



CONTRIBUTED PAPER

Relative effects of recreational activities on a temperate terrestrial wildlife assemblage

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Abstract

Outdoor recreation is one of the fastest growing economic sectors in the world and provides many benefits to people. Assessing possible negative impacts of recreation is nevertheless important for sustainable management. Here, we used camera traps to assess relative effects of various recreational activities—as compared to each other and to environmental conditions—on a terrestrial wildlife assemblage in British Columbia, Canada. Across 13 species, only two negative associations between recreational activities and wildlife detections were observed at weekly scales: mountain biking on moose and grizzly bears. However, finer-scale analysis showed that all species avoided humans on trails, with avoidance strongest for mountain biking and motorized vehicles. Our results imply that environmental factors generally shaped broad-scale patterns of wildlife use, but highlight that recreational activities also have detectable impacts. These impacts can be monitored using the same camera-trapping techniques that are commonly used to monitor wildlife assemblages.

KEYWORDS

Bayesian models, camera traps, grizzly bear, hiking, horseback riding, logging, motorized vehicles, mountain biking, protected areas, tourism

1 | INTRODUCTION

Human transformation of the natural world is a defining characteristic of the Anthropocene (Johnson et al., 2017). This large-scale transformation is most visible in conversion of natural ecosystems, such as forests, to anthropogenic systems, such as agriculture (Hansen et al., 2013). The impacts of these conversions on biodiversity are also increasingly well-studied (Dirzo et al., 2014; Newbold et al., 2015). However, another type of anthropogenic impact is less apparent but potentially of similar importance: the increasing recreation of humans in formerly pristine or unvisited areas of the Earth (Balmford, Green, Anderson, et al., 2015). In ecosystem services terminology

outdoor recreation is considered a “cultural service” (Willis, 2015), and the benefits of such nature-based tourism for incentivizing conservation of natural ecosystems, both within and outside of formal protected areas (PAs), has been emphasized by many authors (e.g., Naidoo, Fisher, Manica, & Balmford, 2016; Willis, 2015).

The flip side to these positive aspects of nature-based tourism and outdoor recreation is the potential for disruption of natural systems, disturbance to wildlife, and degradation of biodiversity (Buckley, 2004; Larson, Reed, Merenlender, & Crooks, 2019; Reed & Merenlender, 2008). Monitoring whether these impacts are indeed occurring is challenging. On the biological side, changes to biodiversity in response to recreation can be more

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subtle than biodiversity losses from land-use change. From the human standpoint, visitor numbers to PAs and other areas can be surprisingly difficult to monitor (Buckley, 2009). Furthermore, disentangling the relative impacts of changes in outdoor recreation as compared to changing environmental conditions requires comparable data on both to be convincingly linked to changes in biodiversity (Coppes et al., 2018). Finally, different species may respond in different ways to changes in both recreation and the broader environment, requiring a multi-species approach to evaluating impacts (Fisher & Burton, 2018; Larson, Reed, Merenlender, & Crooks, 2016).

Motion-triggered cameras (“camera traps”) are increasingly being used to monitor wildlife (Steenweg et al., 2017; Wearn & Glover-Kapfer, 2019). Advances in camera technology (Glover-Kapfer, Soto-Navarro, & Wearn, 2019), decreasing costs of production, and the development of artificial intelligence to automate image identification (Norouzzadeh et al., 2018) are likely to further increase their competitive advantage as a wildlife monitoring tool. With appropriate design considerations, cameras can also noninvasively detect and record human activities (Ladle, Steenweg, Shepherd, & Boyce, 2018; Miller, Leung, & Kays, 2017). To-date however, and notwithstanding legitimate privacy concerns (Sandbrook, Luque-Lora, & Adams, 2018), there has been relatively limited application of camera traps to explicitly monitor anthropogenic landscape use in the same standardized way as wildlife.

Here, we test whether an integrated recreation-environment statistical analysis can disentangle the

relative impacts of each on a temperate wildlife assemblage surveyed via camera traps. We deployed cameras in a grid-based system across roads and trails in a landscape that is also subject to heavy industrial use. We used Bayesian regression models to quantify the relative impacts of recreational usage versus environmental conditions on wildlife at the weekly scale, while also assessing recreation impacts at finer temporal scales. We suggest this approach could prove fruitful for monitoring the relative impacts of nature-based tourism in other contexts.

2 | METHODS

2.1 | Study area

We conducted our study in and around the South Chilcotin Mountains provincial park (hereafter, “SCM”) in southwestern British Columbia, Canada (Figure 1). The provincial park was established in 2010 and covers 568 km² of forested mountains and alpine terrain. The SCM is notable for its diversity of large wildlife species, including predators such as grizzly bear (*Ursus arctos horribilis*), wolverine (*Gulo gulo*), and fisher (*Pekania pennanti*), as well as large ungulates such as moose (*Alces alces*), mule deer (*Odocoileus hemionus*), and mountain goat *Oreamnos americanus* (BC Parks, 2019; McLellan et al., 2019). In addition to the PA, there is a mix of logging, mining, ranching, tourism, and private land holdings in the region. The main drivers of anthropogenic

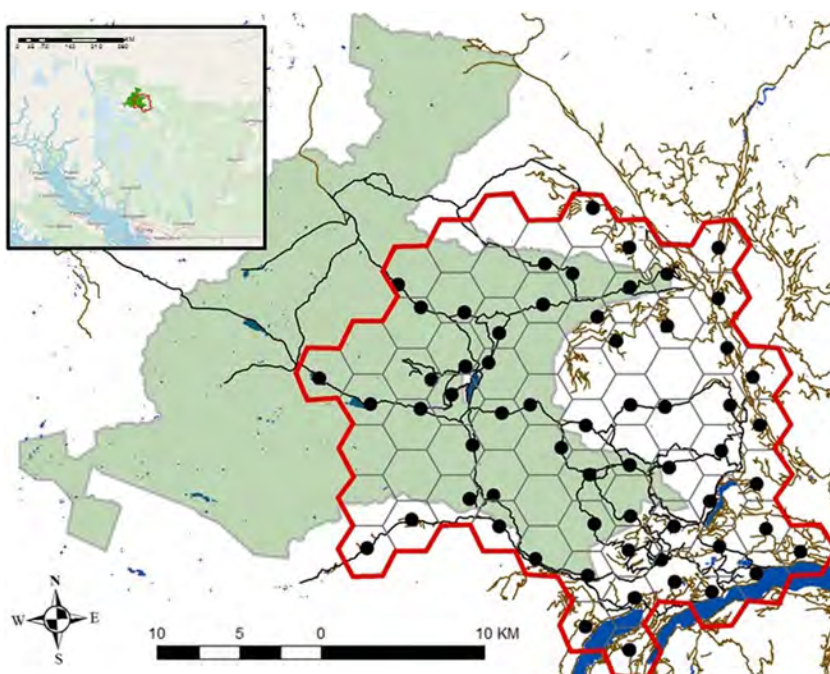


FIGURE 1 Study area (red outline) in and around the South Chilcotin Mountains Provincial Park (green polygon) in southwestern British Columbia, Canada. Black circles indicate camera trap sites. Brown lines = logging roads or cutlines. Black lines = multiuse recreational trails. Blue polygons = water bodies. Hexagonal grid used to deploy cameras at sites (3-km distance between grid cell centers) is also shown

change in the SCM appear to be (1) recreation, particularly increased mountain biking and off-road vehicle use (BC Parks, 2019), and (2) an increasing level of logging (Hermosilla, Wulder, White, Coops, & Hobart, 2015). Other aspects of the expanding human footprint include residential use, hydroelectric development, and mining. Taken together, the human presence in the study area is growing tremendously, and at present land-use managers lack data on how these potential stressors may be affecting wildlife (BC Parks, 2019). Studies in other areas have shown negative impacts of recreation on some species of management concern in the SCM (e.g., wolverine, Heinemeyer et al., 2019; grizzly bear, Ladle et al., 2019). However, such knowledge is lacking for most species, particularly within an intact assemblage facing rapidly growing pressure. Furthermore, recreational impacts may be context-specific as they have been shown to differ for the same species in different areas (e.g., predator species such as coyote have been shown to avoid recreational activity in some areas but not in others; Parsons et al., 2016; Reed & Merenlender, 2008).

2.2 | Camera trap survey

We used camera traps to sample trail use by wildlife and people. To determine sampling locations, we overlaid a hexagonal grid across $\sim 550 \text{ km}^2$ of the SCM, encompassing a significant fraction of the park as well as public and residential lands to the south and east (Figure 1). Centers of grid cells were spaced 3 km apart, which we adjudged a compromise distance allowing for effective sampling of the range of medium- and large-bodied wildlife species using the area (Burton et al., 2015; Kays, Kranstauber, Jansen, et al., 2009). Using provincial databases on the locations of logging roads and other linear infrastructure (<https://www2.gov.bc.ca/gov/content/data/geographic-data-services/topographic-data/roads>), as well as maps of the multiuse trail network (<https://www.trailforks.com/region/south-chilcotin-mountains-provincial-park/>), we placed a motion-triggered trail camera (Browning StrikeForce Pro HD; www.browning.com) at the location on the trail or road that was closest to the grid cell center (Rich, Miller, Robinson, McNutt, & Kelly, 2016; this location hereafter referred to as the “site”). Budgetary constraints restricted the total number of cameras to $n = 61$, and some cells within the study area did not have any logging roads or multiuse trails within them. As a result, 11 of 71 total grid cells within the study area did not have a camera.

Cameras were positioned on a tree about a meter above trail level, a height that allowed a wide range of terrestrial wildlife to be detected. Trail width, distance to trail, and camera height were recorded at each site for

inclusion in subsequent statistical modeling. Most cameras were deployed in late May or early June (when high elevation terrain started to become snow-free) and were active through mid- to end-September, when SIM cards were collected.

We identified to species all pictures containing wildlife observations, including mammals, birds, and amphibians. We counted the number of individuals of each species on each picture and classified them using the Camelot software package (Hendry & Mann, 2017). We also recorded the number and type of human activities detected by cameras and assigned each to one of four categories: hikers, horseback riders, mountain bikers, or motorized vehicles.

2.3 | Statistical analysis

We focused our inference on the 13 species of wildlife with at least 30 independent detections (Table 1), using a minimum threshold between detections of 20 min for independence (Burton et al., 2015). For each week that a camera was operational at a site, we assessed whether each of the 13 species was detected (1 = species detected during site-week, 0 = not detected). We chose the week as our temporal unit of analysis because it reduced the number of observations where the count of species detections was zero (i.e., as opposed to a daily time period), while at the same time providing a large enough sample size ($n = 939$ site-weeks) to model the effects of a substantial number of spatiotemporal predictor variables. A week also falls within the range of repeat sampling occasion lengths used in other camera trap surveys, and is coarse enough to minimize spatiotemporal autocorrelation as compared to, for example, using a daily time period (Burton et al., 2015; Steenweg et al., 2016).

For each species, we modeled the probability of occurrence during a site-week with a Bayesian hierarchical regression model, assuming a binomial response distribution and with site included as a random effect to account for nonindependence of sampling weeks within sites. Independent variables included a set of recreational and environmental variables expected to influence whether a species used (or occurred) at a site. Recreational variables were the number of detections, by site-week, for each of hiking, horseback riding, mountain biking, and motorized vehicles, using the same 20-min independence threshold as for wildlife. Environmental variables included a subset characterized as a single point value assessed at the site, and a subset summarized across the grid cell containing the site, in order to provide both site and landscape-level environmental predictors (detailed description of variables and how they were summarized across a grid cell in Table 2). To account for

TABLE 1 Summary of detections for 13 wildlife species across 61 camera traps in the South Chilcotin mountains used in an assessment of the relative impacts of environmental factors and human recreation on weekly wildlife occurrences

Species	Common name	# sites	# detections ^a	Detections/100 camera-days	Average group size
Wildlife					
<i>Odocoileus hemionus</i>	Mule deer	60	4,070	64.8	1.23
<i>Tamiasciurus hudsonicus</i>	Red squirrel	49	1,033	16.4	1.00
<i>Lepus americanus</i>	Snowshoe hare	60	885	14.1	1.00
<i>Canis lupus</i>	Wolf	36	621	9.9	1.31
<i>Ursus americanus</i>	Black bear	49	340	5.4	1.03
<i>Ursus arctos</i>	Grizzly bear	48	206	3.3	1.07
<i>Canis latrans</i>	Coyote	41	203	3.2	1.10
<i>Lynx canadensis</i>	Canada lynx	41	198	3.2	1.00
<i>Falci pennis canadensis</i>	Spruce grouse	19	135	2.1	1.35
<i>Dendragapus obscurus</i>	Dusky grouse	30	89	1.4	1.22
<i>Martes americana</i>	American marten	13	82	1.3	1.03
<i>Alces</i>	Moose	20	55	0.9	1.06
<i>Puma concolor</i>	Cougar	21	32	0.5	1.03
Humans					
–	Mountain bikers	39	10,081	160.4	3.17
–	Hikers	43	2,522	40.1	2.29
–	Horseback riders	30	1,382	22.0	2.65
–	Motorized vehicles	27	2,490	39.6	1.30

^aIndependent detections, with a 20-min threshold used to define independence.

TABLE 2 Descriptive statistics of environmental, recreation and sampling variables used in hierarchical Bayesian regression models of weekly detections of 13 wildlife species across 61 camera trap sites in the South Chilcotin mountains

Variable	Description	Mean	Min	Max	Category
Road length ^a	Length of roads in cell (km)	36.5	0	174	Environment (cell)
Harvested area ^b	% of grid cell harvested for timber	10.6	0	66	Environment (cell)
Biomass ^b	Average biomass of forest stands in cell (mg/ha)	97.5	7.6	159	Environment (cell)
Open ^b	% grid cell in nonforested habitat	2.3	0	25	Environment (cell)
Elevation ^c	Elevation at site (m)	1,411	796	2,141	Environment (site)
Forested ^c	Whether site is in forest (1/0)	0.69	0	1	Environment (site)
Trail width ^c	Width of trail that site is at (m)	3	0.3	11	Detectability
Distance to trail ^c	Distance to center of trail from camera (m)	4.8	1	9	Detectability
Camera height ^c	Height of camera from bottom of tree (m)	1	0.1	2	Detectability
Mountain bikers	# mountain bikers detected in a week at site	10	0	273	Human trail-use
Hikers	# hikers detected in a week at site	2.6	0	42	Human trail-use
Horseback riders	# horseback riders detected in a week at site	1.4	0	34	Human trail-use
Motorized vehicles	# motorized vehicles detected in a week at site	2.6	0	100	Human trail-use
Lag detection	Species detected at site in previous week (1/0)	– ^d	0	1	Temporal autocorrelation

^aSource: <https://www2.gov.bc.ca/gov/content/data/geographic-data-services/topographic-data/roads>

^bSource: <https://www2.gov.bc.ca/gov/content/industry/forestry/managing-our-forest-resources/forest-inventory/data-management-and-access>

^cField measurement.

^dPrevious week's detections specific to each species, therefore mean across sites not calculated.

possible variation in detectability across sites due to camera positioning, we also included camera height, distance from trail, and trail width as additional predictor variables (Table 2). Finally, we accounted for potential temporal autocorrelation by including a covariate for whether a given species was detected the previous week. All independent variables were standardized by subtracting the mean and dividing by 2 SDs (Gelman, 2008) to allow a direct assessment of the relative magnitude of variable impacts. We assessed possible collinearity among all predictor variables and did not find any correlation stronger than $r = 0.42$, therefore retained all variables in subsequent models. We considered any variable whose 95% Bayesian credible interval did not overlap with 0 to have a statistically significant impact on the probability of species occurrence during a site-week.

Note that the approach above differs from a formal occupancy modeling approach (MacKenzie et al., 2002). Occupancy models assume closure of a site to changes in occupancy status, and attribute variation in detections across sampling occasions (weeks in this case) to “detectability.” However, this is not detectability in the strict sense (i.e., animal missed when present) but rather represents the movement (i.e., local habitat use) process that we wished to model. We therefore felt an occupancy approach was not appropriate conceptually (e.g., Efford & Dawson, 2012; Neilson, Avgar, Burton, Broadley, & Boutin, 2018). Furthermore, a practical limitation is that an occupancy approach would have severely limited the number of independent variables that could have been tested in our models, due to the much smaller sample size that collapsing data to the site, rather than site-week, would have entailed. Occupancy models are also known to have convergence problems for rare and common species, that is, at very low or high estimated occupancies (Neilson et al., 2018). In our view it was preferable to include variables likely to affect both detection and occurrence in the same model, with the associated increase in statistical power this allowed, rather than pursue an occupancy approach. Our detection variables also reflect the true problem of imperfect detection (i.e., missing an animal that in fact passed the camera), which is not what is estimated by an occupancy model in this context, as explained above.

We conducted a second analysis to assess whether recreational activities caused finer-scale temporal displacement of wildlife from trails. For each human detection at a site, we recorded the time until the next detection of each of our focal species, as well as the most recent detection of that species prior to the human detection (Figure S1). The ratio between these times has been termed an “Avoidance-Attraction Ratio” (AAR), with values greater than one indicative of avoidance and

values less than one indicative of attraction (Parsons et al., 2016). We used a hierarchical Bayesian linear regression model, with species as random effects, to test for general recreation impacts across all species. We expected that lag times for each species would be different among recreation types, with hikers and horseback riders (slow-moving and relatively quiet activities) having the lowest AAR, motorized vehicles (fast, loud activity) having the highest, and mountain bikers intermediate. We tested this hypothesis by assessing when Bayesian 95% credible intervals of the main effects of recreation type did not overlap with one another, and also examined species-specific responses via the random-effect estimates of the species coefficients for each recreational type.

Regression models for both analyses were estimated using package *brms* (Burkner, 2017) in the statistical computing software R (R Core Team, 2018).

3 | RESULTS

Our grid of 61 motion-triggered camera trap sites was active for a total of 6,285 nights, and produced a wide array of species detections: 27 mammal, 32 bird, and 1 amphibian species (Figure 2; Table S1). The most frequently-detected mammal was the mule deer, with 4,070 detections, or approximately 2 detections every 3 days (Table 1). The least-frequently-detected mammals were red fox (*Vulpes vulpes*; $n = 3$), river otter (*Lontra canadensis*; $n = 1$), and striped skunk (*Mephitis*; $n = 1$). Mule deer were detected at all but one site, making them the most widely distributed (along with snowshoe hare *Lepus americanus*) as well as abundant wildlife species (Table 1).

Despite the diversity and abundance of wildlife captured on cameras, human activities were detected much more frequently (Table 1). Mountain biking was the most commonly-recorded activity, with over twice as many detections (10,017) as mule deer. Hikers, motorized vehicles, and horseback riders all had higher numbers of detections than any wildlife species other than mule deer.

There was considerable variation in the number and identity of predictor variables that best explained use of trails by different species. Grizzly bears had the greatest number of statistically significant variables (7), while wolves (*Canis lupus*) were the only species where no explanatory variables showed strong evidence of impacting probability of site-week use (Figure 3, Table S2). Human activity did not have particularly strong or consistent negative effects; human activity variables accounted for 6 of the 30 statistically-significant variables, and only two of those, mountain biking for moose and grizzly bears, were negatively associated with



FIGURE 2 Example camera trap photographs of wildlife and human activities in and around the South Chilcotin Mountains Provincial Park in southwestern British Columbia, Canada. Clockwise from upper left: grizzly bear; moose; hikers; mountain bikers

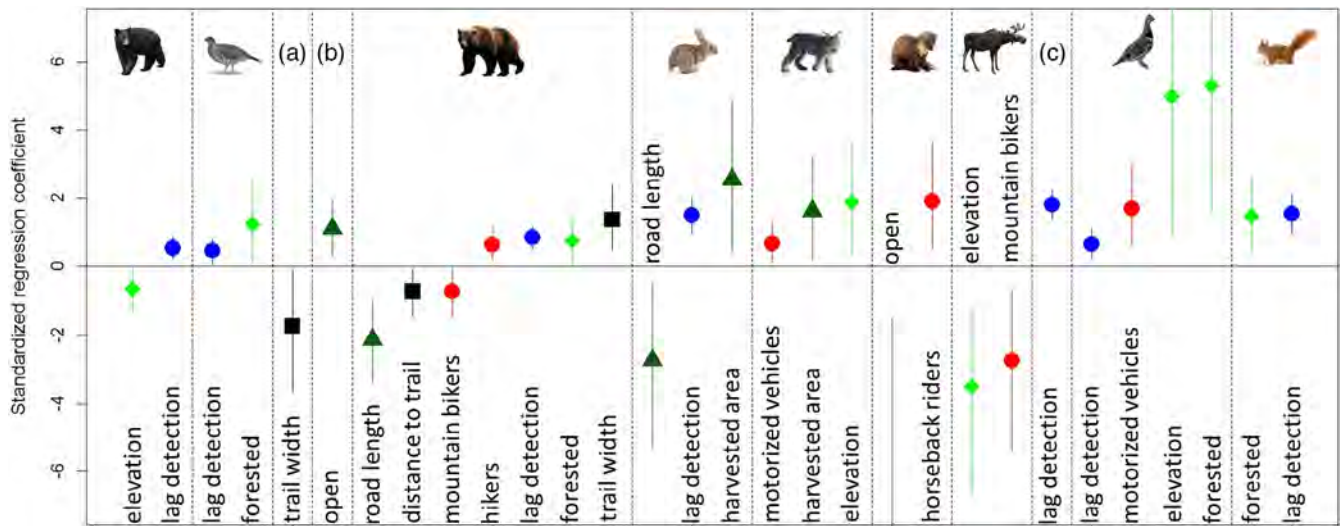


FIGURE 3 Standardized coefficients, from regression models of weekly detections for 13 wildlife species across 61 camera trap sites in the South Chilcotin mountains, whose 95% credible intervals do not overlap with zero. Species from left to right are Black bear, Dusky grouse, A = Cougar, B = Coyote, Grizzly bear, Snowshoe hare, Canada lynx, American marten, Moose, C = Mule deer, Spruce grouse, Red squirrel. Symbols are as follows: blue circles—lagged weekly presence; light green diamonds—site-level environmental variable; dark green triangles—cell-level environmental variable; red circles—human trail-use variable; black squares—trail characteristic. Labels indicate variable name. See Table S2 for details from full models

probability of use. In contrast, environmental variables at the site level ($n = 8$) and the larger landscape surrounding the site ($n = 6$) accounted for half of all statistically significant variables, with temporal autocorrelation ($n = 7$) and trail/camera characteristics ($n = 3$) accounting for the remainder.

All species had AAR values that were significantly greater than 1 for each recreation type, indicating some level of finer-scale trail displacement (Figure 4; note that $\log(1) = 0$ on y-axis). As expected, hiking (median AAR = 2.71) and horseback riding (2.38) had the lowest AAR estimates and were not statistically different from one

another. In contrast to our expectation, mountain bikers (4.96), rather than motorized vehicles (4.16), had the highest AAR estimate. These were not statistically different from one another, but both were statistically greater than the AAR of horseback riding, and, in the case of mountain bikers, also higher than hikers (Table S3). Random effect estimates revealed that most species showed no statistically significant differences among one another for any of the recreation types (Figure 4). The only statistically significant difference was for mountain biking, where mule deer had significantly lower AAR values than those for black bears *Ursus americanus* (Figure 4).

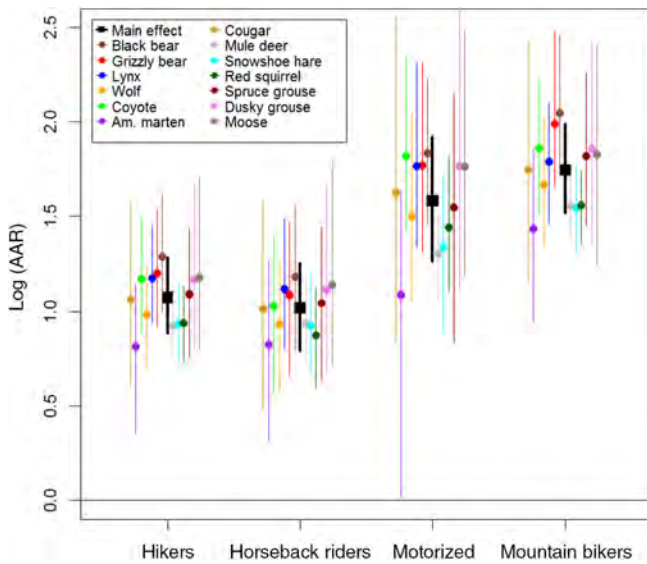


FIGURE 4 Main effects and species random effects from a Bayesian hierarchical regression model of avoidance-attraction ratios (AAR, log-transformed) as a function of recreation type for 13 wildlife species detected at 61 camera trap sites in the South Chilcotin mountains. Vertical lines indicate 95% Bayesian credible intervals on AAR coefficient; intervals greater than zero (i.e., an AAR of 1 in original units) indicate avoidance of a recreation activity

4 | DISCUSSION

Both wildlife and people were frequently detected by camera traps in the South Chilcotin mountains. Despite substantial recreational use of this landscape, environmental variables, rather than human trail use, were predominantly responsible for shaping where and when wildlife were detected at camera traps. We detected few statistically-significant associations between recreational activities and wildlife trail use; at the weekly scale, the only negative associations between human and wildlife activity were between mountain biking and both moose and grizzly bear. Given the ecological and cultural importance of these two species, their relatively slow life histories (e.g., low birth rates), and the increasing popularity of mountain biking in this and other regions, these negative associations warrant further investigation. Moose populations are declining in parts of British Columbia (Kuzyk et al., 2018), and negative impacts of recreation at limited scales have been previously reported (Harris, Nielson, Rinaldi, & Lohuis, 2014). Grizzly bear range has retracted across most of North America (Laliberte & Ripple, 2004), and grizzly bears have been shown to avoid recreational activity in other areas (e.g., Ladle et al., 2019). While recreational disturbance may be of secondary importance for grizzly bear and moose populations, it may nevertheless

represent a significant addition to the cumulative effects of human disturbance facing these and other species (Heinemeyer et al., 2019; Shackelford, Standish, Ripple, & Starzomski, 2018). Furthermore, the variable responses we observed across species suggest that recreational disturbance has the potential to alter species interactions and community structure. For instance, positive associations with human activity for some species have been linked to a “predator shield” effect (e.g., Muhly, Semeniuk, Massolo, Hickman, & Musiani, 2011) which may explain the positive responses observed for some prey and mesopredator species in our study (e.g., lynx, marten, grouse; Figure 4). Further exploration of whether similar effects are observed at time scales other than weekly, as well as an expanded set of explanatory variables that help rule out any omitted variable bias (Blanchet, Cazelles, & Gravel, 2020), would also be fruitful.

While few recreational impacts were observed at the weekly time scale, our finer-scale analysis of wildlife displacement following human activity showed that all species of wildlife avoided all types of human recreational events on trails. This temporal avoidance by wildlife was highest for motorized vehicles and mountain biking, results that are consistent with studies documenting greater levels of wildlife disturbance associated with the noise and speed of motorized vehicles (Ladle et al., 2018). They also suggest that wildlife in the study area may perceive mountain bikers more similarly to motorized vehicles than to nonmotorized recreation. The velocity at which mountain bikes travel along trails, as well as the tremendous growth of the activity, has led to concerns on their impact on wildlife (e.g., Scholten, Moe, & Hegland, 2018), especially after high-profile incidences of conflict with grizzly bears (e.g., Servheen, Manley, Starling, Jacobs, & Waller, 2017).

Group size of mountain bikers was higher than that of hikers (Table 1), which reflects the fact that mountain biking is a more popular activity than hiking at our study site. This raises the possibility that the effects observed among the different recreational types at a weekly scale are a function of absolute abundance of a particular recreational activity, rather than differences among activities such as speed and noise. However, group size of horseback riders was only slightly lower than that of mountain bikers, so group size alone may not explain the finer-scale differences in AAR between these two recreational categories. Ultimately, longer-term data collection and analysis are required to determine whether the results from this one season are consistently observed in the future, and to identify mechanisms underlying observed differences among recreation types. In particular, an approach that models AAR as a function of environmental

variables at and around camera stations may be a useful advance on the work presented here, and help reconcile some of the different effects we observed in our two analyses. In the meantime, given high rates of increase in all types of recreational visits to our study site—as with parks elsewhere—managers should continue to monitor trends in all forms of human recreation for indications of possible impacts on wildlife. Ultimately, monitoring wildlife population trends is also a critical component of ensuring adequate conservation measures in the face of increasing anthropogenic pressures.

While our study characterizes the potential impacts of both motorized and nonmotorized recreational activity on wildlife, there are limitations to the inferences we can make. We focused on estimating wildlife behavioral responses to human activities on trails and roads, but without off-trail sampling we were unable to determine the spatial magnitude to which wildlife may be displaced from roads or trails by anthropogenic activities. As such, it was not possible to establish whether wildlife were avoiding large areas of habitat through which roads or trails run, or briefly moving off trails into adjacent habitat during times of human use. Future studies that incorporate an off-trail camera sampling stratum would improve our ability to distinguish between these scenarios (cf. Kays et al., 2017). Satellite tracking collars on focal species of conservation concern would further allow such differentiation at finer behavioral scales (Suraci, 2020), and would provide a useful complement to the knowledge gained from trail and off-trail camera study designs. In addition, we acknowledge that overall recreational use at our study site is relatively low compared to more accessible sites with higher visitor numbers, which perhaps explain why in aggregate, wildlife and humans were both frequently detected on roads and trails. For example, although detailed records of park visitors to the SCM have not been collected, visitors to all parks in this region of BC (Thompson-Cariboo) were only about 10% of those in parks in the region containing the province's largest city, Vancouver (BC Parks, 2018).

Despite the limitations of trail-based camera sampling, our modeling approach permitted a direct comparison between environmental factors and human trail-use activities, allowing their relative impacts on wildlife to be evaluated. With empirical data linking wildlife to potential stressors, managers and other stakeholders can assess which management interventions are likely to have the greatest impact on the distribution of wildlife in the SCM. This information has been explicitly recognized as fundamental but currently lacking for management of this particular park and surrounding landscape (BC Parks, 2019), as well as many other PAs around the world (Geldmann et al., 2013). Camera traps are being

increasingly used for large-scale monitoring of wildlife (Ahumada et al., 2011; Steenweg et al., 2017); because they can be positioned to also capture human activities, networks of camera traps can help disentangle how recreational activity interacts with environmental conditions to shape the distribution and abundance of wildlife. Furthermore, while remote sensing or field-based measurements of environmental conditions can be incorporated into statistical models of wildlife detections such as we did here, camera traps themselves can also be used to passively monitor vegetation at sites (Bater et al., 2011), providing an additional ground-based data source for local environmental conditions.

The importance of PAs in conservation has long been recognized (Naughton-Treves, Holland, & Brandon, 2005), and is currently undergoing a renaissance as global initiatives to conserve substantial fractions of the earth's surface gain momentum (Dinerstein et al., 2019). While PAs typically protect landscapes and ecosystems from the most disruptive anthropogenic activities, tourism and other recreational uses of PAs is increasing dramatically around the world (Balmford et al., 2015). Although nature-based tourism can play a positive role in biodiversity conservation (Naidoo et al., 2016), monitoring possible impacts of such tourism on wildlife can be difficult (Muntifering et al., 2019), as even basic data such as the number of visits to PAs can be difficult to acquire or noncomprehensive (Buckley, 2009). Networks of camera traps can therefore play a vital role in monitoring the number and type of human activities occurring in and around PAs, with these data being also harnessed to assess impacts of visitors on the wildlife that PAs and their surrounding landscapes are designed to conserve.

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CONFLICT OF INTEREST

The authors declare no conflicts of interest.

AUTHOR CONTRIBUTIONS

Robin Naidoo and A. Cole Burton conceived the study. Robin Naidoo conducted the field work and data analysis. Robin Naidoo and A. Cole Burton wrote the paper. Robin Naidoo and A. Cole Burton acquired funding to support the study.

ETHICS STATEMENT

This research was conducted under protocol A18-0234 from the University of British Columbia's Animal Care Committee.

DATA AVAILABILITY STATEMENT

Data available from the authors upon reasonable request to the WildCAM network (<http://wildcams.ca>).

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REFERENCES

- Ahumada, J. A., Silva, C. E. F., Gajapersad, K., Hallam, C., Hurtado, J., Martin, E., ... Andelman, S. J. (2011). Community structure and diversity of tropical forest mammals: Data from a global camera trap network. *Philosophical Transactions of the Royal Society B*, *366*, 2703–2711.
- Balmford, A., Green, J. M. H., Anderson, M., Beresford, J., Huang, C., Naidoo, R., ..., Manica, M. (2015). Walk on the wild side: Estimating the global magnitude of visits to protected areas. *PLOS Biology*, *13*, e1002074. <https://doi.org/10.1371/journal.pbio.1002074>
- Bater, C. W., Coops, N. C., Wulder, M. A., Hilker, T., Nielsen, S. E., McDermid, G., & Stenhouse, G. B. (2011). Using digital time-lapse cameras to monitor species-specific understory and overstorey phenology in support of wildlife habitat assessment. *Environmental Monitoring and Assessment*, *180*, 1–13.
- BC Parks. (2018). *2017/2018 statistics report*. Victoria, BC: BC Parks.
- BC Parks. (2019). *South Chilcotin Mountains Park and Big Creek Park management plan*. Kamloops, BC: Ministry of Environment and Climate Change Strategy.
- Blanchet, F. G., Cazes, K., & Gravel, D. (2020). Co-occurrence is not evidence of ecological interactions. *Ecology Letters*, *23*, 1050–1063.
- Buckley, R. (Ed.). (2004). *Environmental impacts of ecotourism*. Wallingford, England: CABI.
- Buckley, R. (2009). Parks and tourism. *PLoS Biology*, *7*, e1000143.
- Burkner, P.-C. (2017). Brms: An R package for Bayesian multilevel models using Stan. *Journal of Statistical Software*, *80*, 1–28.
- Burton, A. C., Neilson, E., Moreira, D., Ladle, A., Steenweg, R., Fisher, J. T., ... Boutin, S. (2015). Wildlife camera trapping: A review and recommendations for linking surveys to ecological processes. *Journal of Applied Ecology*, *52*, 675–685.
- Coppes, J., Nopp-Mayr, U., Grünschachner-Berger, V., Storch, I., Suchant, R., & Braunisch, V. (2018). Habitat suitability modulates the response of wildlife to recreation. *Biological Conservation*, *227*, 56–64.
- Dinerstein, E., Vynne, C., Sala, E., Joshi, A. R., Fernando, S., Lovejoy, T. E., ... Wikramanayake, E. (2019). A global deal for nature: Guiding principles, milestones, and targets. *Science Advances*, *5*, eaaw2869.
- Dirzo, R., Young, H. S., Galetti, M., Ceballos, G., Isaac, N. J. B., & Collen, B. (2014). Defaunation in the anthropocene. *Science*, *345*, 401–406.
- Efford, M. G., & Dawson, D. K. (2012). Occupancy in continuous habitat. *Ecosphere*, *3*, art32.
- Fisher, J. T., & Burton, A. C. (2018). Wildlife winners and losers in an oil sands landscape. *Frontiers in Ecology and Evolution*, *16*, 323–328. <https://doi.org/10.1002/fee.1807>
- Geldmann, J., Barnes, M., Coad, L., Craigie, I. D., Hockings, M., & Burgess, N. D. (2013). Effectiveness of terrestrial protected areas in reducing habitat loss and population declines. *Biological Conservation*, *161*, 230–238.
- Gelman, A. (2008). Scaling regression inputs by dividing by two standard deviations. *Statistics in Medicine*, *27*, 2865–2873.
- Glover-Kapfer, P., Soto-Navarro, C. A., & Wearn, O. R. (2019). Camera-trapping version 3.0: Current constraints and future priorities for development. *Remote Sensing in Ecology and Conservation*, *5*, 209–223. <https://doi.org/10.1002/rse2.106>
- Hansen, M. C., Potapov, P. V., Moore, R., Hancher, M., Turubanova, S. A., Tyukavina, A., ... Townshend, J. R. G. (2013). High-resolution global maps of 21st-century forest cover change. *Science*, *342*, 850–853.
- Harris, G., Nielson, R. M., Rinaldi, T., & Lohuis, T. (2014). Effects of winter recreation on northern ungulates with focus on moose (*Alces alces*) and snowmobiles. *European Journal of Wildlife Research*, *60*, 45–58.
- Heinemeyer, K., Squires, J., Hebblewhite, M., O'Keefe, J. J., Holbrook, J. D., & Copeland, J. (2019). Wolverines in winter: Indirect habitat loss and functional responses to backcountry recreation. *Ecosphere*, *10*, e02611.
- Hendry, H., & Mann, C. (2017). Camelot—Intuitive Software for Camera Trap Data Management.
- Hermosilla, T., Wulder, M. A., White, J. C., Coops, N. C., & Hobart, G. W. (2015). Regional detection, characterization, and attribution of annual forest change from 1984 to 2012 using Landsat-derived time-series metrics. *Remote Sensing of Environment*, *170*, 121–132.
- Johnson, C. N., Balmford, A., Brook, B. W., Buettel, J. C., Galetti, M., Guangchun, L., & Wilmshurst, J. M. (2017). Biodiversity losses and conservation responses in the Anthropocene. *Science*, *356*, 270–275.
- Kays, R., Kranstauber, B., Jansen, P., Carbone, C., Rowcliffe, M., Fountain, T., & Tilak, S. (2009). Camera traps as sensor networks for monitoring animal communities. *International Journal of Research and Reviews in Wireless Sensor Networks*, *1*, 19–29.
- Kays, R., Parsons, A. W., Baker, M. C., Kalies, E. L., Forrester, T., Costello, R., ... McShea, W. J. (2017). Does hunting or hiking affect wildlife communities in protected areas? *Journal of Applied Ecology*, *54*, 242–252.
- Kuzyk, G., Hatter, I., Marshall, S., Procter, C., Cadsand, B., Lirette, D., & Klaczek, M. (2018). Moose population dynamics during 20 years of declining harvest in British Columbia. *Alces: A Journal Devoted to the Biology and Management of Moose*, *54*, 101–119.
- Ladle, A., Avgar, T., Wheatley, M., Stenhouse, G. B., Nielsen, S. E., & Boyce, M. S. (2019). Grizzly bear response to

- spatio-temporal variability in human recreational activity. *Journal of Applied Ecology*, 56, 375–386.
- Ladle, A., Steenweg, R., Shepherd, B., & Boyce, M. S. (2018). The role of human outdoor recreation in shaping patterns of grizzly bear-black bear co-occurrence. *PLoS One*, 13, e0191730.
- Laliberte, A. S., & Ripple, W. J. (2004). Range contractions of North American carnivores and ungulates. *Bioscience*, 54, 123–138.
- Larson, C. L., Reed, S. E., Merenlender, A. M., & Crooks, K. R. (2016). Effects of recreation on animals revealed as widespread through a global systematic review. *PLoS One*, 11, e0167259.
- Larson, C. L., Reed, S. E., Merenlender, A. M., & Crooks, K. R. (2019). A meta-analysis of recreation effects on vertebrate species richness and abundance. *Conservation Science and Practice*, 1, e93. <https://doi.org/10.1111/csp1112.1193>
- MacKenzie, D. I., Nichols, J. D., Lachman, G. B., Droege, S., Royle, J. A., & Langtimm, C. A. (2002). Estimating site occupancy rates when detection probabilities are less than one. *Ecology*, 83, 2248–2255.
- McLellan, M. L., McLellan, B. N., Sollmann, R., Lamb, C. T., Apps, C. D., & Wittmer, H. U. (2019). Divergent population trends following the cessation of legal grizzly bear hunting in southwestern British Columbia, Canada. *Biological Conservation*, 233, 247–254.
- Miller, A. B., Leung, Y.-F., & Kays, R. (2017). Coupling visitor and wildlife monitoring in protected areas using camera traps. *Journal of Outdoor Recreation and Tourism*, 17, 44–53.
- Muhly, T. B., Semeniuk, C., Massolo, A., Hickman, L., & Musiani, M. (2011). Human activity helps prey win the predator-prey space race. *PLoS One*, 6, e17050.
- Muntifering, J. R., Linklater, W. L., Naidoo, R., Uri-Khob, S., Preez, P. D., Beytell, P., ... Knight, A. T. (2019). Sustainable close encounters: Integrating tourist and animal behaviour to improve rhinoceros viewing protocols. *Animal Conservation*, 22, 189–197.
- Naidoo, R., Fisher, B., Manica, A., & Balmford, A. (2016). Estimating the economic losses to tourism in Africa from the illegal killing of elephants. *Nature Communications*, 7, 13379. <https://doi.org/10.1038/ncomms13379>
- Naughton-Treves, L., Holland, M. B., & Brandon, K. (2005). The role of protected areas in conserving biodiversity and sustaining local livelihoods. *Annual Review of Ecology and Systematics*, 30, 219–252.
- Neilson, E. W., Avgar, T., Burton, A. C., Broadley, K., & Boutin, S. (2018). Animal movement affects interpretation of occupancy models from camera-trap surveys of unmarked animals. *Ecosphere*, 9, e02092.
- Newbold, T., Hudson, L. N., Hill, S. L. L., Contu, S., Lysenko, I., Senior, R. A., ... Purvis, A. (2015). Global effects of land use on local terrestrial biodiversity. *Nature*, 520, 45–50.
- Norouzzadeh, M. S., Nguyen, A., Kosmala, M., Swanson, A., Palmer, M. S., Packer, C., & Clune, J. (2018). Automatically identifying, counting, and describing wild animals in camera-trap images with deep learning. *PNAS*, 115, E5716–E5725.
- Parsons, A. W., Bland, C., Forrester, T., Baker-Whatton, M. C., Schuttler, S. G., McShea, W. J., ... Kays, R. (2016). The ecological impact of humans and dogs on wildlife in protected areas in eastern North America. *Biological Conservation*, 203, 75–88.
- R Core Team. (2018). R: A language and environment for statistical computing. R Foundation for Statistical Computing. Vienna, Austria. Retrieved from <https://www.R-project.org>
- Reed, S. E., & Merenlender, A. M. (2008). Quiet, nonconsumptive recreation reduces protected area effectiveness. *Conservation Letters*, 1, 146–154.
- Rich, L. N., Miller, D. A. W., Robinson, H. S., McNutt, J. W., & Kelly, M. J. (2016). Using camera trapping and hierarchical occupancy modelling to evaluate the spatial ecology of an African mammal community. *Journal of Applied Ecology*, 53, 1225–1235.
- Sandbrook, C., Luque-Lora, R., & Adams, W. (2018). Human Bycatch: Conservation surveillance and the social implications of camera traps. *Conservation and Society*, 16, 493–504.
- Scholten, J., Moe, S. R., & Hegland, S. J. (2018). Red deer (*Cervus elaphus*) avoid mountain biking trails. *European Journal of Wildlife Research*, 64, 8. <https://doi.org/10.1007/s10344-018-1169-y>
- Servheen, C., Manley, T., Starling, D. M., Jacobs, A., & Waller, J. (2017). *Board of Review Report on the death of Brad treat from a grizzly bear attack*. Missoula, MT: US Fish and Wildlife Service.
- Shackelford, N., Standish, R. J., Ripple, W., & Starzomski, B. M. (2018). Threats to biodiversity from cumulative human impacts in one of North America's last wildlife frontiers. *Conservation Biology*, 32, 672–684.
- Steenweg, R., Hebblewhite, M., Kays, R., Ahumada, J., Fisher, J. T., Burton, C., ... Rich, L. N. (2017). Scaling-up camera traps: Monitoring the planet's biodiversity with networks of remote sensors. *Frontiers in Ecology and the Environment*, 15, 26–34.
- Steenweg, R., Whittington, J., Hebblewhite, M., Forshner, A., Johnston, B., Petersen, D., ... Lukacs, P. M. (2016). Camera-based occupancy monitoring at large scales: Power to detect trends in grizzly bears across the Canadian Rockies. *Biological Conservation*, 201, 192–200.
- Suraci, J. P. (2020). Fine-scale movement decisions by a large carnivore inform conservation planning in human-dominated landscapes. *Landscape Ecology*, 35, 1635–1649.
- Wearn, O. R., & Glover-Kapfer, P. (2019). Snap happy: Camera traps are an effective sampling tool when compared with alternative methods. *Royal Society Open Science*, 6, 181748.
- Willis, C. (2015). The contribution of cultural ecosystem services to understanding the tourism–nature–wellbeing nexus. *Journal of Outdoor Recreation and Tourism*, 10, 38–43.

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Additional supporting information may be found online in the Supporting Information section at the end of this article.

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