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PLANT COMPOSITION AND EROSION POTENTIAL OF A GRAZED WETLAND IN THE SALMON RIVER SUBBASIN, IDAHO

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ABSTRACT.—Wetlands are dynamic habitats with many unique, important functions including filtering sediments and providing diverse habitats for fish and wildlife. Wetlands in the western United States are particularly important because they offer habitat for a number of protected runs of endangered fish species. Historically, livestock grazing has altered wetland and riparian area form and function by facilitating exotic species invasions, altering spatial heterogeneity of vegetation, and increasing erosion. In this study we examined vegetation structure and erosion potential in a wetland meadow exposed to unregulated grazing along Deer Creek in the Salmon River subbasin, Idaho. We characterized the vegetation composition and structure within the study area and attempted to assess potential erosion conditions using the Revised Universal Soil Loss Equation (RUSLE), an empirical approach developed by the U.S. Department of Agriculture–Agricultural Research Service (USDA-ARS). We found no significant spatial variability in species richness and noted a moderate number of exotic species in the total plant composition. Plant cover was higher near slightly entrenched banks, indicating that uncontrolled livestock were primarily occupying gently sloped streambanks and the interior of the meadow. Based on current vegetation composition and RUSLE results, uncontrolled grazing may be negatively impacting the study area. If uncontrolled grazing were excluded or carefully managed in the wetland meadows of the upper portion of the Deer Creek watershed, a reduction in excess sediments to Deer Creek may occur.

Key words: wetland, species diversity, species richness, water erosion, geographic information systems, erosion model, RUSLE, grazing.

Wetlands are among the most diverse and dynamic habitats on earth (Naiman and DéCamps 1997). The high soil moisture and nutrients found in wetlands often facilitate highly productive and biologically diverse plant communities, which provide abundant habitat for fish and wildlife (Fitch and Adams 1998, Jansen and Robertson 2001). Vegetation plays a vital role in wetland ecosystems by creating shade cover, which helps to maintain cool water temperatures (Platts 1981), and by providing organic material to heterotrophic stream organisms (Kauffman and Krueger 1984). Disturbances that alter vegetation structure and organic inputs to streams can significantly impair stream habitats—a major concern in watersheds and wetlands in the Pacific Northwest. Wetland plants are critical to maintaining productive fish habitats because they stabilize streambanks (Kauffman and Krueger 1984, Fitch and Adams 1998) and reduce erosion and sedimentation through the vegetative filtering of sediments (Davis 1977). Furthermore, disturbances have the potential to alter the spatial distribution of vegetation, which can then affect biotic and abiotic interactions (e.g., food chain and soil properties; Adler et al. 2001).

Uncontrolled cattle grazing can be detrimental to riparian and wetland communities (Behnke and Zarn 1976, Carothers 1977, Fitch and Adams 1998). Cattle exhibit a strong preference for riparian zones and tend to congregate in those areas because they offer ample water (Platts 1981, Jansen and Robertson 2001, Martin and Chambers 2001), shade, cooler air temperatures, and abundant quality food (Kauffman and Krueger 1984, Fleischner 1994, Belsky et al. 1999). When grazing in riparian areas livestock may impact several in-channel processes. For example, researchers have noted significant impacts of heavy livestock grazing on channel morphology, including changes in channel depth, channel width, and channel capacity (Myers and Swanson 1992, Clary 1999, Nagle and Clifton 2003). By observing long-term channel changes due to removal of

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livestock grazing, Nagle and Clifton (2003) discovered substantial decreases in mean width and width-to-depth ratio and increases in mean depth and hydraulic radius. Channel bottom embeddedness may also decrease with the removal of livestock grazing pressures in riparian areas (Clary 1999).

Defoliation of plants from livestock grazing has caused high mortality of certain native plant species, leading to reductions in native biological diversity (Kauffman and Krueger 1984, Fitch and Adams 1998, Clary and Kinney 2002). Riparian plant communities are further threatened when grazing promotes exotic plant species that displace native species (Fleischner 1994, Naiman and DéCamps 1997, USDA 1999). Exotic species tend to prefer disturbed soils and overgrazed areas (D'Antonio and Vitousek 1992, Asher and Spurrier 1998) and have the potential to disrupt plant and animal community dynamics. Uncontrolled grazing by livestock can also alter soil structure by causing soil compaction (Clary 1995) and exposure of bare soil; both contribute to increases in soil erosion and sedimentation (Crumpacker 1984, Smith and Rushton 1994, Ford and Grace 1998). Excessive sedimentation in a stream deteriorates fish-spawning habitat by sealing preferred nest areas (Keller and Burnham 1982). Streambank erosion may remove vegetation, thereby causing increased water temperature, which degrades or eliminates crucial fish habitat (Platts 1981).

Plant community composition and structure can be measured in various ways. The use of diversity indices is 1 way to examine plant species richness and species evenness. One such index is the Shannon-Wiener index, which accounts for both species richness and evenness (Magurran 1988). The Shannon-Wiener index is based on the proportion of abundance of different species (Magurran 1988) and is sensitive to variability represented by the number of species in the community (Risvold and Fonda 2001). The value of the index usually falls between 1.5 and 3.5 and rarely exceeds 4.5 (Magurran 1988). A high index value signifies a greater number of species than a low value (Magurran 1988).

Plant community structure can be characterized by assessing the aerial percent cover of vegetation and breaking it down into specific life form categories (Brooks and Matchett 2003). The cover percentage can help to identify areas prone to erosion (Clary and Kinney 2002) and can be used to compare abundances of species of widely different life forms because it is not biased by the size and distribution of individuals (Floyd and Anderson 1987). A vigorous herbaceous plant community holds the soil in place (Clary and Kinney 2002) thereby increasing resistance to erosion and streambank slough-off; however, the removal of vegetation by excessive grazing can substantially increase soil erosion (Crumpacker 1984) and lead to elevated sedimentation in streams (Clary and Kinney 2002).

In addition to the management practices of an area, other factors can influence soil erosion. Amount of precipitation, topographic conditions (length, steepness, and shape of land slope), and conditions of vegetative cover all impact potential for erosion. The Revised Universal Soil Loss Equation (RUSLE), an empirical model for water erosion developed by the USDA-ARS, uses all the aforementioned factors to estimate average annual soil loss from cropland, rangeland, and forests (Renard et al. 1997). Compared to more complex, process-based erosion models, such as the USDA-ARS Water Erosion Prediction Project (WEPP; Laflen et al. 1997), RUSLE requires fewer input data and is less complicated to use. Integration of RUSLE and a geographic information system (GIS) for watershed-scale, distributed erosion modeling has been attempted in numerous studies (Hamlett et al. 1992, Yitayew et al. 1999, Fernandez et al. 2003). Under typical conditions, RUSLE and WEPP may give comparable results (e.g., Elliot 2001). Nonetheless, due to its empirical nature, RUSLE presents several major limitations. RUSLE does not explicitly model hydrological processes and therefore does not differentiate between types of runoff (infiltration-excess versus saturation-excess) and their impacts on erosion. Furthermore, RUSLE does not consider gully and channel erosion, making its applications to large watersheds problematic. RUSLE was developed primarily with data collected from agricultural lands, and its application and testing under rangelands and forestlands have been lacking. Consequently, caution should be exercised in using RUSLE.

The Deer Creek watershed, located in the Salmon River subbasin in western Idaho, contains numerous streams and wetlands valuable to salmonids and other wildlife. Roads and cattle grazing impact many streams and wetlands in the subbasin. In 1998 Deer Creek was listed by the state of Idaho as impaired due to elevated sedimentation under section 303(d) of the Clean Water Act (IDEQ 1998). Habitat development for fish and wildlife with diverse and native vegetation and sustainable water quality for salmonids is of high interest to federal and state politicians, regulatory and land management agencies, Native American tribes, and private individuals.

The purposes of this study were to examine the existing vegetation and assess soil erosion potential in a mountain wetland meadow in the Deer Creek watershed and to facilitate decision-making by the Nez Perce tribe by providing potential consequences of various cattle management strategies. Three specific objectives were to (1) characterize species diversity and vegetation structure within the study area, (2) assess the erosion potential of the area using RUSLE with a GIS, and (3) synthesize information about the vegetation and erosion conditions in order to make recommendations on potential management alternatives for land managers. The RUSLE model was chosen over a process-oriented model, WEPP, primarily for its simplicity and reasonable data requirements. Additionally, this study focused on assessment of the long-term potential of erosion and sedimentation as affected by livestock grazing rather than on identification of flow and sediment transport pathways, which can only be achieved with a physically based model such as WEPP. The 3rd reason RUSLE was selected over WEPP was a practical one. At the time of this study, water balance routines of the WEPP model were inadequate, particularly when used for forested or rangeland settings where canopy interception and saturation-excess runoff may be substantial (W.J. Elliot, Rocky Mountain Research Station, USDA Forest Service, personal communication, 2003).

STUDY SITE

The study area was approximately 24.3 ha (60 acres) of meadow within a wetland complex in the upper portion of the Deer Creek watershed in Nez Perce County, Idaho (Fig. 1). Deer Creek is a 4th-order tributary that flows into the Salmon River at river mile 13.8. Elevation of the study area ranges from 1313 m to 1353 m (USGS 2001). Mean monthly temperatures for this region vary from –7.2°C to 25.4°C, with highest temperatures occurring in August and the lowest in January. Average annual precipitation for the study area is 634 mm, with the majority falling between March and June and 39% of the total precipitation falling as snowfall (WRCC 2002).

The headwaters of Deer Creek are located in the Craig Mountains, which control flow of water through Deer Creek from spring snow melt. In the study site, Deer Creek has been classified (e.g., Rosgen 1994) as a C3–C4 channel with an average gradient of 1%. The upper portion of Deer Creek, including the study site, suffers from poor width-to-depth ratios, low pool-to-riffle ratios, and excessive sediment deposition (USDI BLM 2000).

The lithology of the study area consists primarily of mafic metavolcanic rocks (ICBEMP 1995). Most of the soils in the study area are hydric, with slow to moderate permeability (USDA 1995). The Deer Creek wetland meadows are similar to the intermountain meadows described by Daubenmire (1942) and Garrison et al. (1977). Although no direct measurements of water table were recorded, the meadow has standing water throughout the spring and into the summer months, and patchy areas of saturation in the late summer months. The Deer Creek wetland meadow satisfies the 3-level definition of a wetland as proposed by Mitsch and Gosselink (2000). First, water continues to be observed year after year at the surface and within the root zone of the area. Second, soils have been surveyed as hydric, wetland type soils (USDA 1995). Third, the study area supports numerous species of vegetation adapted to survival in wetland conditions.

Historically, the riparian vegetation in the study region was dominated by graminoids and forbs, some of the major species being *Camassia quamsh, Wyethia amplexicaulis, Poa ampla,* and *Polygonum bistortoides* (Daubenmire 1942). The current wetland meadow is dominated by forbs blended with few sedges and grasses that are all listed as wetland indicators by the USFWS (Table 1). The Salmon River subbasin also includes subalpine meadows, broadleaf riparian vegetation, and shrub-dominated riparian vegetation (Idaho GAP Analysis 1998).

Disturbances within the area primarily originate from unregulated livestock grazing and

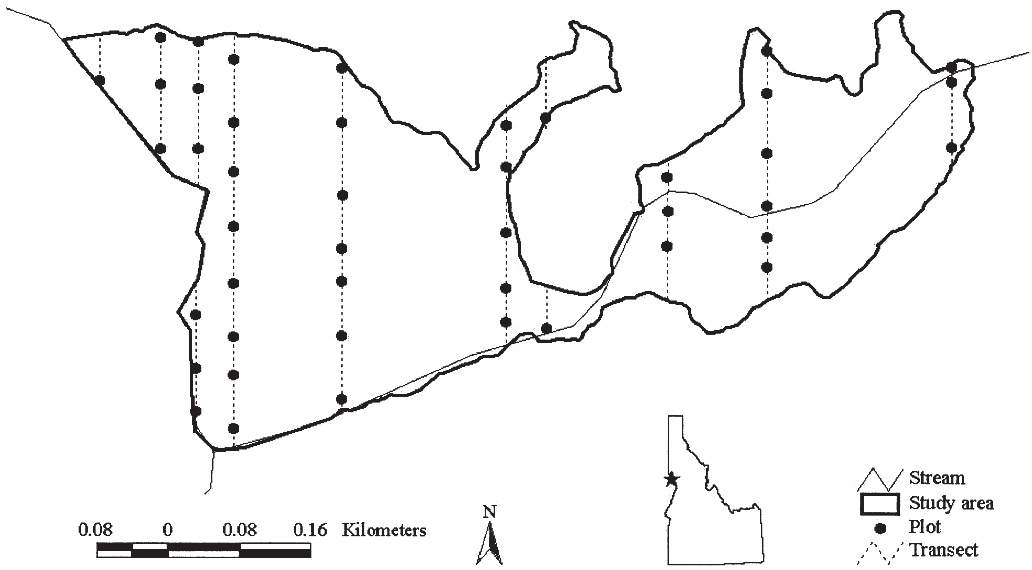

Fig. 1. Boundary of the study area in the Deer Creek watershed, Idaho, including Deer Creek, the west border of the study site, its tributary, and all sampled transects (---) and plots (•).

roads. There are 2 roads through the study area. These roads are not maintained, yet are still used by trucks and off-road vehicles. Livestock grazing in the upper Deer Creek watershed began in the 1880s (Craig Johnson, Bureau of Land Management, personal communication, 2002). Grazing has ceased in the lower watershed, but unpermitted grazing continues in the upper watershed (IDEQ 1998). The excessive sediments entering Deer Creek may occur for 2 reasons: (1) the wide floodplains that border the creek in the upper watershed provide ideal livestock grazing areas and (2) the upper watershed is predominantly owned by the Nez Perce tribe and is not monitored frequently; therefore, grazing prohibitions are not enforced and livestock continue to graze unrestricted. The Nez Perce tribe is currently developing a long-term management strategy, recognizing that uncontrolled cattle grazing in the area may cause problems to both ecological and cultural resources (Darin Saul, Tribal Liaison, personal communication, 2003).

METHODS

Plant Community Survey

We used a double-stratified random sampling scheme (Krebs 1999) to select transects and plots in the field survey. Transects were randomly placed within intervals of 400 m, and, along each transect, plots were randomly placed at intervals of 60 m. We used a GPS unit and compass to position plots in the field.

Thirty-six one-eighth-m^2 sample plots were surveyed over the entire study area. The sample size was determined by referring to previous wetland biomass studies using a significance level of 0.05 and a margin of error of 30 (Ford and Grace 1998, Atkinson and Cairns 2001, Farnsworth and Ellis 2001). All plants in the plots were censused to identify rare, threatened, or endangered species according to the species list in Guard (1995). Percent cover of vegetation within each plot was visually estimated according to Daubenmire's cover class scheme: 0%, 1%, 5%, 25%, 50%, 75%, and 95% (Daubenmire 1959). The vegetation was further categorized as bare ground, grasses, forbs, or shrubs. All field sampling was conducted from 19 June through 22 June 2002.

After identifying plant species, we clipped and dried all the vegetation from every plot to determine aboveground biomass.

Determination of the Shannon-Wiener Index

The plant community diversity of the study area was characterized by calculating the

TABLE 1. List of all species collected from the study site at Deer Creek, Idaho. Abundance, aboveground biomass, wetland indicator status, nativeness, and growth form are given.

Species	Abundance (individual · m^{-2})	Biomass (g · m^{-2})	Wetland indicator status[a]	Exotic (E) or Native (N)	Annual (A) or Perennial (P)
Achillea millefolium	37.1	5.5	FACU	E	P
Aconitum columbianum	14.0	9.5	FACW	N	P
Amelanchier alnifolia	0.2	5.1	FACU	N	P
Antennaria corymbosa	0.5	0.3	FAC–	N	P
Camassia quamash	0.4	0.4	FACW	N	P
Carex feta	13.6	4.6	FACW	N	P
Castilleja tenuis	1.3	0.5	FACU–	N	A
Fragaria vesca	20.2	2.7	NI	N	P
Frasera fastigiata	1.5	29.0	NI	N	P
Galium boreale	1.1	0.2	FACU	N	P
Geranium viscosissimum	5.6	2.8	FACU+	N	A or P
Geum triflorum	12.9	16.3	FACU	N	P
Lotus corniculatus	3.5	5.0	FAC	E	P
Lupinus rivularis	9.3	5.4	FAC	N	P
Maianthemum stellatum	0.4	0.1	FAC–	N	P
Mentha piperita	2.5	1.1	FACW+	E	P
Phalaris arundinacea	53.3	16.6	FACW	E	P
Plantago major	6.9	0.6	FAC+	N	P
Potentilla gracilis	31.3	11.3	FAC	N	P
Ranunculus occidentalis	3.5	1.3	FACW	N	P
Senecio triangularis	11.8	7.7	FACW+	N	P
Taraxacum officinale	5.3	0.6	FACU	E	P
Thalictrum alpinum	0.9	0.1	FACW–	N	P
Trifolium pratense	4.7	0.7	FACU	E	P
Veratrum californicum	2.2	78.0	FACW+	N	P

[a]Indicator status definitions: FACW = facultative wetland, usually occurs in wetlands (estimated probability 67%–99%), but occasionally found in non-wetlands; FAC = facultative, equally likely to occur in wetlands or non-wetlands (estimated probability 34%–66%); FACU = facultative upland, usually occurs in non-wetlands (estimated probability 67%–99%), but occasionally found in wetlands (estimated probability 1%–33%); NI = insufficient information was available to determine an indicator status (USFWS 1988).

Shannon-Wiener index for each plot. The index (H) was calculated based on live shoot biomass according to Smith (1940) and Ram et al. (1989):

$$H = 3.322[\log_{10}N - 1/(N\Sigma N_i \log_{10}N_i)]$$

where N is the total shoot biomass and N_i is the shoot biomass of species i.

Estimation of Soil Erosion Potential

The potential annual soil loss was estimated for the study area using RUSLE and ArcView GIS (ESRI 1996). RUSLE is expressed as $A = RKLSCP$, where A is the average soil loss per unit area per year (t · ha^{-1} · yr^{-1}), R is a rainfall-runoff erosivity factor, K is a soil erodibility factor, L is a slope-length factor, S is a slope steepness factor, C is a cover management factor, and P is a support practice factor.

The rainfall-runoff erosivity factor quantifies the effect of raindrop impact and reflects the amount and rate of runoff associated with the rain (Renard et al. 1997). A modified equation for estimating the rainfall and runoff erosivity for the unique climatic conditions of the Pacific Northwest (USDA-ARS 2003) was used. Two different zones for R (635 mm and 686 mm of annual precipitation) were discovered within the study area. These zones were obtained from Oregon State University's Spatial Climate Analysis Service (OSU 2002) in the form of a 10-m-resolution precipitation map representing a 30-year (1961–1990) long-term average.

The soil erodibility factor reflects the soil's resistance to the combined effect of rainfall and runoff on soil loss (Renard et al. 1997). The soil erodibility coverage at 30-m resolution was obtained from the USDA NRCS SSURGO database (USDA 1995) and was converted to a 10-m resolution map by using the nearest neighbor interpolation (Lillesand and Kiefer 1994). The K-factor map showed 3 different values (0.20, 0.37, 0.43) within the study area.

Slope length is the horizontal distance from the origin of flow to the point where either deposition begins from decreased slope gradient or runoff becomes concentrated in a defined channel (Wischmeier and Smith 1978). Slope steepness accounts for the influence of slope gradient on erosion (Renard et al. 1997). *LS*, the combined form of the *L* and *S* factors, was calculated in ArcView using a 10-m digital elevation model (DEM) for the study area following Fernandez et al. (2003). A modified slope factor equation developed for the Pacific Northwest region (McCool et al. 1993) was used. Furthermore, the upper limit on slope length was set to 150 m (corresponding to 15 grid cells), because, in reality, deposition or concentrated flow would almost always occur when a slope becomes long (Fernandez et al. 2003).

The cover management factor is the ratio of soil loss from an area with specified cover and management to soil loss from an identical area in tilled continuous fallow (Renard et al. 1997). The *C* factor takes into account the aboveground canopy effects, surface effects (e.g., ground cover and surface roughness), and belowground effects (e.g., root mass; Haan et al. 1994). Field data from each plot was averaged to obtain the overall percent canopy cover, percent ground cover, and type and height of canopy in the study area. A *C* value of 0.09 representing current, averaged conditions was used for the entire study area, according to Haan et al. (1994).

The *P* factor represents the ratio of soil loss with a support practice such as contouring or strip cropping to soil loss with straight-row farming up and down the slope (Renard et al. 1997). The entire study area consists of wetland meadow only and is not farmed; therefore, a *P* factor of 1 was used.

The potential annual erosion for the study area was calculated by multiplying the RUSLE variables within ArcView, resulting in a potential annual erosion map.

Statistical Analysis

For statistical analysis we considered each plot as an independent data point. We made scatterplots and performed simple linear regression ($\alpha = 0.05$). *H* value, percent cover of vegetation, *LS* factor, and potential annual water erosion (*A*) were response variables, and distance to stream was the predictor variable to show how livestock impact areas near to and far from the stream. Regression analysis was performed to distinguish any spatial relations of (1) *H* and the percent of vegetation cover versus the distance from the stream and (2) slope steepness and erosion potential as affected by distance from the stream. Additionally, Pearson's correlation analysis was performed for the following variables: *H* value, percent of vegetation cover, *LS* factor, potential annual erosion (*A*), and distance from stream.

RESULTS

Among the 25 species recorded in the 36 plots, no rare, threatened, or endangered species were found, although 6 of the 25 species (14% of the biomass) in the wetland meadow were exotic (Table 1). Distance from the stream was not a significant determinant of the *H* value or percent plant cover. However, the *LS* factor and the annual soil loss potential (*A*) versus distance from stream show an increasing trend (i.e., the closer to the stream, the higher the *LS* and *A* values). This observation was confirmed by the regression analysis.

There was no significant difference in diversity among plots based on the Shannon-Wiener index. Species richness was uniformly distributed with distance from the stream. Consistent with the regression result, correlation analysis showed a significant relationship between (1) the *LS* factor and distance from stream ($P = 0.0005$) and (2) *A* and distance from stream ($P = 0.0026$). In addition, correlations between the *LS* factor and percent cover ($P = 0.0114$), *A* and percent of cover ($P = 0.0107$), and *LS* and *A* ($P < 0.0001$) were also significant (Table 2).

DISCUSSION

No records of intermountain wetland meadow vegetation composition were found in the literature from pregrazing years. The vegetation description of wetland meadows in this region by Daubenmire (1942) suggests that most meadows were historically dominated by forbs and grasses that are currently poorly represented in the plant community (Table 1). Vale (1975) examined very old journal documents and reported that stands of grass were confined to the wet bottom areas in the 1850s. After studying the vegetation composition of riparian areas along the middle Snake River in a National Wildlife Refuge, Johnson et al. (1995) found adjacent wetland meadows to be dominated by perennial herbaceous plants; the grazing history of this site was not given. From studying vegetation composition in grazing

TABLE 2. Pearson's correlation analysis with 5 variables: species richness calculated using the Shannon-Wiener Index (*H*), percent of vegetative cover using Daubenmire's cover class scheme (cover), length and slope factor (*LS*) derived from the RUSLE analysis, potential annual erosion (*A*) from the RUSLE analysis, and distance from stream (distance) measured in ArcView. Shown are the correlation coefficients and *P*-values (in parentheses). Relationships that are significant at $\alpha = 0.05$ are marked with an asterisk (*).

Parameter	*H*	Cover	*LS*	*A*	Distance
H	1.000				
Cover	0.8002*	1.000			
	(<0.0001)				
LS	0.2864	0.3781*	1.000		
	(0.0594)	(0.0114)			
A	0.2622	0.3809*	0.9400*	1.000	
	(0.0856)	(0.0107)	(<0.0001)		
Distance	0.1228	–0.0536	–0.5010*	–0.4428*	1.000
	(0.4269)	(0.7295)	(0.0005)	(0.0026)	

exclosures, Green and Kauffman (1995) suggested that the number of *Achillea millefolium*, *Taraxacum officinale* and other exotics decreases through time.

The vegetation composition of the Deer Creek study site would likely be different without its long history of unregulated grazing and could change again with the removal of grazing or improvement in grazing management. Daubenmire (1942) states that, under heavy grazing, the vegetation of mountain meadows turns into heavy unpalatable forbs. If grazing is halted, a shift in species composition and dominance may occur following Grime's (1979) competition model. Species that are adapted to resource-abundant habitats with disturbances, such as grazing, are displaced by competitive species that tolerate decreased amounts of disturbance (Grime 1979). The competitive ability of a plant may be altered when leaf area is subject to predation (Grime 1979), and some species respond with renewed growth when defoliation occurs. According to Grime's model, species that respond positively to grazing pressures will be replaced when grazing is halted. If grazing stopped in the Deer Creek meadow, an example of Grime's competition model could be *Carex rostrata* replacing *Achillea millefolium*. *Carex rostrata* is a species that thrives without disturbances; is fast growing, tall, and dense; and produces much litter (Green and Kauffman 1995). In contrast, *A. millefolium* thrives in disturbed areas.

Unregulated grazing can destabilize plant communities by enhancing the spread and establishment of invading species (Hobbs and Huenneke 1992, Fleischner 1994). Grazing causes changes in the species dominance in a number of plant communities (Collins et al. 1998), particularly when grazing pressures prevent establishment of less competitive native species seedlings (Carothers 1977) and more competitive exotic species are allowed to flourish. Many exotic species, such as *Phalaris arundinacea*, respond positively to grazing through renewed growth and expansion of new shoots (Grime 1979). If grazing is reduced in the Deer Creek study site, exotic species should respond by declining; however, if grazing continues, many more exotic species may flourish. Exotic species threaten native species and ecosystems in a multitude of ways (D'Antonio and Vitousek 1992) and are a major challenge to the effort of restoration of riparian systems (Young 1994, Dobkin et al. 1998).

The species composition of the wetland meadow under study showed little spatial heterogeneity, as indicated by the nonsignificant relationship between *H* and the distance from the creek. While we are not definite about conditions of the plant community before grazing, because few ungrazed meadows presently exist in the area, the lack of spatial relationships between community composition and distance to the stream is frequently an indicator of changes in habitat diversity as a consequence of grazing (Smith 1940, Adler et al. 2001). Adler et al. (2001) submit that, when grazing is homogenous across a heterogeneous vegetative landscape, the result may be a decrease in spatial heterogeneity of the vegetation. The result of such a scenario is homogenous grazing leading to spatially homogenous vegetation. However, it is worth noting that management decisions based solely on diversity measurements may be equally inappropriate simply because of the need to conserve species-poor habitats (Magurran 1988). Salt marshes generally have fewer species than freshwater marshes, but they still provide important ecosystem functions worth protecting. If managers chose areas to protect only by measuring diversity, areas with fewer species, such as salt marshes, may be overlooked.

Although cattle have been observed in and next to the stream at the study site, a higher percent cover along areas with slightly entrenched streambanks was discovered than in some interior areas of the meadow. A positive correlation between percent cover and *LS* factor shows that livestock were avoiding the streambanks. The slightly entrenched streambank areas had a sharper drop-off down to the stream than the majority of streambank areas. Livestock probably avoided the slightly entrenched streambank areas, because reaching the water or stepping down to the stream from those areas would be more difficult. Sloughing of entrenched streambanks can occur after a rapid drop in flow level that leaves the banks saturated, resulting in positive pore water pressure and highly unstable banks (Fischenich 2001). Drops in flow level are possible after the spring snow melt at the study site.

A 2nd possible contributor to increased erosion potential in the study site may be the decrease in plant cover within the interior meadow. Areas that are bare or sparsely vegetated due to livestock grazing exist throughout the interior of the meadow. Sparsely vegetated areas are prone to higher soil erosion than densely vegetated areas because of lower belowground biomass. Areas with dense vegetation have more roots, rhizomes, and total belowground biomass that stabilize sediment and prevent substrate erosion than areas that are sparsely vegetated (Cronk and Fennessy 2001). Dunaway et al. (1994) found that a vigorous herbaceous plant community provided greater root length density and root mass and resulted in greater resistance to particle erosion. Along with belowground biomass, high aboveground stem and foliage length may also provide protection to the substrate surface under conditions of water inundation (Clary et al. 1996).

The vegetation composition of a wetland meadow may be used to adjust the soil erodibility factor (*K*) in the RUSLE equation. The soil erodibility factor is currently a variable given in the USDA NRCS SSURGO database (USDA 1995) that represents the combined effect of rainfall and runoff on soil loss (Yitayew et al. 1999). Plant composition can affect the erosiveness of a site due to the complexity of the associated underground structures. A wetland dominated by forbs would have considerably fewer underground roots and rhizomes to stabilize soil than a wetland dominated by sedges and grasses that have extensive belowground structures. The *K* factor in this study was not adjusted from the value given in the SSURGO database. Further research should involve adjusting the *K* factor to represent plant composition if RUSLE is to be continually used for wetland meadows.

The strong correlation between *LS* and *A* is simply an outcome reflecting that the RUSLE factors *C* and *P* are constants, *R* varies little, and *K* changes moderately, while *LS* differs substantially across the study area. Potential erosion from the interior of the meadow coupled with erosion from the streambanks and in-channel processes contribute to the amount of sediment going into the stream. According to the Idaho Department of Environmental Quality, sediment is a narrative (not numeric) criterion for listing streams on the 303(d) list. There are 3 conditions to examine before a stream can be listed as impaired due to sediment: (1) there must be an anthropogenic source of sediment; (2) the source must have current, recent, or probable delivery of sediment; and (3) delivery of sediment must be of sufficient quantity and duration to result in an adverse response by the stream. An adverse response is directly measured as an undesirable change in aquatic life (Grafe et al. 2002).

Various methods exist to manage grazing on riparian areas including reducing the number of cattle, reducing the season of grazing, reducing the length of livestock use, and complete removal (Elmore 1992). Difficulties exist in comparing the success of the various methods because federal lands lack standardized field protocols to monitor responses of riparian areas to management practices (Nagle and Clifton 2003). Complete removal of livestock may be the most successful method of restoring a riparian area (Elmore and Beschta 1987, Elmore and Kauffman 1994, Belsky et al. 1999). Complete removal of livestock may involve fencing, enforcement, or both; these options may not be feasible for the tribe. Therefore, further research on other stream reaches and riparian meadows along Deer Creek should be examined and compared to determine which areas are in most need of protection.

In this study we examined the current vegetative composition and erosion potential of an intermountain wetland meadow. We examined only 1 meadow within a wetland complex, but management decision-making requires the consideration of the entire Deer Creek watershed. Based on current vegetation composition and RUSLE model results, uncontrolled grazing may be negatively impacting the study area. The RUSLE model results should be taken with caution. As stated earlier, RUSLE has a number of limitations when used for rangeland or forest. RUSLE emphasizes the effect of surface residue cover on erosion and neglects the importance and impact of the overall plant community on hydrology and erosion. RUSLE does not explicitly account for hydrological processes. Therefore, it does not distinguish between infiltration-excess and saturation-excess runoff, which can have substantially different impacts on water erosion occurrence. Our study area is typified by winter rainy seasons with relatively mild events and soils underlain by basaltic bedrock. The area is thus prone to saturation-excess runoff instead of Hortonian flow. Due to these limitations of RUSLE, future efforts may involve application of a process-based model such as WEPP to the study site and a comparison of the performances of both models.

Additionally, we found a homogenous vegetative community throughout the study area. High percent cover near slightly entrenched streambanks showed that livestock stayed and grazed mainly on gently sloped streambanks and the interior of the meadow. If grazing were properly regulated or completely excluded from this wetland and others along the upper portion of the Deer Creek watershed, the vegetation composition and dominance may change, number of exotic species may decrease, and erosion would likely decrease along with sedimentation into Deer Creek.

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