RELATIONSHIP BETWEEN INTENSITY OF LIVESTOCK GRAZING AND TROUT BIOMASS IN HEADWATERS OF EAST FRONT ROCKY MOUNTAIN STREAMS, MONTANA

by

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ABSTRACT

Livestock grazing is the most common land-use practice in the western United States. Riparian and stream habitats are particularly susceptible to effects of poorly-managed livestock grazing. About 80% of stream and riparian habitats in the western United States are thought to have been damaged by livestock grazing, but because grazing usually pre-dated assessments of fish populations and stream habitats, before and after comparisons are impossible. The spatial and temporal complexity of livestock grazing make it difficult to isolate its effects on instream habitat and channel morphology characteristics. Moreover, instream habitat and channel morphology are also influenced by inherent watershed characteristics (i.e., basin area, gradient, discharge). I assessed the effects of livestock grazing on 25 separate 150-m long sample sites (1400 to 1585 m in elevation) within ten headwater basins along the northeastern Rocky Mountain Front in north-central Montana. I used scat counts as an index of relative grazing intensity to assess the effects of livestock grazing on channel morphology characteristics, stream substrate, instream cover, and trout biomass. To my knowledge, this effort is the first to quantify livestock grazing intensity using scat counts to assess grazing effects on trout biomass. I assessed potential effects that grazing intensity had on habitat condition and fish biomass using linear mixed models, which also accounted for watershed and sample site effects. I found that the proportion of fine sediment in the streambed increased as the number of scats increased ($P < 0.001$), but the area of undercut banks declined as scat counts increased ($P < 0.001$). Estimated trout biomass declined as number of scats increased, even when I accounted random effects of stream and year in a linear mixed-effect model ($P = 0.009$). My results corroborate previous findings that livestock grazing along stream channels may reduce trout biomass, but unlike previous studies I actually quantified grazing intensity using scat counts. Since increased livestock grazing intensities were related to increased levels of fine sediments in streambeds and smaller areas of undercut streambanks, I suggest that these factors may be related to why increased livestock grazing reduced trout biomass.

INTRODUCTION

Livestock grazing is the most common land-use practice in the western United States (Fleischner 1994). Of the 124 million hectares of public land in the contiguous western states (Montana, Wyoming, Colorado, New Mexico, and westward), an estimated 105 million hectares or 85% are grazed by livestock (CAST 1996). In 2012, cattle accounted for 91% of the 7.8 million animal unit months (AUMs) stocked on Bureau of Land Management (BLM) property (Pomarico et al. 2013), showing that cattle are the dominant livestock species stocked on public lands. Sheep and goats accounted for 7.7% of leased AUMs, and horses and burros accounted for 0.5% of AUMs on BLM land (Pomarico et al. 2013).

Grazing can affect water quality, stream channel morphology, hydrology, riparian soils, riparian vegetation, and abundance and diversity of fishes, terrestrial and aquatic macroinvertebrates, amphibians, reptiles, birds, and mammals (reviewed by Fleischner 1994 and Belsky et al. 1999). Riparian and stream habitats are particularly susceptible to degradation from livestock grazing compared to upland habitats (Roath and Kreuger 1982; Fleischner 1994). An estimated 80% of stream and riparian ecosystems in the western United States are damaged by livestock grazing (Belsky et al. 1999). Free-ranging cattle select channel and floodplain habitats over upland habitats (Smith et al. 1992) and feral horses use riparian habitats disproportionately compared to upland habitats (Cran et al. 1997).

Livestock can affect riparian and instream habitat by altering, reducing, or removing riparian vegetation and trampling streambanks (Gunderson 1968; Duff 1977;

Marcuson 1977; Winegar 1977; Roath and Krueger 1982; Kauffman et al. 1983a, 1983b; Elmore and Beschta 1987; Schulz and Leininger 1990; Stevens et al. 1992; Popolizio et al. 1994; Green and Kauffman 1995; Knapp and Matthews 1996). These effects can lead to streambank instability and increased erosion, resulting in significant modifications to channel morphology over time. Accelerated erosion can lead to increased proportions of fine sediment particles in streambed substrates, substrate embeddedness, and higher suspended sediment loads (Winegar 1977; Platts 1982; Kauffman and Krueger 1984; Stevens et al. 1992; Smith et al. 1993; Myers and Swanson 1996; Wood and Armitage 1997; Clary 1999). Livestock grazing can reduce the amount of undercut streambanks (Kauffman et al. 1983b; Gillen et al. 1984; Hubert et al. 1985; Myers and Swanson 1995; Knapp and Matthews 1996), increase bank-full and wetted widths (Duff 1977; Marcuson 1977; Bryant 1982; Kauffman et al. 1983b; Gillen et al. 1984), and reduce stream depths (Bryant 1982; Hubert et al. 1985; Stuber 1985), which increase width-to-depth (W:D) ratios.

Previous studies that assessed the effects of livestock grazing on stream habitats or fish abundance have been hindered by the inability to accurately characterize grazing intensities along stream channels. This has led to somewhat confusing and, often, statistically insignificant findings (Platts 1982; Rinne 1988; Platts 1991). Past studies of grazing effects on fish have used categorical designations of “grazed” or “ungrazed” or grazing exclosures versus grazed stream reaches to evaluate grazing effects. Categorical designation of grazing status assumes that grazing intensity is homogenous within categories, but this assumption is probably violated in many cases due to the variation that has been observed in grazing intensities within and among watersheds (Gillen et al. 1984; Cran et al. 1997; Smith et al. 1992). Moreover, livestock grazing has been pervasive for more than a century in most areas, making it difficult if not impossible to find ungrazed reference reaches.

A method that could be used to quantify livestock grazing intensities along stream channels is counting livestock scats. Scat counts (pellet counts) is a common method used to assess wildlife habitat use (Neff 1968). However, I was unable to find any study that used scat counts to quantify livestock grazing intensity as an explanatory variable for observed fish biomasses or instream habitat conditions. A few studies have used livestock scat counts to estimate domestic livestock grazing intensity, but all these were conducted in upland habitats. Scat counts of domestic cattle to estimate grazing intensities were first used in Utah to compare cattle and deer grazing (Julander 1955). Sheep pellets counts were used to estimate their grazing intensity in Western Australia (Abensperg-Traun 1996). Cattle scat counts were used to evaluate the relationship between various cattle grazing intensities and bee abundance in Mediterranean shrubland habitats (Vulliamy et al. (2006) and also to assess the effects of grazing intensity on Desert Horned Lizards *Phrynosoma platyrhinos* in Utah (Newbold and MacMahon 2008). The absence of scat counts as a method for assessing livestock grazing effects on instream habitat and fish assemblages is probably because its origin in wildlife research.

Evaluating the effects of land-use practices on stream habitat and fish assemblages can be difficult because of confounding with other natural watershed characteristics and processes (Allan 2004). These factors include the physical and chemical properties of the watershed and characteristics of specific reaches within the stream network. Reach level characteristics such as riparian vegetation, stream channel morphology, substrate composition, streambank erosion potential, and fine sediment transport are a function of watershed characteristics and processes, including bed geology and soil types, channel and valley gradient, stream discharge, and basin area (Brush 1961; Osterkamp and Hedman 1977; Medina et al. 2005). These watershed characteristics may vary among watersheds and longitudinally within watersheds as stream networks coalesce from headwaters to valley bottoms (Rahel and Hubert 1991; Clarkson and Wilson 1995; Medina et al. 2005).

Stream reach characteristics, such as channel morphology and streambed substrate composition, can influence trout densities and biomasses. Higher W:D ratios translate to less habitat diversity and instream cover, which reduce trout abundances and biomasses (Lewis 1969; Horan et al. 2000). An increase in fine sediment in stream substrates negatively affects salmonid populations (Platts 1991; Howell 2001). The volume of interstitial spaces among large substrate particles is reduced as levels of fine sediment increase (Ryan 1991). Reduction in interstitial volumes among substrate particles results in a net loss of available habitat for benthic invertebrates, which lowers abundances and biomasses of benthic invertebrates (Wagener and Laperriere 1985). Declines in invertebrate abundances may affect trout abundances and biomasses because aquatic invertebrates are common prey of salmonids (Murphy et al. 1981; Wilzbach et al. 1986; Suttle et al. 2004). High levels of fine sediment can also reduce the spawning success of salmonids by creating anoxic conditions within spawning redds and entombing newly hatched fry (Chapman 1988; Magee et al. 1996; Louhi et al. 2011; Sear et al. 2014; Sternecker et al. 2014).

I evaluated the relative effects of livestock grazing on stream habitat and trout biomass in headwater portions of streams draining the east slopes of the Rocky Mountains in north-central Montana. The objectives of this study were to (1) determine the effects of livestock grazing on instream habitat conditions and (2) assess relationships between livestock grazing and trout biomasses. I hypothesized that high-intensity livestock grazing along streams would alter streambank vegetation and stability, which would subsequently modify streambed substrate and channel morphology characteristics, and reduce trout biomass. I wanted to account for 1) watershed characteristics that also might affect streambed substrate and channel morphology, and 2) the potential variation in livestock grazing intensity within and among watersheds using scat counts as a quantitative indicator of grazing intensity.

STUDY AREA

I sampled 150-m stream reaches at 25 sites (hereafter termed sample sites) within 10 headwater streams located along the northeastern Rocky Mountain Front in north-central Montana (Figure 1; Table 1). Seven of these streams were on the Blackfeet Reservation, which is bordered by Glacier National Park and the Bob Marshall Wilderness Complex to the west and Canada to the north. Because cattle were allowed to graze along Blackfeet Reservation streams throughout the vegetative growing season (season-long), I sampled three additional streams (Dupuyer, North Fork Willow, and Deep creeks) located immediately south of the reservation boundary that also originated on the eastern border of the Bob Marshall Wilderness. These streams had tightly controlled grazing management or no grazing, and served as a priori low-intensity grazing streams. All selected sample streams were thought to support populations of trout that could also be accessed for sampling. Selection of sampling sites within each stream was largely dependent on the ability to perform depletion estimates and habitat surveys in a single day. I purposely selected sample sites within streams that were not adjacent to developed roads (paved or gravel), but in some cases, sampling sites were close to primitive two-track roads.

All sample sites were located in headwater areas of streams because these were the only places where cattle grazing was the primary land-use. At lower elevations, inferences may have been confounded by other land-use practices, such as farming and irrigation withdrawals. All sample sites were located in first and second-order stream segments (Strahler 1957) and elevations of sample sites ranged from 1400 to 1585 m

(Table 1). Because trout species dominate fish assemblages in headwater reaches of this region (Rahel and Hubert 1991), I assumed that comparisons of trout populations and biomasses within and among streams should provide a reasonable metric for testing effects of different grazing intensities. Moreover, the relatively small size (mean wetted width < 10 m) of these streams should increase the probability of detecting grazing effects on riparian and instream habitat (Allan 2004).

Salmonids in the study area included nonnative Brook Trout (*Salvelinus fontinalis*) and Rainbow Trout (*Oncorhynchus mykiss*), and native Westslope Cutthroat Trout (*Oncorhynchus clarkii lewisi*), Bull Trout (*Salvelinus confluentus*; St. Mary River drainage only), and Mountain Whitefish (*Prosopium williamsoni*). Other fish species included Longnose Dace (*Rhinichthys cataractae*), Mottled Sculpin (*Cottus bairdi*), White Sucker (*Catostomus commersoni*), Longnose Sucker (*Catostomus catostomus*), Mountain Sucker (*Catostomus platyrhynchus*), and Lake Chub (*Couesius plumbeus*).

Table 1. Stream sites (*n* = 25), grazing regimes, estimated mean summer discharge, and estimated scat counts during the 2013 field season. Scat surveys were visual counts made within 15 m buffers on both sides of sampling sites. Not Grazed sites had either no evidence of livestock grazing or were not designated grazing allotments, although occasionally a few individuals strayed from adjacent allotments. Rest-Rotation was defined as not grazed in consecutive seasons Season-Long was defined as grazed from May to September, and Fall-Winter was defined as grazed from October to May.

Stream	Site	Total scat (n)	Old scat (n)	Grazing regime	Mean summer discharge (cfs)
Dupuyer	1	3	3	Fall-Winter	16.0
Dupuyer	2	19	5	Fall-Winter	16.0
Dupuyer	3	3	4	Fall-Winter	16.0
Dupuyer	4	8	5	Fall-Winter	13.7
Deep	1	0	0	Not Grazed	18.3
Deep	2	0	0	Not Grazed	18.3
Middle Fork Lee	1	2	0	Not Grazed	0.6
Middle Fork Lee	2	0	0	Not Grazed	0.6
Middle Fork Lee	3	5	3	Not Grazed	0.6
North Fork Whitetail	1	0	0	Not Grazed	5.6
North Fork Whitetail	2	7	0	Not Grazed	5.6
North Fork Willow	1	5	0	Rest-Rotation	2.8
North Fork Willow	2	2	0	Rest-Rotation	2.8
North Fork Willow	3	0	0	Rest-Rotation	2.8
East Fork Lee	1	35	4	Season-Long	2.2
East Fork Lee	2	73	9	Season-Long	2.2
East Fork Lee	3	48	7	Season-Long	2.2
East Fork Lee	4	55	13	Season-Long	2.2
Little Badger	1	209	21	Season-Long	6.5
Roberts	2	361	42	Season-Long	0.7
Roberts	1	676	57	Season-Long	0.7
South Fork Cut Bank	1	21	2	Season-Long	7.7
South Fork Cut Bank	2	69	6	Season-Long	7.7
South Fork Cut Bank	3	96	17	Season-Long	7.7
South Fork Cut Bank	4	96	23	Season-Long	7.7

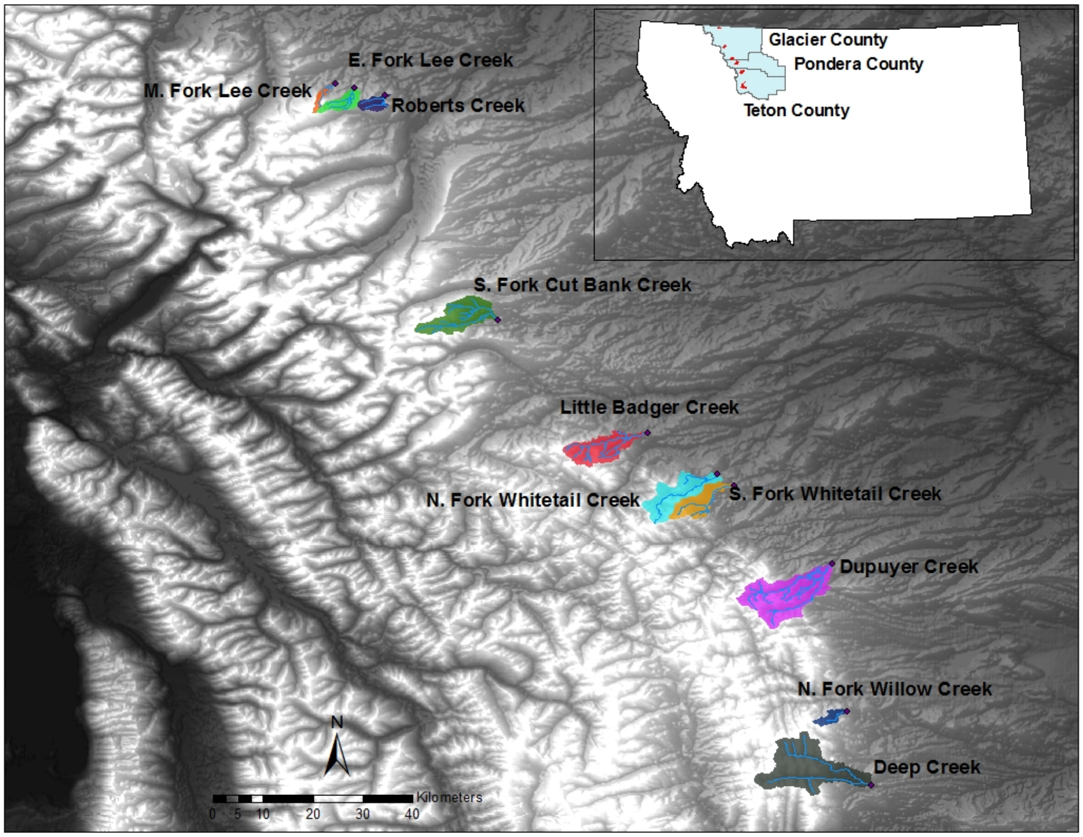

Figure 1. Map showing the watersheds sampled (each individual stream shown had two to four 150-m study sections that were sample at least once during the summers of 2012-2013). Highlighted areas represent the area of the watershed located above the lowest sampling site.

METHODS

Stream Habitat

Stream habitat condition was assessed at 18 sites in 2012 and 7 sites in 2013. Bank-full widths and wetted widths were measured at 16 transects (located perpendicular to the flow at 10-m intervals across the 150-m length of the sampled channel) within each sample site and averaged for the site. Substrate type and stream depth were also recorded at 0.5-m intervals along each transect. I classified stream substrates based on particle sizes: boulder, cobble, mud/silt, sand, or gravel (McMahon et al. 1996). I summed the number of observations for each substrate type and computed the proportion of each substrate type within each sample section. I recorded whether each cross-stream sample transect did or did not bisect a pool and computed the proportion of pool habitats for each sample site. I estimated mean depth of each sample site by pooling all measured depths across the entire sample site. Instream cover was estimated by measuring lengths and widths of undercut banks along both stream banks and measuring the dimensions of large ($\geq$ 15 cm diameter) woody debris within each 150-m long sample site. Undercut banks were visually identified, the horizontal depth of the undercut was measured at the approximate longitudinal center of the undercut, and the entire length of the undercut bank was recorded (Heifetz et al. 1986). The depths and lengths of undercuts were multiplied and summed (total area; m^2) by study site.

Due to time constraints, habitat measurements were only made during one year at each sample site, so I needed to assume that habitat likely did not change between 2012 and 2013. To qualitatively assess this assumption, I summarized USGS streamflow data for gauging stations on the St. Mary (station code 05017500) and Two-Medicine (station code 06091700) rivers for the years 2012 and 2013. Annual runoff (acre/feet) in the St. Mary River was 113% (2012) and 110% (2013) of mean annual runoff (years of record: 1951-2014). Annual runoff (acre/feet) in the Two-Medicine River was 94% (2012) and 104% (2013) of mean annual runoff (1977-2014). Annual runoff was within 13% of normal values during both years and I did not observe noticeable changes in sample sites between years. Consequently, I believe the assumption that habitat likely did not change significantly between the two years was valid.

ArcGIS 10.2 (ESRI 2014) was used to calculate channel gradient at each site. The lower boundary of each section was plotted on a 1:100,000 National Hydrography Dataset (NHD; USGS 2013) layer that had each stream segment routed for distance. A 30-m digital elevation model (DEM; USGS 2002) was used to estimate elevations. Points were plotted 1850 m upstream and downstream from the lower bound of each site on the NHD stream layer. This distance was used to minimize the effect of error introduced by a 3 m degree of accuracy in the elevation dataset. These points were intersected with the DEM layer to estimate elevations at these lower and upper locations. Gradient was calculated as the stream rise divided by section length between the upper and lower bounds. This value was then multiplied by 100 to convert to percent gradient. Two streams (Little Badger and Deep Creek) had sampling sites that were less than 1850 m below the confluence of two streams. Their gradients were calculated over 1109 m and 1759 m, respectively.

Estimated mean summer stream discharges were used as a relative measure of stream size. Mean summer stream discharges were estimated using the historical US Forest Service variable infiltration capacity (VIC) macro-scale hydrologic model for the upper Missouri watershed (USFS 2014) for each sample site. Modeling of mean summer flow using this method predicts observed discharges with reliable accuracy (Wenger et al. 2010). I intersected my sampling sites with the same NHD layer referenced above using ArcGIS 10.2 (ESRI 2014). I cross-referenced the code that identified each unique NHD stream segment (ComID) from intersected NHD stream segments to the ComIDs provided from the VIC dataset. After corresponding ComIDs for both my sampling sites and the VIC database were identified, the VIC database file provided estimates of mean summer discharges for each stream segment I sampled.

Fish Abundance

Fish sampling was conducted during both the summers of 2012 and 2013. Of the 25 total sample sites, I sampled 12 sites during both years, 6 sites only during 2012, and 8 sites only during 2013. I used multiple-pass depletion electrofishing to estimate salmonid abundances and biomasses. Fish were captured using a Smith-Root® LR-20B backpack electrofisher operated at voltages in the range of 400 to 600 V, frequencies of 25 to 30 Hz, and pulse widths of 5 µsec to maximize the number of fish captured while minimizing injury to fish (Dwyer et al. 2001). An electrofishing crew consisted of three people. One crewmember wore the backpack shocker and shocked using a wand anode while dragging a cable cathode. A second crewmember followed the shocker netting all stunned fish and a third crewmember held a dip net in the stream channel below the two other crewmembers and carried a mesh bucket for transporting captured fish.

The assumption of population closure for sample sections was met by (1) using block nets (6.5 mm mesh) at the upper and lower ends of all sample sites; (2) using a second netter during most sampling to prevent fish from moving downstream; and (3) the relatively short time (< 4 hours) it took to complete all sample passes (White et al. 1982). Four to five electrofishing passes were made in an upstream direction during each removal treatment. Passes were made in an upstream motion to avoid a loss in water clarity (reducing capture probability). At least four passes were completed to maximize the probability that the majority (> 60%) of the estimated fish were actually caught (Rosenberger and Dunham 2005; Shepard et al. 2013).

Lengths (TL; mm), weights (g), species, and pass number were recorded for all captured salmonids after each electrofishing pass. Weight measurements were only taken during the 2013 field season. Fish were released outside of the sampling section after data were recorded for each pass.

Livestock Grazing Intensity

To quantify the relative intensity of domestic livestock grazing, three people counted all domestic livestock scats within 15-m wide riparian strips along each side of each of the 25, 150-m long sample sites during 2013 (Julander 1955; Neff 1968). The three observers counted scats by walking adjacent to each other along three, 5-m wide strips along each side of the stream at each sample site. The total area sampled at each site was 4500 m^2 (30-m wide by 150-m long). Scats were identified as either horse or cattle and were summed for each sample site by these two classes. Relative age of each observed scat was also recorded as either from the current grazing season or from past years. Current grazing season scats were green, brown, or moist, or a combination thereof. Past grazing season scats were either dry or gray, or both dry and gray. The sum of current and past season scats were used for all grazing intensity models.

Based on my observations in the field I hypothesized that livestock grazing intensity (i.e., scat counts) could have been affected by riparian vegetation types. I evaluated this hypothesis by estimating the proportions of grass and woody vegetation in riparian areas (300-m wide areas centered on the stream channels; Figure 2) using high resolution (30 cm) Landsat imagery (Xie et al. 2008; Yu et al. 2006; date of imagery July 15, 2011) and ArcMap 10.2 (ESRI 2014). I considered three types of vegetation: open grass vegetation, dense woody vegetation, and other, which included rock/cliff faces. I delineated 150-m wide buffers around each study site in ArcMap, clipped the Landsat imagery within these buffers, outlined the three different vegetation types, estimated and summed areas of each type, and computed the proportions of each type along each study section. The proportions of the “Other” category at all sample sites were extremely limited (range 0 to 5%), so for the final analysis I only used open grass and dense woody vegetation categories. Dense woody vegetation included trees and shrubs that were close enough together that underlying grasses were not clearly visible. Open grass vegetation included all other areas where open grass was clearly visible.

Data Analyses

All sample sites were included in final analyses except for those in South Fork Whitetail Creek (Figure 1), which had substantial beaver (*Castor canadensis*) activity and mid-summer water temperatures (≥ 21°C) that exceeded optimum temperatures for growth and survival of trout (Hokanson et al. 1973; Winkle et al. 1990; Bear et al. 2007). I decided high temperatures would confound biomass and grazing inferences, and past beaver activity would confound instream habitat and grazing inferences. Additionally, minnow and sucker species dominated the fish assemblage in this stream and livestock grazing was not evident, suggesting that other factors were probably limiting trout abundance.

I first tested for correlations between all pairs of habitat, watershed, and grazing intensity variables using a Pearson correlation test (Crawley 2007). I suspected that many of the habitat and watershed variables would be correlated to each other, so I used principal component analysis (PCA) to create components of habitat and watershed variables that minimized correlation among components using "prcomp" with "scale = TRUE" in the R statistical package version 3.0.2 (R Development Core Team 2008). PCA computed the proportion of the overall variance within this habitat and watershed variable dataset that was explained by each component and the relative contribution (loading) that each variable contributed to each component. PCA components were assessed for use as covariates in linear mixed models to explore potential relationships between trout biomass, entered as the dependent variable, and estimates of grazing intensity (scat counts) and stream habitat characteristics (independent variables) integrated through PCA components to minimize multicollinearity.

I used a Box-Cox transformation technique to identify potential data transformations to improve normality in response variables that were used in regression analyses (Sakia 1992). The following equation was used to determine which transformation was most appropriate for normalizing response variables, where lambda (λ) is the maximum likelihood estimate of a normal probability plot and y is the un-transformed response variable.

$$y' = \frac{y^{\lambda} - 1}{\lambda}$$

When λ = 0, the transformation is equal to the natural log transformation of the response variable, as was the case for estimated trout biomasses and volumes of large woody debris. The maximum likelihood estimate of lambda for area of undercut banks was λ = -0.5. Therefore the following equation was used to transform area of undercut banks.

$$y' = \frac{y^{-0.5} - 1}{-0.5}$$

I then used linear mixed models to test for relationships between grazing intensity (scat counts) and each habitat parameter estimate. Habitat and watershed variables were estimated for each sample site because livestock grazing was deemed independent, and highly variable, at the site level, but streams were treated as a random effect to account for potential variation at the stream level.

Abundances of trout from two length classes (TL = 75-200 mm and 201-400 mm) at each sampling site were estimated using the "deplet" function in the fish methods package in program R version 3.0.2 (R Development Core Team 2008; Nelson 2014). This method provided an estimate of the number of trout within each sample site and assumes constant effort for each depletion pass for a closed population. Average weight (g) and length (mm) of all trout within each length class and sample section were calculated. Because fish were not weighed during 2012, length-weight regressions were used to estimate weights of individual fish caught in 2012 based on fish weighed during 2013 (Anderson and Neuman 1996). I estimated biomasses (standing crop; g/m^2) of trout by multiplying the mean weight of fish (g) by the estimated number of fish from two length classes (TL = 75-200 mm and 201-400 mm) in each section and then dividing this total weight (g) by the product of the section length (150 m) and average wetted width (m).

Linear mixed models were used to explore potential relationships between trout biomass, entered as the dependent variable, and estimates of grazing intensity (scat counts) and stream habitat and watershed PCA components. Sample year and stream were treated as random effects to account for potential variation among streams and years (Zuur et al. 2009). These statistical analyses were performed in program R version 3.0.2 with the "nlme" package (Pinheiro 2014).

Simple linear regression was used to evaluate the effect of riparian vegetation type on grazing intensity. This analysis was used only for streams located on Blackfeet Tribal lands, where livestock were free-ranging.

Mixed models with random effects of stream and year were used to address spatial and temporal pseudo-replication and to assess the assumption of independence of errors. Normality of all models was assessed by examining residual versus fitted plots. Model assumptions were assessed by using data transformations or running models with and without influential observations to determine their effect on statistical inferences. Regression diagnostics provided in R were used to identify influential observations based on leverage and outlier tests. Statistical significance was set at $P \leq 0.05$.

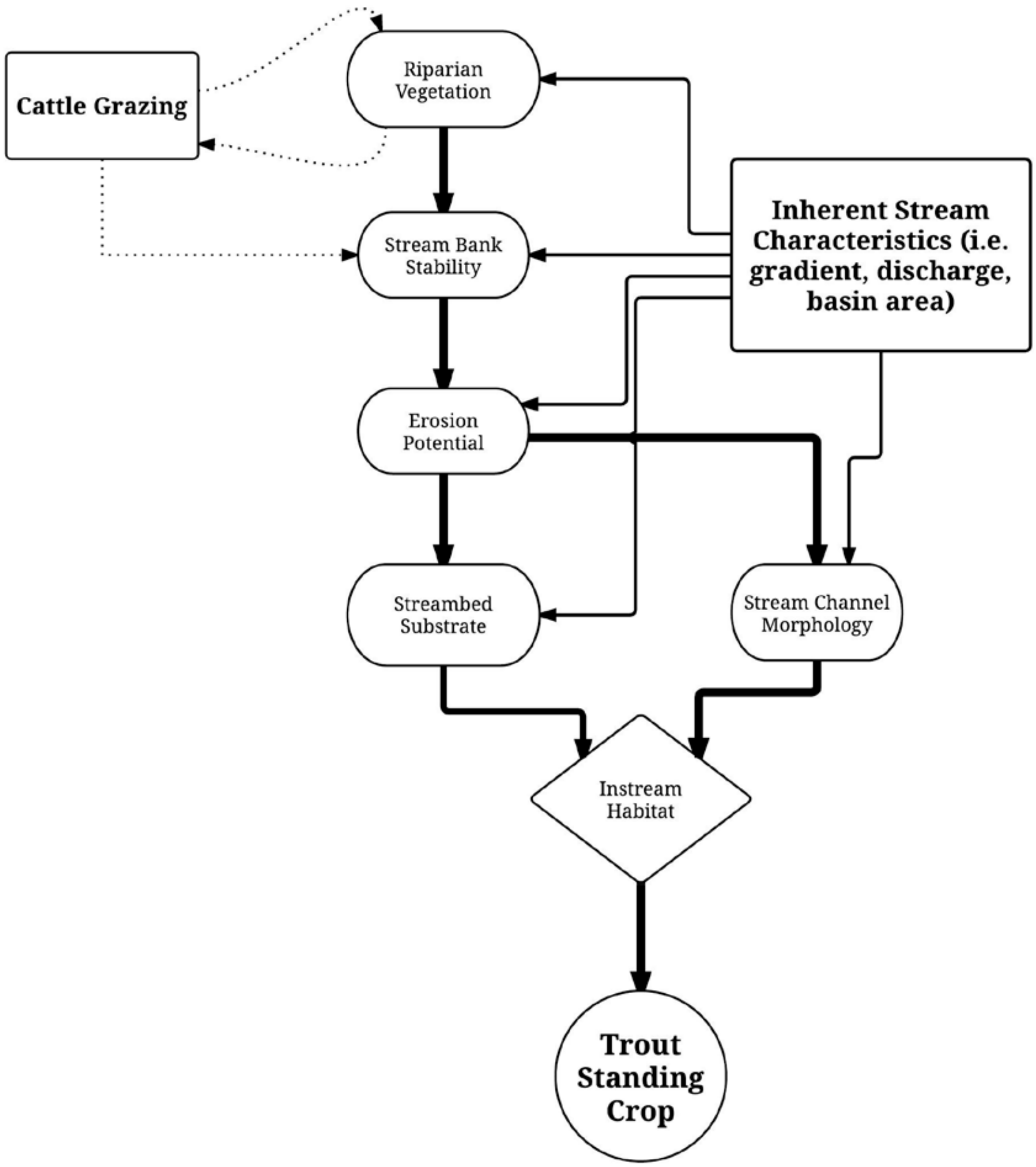

Figure 2. Conceptual model showing assumed relationships among stream habitat, watershed characteristics, and cattle grazing, and their potential effects on trout biomass.

RESULTS

Grazing Intensity

Scat counts were highly variable among sample sites within and among streams (Table 1), ranging from 0 to 676 scats/site. The majority of sites (88%) had fewer than 100 scats (Figure 3). Almost all scats were from domestic cattle (> 99%). Minimal evidence of horse grazing (≤ 2 scats) was present at two sample sites on North Fork Whitetail Creek. No evidence of horse grazing existed at any other sample site. Significantly more scats ($P < 0.001$) were counted in sample sites with higher proportions of open (grass) riparian habitat when livestock was free to range.

Stream Habitat and Grazing Intensity

Many of the habitat and watershed variables were correlated with each other (Table 2). A positive relationship existed between scat count and proportion of fine substrate, even when stream was included as a random effect in a mixed model ($P <$ 0.001; Table 3). The relationship was not significant ($P = 0.18$) when Roberts Creek Site One was removed (Table 3). This site was removed because it had an extremely high number of scats ($n = 676$) and had higher leverage on the regression than other sites. Although there was no reason to believe that the grazing intensity estimate was inaccurate, I acknowledge that the inferences of these models were significantly influenced by including this site. Therefore, all grazing intensity models were evaluated with and without Roberts Creek Site One. A negative relationship existed between scat count and area of undercut banks ($P < 0.001$; Table 3). This relationship was not significant (P = 0.20) when Roberts Creek Site One was removed (Table 3). No significant relationships existed between scat counts and other instream habitat variables (Figure 4; Table 3).

The PCA analysis indicated that the first component (PC1) explained about 54% of the total variability in the habitat and watershed data (Table 4). The second component (PC2) only accounted for about 15% of the variability and loaded heavily on two of the variables that were significantly related to scat counts (proportion fine sediment and area of undercut banks; Table 3; Figure 4). I only used PC1 as a covariate in linear mixed models that related scat counts to fish biomasses because this component explained most of the habitat variation in this dataset. I did not use PC2 because it loaded heavily on undercut banks and fine sediment, which were significantly related to scat counts (Table 3; Figure 4). Scat counts are included in the model as a covariate. PC1 was not significantly correlated to scat count (r = 0.19, P = 0.26) indicating that scat counts may account for the remaining variation in the data.

<u>Trout Biomass and Grazing Intensity</u>

There was a negative relationship between scat count and trout biomass, even after accounting for habitat variability at the site level using PC1 as a covariate and variability among streams and between years as random effects in a mixed model (P = 0.006; Table 5; Figure 5). This analysis indicated that there were significant differences in estimates of stream intercepts, but no significant differences in slopes among streams (P = 0.15). This relationship became marginally insignificant after the removal of Roberts Creek Site One (P = 0.091; Table 5). Random effect of year was not significant

(χ^2 = 0.136, P = 0.712), whereas random effect of stream was significant (χ^2 = 9.355, P = 0.002; Table 5).

Table 2. Pearson correlation matrix showing simple correlation coefficients between watershed and instream habitat variables (n = 25). Correlations with $P \leq 0.05$ are represented in bold. Variable names and corresponding numbers are: 1 - Proportion Fine Sediment, 2 - Basin Area (m^2), 3 - Elevation, 4 - Stream Order, 5 - Average Depth (cm), 6 - Average Wetted Width, 7 - Average Bank-full Width, 8 - Ln(Area of Large Woody Debris), 9 - Area undercut banks , 10 - Average Pool Depth, 11 - Width:Depth, 12 - Mean Summer Discharge, 13 - Percent Gradient.

	1	2	3	4	5	6	7	8	9	10	11	12	13
1	1	**-0.40**	0.05	-0.31	**-0.48**	**-0.43**	-0.37	**-0.43**	**-0.78**	**-0.42**	0.04	**-0.46**	-0.14
2	-	1	**-0.82**	**0.84**	**0.69**	**0.87**	**0.78**	0.22	0.34	**0.71**	0.27	**0.94**	**-0.66**
3	-	-	1	**-0.84**	**-0.45**	**-0.65**	**-0.67**	-0.15	-0.14	**-0.54**	-0.28	**-0.71**	**0.68**
4	-	-	-	1	0.51	**0.73**	**0.72**	0.38	0.18	**0.53**	0.28	**0.80**	**-0.46**
5	-	-	-	-	1	**0.76**	**0.55**	-0.05	**0.48**	**0.84**	-0.16	**0.66**	**-0.50**
6	-	-	-	-	-	1	**0.80**	0.10	0.36	**0.74**	**0.46**	**0.81**	**-0.66**
7	-	-	-	-	-	-	1	**0.40**	0.24	**0.58**	0.38	**0.63**	**-0.55**
8	-	-	-	-	-	-	-	1	0.27	0.01	0.03	0.19	0.18
9	-	-	-	-	-	-	-	-	1	0.38	-0.09	**0.42**	-0.13
10	-	-	-	-	-	-	-	-	-	1	0.03	**0.66**	**-0.59**
11	-	-	-	-	-	-	-	-	-	-	1	0.23	-0.35
12	-	-	-	-	-	-	-	-	-	-	-	1	**-0.56**
13	-	-	-	-	-	-	-	-	-	-	-	-	1

Table 3. Simple linear mixed effects models evaluating the relationships between scat count and stream habitat variables. Models used stream as a random effect to account for spatial autocorrelation of sites within the same stream. These relationships were evaluated with and without Roberts Creek Site One, which had a high scat count. * indicates significant random effect of stream ($P < 0.05$).

Explanatory variable	Response variable	Roberts Creek Site One included				Roberts Creek Site One removed			
		df	Estimate	*t*	*P*	df	Estimate	*t*	*P*
Scat Count	Fine sediment*	15	0.00061	4.489	< 0.001	14	0.00035	1.385	0.188
	Width-to-depth ratio	15	0.00006	0.401	0.694	14	0.00020	0.759	0.460
	Area undercut banks*	15	-0.00196	-6.055	< 0.001	14	-0.00058	-1.317	0.209
	Mean wetted width*	15	0.00208	0.619	0.545	14	0.00367	0.545	0.595
	Mean bank-full width*	15	0.00231	0.303	0.766	14	0.00177	0.121	0.906
	Large woody debris	15	-0.00336	-1.962	0.068	14	-0.0055	-1.803	0.093
	Mean depth*	15	-0.00010	-0.084	0.933	14	-0.00426	-0.189	0.852
	Mean pool depth*	15	-0.01100	0.391	0.701	14	-0.03433	-0.701	0.494

Table 4. Results from principal component analysis of habitat variables showing proportion of total variance explained and relative loadings that each habitat variable contributed to each principal component (PC). Habitat variables with loadings ≥ 0.3 for each component are shown in bold.

	PC1	PC2	PC3
Per cent variation explained	53.73	15.11	10.95
Proportion fine sediment	0.17	**0.33**	**-0.52**
Basin area	**-0.36**	0.06	0.01
Elevation	**0.30**	**-0.30**	0.11
Stream order	**-0.32**	0.23	0.13
Mean depth	**-0.30**	**-0.38**	-0.06
Mean wetted width	**-0.36**	-0.04	-0.08
Mean bank-full width	**-0.32**	0.14	0.19
Large woody debris	-0.13	0.25	**0.62**
Undercut banks	-0.15	**-0.55**	-0.11
Mean pool depth	**-0.31**	-0.23	-0.09
Width-to-depth ratio	-0.11	**0.39**	-0.17
Mean summer discharge	**-0.34**	-0.02	-0.03
Gradient	0.26	-0.11	**0.47**

Table 5. Additive linear mixed model with random effects of stream and sampling year and fixed effects of PC1, scat count, and trout biomass. Ln(trout biomass) is the response variable. This relationship was evaluated with and without Roberts Creek Site One which had a high scat count. Random effect of year (χ^2 = 0.136, P = 0.712) was not significant whereas random effect of stream (χ^2 = 9.355, P = 0.002) was significant.

Explanatory variable	Roberts Creek Site One included				Roberts Creek Site One removed			
	df	Estimate	*t*	*P*	df	Estimate	*t*	*P*
Scat Count	19	-0.00295	-3.165	0.006	18	-0.00271	-1.881	0.0919
PC1	19	0.08319	1.345	0.215	18	0.08236	1.314	0.2276

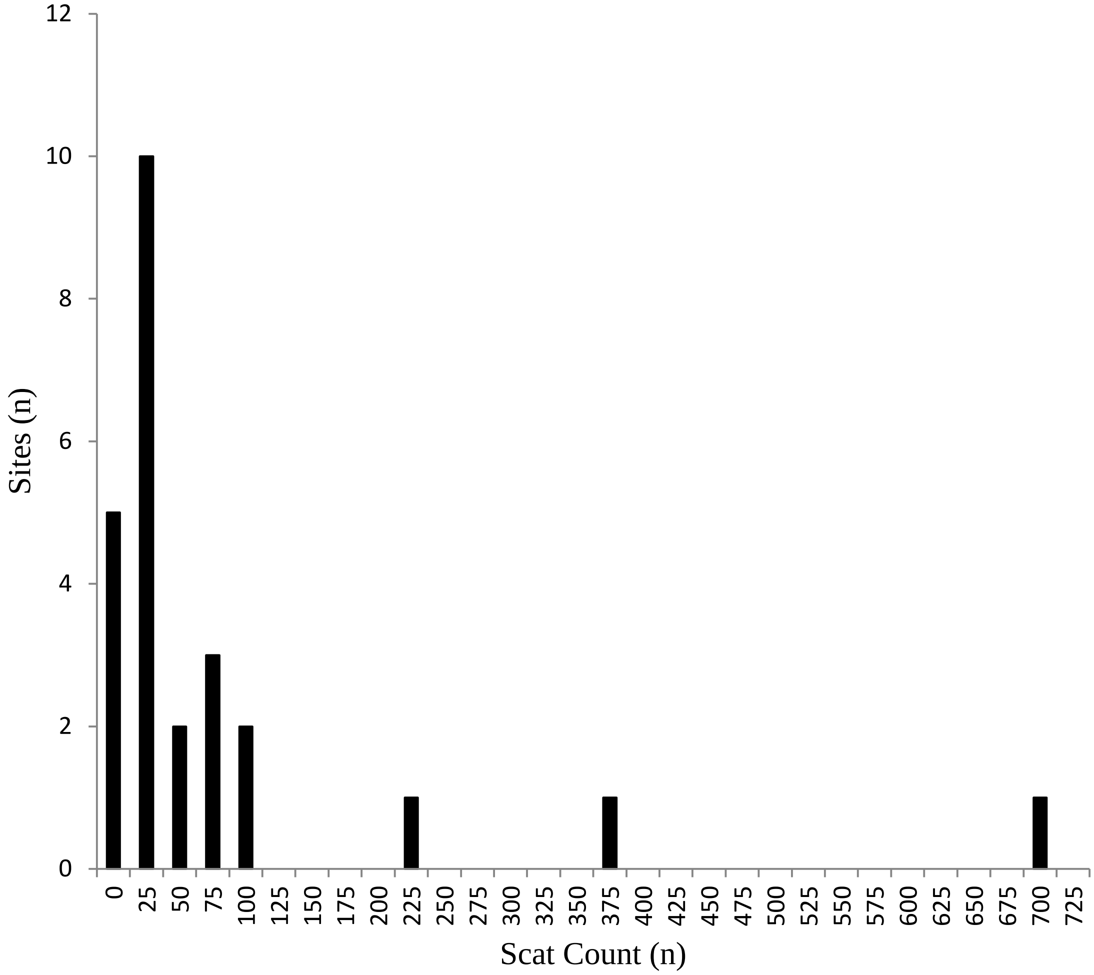

Figure 3. Frequency of scat count (n) by sample site (*n* = 25).

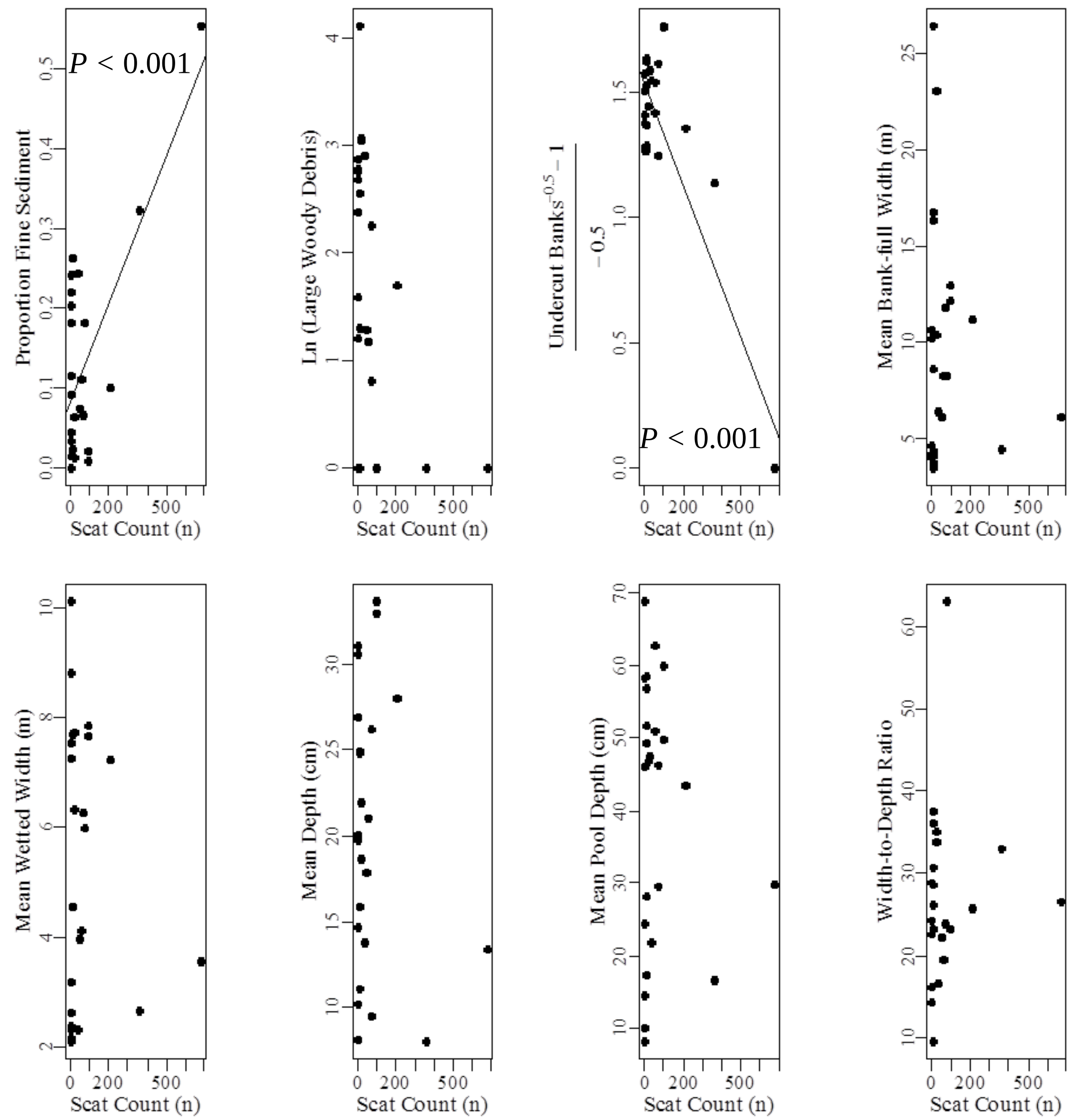

Figure 4. Observed relationships between instream habitat characteristics and grazing intensity (scat count) at all sites (n = 25). Trend lines and P-values are provided for significant simple linear mixed model regression relationships ($P < 0.05$) with fixed effect of scat count and random effect of stream.

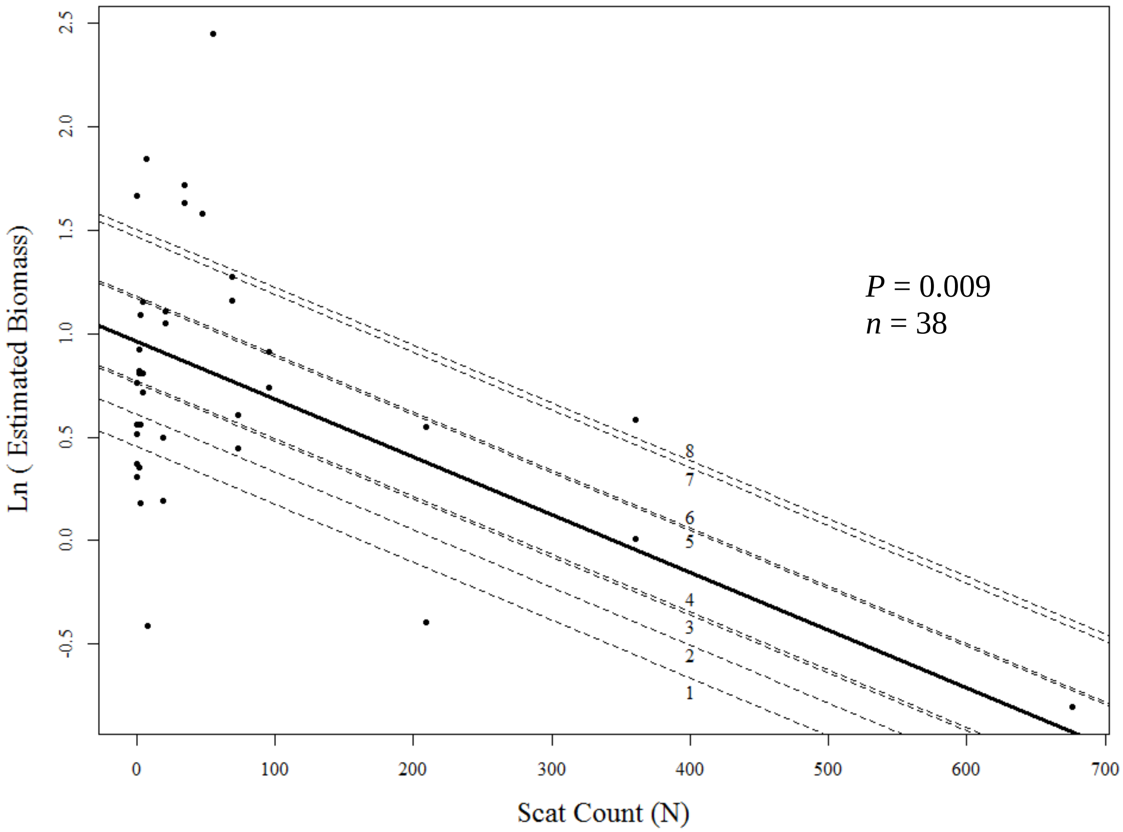

Figure 5. Linear mixed model showing relationship between ln(estimated biomass) and fixed effect of scat count (n = 38) and random effects of year and stream; dashed lines show individual stream regressions (n = 8) and the solid line shows the overall regression (P = 0.009). Each stream was allowed to have a random intercept, and there was no evidence for random slopes (P = 0.15). An individual regression could not be computed for Little Badger Creek because it only had one sampling site. Individual stream regressions are numbered; 1 - Dupuyer Creek, 2 - Deep Creek, 3 - North Fork Willow Creek, 4 - Middle Fork Lee Creek, 5 - Roberts Creek, 6 - South Fork Cut Bank Creek, 7 - East Fork Lee Creek, 8 - North Fork Whitetail Creek.

DISCUSSION

Grazing Intensity

Scat counts along riparian corridors were highly variable; both within and among streams (Table 1). I counted more livestock scats in sample sites with higher proportions of open riparian habitat. Scats I counted in areas of dense woody vegetation were primarily observed on livestock trails. Two previous studies documented that livestock use is highly variable between uplands and riparian areas (Smith et al. 1992; Cran et al. 1997). Similarly, riparian meadows were the most preferred habitat of domestic cattle in northeastern Oregon, despite their rarity (only 3 to 5% of the study area) and heavily forested sites were least preferred (Gillen et al. 1984). Variability of livestock use between riparian vegetation types has also been documented in the Katherine Creek drainage in northeastern Oregon where livestock utilization was 70% in riparian meadows, but only 18% within Black Cottonwood (*Populus trichocarpa*) dominated riparian habitats (Green and Kauffman 1995). The disproportionate use of riparian meadows has been attributed to higher plant productivity and succulence, gentler slope, and close proximity to a water source (Bryant 1982). This observed high variability in grazing intensities across the landscape and along streams highlights the need for a method such as scat counts to better quantify livestock grazing intensities when assessing effects of grazing on aquatic habitats and fish communities.

The potential for livestock grazing to be highly variable along riparian corridors also has important consequences for interpreting past grazing studies. Past studies that evaluated the effects of livestock grazing on fish and fish habitats have used block designs (grazed, ungrazed, grazing management regimes), comparisons of grazing exclosures to grazed stream reaches, or a combination of these approaches (Platts 1982; Keller and Burnham 1982; Kauffman et al. 1983b; Hubert et al. 1985; Rinne 1988; Platts 1991; Overton et al. 1994; Knapp and Matthews 1996; Rinne 1999; Clary 1999; Bayley and Li 2008). Sometimes livestock stocking rates (AUMs) are used to index grazing intensity categories (high, medium, low; Hubert et al. 1985; Clary 1999). All these studies assume grazing intensity is evenly distributed across treatment areas (pastures; Gillen et al. 1984). However, livestock are usually allowed to range freely throughout treatment areas. The spatial variability that I and others have observed in livestock grazing intensity suggests that the assumption of uniform grazing within grazing treatment areas made by these earlier studies may be invalid.

In addition, spatial arrangements of treatments in previous grazing studies (e.g., exclosures located downstream of grazed reaches; Rinne 1988) and legacy effects of historical livestock grazing could have influenced inferences made on how these treatments affected aquatic habitat and fish populations. Inferences from these studies are limited to observed responses after the removal of livestock and can overlook historical or current spatiotemporal factors that may have contributed to current conditions (Platts and Nelson 1985; Smith et al. 1993).

Using livestock scat counts as an index to quantify recent grazing intensity at specific sites of interest is practical. This method does not require any information on livestock stocking rates or grazing management strategies (i.e., AUMs, season of use, pasture locations). Rather, such counts provide a reasonable index of actual livestock use within a particular area over the past two years (Julander 1955; Abensperg-Traun et al. 1996, Vulliamy et al. 2006; Newbold and McMahon 2008). Livestock stocking rates might be related to scat counts, but this relationship should be tested. In addition, counts of livestock scats done at regular intervals (e.g., one to three years) along known-width riparian buffers could be used to index grazing intensity through time. I was unable to evaluate this in my short two-year study.

The number of livestock scats is dependent upon the length of time livestock are in a treatment area so the time when scat counts are made is important. I conducted scat counts for all my sample sites within a five-day period to minimize the probability that scat counts would differ based on livestock residence time. For this study, the relative number of scats I counted among the different sample sites should have provided a reasonable index of grazing intensity among sample sites because livestock grazing was season-long throughout the study area. Ideally, scat counts should be done near the time livestock are removed from a treatment area to ensure all scat from that grazing season are counted. If scat counts are to be used to assess grazing intensity over time, the scat count surveys should be done at the end of the grazing season each year. In my study previous year's scats made up a relatively small proportion of total scat counts (Table 1), probably due to their decomposition. Future efforts should be made to assess scat decomposition rates to evaluate contributions of older scats to total scat counts.

Initially, widespread horse grazing on the Blackfeet Reservation was thought to be a potential source of stream and riparian degradation because of their selection for riparian habitats (Cran et al. 1997). However, despite large numbers of feral and domestic horses on the Blackfeet Reservation, I found little evidence of horse grazing at my sample sites. Horse grazing was only detected along one study stream at an extremely low intensity. I speculate that the conspicuous high concentrations of horses observed along developed roads throughout the Blackfeet Reservation was related to both the fencing along highways and care given to these horses by the Blackfeet people. I was unable to find any studies that assessed potential differences between cattle and horse grazing. The lack of information on horse grazing is probably due to much smaller populations of horses found on both public and private land.

Stream Habitat and Grazing Intensity

I found that the proportion of fine sediment in stream substrates was significantly and positively related to scat counts, and there was a significant stream effect (Table 3; Figure 4). Sediment input has been found to be affected by discharge, watershed geology, stream velocity, and stream channel gradient (Brush 1961; Medina et al. 2005) and I found that proportion of fine sediment was negatively correlated with discharge, basin area, area of woody debris, and stream channel size, but not with gradient (Table 2). Rinne (1988) suggested that substrate composition varies spatially within stream networks and that high-intensity livestock grazing increase proportions of fine sediments. Proportions of fine sediments were significantly reduced after grazing stocking rates were reduced in Stanley Creek, Idaho (Clary 1999) and after season-long cattle grazing was eliminated from a stream in northwestern Nevada (Myers and Swanson 1996). Grazing exclosures had higher levels of fine sediment than grazed sections of Cherry Creek, Wyoming, but lightly grazed reaches had higher levels of fine sediment than heavily grazed reaches in Pete Creek, Wyoming (Hubert et al. 1985). Grazing exclosures had higher proportions of small substrate particles than grazed stream reaches in the Golden Trout Wilderness, California (Knapp and Matthews 1996), even where exclosures were located upstream of grazed sampling reaches. These conflicting findings highlight the difficulty in assessing grazing effects on stream substrate.

I found that the area of undercut banks was negatively related to scat counts, and there was a significant stream effect (Table 3; Figure 4). The significant stream effect indicates that watershed characteristics such as gradient, soil type, substrate composition, and runoff regimes, probably influence formation of undercut banks, making quantitative assessment of grazing effects on undercut banks difficult. In a few previous studies livestock grazing was negatively correlated with undercut banks (Kauffman et al. 1983b; Overton et al. 1994; Myers and Swanson 1995; Knapp and Matthews 1996). However, other studies were not able to document a statistically significant relationship between grazing and undercut banks. Slightly higher, but not statistically significant, levels of bank overhang existed in grazed reaches than in grazing exclosures in Golden Trout Wilderness streams in California (Knapp and Matthews 1996). Amounts of undercut banks were not significantly different between grazed and ungrazed sections of Rock Creek, Montana (Gunderson 1968).

Previous studies have found relationships between livestock grazing and large woody debris (Duff 1977; Marcuson 1977), W:D ratios (Duff 1977; Marcuson 1977; Kauffman et al. 1983b), and stream depths (Bryant 1982; Hubert et al. 1985; Stuber 1985, Overton et al. 1994; Bayley and Li 2008), but I was unable to quantify significant effects of scat counts on any of these variables. I found a marginally significant relationship between grazing intensity and large woody debris (Table 3). Riparian communities at all sampling locations were dominated by Quaking Aspen (*Populus tremuloides*), Black Cottonwood, and various willow species. However, the communities varied by seral stage and some communities had higher proportions of old-growth woody riparian vegetation. This factor may have regulated the size and abundances of LWD recruitment. Additionally, the relationship between grazing and LWD may be complicated by livestock avoiding areas of dense woody vegetation. A lack of LWD at sites with high-intensity grazing may coincide with stream reaches where woody riparian vegetation is naturally sparse or absent. Moreover, the effects of livestock grazing on riparian community dynamics and seral succession probably become more apparent over long periods of time (Green and Kauffman 1995). This relationship could be assessed if long-term data is available.

Trout Biomass and Grazing Intensity

Estimated biomass of trout was significantly and negatively related to scat counts, even after accounting for site habitat characteristics, stream, and year (Table 5 and Figure 5). Expected mean biomass with no grazing (scat count = 0) was estimated from the mixed model to be 2.63 g/m^2, whereas expected biomass at the highest observed grazing intensity (scat count = 676) was 0.36 g/m^2 (95% CI = 0.11 – 0.50 g/m^2). The results of this model indicate expected biomass (g/m^2) over the range of grazing intensities observed in this study (0-676 scats) was reduced by 86% (95% CI = 81 - 96%). My results are consistent with previous findings that high stocking rates of livestock along streams suppresses trout biomass (Gunderson 1968; Marcuson 1977; Hubert et al. 1985; Knapp and Matthews 1996; Bayley and Li 2008), but I actually quantified grazing intensities in this study.

Although trout biomasses were negatively related to grazing intensity, the mechanisms contributing to this relationship are not entirely clear. Livestock grazing probably affects trout biomass indirectly by altering instream habitat. I was able to detect a significant negative relationship between grazing intensity and the amount of undercut banks. Instream cover has been widely associated with trout density and standing crop (Boussu 1954; Lewis 1969). Trout prefer undercut bank habitat and avoid bare and collapsed banks (Matthews 1996). Undercut banks have been shown to affect growth, biomass, and survival of trout (Mathews 1996).

High levels of fine sediment in stream substrates can directly and indirectly affect salmonid biomasses. Fine sediment can reduce spawning success and recruitment by asphyxiating embryos in spawning redds (Magee et al. 1996). Fine sediment can also reduce abundances and biomasses of benthic invertebrates by reducing the amount of interstitial habitat among large substrate particles (Ryan 1991). These effects can reduce trout survival and growth, resulting in lower abundances and biomasses (Suttle et al. 2004).

A direct effect of livestock grazing on salmonids through trampling of redds causing egg mortality has recently been proposed (Gregory and Gamett 2009; Peterson et al. 2010). Redd trampling can cause negative population growth (Peterson et al. 2010).

Moreover, destruction of simulated redds appeared to be related to cattle stocking density (Gregory and Gamett 2009).

Conclusions

In conclusion, scat counts provided a practical method for indexing the intensity of livestock grazing along stream channels. I recommend using scat counts in future livestock grazing studies and as a potentially valuable method for monitoring livestock grazing intensity through space and time. I recommend conducting scat counts at the end of the grazing season to fully assess grazing intensity for that year. I also recommend further evaluation of this technique to assess longevity of scats and to develop relationships between livestock stocking rates and scat counts. My ability to reach strong statistical conclusions from these data was affected by the relatively small number of sample sites, lack of randomization for sample site selection, and a limited number of sites that had high-intensity grazing. Nevertheless, higher grazing intensities were negatively related to the amount of undercut banks and positively correlated with the proportion of fine sediments in the streambed. Higher grazing intensities resulted in lower trout biomasses, even after accounting for site habitat, stream, and year effects.

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