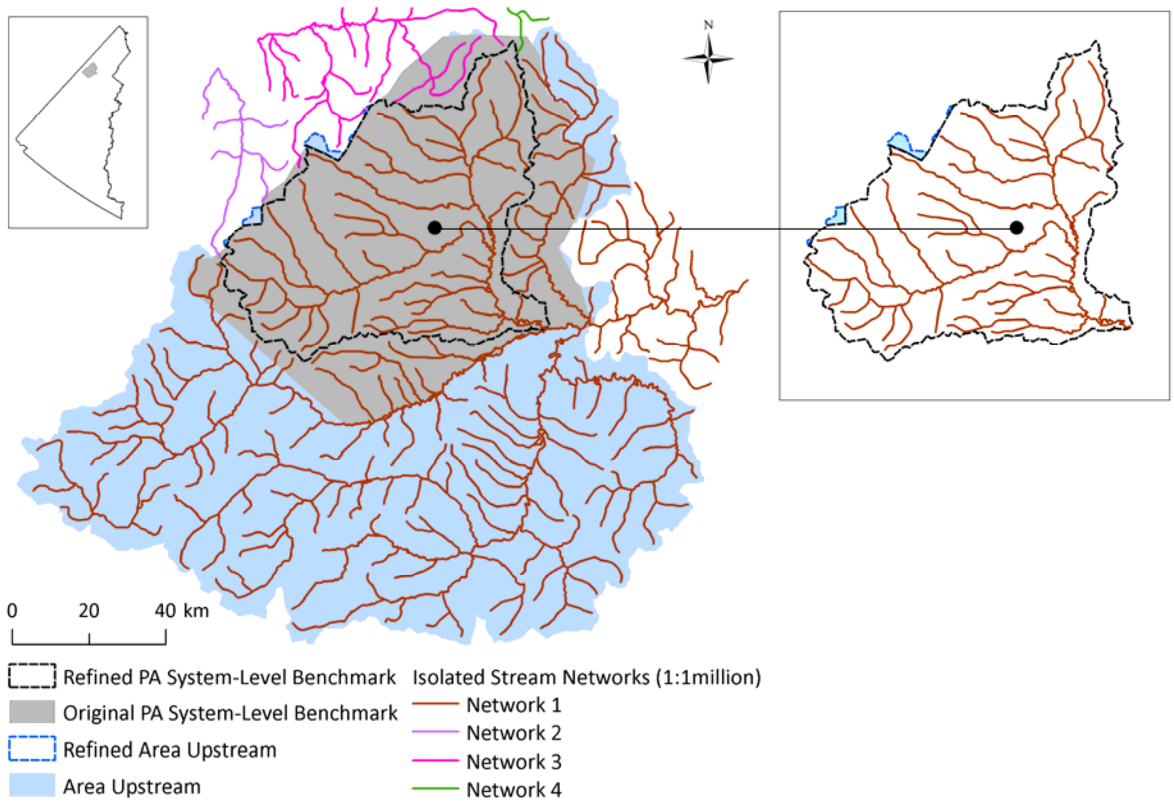

Figure 24. Nj 'iinlii" Jjik (Fishing Branch) Ecological Reserve, Habitat Protection Area, and Wilderness Preserve in northern Yukon is a protected area system-level benchmark that is 1.5 times the MDR (4,225 km^2) with low internal longitudinal hydrologic connectivity (lwDCI = 0.48). The area upstream (10,742 km^2) of the benchmark has a catchment area-weighted intactness of 90%. Refinements to the benchmark boundary can improve DCI to 1, which is a fully connected stream network, and reduce the vulnerability of the benchmark to upstream area (48 km^2), while still maintaining a size ≥ MDR.

When assessing the vulnerability of benchmarks to upstream disturbances, we restricted the evaluation of upstream area and upstream intactness to the ecoregion and associated hydrology units (*i.e.*, HUC8s and/or FDAs). This stratification does a good job of capturing most if not all headwaters associated with the ecoregion; however, there may be the odd case where headwaters extend beyond the hydrology units and vulnerability is underestimated. While disturbances downstream of benchmarks were not explored, they should also be considered given that dams, for example, can impede the migration of wide-ranging fish such as lake sturgeon and spawning salmon (Dungeon *et al.* 2006). It is best to capture all headwaters and downstream reaches associated with a benchmark to minimize the threat of human disturbances (Pringle 2001), but this is often not possible given the large spatial extent of stream networks or limitations imposed by landscape condition. In these situations, other tools such as special management zones can be used to carefully manage human activities in upstream and downstream areas to avoid influencing the ecological integrity of a benchmark and the behaviour of processes and species monitored.

To support the persistence of freshwater biodiversity, it is recommended that longitudinal, lateral, and vertical hydrologic connectivity be addressed in reserve design (Pringle 2001, Nel *et al.* 2011). We address the first two within benchmark design by using catchments assembled along stream networks. However, we did not address vertical connectivity, the movement of groundwater. Despite the ecological significance of groundwater (*e.g.*, Marmonier *et al.* 1993, Brunke and Gonser 1997, Ward 1998, Winter 2007), we were only able to address surface flow in the design of benchmarks because of the challenges of modelling and mapping groundwater flow patterns, although emerging methods for mapping groundwater flow are promising (*e.g.*, Devito *et al.* 2005, Yeh *et al.* 2016). Despite the lack of comprehensive groundwater datasets, mapping is available for some regions within the NWBLCC planning region (*e.g.*, hydrogeologic units for Matanuska-Susitna Valley in south-eastern Alaska by Kikuchi (2012) *unpublished* as cited in Callegary *et al.* 2013). Where available, local information on groundwater flow should be used to inform the management of areas associated with groundwater flows contributing to the hydrology of the benchmark network.

Climate Change – Permafrost and Other Datasets

Permafrost occurs throughout the NWBLCC planning region (Figure 25). Permafrost melt is occurring, and is expected to continue in response to climate warming, with associated changes in hydrology leading to drier landscapes, increased wildfire risk, and altered habitats (Chaplin *et al.* 2014, Streiker 2016). Benchmark networks can be further evaluated and ranked based on minimizing vulnerability to permafrost melt, as well as the ability to serve as controls for monitoring areas of permafrost melt. Geographic extent and quality of permafrost data is variable, from coarse-scale distribution zones available for the full extent of the NWBLCC planning region (*e.g.*, Figure 25), probability of permafrost mapping for the southern Yukon and northern BC (Bonnaventure *et al.* 2012), to more sophisticated modelling of projections for average annual ground temperature in Alaska (Jafarov *et al.* 2012, Rinke *et al.* 2012). In addition to permafrost, there are a growing number of climate datasets and climate-modelling tools that could be used to evaluate benchmark networks, including products from the following organizations: SNAP University of Alaska Fairbanks (*e.g.*, climate-biomes, Rowland *et al.* 2016), Adaptwest, NASA's ABoVE, National Snow and Ice Data Center, Permafrost Laboratory University of Alaska Fairbanks, and Northern Climate Exchange Yukon College.

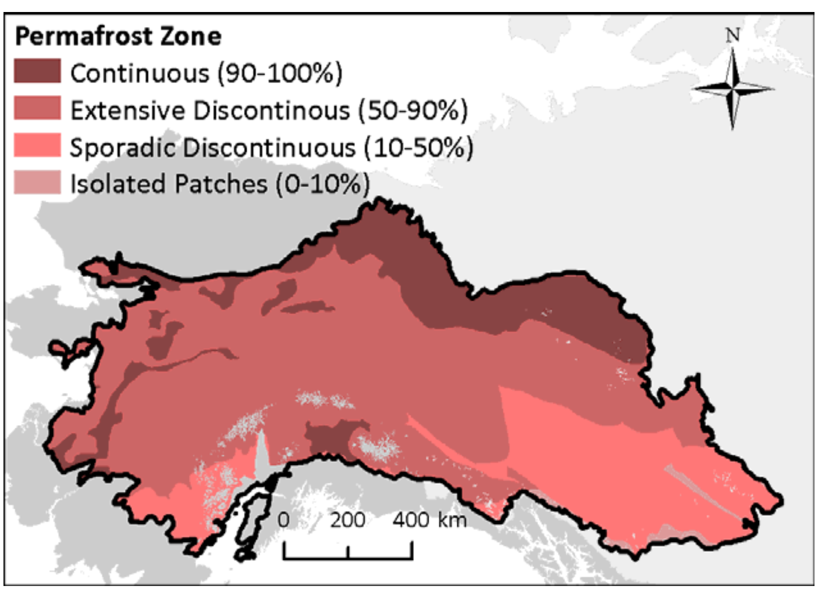

Figure 25. Permafrost cover in the Northwest Boreal LCC based on permafrost zones (Brown *et al.* 2002).

Focal Species

Benchmark networks were ranked based on the protection of habitat for a suite of priority focal species. Given the lack of conservation targets for species in management plans across the NWBLCC (see Focal Species Report; BEACONs 2017b), benchmark networks were ranked based on maximizing representation using spatial overlays. Ideally, the networks would have been assessed using evidence-based targets (Svancara *et al.* 2005). When undertaking regional Landscape Conservation Design, planners should strive to identify evidence-based targets for focal species. Examples of evidence-based targets include the amount of habitat required to (1) support a monitoring program based on a robust statistical sampling design and the habitat requirements of individuals, and (2) support a minimum viable population. While explicit numerical targets were not used in this analysis, the data are available such that users can apply representation targets and re-rank benchmark network options. Also, depending on the conservation goals of the planning exercise, not all focal species may be of interest. Users have the option to rank networks based on a subset of species and/or species datasets.

The ranking of benchmark networks was based solely on static spatial overlays. Dynamic landscape simulation modelling tools (*e.g.*, CONSERV; BEACONs 2015d) can be used to evaluate the representation of habitat through time given an active natural disturbance regime. This requires habitat models such as resource selection functions. This type of evaluation can be used to test the minimum amount of habitat maintained within a benchmark network over a specified period, and in turn, the ability of the network to continuously support a monitoring program or minimum viable population, for example, depending on the conservation objectives. Where reliable species habitat models exist, it is recommended that dynamic modelling be used to test that habitat targets are maintained within benchmark networks through time.

One of the greatest challenges to conservation planning is the availability of high-quality data. There are few species for which we have detailed maps of habitat use. Coarse distribution maps are often inferred from distribution models based on presence-absence surveys from outside the planning region. These surveys are susceptible to sampling bias including false absences assigned to areas that are difficult to access (*e.g*., Matsuoka *et al.* 2011, Reddy and Dávalos 2003). For the analysis, we used the best available data; however, we did not thoroughly assess spatial data with regards to its strengths and limitations. Prior to using the results, users should undertake such an assessment to avoid improper use (*e.g.*, Visconti *et al.* 2013), so that the risk of poor planning and management decisions is minimized.

CONCLUSION

As illustrated by this work, the largely intact NWBLCC planning region has high potential for the establishment of a comprehensive benchmark network, with contributions from the existing protected areas network. There are numerous network options to select from, and emerging datasets from the NWBLCC, and projects such as NASA's ABOVE, will further strengthen the application of this work and the design of a benchmark network for the north. Given the uncertainty of climate change and our limited knowledge regarding the response of biodiversity to human development, the establishment of a benchmark network for the NWBLCC region will allow for the implementation of active adaptive management, so that we may embrace uncertainty and learn-by-doing, to identify truly sustainable management practices that support the wide-range of environmental, cultural and economic values associated with the northwest boreal region.

Acknowledgements

We would like to thank the following individuals and organizations: Dr. Amanda Sesser, John DeLapp, Ben Matheson, Dr. Dawn Magness, and Dr. Amy Pocewicz, and the NWBLCC Steering Committee; Dr. Nancy Fresco, University of Alaska Fairbanks, and Dr. Jill Johnstone, University of Saskatchewan, for assistance with fire and vegetation modelling; Arnold Moy and Mélina Houle for technical support; Dr. Diana Stralberg University of Alberta for assistance with climate change and forest bird refugia; Tim Hammond, Bureau of Land Management, for feedback on earlier drafts; Megan Boldenow and Charleen Veach, U.S. Fish & Wildlife Service, for assistance with the review of management plans and acquisition of data for focal species; and Dr. Jamie Trammell, Marcus Geist, and others at the Alaska Center for Conservation Science, University of Alaska Anchorage, for technical assistance and data. Financial support was provided by Northwest Boreal Landscape Conservation Cooperative and Wilburforce Foundation. We would like to acknowledge other members of the BEACONs Project – Dr. Steve Cumming Laval University, Dr. Shawn Leroux Memorial University, and Dr. Meg Krawchuk Oregon State University.

References

Abell, R., J.D. Allan, and B. Lehner. 2007. Unlocking the potential for protected areas in conserving freshwaters. Biological Conservation 134: 48-63.

AdaptWest Project. 2015. Gridded climatic velocity data for North America at 1km resolution. Available at adaptwest.databasin.org.

Arcese, P. and A.R.E. Sinclair. 1997. The role of protected areas as ecological baselines. Journal of Wildlife Management 61(3):587-602.

Barber, Q., S. Nielsen, and A. Hamann. 2015. Assessing the vulnerability of rare plants using climate change velocity, habitat connectivity, and dispersal ability: a case study in Alberta, Canada. Regional Environmental Change:1-9.

BEACONs. 2015a. Northwest Boreal LCC Catchments Version 1.0. BEACONs Project, University of Alberta, Edmonton AB.

BEACONs. 2015b. Northwest Boreal LCC Harmonized FDA-HUC8 dataset. BEACONs Project, University of Alberta, Edmonton AB.

BEACONs. 2015c. Lake Edge Density (LED) for the NWBLCC Planning Region. BEACONs Project, University of Alberta, Edmonton, Alberta.

BEACONs. 2015d. CONSERV 2.0.22. BEACONs Project, University of Alberta, Edmonton, AB.

BEACONs. 2016a. Benchmark Builder v3.3.15. BEACONs Project, University of Alberta, Edmonton AB.

BEACONs. 2016b. Moose habitat. BEACONs Project, University of Alberta, Edmonton AB.

BEACONs. 2017a. Estimating Minimum Dynamic Reserves for Ecoregions in the Northwest Boreal LCC Planning Region. BEACONs Project, University of Alberta, Edmonton, AB. 35pp.

BEACONs. 2017b. Focal Species and the Design of Ecological Benchmarks to Support the Implementation of Landscape Conservation Design in the Northwest Boreal Landscape Conservation Cooperative. BEACONs Project, University of Alberta, Edmonton, AB. 28pp.

BCMFLNRO (BC Ministry of Forests, Land, and Natural Resources Operations). 2017. Mountain pine beetle projections. Government of British Columbia, Victoria, BC. http://www2.gov.bc.ca/gov/content/industry/forestry/managing-our-forest-resources/forest-health/forest-pests/bark-beetles/mountain-pine-beetle/mpb-projections

Bond, W.J., and J.E., Keeley. 2005. Fire as a global 'herbivore': the ecology and evolution of flammable ecosystems. Trends in Ecology & Evolution 20(7):387-394.

Bonnaventure, P.P., A.G. Lewkowicz, M. Kremer, M.C. Sawada. 2012. A Permafrost Probability Model for the Southern Yukon and Northern British Columbia, Canada. Permafrost and Periglacial Processes 23: 52-68.

Brown, J., O. Ferrians, J. A. Heginbottom, and E. Melnikov. 2002. Circum-Arctic Map of Permafrost and Ground-Ice Conditions, Version 2. Boulder, Colorado USA. NSIDC: National Snow and Ice Data Center. Date Accessed April 28, 2017. ftp://sidads.colorado.edu/pub/DATASETS/fgdc/ggd318_map_circumarctic/

Brunke, M. and T. Gonser. 1997. The ecological significance of exchange processes between rivers and groundwater. Freshwater Biology 37:1-33.

Burnham, K. P., and D. R. Anderson. 2002. Model Selection and Multimodel Inference: a practical information-theoretic approach, 2nd edition. Springer-Verlag, New York.

Callegary, J.B., C.P. Kikuchi, J.C. Koch, M.R. Lilly, and S.A. Leake. 2013. Review: Groundwater in Alaska (USA). Hydrogeology Journal 21:25-39.

Carroll, C., J. J. Lawler, D. R. Roberts, and A. Hamann. 2015. Biotic and Climatic Velocity Identify Contrasting Areas of Vulnerability to Climate Change. PLOS ONE 10:e0140486.

CCEA. 2014. CARTS (Conservation Areas Reporting and Tracking System). Downloaded July 2015. Canadian Council for Ecological Areas, http://www.ccea.org/carts.

CEC (Commission for Environmental Cooperation). 2013. North American Land Cover 2010. Produced by Natural Resources Canada/Canadian Center for Remote Sensing (NRCan/CCRS), United States Geological Survey (USGS); Insituto Nacional de Estadística y Geografía (INEGI), Comisión Nacional para el Conocimiento y Uso de la Biodiversidad (CONABIO) and Comisión Nacional Forestal (CONAFOR). http://www.cec.org/tools-and-resources/map-files/land-cover-2010

Chapin, F. S., III, S. F. Trainor, P. Cochran, H. Huntington, C. Markon, M. McCammon, A. D. McGuire, and M. Serreze, 2014: Ch. 22: Alaska. *Climate Change Impacts in the United States: The Third National Climate Assessment*, J. M. Melillo, Terese (T.C.) Richmond, and G. W. Yohe, Eds., U.S. Global Change Research Program, 514-536. doi:10.7930/J00Z7150. http://nca2014.globalchange.gov/report/regions/alaska#intro-section

Cote, D., G. Dan, C. Bourne, and Y.F. Wiersma. 2009. A new measure of longitudinal connectivity for stream Networks. Landscape Ecology 24:101-113.

Devito, K., I. Creed, T. Gan, C. Mendoza, R. Petrone, U. Silins, and B. Smerdon. 2005. A framewaork for broad-scale classification of hydrologic response units on Boreal Plains: is topography the last thing to consider? Hydrological Processes 19:1705-1714.

Dudgeon, D., A.H. Arthington, M.O. Gessner, Z.-I. Kawabata, D.J. Knowler, C. Lévêque, R.J. Naiman, A.-H. Prieur-Richard, D. Soto, M.L.J. Stiassny, and C.A. Sullivan. 2006. Freshwater biodiversity: importance, threats, status and conservation challenges. Biological Reviews 81:163–182.

Geist, M., M. Aisu, P. Lema and E.J. Trammell. 2017. Spatial estimates of surface mining footprints in northwest boreal ecoregions of Alaska and Canada. Dataset available from http://accs.uaa.alaska.edu/landscape-ecology.

Gomi, T., R.C. Sidle, and J.S. Richardson. 2002. Understanding processes and downstream linkages of headwater systems. Bioscience 52(10):905-916.

Groves, C.R. 2003. Drafting a Conservation Blueprint: A Practitioner's Guide to Planning for Biodiversity. Island Press: Washington DC.

Hamann, A., D.R. Roberts, Q.E. Barber, C. Carroll, and S.E. Nielsen. 2014. Velocity of climate change algorithms for guiding conservation and management. Global Change Biology 21:997–1004.

Hermoso, V., R. Abell, S. Linke, and P. Boon. 2016. The role of protected areas for freshwater biodiversity conservation: challenges and opportunities in a rapidly changing world. Aquatic Conservation 26(S1):3-11.

IPCC. 2014. Climate Change 2014: Synthesis Report. Contribution of Working Groups I, II and III to the Fifth Assessment Report of the Intergovernmental Panel on Climate Change [Core Writing Team, R.K. Pachauri and L.A. Meyer (eds.)]. IPCC, Geneva, Switzerland, 151 pp.

Jafarov, E.E., S. S. Marchenko, and V.E. Romanovsky. 2012. Numerical modeling of permafrost dynamics in Alaska using a high spatial resolution dataset. The Cryosphere 6:613-624.

Lee, P. and R. Cheng 2014. Human Access of Canada's Landscapes. Global Forest Watch Canada, Edmonton, AB. http://globalforestwatch.ca/node/198

Leroux, S.J., F.K.A. Schmiegelow, R.B. Lessard, and S.G. Cumming. 2007a. Minimum dynamic reserves: a framework for determining reserve size in ecosystems structured by large disturbances. Biological Conservation 138: 464-473.

Leroux, S.J., F.K.A. Schmiegelow, R.B. Lessard, and S.G. Cumming. 2007b. Potential spatial overlap of heritage sites and protected areas in a boreal region of northern Canada. Conservation Biology 21(2): 376-386.

Loarie, S.R., and P.B. Duffy, H. Hamilton, G.P. Asner, C.B. Field, and D.D. Ackerly. 2009. The velocity of climate change. Nature Letters 462:1052-1055.

Lowe, W.H., and G.E. Likens. 2005. Moving headwater streams to the head of the class. Bioscience 55(3):196-197.

McGarigal, K. and B.J. Marks. 1994. Fragstats. Spatial pattern analysis program for quantifying landscape structure. Version 2.0. Corvallis: Forest science department, Oregon State University.

Marmonier, P., P. Vervier, J. Gibert, and M.-J. Dole-Olivier. 1993. Biodiversity in ground waters. TREE 8(11):392-395.

Marshall, I.B., Schut, P.H., and Ballard, M. 1999. A National Ecological Framework for Canada: Attribute Data. Agriculture and Agri-Food Canada, Research Branch, Centre for Land and Biological Resources Research, and Environment Canada, State of the Environment Directorate, Ecozone Analysis Branch, Ottawa/Hull. Online: http://sis.agr.gc.ca/cansis/nsdb/ecostrat/1999report/index.html

Matsuoka, S., P. Sólymos, T. Fontaine, and E. Bayne. 2011. Roadside surveys of boreal forest birds: how representative are they and how can we improve current sampling? A Report to Environment Canada. Boreal Avian Modelling Project

Nel, J.L., B. Reyers, D.J. Roux, N.D. Impson, and R.M. Cowling. 2011. Designing a conservation area network that supports the representation and persistence of freshwater biodiversity. Freshwater Biology 56:106-124.

Nowacki, G., P. Spencer, M. Fleming, T. Brock, and T. Jorgenson. 2001. Ecoregions of Alaska. U.S. Geological Survey Open-File Report 02-297. https://agdc.usgs.gov/data/usgs/erosafo/ecoreg

NRCAN (Natural Resources Canada). 2003. Atlas of Canada 1,000,000 National Frameworks Data, Hydrology - Fundamental Drainage Areas. http://geogratis.cgdi.gc.ca

NRCAN (Natural Resources Canada). 2001-2012. Canadian Digital Elevation Data. http://geogratis.cgdi.gc.ca

NRCAN (Natural Resources Canada). 2011-2014. National Hydro Network, Version 14. http://geogratis.cgdi.gc.ca

Price, D., R.I Alfaro, K.J. Brown, M.D. Flannigan, R.A. Fleming, E.H. Hogg *et al.* 2013. Anticipating the consequences of climate change for Canada's boreal forest ecosystems. Environmental Reviews 21:322-365.

Pringle, C.M. 2001. Hydrologic connectivity and the management of biological reserves: a global perspective. Ecological Applications 11(4):981–998.

Reddy, S., and L. M. Dávalos. 2003. Geographical sampling bias and its implications for conservation priorities in Africa. Journal of Biogeography 30:1719–1727.

Rinke, A., H. Matthes, J.H. Christensen, P. Kuhry, V.E. Romanovsky, and K. Dethloff. 2012. Arctic RCM simulations of temperature and precipitation derived indices relevant to future frozen ground conditions. Global and Planetary Change 80-81:136-148.

Rowland, E.L., N. Fresco, D. Reid, and H.A. Cooke. 2016. Examining climate-biome ("cliome") shifts for Yukon and its protected areas. Global Ecology and Conservation 8:1-17.

Schmiegelow, F.K.A., S.G. Cumming, K.A. Lisgo, S.J. Leroux, and M.A. Krawchuk. 2014. Chapter 5 - Catalyzing Large Landscape Conservation in Canada's Boreal Systems: The BEACONs Project Experience.

Pages 97-122 in Conservation Catalysts: The Academy as Nature's Agent *edited by* J.N. Lewitt, Lincoln Institute of Land Policy, Cambridge Massachusetts.

Serra-Diaz, J. M., J. Franklin, M. Ninyerola, F. W. Davis, A. D. Syphard, H. M. Regan, and M. Ikegami. 2014. Bioclimatic velocity: the pace of species exposure to climate change. Diversity and Distributions 20:169-180.

Siipi, H. 2004. Naturalness in biological conservation. Journal of Agricultural and Environmental Ethics 17:457-477.

Sinclair, A.R.E. 1998. Natural regulation of ecosystems in protected areas as ecological baselines. Wildlife Society Bulletin 26:399-409.

Sinclair, A.R.E., S.A.R. Mduma, and P. Arcese. 2002. Protected areas as biodiversity benchmarks for human impact: agriculture and the Serengeti avifauna. Proc. R. Soc. Lond. B 269:2401-2405

Stralberg, D., C. Carroll, S. Nielsen, C. Wilsey, J. Pedlar, and D. McKenney. in review. A bird in the hand: Locations and drivers of multi-species macrorefugia for North American trees and songbirds.

Stralberg, D., S. M. Matsuoka, A. Hamann, E. M. Bayne, P. Sólymos, F. K. A. Schmiegelow, X. Wang, S. G. Cumming, and S. J. Song. 2015. Projecting boreal bird responses to climate change: the signal exceeds the noise. Ecological Applications 25:52–69.

Streicker, J. 2016. Yukon Climate Change Indicators and Key Findings 2015. Northern Climate ExChange, Yukon Research Centre, Yukon College, 84 p. https://www.yukoncollege.yk.ca/downloads/Indicator_Report_Final_web.pdf

Svancara, L.K., R. Brannon, J.M. Scott, C.R. Groves, R.F. Noss, and R.L. Pressey. 2005. Policy-driven versus evidence-based conservation: a review of political targets and biological needs. Bioscience 55(11):989-995.

Trammell, E.J. and M. Aisu. 2014. DRAFT Landscape Condition of Alaska. Developed for Western Governors Association (WGA) Critical Habitat Assessment Tool (CHAT). Alaska Natural Heritage Program, University of Alaska, Anchorage, AK.

U.S. Fish & Wildlife Service. 2013. Final Report: A landscape-scale approach to refuge system planning. https://www.fws.gov/refuges/vision/pdfs/PlanningGuideRev10.pdf

USGS (U.S. Geological Survey). 2014a. National Hydrography Dataset, Alaska. ftp://nhdftp.usgs.gov/DataSets/Staged/States/FileGDB/HighResolution/NHDH_AK_931v210.zip

USGS (U.S. Geological Survey). 2014b. National Elevation Dataset, Alaska. http://nationalmap.gov/elevation.html

USGS (U.S. Geological Survey). 2012. Protected Areas Database of the United States (PAD-US), version 1.3. https://gapanalysis.usgs.gov/padus/data/

Visconti, P. M. Di Marco, J.G. Álvarez-Romero, R.S Januchowski-Hartley, R.L. Pressey, R. Weeks, and C. Rondinini. 2013. Effects of errors and gaps in spatial data sets on assessment of conservation progress. Conservation Biology 27(5):1000-1010.

Ward, J.V. 1998. Riverine landscapes: biodiversity patterns, disturbance regimes, and aquatic conservation. Biological Conservation 83(3): 269-278.

Wang, T., A. Hamann, D.L. Spittlehouse, and C. Carroll. 2016. Locally downscaled and spatially customizable climate data for historical and future periods for North America. PLoS One 11: e0156720.

Weber, M.G. and B.J. Stocks. 1998. Forest fires and sustainability in the boreal forests of Canada. Ambio 27:545-550.

Wipfli, M.S., J.S. Richardson, and R.J. Naiman. 2007. Ecological linkages between headwaters and downstream ecosystems: transport of organic matter, invertebrates, and wood down headwater channels. Journal of the American Water Resources Association 43(1):72-85.

Winter, T.C. 2007. The role of ground water in generating streamflow in headwater areas and in maintaining base flow. Journal of the American Water Resources Association 43(1):15-25.

Yeh, H-F, Y-S Cheng, H-I Lin, and C-H Lee. 2016. Mapping groundwater recharge potential zone using a GIS approach in Hualian River, Taiwan. Sustainable Environment Research 26:33043.

Yukon Government. 2014. Quartz Claims - 1M. Yukon Energy, Mines and Resources, Whitehorse, Yukon.

Zhao, M. and S.W. Running. 2010. Drought-induced reduction in global terrestrial net primary production from 2000 through 2009. Science 329(5994):940-943.