

Assessing Representation of Remote Sensing Derived Forest Structure and Land Cover Across a Network of Protected Areas

Evan, R. Muise¹ Nicholas, C. Coops¹ Txomin Hermosilla² Stephen, S. Ban³

¹ Department of Forest Resources Management, University of British Columbia ² Canadian Forest Service (Pacific Forestry Centre) ³ BC Parks, Ministry of Environment and Climate Change Strategy

Correspondance to:

Evan, R. Muise — evanmuise@student.ubc.ca

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 ([tab:buc-table?](#))).

Table 1: Number of subzones, total area, and percent protected by BEC Zone.

Zone	Zone Name	# of Subzones	Area (ha)	% Protected
BAFA	Boreal Altai Fescue Alpine	2	6,286,778	30.1%
BG	Bunchgrass	2	257,072	11.8%
BWBS	Boreal White and Black Spruce	5	16,404,142	8.6%
CDF	Coastal Douglas-fir	1	251,232	4.8%
CMA	Coastal Mountain-heather Alpine	3	3,574,039	17.9%
CWH	Coastal Western Hemlock	10	10,795,067	19.5%
ESSF	Engelmann Spruce – Subalpine Fir	43	17,465,113	17.8%

Zone	Zone Name	# of Subzones	Area (ha)	% Protected
ICH	Interior Cedar – Hemlock	12	5,538,842	10.2%
IDF	Interior Douglas-fir	12	4,488,085	5.9%
IMA	Interior Mountain-heather Alpine	2	1,257,949	29.2%
MH	Mountain Hemlock	6	4,059,301	19.8%
MS	Montane Spruce	8	2,863,394	9.4%
PP	Ponderosa Pine	1	294,985	7.1%
SBPS	Sub-Boreal Pine – Spruce	4	2,265,365	9.5%
SBS	Sub-Boreal Spruce	11	10,337,497	6.7%
SWB	Spruce – Willow – Birch	6	8,655,855	23.3%

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. 1).

112 [Figure 1 about here.]

113 Data

114 Biogeoclimatic Ecosystem Classification and Protected Areas

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

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

131 **Digital Elevation Model**

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

135 **Landsat derived datasets**

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

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

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

165 Forest cover classes (deciduous, broadleaf, mixed-wood, and wetland-treed) were used to generate land
166 cover masks to restrict the comparison of forest structural attributes to treed pixels. Pixels with harvest
167 activity disturbances detected post-1985 were also removed from forest structural attribute rasters in both
168 PA and UA, in order to restrict analysis to non-anthropogenically disturbed areas. All datasets are
169 displayed in fig. 2.

170 [Figure 2 about here.]

171 Analysis

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

178 representation in the current BC PA network. All data manipulation and analysis was conducted in the **R**
179 (R Core Team 2021) or **Python** programming languages.

180 **Ecosystems, Land Cover, and Disturbances**

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

190 **Forest Structural Attributes**

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

200 **Results**

201 **Ecosystems, Land Cover, and Disturbances**

202 British Columbia's ecosystems are protected at varying rates across the province (fig. 3). Of the 16
203 ecosystems present in BC, seven are protected at rates above the Aichi biodiversity target (17%). Only two
204 zones (Boreal Altai Fescue Alpine and Interior Mountain-heather Alpine) are currently protected at rates
205 above the Canadian 2025 protection targets (25%). Zones with Douglas-fir (*Pseudotsuga menziesii*) as
206 dominant old-growth components (Coastal Douglas-fir and Interior Douglas-fir) are the least
207 proportionally represented zones in British Columbia, with 4.9% and 6.4% protected, respectively (fig. 3).

208 [Figure 3 about here.]

209 As elevation increases in BC, increasing terrestrial area is protected within the PA network until ~4000m,
210 upon which all terrestrial area is protected (fig. 4). When comparing between PA and UA, zones are
211 protected at differing proportions. Zones commonly found at high elevations, such as the Boreal Altai
212 Fescue Alpine, are predominantly located in protected areas, however, little terrestrial area is found at
213 these elevations. In low elevations, proportions of area protected also differ, with Coastal Western
214 Hemlock having a large proportion of coverage in PA, while in UA, Boreal Black and White Spruce are
215 underrepresented. Generally, the remaining ecosystems are found at similar rates in both PA and UA
216 (fig. 4).

217 [Figure 4 about here.]

218 Protected land cover also varies by proportion (fig. 5). Non-vegetated classes of snow/ice, exposed/barren
219 land, and rock/rubble have higher than average proportions protected while mixed wood and broadleaf
220 land cover classes are underrepresented. All other classes are found at rates similar to the overall
221 proportion of the province protected (~15%; fig. 5).

222 [Figure 5 about here.]

223 Similar to BEC zones (fig. 4), land cover also varies with elevation (fig. 6). Expectedly, snow/ice make up a
224 large proportion of PA at higher elevations. At lower elevations in UA, mixed wood forest is a more

225 common forest type than in PA, while wetland classes (wetland, wetland-treed) are less frequent in the
226 400-900m elevation range in UA compared to PA.

227 [Figure 6 about here.]

228 Examining the elevation distributions of BEC zones and land cover classes shows elevation variation in
229 some classes and ecosystems (fig. 7). Generally, BEC zones are found at similar elevation profiles in both
230 PA and UA. Alpine BEC zones (Interior Mountain-heather Alpine, Boreal Altai Fescue Alpine, and
231 Coastal Mountain-heather Alpine) are found at similar elevations across PA and UA, while other zones
232 such as Sub-Boreal Pine – Spruce, Ponderosa Pine, and Bunchgrass vary in their elevation profiles. Land
233 cover classes show differences in the wetland, wetland-treed, and mixed wood classes. The wetland
234 classes are found at lower elevations in PA than UA, while the mixed wood class has more variation in PA.

235 [Figure 7 about here.]

236 Overall, the burned area of forested cells is similar between PA (2.5% overall) and UA (2.3%), while
237 harvesting is much higher in UA (7.2%) than in PA (0.33%). Harvesting is more common at lower latitudes
238 in UA than at higher latitudes. Fire shows similar, but not identical patterns across varying latitudes, with
239 higher wildfire proportions at high latitudes and between 51-53°N (fig. 8).

240 [Figure 8 about here.]

241 **Forest Structural Attributes**

242 fig. 9 shows the subzonal proportional significance ($p < 0.01$) grouped by ecosystem for the 496
243 comparisons of forest structural variables. Higher percentages confirm ecosystems which had increased
244 number of dissimilar subzones for the specific indicator, and shows that at least half of all subzones in
245 each ecosystem are significantly different (the exception being Ponderosa Pine, which consists of a single
246 subzone that is not significantly different in canopy structure). Median proportional significance values
247 for canopy height, canopy cover, and aboveground biomass are universally significantly different between
248 PA and UA within the same ecosystem.

249 [Figure 9 about here.]

250 Forest structural attributes vary between PA and UA in BC (fig. 10). The largest differences between PA
251 and UA are found in canopy structure in the Coastal Douglas-fir BEC zone, with the protected area having
252 much higher canopy structure values. As shown in fig. 9, forests are commonly significantly different
253 when comparing PA vs UA across all attributes. When examining the forests on an BEC zone level, only
254 one BEC zone has a >5% difference in vertical forest structure (co-efficient of variation in vegetation
255 returns), six BEC zones have >5% difference in canopy cover, and five BEC zones have a >5% difference in
256 canopy height. Ponderosa pine has large differences in canopy cover and canopy height (>5%), but minor
257 differences in elevation covariance (only 0.25%; tbl. ??). PA in the Ponderosa Pine, Interior Mountain
258 Heather Alpine, and Coastal Douglas-fir have more aboveground biomass than in UA in corresponding
259 areas (fig. 10).

260 Zone

261 Elevation Covariance

262 Canopy Cover (%)

263 Canopy Height (m)

264 PA

265 UA

266 % Difference

267 PA

268 UA

269 % Difference

270 PA

271 UA

272 % Difference

273 BAFA

274 0.39

275 0.39

276 0.01%

277 46.94%

278 48.47%

279 3.17%

280 14.03

281 13.11

282 -7%

283 BG

284 0.38

285 0.38

286 -1.41%

287 61.8%

288 58.71%

289 -5.26%

290 23.00

291 21.58

292 -6.59%

293 BWBS

294 0.37

295 0.38

296 2.62%

297 67.18%

298 64.72%

299 -3.8%

300 **15.83**

301 **15.69**

302 **-0.9%**

303 **CDF**

304 **0.31**

305 **0.33**

306 **6.35%**

307 **89.38%**

308 **83.69%**

309 **-6.79%**

310 **27.35**

311 **26.58**

312 **-2.89%**

313 **CMA**

314 **0.38**

315 **0.39**

316 **1.63%**

317 **62.65%**

318 **64.01%**

319 **2.13%**

320 **19.91**

321 **20.01**

322 **0.48%**

323 **CWH**

324 0.34

325 0.33

326 -3.83%

327 83.94%

328 85.26%

329 1.55%

330 21.58

331 22.34

332 3.4%

333 ESSF

334 0.37

335 0.37

336 0.34%

337 61.7%

338 64.97%

339 5.04%

340 18.90

341 18.91

342 0.07%

343 ICH

344 0.36

345 0.36

346 -1.75%

347 81.24%

348 83.52%

349 2.73%

350 22.39

351 22.18

352 -0.98%

353 IDF

354 0.36

355 0.36

356 -0.3%

357 67.42%

358 67.9%

359 0.71%

360 21.98

361 22.49

362 2.29%

363 IMA

364 0.38

365 0.36

366 -3.72%

367 68.17%

368 62.07%

369 -9.83%

370 22.53

371 21.06

372 -6.98%

373 MH

374 0.36

375 0.36

376 0.25%

377 76.87%

378 77.85%

379 1.26%

380 19.42

381 18.31

382 -6.07%

383 MS

384 0.35

385 0.35

386 0.31%

387 57.99%

388 60.41%

389 4.01%

390 20.64

391 20.86

392 1.04%

393 PP

394 0.36

395 0.37

396 0.25%

397 57.92%

398 48.93%

399 -18.36%

400 19.88

401 18.03

402 -10.24%

403 SBPS

404 0.36

405 0.35

406 -1.97%

407 32.98%

408 34.63%

409 4.76%

410 18.00

411 18.70

412 3.75%

413 SBS

414 0.37

415 0.37

416 -0.4%

417 62.24%

418 67.25%

419 7.45%

420 18.51
421 18.67
422 0.82%
423 SWB
424 0.39
425 0.39
426 -1.22%
427 56.67%
428 57.71%
429 1.8%
430 13.78
431 13.67
432 -0.83%
433 : Mean values of forest structural attributes in protected areas (PA), unprotected areas (UA), as well as the
434 percent difference between the means. Zones with more than a 5% difference are bolded.
435 {#tbl:vector-table}

436 [Figure 10 about here.]

437 Discussion

438 The recent global availability of freely available, open-source, consistent, and accurate remote sensing data
439 products allow researchers to examine issues of representation of PA compared to UA, and regional
440 ecosystems in novel ways (Soverel et al. 2010, Hansen and Phillips 2018, Bolton et al. 2019). Additionally,
441 the capacity to track forest structural attributes, a key indicator of forest biodiversity (Guo et al. 2017),
442 across wide swaths allows for informed decisions on potential locations of new PA which capture
443 previously underrepresented forest structure conditions (Noss 1999). By applying this analysis to an entire

444 PA network across BEC zones (or other ecological classifications), it becomes possible to determine not
445 only which BEC zones need additional representation (the proportional metric), but also what types of
446 forest structures should be represented to ensure adequate biodiversity protection (Lemieux and Scott
447 2005).

448 Internationally, biodiversity preservation targets aim to protect a proportion of the total terrestrial area
449 (CBD 2010). Frequently, new protected areas are placed in high-elevation, low-productivity ecosystems
450 both globally (Joppa and Pfaff 2009, Venter et al. 2014, 2018), and in BC, as confirmed by our analysis of
451 ecosystem (fig. 3) and land cover (fig. 5) proportions. Alpine ecosystems are more commonly protected
452 (fig. 3), as are the land covers commonly present within them (rock/rubble, snow/ice, exposed/barren
453 land; fig. 5). As elevation increases, these ecosystems and land covers begin to dominate the proportional
454 representation (see fig. 4 and fig. 6). Differences between elevation profiles in land cover classes and BEC
455 zones were also found, with the starker difference being that wetland classes were found at lower
456 elevations in PA (fig. 7)

457 In high elevation ecosystems, Boreal Altai Fescue Alpine dominates the PA proportions above 3000m,
458 replacing the Coastal Mountain-Heather Alpine ecosystem found in UA (fig. 4). These zones were both
459 protected at rates above the average (fig. 3), and above the Aichi biodiversity targets. Interior
460 mountain-heather alpine had large differences in canopy cover and canopy height, while Boreal Altai
461 Fescue Alpine only showed large differences in height. The Coastal Mountain-heather Alpine did not any
462 have large forest structural attribute differences (tbl. ??).

463 Distribution of disturbances followed a similar pattern to that reported by Bolton et al. (2019). Thus, the
464 area affected by wildfires is comparable between PA and UA and at mid latitudes (51-53°N), while
465 harvesting activity is more prevalent in UA and at low latitudes (fig. 8).

466 Our analysis shows that the majority of structural attributes were significantly different between the
467 protected and unprotected forest stands across BEC subzones (fig. 9). In the south, Coastal Douglas-fir, a
468 zone with a single subzone, had the large variation between PA and UA in two of four forest structural
469 attributes examined. The unprotected forests were significantly less tall, had significantly less canopy
470 cover, and significantly higher elevation covariance (vertical forest structure; fig. 10). In addition, it was
471 the least protected BEC zone by area, with only 4.9% of the total terrestrial area protected. In this specific
472 BEC zone, not only does additional area need to be protected to meet national goals, different forest
473 structures need to be included in new protected areas (Paillet et al. 2010).

474 Utilizing this information on the proportion of BEC zones protected (fig. 3), as well as their forest
475 structural attributes (tbl. ??), it is possible to identify which forest structures need to be added to the PA
476 network in BC. Those BEC zones with large differences (identified as being >5% change from PA to UA)
477 suggest additional protection is needed to encapsulate these underrepresented forest structures. For
478 example: the forests in the Bunchgrass zone have large differences in both canopy cover and canopy
479 height, with the PA having larger values in both attributes (tbl. ??). New PA in this BEC zone should
480 contain forests with shorter and more open forests. A future avenue of research could be to incorporate
481 forest structural attributes into spatially optimized PA placement approaches (Christensen et al. 2009).

482 The advent of free and open-source global datasets can allow for the monitoring of protected area health
483 across the globe (Nagendra et al. 2013). Analyzing large amounts of free and open-source data using
484 open-source software approaches offers previously unseen perspectives into protected area
485 representativeness. There are some challenges associated with this, namely: optical imagery archives
486 being scarce in some regions due to imagery acquisition policies (Wulder et al. 2016), clouds and
487 atmospheric interference, lack of aerial lidar data available, and varying hierarchies of land cover
488 classifications in differing regions. New data and satellite missions are being introduced that can meet
489 these challenges at a spatial resolution of 30 m or less such as Landsat-9 and Sentinel-2, as well as
490 spaceborne lidar such as GEDI (Dubayah et al. 2020) and ICESat-2 (Neuenschwander et al. 2020), which
491 can provide global coverage of various forest structural attributes (Potapov et al. 2021) through similar
492 imputation methods to Matasci et al. (2018a), global land cover maps (Potapov et al. 2020, Zanaga et al.
493 2021), and forest disturbance maps (Hansen et al. 2013). These novel datasets provide clear opportunities
494 for regional to global analyses of PA vs UA to be conducted concerning forest structure.

495 Future research monitoring protected area health using satellite remote sensing could focus on
496 implementing essential biodiversity variables (Pereira et al. 2013) into their monitoring scheme.
497 Advancing research towards these variables would not only benefit PA monitoring projects, but also
498 biodiversity monitoring projects across the globe. Beyond this, examining the recovery of forest structural
499 attribute following disturbances in both PA and UA could assess the effectiveness of PA for promoting
500 regeneration.

501 **Conclusion**

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

511 **Acknowledgments**

512 This research was funded through the Living Lab for Climate Change and Conservation program of the
513 British Columbia Parks (grant ID: TP21JHQ011) and in part by NSERC support of Coops
514 (RGPIN-2018-03851). Remote sensing data products utilized in this research are free and open and
515 available for download at: https://ca.nfis.org/maps_eng.html. We thank Dr Michael Wulder and Dr
516 Joanne White for development and early access to these National Terrestrial Ecosystem Mapping System
517 (NTEMS) products. We also thank an anonymous reviewer for their comments which helped improve the
518 manuscript.

519 **Literature Cited**

- 520 Alsdorf, D. E., E. Rodriguez, and D. P. Lettenmaier. 2007. Measuring surface water from space. *Reviews of
521 Geophysics* 45:RG2002.
- 522 BC Ministry of Forests. 2003. British Columbia's forests and their management.
- 523 BC Parks. 2012. Ecological Integrity in British Columbia's Parks and Protected Areas.
- 524 Bolton, D. K., N. C. Coops, T. Hermosilla, M. A. Wulder, J. C. White, and C. J. Ferster. 2019. Uncovering
525 regional variability in disturbance trends between parks and greater park ecosystems across Canada
526 (1985). *Scientific Reports* 9:1323.
- 527 Bonferroni, C. E. 1936. Teoria statistica delle classi e calcolo delle probabilità. Pages 3–62.
- 528 Brooks, T. M., M. I. Bakarr, T. Boucher, G. A. B. Da Fonseca, C. Hilton-Taylor, J. M. Hoekstra, T. Moritz, S.
529 Olivieri, J. Parrish, R. L. Pressey, A. S. L. Rodrigues, W. Sechrest, A. Stattersfield, W. Strahm, and S. N.
530 Stuart. 2004. Coverage Provided by the Global Protected-Area System: Is It Enough? *BioScience*
531 54:1081.
- 532 Buchanan, G. M., A. E. Beresford, M. Hebblewhite, F. J. Escobedo, H. M. D. Klerk, P. F. Donald, P.
533 Escribano, L. P. Koh, J. Martínez-López, N. Pettorelli, A. K. Skidmore, Z. Szantoi, K. Tabor, M.
534 Wegmann, and S. Wich. 2018. Free satellite data key to conservation. *Science* 361:139–140.
- 535 Burkhard, B., F. Kroll, S. Nedkov, and F. Müller. 2012. Mapping ecosystem service supply, demand and
536 budgets. *Ecological Indicators* 21:17–29.

- 537 Butchart, S. H. M., M. Clarke, R. J. Smith, R. E. Sykes, J. P. W. Scharlemann, M. Harfoot, G. M. Buchanan,
538 A. Angulo, A. Balmford, B. Bertzky, T. M. Brooks, K. E. Carpenter, M. T. Comeros-Raynal, J. Cornell, G.
539 F. Ficetola, L. D. C. Fishpool, R. A. Fuller, J. Geldmann, H. Harwell, C. Hilton-Taylor, M. Hoffmann, A.
540 Joolia, L. Joppa, N. Kingston, I. May, A. Milam, B. Polidoro, G. Ralph, N. Richman, C. Rondinini, D. B.
541 Segan, B. Skolnik, M. D. Spalding, S. N. Stuart, A. Symes, J. Taylor, P. Visconti, J. E. M. Watson, L.
542 Wood, and N. D. Burgess. 2015. Shortfalls and Solutions for Meeting National and Global
543 Conservation Area Targets. *Conservation Letters* 8:329–337.
- 544 CBD. 2004. CoP 7 decision VII/30. Strategic plan: future evaluation of progress. Goal 1 promote the
545 conservation of the biological diversity of ecosystems, habitats and biomes; Target 1.1.
- 546 CBD. 2010. The strategic plan for biodiversity 2011-2020 and the Aichi biodiversity targets.
- 547 Chape, S., J. Harrison, M. Spalding, and I. Lysenko. 2005. Measuring the extent and effectiveness of
548 protected areas as an indicator for meeting global biodiversity targets. *Philosophical Transactions of*
549 *the Royal Society B: Biological Sciences* 360:443–455.
- 550 Christensen, V., Z. Ferdaña, and J. Steenbeek. 2009. Spatial optimization of protected area placement
551 incorporating ecological, social and economical criteria. *Ecological Modelling* 220:2583–2593.
- 552 Coad, L., F. Leverington, K. Knights, J. Geldmann, A. Eassom, V. Kapos, N. Kingston, M. de Lima, C.
553 Zamora, I. Cuardros, C. Nolte, N. D. Burgess, and M. Hockings. 2015. Measuring impact of protected
554 area management interventions: Current and future use of the global database of protected area
555 management effectiveness. *Philosophical Transactions of the Royal Society B: Biological Sciences*
556 370:20140281.
- 557 Cohen, W. B., and S. N. Goward. 2004. Landsat's role in ecological applications of remote sensing.
558 *Bioscience* 54:535–545.
- 559 Defries, R., A. Hansen, A. Newton, and M. Hansen. 2005. Increasing isolation of protected areas in
560 tropical forests over the past twenty years. *Ecological Applications* 15:19–26.
- 561 Dinerstein, E., D. Olson, A. Joshi, C. Vynne, N. D. Burgess, E. Wikramanayake, N. Hahn, S. Palminteri, P.
562 Hedao, R. Noss, M. Hansen, H. Locke, E. C. Ellis, B. Jones, C. V. Barber, R. Hayes, C. Kormos, V.
563 Martin, E. Crist, W. Sechrest, L. Price, J. E. M. Baillie, D. Weeden, K. Suckling, C. Davis, N. Sizer, R.
564 Moore, D. Thau, T. Birch, P. Potapov, S. Turubanova, A. Tyukavina, N. de Souza, L. Pintea, J. C. Brito,
565 O. A. Llewellyn, A. G. Miller, A. Patzelt, S. A. Ghazanfar, J. Timberlake, H. Klöser, Y. Shennan-Farpón,

- 566 R. Kindt, J.-P. B. Lillesø, P. van Breugel, L. Graudal, M. Voge, K. F. Al-Shammari, and M. Saleem. 2017.
567 An Ecoregion-Based Approach to Protecting Half the Terrestrial Realm. *BioScience* 67:534–545.
- 568 Dinerstein, E., C. Vynne, E. Sala, A. R. Joshi, S. Fernando, T. E. Lovejoy, J. Mayorga, D. Olson, G. P. Asner,
569 J. E. M. Baillie, N. D. Burgess, K. Burkart, R. F. Noss, Y. P. Zhang, A. Baccini, T. Birch, N. Hahn, L. N.
570 Joppa, and E. Wikramanayake. 2019. A Global Deal For Nature: Guiding principles, milestones, and
571 targets. *Science Advances* 5:eaaw2869.
- 572 Dubayah, R., J. B. Blair, S. Goetz, L. Fatoyinbo, M. Hansen, S. Healey, M. Hofton, G. Hurt, J. Kellner, S.
573 Luthcke, J. Armston, H. Tang, L. Duncanson, S. Hancock, P. Jantz, S. Marselis, P. L. Patterson, W. Qi,
574 and C. Silva. 2020. The Global Ecosystem Dynamics Investigation: High-resolution laser ranging of
575 the Earth's forests and topography. *Science of Remote Sensing* 1:100002.
- 576 ECCC. 2021, May 3. Canada Target 1 Challenge.
- 577 Eklund, J., L. Coad, J. Geldmann, and M. Cabeza. 2019. What constitutes a useful measure of protected
578 area effectiveness? A case study of management inputs and protected area impacts in Madagascar.
579 *Conservation Science and Practice* 1:e107.
- 580 Environmental Reporting BC. 2016. Protected Lands and Waters in British Columbia.
581 <http://www.env.gov.bc.ca/soe/indicators/land/protected-lands-and-waters.html>.
- 582 Feeley, K. J., T. W. Gillespie, and J. W. Terborgh. 2005. The Utility of Spectral Indices from Landsat ETM+
583 for Measuring the Structure and Composition of Tropical Dry Forests. *Biotropica* 37:508–519.
- 584 Ferraro, P. J. 2009. Counterfactual thinking and impact evaluation in environmental policy. *New
585 Directions for Evaluation* 2009:75–84.
- 586 Fraser, R. H., I. Olthof, and D. Pouliot. 2009. Monitoring land cover change and ecological integrity in
587 Canada's national parks. *Remote Sensing of Environment* 113:1397–1409.
- 588 Gao, T., M. Hedblom, T. Emilsson, and A. B. Nielsen. 2014. The role of forest stand structure as
589 biodiversity indicator. *Forest Ecology and Management* 330:82–93.
- 590 Gaston, K. J., K. Charman, S. F. Jackson, P. R. Armsworth, A. Bonn, R. A. Briers, C. S. Q. Callaghan, R.
591 Catchpole, J. Hopkins, W. E. Kunin, J. Latham, P. Opdam, R. Stoneman, D. A. Stroud, and R. Tratt.
592 2006. The ecological effectiveness of protected areas: The United Kingdom. *Biological Conservation*
593 132:76–87.

- 594 Gaston, K. J., S. F. Jackson, L. Cantú-Salazar, and G. Cruz-Piñón. 2008. The Ecological Performance of
595 Protected Areas. *Annual Review of Ecology, Evolution, and Systematics* 39:93–113.
- 596 Geldmann, J., A. Manica, N. D. Burgess, L. Coad, and A. Balmford. 2019. A global-level assessment of the
597 effectiveness of protected areas at resisting anthropogenic pressures. *Proceedings of the National
598 Academy of Sciences* 116:23209–23215.
- 599 Gillespie, T. W. 2005. Predicting Woody-Plant Species Richness in Tropical Dry Forests: A Case Study from
600 South Florida, Usa. *Ecological Applications* 15:27–37.
- 601 Goetz, S., D. Steinberg, R. Dubayah, and B. Blair. 2007. Laser remote sensing of canopy habitat
602 heterogeneity as a predictor of bird species richness in an eastern temperate forest, USA. *Remote
603 Sensing of Environment* 108:254–263.
- 604 Government of Canada,. 2019, September 4. Canada National Park Act.
- 605 Guo, X., N. C. Coops, P. Tompalski, S. E. Nielsen, C. W. Bater, and J. John Stadt. 2017. Regional mapping
606 of vegetation structure for biodiversity monitoring using airborne lidar data. *Ecological Informatics*
607 38:50–61.
- 608 Hamann, A., P. Smets, A. D. Yanchuk, and S. N. Aitken. 2005. An ecogeographic framework for in situ
609 conservation of forest trees in British Columbia. *Canadian Journal of Forest Research* 35:2553–2561.
- 610 Hansen, A. J., B. P. Noble, J. Veneros, A. East, S. J. Goetz, C. Supples, J. E. M. Watson, P. A. Jantz, R. Pillay,
611 W. Jetz, S. Ferrier, H. S. Grantham, T. D. Evans, J. Ervin, O. Venter, and A. L. S. Virnig. 2021. Toward
612 monitoring forest ecosystem integrity within the post-2020 Global Biodiversity Framework.
613 *Conservation Letters* 14:e12822.
- 614 Hansen, A. J., and L. Phillips. 2018. Trends in vital signs for Greater Yellowstone: Application of a
615 Wildland Health Index. *Ecosphere* 9:e02380.
- 616 Hansen, M. C., P. V. Potapov, R. Moore, M. Hancher, S. A. Turubanova, A. Tyukavina, D. Thau, S. V.
617 Stehman, S. J. Goetz, T. R. Loveland, A. Kommareddy, A. Egorov, L. Chini, C. O. Justice, and J. R. G.
618 Townshend. 2013. High-Resolution Global Maps of 21st-Century Forest Cover Change. *Science*
619 342:850–853.
- 620 Hazen, H. D., and P. J. Anthamatten. 2004. Representation of ecological regions by protected areas at the
621 global scale. *Physical Geography* 25:499–512.

- 622 Hermosilla, T., M. A. Wulder, J. C. White, N. C. Coops, and G. W. Hobart. 2015a. An integrated Landsat
623 time series protocol for change detection and generation of annual gap-free surface reflectance
624 composites. *Remote Sensing of Environment* 158:220–234.
- 625 Hermosilla, T., M. A. Wulder, J. C. White, N. C. Coops, and G. W. Hobart. 2015b. Regional detection,
626 characterization, and attribution of annual forest change from 1984 to 2012 using landsat-derived
627 time-series metrics. *Remote Sensing of Environment* 170:121132.
- 628 Hermosilla, T., M. A. Wulder, J. C. White, N. C. Coops, and G. W. Hobart. 2018. Disturbance-Informed
629 Annual Land Cover Classification Maps of Canada's Forested Ecosystems for a 29-Year Landsat Time
630 Series. *Canadian Journal of Remote Sensing* 44:67–87.
- 631 Hermosilla, T., M. A. Wulder, J. C. White, N. C. Coops, G. W. Hobart, and L. B. Campbell. 2016. Mass data
632 processing of time series Landsat imagery: Pixels to data products for forest monitoring. *International
633 Journal of Digital Earth* 9:1035–1054.
- 634 Joppa, L. N., and A. Pfaff. 2009. High and Far: Biases in the Location of Protected Areas. *PLOS ONE*
635 4:e8273.
- 636 Kerr, J. T., and M. Ostrovsky. 2003. From space to species: ecological applications for remote sensing.
637 *Trends in Ecology & Evolution* 18:299–305.
- 638 Lemieux, C. J., and D. J. Scott. 2005. Climate change, biodiversity conservation and protected area
639 planning in Canada. *The Canadian Geographer / Le Géographe canadien* 49:384–397.
- 640 Lim, K., P. Treitz, M. Wulder, B. St-Onge, and M. Flood. 2003. LiDAR remote sensing of forest structure.
641 *Progress in Physical Geography: Earth and Environment* 27:88–106.
- 642 Lucas, R., K. Medcalf, A. Brown, P. Bunting, J. Breyer, D. Clewley, S. Keyworth, and P. Blackmore. 2011.
643 Updating the Phase 1 habitat map of Wales, UK, using satellite sensor data. *ISPRS Journal of
644 Photogrammetry and Remote Sensing* 66:81–102.
- 645 Matasci, G., T. Hermosilla, M. A. Wulder, J. C. White, N. C. Coops, G. W. Hobart, D. K. Bolton, P.
646 Tompalski, and C. W. Bater. 2018a. Three decades of forest structural dynamics over Canada's forested
647 ecosystems using Landsat time-series and lidar plots. *Remote Sensing of Environment* 216:697–714.
- 648 Matasci, G., T. Hermosilla, M. A. Wulder, J. C. White, N. C. Coops, G. W. Hobart, and H. S. J. Zald. 2018b.
649 Large-area mapping of Canadian boreal forest cover, height, biomass and other structural attributes

- 650 using Landsat composites and lidar plots. *Remote Sensing of Environment* 209:90–106.
- 651 Maxwell, S. L., V. Cazalis, N. Dudley, M. Hoffmann, A. S. L. Rodrigues, S. Stolton, P. Visconti, S. Woodley,
652 N. Kingston, E. Lewis, M. Maron, B. B. N. Strassburg, A. Wenger, H. D. Jonas, O. Venter, and J. E. M.
653 Watson. 2020. Area-based conservation in the twenty-first century. *Nature* 586:217–227.
- 654 McDermid, G. J., S. E. Franklin, and E. F. LeDrew. 2005. Remote sensing for large-area habitat mapping.
655 *Progress in Physical Geography: Earth and Environment* 29:449–474.
- 656 Meidinger, D. V., and J. Pojar, editors. 1991. *Ecosystems of British Columbia*. Research Branch, Ministry of
657 Forests, Victoria, B.C.
- 658 Myneni, R. B., J. Dong, C. J. Tucker, R. K. Kaufmann, P. E. Kauppi, J. Liski, L. Zhou, V. Alexeyev, and M. K.
659 Hughes. 2001. A large carbon sink in the woody biomass of northern forests. *Proceedings of the*
660 *National Academy of Sciences of the United States of America* 98:14784–14789.
- 661 Nagendra, H. 2001. Using remote sensing to assess biodiversity. *International Journal of Remote Sensing*
662 22:2377–2400.
- 663 Nagendra, H. 2008. Do parks work? Impact of protected areas on land cover clearing. *Ambio* 37:330–337.
- 664 Nagendra, H., R. Lucas, J. P. Honrado, R. H. G. Jongman, C. Tarantino, M. Adamo, and P. Mairotta. 2013.
665 Remote sensing for conservation monitoring: Assessing protected areas, habitat extent, habitat
666 condition, species diversity, and threats. *Ecological Indicators* 33:45–59.
- 667 Nagendra, H., D. Rocchini, R. Ghate, B. Sharma, and S. Pareeth. 2010. Assessing Plant Diversity in a Dry
668 Tropical Forest: Comparing the Utility of Landsat and Ikonos Satellite Images. *Remote Sensing*
669 2:478–496.
- 670 Neuenschwander, A., E. Guenther, J. C. White, L. Duncanson, and P. Montesano. 2020. Validation of
671 ICESat-2 terrain and canopy heights in boreal forests. *Remote Sensing of Environment* 251:112110.
- 672 Noss, R. F. 1999. Assessing and monitoring forest biodiversity: A suggested framework and indicators.
673 *Forest Ecology and Management* 115:135–146.
- 674 Olthof, I., D. Pouliot, R. Fraser, A. Clouston, S. Wang, W. Chen, J. Oraziotti, J. Poitevin, D. McLennan, J.
675 Kerr, and M. Sawada. 2006. Using Satellite Remote Sensing to Assess and Monitor Ecosystem Integrity
676 and Climate Change in Canada's National Parks. 2006 IEEE International Symposium on Geoscience
677 and Remote Sensing. IEEE.

- 678 Paillet, Y., L. Berges, J. Hjalten, P. Odor, C. Avon, M. Bernhardt-Roemermann, R.-J. Bijlsma, L. De Bruyn,
679 M. Fuhr, U. Grandin, R. Kanka, L. Lundin, S. Luque, T. Magura, S. Matesanz, I. Meszaros, M.-. Teresa
680 Sebastia, W. Schmidt, T. Standovar, B. Tothmeresz, A. Uotila, F. Valladares, K. Vellak, and R. Virtanen.
681 2010. Biodiversity differences between managed and unmanaged forests: Meta-analysis of species
682 richness in europe. *Conservation Biology* 24:101–112.
- 683 Parks Canada. 2019. Ecological Integrity. <https://www.pc.gc.ca/en/nature/science/conservation/ie-ei>.
- 684 Parmenter, A. W., A. Hansen, R. E. Kennedy, W. Cohen, U. Langner, R. Lawrence, B. Maxwell, A. Gallant,
685 and R. Aspinall. 2003. Land use and land cover change in the Greater Yellowstone Ecosystem: 1975.
686 *Ecological Applications* 13:687–703.
- 687 Parrish, J. D., D. P. Braun, and R. S. Unnasch. 2003. Are We Conserving What We Say We Are? Measuring
688 Ecological Integrity within Protected Areas. *BioScience* 53:851.
- 689 Pereira, H. M., S. Ferrier, M. Walters, G. N. Geller, R. H. G. Jongman, R. J. Scholes, M. W. Bruford, N.
690 Brummitt, S. H. M. Butchart, A. C. Cardoso, N. C. Coops, E. Dulloo, D. P. Faith, J. Freyhof, R. D.
691 Gregory, C. Heip, R. Hoft, G. Hurttt, W. Jetz, D. S. Karp, M. A. McGeoch, D. Obura, Y. Onoda, N.
692 Pettorelli, B. Reyers, R. Sayre, J. P. W. Scharlemann, S. N. Stuart, E. Turak, M. Walpole, and M.
693 Wegmann. 2013. Essential Biodiversity Variables. *Science* 339:277–278.
- 694 Pojar, J., K. Klinka, and D. V. Meidinger. 1987. Biogeoclimatic ecosystem classification in British
695 Columbia. *Forest Ecology and Management* 22:119–154.
- 696 Potapov, P., M. C. Hansen, I. Kommareddy, A. Kommareddy, S. Turubanova, A. Pickens, B. Adusei, A.
697 Tyukavina, and Q. Ying. 2020. Landsat Analysis Ready Data for Global Land Cover and Land Cover
698 Change Mapping. *Remote Sensing* 12:426.
- 699 Potapov, P., X. Li, A. Hernandez-Serna, A. Tyukavina, M. C. Hansen, A. Kommareddy, A. Pickens, S.
700 Turubanova, H. Tang, C. E. Silva, J. Armston, R. Dubayah, J. B. Blair, and M. Hofton. 2021. Mapping
701 global forest canopy height through integration of GEDI and Landsat data. *Remote Sensing of
702 Environment* 253:112165.
- 703 Pôcas, I., M. Cunha, and L. S. Pereira. 2011. Remote sensing based indicators of changes in a mountain
704 rural landscape of Northeast Portugal. *Applied Geography* 31:871–880.

- 705 R Core Team. 2021. R: A language and environment for statistical computing. R Foundation for Statistical
706 Computing, Vienna, Austria.
- 707 Ribas, L. G. dos S., R. L. Pressey, R. Loyola, and L. M. Bini. 2020. A global comparative analysis of impact
708 evaluation methods in estimating the effectiveness of protected areas. *Biological Conservation*
709 246:108595.
- 710 Rocchini, D., N. Balkenhol, G. A. Carter, G. M. Foody, T. W. Gillespie, K. S. He, S. Kark, N. Levin, K. Lucas,
711 M. Luoto, H. Nagendra, J. Oldeland, C. Ricotta, J. Southworth, and M. Neteler. 2010. Remotely sensed
712 spectral heterogeneity as a proxy of species diversity: Recent advances and open challenges. *Ecological
713 Informatics* 5:318–329.
- 714 Running, S. W., R. R. Nemani, F. A. Heinsch, M. S. Zhao, M. Reeves, and H. Hashimoto. 2004. A
715 continuous satellite-derived measure of global terrestrial primary production. *Bioscience* 54:547–560.
- 716 Skidmore, A. K., N. C. Coops, E. Neinavaz, A. Ali, M. E. Schaepman, M. Paganini, W. D. Kissling, P.
717 Vihervaara, R. Darvishzadeh, H. Feilhauer, M. Fernandez, N. Fernández, N. Gorelick, I. Geijzendorffer,
718 U. Heiden, M. Heurich, D. Hoborn, S. Holzwarth, F. E. Muller-Karger, R. Van De Kerchove, A. Lausch,
719 P. J. Leitão, M. C. Lock, C. A. Mücher, B. O'Connor, D. Rocchini, W. Turner, J. K. Vis, T. Wang, M.
720 Wegmann, and V. Wingate. 2021. Priority list of biodiversity metrics to observe from space. *Nature
721 Ecology & Evolution*.
- 722 Soverel, N. O., N. C. Coops, J. C. White, and M. A. Wulder. 2010. Characterizing the forest fragmentation
723 of Canada's national parks. *Environmental Monitoring and Assessment* 164:481–499.
- 724 Tachikawa, T., M. Kaku, A. Iwasaki, D. B. Gesch, M. J. Oimoen, Z. Zhang, J. J. Danielson, T. Krieger, B.
725 Curtis, J. Haase, M. Abrams, and C. Carabajal. 2011. ASTER global digital elevation model version 2 -
726 summary of validation results. Page 27.
- 727 Teucher, A., S. Hazlitt, and S. Albers. 2021. Bcmaps: Map layers and spatial utilities for british columbia.
- 728 Turner, W., C. Rondinini, N. Pettorelli, B. Mora, A. K. Leidner, Z. Szantoi, G. Buchanan, S. Dech, J. Dwyer,
729 M. Herold, L. P. Koh, P. Leimgruber, H. Taubenboeck, M. Wegmann, M. Wikelski, and C. Woodcock.
730 2015. Free and open-access satellite data are key to biodiversity conservation. *Biological Conservation*
731 182:173–176.

- 732 Turner, W., S. Spector, N. Gardiner, M. Fladeland, E. Sterling, and M. Steininger. 2003. Remote sensing for
733 biodiversity science and conservation. *Trends in Ecology & Evolution* 18:306–314.
- 734 Venter, O., R. A. Fuller, D. B. Segar, J. Carwardine, T. Brooks, S. H. M. Butchart, M. Di Marco, T. Iwamura,
735 L. Joseph, D. O’Grady, H. P. Possingham, C. Rondinini, R. J. Smith, M. Venter, and J. E. M. Watson.
736 2014. Targeting Global Protected Area Expansion for Imperiled Biodiversity. *PLoS Biology*
737 12:e1001891.
- 738 Venter, O., A. Magrach, N. Outram, C. J. Klein, H. P. Possingham, M. Di Marco, and J. E. M. Watson. 2018.
739 Bias in protected-area location and its effects on long-term aspirations of biodiversity conventions.
740 *Conservation Biology: The Journal of the Society for Conservation Biology* 32:127–134.
- 741 Wang, T., P. Smets, C. Chourmouzis, S. N. Aitken, and D. Kolotelo. 2020. Conservation status of native tree
742 species in British Columbia. *Global Ecology and Conservation* 24:e01362.
- 743 Watson, J. E. M., N. Dudley, D. B. Segar, and M. Hockings. 2014. The performance and potential of
744 protected areas. *Nature* 515:67–73.
- 745 White, Joanne. C., M. A. Wulder, G. W. Hobart, J. E. Luther, T. Hermosilla, P. Griffiths, N. C. Coops, R. J.
746 Hall, P. Hostert, A. Dyk, and L. Guindon. 2014. Pixel-Based Image Compositing for Large-Area Dense
747 Time Series Applications and Science. *Canadian Journal of Remote Sensing* 40:192–212.
- 748 Wiens, J., R. Sutter, M. Anderson, J. Blanchard, A. Barnett, N. aguilar-amuchastegui, C. Avery, and S.
749 Laine. 2009. Selecting and conserving lands for biodiversity: The role of remote sensing. *Remote
750 Sensing of Environment* 113:1370–1381.
- 751 Woodley, S. 1993. Monitoring and Measuring Ecosystem Integrity in Canadian National Parks. *Ecological
752 Integrity and the Management of Ecosystems*. Taylor & Francis.
- 753 Wulder, M. A., J. G. Masek, W. B. Cohen, T. R. Loveland, and C. E. Woodcock. 2012a. Opening the archive:
754 How free data has enabled the science and monitoring promise of Landsat. *Remote Sensing of
755 Environment* 122:2–10.
- 756 Wulder, M. A., J. C. White, T. R. Loveland, C. E. Woodcock, A. S. Belward, W. B. Cohen, E. A. Fosnight, J.
757 Shaw, J. G. Masek, and D. P. Roy. 2016. The global Landsat archive: Status, consolidation, and
758 direction. *Remote Sensing of Environment* 185:271–283.

- 759 Wulder, M. A., J. C. White, R. F. Nelson, E. Næsset, H. O. Ørka, N. C. Coops, T. Hilker, C. W. Bater, and T.
760 Gobakken. 2012b. Lidar sampling for large-area forest characterization: A review. *Remote Sensing of*
761 *Environment* 121:196–209.
- 762 Zanaga, D., R. Van De Kerchove, W. De Keersmaecker, N. Souverijns, C. Brockmann, R. Quast, J. Wevers,
763 A. Grosu, A. Paccini, S. Vergnaud, O. Cartus, M. Santoro, S. Fritz, I. Georgieva, M. Lesiv, S. Carter, M.
764 Herold, L. Li, N.-E. Tsendbazar, F. Ramoino, and O. Arino. 2021. *ESA WorldCover 10 m 2020 v100*.
- 765 Zhang, X. Y., M. A. Friedl, C. B. Schaaf, A. H. Strahler, J. C. F. Hodges, F. Gao, B. C. Reed, and A. Huete.
766 2003. Monitoring vegetation phenology using MODIS. *Remote Sensing of Environment* 84:471–475.

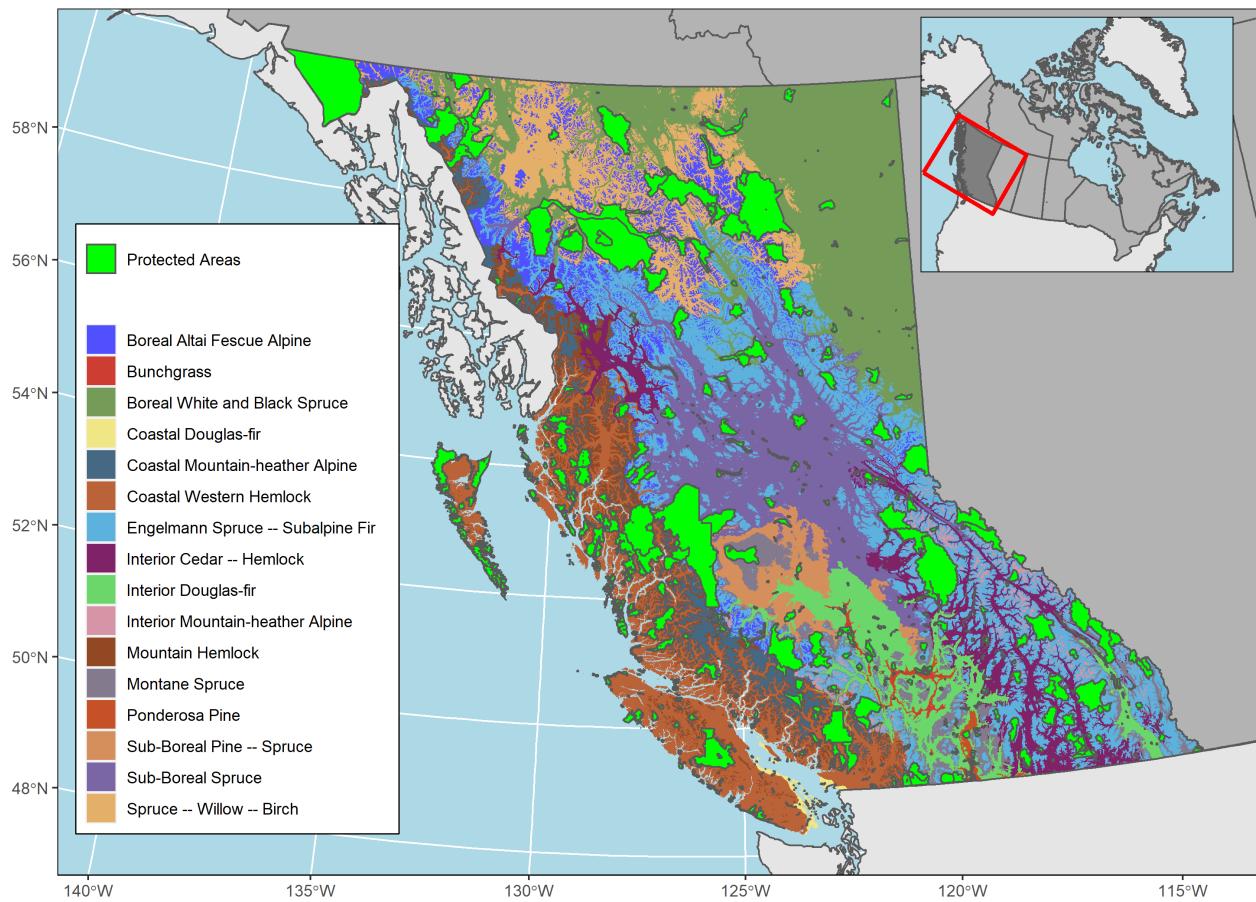


Figure 1: Terrestrial British Columbia including BEC zones and the location of PA selected in this study.

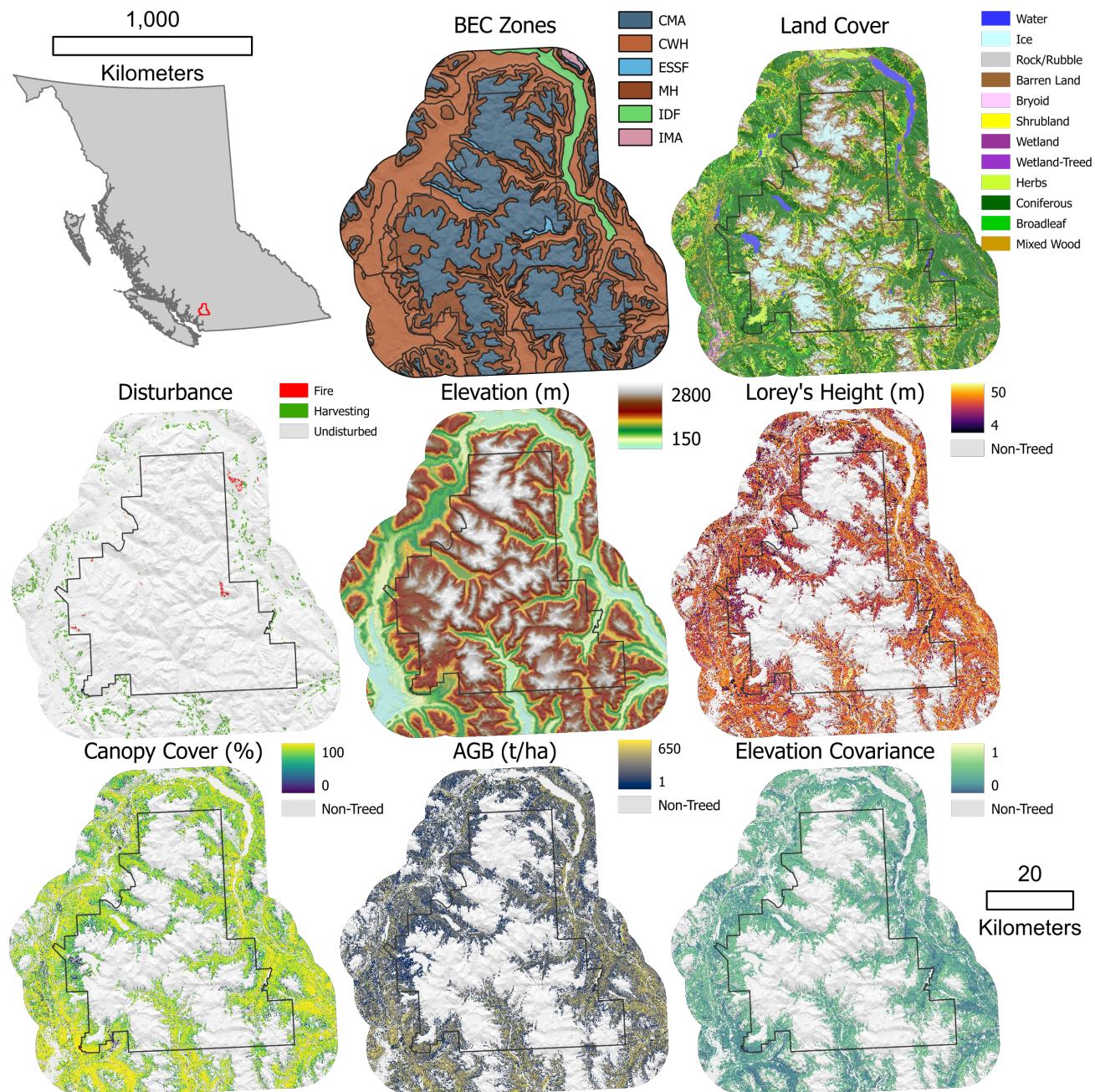


Figure 2: Visualizations for all layers included in the analysis for Garibaldi Park and surrounding region (red outline) in BC for 2015.

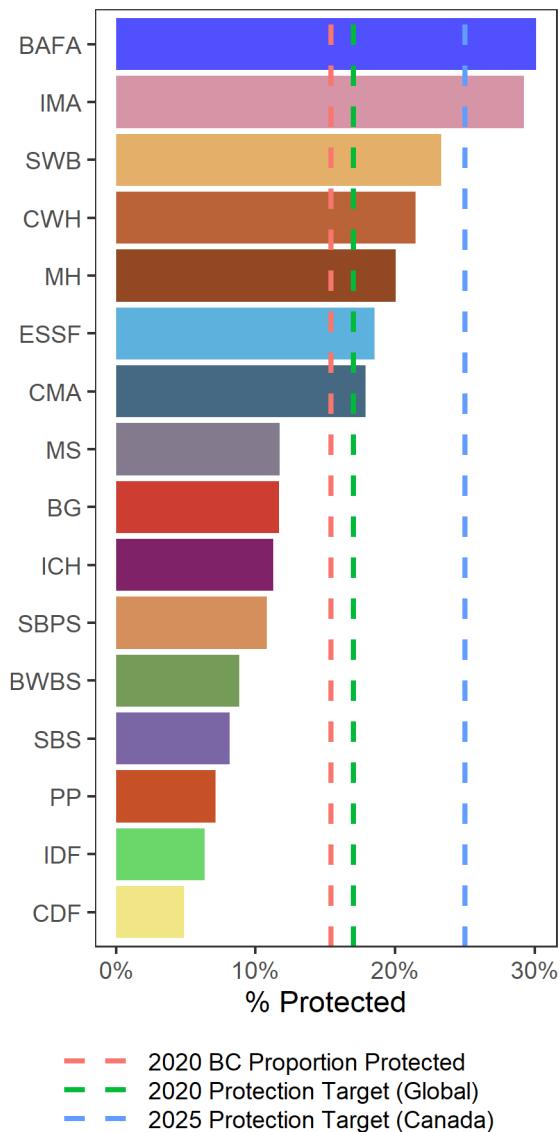


Figure 3: Areal proportion of biogeoclimatic ecosystem classification (BEC) zones protected in British Columbia (See Table 1 for full BEC zone names).

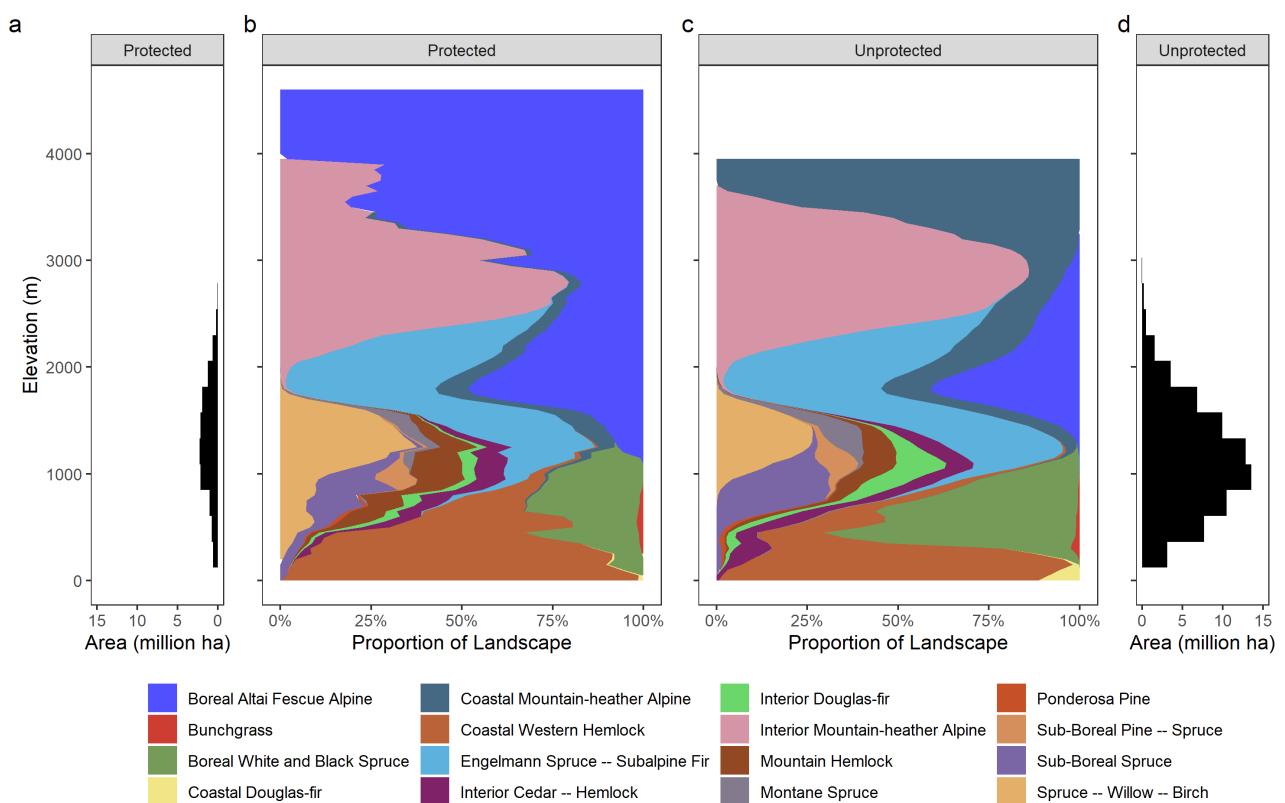


Figure 4: Histogram of area protected in British Columbia by Elevation (a). Proportion of Biogeoclimatic Ecosystem Classification (BEC) zone by elevation for both protected areas (b), and unprotected areas (c). Histogram of area unprotected in British Columbia by Elevation (d).

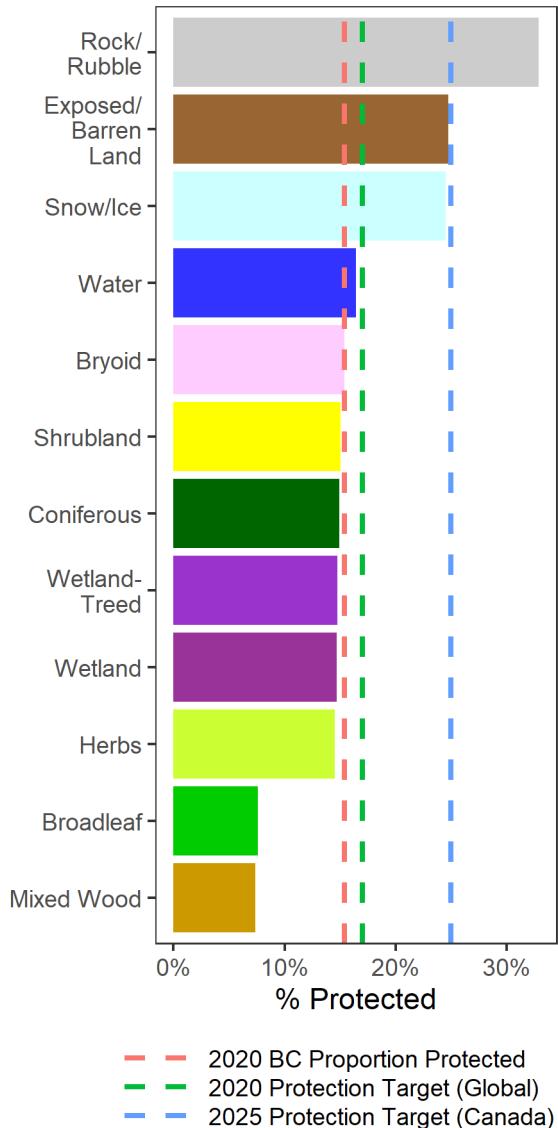


Figure 5: Areal proportion of land cover classes protected in BC.

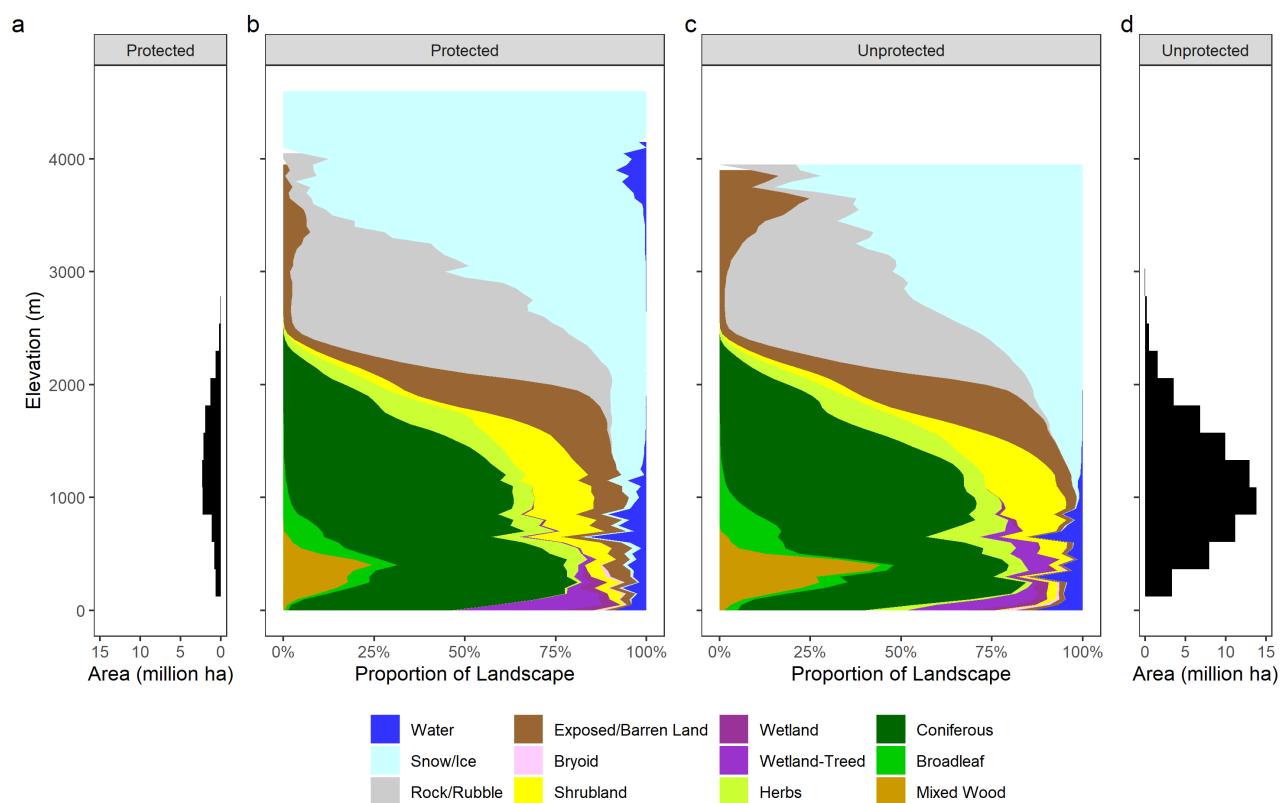


Figure 6: Histogram of area protected in British Columbia by Elevation (a) Proportion of land cover by elevation for both protected areas (b), and unprotected areas (c). Histogram of area unprotected in British Columbia by Elevation (d).

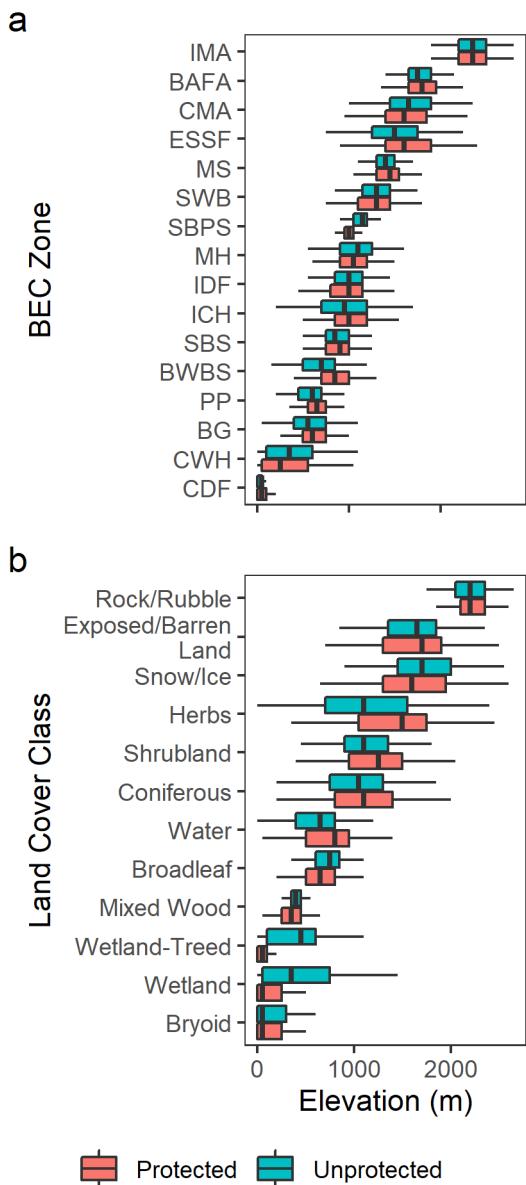


Figure 7: Elevation boxplots for BEC zones (a), and land cover classes (b). Whiskers indicate first quartile minus the interquartile range and third quartile to the interquartile range. Box and interior vertical line indicate first quartile, median, and third quartile, respectively.

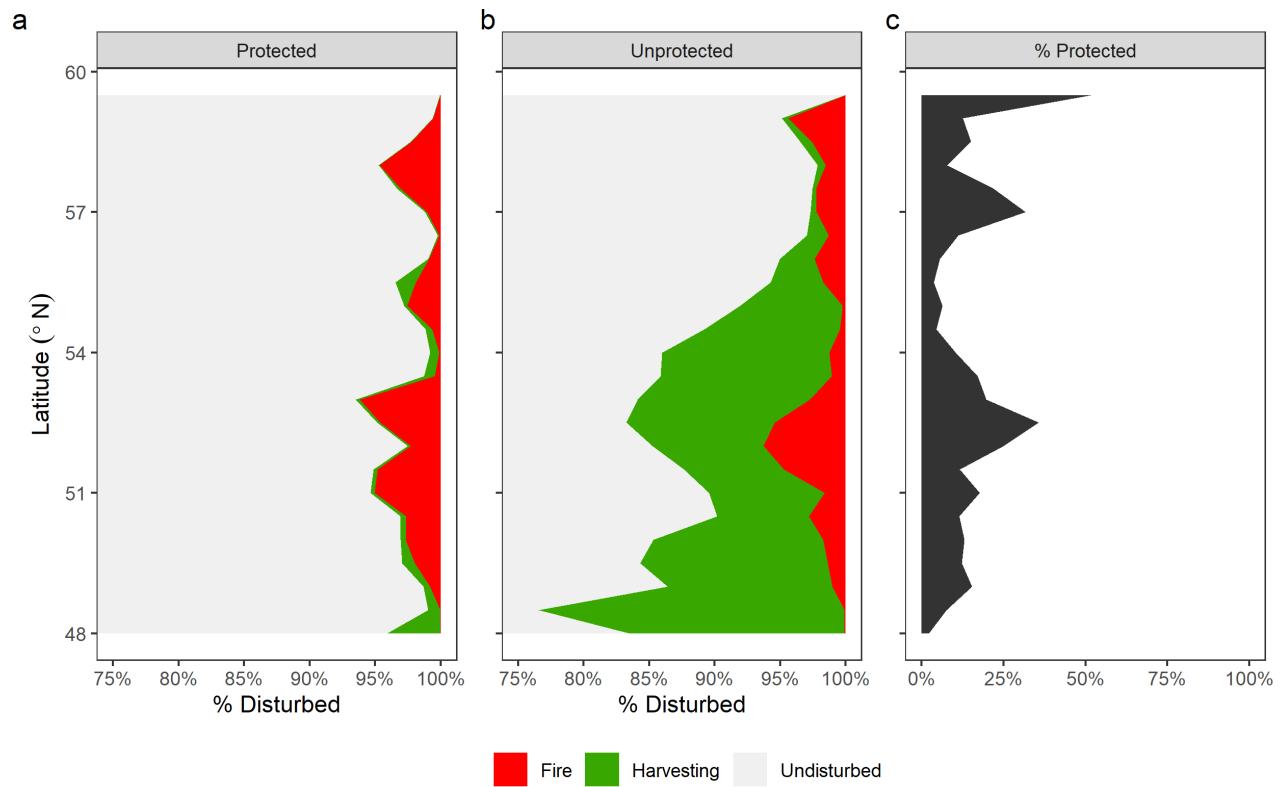


Figure 8: Proportion of area disturbed by latitude from 1984 to 2019 in protected areas (a), and unprotected areas (b). Proportion of terrestrial area that is protected at each latitude (c).

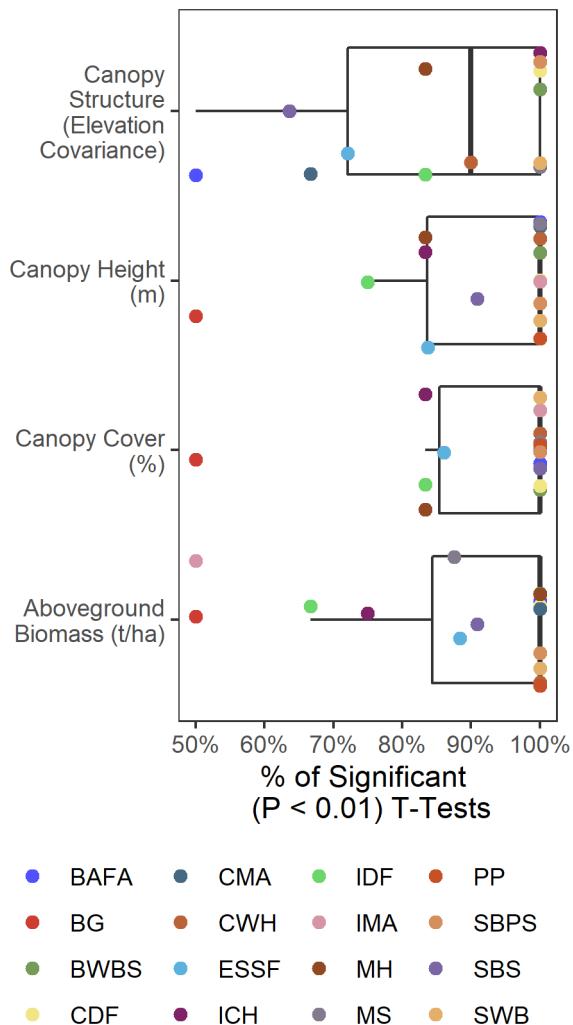


Figure 9: Boxplot of proportion of ecosystem subzone which have significant p-values from a two-tailed t-test with the Bonferroni correction ($n = 496$) applied at a significance level of 0.05. Boxplot vertical lines indicate the first quartile, the median, and the third quartile. The whisker extends from the first quartile to the smallest value no further than $1.5 * \text{interquartile range}$ from the first quartile.

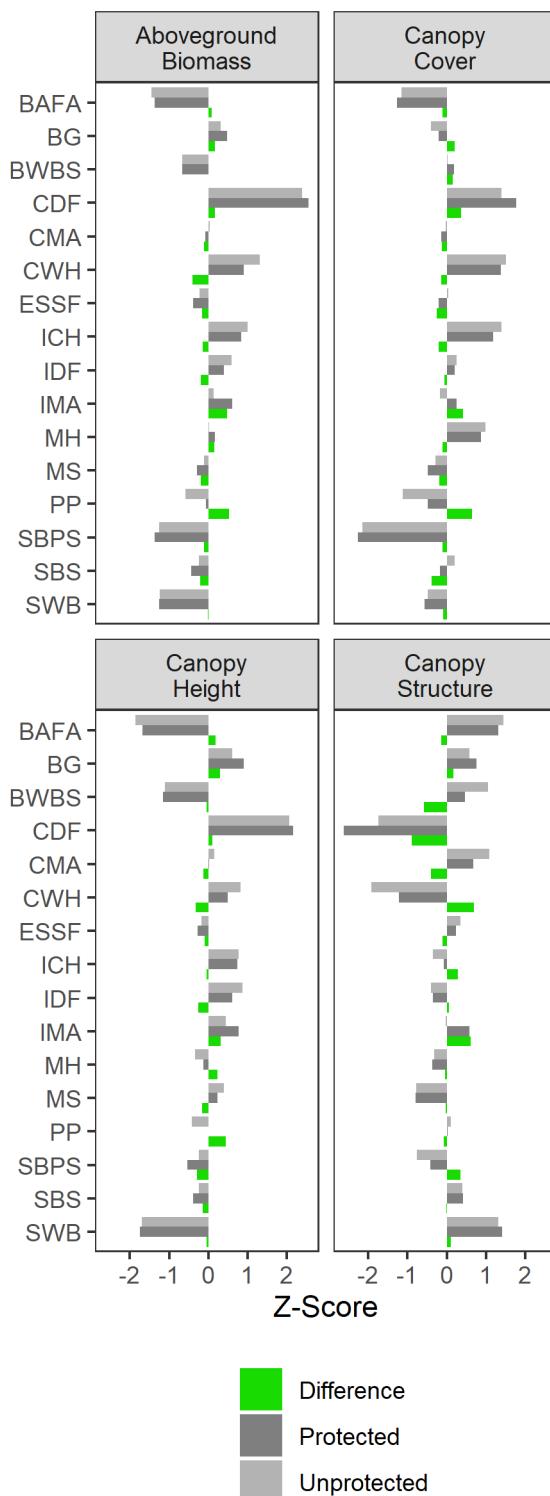


Figure 10: Z-Scores of forest structural attributes in PA, UA, and their differences across BEC zones in BC.