

Template to prepare preprints and manuscripts using markdown and github actions

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Protected areas (PA) are an effective means of conserving biodiversity and protecting suites of valuable ecosystem services. Currently, many nations and international governments use proportional area protected as a critical metric for assessing progress towards biodiversity conservation. However, this and other common metrics do not assess the effectiveness of PA networks, nor do they assess how representative PA are of the ecosystems they aim to protect. Topography, stand structure, and land cover are all key drivers of biodiversity within forest environments, and are well suited as indicators to assess the representation of PA. Here we examine the protected area network in British Columbia, Canada, through these drivers derived from freely available data and remote sensing products across the provincial biogeoclimatic ecosystem classification system. We examine biases in the protected area network by elevation, forest disturbances, and forest structural attributes, including height, cover, and biomass by comparing a random sample of protected and unprotected pixels. Results indicate that PA are commonly biased towards high elevation and alpine land covers, and that forest structural attributes of the park network are often significantly different in protected vs unprotected areas (426 out of 496 forest structural attributes found to be different; $p < 0.01$). Analysis of forest structural attributes suggests that establishing additional PA could ensure representation of various forest structure regimes across British Columbia's ecosystems. We conclude that these approaches using free and open remote sensing data are highly transferable, and can be accomplished using consistent datasets to assess PA representations globally.

1 Introduction

2 Protected areas (hereafter PA) are an integral component of biological conservation, designed to preserve
3 ecosystem services and biodiversity both inside the PA and in some cases the surrounding regions (Chape
4 et al. 2005, Watson et al. 2014). In recent decades, there has been a growing consensus of the need to
5 conserve varying portions of the terrestrial area of the globe, with areal goals increasing over time (CBD
6 2004, 2010). In the 2010s, the Aichi biodiversity target sought to protect 17% of the entire globe (CBD
7 2010). Nationwide, the Canadian government has set the goal of protecting 25% of Canada's terrestrial
8 area by 2025 (ECCC 2021). While increasing proportional ecosystem protection does in turn aid
9 conservation, it does not guarantee the representativeness of the entire ecosystem, nor that all biodiversity
10 within the PA will be effectively conserved (Hazen and Anthamatten 2004).

11 Many conservation goals, both global and regional, are commonly based on the proportion of area
12 protected, at least partly due to its ease of use and calculation (Brooks et al. 2004, CBD 2010). However,
13 while the area protected is a simple metric to report, other metrics can be more informative, with the
14 potential to convey how effective a given PA is for protecting the inherent ecosystem services or
15 biodiversity in the area (Chape et al. 2005, Butchart et al. 2015, Maxwell et al. 2020). Beyond areal extent,
16 it is also relevant to consider the biases in PA placement, which are frequently located in fiscally cheaper,
17 low productivity regions both globally (Joppa and Pfaff 2009, Venter et al. 2014, 2018) and regionally, as is
18 the case in British Columbia (BC), Canada (Hamann et al. 2005, Environmental Reporting BC 2016, Wang
19 et al. 2020). The area metric heavily underestimates the global protected area required to adequately
20 protect biodiversity, which research indicates is up to 50% of each ecoregion (Dinerstein et al. 2017,
21 Dinerstein et al. 2019).

22 In response to the proportion protected approach, a number of other methodologies have been developed
23 to evaluate the effectiveness of PAs before these larger global targets have been met (Parrish et al. 2003,
24 Gaston et al. 2006, 2008, Hansen and Phillips 2018, Bolton et al. 2019). One recently identified concept in
25 Canadian park management is ecological integrity. Ecological integrity is defined as an ecosystem having
26 the expected “living and non-living pieces for the region,” and where ecological processes occur at the
27 expected frequency and intensity for the region (Parks Canada 2019). Many potential ecological integrity
28 indicators have been examined to capture biodiversity related processes within PA (Hansen and Phillips
29 2018, Hansen et al. 2021). These indicators can then be interpreted manually or automatically, most often

30 through examining temporal trends within the PA or by comparing the indicators to areas in known
31 healthy reference ecosystems (Woodley 1993).

32 Frequently, comparisons between PA and unprotected areas (UA) have been drawn in order to assess PA
33 performance and health (Defries et al. 2005). This allows for the PA or PA network to be taken in context
34 of surrounding and/or similar ecosystems (Wiens et al. 2009). There are, however, challenges associated
35 with comparing the effectiveness of a PA network directly with UA. It can be difficult to identify suitable
36 UA for comparison due to the increased prevalence of human pressure in UA (Geldmann et al. 2019), and
37 the bias for PA to be in areas that would not have faced increased human pressure due to their remoteness
38 (Joppa and Pfaff 2009).

39 Ferraro (2009) prescribes the use of the counterfactual - comparing what has happened post-PA
40 implementation with what would have happened if the PA were not implemented. The counterfactual
41 method has recently been adopted in the literature as a more accurate method for assessing protected area
42 management effectiveness (Ribas et al. 2020). This method is frequently employed using matching
43 methods to select UA which directly correspond to the PA being analyzed, often matching based on
44 topography, climate, land cover, and other variables (Coad et al. 2015, Eklund et al. 2019, Geldmann et al.
45 2019). Collecting field data across both the PA and its counterfactual UA can often be time-and-cost
46 prohibitive. The increasing prevalence of freely available imagery has led to satellite remote sensing
47 becoming an essential tool for PA monitoring (Nagendra et al. 2013).

48 The opening of the Landsat archive in 2008 (Wulder et al. 2012a) has played a significant role in the use of
49 satellite imagery in conservation monitoring (Nagendra 2008, Turner et al. 2015). The availability of 30-m
50 spatial resolution data since 1984 allows for assessment of temporal trends in satellite derived indicators
51 (Nagendra et al. 2013, Hansen and Phillips 2018, Bolton et al. 2019), while the global coverage allows for
52 comparisons between similar and differing ecosystems (Nagendra 2008, Wulder et al. 2012a). Leveraging
53 free and open-source optical remote sensing data products has allowed users to increasingly undertake
54 comparisons across an entire jurisdiction's PA network (Fraser et al. 2009, Soverel et al. 2010, Pôças et al.
55 2011, Bolton et al. 2019, Skidmore et al. 2021), comparing them to ecologically similar UA (Turner et al.
56 2015, Buchanan et al. 2018). These comparisons allow for an assessment of the effectiveness of a given PA
57 or the entire PA network at representing regional biodiversity trends (Soverel et al. 2010, Turner et al.
58 2015, Bolton et al. 2019).

59 Optical remote sensing technologies have offered a key approach to deriving indicators (Nagendra 2008,

60 Fraser et al. 2009, Soverel et al. 2010, Burkhard et al. 2012, Pereira et al. 2013, Bolton et al. 2019) and
61 detecting key terrestrial processes (Turner et al. 2003) to assess PA effectiveness at conserving ecological
62 integrity (Nagendra 2001, Nagendra et al. 2013). These indicators derived from remote sensing
63 technologies can be categorized and monitored at broad spatial extents and across temporal scales.
64 Commonly used indicators include land cover proportion (e.g., forest type, wetland, and unvegetated,
65 Parmenter et al. 2003, Olthof et al. 2006), tree species (Nagendra 2001), habitat classification (McDermid
66 et al. 2005, Lucas et al. 2011), spectral information (Feeley et al. 2005, Gillespie 2005, Nagendra et al.
67 2010), spectral heterogeneity (Rocchini et al. 2010), and ecosystem structure (Cohen and Goward 2004,
68 Goetz et al. 2007, Soverel et al. 2010, Pôças et al. 2011) and function (Skidmore et al. 2021). Moreover,
69 remote sensing technologies enable the monitoring of terrestrial processes, such as natural and
70 anthropogenic disturbance regimes (Kerr and Ostrovsky 2003, Alsdorf et al. 2007, Hermosilla et al. 2015b,
71 Bolton et al. 2019), alongside biogeochemical cycles (Myneni et al. 2001), vegetation productivity
72 (Running et al. 2004), and vegetation dynamics (Zhang et al. 2003). Diversity in forest structural attribute
73 measurements, often derived from light detection and ranging (lidar) is also a strong indicator of
74 biodiversity, providing habitat, influencing food quality, and mediating microclimates (Gao et al. 2014,
75 Guo et al. 2017).

76 Lidar enables the accurate characterization of treed vegetation structure (e.g. canopy height, canopy cover,
77 basal area) across forested areas by measuring the time it takes for an emitted pulse of light to return to the
78 sensor (Lim et al. 2003). While the natural variation in vertical and horizontal forest structure has been
79 extensively explored using lidar, comparisons between PA and UA have been less frequently drawn using
80 these methods when compared to optical remote sensing (Nagendra et al. 2013). The lack of previous
81 comparisons has likely been due to the frequently limited extents of lidar acquisitions, a problem which
82 has recently been solved by generating wall-to-wall metrics. These wall-to-wall metrics can be created by
83 combining lidar data with times series of Landsat data, generating forest structural attributes across large
84 regions and even entire countries (Wulder et al. 2012b, Matasci et al. 2018a).

85 As Canada progresses towards the national goal of 25% of terrestrial area protected by 2025, there is a
86 growing need to better understand how PA compare to UA with respect to location, ecological
87 classifications, elevations, productivity, and forest structure. In this study, we (1) examine the hypothesis
88 that BC's PA network is biased towards high-elevation, low-productivity regions of the province using free
89 and open remote sensing data products, and (2) identify underrepresented forest structures in PA in the

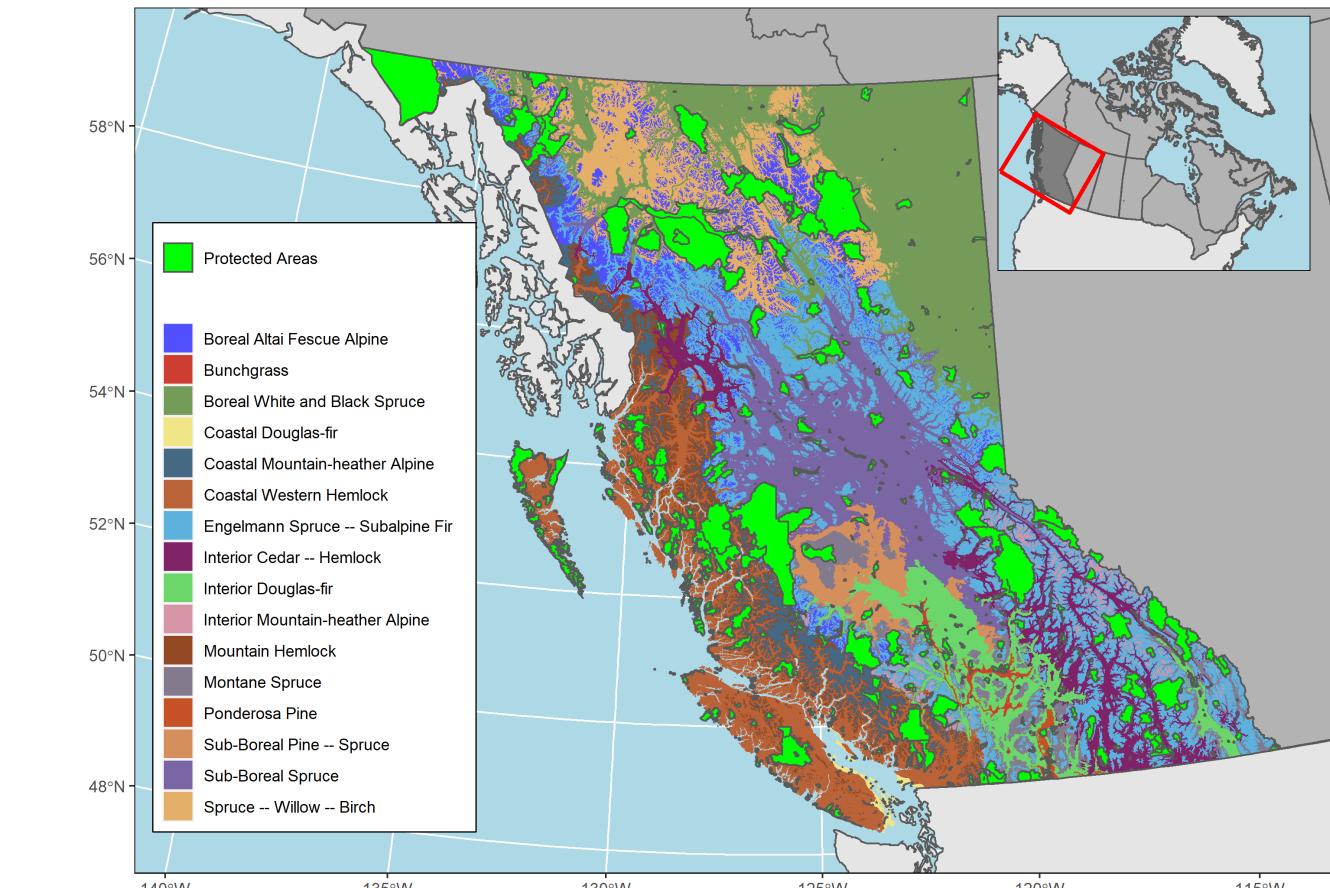
90 province. To accomplish this, we examined the bias in PA placement by comparing ecoregional PA
91 coverage and land cover classes by elevation, and disturbances by latitude across protected and UA in BC.
92 We examine representative forest structural attributes by comparing the distribution of key indicators by
93 ecological zone to determine the differences between PA and UA to find the most and least similar
94 represented forest structures throughout the network. We conclude by highlighting the usefulness of these
95 globally available, high quality, consistent, and transferable datasets and methods for assessing PA
96 effectiveness.

97 **Methods**

98 **Study Area**

99 The province of British Columbia, Canada, covers 94.4 million ha, of which approximately 64% is forested
100 (BC Ministry of Forests 2003), and encapsulates a wide variety of biomes and ecosystems. This diversity of
101 ecosystems is in part due to the large area as well as variations in topography and climate. The existing
102 Biogeoclimatic Ecosystem Classification (BEC) system disaggregates BC ecosystems into zones (Pojar et al.
103 1987). The broadest classification delineates 16 zones, which are further broken down into subzones,
104 variants, and phases based on microclimate, precipitation, and topography (Pojar et al. 1987, Meidinger
105 and Pojar 1991). As a result, BEC zones vary widely in size (ranging from 0.25 million ha to 17.5 million
106 ha), and in number of subzones (from 1 to 43; see Table 1).

107 Both the BC (BC Parks 2012) and Canada-wide (Government of Canada, 2019) PA mandates commit to
108 conserving ecological integrity across the network. The PA network in BC is designed to serve both
109 ecological conservation and human recreation aims (BC Parks 2012) and consists of a network of PA and
110 PA complexes (multiple nearby PA which share the same conservation goals), with large variations in size,
111 ranging from 0.02 to 987,899 ha (fig. ??).



112 {fig:study-
 113 area}
 114 Data
 115 Biogeoclimatic Ecosystem Classification and Protected Areas

116 Boundaries for BEC zones and subzones were acquired using the **bcmaps** R package (Teucher et al. 2021).
 117 Two BEC subzones were entirely subsumed by PA (Boreal White and Black Spruce - Very Wet Cool and
 118 Spruce – Willow – Birch - Very Wet Cool Shrub), whereas the Sub-Boreal Pine – Spruce - Moist Cool
 119 subzone has no PA representation.
 120 Boundaries for all PA in BC were obtained from the Canadian Protected and Conserved Areas Database
 121 (available from https://cws-scf.ca/CPCAD-BDCAPC_Dec2020.gdb.zip), current as of December 2020,
 122 and includes the International Union for Conservation of Nature (IUCN) classification for each PA. PA
 123 were selected for analysis following the criteria outlined in Bolton et al. (2019). Only parks which
 124 belonged to IUCN classes Ia, Ib, II, and IV were selected, as these categories are considered strictly

125 protected. Protected areas < 100 ha in size were also excluded from the analysis, as these mainly occurred
126 in urbanized areas. After selection, 745 suitable parks managed under various jurisdictions (provincial,
127 federal, NGOs), comprising 15.4% of the total terrestrial area of British Columbia, were studied
128 (Environmental Reporting BC 2016). An equal sample of pixels equal to the area of PA or UA - whichever
129 was lower - was randomly selected from both PA and UA for each BEC subzone. This sampling regime
130 accounts for bias in topography, climate, and climax species due to the methods used to delineate BEC
131 zones and subzones (Pojar et al. 1987).

132 **Digital Elevation Model**

133 The Advanced Spaceborne Thermal Emission and Reflection Radiometer (ASTER) digital elevation model
134 (GDEM V2, 30 m) was used to examine biases in protected area land cover and ecological classification by
135 elevation (Tachikawa et al. 2011).

136 **Landsat derived datasets**

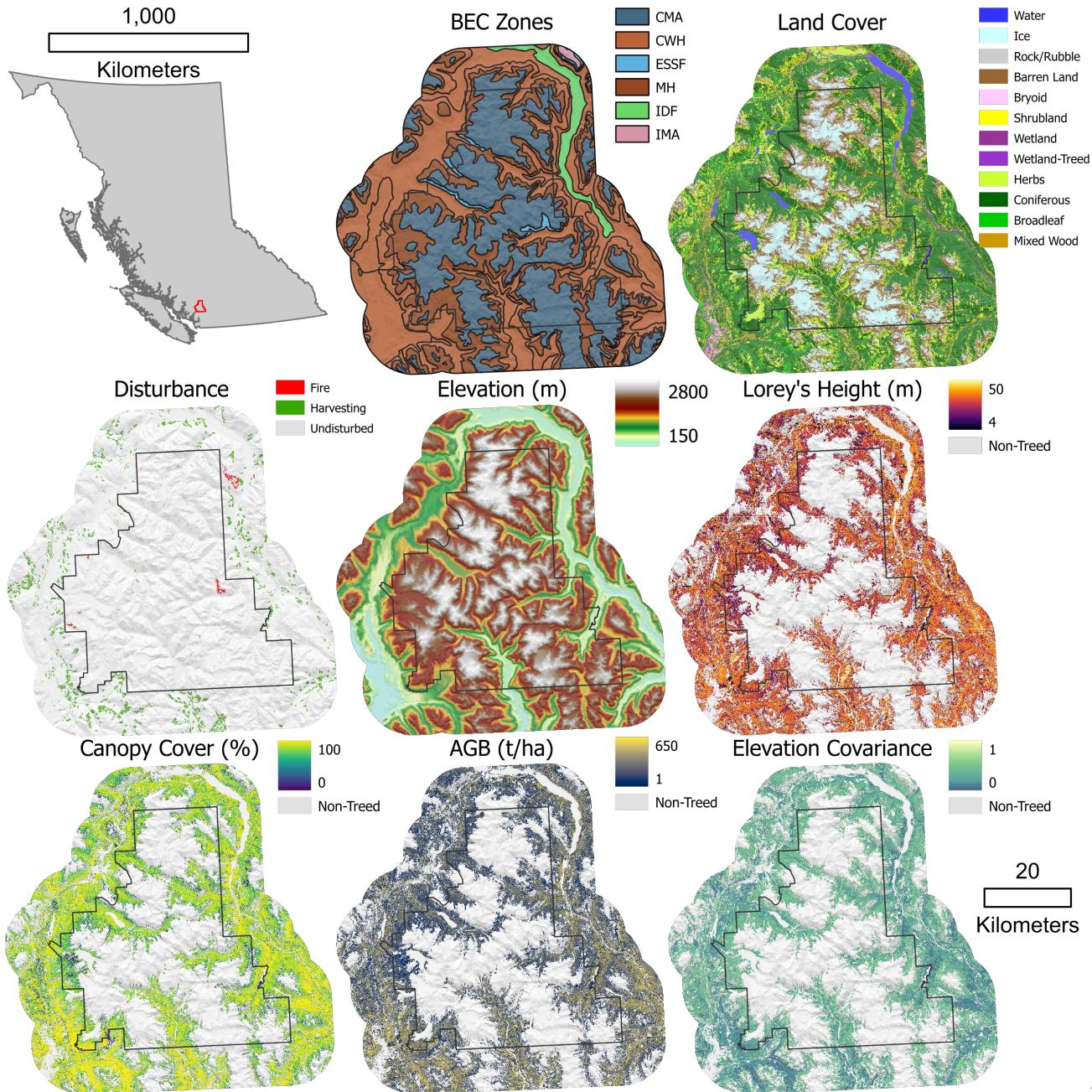
137 Land cover, forest disturbances, and forest structural attributes for BC were derived from the annual
138 Landsat best-available-pixel (BAP) composites from 1984 to 2019 at 30-m spatial resolution generated
139 using the Composite2Change (C2C) approach (Hermosilla et al. 2016). These composites are generated by
140 annually selecting the optimal observations, free from atmospheric effects (haze, clouds, cloud shadows),
141 for each pixel from the catalog of available Landsat-5 Thematic Mapper (TM), Landsat-7 Enhanced
142 Thematic Mapper Plus (ETM+), and Landsat-8 Operational Land Imager (OLI) imagery acquired during
143 Canada's growing season using the scoring functions defined in White et al. (2014). The annual BAP
144 composites are further refined by applying a spectral trend analysis over the Normalized Burn Ratio
145 (NBR) at pixel level in order to remove unscreened noise, detect changes and fill data gaps with
146 temporally-interpolated values, resulting in annual, gap-free, surface-reflectance image composites
147 (Hermosilla et al. 2015b). During this process, forest disturbances are detected, characterized and
148 attributed to a disturbance agent (i.e., wildfire, harvest, non-stand replacing disturbances) using a
149 Random Forests classification model via the object-based analysis approach (Hermosilla et al. 2015a) with
150 an overall accuracy of 92% \pm 2% (Hermosilla et al. 2016).

151 Annual land cover information for Canada was produced using the BAP composites following the Virtual

152 Land Cover Engine framework (Hermosilla et al. 2018). This framework integrates post-classification
153 probabilities, forest disturbance information and forest successional knowledge with a Hidden Markov
154 Model to ensure logical land cover transitions between years. The classification comprises 12 land cover
155 classes organized as either non-vegetated or vegetated. Non-vegetated classes included water, snow/ice,
156 rock/rubble, and exposed/barren land. Vegetated land cover classes discriminated among non-treed and
157 treed vegetation (land-cover level). Vegetated non-treed classes comprised bryoids, herbs, wetland, and
158 shrubs. Vegetated treed land cover classes included wetland-treed, coniferous, broadleaf, and mixed wood.
159 Independent validation of the land cover maps indicated an overall accuracy of $70.3\% \pm 2.5\%$.

160 Wall-to-wall, 30-m forest structure metrics (i.e., Lorey's height, total aboveground biomass, elevation
161 covariance, and canopy cover) were also annually derived from the BAP composites using the imputation
162 method described in Matasci et al. (2018b, 2018a). This method uses lidar and field plot data to estimate
163 forest structure metrics from topographic and Landsat spectral predictors, using a k-Nearest Neighbor
164 approach. Reported accuracy for the structure metrics indicated a RMSE% ranging from 24.5% to 65.8%
165 and a R^2 ranging from 0.125 to 0.699 (Matasci et al. 2018a).

166 Forest cover classes (deciduous, broadleaf, mixed-wood, and wetland-treed) were used to generate land
167 cover masks to restrict the comparison of forest structural attributes to treed pixels. Pixels with harvest
168 activity disturbances detected post-1985 were also removed from forest structural attribute rasters in both
169 PA and UA, in order to restrict analysis to non-anthropogenically disturbed areas. All datasets are
170 displayed in fig. ??.



172 Analysis

173 To determine bias in ecosystem representation in BC's PA network, we compared BEC zone, land cover,
 174 and disturbance proportions within and outside the PA network. We employ counterfactual thinking by
 175 examining BEC zone, and land cover as a function of elevation, and secondly compiled disturbance rates
 176 on a latitudinal gradient across the province. Forest structural attributes were then examined at a finer
 177 ecosystem classification level, statistically comparing PA vs UA across similar ecosystems. Forest

178 structural means across BC zones were calculated to determine which forest structures need additional
179 representation in the current BC PA network. All data manipulation and analysis was conducted in the **R**
180 (R Core Team 2021) or **Python** programming languages.

181 **Ecosystems, Land Cover, and Disturbances**

182 BEC zones and land cover classifications were aggregated to both PA and UA in order to determine the
183 proportion of each zone under the protected classifications, to examine progress towards the Aichi
184 biodiversity targets. In this analysis, zones were used to examine categorical data (land cover and
185 disturbance) for the period of 1984-2019. Land cover and BEC zones were further examined along an
186 elevation gradient, at 50m increments. Histograms of area by elevation were generated in order to
187 examine the areal magnitude alongside the proportional coverage of land cover and BEC zones. This
188 allows us to examine the amount of area protected at each elevation, as well as the differences between PA
189 and UA. Forest disturbances (including harvesting) were aggregated along a latitudinal gradient at
190 increments of 0.5°.

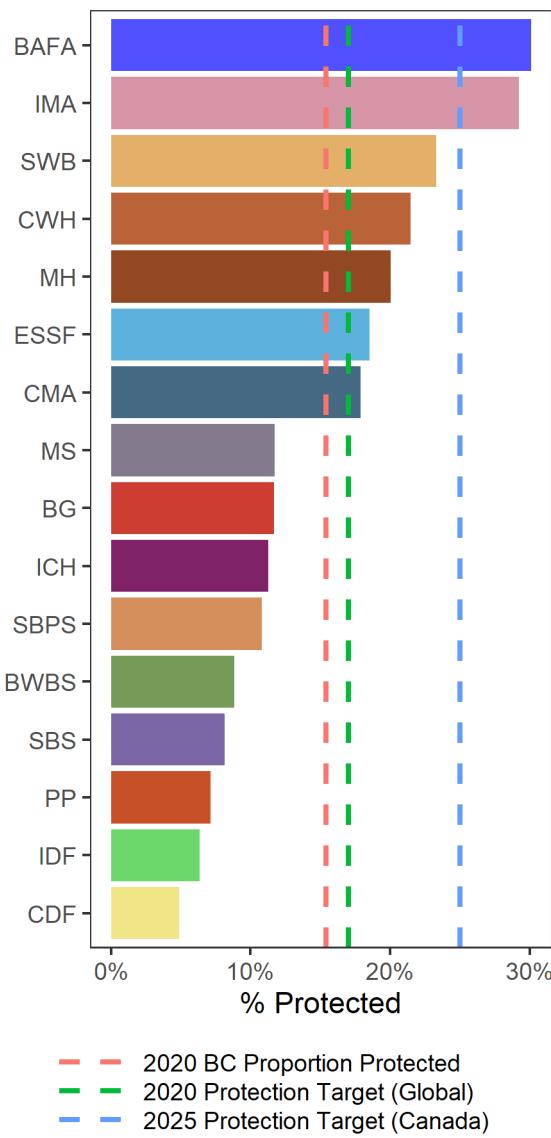
191 **Forest Structural Attributes**

192 T-tests for PA vs UA were conducted on all pixels selected for analysis by BEC subzone and forest
193 structural attribute for 2015, and the Bonferroni correction was applied. The Bonferroni correction avoids
194 spuriously significant results in multiple comparison tests by dividing the significant p-value (0.01) by the
195 number of tests (Bonferroni 1936). Within each BEC zone, higher proportions of significant tests will
196 indicate dissimilar subzones in each forest structural attribute. The mean values for PA and UA forest
197 structural attributes were calculated, in order to examine the differences in their distribution and
198 determine which structures and zones differ between PA and UA. Values were also converted into z-scores
199 to determine the greatest standardized vector magnitude when comparing canopy cover, elevation
200 covariance, and forest height between PA and UA.

201 **Results**

202 **Ecosystems, Land Cover, and Disturbances**

203 British Columbia's ecosystems are protected at varying rates across the province (fig. ??). Of the 16
204 ecosystems present in BC, seven are protected at rates above the Aichi biodiversity target (17%). Only two
205 zones (Boreal Altai Fescue Alpine and Interior Mountain-heather Alpine) are currently protected at rates
206 above the Canadian 2025 protection targets (25%). Zones with Douglas-fir (*Pseudotsuga menziesii*) as
207 dominant old-growth components (Coastal Douglas-fir and Interior Douglas-fir) are the least
208 proportionally represented zones in British Columbia, with 4.9% and 6.4% protected, respectively (fig. ??).

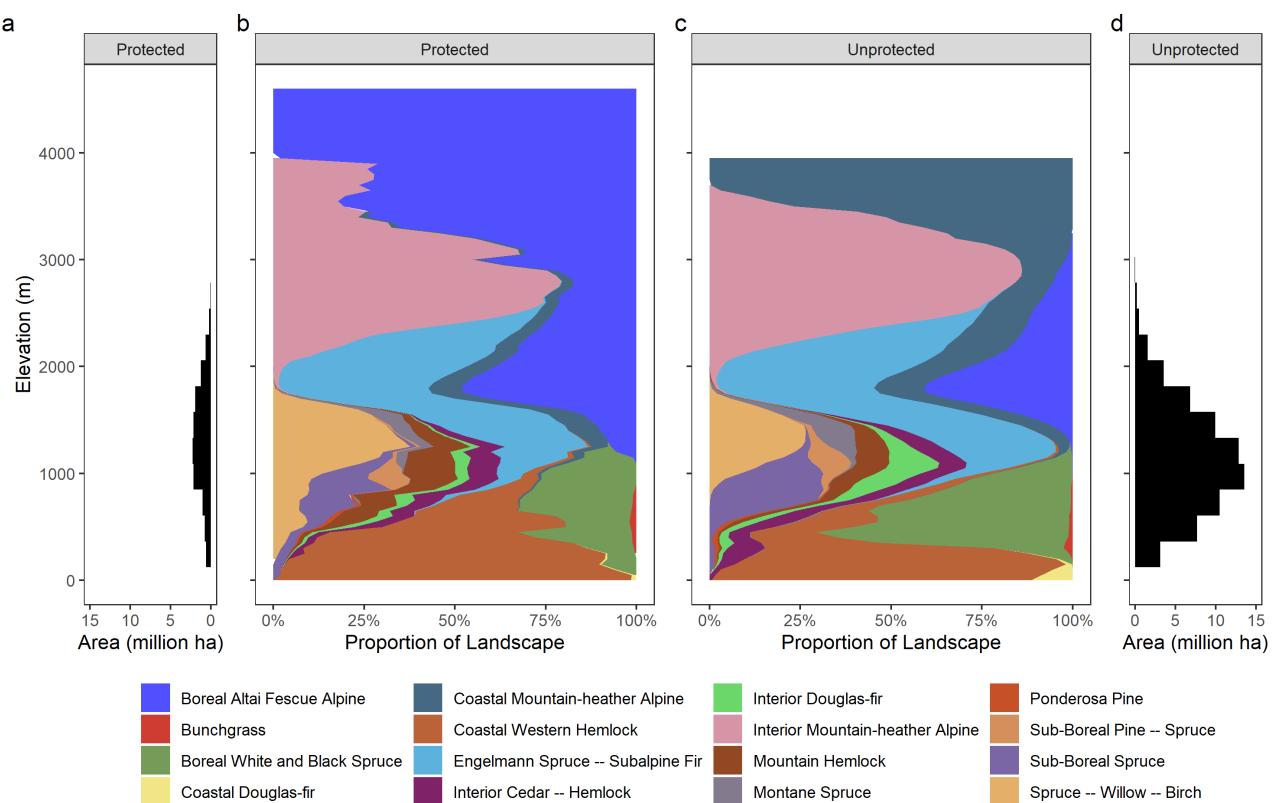


209

{fig. ??}

210 As elevation increases in BC, increasing terrestrial area is protected within the PA network until ~4000m,

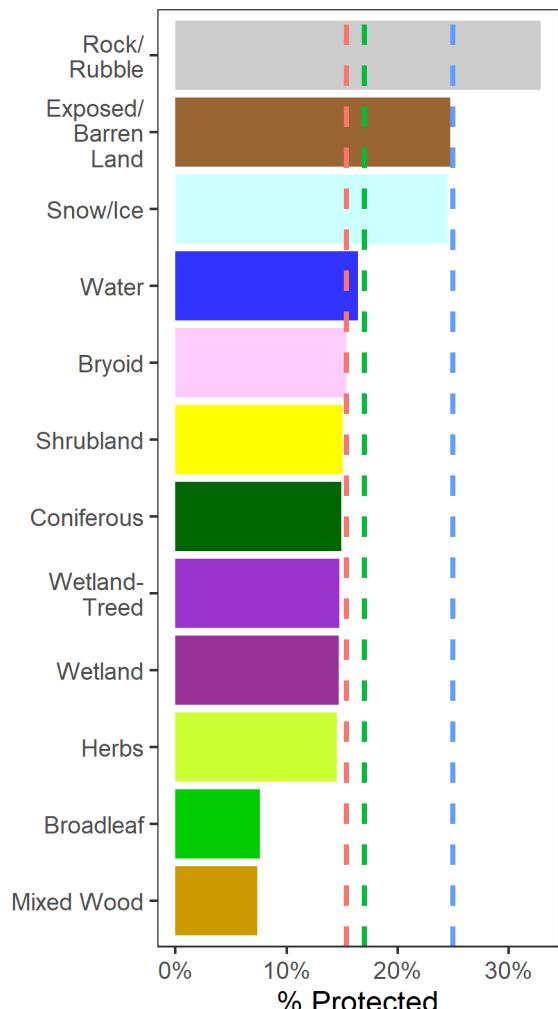
211 upon which all terrestrial area is protected (fig. ??). When comparing between PA and UA, zones are
 212 protected at differing proportions. Zones commonly found at high elevations, such as the Boreal Altai
 213 Fescue Alpine, are predominantly located in protected areas, however, little terrestrial area is found at
 214 these elevations. In low elevations, proportions of area protected also differ, with Coastal Western
 215 Hemlock having a large proportion of coverage in PA, while in UA, Boreal Black and White Spruce are
 216 underrepresented. Generally, the remaining ecosystems are found at similar rates in both PA and UA
 217 (fig. ??).



218

{fig. ??}

219 Protected land cover also varies by proportion (fig. ??). Non-vegetated classes of snow/ice, exposed/barren
 220 land, and rock/rubble have higher than average proportions protected while mixed wood and broadleaf
 221 land cover classes are underrepresented. All other classes are found at rates similar to the overall
 222 proportion of the province protected (~15%; fig. ??).

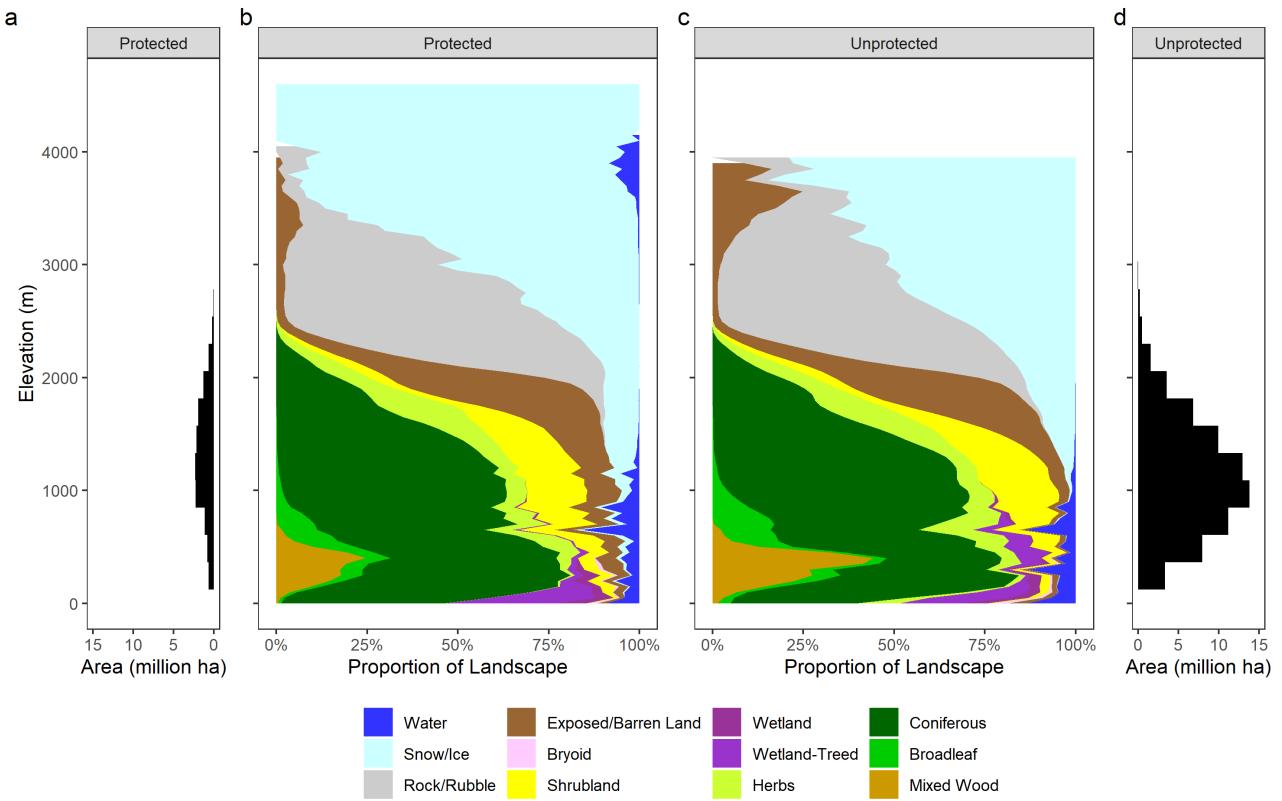


— 2020 BC Proportion Protected
— 2020 Protection Target (Global)
— 2025 Protection Target (Canada)

223

{fig. ??}

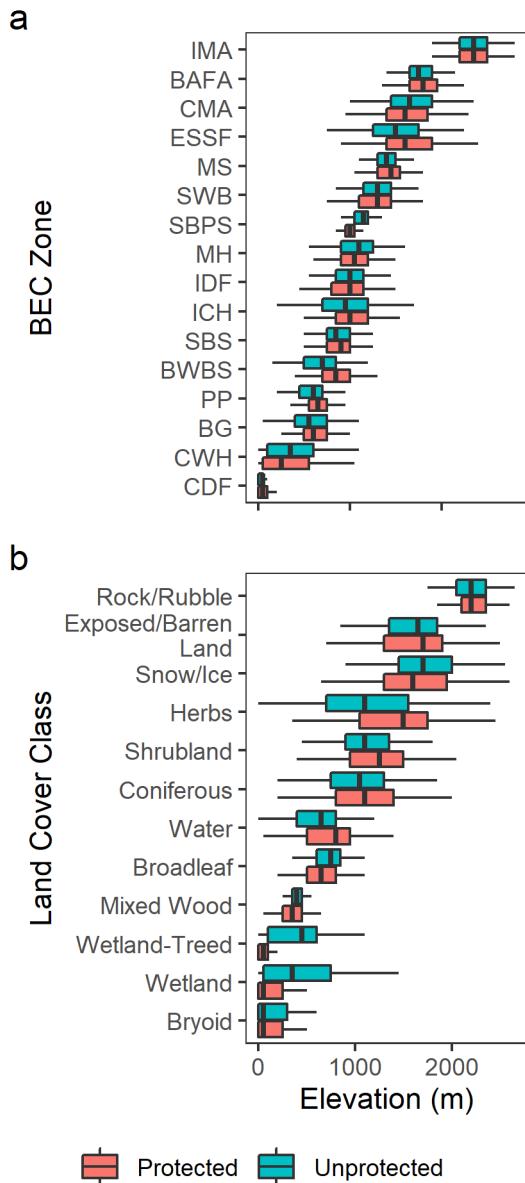
- 224 Similar to BEC zones (fig. ??), land cover also varies with elevation (fig. ??). Expectedly, snow/ice make up
 225 a large proportion of PA at higher elevations. At lower elevations in UA, mixed wood forest is a more
 226 common forest type than in PA, while wetland classes (wetland, wetland-treed) are less frequent in the
 227 400-900m elevation range in UA compared to PA.



228

{fig. ??}

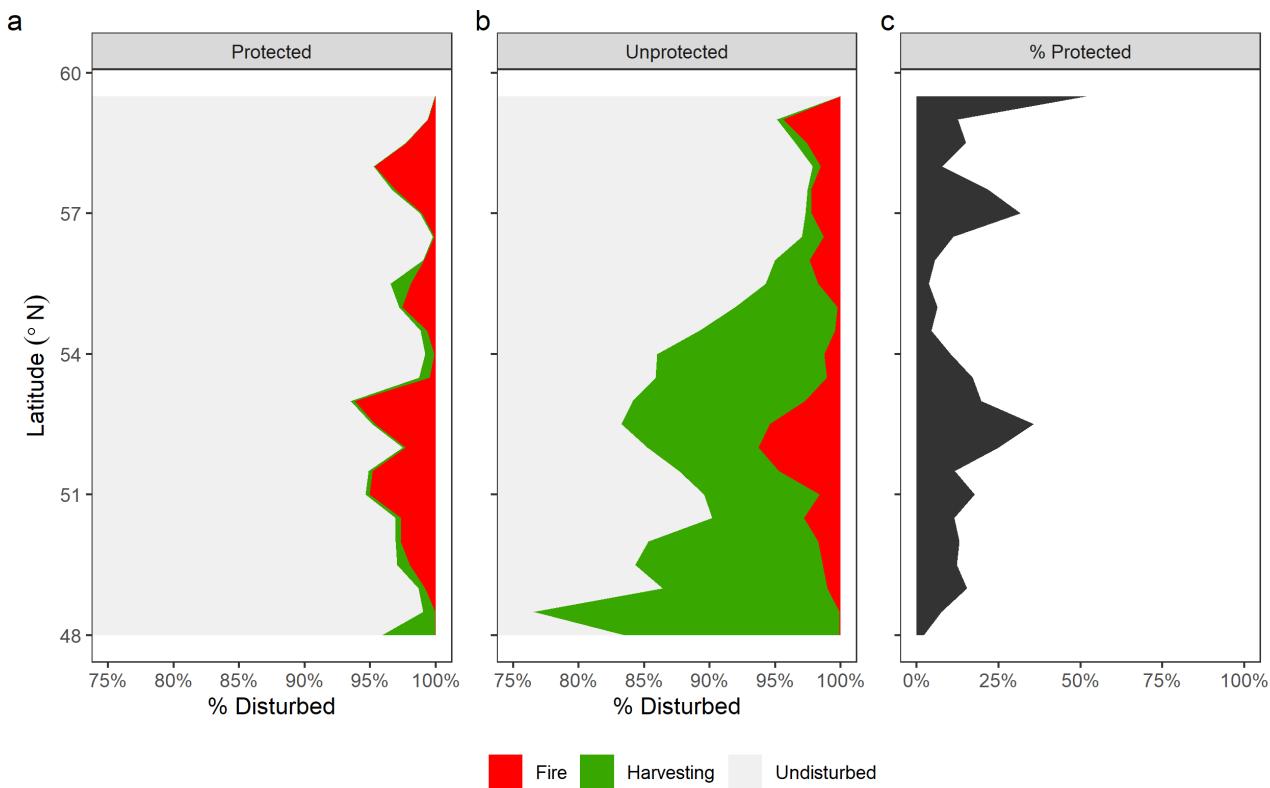
229 Examining the elevation distributions of BEC zones and land cover classes shows elevation variation in
 230 some classes and ecosystems (fig. ??). Generally, BEC zones are found at similar elevation profiles in both
 231 PA and UA. Alpine BEC zones (Interior Mountain-heather Alpine, Boreal Altai Fescue Alpine, and
 232 Coastal Mountain-heather Alpine) are found at similar elevations across PA and UA, while other zones
 233 such as Sub-Boreal Pine – Spruce, Ponderosa Pine, and Bunchgrass vary in their elevation profiles. Land
 234 cover classes show differences in the wetland, wetland-treed, and mixed wood classes. The wetland
 235 classes are found at lower elevations in PA than UA, while the mixed wood class has more variation in PA.



236

{fig. ??}

237 Overall, the burned area of forested cells is similar between PA (2.5% overall) and UA (2.3%), while
 238 harvesting is much higher in UA (7.2%) than in PA (0.33%). Harvesting is more common at lower latitudes
 239 in UA than at higher latitudes. Fire shows similar, but not identical patterns across varying latitudes, with
 240 higher wildfire proportions at high latitudes and between 51-53°N (fig. ??).

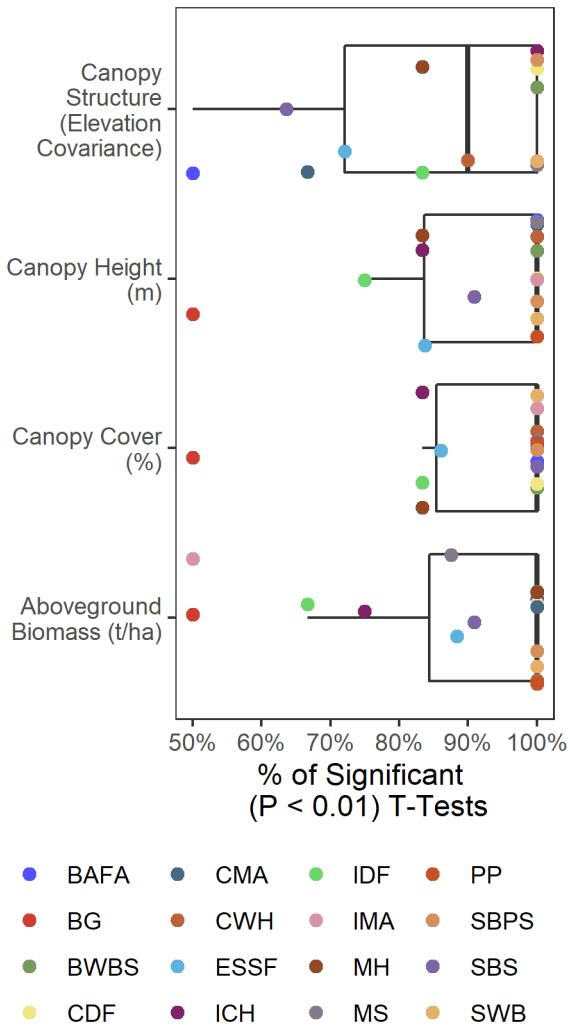


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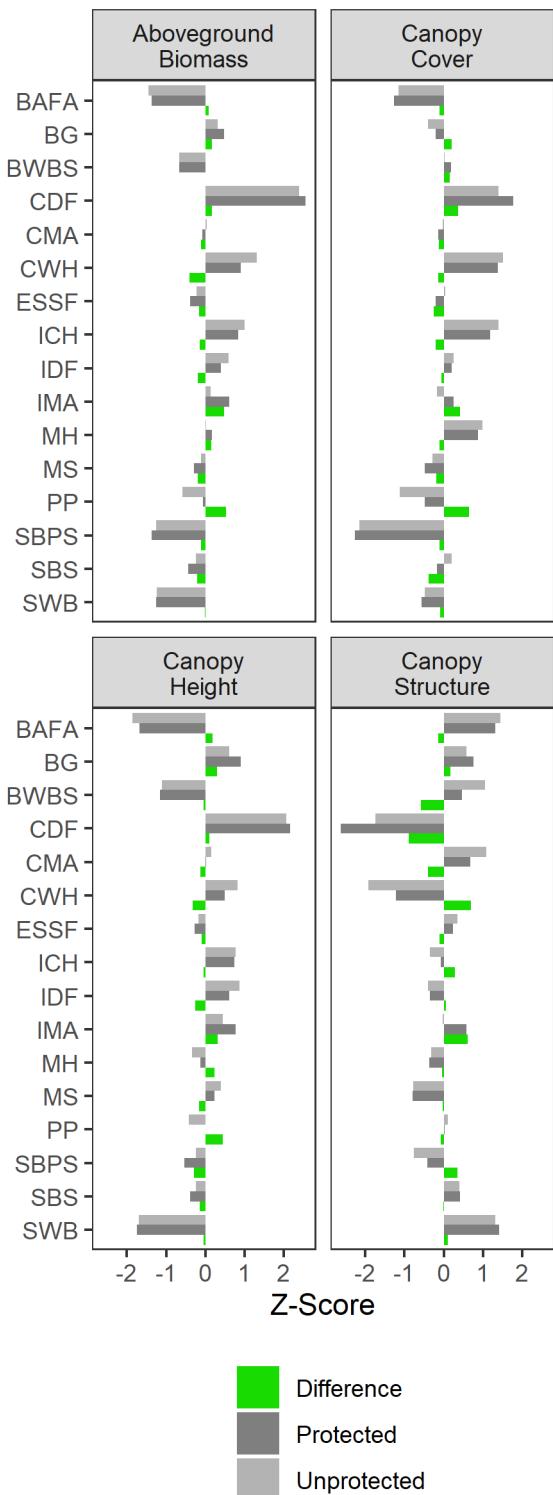
{fig. ??}

242 Forest Structural Attributes

243 fig. ?? shows the subzonal proportional significance ($p < 0.01$) grouped by ecosystem for the 496
 244 comparisons of forest structural variables. Higher percentages confirm ecosystems which had increased
 245 number of dissimilar subzones for the specific indicator, and shows that at least half of all subzones in
 246 each ecosystem are significantly different (the exception being Ponderosa Pine, which consists of a single
 247 subzone that is not significantly different in canopy structure). Median proportional significance values
 248 for canopy height, canopy cover, and aboveground biomass are universally significantly different between
 249 PA and UA within the same ecosystem.



251 Forest structural attributes vary between PA and UA in BC (fig. ??). The largest differences between PA
 252 and UA are found in canopy structure in the Coastal Douglas-fir BEC zone, with the protected area having
 253 much higher canopy structure values. As shown in fig. ??, forests are commonly significantly different
 254 when comparing PA vs UA across all attributes. When examining the forests on an BEC zone level, only
 255 one BEC zone has a >5% difference in vertical forest structure (co-efficient of variation in vegetation
 256 returns), six BEC zones have >5% difference in canopy cover, and five BEC zones have a >5% difference in
 257 canopy height. Ponderosa pine has large differences in canopy cover and canopy height (>5%), but minor
 258 differences in elevation covariance (only 0.25%; Table 2). PA in the Ponderosa Pine, Interior Mountain
 259 Heather Alpine, and Coastal Douglas-fir have more aboveground biomass than in UA in corresponding
 260 areas (fig. ??).



262 **Discussion**

263 The recent global availability of freely available, open-source, consistent, and accurate remote sensing data
264 products allow researchers to examine issues of representation of PA compared to UA, and regional
265 ecosystems in novel ways (Soverel et al. 2010, Hansen and Phillips 2018, Bolton et al. 2019). Additionally,
266 the capacity to track forest structural attributes, a key indicator of forest biodiversity (Guo et al. 2017),
267 across wide swaths allows for informed decisions on potential locations of new PA which capture
268 previously underrepresented forest structure conditions (Noss 1999). By applying this analysis to an entire
269 PA network across BEC zones (or other ecological classifications), it becomes possible to determine not
270 only which BEC zones need additional representation (the proportional metric), but also what types of
271 forest structures should be represented to ensure adequate biodiversity protection (Lemieux and Scott
272 2005).

273 Internationally, biodiversity preservation targets aim to protect a proportion of the total terrestrial area
274 (CBD 2010). Frequently, new protected areas are placed in high-elevation, low-productivity ecosystems
275 both globally (Joppa and Pfaff 2009, Venter et al. 2014, 2018), and in BC, as confirmed by our analysis of
276 ecosystem (fig. ??) and land cover (fig. ??) proportions. Alpine ecosystems are more commonly protected
277 (fig. ??), as are the land covers commonly present within them (rock/rubble, snow/ice, exposed/barren
278 land; fig. ??). As elevation increases, these ecosystems and land covers begin to dominate the proportional
279 representation (see fig. ?? and fig. ??). Differences between elevation profiles in land cover classes and
280 BEC zones were also found, with the starker difference being that wetland classes were found at lower
281 elevations in PA (fig. ??)

282 In high elevation ecosystems, Boreal Altai Fescue Alpine dominates the PA proportions above 3000m,
283 replacing the Coastal Mountain-Heather Alpine ecosystem found in UA (fig. ??). These zones were both
284 protected at rates above the average (fig. ??), and above the Aichi biodiversity targets. Interior
285 mountain-heather alpine had large differences in canopy cover and canopy height, while Boreal Altai
286 Fescue Alpine only showed large differences in height. The Coastal Mountain-heather Alpine did not any
287 have large forest structural attribute differences (Table 2).

288 Distribution of disturbances followed a similar pattern to that reported by Bolton et al. (2019). Thus, the
289 area affected by wildfires is comparable between PA and UA and at mid latitudes (51-53°N), while
290 harvesting activity is more prevalent in UA and at low latitudes (fig. ??).

291 Our analysis shows that the majority of structural attributes were significantly different between the
292 protected and unprotected forest stands across BEC subzones (fig. ??). In the south, Coastal Douglas-fir, a
293 zone with a single subzone, had the large variation between PA and UA in two of four forest structural
294 attributes examined. The unprotected forests were significantly less tall, had significantly less canopy
295 cover, and significantly higher elevation covariance (vertical forest structure; fig. ??). In addition, it was
296 the least protected BEC zone by area, with only 4.9% of the total terrestrial area protected. In this specific
297 BEC zone, not only does additional area need to be protected to meet national goals, different forest
298 structures need to be included in new protected areas (Paillet et al. 2010).

299 Utilizing this information on the proportion of BEC zones protected (fig. ??), as well as their forest
300 structural attributes (Table 2), it is possible to identify which forest structures need to be added to the PA
301 network in BC. Those BEC zones with large differences (identified as being >5% change from PA to UA)
302 suggest additional protection is needed to encapsulate these underrepresented forest structures. For
303 example: the forests in the Bunchgrass zone have large differences in both canopy cover and canopy
304 height, with the PA having larger values in both attributes (Table 2). New PA in this BEC zone should
305 contain forests with shorter and more open forests. A future avenue of research could be to incorporate
306 forest structural attributes into spatially optimized PA placement approaches (Christensen et al. 2009).

307 The advent of free and open-source global datasets can allow for the monitoring of protected area health
308 across the globe (Nagendra et al. 2013). Analyzing large amounts of free and open-source data using
309 open-source software approaches offers previously unseen perspectives into protected area
310 representativeness. There are some challenges associated with this, namely: optical imagery archives
311 being scarce in some regions due to imagery acquisition policies (Wulder et al. 2016), clouds and
312 atmospheric interference, lack of aerial lidar data available, and varying hierarchies of land cover
313 classifications in differing regions. New data and satellite missions are being introduced that can meet
314 these challenges at a spatial resolution of 30 m or less such as Landsat-9 and Sentinel-2, as well as
315 spaceborne lidar such as GEDI (Dubayah et al. 2020) and ICESat-2 (Neuenschwander et al. 2020), which
316 can provide global coverage of various forest structural attributes (Potapov et al. 2021) through similar
317 imputation methods to Matasci et al. (2018a), global land cover maps (Potapov et al. 2020, Zanaga et al.
318 2021), and forest disturbance maps (Hansen et al. 2013). These novel datasets provide clear opportunities
319 for regional to global analyses of PA vs UA to be conducted concerning forest structure.

320 Future research monitoring protected area health using satellite remote sensing could focus on

321 implementing essential biodiversity variables (Pereira et al. 2013) into their monitoring scheme.
322 Advancing research towards these variables would not only benefit PA monitoring projects, but also
323 biodiversity monitoring projects across the globe. Beyond this, examining the recovery of forest structural
324 attribute following disturbances in both PA and UA could assess the effectiveness of PA for promoting
325 regeneration.

326 Conclusion

327 In conclusion, we identified biases in the BC PA network for PA to be placed in high-elevation BEC zones,
328 commonly dominated by low-productivity land covers. We examined the disturbance regimes of PA vs UA
329 by latitude, finding that wildfires are similar, while harvesting differs across the province. We then
330 compared the forest structural attributes across all BEC subzones, finding that the majority of subzones
331 have significantly different forest structures. Beyond this, we identified BEC zones with large variation in
332 mean forest structural attributes. When new PA locations are decided upon in BC, they should take forest
333 structure into consideration, as wall-to-wall coverage of forest structural attributes becomes available.
334 Novel datasets can allow this methodology to be applied across large regions, in order to identify PA biases
335 and underrepresented forest structures.

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