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Debris Barriers Reduce the Effects of Livestock Grazing Along Streams After Timber Harvest☆

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A B S T R A C T

Timber harvesting in riparian zones without leaving a buffer can increase the likelihood of livestock grazing along streams. The wet soil around small streams can accentuate the negative impact of grazing, affecting vegetation, as well as other ecosystem characteristics. In this study, we tested the effectiveness of using coarse woody debris, a readily available barrier method to reduce the effects of livestock grazing after timber harvesting on small stream vegetative and other ecosystem values over two grazing seasons. We placed debris barriers within four recently harvested cutblocks where livestock graze seasonally on extensive, forested rangeland in the Okanagan region of Interior British Columbia, Canada. We sampled cover, species richness, bare soil, litter, biomass, trampling, and manure within 0.25m^2 quadrats to determine the effectiveness of the barriers on these variables over two grazing seasons. We used log response ratios to compare the pretreatment and post-treatment values of these variables in control plots and plots with debris barriers. We also used log response ratios to test the effect of debris barriers on biomass utilization within and outside 1-m^2 cages. Results varied by site: debris barriers resulted in improved richness and litter in two sites each, reduced trampling in three sites, and reduced bare soil in one site when compared with plots without barriers. Barriers also increased cover in one site but had no influence on manure. Biomass utilization was significantly reduced by debris barriers in uncaged grazed plots compared with caged plots. Debris barriers can be a convenient tool to mitigate the potential negative effects of livestock grazing after timber harvest in and around small headwater streams.

Introduction

Riparian zones, especially in headwater streams, play a critical role in protecting resource values (Lowe & Likens 2005). Vegetation along streams can provide shade, stabilize banks, trap sediments, and filter pollutants (Dadkhah & Gifford 1980; Vought et al. 1995), while leaf litter can provide food for aquatic invertebrates (Correll 2005; Hrodey et al. 2009). Forest harvesting without leaving a riparian buffer on streams and other moisture receiving areas, and where there are overlapping grazing tenures, increases the likelihood of disturbance to these riparian areas (Armour et al. 1994). These harvested cutblocks become ideal sites for livestock due to the potential for increased forage and available water until the next generation of trees is established (Dwire et al. 2006; Johnson et al. 2016).

Increased access by livestock to riparian zones can have negative effects on soil properties (Fleischner 1994; Clary 1995) and hydrologic conditions (Belsky et al. 1999). Livestock can use the stream channel as a trail, resulting in a reduction in plant cover and biomass, increase in stream temperatures and soil compaction, and addition of sediments and nutrients to the system (Belsky et al. 1999; Clary & Kinney 2002). Trampling and overgrazing can lead to shifts in the plant community, and reduction in plant cover and richness, by reducing the amount of available moisture for plant production (Willatt & Pullar 1984; Herbst et al. 2011). Vegetation shifts from deeper rooted stabilizing species, with strong root systems to shallower rooted wetland upland and non-native species, have been observed along degraded streambanks (Dobarro et al. 2013; Dickard 2015). Reduced infiltration rates caused by trampling changes the timing of spring flows and leads to increased runoff and erosion, affecting water quality and the land's ability to store and release water later in the season (Gifford & Hawkins 1978). Sediment inputs into the streambed due to trampling can remove habitat for aquatic invertebrates and spawning areas for fish (Armour et al. 1994; Fleischner 1994). In addition,

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nutrients from manure can have a negative effect on water quality that can increase the number of pathogens such as *Escherichia coli* and campylobacter, which are associated with water-borne diseases (Derlet et al. 2009; Lewis et al. 2019). Therefore, adequate management of riparian areas requires assessing different aspects of riparian proper functioning condition (Swanson et al. 2017).

Studies have provided evidence for several conservation management methods that can reduce the harm caused by livestock to riparian ecosystems. Briske et al. (2011) noted that management efforts such as reduced stocking rates can increase rangeland production, as plant production decreases with stocking rate (Raynor et al. 2019), although individual plant responses are complex and may depend on the ability of the plant to adjust its reproductive strategies (Guo et al. 2020). Reduced stocking rates also decreased species richness, with inconsistent effects on plant cover and diversity (Briske et al. 2011). Deferment of the grazing season in nonarid conditions can maintain plant cover and productivity, although during drought, this deferment may need to exceed 2 years to show any effects (Briske et al. 2011). Changing grazing systems have also been explored for their effects on vegetation properties. Briske et al. (2011) observed that 83% of studies showed no difference between rotational or continuous grazing on biomass production. George et al. (2011) in a review of riparian rangeland management practices found that the duration, frequency, or seasonality of grazing can affect riparian vegetation composition, but their effects on biomass production varied by site. Importantly, they showed that using livestock distribution practices such as water placements, supplemental feeding, or herding reduced the residence time of livestock in the riparian zone and, thus, livestock use and impacts on riparian vegetation (George et al. 2011). Livestock exclusion via fencing can also be effective in improving riparian health, although it can be combined with aforementioned distribution practices to improve efficacy (George et al. 2011; Derose et al. 2020).

The riparian forest acts as a source of coarse woody debris that is important to channel morphology (Correll 2005). Forested riparian zones with vegetative ground cover and litter provide a buffer from potential upland nonpoint source pollutants (Peterjohn & Correll 1984; Fortier et al. 2016). Using coarse woody debris to reduce the presence and effects of livestock is a known, yet unquantified practice that needs further confirmation of its effectiveness where the timbered reserve zone has been removed through harvesting. A pilot study conducted in 2011 in the Aberdeen Plateau, southeast of Vernon, British Columbia showed positive observed effects for riparian health (Andrew Pantel, personal communication). More focused data collection and analysis are necessary to understand the potential utility of debris barriers along unprotected small streams in cutblocks. Coarse woody debris barriers may provide protection by minimizing the direct access and linear movement within small streams while not requiring the costs of building and maintaining fences.

Currently, many management practices are employed in British Columbia, Canada to manage livestock in community watersheds and to protect riparian values. These include off-stream watering, cross fencing, salting, herding on horseback, and rotational grazing, which have proven effective in increasing riparian vegetative cover (Rigge et al. 2014; Duhaime 2019). All these practices require a financial and time commitment on the part of the province and the range agreement holder. It is not practical or necessary to fence out livestock from streams on extensive rangelands in most cases, when large herds disperse into smaller groups within large pastures. Past studies have shown livestock prefer to use off-stream water sources where available, which minimizes direct access to surface water sources (Godwin & Miner 1996). Riding horses to herd livestock to get good distribution and use of available forage is effective (Carter et al. 2017), but comes with a time commitment on the part of the range agreement holder that can conflict with other ranch duties during the grazing season.

Here, we tested the effectiveness of using coarse woody debris criss-crossed over the stream channel to reduce the effects of livestock grazing on small streams following timber harvest. This is a low-cost, operational management option that can be applied on small, headwater streams and nonclassified drainages. We chose four sites for this study southeast of Vernon in the Okanagan region of British Columbia. The headwater streams (< 1.5 m wide) within the sites flow into larger streams that are the sources of domestic water supply in Vernon, Kelowna and Lake Country in the Okanagan Valley. For this reason, resource users of these community watersheds are monitored closely by resource stewardship staff, water purveyors, and the public to ensure that standards are being met. The sites were harvested between 2005 and 2011 and selected on the basis of having active range agreement overlap. Sites were sampled over the 2016 and 2017 grazing seasons.

We hypothesized that criss-cross logging debris barriers would result in a decrease in evidence of livestock on the small streams, within the riparian zone of recently harvested cutblocks, as evidenced by reduction in trampling, bare soil, and manure. Reduced livestock presence would result in increased plant cover, species richness, and litter. We also hypothesized that debris barriers would reduce the difference in biomass yield between caged and uncaged plots.

Methods

Study area

The study area was located southeast of Vernon, British Columbia, Canada in an area known as the Aberdeen Plateau (Fig. 1). The area is forested, and uses of the land include timber harvesting, livestock grazing, recreation, water storage and delivery, mining, and gravel extraction. The four sites selected for this research occur in the montane spruce biogeoclimatic zone. The Okanagan Dry Mild Montane Spruce Variant (MSdm1) is typically found in the Okanagan Highlands with an elevational range of 1 300−1 600 m (Lloyd et al. 1990), characterized by cold winters and moderately short summers. Slope at the sites ranged from between 1% and 7%. Livestock were moved from pasture to pasture as per the dates in the agreement holder's range use plan. Targeted use of the four sites was not controlled in any way and varied between sites and years (Table 1). Actual levels of use of the sites depended on many factors including weather and management by the agreement holder.

Recently harvested cutblocks with small streams within them were chosen as study sites. Brunette 1 is a 30-ha clearcut that was harvested in 2011 and grazed in the spring and fall. Brunette 2, a 45-ha clearcut, was harvested in 2010 and grazed in late summer (see Table 1). Echo is a 37-ha clearcut harvested in 2009, while Crescent is a 48-ha clearcut harvested in 2005. Both sites are grazed in the fall (see Table 1). Before harvesting, the site overstory was occupied by lodgepole pine and spruce, while the understory consisted primarily of grasses such as bluejoint reedgrass *(Calamagrostis canadensis),* with some sedge species (i.e., *Carex rossii* and *Carex disperma*) and forbs such as *Fragaria virginiana* and *Geum macrophyllum*. All four sites were along forest service roads seeded with domestic forage species that were used by livestock to move throughout and between pastures. Livestock use of the cutblocks was ascertained before selecting them for use in this study.

Experimental design

In each of the four study sites, two treatments were established with each treatment being replicated once at each site, such that

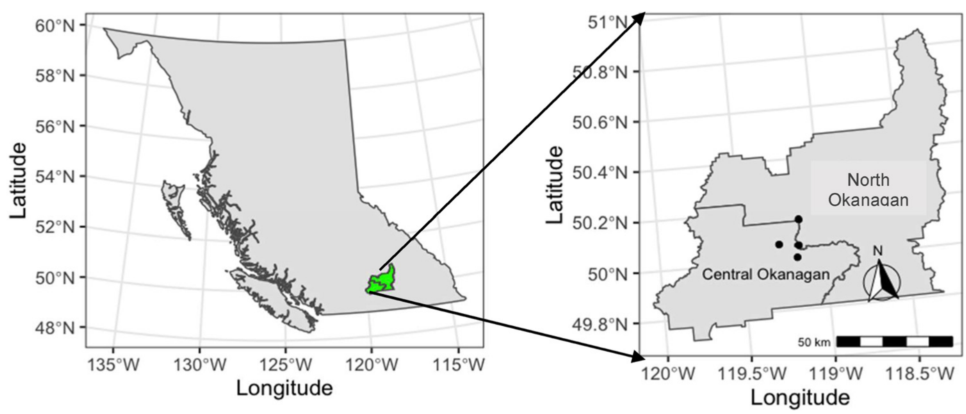

Figure 1. Locations of the four study sites within Vernon and Central Okanagan in British Columbia. **A,** Map of British Columbia, with Vernon and Central Okanagan in green. **B,** Site locations in black. Map plotted using data from the bcmaps package (Teucher et al., 2018), which presents spatial data in the BC Albers projection.

Table 1
Grazing schedule as per range use plan of range agreement holder.

Site (size of cutblock)	Elevation (m)	Size of pasture (ha)	Timing, season of use (all yr)	Estimate of use	No. & class of livestock (AUM)
Brunette 1 (30 ha)	1 428	1 816	July 1-14, Sept 15-Oct 7	Moderate	300 cow/calf pairs & 15 bulls
Brunette 2 (45 ha)	1 380	4 905	Aug 7-Aug 31	Light	300 cow/calf pairs & 15 bulls
Crescent (48 ha)	1 475	6 544	Sept 1-Sept 30	Light	300 cow/calf pairs & 15 bulls
Echo (37 ha)	1 375	5 106	Sept 1-Oct 7	Light - Moderate	300 cow/calf pairs & 21 bulls

AUM indicates animal unit months.

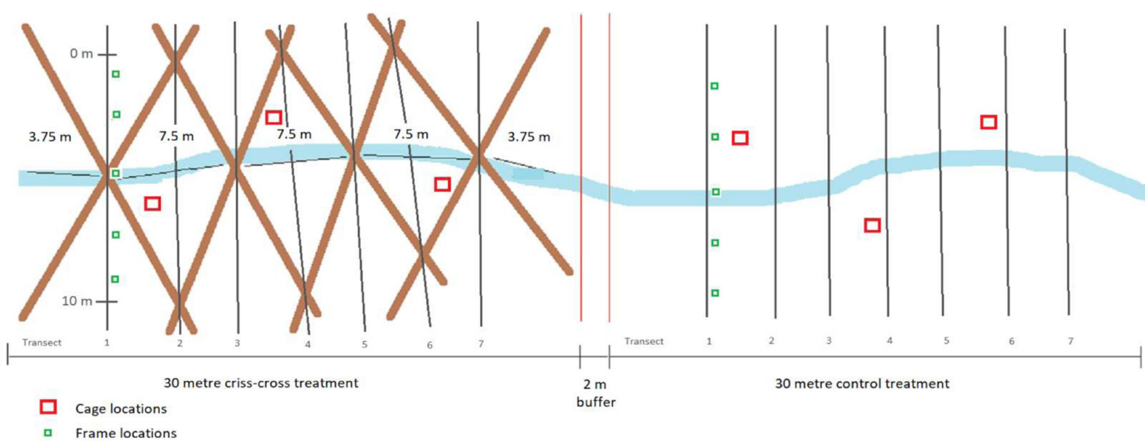

Figure 2. Experimental design showing one replicate of debris barrier treatment and control within a site. Vegetation, substrate, tramples, bare soil, and manure were measured in 0.25m^2 frames at 1 m, 3 m, 5 m, 7 m, and 9 m along the width of the transects (green squares). Biomass was collected from 1-m square cages (red squares) and corresponding 1-m square uncaged plots within each treatment.

each site had four experimental units. One treatment in each site (i.e., two experimental units) was left untouched as a control, and the other had tree trunks (logs) as debris barriers criss-crossed across the stream channel. The treatments alternated from the downstream end based on a coin flip that determined which one would go first. Coarse woody debris treatments were designed to have four X's within the 30-m treatment length along the stream channel (Fig. 2). The logs used to construct barriers were based on site availability and ranged from 13 m to 15 m in length, with diameters at the tapered ends between 20 cm and 50 cm. The barriers were carefully put in place by an excavator with the center of the X's at 3.75 m, 11.25 m, 18.75 m, and 26.25 m, respectively, from the downstream end of the 30-m treatment. The width of each treatment was 10 m, 5 m in both directions perpendicular to the stream channel centreline (see Figs. 2 and 3). Target height for the barriers was 0.75 m but varied between 0.3 m and 1.2 m due to topography and obstacles such as rocks and stumps at each site.

Sampling

Plant vegetation

Each experimental unit (plot) was 30 m long and 10 m wide (see Fig. 2), and the plant community was representative of the riparian area. A 10-m transect line was systematically set up at 3.75-m intervals along the 30-m treatment length of the plot and perpendicular to the stream, resulting in seven transects per treatment (3.75 m, 7.5 m, 11.25 m, 15 m, 18.75 m, 22.5 m, 26.25 m). There were no transects at 0 m and 30 m. Five quadrats were taken along each transect. Within each quadrat, we estimated vegetation, litter, tramples, bare soil, and manure inside a 0.25m^2 frame at 1 m, 3 m, 5 m, 7 m, and 9 m. Frame 3 (at 5 m) was always at the center line of the stream. Absolute cover of vegetation by species and of substrates, tramples (cover of hoof marks in the soil), bare soil, and manure were measured at these five locations within the frame (see Fig. 2), such that the total cover could add

Figure 3. Photographs showing the setup of coarse woody debris treatments in the four sites.

up to more than 100%. This was replicated once within each site, resulting in 35 quadrats that were measured in each treatment for a total of 140 quadrats per site. Baseline measurements were taken in June and July 2016, before grazing and before the treatments were established. Subsequent post-treatment measurements were also taken in the fall of 2016 and 2017 following livestock use. Fall measurements were taken the last wk of September and the first wk of October in 2016 and the last 2 wk of September in 2017.

Biomass yield

At the beginning of each grazing season in the spring, cages were set up at each location to measure biomass. Three cages were set up within each treatment. Within the coarse woody debris treatments, cages were located inside each of the three diamond restriction areas (see Fig. 2). Cages in the control treatments were placed such that if a criss-cross treatment were projected onto the ground as in the barrier treatment, there would be one in each of the diamond restriction areas (see Fig. 2). The location of each 1×1 m caged subplot was selected so that a paired uncaged subplot was found within 5 m. Paired caged/uncaged subplots were chosen to reflect similar species and density within the plant community. The center of each paired subplot was marked with a nail and washer and the distance (m) and azimuth was recorded from the center of each cage location to the uncaged pair. These subplots were clipped in late September or early October and may have included some regrowth from after grazing. A 0.5 m^2 wire hoop was laid down with the nail and washer being the center point of the hoop. Each hoop was clipped to ground level with clipping shears. All herbaceous vegetation was bagged, oven dried, and weighed to the nearest gram. Oven drying was completed at 65°Celsius for 24 h or until constant weight was reached.

Data analysis

For all response variables except biomass, we calculated the log response ratio (LRR) for each plot, comparing the values in Fall 2016 and 2017 with the baseline pretreatment data. The LRR was calculated as $(\ln \frac{Fall\ 2016/2017}{Baseline})$. In order to avoid dividing by 0, we added 1 to both numerator and denominator. We then used the LRR as an input response variable in linear models. The LRR allows us to test the response of the vegetation and nonvegetation variables before and after treatment was applied. A positive LRR indicates an increase of the variable after treatment. The general experimental design was based on three fixed factors: treatments (control and debris), sampling dates (Fall 2016 and Fall 2017), and the four sites. We used linear models with the Gaussian distribution to test the relationship between vegetative and nonvegetative response variables and the three experimental factors, using quadrat, nested within transect as a random factor, using the lme4 package (Bates et al. 2007) in R (R Core Team 2019). We used the analysis of variance (ANOVA) function to obtain type III ANOVA tables of main effects and interactions.

We also used the LRR to calculate the response of biomass to treatment. Here, we calculated the LRR of biomass as $(\ln \frac{\text{Caged}}{\text{Not caged}})$ of the paired subplots within each replicate in our experimental design. We expected the LRR to be positive as caged subplots would have a higher biomass than uncaged subplots. A high LRR of biomass indicates a high level of biomass utilization in uncaged plots compared with caged plots. No baseline measures were collected for biomass yield; therefore, we tested the response of LRR to treatment, sampling dates (Fall 2016 and 2017), and site. When main effects or interactions were significant for all models, we computed estimated marginal means using the "emmeans" package after adjusting for multiple comparisons (Lenth et al. 2018) to identify which treatment levels were significantly different from one another. We were interested in treatment effects on vegetative and nonvegetative variables and in two-way and three-way interactions between treatment and any other factors (i.e., site or date).

Results

Nonvegetation variables

Litter LRR was significantly higher with debris barriers than control in Brunette 1 and Brunette 2, but not in Crescent and Echo (Table 2, Fig. 4a), while bare soil LRR was significantly lower with debris treatments in only the Echo site (see Table 2, Fig. 4b). Of the nonvegetation variables, litter, trample, and bare soil showed significant treatment-site interactions (see Table 2). Trample LRR was generally lower with debris treatments in all sites except Brunette 1 (see Table 2, Fig. 5a). Manure LRR did not respond to any of the experimental factors (see Table 2, Fig. 5b).

Table 2
Results from three-way model testing differences in nonvegetation variables (tramples, bare soil, manure and litter) among sites (Brunette 1, Brunette 2, Echo, Crescent); treatments (Control and Barrier); and sampling date (Baseline, Fall 2016, Fall 2017). *P* values < 0.05 in bold. Denominator degree of freedom = 1 070 for all models.

	df	Tramples		Bare soil		Manure		Litter	
		F value	*P* value	*F* value	*P* value	*F* value	*P* value	*F* value	*P* value
Site	3	51.400	**< 0.001**	38.894	**< 0.001**	0.908	0.437	20.909	**< 0.001**
Treatment	1	23.671	**< 0.001**	0.437	0.509	0.071	0.790	4.420	**0.036**
Date	1	30.475	**< 0.001**	1.491	0.222	0.131	0.718	56.580	**< 0.001**
Site × treatment	3	3.141	**0.025**	4.217	**0.006**	0.590	0.622	4.853	**0.002**
Site × date	3	27.630	**< 0.001**	0.342	0.795	1.425	0.234	8.990	**< 0.001**
Treatment × date	1	2.696	0.101	2.112	0.146	0.011	0.918	2.536	0.112
Site × treatment × date	3	0.416	0.741	0.492	0.688	0.449	0.718	0.309	0.819

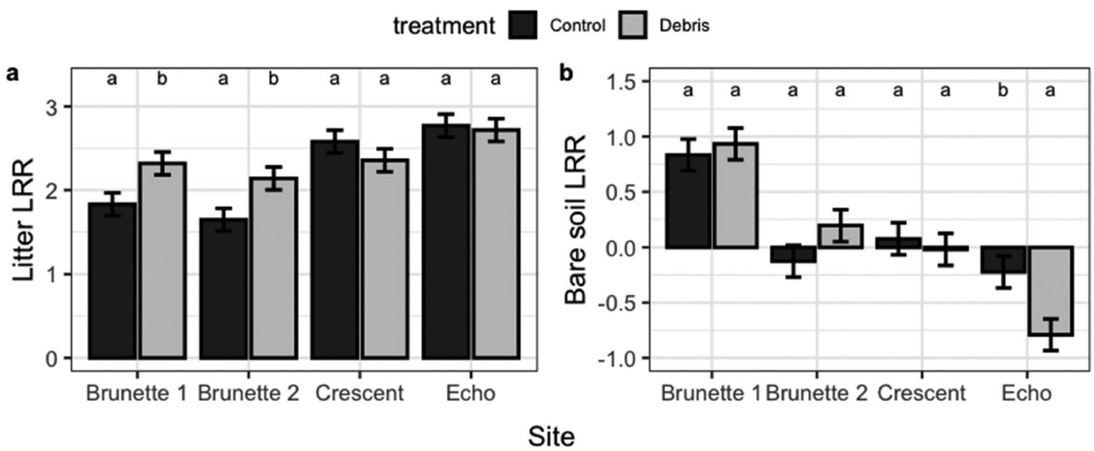

Figure 4. Log response ratio of litter (**a**) and bare soil (**b**) measured in 0.25 m^2 subplots within debris and control treatments in four sites in interior British Columbia. Letters indicate significant differences between treatments after analysis of variance/Tukey post hoc test with *P* < .05 for significant differences. Error bars represent standard errors of the mean.

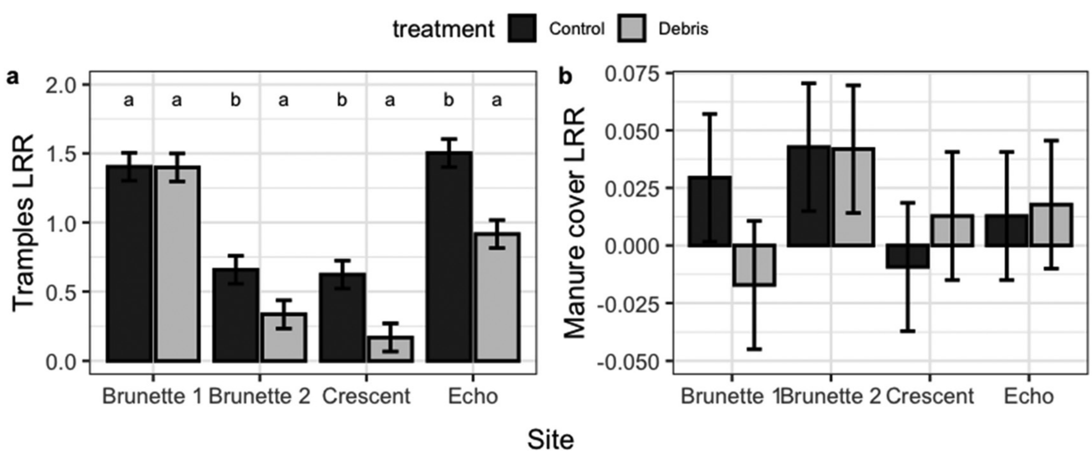

Figure 5. Log response ratio of trample cover (**a**) and manure cover (**b**) measured in 0.25 m^2 subplots within debris and control treatments in four sites in interior British Columbia. Letters indicate significant differences between treatments after analysis of variance/Tukey post hoc test with *P* < .05 for significant differences. Error bars represent standard errors of the mean.

Vegetation variables

Both species richness and cover LRRs displayed significant interaction between site and treatment (Table 3). Post-hoc analysis revealed that species richness LRR was significantly different between debris barriers and control in three sites: Brunette 1, Brunette 2, and Echo. Brunette 2 and Echo displayed higher species richness with debris barriers, and lower values in control plots compared with baseline (Figs. 6a and 6b). Species richness in Brunette 1 was higher in both debris and control compared with baseline as shown by positive LRRs, although debris barriers surprisingly displayed lower LRR than control. LRR of plant cover was significantly different between treatments, displaying a higher value with debris barriers, although the impact of treatments varied by site (see Table 3). Overall plant cover was reduced compared with baseline in all sites, as shown by negative LRR values, but reduction in plant cover was significantly lower with debris barriers, compared with the control in only Brunette 2 (see Fig. 6b).

Across all the treatments, sites, and dates, biomass was higher in caged plots with a mean and standard deviation of 49.194 g ± 29.975 g, compared with uncaged plots: 37.281 g ± 28.046 g; thus, the LRR of biomass was positive for most paired caged/uncaged plots. The LRR of biomass was significantly influenced by treatment ($F_{1,78} = 9.580$, $P = .003$) and interactions between site and

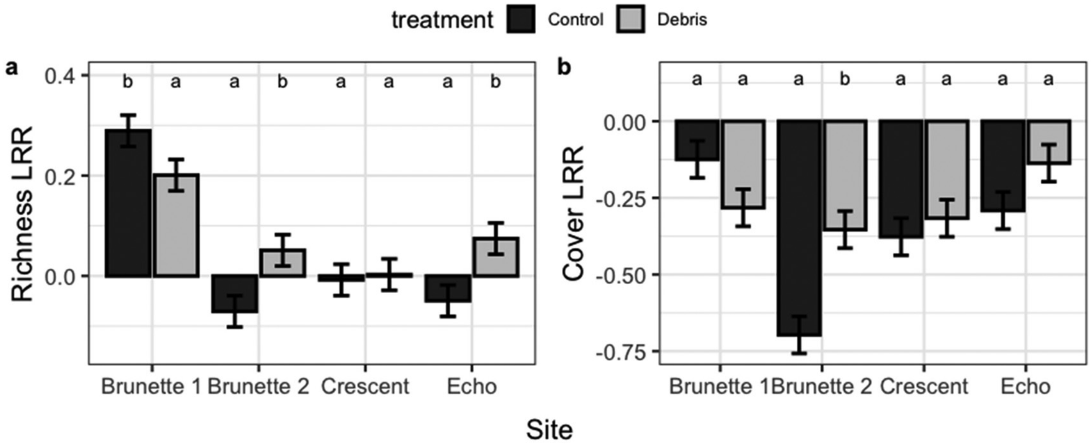

Figure 6. Log response ratio of species richness (**a**) and plant cover (**b**) measured in 0.25 m^2 subplots within debris and control treatments in four sites in interior British Columbia. Letters indicate significant differences between treatments after analysis of variance/Tukey post hoc test with $P < .05$ for significant differences. Error bars represent standard errors of the mean.

Table 3
Results from three-way model testing differences in vegetation variables (cover and species richness) among sites (Brunette 1, Brunette 2, Echo, Crescent); treatments (Control and Barrier); and sampling date (Fall 2016, Fall 2017). *P* values < 0.05 in bold. Denominator degree of freedom = 1 070 for all models.

	df	Richness		Cover	
		F value	*P* value	*F* value	*P* value
Site	3	31.225	**< 0.001**	13.348	**< 0.001**
Treatment	1	3.631	0.057	5.931	**0.015**
Date	1	13.009	**< 0.001**	0.501	0.479
Site × treatment	3	5.352	**0.001**	6.424	**< 0.001**
Site × date	3	1.797	0.146	10.485	**< 0.001**
Treatment × date	1	0.250	0.617	0.360	0.549
Site × treatment × date	3	0.770	0.511	0.511	0.675

date ($F_{1,78} = 7.663$, $P < .001$) and treatment and date ($F_{1,78} = 5.712$, $P = .019$). All other factors were not significant (**Table S1, available online at ...**). The LRR of the control treatment was statistically higher than the LRR of the debris treatment in Fall 2016, but not in Fall 2017 (Fig. 7). LRR of biomass was highest in Fall 2016 in Brunette 1 and lowest in Fall 2017 in the same site (**Fig. S1, available online at ...**). Other sites did not significantly differ in LRR between years.

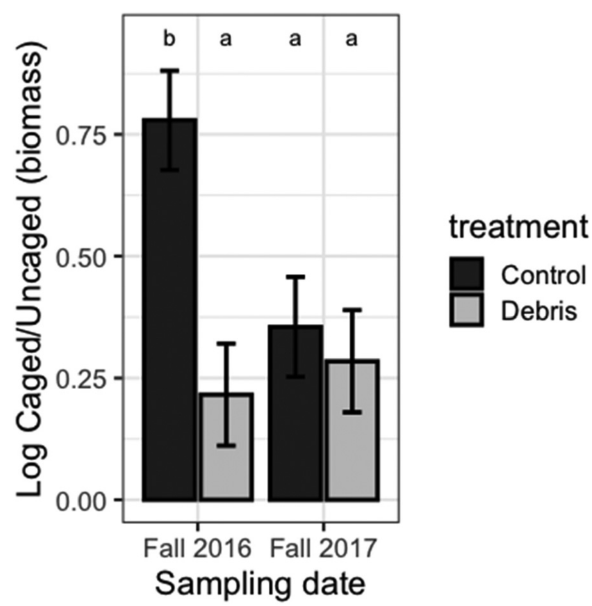

Figure 7. Log response ratio of biomass measured in 1 m^2 subplots within control and debris treatments. Letters indicate significant differences between treatments after analysis of variance/Tukey post hoc test with $P < .05$ for significant differences. Error bars represent standard errors of the mean. Log response ratio calculated as ln (Caged/Uncaged) for paired cages within each replicate.

Discussion

In this study, we demonstrated that debris treatments influenced livestock distribution, generally resulting in reduced trampling and bare soil, which increased litter and species richness and reduced biomass consumption, showing that the use of debris barriers can reduce the impacts of grazing after logging in riparian zone management. Despite the short duration of this study, positive effects of debris barriers begin to emerge, although these effects can vary due to grazing intensity.

Response of nonvegetation variables to debris barriers

Trampling was reduced by debris barriers in three of our four study sites. Barriers can reduce the time spent by cattle in a riparian zone, and debris windrows have proven effective in certain riparian pastures and can result in increased vegetation cover, a result we observed in this study (see next section) (Rigge et al. 2014; Sullivan et al. 2021).

Debris barriers resulted in increased litter accumulation in two out of the four study sites. Litter LRR differed by treatment in both Brunette 1 and Brunette 2, but not in the other two sites. Studies have reported decreases in standing litter due to livestock grazing across different ecosystems due to continuous removal of biomass and an increase in litter decomposition rate (Neufeld 2008; Li et al. 2018). Litter can influence the understory microclimate due to its modulation of soil temperature and moisture; thus, litter accumulation can be viewed as a positive effect of grazing exclusion by debris barriers (Dormaar et al. 1997; Jacobo et al. 2006).

Barriers only significantly reduced bare soil compared with controls, in the Echo site, contrary to studies that show consistent increases in ground cover with increased grazing (Yeo 2005; Teuber et al. 2013; Goosey et al. 2019; Valdez-Cepeda et al. 2021), although Ferreira et al. (2020) did not observe exclosure effects after the first year of a 4-yr experiment. The lack of a treatment effect in three sites could be a result of inadequate grazing pressure. It is possible that at current grazing intensities, more than 1 yr of observation may be required to show the effects of grazing on bare ground. Surprisingly, the difference in trampling between treatments did not result in differences in manure.

Response of vegetation to debris barriers

Vegetation cover showed a smaller decrease with debris treatment compared with controls in only one out of four sites. Riparian management methods such as rotational grazing, supplementary water or nutrients, and the use of physical barriers to restrict or redistribute livestock have led to increases in vegetation cover (Al-Rowaily et al. 2015; Swanson et al. 2015; Sullivan et al. 2021). Here, we show that in one site, the introduction of readily available tree trunks as debris barriers can reduce trampling, leading to an increase in vegetation cover. Carline and Walsh (2007) showed that grazing exclusion within time scales of 3–5 yr yielded 50–100% increase in plant cover. Successfully maintaining the vegetation cover by limiting grazing should reduce soil compaction and result in lower runoff volumes and lower volume of overland flow (George et al. 2011; Tufekcioglu et al. 2013). Despite the lack of significant results in three sites, our early findings here are encouraging but suggest the need for longer study duration.

Species richness was increased by the debris treatment compared with baseline in the lightly grazed sites, while the heavily grazed site displayed a greater increase in richness with barriers compared with the control. In a review, Fleischner (1994) concluded that a reduction in species richness was an ecological cost of grazing and it took up to 30 yr after livestock removal for species richness to increase back to normal levels. On the other hand, Koerner et al. (2018) in a global study found that the effect of grazing on species richness depends on dominance: Grazing dominant species led to an increase of less dominant species by making resources more available to them, resulting in increased species richness. The impact of grazing on plant richness and diversity can therefore depend on a host of factors, including, aridity, community composition, evolutionary history, or grazing intensity (Milchunas & Lauenroth 1993; Gao & Carmel 2020). In this study, Brunette 1 was the most heavily used site and control (grazed) plots in this site exhibited a more positive species richness response than plots with debris barriers, suggesting that there are complex community- or site-level responses to grazing intensities that require further experimental manipulation (Douglas & Kauffman 1995; Jackson & Allen-Diaz 2006). Therefore, rather than merely observing the effects of grazing exclusion on diversity, shifts in community composition should also be studied over longer periods of time.

Grazing exclusion is expected to result in reduced biomass utilization and, thus, increased biomass accumulation (Schultz et al. 2011). Debris barriers were effective at reducing biomass utilization in uncaged subplots in 2016, but not in 2017. Brunette 1 is the most heavily grazed site and was grazed according to schedule in 2016, but in 2017 there was a change in how the livestock were herded within their rotation in this site. This resulted in significant differences in utilization within Brunette 1 between 2016 and 2017, but not in the other sites, as shown by significant site by sampling date interactions. Thus, heavy grazing in Brunette 1 may explain the overall treatment effect found in 2016 and suggests that the impact of debris barriers is more distinct at high levels of grazing. Kota and Bartos (2010) also showed that after 1 yr, slash barriers were successful in preventing excessive browsing of quaking aspen suckers by cattle. Our study thus provides evidence of the ability of debris barriers to reduce biomass utilization and can help minimize livestock impacts on small streams within cutblocks.

In this study, we used a systematic sampling scheme in this study, which may result in differential impacts of barriers (e.g., between plots close and farther away from the stream center). However, this was minimal in our study as exploratory analyses of plot-level differences between responses of vegetative and nonvegetative variables showed a significant plot effect for only bare soil (**Table S2, available online at ...**). Other sampling methods, such as belt or line transects set up at specified equal distances from the stream or debris center, may prevent this issue from arising. In addition, increasing quadrat size may help in sampling a greater proportion of the study site.

Logging debris barriers can take on different forms and structures based on the materials used, site topography, and the presence of obstacles, which may also partially explain the between site differences in the results (Bailey 2005). Logging barriers are also less expensive and more readily available onsite than constructed fences (Redick & Jacobs 2020). They are also only required until the trees grow and they shade out the forage, making the area less attractive to livestock, which then move on to the next cutblock in succession on a rotating basis. In addition, this study did not account for site-level differences in forage quality and quantity which can influence behavioral responses of livestock (Vavra & Sheehy 1996; Bailey 2005).

Conclusion

This 1-yr study demonstrates that even within short timeframes, using woody debris barriers (a readily available source of protection from livestock grazing after logging) can be effective in riparian zones. The variation in response of vegetative and nonvegetative variables between sites may be due to the low grazing intensity or short-term nature of the study. Over longer time frames and/or at higher grazing intensities, we expect more consistent and stronger treatment effects of barriers.

Although the target was for the barriers to be a minimum of 0.75 m off the ground, it proved impossible to achieve this height across all debris treatments and sites. Utilization occurred within the barrier treatment as designed; it was not meant to provide full exclusion. These results should lead to better riparian health and hydrologic function on areas where livestock graze and timber has been harvested to the edge of small streams.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

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Supplementary materials

Supplementary material associated with this article can be found, in the online version, at doi:10.1016/j.rama.2021.11.002.

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