Research / Evaluate Restoration of NE Oregon Streams

Effects of Livestock Exclosures (Corridor Fencing) on Riparian Vegetation, Stream Geomorphic Features, and Fish Populations

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FINAL REPORT
Research/Evaluate Restoration of NE Oregon Streams: Effects of livestock exclosures (corridor fencing) on riparian vegetation, stream geomorphic features, and fish populations

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(Cover and this Figure are the Middle Fork John Day River at Phipps meadow in grazed (cover) and exclosed reaches (above).

SIGNIFICANT FINDINGS OF RESEARCH

Corridor fencing to exclude livestock grazing along Interior Columbia Basin streams is a widely used approach to restore salmonids and their habitats. Yet few studies have quantified the ecosystem response to this treatment. Vegetation, geomorphic features, and fish populations were sampled in 11 riparian/stream ecosystems in Northeast Oregon. At each stream we sampled two paired reaches – a reach grazed by livestock and one exclosed by fencing. The following are the highlights of this study.

Vegetation

- We found significant differences in the cover, composition and structure of vegetation in all grazed/exclosed reaches. In the majority of exclosed reaches there were increases in the cover of forbs, shrubs, and sedges. Exposed bare ground was more extensive in grazed reaches.

- Wetland indicator scores, based upon streamside vegetation composition, indicate that cessation of livestock grazing results in a shift to more mesic wetland riparian vegetation.

- Species adapted to herbivory and drier environments were more abundant in grazed riparian areas.

- Plant species diversity and richness were higher in exclosed stream reaches.

- Composition of shrubs varied between streams. In 88% of the streams where exclosures were over 5 years old, the cover of riparian obligate shrubs (e.g., willows, alder, etc.) was greater in exclosures, with the greatest differences in exclosures over 20 years old

Geomorphology

- Considering all sites together, livestock exclusion resulted in statistically significant improvements in channel geomorphology. The channels in the exclosed reaches are narrower, deeper, and have more pool area than the channels in the grazed reaches.

- At the level of individual sites, in most cases the exclosed reach was clearly narrower, deeper and had more pool area than the grazed reach at the same site.

- We did not detect any differences between fenced and grazed reaches in maximum pool depth or residual pool depth.

- Geomorphic response to livestock exclusion appears to be influenced by multiple factors, including age, vegetation cover, hydraulic conditions, and site geomorphology. Younger exclosures show less vegetation difference with the paired grazed reach and are less likely to show geomorphic adjustment. Other conditions, such as stream power, channel constraint and sediment supply may also limit the effectiveness of restoration projects.

Fish

- Densities of young-of-the-year redband trout (*Oncorhynchus mykiss)* were significantly greater in exclosed reaches compared to grazed reaches. Moreover, the effects of fencing were negatively associated with the dominant warmwater fishes, redside shiners (*Richardsonius balteatus)* and speckled dace (*Rhinichthys cataractae)*, which are relatively uncommon in the best trout habitats.

- Conversely, we could not detect significant differences in densities of combined juvenile and adult life stages of salmonids between exclosures and grazed reaches, suggesting that recruitment bottlenecks exist and/or diurnal migrations within home ranges that extend beyond exclosure lengths may be occurring.

- Fish responses to cattle exclosures were weak because the best experimental design that we could construct from existing exclosures was limited by their lengths which were very small compared to the total stream habitat available and to the home ranges of the fish species of interest.

- Another limitation was that six out of the nine exclosures were below reaches of stream that were disturbed by grazing. The downstream effects of livestock grazing on the water column would more likely compromise benefits of fencing a relatively short distance of a small exclosure.

- The foregoing limitation partly explains why temperatures between grazed and fenced sections of streams were not significantly different.

Table 1. Generalized ecosystem response within exclosures based upon comparisons of 11 grazed/exclosed reaches in northeastern Oregon.	
Ecosystem Component	Apparent response in exclosed riparian zones and stream reaches*
Shrub cover	**?**
Herbaceous cover	?
Species richness of vegetation	?
Plant Species diversity	?
Wetland species composition dominance	?
Bare ground	**?**
Channel Depth	?
Channel Width	**?**
Pool area	?
Water temperature	**?**
Young of the year salmonids	?
Adult salmonids	**?**
Warm water fishes	**?**
* A ? indicates an increase in abundance or cover in exclosures; a **?** indicates a decrease in abundance or cover in exclosures and a **?** indicates no difference between exclosures and grazed reaches.	

MANAGEMENT RECOMMENDATIONS

- Grazing exclosures are a simple, holistic, and effective restoration strategy. Changes in vegetation composition structure as well as geomorphic features suggest that livestock exclusion succeeds in restoring many important components of productive wildlife and fish habitats. A significant increase in young of the year salmonid density was evident across exclosures but a difference was not detectable for larger fish whose home ranges greatly exceeded exclosure lengths of this study.

- Response of vegetation and geomorphology was greatest in the oldest exclosures suggesting the quality of fish and wildlife habitats increase with increasing exclosure age. Land management agencies and landowners should be encouraged to maintain exclosures as long-term investments in habitat restoration.

- Small exclosures that cover only a few hundred meters of channel length may result in locally improved vegetation cover, channel geomorphology, and young-of-the-year salmonids but improvements in the density of adult fish populations or water temperature were not detected. Effective restoration of water quality and fish populations will require exclosures to be significantly longer than most of the exclosures currently in place in the study area.

- The size and age of exclosures in this study are representative of exclosures throughout the interior Columbia Basin. We conclude the scale of the exclosures sampled in this study, (in terms of size and time) is too small to produce anticipated improvements in juvenile and adult coldwater fishes. Larger areas of livestock exclusion for long time periods will be necessary to restore salmonids. We suggest that more effective and efficient restoration can be accomplished by a strategic approach at the sub-basin scale, taking into account the lengths of, and distances between, exclosures and their locations with respect to the migratory patterns of salmonids in each sub-basin.

- Many key questions about how and where to do restoration projects remain unanswered. Despite our strenuous efforts in site selection, the ex post facto research design of this study limited the strength of the research results. Monitoring of effectiveness of restoration projects must be improved. Pre-construction monitoring and ten to twenty years of post-construction monitoring should be initiated in a large number of new restoration projects. Monitoring should focus on ecosystem, habitat, and fish population changes. Further monitoring and research will lead to better decisions about location, scale, and methods of restoration projects.

Acknowledgments

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Introduction

The Pacific Northwest Electric Power Planning and Conservation Act of 1980 indicated "The council shall properly develop and adopt...a program to protect, mitigate, and enhance fish and wildlife, including related spawning grounds and habitat on the Columbia River and its tributaries." As a result, the Bonneville Power Administration (BPA) has spent millions of dollars on various instream projects throughout the Columbia Basin with the goal of increasing system-wide production of anadromous fisheries through a combination of habitat restoration and enhancement measures.

For two decades, numerous BPA-funded projects have been initiated in the upper Columbia River Basin for the express intent of improving the aquatic habitats of anadromous salmonids. Largely missing from most of these projects has been any rigorous evaluation of project success or failure. Some field reviews of some habitat projects have been undertaken (e.g., Beschta et al. 1991, Kauffman et al. 1993) and provide an overview of major problems and opportunities associated with selected projects. However, there continues to be a lack of quantifiable information, collected in a systematic manner that could be used as the basis for scientifically assessing the effects of individual projects on riparian/aquatic habitats, functions, or processes.

Recent publications (e.g., NRC 1992, ISG 1996, NRC 1996, Beschta 1997, and Kauffman et al. 1997) have identified and summarized important concepts associated with the restoration and improvement of aquatic ecosystems. While such conceptual approaches provide an important structure for those undertaking restoration efforts, there remains a paucity of basic information throughout the upper Columbia Basin on the hydrologic, geomorphic, and biologic responses that occur from various enhancement approaches. Basic data on the spatial and temporal responses of restoration approaches would provide: (1) a better understanding of project effects upon aquatic habitats and associated riparian functions; (2) a means of determining rates of aquatic habitat improvement; and (3) a basis for projecting future trends of habitat recovery.

The proposed research is intended to provide an improved understanding of both the effects and effectiveness of a commonly used habitat enhancement approach in the upper Columbia River Basin. This is the exclusion of domestic livestock from streamside communities and streams via corridor fencing (exclosures).

This final report is broken into three separate chapters. The first chapter covers the vegetation change associated with livestock exclusion. The second chapter focuses on the physical geomorphic changes to the streambank and channel. The final chapter covers the response of salmonids and warmwater fishes to livestock exclusion at the spatial scales of exclosures as is commonly constructed today. It is expected that this study will provide an important scientific basis, currently lacking, for understanding the ecological principles of restoration/enhancement of sustainable aquatic habitats for salmonids. Thus, the results of this work are likely to have important ramifications for habitat improvement projects within and beyond the general geographic region of northeastern Oregon.

Objectives of Research

Objective 1. Along a variety of differing riparian/stream ecosystems in Northeastern Oregon, quantify the geomorphic, hydrologic, fisheries, and riparian vegetation responses to the passive restoration approach of livestock exclusion/corridor fencing.

Objective 2: Based upon our research findings, to the greatest extent possible, develop management recommendations and delineate additional research needs to determine which type of approaches will yield the highest probability for the enhancement of salmonid habitats.

Study Sites

To examine how vegetation, geomorphic structure, and fisheries populations differed between grazed and ungrazed stream reaches a total of 11 Northeast Oregon streams were selected for study in 2000 (Figure1). Each study stream consisted of two reaches a grazed reach and an exclosed reach. Grazed reaches were those in which livestock grazing (principally cattle) was a dominant use in the riparian zone and surrounding uplands. Exclosed reaches were those where livestock grazing has been eliminated through the construction of riparian exclosures or corridor fences. Ages of the exclosures ranged from ~3 to 37 years (Table 1). Criteria for site selection included paired reaches (grazed and exclosed) that were as geomorphically similar as possible, streams with salmonids, knowledge of the history of the exclosure, and owner permission. These criteria increased the likelihood that differences between grazed and exclosed areas were largely due to differences in land use (i.e., the ecosystem response to livestock exclusion). Because most exclosures were not built with consideration of research needs these criteria severely limited the number of suitable sites. We visited 100s of sites but rejected the vast majority because potential grazed/exclosed reaches did not meet our strict criteria for selection. As is the case for most livestock exclosures, occasional trespass grazing occurred for many of the sites. Uplands were dominated by ponderosa pine (*Pinus ponderosa*) and mixed conifer forests. Land ownership was public and private.

At each site, the exclosed reach was always designated reach B, and the control reach was designated reach A if located upstream of reach B or C if located downstream of reach B (Table 1). At all sites, drainage areas were less than 100 km^2 (40 mi^2), and the streams were approximately second to third order. At all sites, the channel was an unconstrained, sinuous, alluvial channel with a gravel bed (Table 1). Channel gradients were all less than 2%, and all channels fell within the pool-riffle type (Montgomery and Buffington 1998).

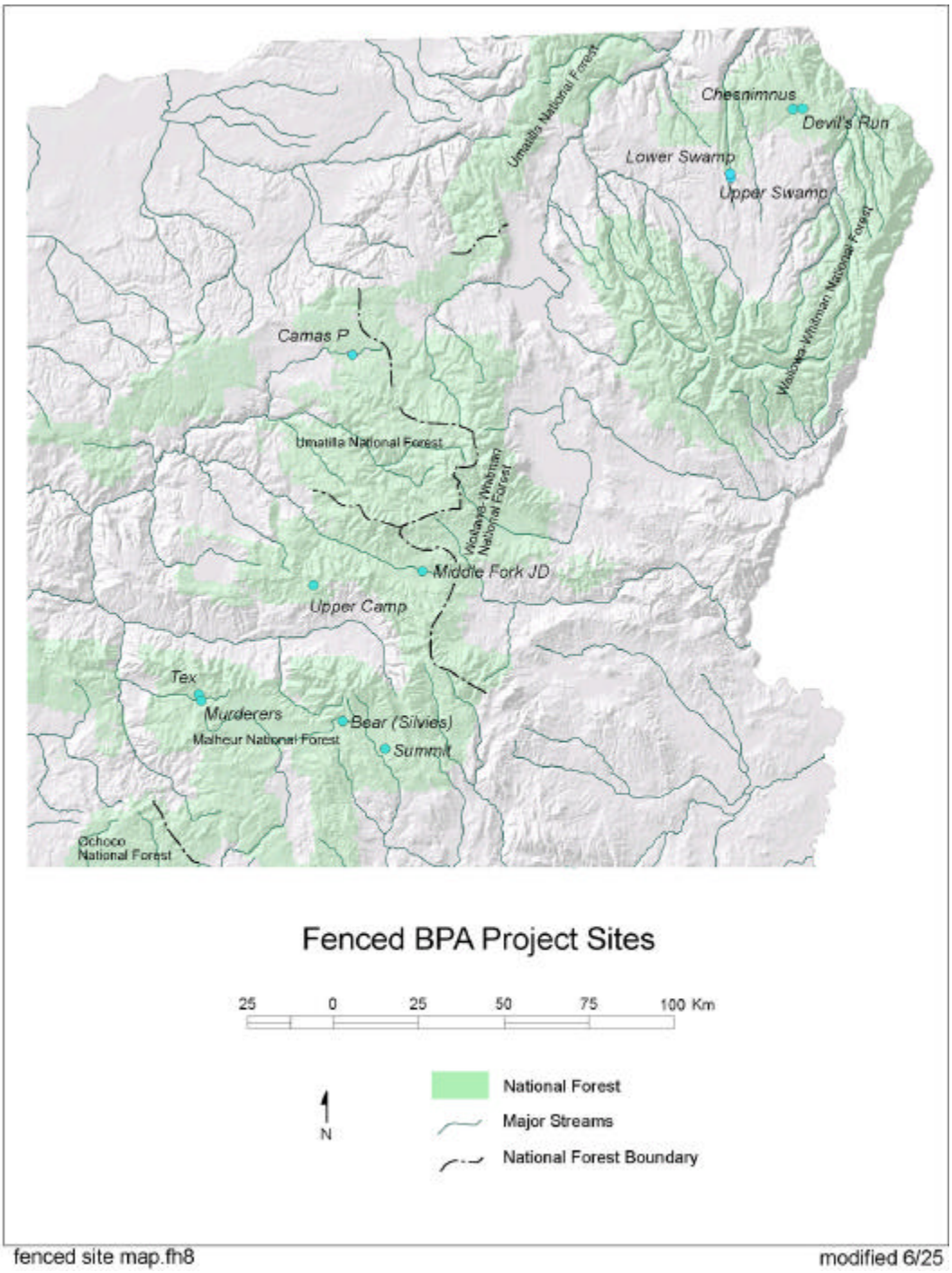

Figure 1. Location map of fenced reaches selected for study of the ecosystem response to livestock exclusion in Northeastern Oregon.

Table 1. Site characteristics of the 11 stream reaches selected for study in Northeast Oregon. (see Table 1-1 for reach lengths and additional descriptions).

Site	Abbreviation	Sampled Reaches	Site Elevation (m)	mean annual ppt (in)	Drainage Area(km^2)	sinuosity	valley gradient	channel gradient	bed material D_{50} (mm)
Bear Creek (Silvies)	BearSi	B,C	1554	27	39.2	1.55	0.0099	0.0064	28
Camas Creek	CamasP	A,B	1240	25	95.3	1.22	0.0072	0.0058	55
Chesnimnus Creek	Chesni	A,B	1305	19	40.5	1.27	0.0177	0.0139	75
Camp Creek	CmpUp	A,B	1467	25	16.4	1.33	0.0343	0.0258	59
Devil's Run Creek	Devils	B,C	1285	19	29.0	1.75	0.0186	0.0105	54
Middle Fk John Day (Phipps Mdw)	MFPhip	A,B	1292	21	97.6	1.91	0.0043	0.0023	24
Murderers Creek	Murder	A,B	1347	19	36.3	1.58	0.0052	0.0034	10
Summit Creek	Summit	B,C	1506	23	77.3	1.45	0.0090	0.0062	54
Lower Swamp Creek	SwmpLo	B,C	1123	19	79.9	1.29	0.0060	0.0046	31
Upper Swamp Creek	SwmpUp	A,B	1142	19	74.6	1.33	0.0077	0.0056	28
Tex Creek	Tex	A,B	1359	19	31.6	1.24	0.0111	0.0090	33

Chapter 1 RIPARIAN VEGETATION COMPOSITION IN PAIRED GRAZED AND UNGRAZED STREAM REACHES IN NORTHEASTERN OREGON

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Abstract

Exclusion of cattle along stream riparian corridors has been suggested to be an effective means of riparian/stream restoration benefiting both terrestrial wildlife as well as the aquatic biota. Construction of corridor fencing to exclude livestock has been accomplished along hundreds of kilometers of streams in the Columbia Basin, yet no studies have been undertaken to evaluate their efficacy. We sampled riparian vegetation composition along 11-paired grazed and exclosed (ungrazed) reaches in Northeastern Oregon streams. Exclosure ages ranged from 3 to >30 years and grazing treatment ranged from light grazing every one out of three years to heavy season-long grazing. Rather than examine one type of grazing strategy we were interested if vegetation patterns could be detected between riparian areas with and without livestock influences. Each reach consisted of 20 to 30 channel units. In the middle of each channel unit on each streambank, we sampled herbaceous vegetation composition in a 1 x 4 meter plot placed at the streambank edge (the greenline). Shrub cover was measured along the greenline of each streambank of each channel unit. Stream cover and cover of emergents was also measured. Species diversity was higher in exclosed reaches for all streams. Analyses detected a significant increase in the abundance of native sedges (*Carex spp*) in ungrazed areas. In contrast exotic species adapted to grazing such as *Poa pratensis* and *Trifolium repens* were more abundant in grazed stream reaches. In exclosures, the wetland indicator scores of the vegetation composition significantly decreased compared to grazed reaches. This indicates that along exclosed stream reaches wetland plant communities are replacing ones adapted to drier environments. Shrub responses were slower than the responses of the herbaceous composition. Riparian-obligate shrub cover along the streambank was higher in 7 of 8 of the exclosures that were older than 5 years. We conclude that cessation of livestock grazing resulted in shifts in vegetation structure and composition that would be favorable to the native aquatic and terrestrial biota. We found that cessation of livestock grazing was effective in increasing the biological diversity of the streamside riparian plant communities. The benefits, ecosystem services, and values of the riparian/stream exclsures increase through time and may not be fully realized until decades after exclusion. With increases in structure, we would expect vegetation to positively affect other ecosystem processes such as allocthonous inputs and sediment retention, thereby affecting the aquatic biota, water quality and stream geomorphology.

Introduction

Riparian vegetation is a keystone ecosystem feature that has a strong influence on stream ecosystems including the quality of fish habitat. Riparian zones are the most productive and biologically diverse wildlife habitats in the Columbia Basin (Kauffman et al 2001). In this sense, fisheries enhancement, wildlife habitat enhancement, and riparian vegetation recovery are one in the same. The restoration of fisheries and wildlife habitats must include the restoration of riparian vegetation structure and function to be successful.

Kauffman et al. (1996) suggested the first logical step in riparian restoration is the implementation of "passive restoration" defined as the cessation of those activities that are causing ecosystems degradation or preventing recovery. Because the vegetation of riparian zones are adapted to frequent fluvial disturbances (Gregory et al. 1991), many riparian species posses adaptations facilitating a rapid recovery following both natural and anthropogenic disturbances. In terms of anthropogenic disturbances, livestock grazing is the most widespread land use in the western USA and has been suggested to be a significant influence affecting riparian ecosystem structure, diversity, and function (Kauffman and Krueger 1983, Elmore and Kauffman 1991). As such, cessation of livestock grazing along salmonid bearing streams has been a common approach to fish and wildlife habitat restoration in the Interior Colombia Basin of Oregon and Washington.

Few studies have quantified the vegetation differences between grazed and exclosed stream reaches in a wide diversity of stream types in the Pacific Northwest. The objective of this study was to quantify changes in the composition, structure, and mass of the riparian vegetation along 11 experimental streams where passive restoration (Corridor fencing or livestock exclosures) has occurred. We measured species composition and diversity in paired reaches that included livestock exclusion with an adjacent reach where the riparian zones were grazed by domestic livestock.

Methods

To examine how streamside riparian vegetation differed between grazed and ungrazed stream reaches a total of 11 Northeast Oregon streams were selected for study in 2000. Each study stream consisted of two reaches a grazed reach and an exclosed reach. Grazed reaches were those in which livestock grazing (principally cattle) was a dominant use in the riparian zone and surrounding uplands. Exclosed reaches were those where livestock grazing has been eliminated through the construction of riparian exclosures or corridor fences. Ages of the exclosures ranged from ~3 to 37 years (Table 1). Criteria for site selection included paired reaches (grazed and exclosed) that were as geomorphically similar as possible, streams with salmonids, knowledge of the history of the exclosure, and owner permission. These criteria increased the likelihood that differences between grazed and exclosed areas were largely due to differences in land use. Because most exclosures were not built with consideration of monitoring and evaluation, these criteria severely limited the number of suitable sites. As is the case for

most livestock exclosures, occasional trespass grazing occurred for many of the sites. Uplands were dominated by ponderosa pine (*Pinus ponderosa*) and mixed conifer forests. Land ownership consisted both of public and private ownership.

Table 1. Study streams, data of exclosure of the 11 sites where vegetation in paired grazed and ungrazed reaches was measured.

Site Name	**Date of exclosure**	**USGS Quad**
Bear Creek (Silvies)	1998	Big Canyon, Logan Valley West
Camas Creek	1995	Lehman Springs
Chesnimnus Creek	1986-87	Poison Point
Camp Creek	1963-64	Cougar Rock
Devil's Run Creek	1990	Poison Point
Mid. Fk John Day (Phipps meadow)	1997	Austin
Murderers Creek	1965-75	Big Weasel Springs
Summit Creek	1978	Dollar Basin, Logan Valley East
Lower Swamp Creek	1992	Elk Mountain
Upper Swamp Creek	1987	Elk Mountain
Tex Creek	1977	Big Weasel Springs

Each study reach was first delineated into channel units. We sampled vegetation composition and cover in at least 20 channel units for each grazed and exclosed reach. Reach length for each site ranged from 99 to 340m.

Vegetation community composition was determined for each channel unit on both sides of the stream (N ˜ 40 plots/reach). These were determined by calculating the percent cover each plant species that occurred in a 1 X 4 m plot. These plots were placed so that the center of the plot was directly adjacent to the center of the major characteristic defining that channel unit (e.g., the middle of the riffle or the middle of the pool). Plots were placed with the innermost edge on the green line (i.e., the transitional point where terrestrial vegetation dominates ground cover). All plant species with a canopy cover 5% or more within the plot were recorded.

From the microplot data, we calculated species richness (number of species per experimental reach), Species diversity, and similarity. Species diversity was calculated using the Shannon Index where:

$H' = -? \; p_i \ln(p_i)$.

The quantity p_i is the proportion of cover of the ith species relative to the sum of cover for all species. We also report species diversity as the exp H' which is equivalent to the number of equally common species required to produce the value of H'(Magurran 1988).

The similarity between grazed and exclosed reaches was calculated using Sorenson's quantitative measure of similarity (Magurran 1988). Similarity ranges from 0 (no species in common) to 1 (all species and their cover are identical). Similarity (C_N) was calculated using the formula:

$C_N=2jN/aN+bN$

Where:
jN = sum of the lower of the two abundances (cover) of all species occurring on the grazed and exclosed reaches.
aN= The sum of plant cover in the exclosed reach
bN= The sum of plant cover in the grazed reach

The prevalence index also referred to as the wetland score was calculated for each grazed and exclosed reach to determine predominance of hydrophytic (wetland) vegetation (Wentworth et al. 1988). The prevalence index was computed by weighting the species cover from plots with index values for wetland indicator categories (Table 2). Wetland indicator values were assigned to each species using the *National List of Plant Species that Occur in Wetlands: Northwest (Region 9)*; (USDA, NRCS. 2001). By assigning the composition to the USFWS wetland indictor scores we derived a wetland score for each grazed and exclosed reach of the study. Wetland scores can range from 1(all species wetland-obligate) to 5 (all species upland-obligate).

Table 2. Indicator categories, probability ranges, and indicator index values for species occurrence in wetlands.

Wetland Indicator Category	Probability of Occurrence in Wetlands	Indicator Index
Obligate wetland (OBL)	>99%	1
Facultative wetland (FACW)	67-99%	2
Facultative (FAC)	34-66%	3
Facultative upland (FACU)	1-33%	4
Upland (UPL)	<1%	5

Prevalence indices were calculated as follows:

$$\text{The Wetland Score or Prevalence Index} = \sum A_i W_i / \sum W_i$$

Where: A_i = abundance (cover) of species i
W_i = indicator index value for species i
i = species

To statistically determine if there were differences between grazed and nongrazed areas, the study reach was the individual unit. The sign test was used to test for differences in prevalence indices and cover between treatments and controls (Snedecor and Cochran

1967). Sign tests were conducted using SPSS (nonparametric test, two related samples procedure, SPSS 2000).

Vegetation cover over the stream was measured at each channel unit. This measure employed the use of a concave spherical densiometer (Lemmon, 1957). The densiometer was taped so that there was a "V" exposing only 17 of the grid line intersections following the methods of Platts et al. (1987). Cover was measured approximately 30 cm above the surface of the water and approximately 30 cm from each bank as well as the middle of the stream. One reading at each bank was taken and four readings were taken in the middle of the stream. The four readings in the middle included readings up stream, down stream, towards right bank and towards left bank. If emergents were present in the active channel, their cover relative to the entire water surface area of the channel unit was estimated.

Shrub and tree composition was measured via the line intercept technique. In each channel unit there were 2 transects running the entire length of the unit; one on each side of the creek and running along the green line. Total length of shrub transects in the study reaches ranged from 188 to 688m

All trees and shrubs that were overhanging the green line on each bank were recorded. For each tree or shrub determined to overhang the green line, the distance of canopy coverage was recorded. All shrubs were identified to species. The cover of every individual shrub or tree overhanging the green line was measured regardless if there was overlap with other individuals. From these data, woody plant composition and streamside cover was determined.

Results

At all sites we found differences in vegetation composition and structure that we attribute to livestock exclusion. Comparing the paired grazed-exclosed reaches, we found that the sedges (*Carex spp*) had a greater cover (>5% increase) in 63% of the restored (fenced) vs grazed reaches. Forbs were more prevalent in 73% of the restored stream reaches. Shrub cover was greater in 88% of the exclosed reaches >5 years old. Bare ground was higher in 63% of the grazed reaches (Table 3 and 4).

Species richness was higher in exclosures of 82% of the experimental streams while species diversity was higher in all 11 of the exclosed reaches (Table 5). The greatest species richness was found in exclosed reaches of Camp Creek (S=50) and Camas, Chesnimnus and Devil Creeks (N=45). The greatest differences in species richness were in the heavily grazed Devil's Creek where 17 more plant species were encountered in the exclosures than grazed areas. Species diversity (exp H') was as low 4.71 and 5.45 in the grazed reaches of the Middle Fk John Day and Murderer's Creek. However, the exclosed reaches of these two streams had species diversity values of 14.62 and 12.70, indicating that livestock were limiting species diversity on these sites. Similarly, there was a great

increase in the species diversity in the grazed and exclosed reach at Summit creek (9.12 and 17.74, respectively). Sites with the fewest changes included the recently established Bear and Camas creek sites as well as the forested Tex creek. With a dense overstory canopy and light grazing regime, understory composition was most similar at Tex Creek (a similarity index of 0.73).

The changes in species richness and diversity reflect real changes in species composition between the grazed and exclosed reaches. Plant species composition were least similar between reaches (0.34-0.38) at the Devils, Lower Swamp, Mid Fk John Day, and Summit Creeks. These were all relatively low gradient reaches where sedges and more hydric species were found in greater abundance in exclosures. The relatively young age of some of these exclosures with this stream type, demonstrates that species composition shifts of the herbaceous component can occur in a relatively short period of time. Comparing grazed and exclosed reaches we found the most similar composition to be in the forested reaches with a higher gradient (Tex and Camp creek with a high shrub/tree cover) and in some sites with recently established exclosures.

Wetland scores were consistently and significantly lower in the exclosed reaches compared to the grazed reaches (Fig. 1, Table 5). Wetland-obligate and Facultative wetland species were found in greater abundances in exclosures. This indicates a greater amount of water available for riparian plant use following fencing. Combining all sites, we found a statistically significant increase in native sedge (*Carex spp*) abundance (largely wetland-obligate and facultative-wetland species (Figs. 2 and 3). The least amount of compositional change was in Tex Creek (a partially constrained forested reach). The greatest differences were in unconstrained. low gradient reaches (e.g., Mid Fk John Day, Summit Creek, and Lower Swamp Creek). For example, at Summit Creek, the wetland scores for the grazed (3.32) and exclosed reaches (1.95) indicate that the grazed reach is dominated by facultative and facultative upland species while the streambank in the exclosures is dominated by wetland-obligate and facultative wetland species. This is apparent through examination of the dominants in this stream where *Carex* species (wetland obligates) dominated in exclosures (55% cover) while Facultative species *Poa pratensis* and *Trifolium longipes* dominated the grazed reach (50% cover; Table 3). This was a consistent trend in the majority of the stream reaches sampled; hydric species such as *Carex spp. Glyceria spp*., and *Scirpus microcarpus* were more abundant is exclosures while species more adapted to grazing and drier conditions (e.g., *Poa pratensis, Phleum pratense, Trifolium spp and Taraxacum officinale*,) were usually more abundant in grazed reaches (Figs. 2 and 3). While there were shifts in forb composition there was also an overall increase in forb cover in 73% of the exclosures contributing to increases in ground cover as well as species diversity. Similar to changes in vegetation cover, bare ground was lower in 63% of the exclosures compared to their adjacent grazed reaches (Table 4).

Shrub Composition and Cover

Stream reaches varied not only in composition but also in their potential to support large expanses of wood-dominated communities (Table 6). Streamside cover of woody vegetation ranged from 0.7% in the meadow-dominated Mid-Fk John Day to 129% in the forested Tex creek. The most abundant woody species was thin-leaf alder which was present in 10 of the 11 sites. Willows (*Salix spp*) were present in all sites. A total of 6 willow species were encountered in the study but never in great abundance. In total, we encountered 28 shrub and tree species in the study.

We found that shrub cover was higher in exclosures in 6 of the 11 sampled reaches. Examining riparian-obligate shrubs we found higher cover in 7 of 11 study streams. Differences can be explained by the age of the exclosure. Of the 4 study streams where cover was equal or less in exclosures compared to grazed areas, 3 were less than 5 years old. In 7 of 8 study streams where exclosures were >5 years in age, shrub cover was greater in the exclosures. The greatest differences in shrub cover were in the oldest 4 exclosures (>20 years). For example, woody vegetation cover at Summit creek was 26 and 6% in exclosed and grazed areas, respectively, while cover at Camp creek was 74 and 35%, respectively (Table 6). This finding indicates that the full effects of vegetation response to livestock exclusion may require over 20 years for the full expression to occur. The benefits, ecosystem services, and values of the stream reaches excluded from livestock increase through time and may not be fully realized until decades after exclusion.

Table 3. A summary of the number of experimental reaches where vegetation parameters differed by 5% or greater between grazed and exclosed reaches.

	> Abundance in exclosures	> Abundance in grazed reaches
Sedges	7 (63%)	0
Rushes	4 (36%)	2 (18%)
Grasses	2(18%)	5 (45%)
Forbs	8 (73%)	2(18%)
Shrubs	7 (63%)	1 (9%)
Trees	1 (9%)	0
Bare Ground	2 (18%)	7(63%)
Emergent	3(27%)	0
Overstory	5 (45%)	2(18%)
Species richness	7 (63%)	1 (9%)
Species diversity	10 (91%)	0

Table 4. Vegetation cover (%) of riparian vegetation life forms in 11 paired exclosed (restoration) and grazed reaches in Northeastern Oregon. All sites were treated with corridor fencing.

	Cover (%)											
	Bear		Camas		Chesnimnus		Devils		Lower Swamp		Upper Swamp	
	Exclosed	Grazed	Exclosed	Grazed	Exclosed	Grazed	Exclosed	Grazed	Exclosed	Grazed	Exclosed	Grazed
Sedges	**29.6**	18.4	**20.6**	17.1	**9.2**	1.5	**14.7**	0.7	**38.7**	4.6	**33.8**	4.7
Rushes	**2.2**	9.5	**8.0**	4.2	**0.4**	0.3	**0.9**	0.4	**14.9**	0.4	**13.5**	0.9
Grasses	**23.1**	30.3	**37.9**	33.9	**33.1**	33.0	**41.9**	58.1	**39.8**	68.1	**35.9**	35.2
Forbs	**49.0**	68.7	**40.5**	41.6	**50.5**	36.2	**59.5**	46.0	**35.2**	20.4	**43.1**	31.7
Shrubs	**2.6**	3.2	**2.9**	2.5	**30.8**	25.4	**18.9**	1.2	**18.9**	58.8	**13.9**	6.9
Trees	**8.8**	6.4	**1.0**	1.9	**6.6**	10.9	**5.7**	2.9	**1.2**	3.2	**0.0**	2.6
Bare Ground	**25.4**	9.6	**25.1**	19.7	**14.3**	24.1	**4.7**	12.2	**1.7**	8.0	**0.4**	36.2
Emergent	**1.4**	0.5	**5.3**	6.8	**0.6**	1.0	**3.1**	0.8	**23.5**	2.5	**11.8**	2.9
Overstory	**18.0**	19.8	**56.8**	3.7	**39.8**	48.3	**38.7**	10.4	**51.5**	78.5	**31.2**	35.8

	Cover (%)									
	Mid Fk John Day		Murderers		Summit		Tex		Camp	
	Exclosed	Grazed	Exclosed	Grazed	Exclosed	Grazed	Exclosed	Grazed	Exclosed	Grazed
Sedges	**46.6**	12.7	**33.4**	29.4	**54.7**	20.9	**5.7**	4.4	**12.7**	10.6
Rushes	**14.2**	1.6	**51.8**	68.6	**10.2**	6.5	**0.1**	3.3	**26.7**	18.0
Grasses	**37.5**	66.5	**42.0**	30.6	**11.1**	11.1	**41.3**	35.6	**52.5**	61.9
Forbs	**48.5**	28.0	**50.9**	21.1	**43.9**	50.4	**61.6**	40.4	**74.3**	67.6
Shrubs	**0.3**	0.6	**11.2**	0.1	**17.4**	5.6	**64.7**	51.9	**32.7**	15.1
Trees	**0.0**	0.0	**0.0**	0.0	**6.4**	0.2	**7.0**	5.9	**8.7**	6.6
Bare Ground	**15.9**	15.4	**4.4**	6.0	**10.6**	22.9	**1.9**	10.9	**0.6**	10.0
Emergent	**13.8**	12.8	**13.3**	8.2	**3.6**	4.4	**3.7**	0.6	**5.3**	1.4
Overstory	**2.7**	0.4	**32.4**	13.2	**22.1**	4.2	**70.3**	67.2	**56.8**	43.9

Note: Bear, Camas, and MF John Day are relatively recent restoration programs (<5 years). The others are 6-36 years in age.

Discussion

The data are clearly indicating an increased connectivity of riparian vegetation and their associated aquatic system. Coupled with other studies in Northeast Oregon riparian zones, we suggest that the restoration of the riparian areas associated with the exclusion of domestic livestock grazing results in a number of improved ecosystem services. For example, Kauffman et al (in prep) has found that root biomass dramatically increased in exclosed riparian meadows compared to grazed meadows. In addition they reported that Carex-dominated meadows had much higher root mass than dry (*Poa pratensis*) - dominated meadows. This suggests that the exclosed streamside communities in this reach will have a greater root mass than grazed reaches resulting in greater erosion resistance and an enhancement in streambank structure.

The source of much of the energy and nutrients of headwater streams originates from streamside vegetation. Riparian meadows and forested reaches play an important function as a source of allocthonous materials to aquatic systems (Brookshire 2001). In meadow-dominated reaches during peak flows when streamside communities are inundated, plant materials become a source of organic C and nutrients to the stream. Exclosures likely have an increase in streamside biomass and hence organic inputs into the aquatic biota for three reasons: (1) there is no removal of streamside vegetation via herbivory (2); there is an increase in species of higher productivity (e.g. *Carex spp.* compared to *Poa pratensis*); and (3) there is an increase in shrub and tree cover providing shade and direct cover alsong banks for fish. With an increase in streamside biomass and overstory cover, we would expect increases in the seasonal duration of inputs as well as the total quantity that enters the stream. In addition, there is likely an increase in the diversity of inputs of allocthonous inputs (timing and composition) associated with the increase in plant species diversity.

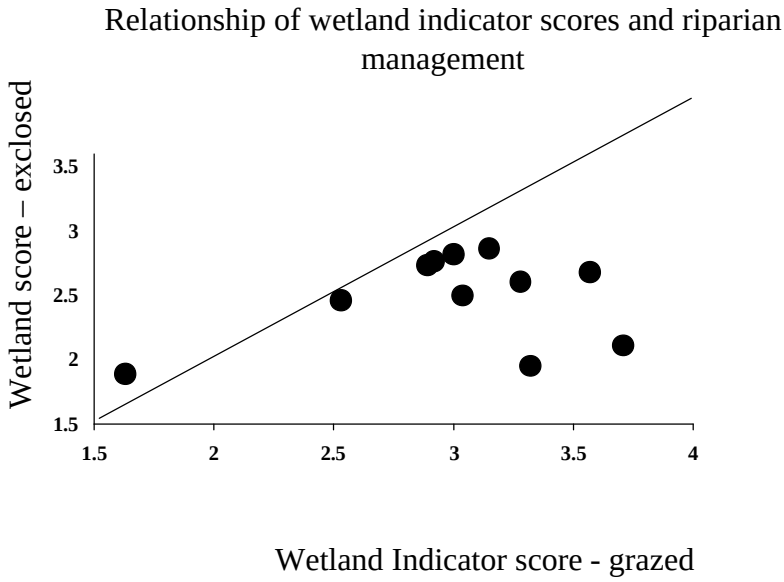

Figure 1. The relationship of wetland indicator scores between grazed and fenced sections of 11 study reaches in Northeast Oregon 2000. . If there was no difference in the wetland scores between grazed and fenced areas the numbers would be expected to fall on or near the line. Occurrence below the line indicates a higher abundance of wetland species in fenced areas compared to grazed reaches.

Comparison of fenced vs grazed cover of wetland obligate sedges (*Carex spp*)

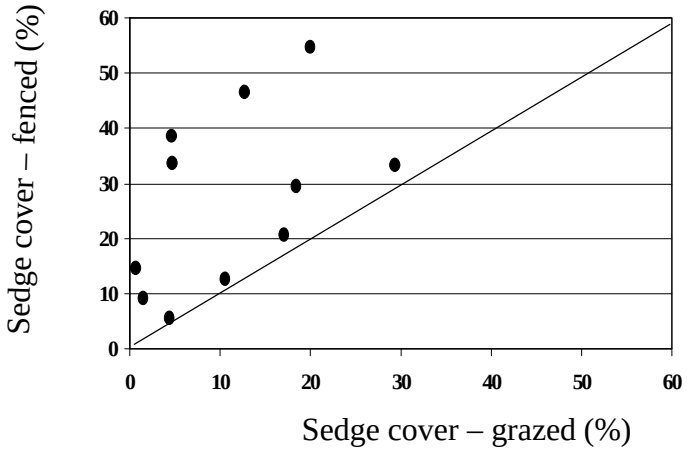

Figure 2. The relationship between sedge cover in grazed and ungrazed riparian reaches of 11 BPA project areas of Northeast Oregon, 2000. If there was no change in the abundance of sedges between grazed and fenced areas the numbers would be expected to fall on or near the line. Occurrence above the line indicates a higher abundance of sedges in fenced areas compared to grazed reaches.

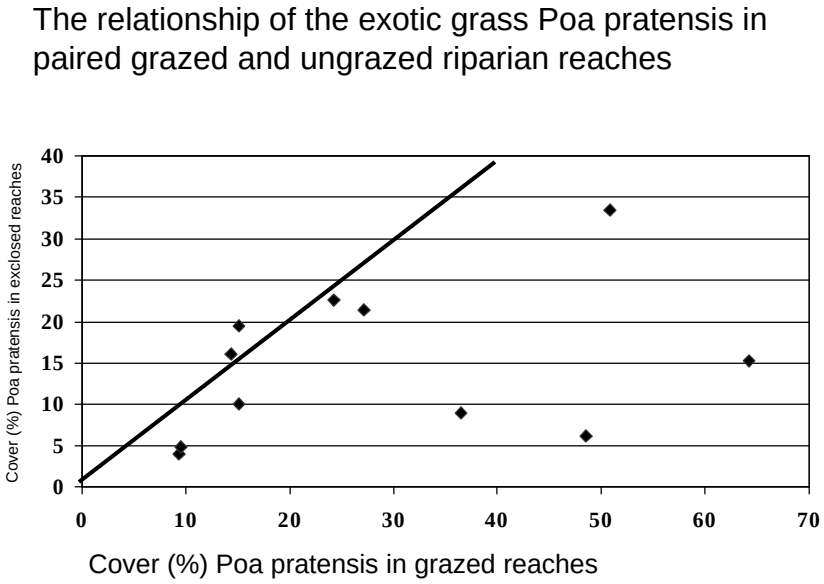

Figure 3. The relationship of an exotic grass, *Poa pratensis*, adapted to herbivory, between grazed and fenced sections of 11 study reaches in Northeast Oregon 2000. If there was no difference in the abundance of this species between grazed and exclosed areas the numbers would be expected to fall on or near the line. Occurrence below the line indicates a higher abundance of *P. pratensis* in grazed reaches.

Table 5. Plant species richness (S) , species diversity (H'and exp H'), similarity (C_S), wetland score or prevalence indices (PI), and dominant species and cover for paired exclosed and stream reaches in Northeastern Oregon.

Site	S	H'	exp H'	C_S	PI	Dominant species (% cover)
Bear						
Exclosed	40	3.10	22.10	0.62	2.82	Poa pratensis (10), Fragaria virginiana (10), Carex lanuginosa (9)
Grazed	41	3.02	20.54		3.00	Equisetum arvense (19), Trifolium repens (16), Poa pratensis (15)
Camas						
Exclosed	45	3.17	23.83	0.55	2.73	Poa pratensis (19), Carex lanuginosa (7), Juncus balticus (5)
Grazed	45	3.11	22.42		2.89	Poa pratensis (15), Trifolium repens (7), Phleum pratense (7)
Chesnimnus						
Exclosed	45	2.80	16.48	0.41	2.60	Equisetum arvense (30), Poa pratensis (22), Salix alba (18)
Grazed	36	2.61	13.54		3.28	Poa pratensis (27), Pseudotsuga menziesii (11), Trifolium repens (11)
Devils						
Exclosed	45	2.90	18.09	0.35	2.68	Myosotis scorpiodes (32), Agrostis stolonifera (15), Alnus incana (10)
Grazed	28	2.17	8.76		3.57	Poa pratensis (37), Trifolium repens (19), Phleum pratense (17)
Lower Swamp						
Exclosed	36	2.74	15.50	0.34	2.50	Agrostis stolonifera (25), Carex utriculata (20), Alnus incana (18)
Grazed	34	2.02	7.50		3.04	Alnus incana (52), Poa pratensis (49), Agrostis stolonifera (10)
Upper Swamp						
Exclosed	35	2.92	18.46	0.42	2.46	Agrostis stolonifera (18), Erigeron philadelphicus (15), Alnus incana (13)
Grazed	30	2.67	14.37		2.53	Juncus balticus (13, Poa pratensis (9, Trifolium repens (9
Mid Fk J. Day						
Exclosed	32	2.68	14.62	0.34	2.11	Carex lanuginosa (30), Solidago canadensis (15), Deschampsia cespitosa (15)
Grazed	23	1.55	4.71		3.71	Poa pratensis (64), Solidago canadensis (11), Carex lanuginosa (10)
Murderers						
Exclosed	32	2.54	12.70	0.59	1.86	Juncus balticus (44), Poa pratensis (23), Carex utriculata (21)
Grazed	21	1.74	5.45		1.63	Juncus balticus (67), Carex nebrascensis (27), Poa pratensis (24)
Summit						
Exclosed	37	2.88	17.74	0.38	1.95	Carex nebrascensis (19), Carex lanuginosa (16), Carex utriculata (14)
Grazed	29	2.21	9.12		3.32	Trifolium longipes (31), Poa pratensis (10), Carex lanuginosa (9)

Tex						
Exclosed	36	2.63	13.86	0.73	2.76	Alnus incana (54), Poa pratensis (16), Aster foliaceus (11)
Grazed	39	2.53	12.59		2.92	Alnus incana (40), Poa pratensis (14), Equisetum arvense (8)
Camp						
Exclosed	51	3.01	20.21	0.63	2.86	Poa pratensis (34), Alnus incana (30), Juncus balticus (24)
Grazed	41	2.80	16.50		3.15	Poa pratensis (51), Trifolium longipes (26), Juncus balticus (16)

	Wetland Indicator status	Bear		Camas		Chesnimnus		Devils		Lower Swamp		Upper Swamp	
Species		Excl	Graz	Excl	Graz	Excl	Graz	Excl	Graz	Excl	Graz	Excl	Graz
Acer glabrum Torr.	FAC						0.28						
Alnus incana ssp. tenuifolia (Nutt.) Breitung	FACW	2.99	3.23	0.00	0.12	2.24	4.61	12.20	1.01	28.97	62.87	29.90	20.25
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roemer	FACU									0.00	1.33		
Artemisia tridentata Nutt. var. vaseyana (Rydb.) Boivin	UPL												
Cornus sericea ssp. sericea L.	FACW					0.00	2.86	0.27	0.00				
Crataegus douglasii Lindl.	FAC					7.57	16.14			0.00	0.93	0.82	0.00
Juniperus occidentalis Hook.	UPL												
Larix occidentalis Nutt.	FACU					0.00	0.24						
Pinus contorta Dougl. ex Loud.	FAC	8.76	8.88	0.81	1.28			2.09	0.00	0.83	2.24		
Picea engelmannii Parry ex Engelm.	FAC								1.10			0.00	4.00
Pinus ponderosa P.& C. Lawson	FACU					1.90	0.00	2.40	0.00	0.00	1.11		
Populus balsamifera L.	FAC					4.47	0.00						
Populus tremuloides Michx.	FAC							0.80	0.00				
Pseudotsuga menziesii (Mirbel) Franco	UPL					0.00	23.73	0.00	1.01				
Ribes aureum Pursh	FAC											0.00	0.52
Ribes cereum Dougl.	FACU					0.00	1.79					0.00	0.79
Ribes hudsonianum Richards.	OBL					1.26	4.10	6.48	0.00				
Ribes lacustre (Pers.) Poir.	FAC					0.00	2.18	3.73	0.00			0.36	0.00
Rosa woodsii Lindl.	FACU					1.3	0.6	0.1	0.0	0.0	1.4	0.0	1.4
Salix spp (english willow)	FACW					20.8	0.0						
Salix bebbiana Sarg.	FACW							0.9	0.0				
Salix boothii Dorn	OBL			2.3	0.0								
Salix exigua Nutt.	OBL					2.7	0.0						
Salix geyeriana Anderss.	FACW	0.0	0.05	0.0	1.2								
Salix lucida Muhl.	FACW	0.2	0.3	0.0	2.5	0.7	0.0	0.0	0.9				
Symphoricarpos albus (L.) Blake	FACU					0.0	0.5	4.8	0.0	0.8	4.8	0.6	0.9
SALIX SP.	FACW												

unknown 1

Total shrub cover (%)	11.9	12.4	3.1	5.1	43.0	57.1	33.9	4.0	30.6	74.6	31.6	27.9
Riparian obligate cover (%)	3.2	3.6	2.3	3.8	32.2	11.6	19.9	1.9	29.0	62.9	29.9	20.3
shrub spp richness	3.0	4.0	2.0	4.0	9.0	11.0	10.0	4.0	3.0	7.0	4.0	6.0
Reach length (m)	260.7	219.5	340.4	320.8	163.2	229.0	112.7	109.2	168.1	198.9	152.2	145.1

Table 6. (continued)

Species	Wetland Indicator status	MF John Day		Murderers		Summit		Tex		Camp	
		Excl	Graz	Excl	Graz	Excl	Graz	Excl	Graz	Excl	Graz
Acer glabrum Torr.	FAC										
Alnus incana ssp. tenuifolia (Nutt.) Breitung	FACW			9.43	0.60	19.53	2.19	72.61	40.85	54.75	18.39
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roemer	FACU										
Artemisia tridentata Nutt. var. vaseyana (Rydb.) Boivin	UPL					0.00	1.31				
Cornus sericea ssp. sericea L.	FACW	0.00	0.34					26.90	1.37	0.00	0.30
Crataegus douglasii Lindl.	FAC										
Juniperus occidentalis Hook.	UPL								0.34		
Larix occidentalis Nutt.	FACU									3.03	3.64
Pinus contorta Dougl. ex Loud.	FAC				0.00	5.30	1.40	0.00	0.00	6.77	3.29
Picea engelmannii Parry ex Engelm.	FAC										
Pinus ponderosa P.& C. Lawson	FACU							0.45	3.66		
Populus balsamifera L.	FAC										
Populus tremuloides Michx.	FAC										
Pseudotsuga menziesii (Mirbel) Franco	UPL							1.73	0.00		
Ribes aureum Pursh	FAC										
Ribes cereum Dougl.	FACU									1.62	0.05
Ribes hudsonianum Richards.	OBL							20.58	13.55	6.36	2.74
Ribes lacustre (Pers.) Poir.	FAC								1.01		1.84
Rosa woodsii Lindl.	FACU								0.4	0.7	
Salix spp (english willow)	FACW										
Salix bebbiana Sarg.	FACW										
Salix boothii Dorn	OBL			0.7	1.0	0.0			0.7		
Salix exigua Nutt.	OBL					0.8	0.5	0.0		0.0	0.3
Salix geyeriana Anderss.	FACW	0.0	1.1	0.5	0.0	0.0	0.6	0.0	0.7		
Salix lucida Muhl.	FACW	0.7	0.2	1.0	0.0	0.3	0.1	2.1	1.9	0.4	
Symphoricarpos albus (L.) Blake	FACU							4.9	8.8	0.0	4.0
SALIX SP.	FACW								0.5		
unknown 1									0.4		

Total shrub cover (%)	0.7	1.6	11.7	1.6	25.9	6.2	129.2	74.1	73.6	34.6
Riparian obligate cover (%)	0.7	1.6	11.7	1.6	20.6	3.5	103.2	59.6	61.5	21.8
Shrub Species richness	1.0	3.0	4.0	2.0	4.0	6.0	7.0	13.0	7.0	9.0
Reach length (m)	153.7	220.6	169.6	149.5	202.8	278.8	168.1	207.7	99.0	100.3

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Appendix 1. Plant species encountered in the 11 study reaches . The scientific name is followed by the Alpha code used in some tables.

Scientific Name	Alpha Code	Scientific Name	Alpha Code	Scientific Name	Alpha Code
Aconitum columbianum Nutt.	ACCO	Erigeron philadelphicus L.	ERPH	Populus balsamifera L.	POBA
Acer glabrum Torr.	ACGL	Festuca arundinacea Schreb.	FEAR	Populus tremuloides Michx.	POTR
Achillea millefolium L.	ACMI	Festuca idahoensis Elmer	FEID	Potentilla gracilis Dougl. ex Hook.	POGR
Agrostis stolonifera L.	AGST	Festuca rubra L.	FERU	Polemonium occidentale Greene	POOOC
Agastache urticifolia (Benth.) Kuntze	AGUR	Fragaria virginiana Duchesne	FRVI	Poa pratensis L.	POPR
Alopecurus aequalis Sobol.	ALAE	Galium aparine L.	GAAP	Prunella vulgaris L.	PRVU
Alnus incana ssp. tenuifolia (Nutt.) Breitung	ALIN	Galium bifolium S. Wats.	GABI	Pseudotsuga menziesii (Mirbel) Franco	PSME
Alopecurus pratensis L.	ALPR	Galium boreale L.	GABO	Ranunculus macounii Britt.	RAMA
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roemer	AMAL	Geum macrophyllum Willd.	GEMA	Ribes aureum Pursh	RIAU
Angelica arguta Nutt.	ANAR	Geranium richardsonii Fisch. & Trautv.	GERI	Ribes cereum Dougl.	RICE
Antennaria rosea Greene	ANRO	Glyceria striata (Lam.) A.S. Hitchc.	GLST	Ribes hudsonianum Richards.	RIHU
Aquilegia formosa Fisch. ex DC.	AQFO	Heracleum lanatum Michx.	HELA	Ribes lacustre (Pers.) Poir.	RILA
Arnica chamissonis Less.	ARCH	Hypericum anagalloides Cham. & Schlecht.	HYAN	Rorippa nasturtium-aquaticum (L.) Hayek	RONA-AQ
Arnica cordifolia Hook.	ARCO	Hypericum perforatum L.	HYPE	Rosa woodsii Lindl.	ROWO
Artemisia ludoviciana Nutt.	ARLU	Iris missouriensis Nutt.	IRMI	Rumex acetosella L.	RUAC
Artemisia tridentata Nutt. var. vaseyana (Rydb.) Boivin	ARTR-V	Juncus balticus Willd.	JUBA	Rumex crispus L.	RUCR
Aster foliaceus Lindl. ex DC.	ASFO	Juncus ensifolius Wikstr.	JUEN	Salix alba L.	SAAL
Bromus carinatus Hook. & Arn.	BRCA	Juncus nevadensis S. Wats.	JUNE	Saxifraga arguta auct. non D. Don	SAAR
Cardamine cordifolia Gray	CACO	Juniperus occidentalis Hook.	JUOC	Salix bebbiana Sarg.	SABE
Carex geyeri Boott	CAGE	Larix occidentalis Nutt.	LAOC	Salix boothii Dorn	SABO
Carex lanuginosa Michx.	CALA	Leucanthemum vulgare Lam.	LEVU	Salix exigua Nutt.	SAEX
Carex lenticularis var. lipocarpa (Holm) L.A. Standley	CALE	Ligusticum grayi Coult. & Rose	LIGR	Salix geyeriana Anderss.	SAGE
Carex luzulina Olney	CALU	Lolium multiflorum Lam.	LOMU	Salix lucida Muhl.	SALU
Carex microptera Mackenzie	CAMI	Lolium perenne L.	LOPE	Saxifraga rhomboidea Greene	SARH
Carex nebrascensis Dewey	CANE	Lotus purshianus F.E. & E.G. Clem.	LOPU	Scirpus microcarpus J.& K. Presl	SCMI
Carex praegracilis W. Boott	CAPR	Lupinus leucophyllus Dougl. ex Lindl.	LULE	Senecio pseudaureus Rydb.	SEPS
Carex stipata Muhl. ex Willd.	CAST	Mentha arvensis L.	MEAR	Senecio triangularis Hook.	SETR
Carex utriculata Boott	CAUT	Mertensia ciliata (James ex Torr.) G. Don	MECI	Sidalcea oregana (Nutt. ex Torr. & Gray) Gray	SIOR
Cerastium arvense L.	CEAR	Medicago lupulina L.	MELU	Smilacina stellata (L.) Desf.	SMST
Chrysothamnus nauseosus (Pallas ex Pursh) Britt.	CHNA	Mimulus guttatus DC.	MIGU	Solidago canadensis L.	SOCA
Cirsium callilepis (Greene) Jepson	CICA	Mimulus moschatus Dougl. ex Lindl.	MIMO	Stellaria longipes Goldie	STLO
Cirsium scariosum Nutt.	CISC	Monardella odoratissima Benth.	MOOD	Stipa occidentalis Thurb. ex S. Wats.	STOC
Cornus sericea ssp. sericea L.	COSE	Myosotis scorpioides L.	MYSC	Symphoricarpos albus (L.) Blake	SYAL
Crataegus douglasii Lindl.	CRDO	Osmorhiza occidentalis (Nutt. ex Torr. & Gray) Torr.	OSOC	Taraxacum officinale G.H. Weber ex Wiggers	TAOF
Dactylis glomerata L.	DAGL	Phleum pratense L.	PHPR	Thermopsis montana Nutt.	THMO
Deschampsia cespitosa (L.) Beauv.	DECE	Pinus contorta Dougl. ex Loud.	PICO	Thalictrum occidentale Gray	THOC
Dipsacus sylvestris Huds.	DISY	Picea engelmannii Parry ex Engelm.	PIEN	Trifolium cyathiferum Lindl.	TRCY
Eleocharis palustris (L.) Roemer & J.A. Schultes	ELPA	Pinus ponderosa P.& C. Lawson	PIPO	Trifolium longipes Nutt.	TRLO
Epilobium ciliatum Raf.	EPCI	Platanthera dilatata (Pursh) Lindl. ex Beck	PLDI	Trifolium repens L.	TRRE

Equisetum arvense L.	EQAR	Plantago lanceolata L.	PLLA	Urtica dioica L.	URDI
Equisetum laevigatum A. Braun	EQLA	Plantago major L.	PLMA	Veronica americana Schwein. ex Benth.	VEAM
				Veronica anagallis-aquatica L.	VEAN
				Veratrum californicum Dur.	VECA
				Verbascum thapsus L.	VETH
				Viola adunca Sm.	VIAD
				Vicia americana Muhl. ex Willd.	VIAM

Appendix 2. Plant species cover (%) the 11 study reaches. Numbers are the mean and Standard error based upon their cover in at least 20 2 X 5 m plots for each study reach.

	Bear					Upper Swamp			
	Exclosed		Grazed			Exclosed		Grazed	
	Mean	SE	Mean	SE		Mean	SE	Mean	SE
Achillea millefolium L.	2.0	0.8	4.0	1.3	Achillea millefolium L.	0.1	0.1	0.1	0.1
Agrostis stolonifera L.	1.6	0.7	5.1	2.0	Agrostis stolonifera L.	17.9	2.6	8.0	2.0
Alopecurus aequalis Sobol.	0.1	0.1			Alopecurus aequalis Sobol.			1.2	0.5
Alnus incana ssp. tenuifolia (Nutt.) Breitung	1.3	0.8	1.9	1.0	Alnus incana ssp. tenuifolia (Nutt.) Breitung	12.9	3.6	6.6	2.5
Antennaria rosea Greene	0.0	0.0			Carex geyeri Boott	1.9	1.4		
Aquilegia formosa Fisch. ex DC.	0.0	0.0	0.1	0.1	Carex lanuginosa Michx.	4.9	2.1		
Arnica chamissonis Less.	2.3	0.7	2.2	0.7	Carex nebrascensis Dewey	6.6	2.6	2.5	1.1
Aster foliaceus Lindl. ex DC.	5.5	1.5	2.5	0.7	Carex utriculata Boott	12.3	3.9	1.0	0.7
Bromus carinatus Hook. & Arn.	2.7	1.0			Cirsium callilepis (Greene) Jepson	2.4	0.7		
Carex lanuginosa Michx.	9.0	2.5	9.4	3.1	Crataegus douglasii Lindl.	0.4	0.4		
Carex lenticularis var. lipocarpa (Holm) L.A. Standley	2.2	1.0	0.2	0.2	Dipsacus sylvestris Huds.	2.8	1.0	0.4	0.4
Carex microptera Mackenzie	6.1	2.5	4.3	1.5	Epilobium ciliatum Raf.	3.4	0.9	0.9	0.4
Carex nebrascensis Dewey	1.5	0.8	0.8	0.8	Equisetum arvense L.	2.1	1.1	1.0	0.7
Carex utriculata Boott	8.6	2.7	0.9	0.7	Erigeron philadelphicus L.	15.4	2.8	0.3	0.2
Cirsium scariosum Nutt.	0.0	0.0	0.1	0.1	Fragaria virginiana Duchesne			0.2	0.2
Deschampsia cespitosa (L.) Beauv.	3.6	1.3	6.0	1.5	Galium aparine L.	3.1	1.7	1.1	0.7
Eleocharis palustris (L.) Roemer & J.A. Schultes	0.1	0.1	0.6	0.5	Galium bifolium S. Wats.	0.2	0.2		
Epilobium ciliatum Raf.	0.6	0.3	0.1	0.1	Geum macrophyllum Willd.	0.9	0.4	0.1	0.1
Equisetum arvense L.	9.4	2.7	18.7	3.2	Glyceria striata (Lam.) A.S. Hitchc.	9.2	2.1	0.1	0.1
Erigeron philadelphicus L.	0.1	0.1	1.2	0.4	Juncus balticus Willd.	11.9	3.2	12.7	2.9
Festuca idahoensis Elmer	0.4	0.3			Juncus ensifolius Wikstr.	1.1	0.5		
Festuca rubra L.	1.4	1.1			Mentha arvensis L.	2.6	0.8	0.1	0.1
Fragaria virginiana Duchesne	9.8	2.6	5.7	1.8	Mimulus guttatus DC.	0.8	0.3		
Galium boreale L.	2.0	1.1	0.9	0.5	Myosotis scorpioides L.	2.7	1.3		
Glyceria striata (Lam.) A.S. Hitchc.	0.1	0.1	0.1	0.1	Phleum pratense L.	3.9	1.0	1.0	0.5
Hypericum anagalloides Cham. & Schlecht.	3.9	1.4	8.3	3.0	Picea engelmannii Parry ex Engelm.			2.6	1.7
Juncus balticus Willd.	1.2	0.5	8.0	2.8	Plantago major L.			3.7	0.7
Juncus ensifolius Wikstr.	0.4	0.3	0.5	0.3	Potentilla gracilis Dougl. ex Hook.	0.9	0.6		
Juncus nevadensis S. Wats.	0.0	0.0	0.2	0.2	Poa pratensis L.	4.1	1.3	9.3	2.8
Lolium multiflorum Lam.	0.5	0.3			Ribes cereum Dougl.			0.1	0.1
Mentha arvensis L.	0.3	0.2	0.5	0.3	Ribes lacustre (Pers.) Poir.	0.2	0.2		

Mimulus moschatus Dougl. ex Lindl.	0.1	0.1			Rosa woodsii Lindl.			0.4	0.4
Phleum pratense L.	0.0	0.0	0.9	0.5	Scirpus microcarpus J.& K. Presl	4.0	2.4	0.1	0.1
Pinus contorta Dougl. ex Loud.	7.5	3.5	6.3	2.2	Senecio pseudaureus Rydb.	0.2	0.2	0.4	0.3
Plantago major L.	0.0	0.0	0.3	0.2	Smilacina stellata (L.) Desf.	0.1	0.1		
Potentilla gracilis Dougl. ex Hook.	0.2	0.2	0.3	0.3	Stipa occidentalis Thurb. ex S. Wats.	0.1	0.1		
Poa pratensis L.	10.0	2.1	15.1	3.2	Symphoricarpos albus (L.) Blake	0.3	0.3		
Prunella vulgaris L.	2.7	1.5	7.7	2.3	Taraxacum officinale G.H. Weber ex Wiggers			0.9	0.4
Ranunculus macounii Britt.	0.2	0.2	3.8	1.9	Trifolium longipes Nutt.	1.8	1.0	3.2	1.4
Rumex acetosella L.			0.2	0.2	Trifolium repens L.	0.7	0.4	8.7	2.1
Salix geyeriana Anderss.			0.8	0.4	Veronica americana Schwein. ex Benth.			3.4	1.7
Salix lucida Muhl.	0.1	0.1	0.3	0.3	Veronica anagallis-aquatica L.	0.1	0.1	0.9	0.4
Scirpus microcarpus J.& K. Presl	1.1	1.1	1.0	0.8	Veratrum californicum Dur.	0.8	0.4	0.3	0.2
Senecio pseudaureus Rydb.	0.9	0.6							
Solidago canadensis L.	0.7	0.5							
Sidalcea oregana (Nutt. ex Torr. & Gray) Gray			0.2	0.2					
Stellaria longipes Goldie	0.4	0.2							
Stipa occidentalis Thurb. ex S. Wats.	0.2	0.2	0.5	0.5					
Taraxacum officinale G.H. Weber ex Wiggers	0.4	0.3	4.4	1.6					
Trifolium longipes Nutt.	9.3	2.4	8.5	2.4					
Trifolium repens L.	1.1	0.8	16.2	3.7					

	Camas					Camp Creek			
	Exclosed		Grazed			Exclosed		Grazed	
	Mean	SE	Mean	SE		Mean	SE	Mean	SE
Achillea millefolium L.	2.5	1.1	3.4	1.0	Aconitum columbianum Nutt.			0.4	0.3
Agrostis stolonifera L.	5.0	2.0	4.8	1.0	Achillea millefolium L.	1.0	0.5	1.7	0.7
Alopecurus aequalis Sobol.	0.1	0.1			Agrostis stolonifera L.	3.3	0.8	5.6	2.1
Alopecurus pratensis L.			2.7	1.8	Alopecurus aequalis Sobol.			0.0	0.0
Arnica chamissonis Less.	0.4	0.2	0.3	0.2	Alnus incana ssp. tenuifolia (Nutt.) Breitung	30.5	4.4	10.1	3.8
Aster foliaceus Lindl. ex DC.	0.7	0.3	0.2	0.2	Angelica arguta Nutt.	6.3	2.5	0.1	0.1
Carex geyeri Boott			0.5	0.5	Aquilegia formosa Fisch. ex DC.	0.1	0.1	0.4	0.4
Carex lanuginosa Michx.	7.3	3.0	5.6	2.0	Aster foliaceus Lindl. ex DC.	2.6	0.6	9.4	1.8
Carex lenticularis var. lipocarpa (Holm) L.A. Standley	3.9	1.8	0.6	0.4	Carex lanuginosa Michx.	0.2	0.2	0.2	0.2
Carex microptera Mackenzie	0.1	0.1			Carex lenticularis var. lipocarpa (Holm) L.A. Standley	3.9	1.0	2.8	0.8
Carex utriculata Boott	2.4	2.4			Carex microptera Mackenzie	4.4	0.9	0.3	0.2
Cirsium callilepis (Greene) Jepson	0.4	0.3			Carex praegracilis W. Boott	0.3	0.2	0.3	0.3
Deschampsia cespitosa (L.) Beauv.	2.3	1.6	0.1	0.1	Cirsium scariosum Nutt.	0.1	0.1		
Eleocharis palustris (L.) Roemer & J.A. Schultes	2.1	1.2	4.0	1.9	Dactylis glomerata L.			0.1	0.1
Epilobium ciliatum Raf.	3.7	0.8	1.9	0.5	Eleocharis palustris (L.) Roemer & J.A. Schultes	1.7	0.9		
Equisetum arvense L.	3.9	1.7	1.7	0.6	Epilobium ciliatum Raf.	0.4	0.2	1.7	0.5
Equisetum laevigatum A. Braun	1.0	0.6			Equisetum arvense L.	9.4	2.9	6.0	1.7
Erigeron philadelphicus L.	4.5	1.8	0.7	0.4	Equisetum laevigatum A. Braun	0.1	0.1		
Festuca rubra L.	3.9	1.9			Erigeron philadelphicus L.	10.9	2.2	0.7	0.3
Fragaria virginiana Duchesne	0.1	0.1	0.4	0.3	Fragaria virginiana Duchesne	7.9	2.8	10.7	3.5
Galium boreale L.	2.1	0.7	0.6	0.3	Galium aparine L.	14.9	3.0	2.2	1.1
Glyceria striata (Lam.) A.S. Hitchc.	0.7	0.3	0.5	0.2	Galium boreale L.	0.9	0.5	0.3	0.2
Juncus balticus Willd.	5.0	2.2	3.2	1.5	Glyceria striata (Lam.) A.S. Hitchc.	4.2	0.9	5.1	1.2
Juncus ensifolius Wikstr.	0.6	0.3	0.3	0.3	Heracleum lanatum Michx.	7.4	2.2		
Lolium multiflorum Lam.	0.5	0.2	0.3	0.2	Hypericum anagalloides Cham. & Schlecht.	0.1	0.1		
Lotus purshianus F.E. & E.G. Clem.			4.0	2.6	Juncus balticus Willd.	24.4	4.6	15.7	3.6
Lupinus leucophyllus Dougl. ex Lindl.	1.9	1.2	1.3	0.7	Juncus ensifolius Wikstr.	1.1	0.5	0.5	0.3

Mentha arvensis L.	4.9	1.1	5.1	1.4	Juncus nevadensis S. Wats.	0.5	0.3		
Medicago lupulina L.			0.3	0.2	Juniperus occidentalis Hook.	0.4	0.4		
Mimulus guttatus DC.	1.5	0.7	0.6	0.4	Larix occidentalis Nutt.	2.8	2.0	4.5	2.5
Phleum pratense L.	3.8	1.2	6.8	2.1	Lolium multiflorum Lam.	0.9	0.6		
Pinus contorta Dougl. ex Loud.	1.0	1.0	1.9	1.3	Lotus purshianus F.E. & E.G. Clem.	0.2	0.2		
Plantago lanceolata L.			0.3	0.2	Mentha arvensis L.	6.7	1.9	3.3	1.2
Plantago major L.			0.8	0.4	Mimulus guttatus DC.	0.5	0.3	0.5	0.3
Potentilla gracilis Dougl. ex Hook.	1.8	0.6	3.1	0.8	Mimulus moschatus Dougl. ex Lindl.	0.1	0.1	0.4	0.2
Poa pratensis L.	19.4	4.2	15.1	3.3	Phleum pratense L.	7.6	1.3	5.3	0.9
Prunella vulgaris L.	0.1	0.1	0.4	0.3	Pinus contorta Dougl. ex Loud.	5.8	2.7	2.0	2.0
Rumex acetosella L.			0.1	0.1	Platanthera dilatata (Pursh) Lindl. ex Beck	0.2	0.2		
Rumex crispus L.	0.6	0.4	0.1	0.1	Plantago major L.			0.7	0.3
Salix boothii Dorn	2.6	2.1			Potentilla gracilis Dougl. ex Hook.	0.2	0.2	0.4	0.4
Salix geyeriana Anderss.	0.1	0.1	0.5	0.4	Polemonium occidentale Greene	0.4	0.2		
Salix lucida Muhl.	0.1	0.1	1.9	1.5	Poa pratensis L.	33.5	3.1	50.8	3.1
Scirpus microcarpus J.& K. Presl	4.6	1.9	6.7	2.4	Prunella vulgaris L.	0.7	0.3	7.0	1.2
Senecio pseudaureus Rydb.			0.7	0.5	Ribes cereum Dougl.	0.3	0.2	0.7	0.5
Sidalcea oregana (Nutt. ex Torr. & Gray) Gray	0.4	0.3	0.3	0.2	Ribes hudsonianum Richards.	2.6	1.3	1.9	0.9
Smilacina stellata (L.) Desf.	0.1	0.1			Salix lucida Muhl.	0.3	0.3		
Solidago canadensis L.	0.7	0.4			Scirpus microcarpus J.& K. Presl	1.0	0.8	4.2	1.7
Stipa occidentalis Thurb. ex S. Wats.			0.1	0.1	Senecio pseudaureus Rydb.	3.9	0.9	2.6	1.0
Taraxacum officinale G.H. Weber ex Wiggers	0.2	0.2	0.2	0.2	Sidalcea oregana (Nutt. ex Torr. & Gray) Gray	0.1	0.1		
Thermopsis montana Nutt.	2.2	1.4			Smilacina stellata (L.) Desf.	4.8	2.4	0.4	0.3
Thalictrum occidentale Gray	0.5	0.3	0.1	0.1	Symphoricarpos albus (L.) Blake			5.9	2.8
Trifolium cyathiferum Lindl.	0.5	0.3	0.3	0.2	Taraxacum officinale G.H. Weber ex Wiggers	1.4	0.4	3.8	1.1
Thermopsis montana Nutt.			2.7	1.8	Trifolium longipes Nutt.	21.2	4.3	25.7	3.9
Trifolium repens L.	0.9	0.5	7.0	2.1	Trifolium repens L.			2.4	1.2
Veratrum californicum Dur.	0.1	0.1	0.4	0.4	Urtica dioica L.	0.1	0.1		
					Viola adunca Sm.	0.5	0.3		
					Vicia americana Muhl. ex Willd.	0.5	0.2		

	Chesnimnus					Tex Creek			
	Exclosed		Grazed			Exclosed		Grazed	
	Mean	SE	Mean	SE		Mean	SE	Mean	SE
Aconitum columbianum Nutt.			0.1	0.1	Aconitum columbianum Nutt.	0.1	0.1		
Acer glabrum Torr.			1.1	1.1	Achillea millefolium L.	0.4	0.4	1.0	0.5
Achillea millefolium L.			0.3	0.2	Agrostis stolonifera L.	4.3	1.5	4.4	1.6
Agrostis stolonifera L.	0.3	0.2	0.4	0.2	Agastache urticifolia (Benth.) Kuntze			0.3	0.3
Alopecurus aequalis Sobol.	0.3	0.3	0.3	0.2	Alnus incana ssp. tenuifolia (Nutt.) Breitung	53.8	6.1	40.4	7.2
Alnus incana ssp. tenuifolia (Nutt.) Breitung	1.3	1.0	5.2	3.3	Amelanchier alnifolia (Nutt.) Nutt. ex M. Roemer			0.3	0.3
Alopecurus pratensis L.			0.5	0.3	Angelica arguta Nutt.	0.3	0.2		
Angelica arguta Nutt.	0.9	0.9	2.9	1.1	Antennaria rosea Greene	0.8	0.8		
Arnica cordifolia Hook.	2.3	1.6	1.2	0.5	Aster foliaceus Lindl. ex DC.	11.4	4.1	2.1	1.0
Artemisia ludoviciana Nutt.	0.8	0.4			Cardamine cordifolia Gray	0.3	0.3		
Carex lanuginosa Michx.	4.6	2.7			Carex geyeri Boott	2.5	2.5		
Carex microptera Mackenzie	0.3	0.2	0.6	0.4	Carex lanuginosa Michx.	0.9	0.9		
Carex utriculata Boott	0.9	0.9			Carex microptera Mackenzie	2.2	1.0	3.8	2.1
Cirsium callilepis (Greene) Jepson	0.2	0.2			Carex praegracilis W. Boott			0.0	0.0
Cornus sericea ssp. sericea L.			1.8	1.8	Cornus sericea ssp. sericea L.	2.4	0.9	1.5	1.2
Crataegus douglasii Lindl.	5.5	2.9	7.6	3.5	Crataegus douglasii Lindl.	0.6	0.6		
Deschampsia cespitosa (L.) Beauv.	0.2	0.2	0.1	0.1	Epilobium ciliatum Raf.	0.9	0.6	0.3	0.3
Eleocharis palustris (L.) Roemer & J.A. Schultes	1.1	0.6	0.5	0.3	Equisetum arvense L.	9.4	3.4	8.3	3.7
Epilobium ciliatum Raf.	0.5	0.3	0.2	0.2	Festuca arundinacea Schreb.	0.3	0.3	0.3	0.2
Equisetum arvense L.	30.0	4.1	0.6	0.4	Galium aparine L.	7.6	2.8	0.8	0.4
Erigeron philadelphicus L.	4.7	1.0			Galium boreale L.			0.9	0.5
Fragaria virginiana Duchesne	0.4	0.3	1.7	0.9	Geum macrophyllum Willd.	0.3	0.2	0.2	0.2
Galium aparine L.	1.9	1.5			Geranium richardsonii Fisch. & Trautv.			0.3	0.2
Glyceria striata (Lam.) A.S. Hitchc.	1.1	0.6	0.2	0.2	Glyceria striata (Lam.) A.S. Hitchc.	9.5	2.2	2.9	1.6
Juncus balticus Willd.			0.1	0.1	Heracleum lanatum Michx.	7.7	2.5		
Juncus ensifolius Wikstr.	0.3	0.2	0.1	0.1	Juncus balticus Willd.			3.0	2.2
Lolium multiflorum Lam.	4.4	2.6			Juncus nevadensis S. Wats.			0.4	0.2
Mentha arvensis L.	1.8	0.5	0.8	0.3	Juniperus occidentalis Hook.			0.3	0.3
Monardella odoratissima Benth.	0.2	0.2			Ligusticum grayi Coult. & Rose			0.0	0.0

Myosotis scorpioides L.	7.1	2.2	8.4	2.3	Mentha arvensis L.	6.5	1.9	3.1	1.2
Phleum pratense L.	3.3	1.0	1.7	0.5	Medicago lupulina L.			4.3	2.1
Pinus contorta Dougl. ex Loud.	0.2	0.2			Myosotis scorpioides L.	2.1	2.1		
					Osmorhiza occidentalis (Nutt. ex Torr. & Gray)				
Plantago major L.	0.7	0.3	4.7	1.0	Torr.			0.4	0.4
Populus balsamifera L.	4.9	2.6			Phleum pratense L.	0.1	0.1	0.4	0.2
Potentilla gracilis Dougl. ex Hook.	0.8	0.4	0.5	0.3	Pinus ponderosa P.& C. Lawson	2.8	1.8	5.0	3.5
					Potentilla gracilis				
Poa pratensis L.	21.4	4.3	27.2	4.1	Dougl. ex Hook.	0.3	0.2		
Pseudotsuga menziesii (Mirbel) Franco			11.0	3.2	Poa pratensis L.	16.1	3.6	14.3	3.1
					Pseudotsuga menziesii				
Ribes aureum Pursh			1.1	0.8	(Mirbel) Franco	2.1	2.1		
					Ribes hudsonianum				
Ribes cereum Dougl.			2.8	1.7	Richards.	9.8	3.2	6.5	2.5
Ribes hudsonianum Richards.	0.2	0.2	2.5	1.6	Ribes lacustre (Pers.) Poir.			3.0	2.0
Ribes lacustre (Pers.) Poir.			0.4	0.3	Rosa woodsii Lindl.			0.9	0.8
Rosa woodsii Lindl.	1.3	1.3	0.7	0.7	Rumex crispus L.	0.9	0.6		
Rumex crispus L.	0.1	0.1			Salix bebbiana Sarg.	0.1	0.1		
					Salix geyeriana				
Salix alba L.	17.7	4.9			Anderss.			0.1	0.1
Salix bebbiana Sarg.	0.6	0.6			Salix lucida Muhl.	0.3	0.3	0.3	0.2
					Smilacina stellata (L.)				
Salix exigua Nutt.	3.1	2.3			Desf.	4.4	2.6	2.1	1.1
					Stipa occidentalis				
Salix lucida Muhl.	2.5	1.8	0.2	0.2	Thurb. ex S. Wats.			0.1	0.1
Scirpus microcarpus J.& K. Presl	1.8	1.5			Symphoricarpos albus (L.) Blake	4.6	2.0	7.6	2.4
					Taraxacum officinale				
Sidalcea oregana (Nutt. ex Torr. & Gray) Gray	0.1	0.1			G.H. Weber ex Wiggers	0.2	0.2	0.7	0.3
					Thalictrum occidentale				
Solidago canadensis L.	0.1	0.1			Gray			0.4	0.2
Stipa occidentalis Thurb. ex S. Wats.	0.5	0.3			Trifolium longipes Nutt.			1.3	0.7
Symphoricarpos albus (L.) Blake	1.2	0.8	0.6	0.4	Trifolium repens L.			0.6	0.4
					Veronica americana				
Trifolium repens L.	1.8	0.9	10.9	3.3	Schwein. ex Benth.	1.2	0.7		
Veronica americana Schwein. ex Benth.	0.7	0.5			Vicia americana Muhl. ex Willd.	1.9	1.9		
Veratrum californicum Dur.	0.4	0.3	0.1	0.1					

	Devil's					Summit Creek			
	Exclosed		Grazed			Exclosed		Grazed	
	Mean	SE	Mean	SE		Mean	SE	Mean	SE
Achillea millefolium L.	1.5	0.6	1.3	0.5	Achillea millefolium L.	1.6	0.7	2.0	0.9
Agrostis stolonifera L.	15.0	2.7	1.8	0.5	Agrostis stolonifera L.			0.3	0.3
Alopecurus aequalis Sobol.	0.4	0.4	0.3	0.3	Alnus incana ssp. tenuifolia (Nutt.) Breitung	13.6	4.2	0.1	0.1
Alnus incana ssp. tenuifolia (Nutt.) Breitung	9.7	2.7	0.5	0.5	Antennaria rosea Greene	1.0	0.7	0.4	0.3
Angelica arguta Nutt.	0.8	0.4	0.1	0.1	Arnica chamissonis Less.	2.6	1.1		
Carex geyeri Boott	3.4	1.4			Artemisia ludoviciana Nutt.	0.1	0.1		
Carex lanuginosa Michx.	6.4	2.5			Artemisia tridentata Nutt. var. vaseyana (Rydb.) Boivin			3.8	2.0
Carex lenticularis var. lipocarpa (Holm) L.A. Standley	1.3	0.5			Aster foliaceus Lindl. ex DC.	0.2	0.2	1.5	0.8
Carex microptera Mackenzie	1.6	0.6	0.2	0.2	Carex lanuginosa Michx.	16.1	3.9	8.5	3.3
Carex utriculata Boott	0.3	0.3			Carex microptera Mackenzie	0.9	0.6	0.8	0.8
Cirsium callilepis (Greene) Jepson	0.2	0.2			Carex nebrascensis Dewey	18.6	4.5	5.4	1.7
Cornus sericea ssp. sericea L.	0.2	0.2			Carex utriculata Boott	13.9	3.8		
Deschampsia cespitosa (L.) Beauv.	0.2	0.2			Chrysothamnus nauseosus (Pallas ex Pursh) Britt.			0.3	0.3
Eleocharis palustris (L.) Roemer & J.A. Schultes	0.6	0.6			Cirsium callilepis (Greene) Jepson	0.5	0.4		
Epilobium ciliatum Raf.	2.7	1.3	1.0	0.4	Eleocharis palustris (L.) Roemer & J.A. Schultes	7.7	2.5	3.7	1.0
Equisetum arvense L.	4.7	2.1	3.4	1.4	Epilobium ciliatum Raf.	2.5	1.4	0.3	0.2
Fragaria virginiana Duchesne	1.7	0.9	1.7	0.9	Equisetum arvense L.			0.2	0.2
Galium aparine L.	1.9	0.9			Erigeron philadelphicus L.	0.7	0.5	0.2	0.2
Galium boreale L.			0.1	0.1	Festuca idahoensis Elmer	0.5	0.3		
Geum macrophyllum Willd.			0.1	0.1	Festuca rubra L.	0.3	0.3		
Glyceria striata (Lam.) A.S. Hitchc.	3.6	0.9	0.6	0.3	Fragaria virginiana Duchesne	0.7	0.4	1.0	0.5
Heracleum lanatum Michx.	0.1	0.1			Galium boreale L.	0.5	0.3		
Juncus ensifolius Wikstr.	0.6	0.3	0.2	0.2	Glyceria striata (Lam.) A.S. Hitchc.	1.7	1.3		
Lolium multiflorum Lam.			0.4	0.2	Hypericum anagalloides Cham. & Schlecht.	1.3	0.7	2.1	0.9
Lolium perenne L.	0.2	0.2			Hypericum perforatum L.	0.3	0.3	0.0	0.0
Mentha arvensis L.	4.0	0.8	0.1	0.1	Juncus balticus Willd.	5.8	2.1	6.0	1.7
Mertensia ciliata (James ex Torr.) G. Don	0.2	0.2			Juncus ensifolius Wikstr.	1.3	0.6	0.2	0.2
Mimulus moschatus Dougl. ex Lindl.	0.1	0.1			Juncus nevadensis S. Wats.	0.2	0.2		
Myosotis scorpioides L.	32.1	4.1	8.0	1.4	Lolium multiflorum Lam.	0.3	0.3		

Phleum pratense L.	6.6	1.6	17.1	2.4	Mentha arvensis L.	5.8	1.3	0.7	0.3
Pinus contorta Dougl. ex Loud.	5.6	3.3			Medicago lupulina L.				
Picea engelmannii Parry ex Engelm.			1.6	1.6	Mimulus guttatus DC.	0.2	0.2	0.1	0.1
Plantago major L.			2.3	0.9	Mimulus moschatus Dougl. ex Lindl.	0.1	0.1		
Populus balsamifera L.	0.6	0.6			Phleum pratense L.	0.1	0.1		
Potentilla gracilis Dougl. ex Hook.	0.1	0.1	1.3	0.5	Pinus contorta Dougl. ex Loud.	6.3	3.2		
Poa pratensis L.	8.9	2.4	36.5	2.8	Potentilla gracilis Dougl. ex Hook.	3.1	1.5	0.1	0.1
Pseudotsuga menziesii (Mirbel) Franco			1.3	1.3	Polemonium occidentale Greene				
Ribes aureum Pursh	4.9	2.8			Poa pratensis L.	4.8	2.0	9.5	2.8
Ribes lacustre (Pers.) Poir.	1.7	1.0			Prunella vulgaris L.			2.9	1.3
Rumex acetosella L.	0.2	0.2			Salix exigua Nutt.	2.5	1.6		
Saxifraga arguta auct. non D. Don			3.6	1.9	Salix geyeriana Anderss.	0.1	0.1		
Salix bebbiana Sarg.	0.4	0.4			Salix lucida Muhl.	1.5	1.4		
Salix boothii Dorn	0.1	0.1			Solidago canadensis L.	7.2	3.1		
Salix exigua Nutt.	0.2	0.2			Stipa occidentalis Thurb. ex S. Wats.			0.2	0.2
Salix lucida Muhl.			0.6	0.6	Taraxacum officinale G.H. Weber ex Wiggers			0.2	0.2
Scirpus microcarpus J.& K. Presl	0.1	0.1	0.2	0.2	Trifolium longipes Nutt.	5.0	1.7	30.5	5.4
Senecio pseudaureus Rydb.	0.7	0.4							
Senecio triangularis Hook.	2.5	1.0	1.5	0.9					
Stipa occidentalis Thurb. ex S. Wats.	0.3	0.2							
Symphoricarpos albus (L.) Blake	2.9	2.0							
Taraxacum officinale G.H. Weber ex Wiggers	0.3	0.2							
Trifolium repens L.	5.2	2.5	19.4	3.1					
Veratrum californicum Dur.	0.6	0.3	1.4	0.9					

	Lower Swamp Creek					Murderer's Creek			
	Exclosed		Grazed			Exclosed		Grazed	
	Mean	SE	Mean	SE		Mean	SE	Mean	SE
Achillea millefolium L.	0.4	0.3	0.3	0.2	Achillea millefolium L.	0.3	0.3	1.5	1.1
Agrostis stolonifera L.	24.9	4.5	10.0	2.2	Agrostis stolonifera L.	1.5	0.5	1.3	0.5
Agastache urticifolia (Benth.) Kuntze			0.1	0.1	Alnus incana ssp. tenuifolia (Nutt.) Breitung	8.1	3.4		
Alnus incana ssp. tenuifolia (Nutt.) Breitung	18.4	4.7	52.3	6.5	Antennaria rosea Greene			0.2	0.2
Amelanchier alnifolia (Nutt.) Nutt. ex M. Roemer			0.3	0.3	Aster foliaceus Lindl. ex DC.	9.2	3.2	2.0	1.2
Carex geyeri Boott	7.7	3.0	1.0	0.9	Carex lanuginosa Michx.	2.4	1.5		
Carex microptera Mackenzie	0.2	0.2			Carex microptera Mackenzie	0.5	0.3		
Carex nebrascensis Dewey	1.3	1.3			Carex nebrascensis Dewey	5.2	1.8	26.5	5.7
Carex utriculata Boott	21.3	5.6	2.8	2.4	Carex utriculata Boott	20.5	4.4	4.6	2.3
Carex stipata Muhl. ex Willd.	0.8	0.5			Cirsium callilepis (Greene) Jepson	0.9	0.3		
Cirsium callilepis (Greene) Jepson	2.4	0.7			Dipsacus sylvestris Huds.	0.1	0.1		
Cerastium arvense L.	0.8	0.4			Epilobium ciliatum Raf.	4.3	1.0	2.1	0.6
Crataegus douglasii Lindl.			2.1	1.3	Equisetum arvense L.	0.5	0.3		
Dipsacus sylvestris Huds.	0.3	0.3			Erigeron philadelphicus L.	0.4	0.3	0.1	0.1
Epilobium ciliatum Raf.	3.2	0.8	0.2	0.2	Festuca rubra L.	2.8	2.4	2.4	1.9
Equisetum arvense L.	1.9	0.9			Galium aparine L.	0.9	0.6		
Erigeron philadelphicus L.	11.6	2.7			Galium boreale L.	0.2	0.2		
Fragaria virginiana Duchesne	0.6	0.3	0.9	0.4	Geum macrophyllum Willd.	1.3	0.5		
Galium aparine L.	3.1	1.5	0.5	0.3	Glyceria striata (Lam.) A.S. Hitchc.	13.5	3.5	2.3	1.1
Galium boreale L.			0.2	0.2	Juncus balticus Willd.	43.7	5.9	66.9	5.0
Geum macrophyllum Willd.	1.1	0.6	3.1	0.9	Juncus nevadensis S. Wats.			0.3	0.3
Glyceria striata (Lam.) A.S. Hitchc.	6.3	2.5	0.6	0.6	Mentha arvensis L.	6.6	2.3		
Juncus balticus Willd.	14.3	4.4	0.3	0.3	Medicago lupulina L.	1.1	0.6	1.3	0.9
Juncus ensifolius Wikstr.	0.4	0.3			Mimulus guttatus DC.	0.5	0.3		
Mentha arvensis L.	0.8	0.4	0.2	0.2	Myosotis scorpioides L.	1.7	0.8		
Myosotis scorpioides L.			1.1	0.4	Plantago major L.	0.5	0.4		
Phleum pratense L.	1.8	0.6	7.2	1.4	Potentilla gracilis Dougl. ex Hook.			1.9	1.5
Pinus contorta Dougl. ex Loud.	1.2	0.9	2.9	2.1	Polemonium occidentale Greene	4.1	2.0		
Pinus ponderosa P.& C. Lawson			0.3	0.2	Poa pratensis L.	22.6	4.3	24.3	3.6
Plantago major L.			0.5	0.3	Rorippa nasturtium-aquaticum (L.) Hayek	1.3	0.6		
Potentilla gracilis Dougl. ex Hook.	1.4	0.5	2.3	1.6	Salix boothii Dorn	2.3	1.1	0.1	0.1
Poa pratensis L.	6.2	2.2	48.5	3.7	Salix geyeriana Anderss.	0.2	0.2		
Ribes lacustre (Pers.) Poir.	0.6	0.6	0.1	0.1	Scirpus microcarpus J.& K. Presl	1.3	1.1	0.1	0.1

Rosa woodsii Lindl.			1.2	0.6	Smilacina stellata (L.) Desf.			0.9	0.6
Salix geyeriana Anderss.	3.0	1.7			Trifolium longipes Nutt.			0.6	0.6
Scirpus microcarpus J.& K. Presl	2.8	1.9	0.6	0.6	Veronica americana Schwein. ex Benth.	2.6	0.8	2.0	0.8
Senecio pseudaureus Rydb.	1.0	0.5	2.4	0.9	Viola adunca Sm.	0.7	0.4	0.2	0.2
Smilacina stellata (L.) Desf.	0.5	0.3							
Stipa occidentalis Thurb. ex S. Wats.			1.5	0.9					
Symphoricarpos albus (L.) Blake	0.4	0.4	2.1	1.0					
Taraxacum officinale G.H. Weber ex Wiggers			0.1	0.1					
Thalictrum occidentale Gray	0.3	0.2	0.5	0.3					
Trifolium repens L.			1.5	0.7					
Urtica dioica L.	0.3	0.3							
Veronica americana Schwein. ex Benth.	0.1	0.1							
Veronica anagallis-aquatica L.	1.7	0.7	0.7	0.4					
Veratrum californicum Dur.	0.7	0.4	1.2	0.4					

	Mid Fk John Day River			
	Exclosed		Grazed	
	Mean	SE	Mean	SE
Achillea millefolium L.	10.5	3.2	0.7	0.4
Agrostis stolonifera L.	2.8	1.1	0.8	0.6
Arnica chamissonis Less.	2.9	1.0	1.5	0.6
Aster foliaceus Lindl. ex DC.	0.3	0.2		
Carex lanuginosa Michx.	30.2	4.6	10.4	3.4
Carex nebrascensis Dewey	7.9	2.6		
Carex utriculata Boott	4.8	2.1	0.1	0.1
Cerastium arvense L.			1.6	0.6
Cirsium callilepis (Greene) Jepson			5.5	1.5
Cornus sericea ssp. sericea L.			0.2	0.2
Dactylis glomerata L.	0.2	0.2		
Deschampsia cespitosa (L.) Beauv.	15.1	3.4		
Eleocharis palustris (L.) Roemer & J.A. Schultes	4.2	1.7		
Epilobium ciliatum Raf.	3.3	0.7	0.6	0.5
Equisetum arvense L.	7.6	2.5		
Equisetum laevigatum A. Braun	0.4	0.3		
Erigeron philadelphicus L.	1.2	0.4	0.7	0.3
Galium boreale L.	0.3	0.3	0.2	0.2
Geum macrophyllum Willd.	0.2	0.2		
Glyceria striata (Lam.) A.S. Hitchc.	1.7	0.5		
Hypericum anagalloides Cham. & Schlecht.	0.2	0.2		
Iris missouriensis Nutt.	0.5	0.4		
Juncus balticus Willd.	13.4	3.9	1.5	0.9
Lolium multiflorum Lam.	1.3	1.0	0.4	0.4
Mentha arvensis L.	3.2	0.8	0.1	0.1
Phleum pratense L.	0.4	0.2	1.8	1.4
Potentilla gracilis Dougl. ex Hook.	5.8	2.2	0.1	0.1
Poa pratensis L.	15.2	3.8	64.2	3.2
Ranunculus macounii Britt.	0.4	0.3		
Salix geyeriana Anderss.			0.1	0.1
Salix lucida Muhl.	0.1	0.1	0.3	0.3
Saxifraga rhomboidea Greene	0.4	0.4		
Sidalcea oregana (Nutt.	0.1	0.1		

ex Torr. & Gray) Gray				
Smilacina stellata (L.) Desf.	1.3	0.9	0.1	0.1
Solidago canadensis L.	15.4	4.4	11.3	2.6
Taraxacum officinale G.H. Weber ex Wiggers	0.3	0.2	3.4	0.6
Verbascum thapsus L.			0.9	0.4

Chapter 2 Geomorphic Response to Exclosures

Patricia F. McDowell
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Introduction

We used a paired reach approach, in which adjacent treated (fenced) and untreated (grazed) reaches were measured. The goal was to determine whether or not treated reaches have geomorphic characteristics that are more positive for fish habitat and water quality than control reaches. We focused on geomorphic characteristics such as channel width, depth and pool abundance and depth that are know to be important for fish habitat (e.g., Beschta and Platts, 1986; Bisson et al., 1988; Fausch and Northcote, 1992). We aimed to address two general research questions.

1. Does passive restoration (fencing) have positive effects on channel geomorphology? More specifically, do fenced reaches display better geomorphic characteristics than adjacent grazed reaches?

2. Does response to fencing vary among sites? If so, what kinds of streams are most positively affected by fencing?

The field procedure was a modification of standard procedures for measuring channel reach geomorphology (Fitzpatrick and others, 1998; Kaufmann and Robison, 1998; Lisle, 1987; Moore, Jones, and Dambacher, 1998). At each reach, we took multiple measurements of width and depth at regularly spaced intervals along the length of the reach (Table 2-1). In addition, we identified channel units (habitat units; pool, riffle, etc.) and measured dimensions of each unit in the reach.

Table 2-1. Geomorphic response variables measured at each reach

Symbol	Geomorphic variable	Measurements at a reach
Wbf	Bankfull width	At every 1-2 channel widths
Dbf	Bankfull depth	At every 1-2 channel widths
Wbf:Dbf	Bankfull width to depth ratio	Calculated
Ww	Wetted (summer flow) width	At < every 1 channel width
Dw	Wetted (summer flow) depth	At < every 1 channel width
Ww:Dw	Wetted (summer flow) depth	Calculated
Dmax	Maximum depth (wetted) of pools	At every pool
Dres	Residual pool depth	At every pool
% Pools	% channel area in pools	One value per reach

At each reach, we also measured or extracted data on several geomorphic control variables: channel slope, valley slope, gravel size, bank material, and channel-constraining geomorphic surfaces. A sample of 100 gravels was measured at three riffle cross sections in

each reach to determine reach-average gravel size. The geomorphic surfaces at the channel banks (bar, floodplain, terrace, alluvial fan, colluvial footslope) were identified and measured along the length of each reach. Bank vegetation characteristics were recorded by the vegetation team. In addition to the field-measured variables described above, reach location was mapped and channel sinuosity was measured on digital orthophotographs (U.S. Geological Survey, 2002) for each site.

We hypothesized that fenced reaches would be narrower and deeper than grazed reaches, and that they would have more pool area and deeper pools than grazed reaches. Narrower and deeper channel tend to provide better physical habitat for fish, such as deeper pools and cooler stream temperatures. We hypothesized that the level of response to fencing would be influenced by factors such as age of fencing, vegetation cover on streambanks, stream competence, and channel unconstraint.

Evaluating the Effects of Fencing on Channel Geomorphology

Approach

Initially we wished to test for a difference between the treated (fenced) and control (grazed) reach at each site, for each geomorphic variable (except %Pools which has only one measurement per reach). We found, however, that the data were not suitable for a site-level approach to statistical testing of differences between the two reaches. The data violate the assumptions of independence, normality, and homogeneity of variance that underlie most statistical tests. Because measurements were made at regular intervals along the channel length, most of the data series are not independent but are autocorrelated at one, two or more lags. The data are typically not normally distributed, although most or all data series could be transformed to a normal distribution with a log or square root transformation. Finally, at most sites the variances of the data series from the two reaches were not approximately equal. In the majority of sites, the control reach sample had larger variance than the treated reach sample, but at some sites the treated reach had larger variance. Parametric tests such as analysis of variance or analysis of covariance are based on these assumptions, as are non-parametric tests such as the Mann-Whitney U test and the Kruskal-Wallis one-way analysis of variance by ranks (Underwood, 1997; Sheskin 1997). There are strategies that may be used to overcome these violations of assumptions, but we found that different strategies would be required to correct the specific data problems for each site, so interpretability of the results across sites would be difficult. We therefore decided to use two alternative approaches to testing for effects of treatment. First, we did a statistical test for each geomorphic response variable combining all sites. Second, we used data visualization techniques to examine differences between paired reaches at a site.

Statistical Testing

For all of the response variables except %Pools (Table 2-1), we used regression analysis to determine whether there was a response to treatment when all eleven sites were included in the analysis. Using regression analysis rather than analysis of variance allowed us to correct for autocorrelation by including a continuous independent variable consisting of lag-1 values of the response variable, to correct for autocorrelation. Independent variables were treatment (1 = fenced reach, 0 = grazed reach), dummy variables representing the sites, and a lag-1 variable of the response variable, to account for autocorrelation in the response variable. Each site except BearSi was represented by a dummy variable set equal to 1 for measurements from that site and set to 0 for measurements from all other sites. BearSi was represented by the constant in the regression model. The response variable and the lag-1 variable were transformed to correct for a non-normal distribution in this variable. A log transformation was used for all variables except Dmax and Dres, for which a square root transformation fitted the data best. Plots of residuals from the regression models were examined to confirm that the assumptions of normality, independence and inhomogeneity were not violated. The regression model for each response variable took the following form:
logWbf = f [treatment, site 1, site 2,…site n, log(Wbf-1)].
We developed a best model for reach dependent variable, by first including all twelve independent variables, and then eliminating any independent variables that were not significant at levels of $p<0.05$.

The results showed that there is a statistically significant difference between treated and control reaches for all variables except Dmax and Dres, and that treated (fenced) reaches are narrower and deeper than control (grazed) reaches. The best model for each response variable is listed in Table 2-2. The treatment variable (1=fenced, 0=grazed) was statistically significant ($P < 0.000$) for all response variables except Dmax and Dres. The sign of the coefficient for the treatment variable was in the expected direction in each case. The site variables were also significant in most but not all cases. This indicates that there are differences among sites in the value of the response variable, probably due to site characteristics such as drainage area, channel slope, bed material, bank material, etc.

Table 2-2. Statistical results for effects of treatment

Response variable	Adj. R-squared	P for model	P for treatment	Sign of coefficient for treatment and interpretation
Wbf	78.3	0.000	0.000	- (narrower in fenced reach)
Dbf	68.0	0.000	0.031	+ (deeper in fenced reach)
Wbf:Dbf	49.0	0.000	0.000	- (lower in fenced reach)
Ww	71.4	0.000	0.000	- (narrower in fenced reach)
Dw	50.7	0.000	0.003	+ (deeper in fenced reach)
Ww:Dw	46.3	0.000	0.000	- (lower in fenced reach)
Dmax	41.2	0.000	0.805*	
Dres	25.1	0.000	0.798*	

* not significant

Data Visualization Analysis

We used data visualization to examine the effects of treatment on channel geomorphology at the site level. Data visualization confirmed the overall conclusion that sites have responded to treatment by becoming narrower and deeper. Three data visualization techniques were used (Cleveland, 1993). Box-and-whiskers plots were used to examine width and depth variables, because these variables had many measurements (>100) per reach. Dot plots were used for Dmax and Dres, where there were only ten to twenty-five measurements per reach. A bivariate scatter plot was used for %pools, which has only one value per reach. We examined response variables in two groups: those that reflect channel narrowing (Wbf, Wbf:Dbf, Ww, and Ww:Dw), and those that reflect changes in bed morphology (%Pools, Dbf, Dw, Dmax and Dres). We interpreted box-and-whiskers plots in the following way. If the median for the site with smaller values was less than the 1st quartile value of the site with larger values, or the median of the site with larger values was larger than the 3rd quartile of the site with smaller values, we concluded that there was a clear difference between the two reaches. If the preceding criterion was not met but the median, quartile and standard span values for the treated reach were all displaced in the same direction displaced from those of the control reach, we concluded that there was a weak difference between the two reaches. Data visualization examples and summaries are presented here. Additional data visualizations are in the appendix.

Channel Narrowing

Channel narrowing has clearly occurred in the treated reaches compared to the control reaches (Table 2-3). Most sites are narrower and have lower width-to-depth ratios in the treated reaches.

Table 2-3. Summary of data visualization results for channel narrowing.

	Wbf	Wbf:Dbf	Ww	Ww:Dw
Hypotheses	⇩	⇩	⇩	⇩
BearSi	=	⇓	⇓	⇓
CamasP	⇧	=	=	=
CampUp	⇩	⇩	⇓	⇓
Chesni	⇩	⇩	⇩	⇩
Devils	⇩	⇩	⇩	⇓
MFPhip	⇩	⇓	=	=
Murder	⇩	⇩	⇓	⇓
Summit	⇩	⇩	⇩	⇩
SwmpLo	⇩	⇩	⇩	⇩
SwmpUp	⇩	⇩	⇩	⇩
Tex	=	=	⇑	⇑

Symbols: Up arrow indicates values for treated (fenced) reach are larger than values for control reach; down arrow indicates values for treated reach are smaller than for control reach. Large arrow indicates a clear difference, small arrow indicates a weak difference, and equal sign indicates no difference between reaches. Dark shading indicates sites that clearly support the hypothesis, and light shading indicates sites that weakly support the hypothesis.

We hypothesized that bankfull channel width (Wbf) would be smaller in the treated reach compared to the control reach at each site, because we expect that in the treated (fenced) reach, banks will stabilize, bank vegetation cover will increase, and sediment will be trapped, resulting in channel narrowing. Eight of the eleven sites (CampUp, Chesni, Devils, MFPhip, Murder, Summit, SwmpLo, SwmpUp) have clearly smaller Wbf values in the treated reach compared to the control reach (Fig. 2-1; Table 2-3). Two sites have similar values in the treated and control reach, and one site (CamasP) is wider in the treated reach than in the control reach. Most of the eight sites that fit the hypothesis show large differences between the treated reach and the control reach; the treated reach's 3rd quartile value is less than the control reach's 1st quartile value, and the treated reach's median value is 20% or more smaller than the control reach median.

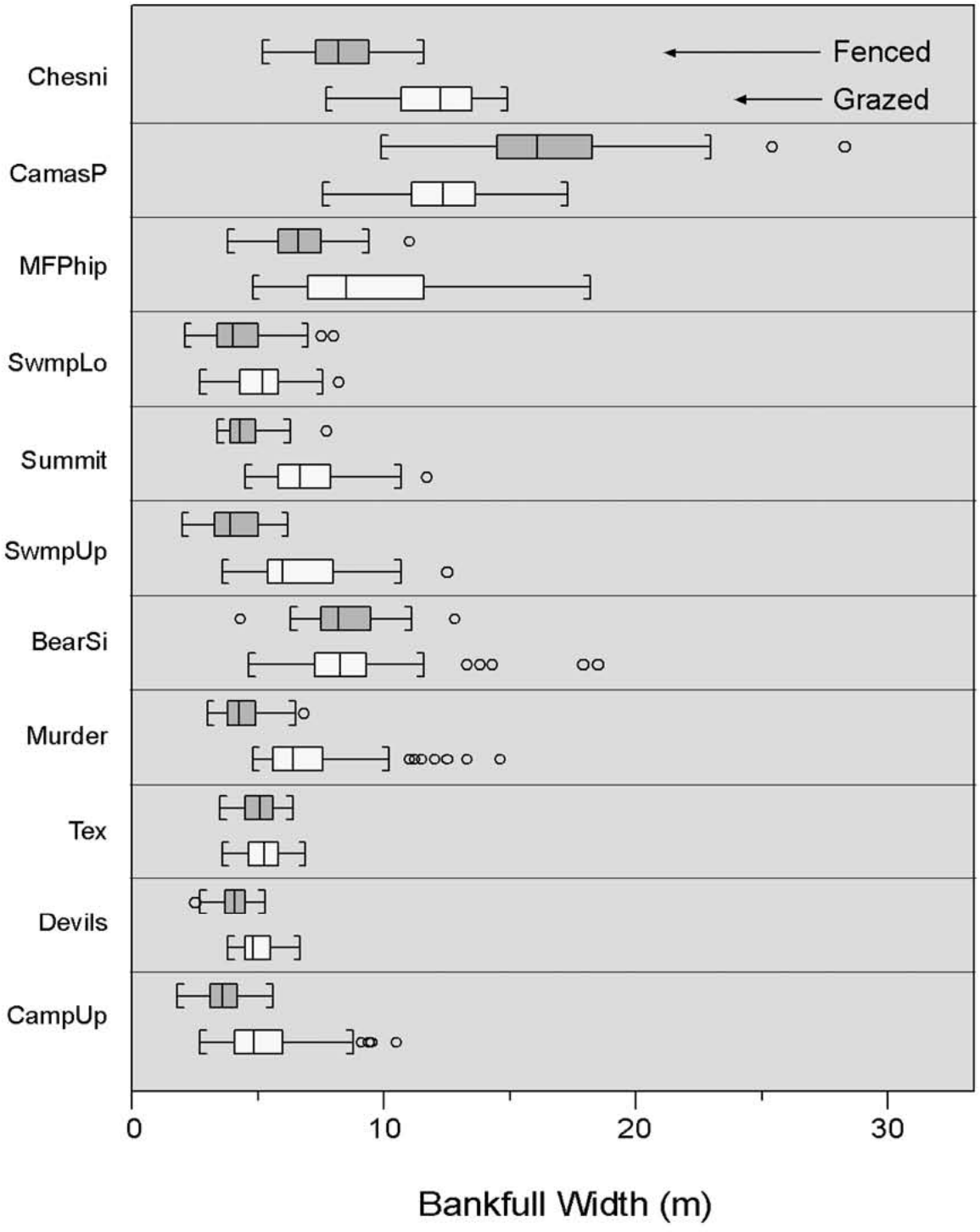


Fig. 2-1. Wbf in treated vs. control reaches.

We hypothesized that bankfull width-to-depth ratio (Wbf:Dbf) would be lower in the treated reach than in the control reach. A lower width-to-depth ratio indicated a narrower and deeper channel. Seven out of eleven sites (CampUp, Chesni, Devils, Murder, Summit, SwmpLo, SwmpUp) had clearly smaller Wbf:Dbf values in the treated reach than in the control reach (see appendix). In the remaining four sites there was no clear difference between the treated and control reaches, but two sites of these four (BearSi, MFPhip) showed somewhat smaller Wbf:Dbf values in the treated reach than the control reach. None of the sites had larger Wbf:Dbf values in the treated reach than in the control reach.

We hypothesized that wetted width (Ww) would be smaller in the treatment reach than in the control reach. This was true at five of eleven sites (see appendix); at the remaining six there was little difference between the treated and control reaches. No sites had larger Ww in the treated reach, except CamasP where the variance and the 3rd quartile values were larger in the treated reach. Of the six sites without a clear difference between the treated and control reach, values for the treated reach were slightly smaller in three sites (BearSi, CampUp, Murder).

We hypothesized that wetted width-to-depth ratio (Ww:Dw) would be smaller in the treated reach than in the control each. This hypothesis was clearly supported at three out of eleven sites (Chesni, Summit, SwmpLo; see appendix). At the remaining sites there was no clear difference between reaches, but at most of these Ww:Dw in the treated reach was somewhat smaller than in the treated reach.

Seven of eleven sites showed clear narrowing (Table 2-3). Three sites (BearSi, CamasP, and Tex) did not show a difference in width in the treated reach compared to the control reach. One site, MFPhip, showed narrower bankfull dimensions but not narrower wetted dimensions. For two reasons we interpret these results as indicating clear response at MFPhip. First, bankfull dimensions are more representative of channel change because they represent the entire channel form that is shaped by effective discharges. Second, while bankfull dimensions are independent of the discharge at the time of measurement, wetted dimensions are discharge dependent. Wetted dimensions may vary within a site, on a given day, due to losing and gaining subreaches. At MFPhip several springs discharge within the treated reach, which is downstream of the control reach. The treated reach therefore has significantly more discharge than the control reach, and this may account for the lack of difference in wetted dimensions.

Bed Morphology Adjustment

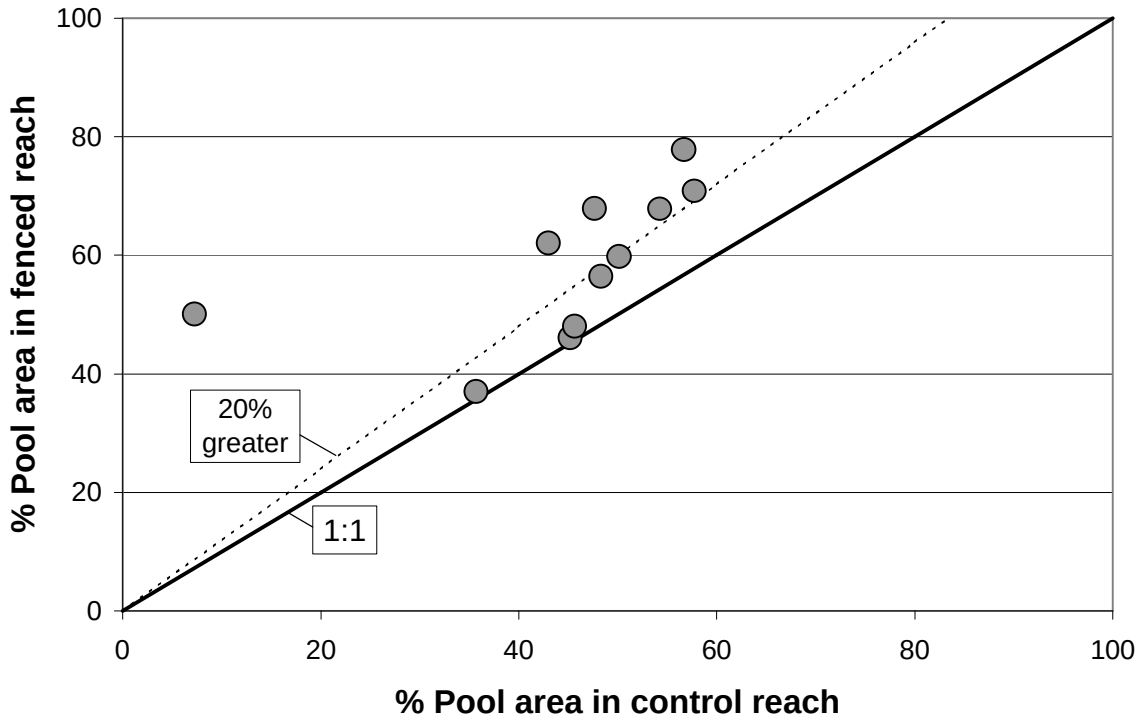
There is less strong evidence for changes in bed morphology in response to fencing than for channel narrowing (Table 2-4).

Table 2-4. Summary of data visualization results for bed morphology
Symbols are same as on Table 2-3.

	%Pool	Dmax	Dres	Dbf	Dw
Hypothesis	⇧	⇧	⇧	?	⇧
BearSi	⇧	↑	↑	↑	↑
CamasP	=	=	=	↑	=
CampUp	=	⇩	⇩	↑	↑
Chesni	⇧	=	=	↑	↑
Devils	↑	=	=	↑	=
MFPhip	⇧	=	⇩	⇩	↑
Murder	↑	⇩	⇩	=	=
Summit	⇧	=	=	=	↑
SwmpLo	⇧	⇩	⇩	↑	=
SwmpUp	⇧	=	=	↑	=
Tex	=	=	=	⇩	=

We hypothesized that percentage of the channel area in pools (%pool) would increase in treated (fenced) reaches. Six of eleven sites show a strong response in %pool, two sites show a weaker positive response, and the remaining three sites show no difference between the treated and control reaches (Fig. 2-2). We hypothesized that pool maximum depth (Dmax) and pool residual depth (Dres) would be larger in the treated (fenced) reaches than in the control reaches, but these hypotheses were not supported by the data. Most sites show no clear difference between the treated and control reach in either of these variables, and some sites have shallower pools in the treated reach. BearSi is the only site with a positive response on the pool depth variables.

Fig. 2-2. Pool area in treated vs. control reaches



We did not have a specific hypothesis for Dbf. In sites where bed material supply is high and bed incision is limited, channel narrowing often is accompanied by an increase in bankfull depth. In sites where incision has occurred or is possible, however, an increase in bankfull depth indicates channel incision, and this may result in loss of the hydrologic connection between channel and floodplain. Nine of the eleven sites showed no clear difference between the treated reach and the control reach (Table 2-4). At most of these nine, Dbf in the treated reach was somewhat higher than in the control reach. At CamasP, Dbf was clearly larger in the treated reach than in the control reach. At MFPhip, the treated reach has Dbf values that are clearly less than those of the control reach. The channel at MFPhip is incised, especially in the control reach. The difference in Dw is a positive sign, suggesting that the treated reach is recovering from incision or has stopped progressing in incision, compared to the control reach.

We hypothesized that Dw would be larger in the treated (fenced) reach than in the control reach. This hypothesis was not strongly supported by the data, although there is a tendency for larger Dw values in the treated reaches. Five of eleven sites show slightly larger Dw values in the treated reach, and the remainder showed approximately equal value sin the treated and control reaches. No sites had smaller Dw values in the treated reach.

We hypothesized that maximum pool depth (Dmax) and residual pool depth (Dres) would be greater in the fenced reach than in the control reach. The data showed no clear differences in these characteristics between paired reaches (see appendix).

Overall, six of eleven sites – BearSi, Chesni, MFPhip, Summit, SwmpLo, and SwmpUp -- showed some response in bed morphology, primarily in increased pool area in the treated (fenced) reach. BearSi showed the strongest response in bed morphology.

Controls of Geomorphic Response

Only three of eleven sites – BearSi, CamasP, and Tex -- did not respond as expected in channel narrowing. Five of eleven sites – CamasP, CampUp, Devils, Murder, and Tex – did not respond as expected in bed morphology adjustments. Why did these sites not show the expected response? A number of possible reasons, listed below, have been identified through theory in fluvial geomorphology and previous research (Church, 1996; Thorne, 1997; Magilligan and McDowell 1997; McDowell and Magilligan 1997; Montgomery and Buffington, 1998).

- Inadequate bank vegetation: Bank vegetation is not established or maintained, so vegetation does not trap sediment for channel narrowing.
- Treatment age: Fencing treatment may show little response in the first few years, since vegetation establishment and geomorphic adjustment may require several years and the occurrence of channel-forming discharge.
- Inadequate sediment supply: The amount of sediment delivered to the reach from upstream is inadequate to build bars (bed load) and banks (suspended load).
- Low competence: The streamflow does not generate adequate stream power and shear stress to ability to reshape bed and banks.
- Channel boundary conditions: A constrained channel, with resistant boundaries such as bedrock, hillsides, terraces or alluvial fans, may have limited ability to build its banks or adjust bed morphology.
- Site history: If the two reaches were significantly different in channel morphology before fencing, adjustment in response to fencing my not be detectable. To eliminate this factor, we selected reaches to be as similar as possible, but we recognize that there may have been initial differences unknown to us.

Because only eleven sites are included in our study and there are several possible controls, a formal statistical or mathematical analysis was not possible. We took an informal approach in analyzing the potential controls of response at our eleven sites. Table 2-5 and 2-6 summarize data on potential controls of response to treatment. It was not possible to obtain direct data on some of the potential controls, particularly sediment supply. We used abundance of bars as an indicator of bedload supply, but there is no feasible way of estimating suspended sediment supply within the scope of this project.

Table 2-5. Reach characteristics potentially controlling geomorphic response to treatment

Site	Narrower in	% Graminoid Cover		% Willow Cover		% Bare ground		% Constraint[1]	
	fenced reach?	Fenced	Grazed	Fenced	Grazed	Fenced	Grazed	Fenced	Grazed
BearSi	no -- similar	47	61	12	12	11	5	0	5
CamasP	no -- similar	67	63	3	5	9	11	0	0
CampUp	yes	94	92	74	35	0	6	1	0
Chesni	yes	28	25	43	57	11	17	19	6
Devils	yes	58	65	34	4	1	8	11	0
MFPhip	yes	96	81	1	2	15	11	0	0
Murder	yes	114	128	12	2	2	2	4	3
Summit	yes	83	29	26	6	4	16	3	2
SwmpLo	yes	96	75	31	75	0	3	0	26
SwmpUp	yes	78	42	32	28	0	33	0	0
Tex	no -- similar	42	32	129	74	0	6	8	15

1. % constraint is the percentage of channel length bordered by high terraces, alluvial fans, footslope, upland or road grade.

Table 2-6. Site characteristics potentially controlling geomorphic response to fencing

Site	Adjusted bed morphology?	Age of fencing in 2000	Valley gradient	Channel gradient	Unit stream power[1], N/m^2	Bed material[2] (D50, mm)	Competence index[3]	Bars as a % of length	Bank material cohesion[4]
BearSi	yes	2	0.0099	0.0064	61	28	2.18	27	moderate
CamasP	no	5	0.0072	0.0058	24	55	0.44	43	low
Chesni	yes	14	0.0177	0.0139	87	75	1.16	38	moderate
CampUp	no	36	0.0343	0.0259	151	59	2.56	5	low
Devils	no	10	0.0186	0.0105	58	54	1.08	5	high
MFPhip	yes	3	0.0043	0.0023	29	24	1.19	5	high
Murder	no	30	0.0052	0.0034	23	10	2.30	4	high
Summit	yes	22	0.0090	0.0075	87	54	1.62	26	moderate
SwmpLo	yes	13	0.0060	0.0046	28	31	0.90	0	high
SwmpUp	yes	13	0.0077	0.0056	30	28	1.09	0	high
Tex	no	23	0.0111	0.0092	47	33	1.42	3	low

1. Unit stream power was estimated for the 2-yr flow event using Wbf. Discharge of the 2-yr flow event was estimated from regional flood frequency curves in Harris and Hubbard (1983).
2. D50 values are the average of three or more gravel counts done in each reach.
3. Competence index is unit stream power divided by D50.
4. Bank material cohesion was assessed by observation in the field.

Channel Narrowing

Bank vegetation, treatment age and sediment supply are expected to be the most important controls of channel narrowing. From the vegetation data collected on site, we extracted three vegetation characteristics to consider as potential controls on channel narrowing: % graminoid cover, % willow and willow-like shrub cover, and % bare ground. These vegetation characteristics were defined in terms of their physical and hydraulic functions rather than their ecological functions. Graminoids (grasses, sedges and rushes) can be effective in protecting sediment surfaces from erosion because they produce many dense, highly flexible stems that lie flat under high flows, providing a protective covering to the bank or bar surface. Graminoids also typically have dense root mats near the soil surface that adds cohesion to reduce erosion and bank failure. Willows produce multiple flexible stems that may lie nearly flat under high flows, protecting the bank or bar surface from erosion. Willows are somewhat less flexible than graminoids, however, and willows may therefore have the additional effect of providing roughness and reducing velocity and shear stress near the bank and bar surface. This may lead to trapping of fine sediments. Most willow species also have dense fine root structures that increase bank cohesion. The root-sprouting Coyote willow (*Salix exigua*) is probably most effective in protecting surfaces and increasing cohesion. Other native willows that are crown-sprouting also protect against erosion and trap sediment to some extent. Two other riparian shrubs observed at the study sites, creek dogwood (*Cornus sericea*) and stinking currant (*Ribes hudsonianum*), also have multiple stems and can layer, producing extensive thickets of stems. The exotic English willow that has been planted at Chesni is tree-like and does not appear to provide hydraulic protection or sediment trapping. The common riparian shrub mountain alder (*Alnus incana*) does not have the flexibility or multiple stems of willows. We therefore included the following species in the variable % willow and willow-like shrub cover: *S. exigua, S. bebbiana, S. boothii, S. geyeriana, S. lasiolepus, S. lucida, C. sericea, R. hudsonianum*. Bare ground, the third vegetation variable, represents lack of protection against erosion. Data on these three variables are summarized in Table 2-5. For graminoids and willows, we used % cover of canopy because it was not feasible to collect data on stem density in the field. We assume that stem density is correlated with % cover.

For vegetation cover to be influential in channel narrowing in the fenced reach, we would expect % graminoid cover and/or % willow cover to be higher in the fenced reach than in the grazed reach, and % bare ground to be lower in the fenced reach than in the grazed reach. At BearSi and CamasP, two of the sites that failed to show a response in channel narrowing, we observe no vegetation effect from fencing that might lead to channel narrowing. At BearSi, graminoid cover is lower and bare ground is higher in the fenced reach than in the grazed reach (Table 2-5). Willow cover is very low in both reaches. At CamasP, vegetation characteristics are about equal in both reaches (Table 2-5). The lack of vegetation response at BearSi and CamasP is probably due to the young age of the exclosures (these are the two youngest exclosures). At Tex, the third site that failed to show a response in channel narrowing, the fenced reach is higher in both graminoid cover and willow-like shrub cover, and lower in bare ground, than the grazed reach (Table 2-5). Therefore, the vegetation effect is operating as expected in the fenced reach at Tex, but channel narrowing has not occurred. Treatment has been in place at Tex for 23 years, so there has been adequate time to see vegetation and geomorphic response. Willow-like shrub cover is higher in both the fenced and grazed reaches at Tex than at any other sites, although no *Salix* species are present at Tex, only *R. hudsonianum*

and *C. sericea*. It is not clear why Tex shows little channel narrowing in response to fencing, although it may be due to inherent geomorphic differences between the two reaches. The grazed reach at Tex is sinuous but constrained at places. In contrast, the fenced reach is nearly straight, and is located close to the left valley wall for most of its length. Although these differences do not appear in our measured values for constraint (Table 2-5), there may be geomorphic differences between the fenced and grazed reaches at Tex that have limited channel narrowing in the fenced reach.

Vegetation does not appear to be an important control at most of the eight sites that developed narrower channels in response to fencing. All eight sites have low values for willow-like shrubs, and only two (CampUp, Devils) show higher shrub cover in the fenced reach than in the grazed reach. Differences in graminoid cover between the paired reaches are mixed – higher at some site, but lower or equal at others. The most consistent difference lies in % bare ground, which is lower in the fenced reach at all sites. Age of exclosure does appear to be an important influence of response. Of the three sites younger than ten years, only MFPhip has narrowed in the fenced reach. MFPhip is a natural meadow site without woody vegetation, and significantly denser and taller sedge cover has developed along the channel margin in the fenced reach.

Adjustment of Bed Morphology

We hypothesized that bed morphologic adjustment should be controlled primarily by competence, sediment supply (particularly bed load supply, and constraint. We used the ratio between unit stream power and bed material size (D50, 50th percentile) as an indicator of competence. Reaches with relatively high unit stream power for the bed material size should be able to easily mobilize and re-shape their beds. Streams with relatively low unit stream power for the bed material size may be somewhat limited in competence, and therefore may not be able to adjust bed morphology. Of the five sites that did not adjust bed morphology, response at only one site, CamasP, appears to be associated with incompetence. CamasP has the lowest competence index of all eleven sites. The low competence at CamasP is related to low unit stream power rather than large bed material size. CamasP has low unit stream power because it has an extremely wide channel, the widest of all sites. The channel at CamasP may have been overwidened due to management or disturbance before fencing occurred. Since CamasP has been fenced very recently, there may not have been time for the channel to respond and reduce channel width. Murder and Tex have competence index values well within the range for sites that did adjust their bed morphology, and Devil's competence index is only slightly lower than average. CampUp has the highest competence index of all eleven sites. The lack of bed adjustment in CampUp is probably due to its high valley gradient (3.4%) and channel gradient (2.5%). These gradient values put CampUp outside the range of gradients for pool-riffle development; the natural channel type at CampUp is plane-bed.

Constraint is another factor that may limit bed morphologic adjustment. At Devils, the fenced reach is constrained by a high terrace. The remaining two sites that did not adjust bed morphology (Murder and Tex) do not appear to be influenced by constraint or low competence. At Murder, the channel bed is dominated by sand and fine gravels, and it has bed morphology that is intermediate between pool-riffle and dune-ripple (Montgomery and Buffington, 1998). We observed channel bed units that were complexes of small pools and small mobile dunes. Dune-ripple channels do not typically develop large, stable pools because of frequent movement

of dunes and ripples under discharge levels below bankfull. At Tex, we see no clear explanation for lack of adjustment in bed morphology, although low sinuosity in the fenced reach (discussed above) may have limited pool development. Although constraint appears to limit bed morphologic development, we note that the fenced reach at Chesni is relatively constrained (by hillslopes) but pool development has occurred. Supply of bedload sediment (indicated by bar abundance, Table 2-5) does not appear to influence bed morphologic development in this data set. Some sites with high sediment supply, such as CamasP, have not adjusted bed morphology, while many sites with low sediment supply have adjusted bed morphology.

In summary, the most important influences on development of bed morphology in the eleven sites we examined are competence, channel constraint by resistant landforms, and channel type. In particular, gravel-bedded pool-riffle channels are most able to develop pools, and steeper (plane-bed) or finer-grained (dune-ripple) channels are less able.

Summary and Conclusions

Considering all sites together, fencing results in clear, statistically significant improvements in channel geomorphology. The channels in the fenced reaches are narrower, deeper, and have more pool area than the channels in the grazed reaches.

At the level of individual sites, in most cases the fenced reach is clearly narrower, deeper and has more pool area than the grazed reach at the same site. The difference between fenced and grazed reaches is strongest for bankfull width, bankfull width-to-depth ratio, and pool area. Eight of eleven sites show evidence of a narrower channel in the fenced reach than in the grazed reach. Evidence for adjustments in bed morphology in the fenced reaches is less clear, apart from greater pool area in the fenced reaches.

Geomorphic response to fencing appears to be influenced by multiple factors, including age, vegetation cover, hydraulic conditions, and site geomorphology. Vegetation response to fencing (increased grass and shrub cover and reduced bare ground) is an important factor associated with geomorphic adjustments. Development of vegetation cover, and geomorphic response that is associated with vegetation cover, appears to take five years or more to be effective. Our data also show that some sites may respond geomorphically to fencing even if there is not a clear different in vegetation characteristics. Characteristics such as channel constraint, stream power, and sediment supply also influence the ability of a site to respond to fencing treatment. There are two implications of this finding. First, site selection for restoration projects should be based on an assessment of local ecological and geomorphic conditions to ensure success. Second, our understanding of where and when restoration will be successful is inadequate.

Understanding whether and why restoration works are important considerations. Current rates of investment in restoration are high. Habitat restoration is currently favored over other approaches for restoring anadromous fish listed under the Endangered Species Act. The results of this study show that there is reason to be optimistic about fencing as a treatment for restoration of aquatic and riparian ecosystems. Yet the results also show that geomorphic response to fencing is complex. Some sites respond, while at other sites response is weak or absent. Multiple factors, and perhaps interactions among factors, determine differences in

response. If response of river and riparian systems to a simple, holistic, passive restoration approach such as fencing is complex, response to less holistic, more active restoration strategies is undoubtedly even more complex.

Many key questions about how and where to do restoration projects remain unanswered. Despite our strenuous efforts in site selection, the ex post facto research design of this study limited the strength of the research results. Monitoring of restoration projects must be improved. Pre-construction monitoring and ten to twenty years of post-construction monitoring should be initiated in a large number of new restoration projects. Monitoring should focus on ecosystem and habitat changes.

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Appendix GA: Data visualizations for geomorphic analysis

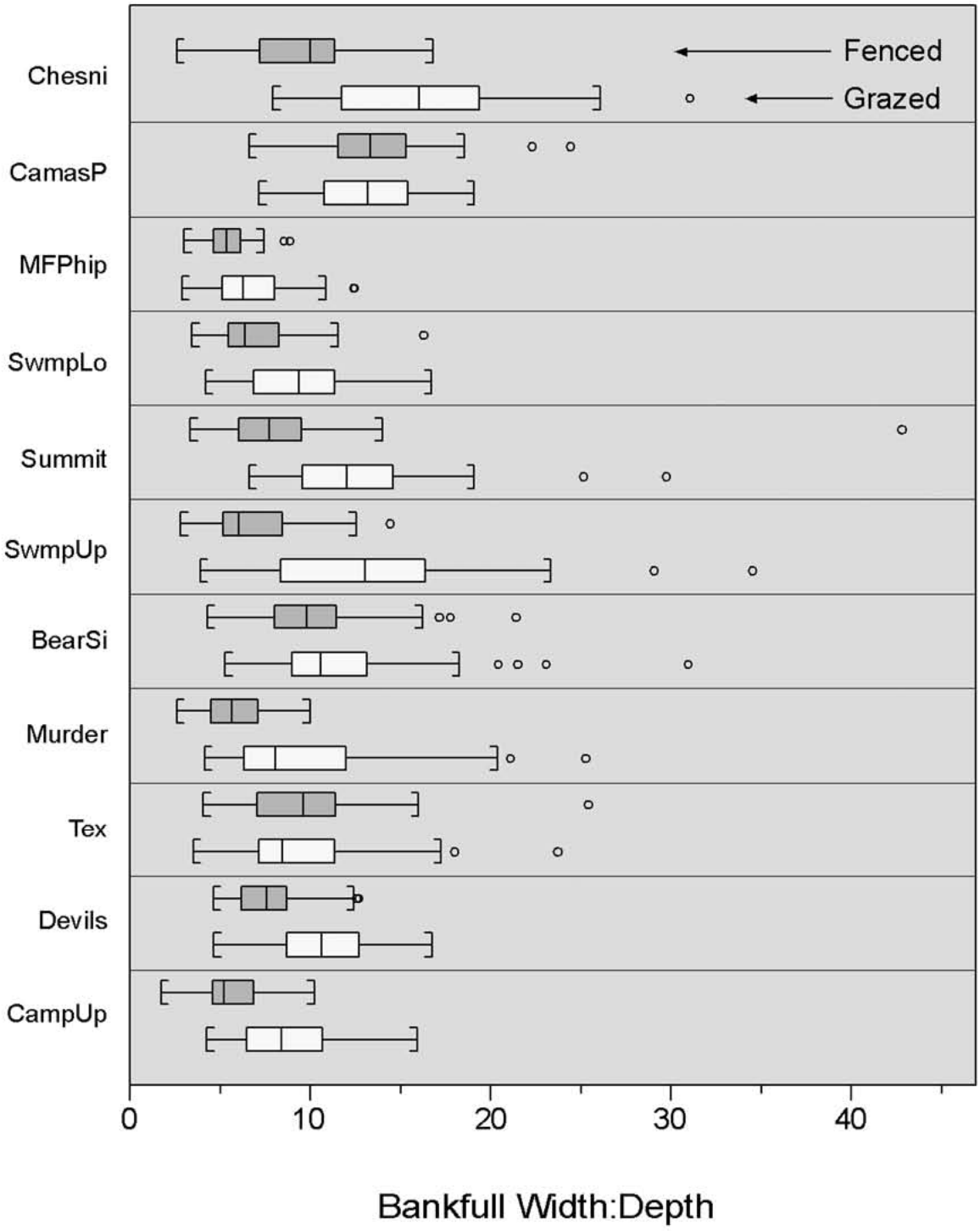

Fig. GA-1. Wbf:Dbf in treated vs. control reaches

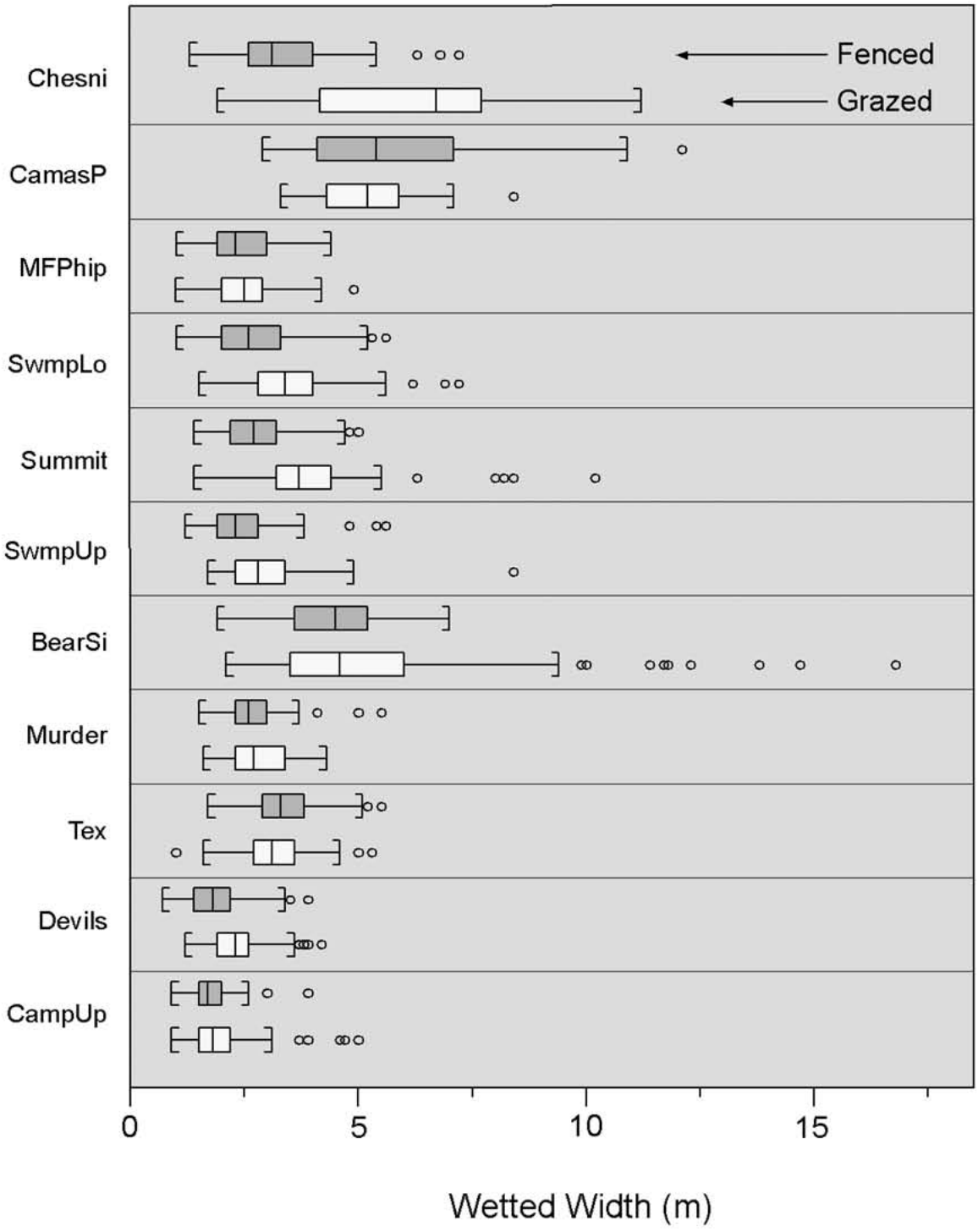

Fig. GA-2. Ww in treated vs. control reaches

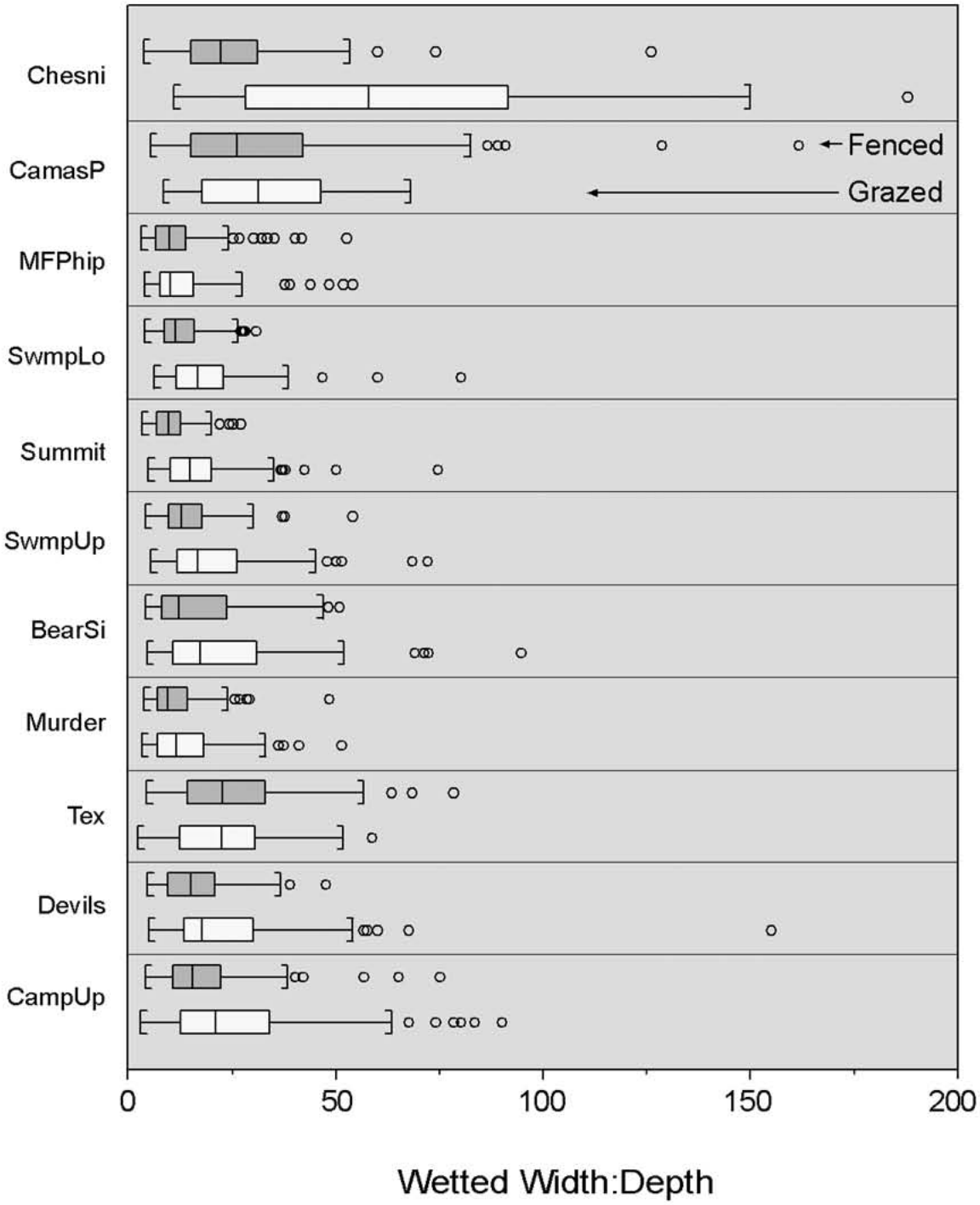

Fig. GA-3. Ww:Dw in treated vs. control reaches

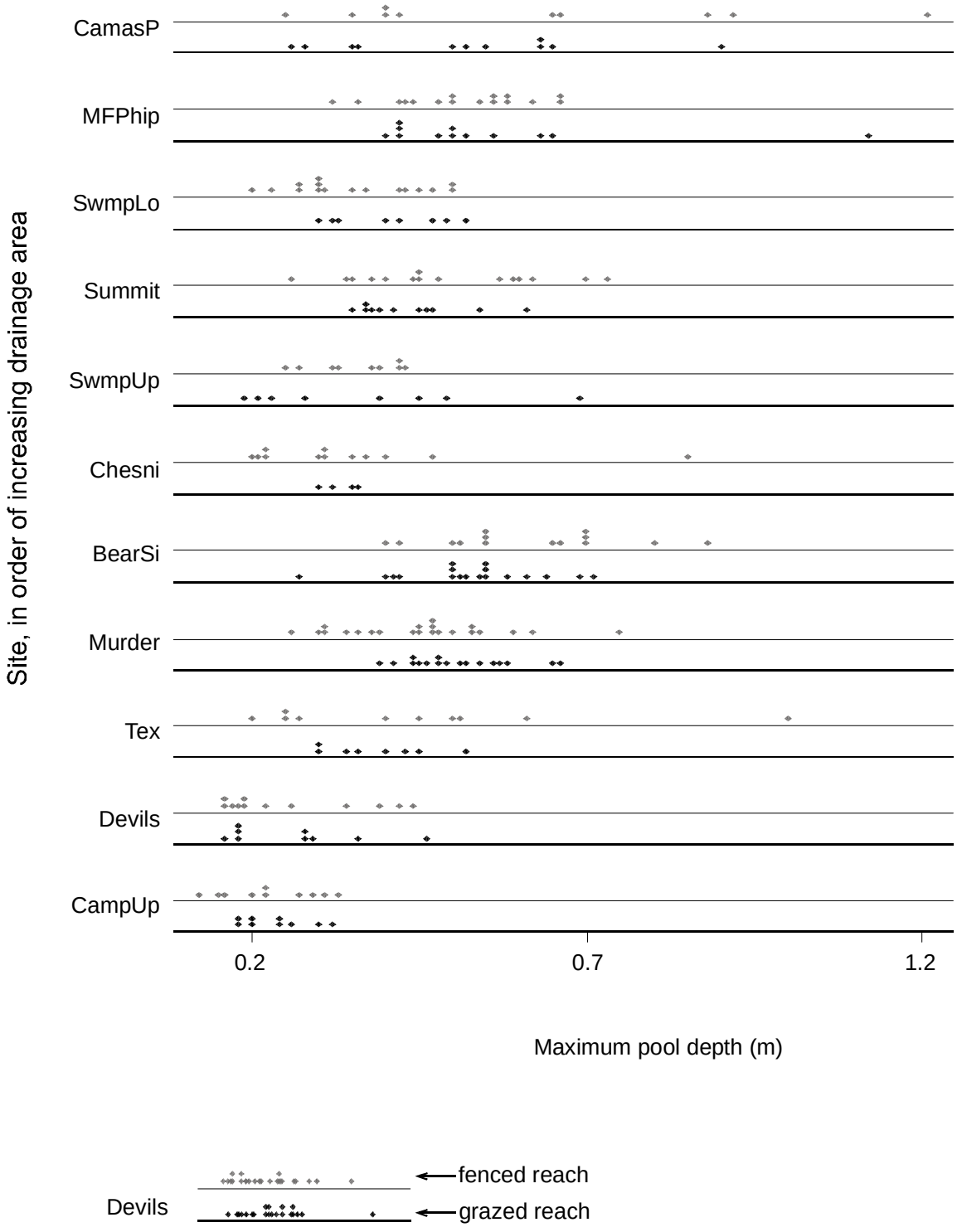

Figure GA-4. Maximum pool depth in treated vs. control reaches

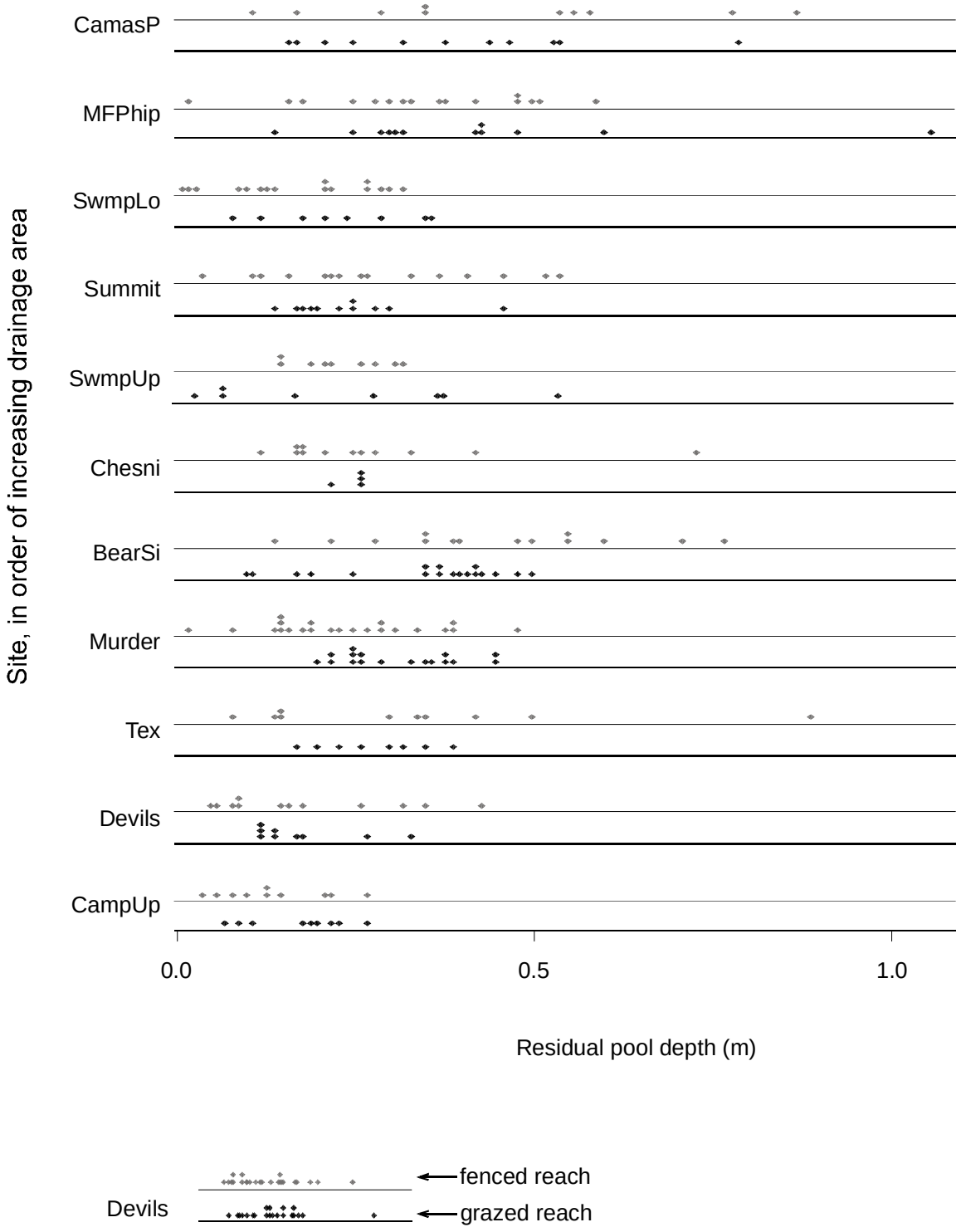

Fig. GA-5. Residual pool depth in treated vs. control reaches.

Chapter 3 ANALYSES OF FISH RESPONSES TO EXCLOSURES

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Introduction

During the past 40 years Federal and State agencies, with the cooperation of landowners, have constructed fenced exclosures along reaches of second to fourth-order streams in Northeast Oregon (Fig. 1). These exclosures protect riparian vegetation and banks from grazing and other effects by livestock. A major reason for the investment in exclosure construction and maintenance was to improve habitat for fishes, in particular salmonids, species now federally listed as threatened and endangered within the region. Most of these projects were *Ad Hoc*, not based upon scientific strategy or effectiveness, but on landowner participation. Therefore the location of the project within the watershed and the extent of the enclosure needed to effect the desired changes in fish habitat were of secondary importance. Other stream rehabilitation tools became popular in Oregon in the 1970's and 1980's, particularly the installation of log weirs or drop log structures. The hope was that the success reported in the upper Midwest would be repeated (White, 1996; Stream Enhancement Research Committee. 1980; Reeves and Roelofs 1982). Again, very little was done to determine scientifically what the critical limiting factors were.

The overall goal of this study was to determine the effectiveness of these tools and the *Ad Hoc* process in restoring salmonid habitats in streams. We took a comparative approach . Our study was designed to determine whether densities of fish were different in exclosed reaches compared to densities in reaches exposed to livestock. We also recorded long term stream temperatures within and without the grazing exclosures as shade from the riparian canopy reduces solar radiation to the stream. The locations and sizes of the exclosures were not designed to facilitate such an evaluation. Therefore, a pre-sampling survey determined what type of post-hoc experimental design could be constructed in order to test the hypothesis. We visited accessible exclosure sites during Spring 2000, and determined whether reaches upstream or downstream of each exclosure would serve as a control (i.e. grazed) reach to compare with the treatment (exclosure). Each control reach needed to be geomorphically and hydrologically comparable to that of the adjacent exclosure. We also determined whether the stream reaches were representative of typical salmonid streams in the area. We concentrated on geomorphically unconstrained reaches where the majority of exclosures have been constructed.

The following sections describe the design and model construction to test the hypothesis that there is no difference between observed numbers of fish in pools within vegetation exclosures and those in adjacent reaches open to livestock, from a set of paired summer samples in NE Oregon streams (Fig. 1). This is followed by the results of the analysis and discussion.

Sampling Design

Information from the pre-sampling survey comprised sets in which the control was downstream or upstream (Table 1, Fig. 2A). We used the following coding system to describe the restoration patterns we encountered:
(1) an AB set in which six streams (totalling 129 pools) were sampled in exclosure reaches (B) and in adjacent grazed reaches (controls) upstream (A), and
(2) a BC set in which three streams (totalling 121 pools) were sampled in exclosure reaches (B)

and in adjacent grazed reaches (controls) downstream (C).

The following streams fell into these categories.

(1) AB set; Camas, Phipps Meadow (Middle Fork John Day R.),
Tex, Murderers (at Lemon Gatherer reach), Camp, and Chesnimnus creeks.

(2) BC set: Bear, Summit, and Devils Run.

The "innovative" program limited us to one field season and we were obligated to select some research sites on the run. This meant that several sites were selected after mid-August, when the streams were cooler. Information gathered from these sites would be outside of the critical period when differences between fenced and unfenced sections of stream would be most extreme in terms of temperature and numbers of redband trout. The data would be out of context with the rest of the data set. Therefore the fish team sampled fewer streams and sites than the riparian and geomorphology teams because seasonal phenologies were not critical to the efforts of the latter teams, while seasonal changes in fish distribution were critical to the fish team.

Although the design could not be randomized, we attempted to select stream reaches that were typical of the region in terms of gradient, hydrology, bank stability, geomorphic setting, and vegetation when exposed to or protected from livestock. However, no stream reaches could be located that had extensive natural reaches unaffected by current or past human activity.

Stream Temperature:

The objective were to (1) record cumulative effects of shade and insolation through fenced and unfenced sections, and (2) remove any confounding effects of temperature when comparing exclosure and grazed reaches. For these objectives, we obtained (1) continuous records during the growing season from late-June to mid-September and (2) point temperature measurements recorded manually at the time and pool in which fishes were being counted. For continuous records water temperature loggers were placed at the top of the study section, at the border between the fenced and unfenced sections of stream and at the bottom of the study site (Fig. 2B). Point measurements involved taking the temperature using a calibrated thermocouple device or mercury thermometer at the outlet of the pool. When a cool spot was identified by the snorkeler, a minimum temperature was recorded at the location of the seep.

Fish Inventories:

Fenced and unfenced sections of each study stream were sampled simultaneously by two teams, consisting of a snorkeler and a data recorder. Each team went down to the bottom of each section and moved upstream. Every pool and riffles deeper than 0.25 m in depth was sampled. Shoreline and edge habitats were sampled from the bank by the data recorder and he helped the diver to search for young-of-the-year trout and larval fishes. Numbers and size/age estimates were made of all fishes. The interobserver variation was estimated during the same afternoon among 3 pools; the percentage errors (CV) estimated here were small compared to the mean percentage difference between treatment and control reaches when the exclosure effect was significant. Our snorkeler comparison also accounted for the differences in temporal variability within 12 pools on successive days with different snorkelers. As with the single afternoon experiment, no bias among divers was indicated (Appendix 1). The physical characteristics of the pool: wetted width, length, maximum stream depth, dominant substrate, percent of undercut banks present, pieces of large wood (at least 10 cm diameter by 1m length), percent canopy, air temperature exposed to the sun, air temperature within the riparian canopy and stream

temperature were measured. Characteristics of the dominant over-and understory were described.

Response variables and model selection:

The study was primarily interested in estimating inferences on population sizes and implied production of coldwater fish (salmon and trout species) associated with livestock presence or absence. We were also interested in similar effects of other fish groups distinguished by their different temperature tolerances and life histories. Our observational tool, standardized snorkeling, provides estimates of numbers of individuals in a visible range (estimated at each pool by the snorkeler as the lateral distance each side in which fish can be enumerated). The snorkeler estimated numbers of fish by species (Table 2) and size range in each pool greater than 20 cm (8 inches) maximum depth.

A generalized linear model, using the negative binomial distribution (Venables and Ripley 1999), was used to analyze count data. This model is appropriate because it accounts for variance exceeding that prescribed by the spatially random, Poisson distribution and handles variable proportions of zero observations more reliably than attempting to normalize the data through a transformation.

Given the length of each pool, the observed counts can then be transformed to a density measure based on observable area. In order to provide a comparable statistical unit, all counts were corrected to the median pool area basis, which was 30 m^2. We chose this area, rather than an arbitrary area such as 100 or 1000 m^2, because the variance of the negative binomial model employed used depends on mean counts.

The statistical model is described in the following section.

The question of which fish counts are analyzed is critical. We cannot look at a multitude of responses such as every species and life stage combination, because many will be correlated and narrowly defined units will have too many zero observations for a valid statistical analysis. However, fish populations behave very differently during their first year (young-of the year, YOY) than during subsequent juvenile and adult stages, occupying different microhabitats and attempting to grow quickly to avoid high mortality rates.

We observed territoriality among adult and juvenile fishes. For example adult chinook or redband trout would not tolerate smaller fish near them. Fish count data could underestimate favorable habitat conditions when defended by smaller numbers of larger fish. Because sizes of counted fish were estimated and abundant length-weight conversion data are available, we calculated observed biomass estimates for the juvenile-adult groups. This would theoretically compensate for smaller numbers of larger fish occupying more favorable habitats.

Proxies of biomass density can also be regarded as better reflecting biological production and recruitment to smolt or adult stages. Cumulative mortality, whose annual rates are higher for younger fish, results in average recruitment levels being more directly related to numbers of older age-classes compared to younger fish. Except for age-0 (YOY) fish, older age groups are too infrequently distributed among pools and often too difficult to age by sight to provide separate indices of potential recruitment. However, combining fish aged older than 1 year as biomass estimates is expected to provide a better proxy to mean recruitment potential than counts of mixed age classes, because the greater survival to recruitment of older (larger) fish justified their greater weighting per individual than younger fish. We analyzed counts (as observed # fish/30m^2) for:

(1) YOY coldwater fish (all salmonids except for the coolwater mountain whitefish, dominated by redband and juvenile steelhead),
(2) combined adult and juvenile coldwater fish,
(3) YOY warmwater fish (dominated by redside shiner, speckled dace and bridgelip sucker),
(4) combined adult and juvenile warmwater fish (dominated by speckled dace and redside shiner);
and total observed biomass (as estimated weight of fish (g)/30m^2) for:
(5) combined adult and juvenile coldwater fish, and
(6) combined adult and juvenile warmwater fish (dominated by bridgelip sucker, followed by redside shiner, northern pikeminnow, and speckled dace).

Insufficient numbers of coolwater fishes were present to warrant a separate analysis, but analyses of combined coolwater and warmwater fishes corresponding to the above categories were conducted.

Correlations of candidate responses provide a guide to what responses could be interpreted independently. Table 3 shows a selection of Pearson correlations of log-transformed (log [(fish count or biomass density) + 1]) responses for AB and BC data sets. Adult-juvenile numbers
and biomass of coldwater versus warmwater are strongly inversely correlated in AB and BC sets. The corresponding YOY counts are not significantly correlated. Neither are YOY numbers of coldwater fish significantly correlated with numbers of older coldwater fish.

The model and its explanatory variables:

The explanatory variable of interest in this project is the exclosure treatment versus its control (the dichotomous variable AB or BC). Other, nuisance variables were included that could also influence the fish response. The obvious nuisance variable was stream, because population levels among streams typically vary as a function of effects on a much larger scale than that of the exclosure. For example, Bear Creek has lower population levels because its lower reaches, where fish retreat during winter, are negatively impacted by severe icing events (Jeff Neal, ODFW, pers. comm.).

A more contentious nuisance variable is water temperature, which is not favorable for coldwater fish in the mid 20's (^{o}C) and above. It could be argued that mature riparian vegetation over long stream segments would shade a sufficient water area to keep temperatures at moderate levels. However, protected reaches were short, and we found a similar frequency of cold seeps in exclosure reaches as in grazed ones. Therefore, the potential for reducing unexplained variance by including water temperature on a pool-by-pool basis was justified.

Conversely, other habitat features were found to be a function of the effect of exclosure, such as water depth, overhanging banks, or vegetation cover (see other chapters) were not considered as valid nuisance variables in this analysis.

The full statistical model fit to the fish count data was:

(1) $$Y = \exp(b_0 + b_1 x_1 + b_{2(i)} x_{2(i)} + b_{12\,(i)} x_1 x_{2(i)} + b_3 x_3 + b_{13}\, x_1 x_3 + b_{23}\, x_{2(I)} x_3)$$

where Y = number of fish per 30 m^2 of pool area

b = fitted coefficients, with subscripts identifying associated x variable(s),

x_1 = 1 for exclosure, = 0 for control (open to grazing),

$x_{2(i)}$ = dummy variable for stream, i . (e.g., $x_{2(6)}$ = 1 for Tex Cr., with all other $x_{2(i)}$ =0)

$x_1x_{2(i)}$ = first order interaction terms between exclosure and stream, *i*,

x_3 = minimum water temperature during sample, °C,

x_1x_3 = first order interaction term between exclosure and temperature, x_3

$x_{2(i)}x_3$ = first order interaction terms between stream, *i*, and temperature, x_3

with the variance function corresponding to the negative binomial model:

(2) $var\ (Y) = m + m^2/q$

where m = mean of count, Y

m^2/q = variance additional to Poission (random) variance

q = fitted coefficient (theta)

In this study we are interested in whether the coefficient, b_1, that estimates the magnitude and sign of any difference between exclosure and control effects, is significantly different from zero. Therefore we are allowing for the possibility that exclosures may have more or less fish than the control. All other explanatory variables are incidental ('nuisance') to our hypothesis, but need to be included because interactions with them may confound our interpretation. For example, any interaction between exclosure/control (x_1) and a nuisance variable calls into question the interpretation of any significance indicated by b_1. If the model does not indicate significant interactions, those terms are removed and the reduced model is refitted. The process is repeated after any non-significant nuisance variables are removed. Nuisance variables remaining in the final model provide information on other effects on fish density, but do not affect inferences on, b_1, (exclosure vs. control) if there are no significant interactions. For example, Bear, Chesnimnus, Devil, and Murderers Creeks may have marked effects on salmonid densities, but may also indicate a significant difference in densities between exclosure and control reaches within streams.

The positively-skewed observed biomass density data (*biod* as g/30m^2) were log-transformed ($Y(biod) = \log_{10}$ [*biod*(g/30m^2) + 1]) and analyzed using the ordinary linear equivalent of model (1).

Results:

Stream Temperature Comparisons Between Exclosures and Unfenced Sections

The continuous stream temperature data represent, on average, the two hottest weeks of the year in this region, August 7-14 (Fig. 3), and 14-21, 2000 (Tables 4-9). . There were no significant differences in daily maximum and minimum stream temperatures between grazed and exclosed reaches among sites from the first data set (Fig. 3). The same may be said for the second data set, in which we tested for differences in the 7-day average and maxima and minima.

In general, there was little difference in stream temperatures between treatment (fenced) and control (unfenced, grazed segments) of each stream. The only exception was Phipps Meadow of the Middle Fork John Day River. This site was unusual in that large coldwater springs arise in the middle portion of the exclosure that was downstream of the grazed reach. Spring inputs varied between 16-17 ^{o}C. Average daily stream temperatures were mediated by the presence of the springs (Table 4). It lowered the maximum, but raised the minimum stream temperatures during this period. Most salmonids were concentrated near these springs, and near seeps in other streams that were not large enough to affect pool temperature at the downstream end.

Bear Creek of the Silvies Basin was the other stream where the fenced section was upstream of the section of stream exposed to livestock grazing. There were no significant differences in stream temperature among sections (Table 5). Of all the study sites, it was at the highest elevation, 2523m above sea level and stream temperatures reflected that circumstance. It was the coolest of all the sites.

The upstream fenced section of Summit Creek was formed by an alluvial fan from which many springs and seeps drained. For that reason, the entire upper section was uniform in temperature (Table 6). Unfortunately, the temperature logger was missing from the lower section. However, temperatures from point samples taken downstream indicated that there were lots of seeps and temperature patterns in enclosed and exposed reaches of streams were similar for the same length of stream segment (Fig. 8).

There were no significant differences in average stream temperatures recorded by the temperature loggers between sections of Camas Creek (Table 7). Two cool upwelling seeps were found in ponds, presumably formed from hyporheic flow downwelled from the riffles above. The ponds were formed by log weirs upstream and downstream controls. The upwelled water was 1 ^{o}C cooler at the surface than 1m upstream, but the upwelled was 3 ^{o}C cooler 5 cm above the bottom and 5 ^{o}C cooler just above the substrate. All redband trout were concentrated near the bottom proximal to the source of inflow.

Tex Creek became intermittent. Temperature loggers were only partially covered with water in both the upper and lower sections. This accounts for the soaring highs and the dramatic lows in temperature during the diel cycle (Table 8). The temperature logger at the fence line between the two sections probably represents more accurately the water temperatures. However, most of the stream is dotted with isolated pools.

Like Tex Creek, Murderers Creek commonly desiccates from the lower end of the stream upward (Tim Unterwegner, personal communication ODFW). No significant differences in temperature were observed (Table 9). It also appeared that there was more water in the unfenced than Fenced sections of stream.

Point temperature measurements

In general, point data patterns appeared similar to those from temperature loggers, except that the ranges from point estimates were larger (Fig. 3). This was due in part to two factors: temperature anomalies among sites (point estimate data from groundwater inputs were included), and the fact that stream temperatures increased during the sampling season (Fig. 3). Time of day influences water temperatures (Fig. 4). There are hints that fenced reaches were slightly cooler than reaches exposed to grazing at Camas, Chesnimnus, and Murderers Creeks. But overall, the differences were not significant. It is clear that the springs of Phipps Meadow moderated stream temperature downstream within the exclosure. Streams sampled in September were cooler because of incipient fall weather (e.g., Camp Creek and Devils Run). Again, there were no significant differences between fenced and unfenced reaches. Later in the year, the exclosed reach of Bear Creek registered a lower mean temperature while the reach exposed to grazing had broader ranges, but this was because, we extended the fish survey in the grazed reach for 3 km to the mouth of the Silvies River. In contrast, the later samples of Bear Creek were warmer. We suggest that the temperature loggers were more characteristic of the mean effects of the riparian canopy because it summarized an accumulated data series over several days; whereas, the point samples reflect short-term weather conditions.

Channel Morphology:

We found two characteristics that were distinctive (see geomorphology chapter). Thalweg depths of pools were deeper in reaches enclosed by fencing in 13 out of 16 paired comparisons (average of three deepest pools (Fig. 5A). Deep pools (>1m) are considered critical habitat elements for salmonids in summer (McIntosh et al. 2000). Likewise, the width/depth ratio, was greater in the reaches exposed to grazing in 12 of 15 paired comparisons (Fig. 5B). Quality of habitat for fishes, in particular salmonids, tends to decline as width to depth ratio increases (Brown 1969, Crittenden 1978, Beschta et al. 1987, Walling and Webb 1992**).**

Fish Density Comparisons Between Exclosures and Unfenced Sections

From the nine streams sampled (Table 1), sixteen fish species were identified (Table 2), plus small numbers of unidentified lamprey, minnow, and sculpin species.

Water temperature is an important candidate variable affecting salmonids (Myrick and Cech 2000, Wurtsbaugh and Davis 1977) as indicated by our pooled results for young-of-the-year (YOY) (Fig. 6A) and marginally for juveniles/adults (Fig. 6B). Because temperature increased during the day's observation period (Fig. 4), we attempted to sample the exclosure and control reaches simultaneously (Fig. 7A) to reduce the likelihood of statistically significant interactions with temperature confounding the interpretation of any exclosure effect.

Cold seeps were observed to attract salmonids within pools, so when snorkelers detected these, the minimum temperature over the seep was measured and used in the analyses instead of the usual measurement near the pool outlet. The role of seeps were especially evident in Camas and Summit creeks and the Middle Fork of the John Day River at Phipps Meadow. Fish congregate around seeps at higher densities than elsewhere in the reach. To determine the importance of seeps, we spooked fish from this microhabitat. Unlike, trout from cool streams, fish from warm streams return immediately to site of the seep following a disturbance, despite the presence of the diver.

There was no significant difference between the frequency of seeps in exclosure and control reaches, and in some cases significant seeps occurred in the same stream (Fig. 7B). However, one major seep, large enough to be described as a spring, was present in the upstream end of the exclosure at Phipps Meadow (Fig. 7C).

Temperature ranges for all streams from point samples are summarized in Fig. 8. In an attempt to increase the power of the statistical analysis, additional fish counts were later made in three streams. Because of the limited reach lengths available in exclosures or controls, these additional samples were subject to bias due to reduced control of temperature differences (see hatched rectangles in Fig. 8).

Analyses of YOY (age 0) salmonid density

When all samples (including additional samples) were analyzed, the AB set indicated a strong effect of exclosure (Table 10), but the more limited set of BC streams did not (Table 11). In the latter case, reanalyses excluding either temperature or treatment variables produced significant ($P<0.05$) effects of treatment and temperature, respectively. When AB was combined with BC, significant interactions prevented general conclusions being made.

These weak results when type BC was included can be ascribed to the unbalanced design due to the addition of extra samples that influenced two of the three type BC streams, Summit and Bear Creeks, and one of the AB streams, Phipps Meadow (see hatched rectangles in Fig. 8). Therefore, to restore the design balance and remove the potential bias of temperature mentioned

above, only the first-day samples (open rectangles in Fig. 8) were analyzed, comprising 171 pools from all nine streams. No interactions were encountered, and a highly significant effect of treatment, indicating higher densities in exclosures across the complete set of streams, was found (Table 12). The magnitude of differences in densities between fenced and grazed reaches and among streams is illustrated in Fig. 9. Because consecutive pools were sampled, it could be argued that the samples were not independent because of local movement of fish, thereby exaggerating the significance levels estimated. However, no serial correlation was indicated when the data set was ordered by proximity of samples (Fig. 10A). Also, residuals from the analysis were well-behaved (Fig. 10B).

Another issue raised earlier was the unusually large cool-water spring that was associated with the exclosure at Phipps Meadow. Although a significant interaction with temperature was not detected across all streams, it could be argued that the presence of this strong association was at least as likely to influence salmonid densities as the exclosure effect at Phipps Meadow. Therefore, the first-day data set was reanalyzed but with Phipps Meadow excluded. The results (Table 13) were still highly significant with respect the exclosures harboring larger densities of YOY salmonids.

The change in magnitude of the fish response to the effect of fencing off an exclosure was an average increase of 150% over the control reach density (Table 13). Mean densities vary by stream, as can be expected (Table 13) but the lack of interactions between stream and treatment (and with temperature) support the conclusion that the percentage change was consistent across streams.

Analyses with other response variables:

Similar analyses were carried out with numbers and biomass of juvenile plus adult salmonids, and with equivalents from the coolwater and warmwater fish observations. No indication of significant effects of exclosure on numbers or biomass of juvenile plus adult salmonids was encountered. This does not prove that there is no difference, but merely states that we were unable to disprove the null hypothesis that they were the same (see Discussion).

YOY densities of warmwater fish, dominated by speckled dace, indicated a highly significant effect of exclosure ($t = 4.8$) with the BC streams, but with opposite sign to the YOY salmonids, indicating that the warmwater fish prefer the unprotected control reaches. This can be regarded as an independent result because there was a poor correlation between YOY salmonids and warmwater fish (Table 3). The corresponding analysis of AB streams was invalidated statistically because of interactions between stream and treatment. Warmwater juveniles and adults combined, also dominated by speckled dace, indicated a highly significant effect of exclosure ($t = 4.2$), again with higher numbers in grazed reaches.

Coolwater fishes were not sufficiently widespread to permit valid statistical analyses because of excessive zero samples, but a combined coolwater and warmwater group produced similar results as the warmwater group.

Discussion

The salmonid YOY results show strong statistical indications of a preference for reaches where livestock entry has been prevented. Although YOY fish are harder to count than larger fish, we could not determine any reason why counts should have different biases in exclosed and control reaches. When YOY salmonids are close to the stream edges, and further from the visual scope of the snorkeler, they can be, and were easily noticed and estimated out by the snorkeler's

assistant. However, fish along the margins are less easy to spot in exclosures because of the development of more overhanging bank vegetation. If this bias existed, discounting it would produce stronger differences than those reported. No bias among snorkelers was indicated for YOY salmonids or other groups (Appendix 1).

The failure to detect a similarly positive effect of exclosures on juvenile and adult salmonids invites several interpretations. First, the greater mobility of fish over one year old gives them the capacity to travel the length of an exclosure in a short time. For example, a 20-cm fish traveling at a typical cruising speed of 1 Body-length s^{-1} can traverse a 200m reach in almost 17 minutes (see reach lengths in Table 1). Fish may be foraging for food outside the exclosures as well as within, with the knowledge that a refuge in terms of cover is available close by. Also, cool-water seeps sometimes exist outside of exclosures, providing a thermal refuge. Home ranges of redband trout in basins of the Steens mountains can range from 30 m to over 2 km (Li et al. 2002), and 300 m is the average home range of cutthroat trout in the Rocky Mountains (Young 1996, Hildebrand and Kirschner 2000). A second explanation is that there may be a bottleneck for survival within the reach because of lack of habitat. Therefore, older fish are not recruited into the local population and either die at greater rates or emigrate from the system. This type of density-dependent response is described by the Ideal Free Distribution (Fretwell 1972, Fretwell and Lucas 1970, Giannico 2000, Giannico and Healy 1998).

Seth White of the University of Wyoming (personal communication) describes a parallel situation for cutthroat trout (*O. clarki*) in grazed stream reaches. There, cattle can loosen impacted gravels locally, thereby making local improving local spawning habitat, but the process creates a death trap because poor habitat and water quality is created downstream such that young-of-the-year progeny often cannot reach good quality habitat. Another case of habitat separation is encountered in Tex Creek and Murderers Creek, where trout become isolated in pools in summer because the streams normally desiccate and become intermittent due to low discharge (Tim Unterwegner, District Biologist-John Day, ODFW, personal communication). There, we encountered elevated densities of YOY salmonids in early summer when stream temperatures were relatively cool.

With respect to the effect of greater quantities of YOY salmonids on the population as a whole, it is difficult to argue that adult, or even smolt population sizes will be directly related to YOY numbers because of their high and variable mortality rates.

In conclusion, the experimental design that we attempted to construct from the existing exclosures is weak, because the lengths of the exclosures are very small compared to the total stream habitat available and compared to the home ranges of the fish species of interest. Cumulative effects upstream were conducted downstream whether or not the fencing pattern was AB (grazed section above exclosure) or BC (grazed section below exclosure) and the exclosures were too small to make a difference. Moreover, six out of the nine fenced exclosures were sited below reaches of stream that were disturbed by grazing, sometimes the extent of the grazed section extended for a km or more. Therefore, the benefits of fencing were mitigated against. This artifact explains why temperature between grazed and fenced sections of streams were not significantly different among fenced and exposed reaches of stream. We suggest that a more efficient restoration can be accomplished a sub-basin by sub-basin strategic approach taking into account influences of scale and location as suggested by Fausch et al. (2002) and to follow the monitoring advice suggested by Bayley (2002) and the recommendations of the Independent Science Panel of the Salmon Recovery office of the state of Washington (2000, 2002).

Table 1. Streams used for analysis of fish in exclosures and controls.

Stream Name	Length of reach in exclosure (m)	Control Reach	Code
Camas Creek	436	upstream	AB
Camp Creek	123	upstream	AB
Chesnimnus Creek	227	upstream	AB
Murderers Creek	271	upstream	AB
Tex Creek	211	upstream	AB
Phipps Meadow (Middle Fork)	306	upstream	AB
Devils Creek	123	downstream	BC
Summit Creek	301	downstream	BC
Bear Creek	339	downstream	BC

Table 2 Fish observed by temperature -Guild

Common Name	Scientific Name	Total No.
Coldwater (all salmonids)**:**		
Rainbow/Steelhead trout	*Oncorhynchus mykiss*	5730
Chinook salmon	*Oncorhynchus tschawytscha*	195
Brook trout	*Salvelinus fontinalis*	13
Westslope cutthroat trout	*Oncorhynchus clarki*	4
Bull trout	*Salvelinus confluentus*	1
Coolwater:		
Mountain whitefish	*Prosopium williamsoni*	253
Shorthead sculpin	*Cottus confusus*	84
Torrent sculpin	*Cottus rhotheus*	81
Paiute sculpin	*Cottus beldingi*	27
Unid. sculpin	*Cottus spp*.	2
Mountain sucker	*Catostomus platyrhynchus*	15
Unidentified lamprey	*Petromyzontidae sp*.	11
Warmwater:		
Speckled dace	*Rhinichthys osculus*	42300
Redside shiner	*Richardsonius balteatus*	13400
Bridgelip sucker	*Catostomus columbianus*	6883
Northern pikeminnow	Ptychocheilus oregonensis	1150
Longnose dace	*Rhinichthys cataractae*	203
Largescale sucker	*Catostomus macrocheilus*	62
Unid. minnow	Cyprinidae spp.	6
Smallmouth bass	*Micropterus dolomieu*	0 (none observed)

Table 3. Pearson correlations among response variables (log (fish count or biomass density + 1))

A vs B data set (127 df):

YOY numbers of coldwater vs warmwater:,	r = 0.118	p = 0.18
Adult-juvenile biomass of coldwater vs warmwater:	r = -0.453	p < 0.0001***
Adult-juvenile numbers of coldwater vs warmwater:	r = -0.653	p < 0.0001***
YOY numbers vs adult-juvenile numbers of coldwater:	r = 0.171	p = 0.052

C vs B data set (119 df):

YOY numbers of coldwater vs warmwater:	r = -0.123	p = 0.18
Adult-juvenile biomass of coldwater vs warmwater:	r = -0.370	p < 0.0001***
Adult-juvenile numbers of coldwater vs warmwater:	r = -0.653	p = 0.0003 ***
YOY numbers vs adult-juvenile numbers of coldwater:	r = 0.110	p = 0.23

Table 4. Seven day average, maximum and minimum stream temperatures of Phipps Meadow (Middle Fork John Day River), August 14-21, 2000.

Site	7 day average ± s.d.	Max/Min Temperature
Above Exclosure	18.5 ± 4.9	27.1/8.7 °C
Middle Phipps	19.1 ± 3.5	25.6/11.4 °C
Phipps Springs	15.9 ± 2.9	21.2/12.3 °C
Lower Phipps	19.7 ± 5.0	29.7/11 °C

Table 5. Seven day average, maximum and minimum stream temperatures of Bear Creek (Silvies Drainage), August 14-21, 2000.

Site	7 day average ± s.d.	Max/Min Temperature
Upper Enclosure	15.1 ± 3.8	22.0/8.5 °C
At Fence Line	15.7 ± 3.9	22.5/9.0 °C
Lower Grazed	15.9 ± 3.8	22.6/9.3 °C

Table 6. Seven day average, maximum and minimum stream temperatures of Summit Creek (Malheur Drainage), August 14-21, 2000.

Site	7 day average ± s.d.	Max/Min Temperature
Upper Enclosure	16.9 ± 3.7	24.4/10.5 °C
At Fence Line	16.8 ± 3.7	23.8/10.4 °C
Lower Grazed	missing	°C

Table 7. Seven day average, maximum and minimum stream temperatures of Camas Creek (North Fork John Day Drainage), August 14-21, 2000.

Site	7 day average ± s.d.	Max/Min Temperature
Upper Grazed	17.9 ± 4.2	26.5/10.8 °C
At Fence Line	18.4 ± 5.0	27.2/10.0 °C

Lower Enclosure	17.9 ± 4.4	25.8/10.2 °C

Table 8. Seven day average, maximum and minimum stream temperatures of Tex Creek (Murderers Creek Drainage of the South Fork John Day River), August 14-21, 2000.

Site	7 day average ± s.d.	Max/Min Temperature
Upper Grazed	14.1 ± 7.1	38.1/2.6 °C
At Fence Line	15.3 ± 2.8	21.5/11.2 °C
Lower Enclosure	16.1 ± 7.8	38.1/4.8 °C

Table 9. Seven day average, maximum and minimum stream temperatures of Murderers Creek (at Lemon Gatherer), South Fork John Day River), August 14-21, 2000.

Site	7 day average ± s.d.	Max/Min Temperature
Upper Grazed	13.6 ± 2.4	18.1/9.0 °C
At Fence Line	14.4 ± 3.1	20.7/9.0 °C
Lower Enclosure	14.2 ± 3.3	20.6/8.7 °C

Table 10. Density of YOY salmonids from AB types (6 streams) using all samples.
Analysis**:** 129 pool observations with Fenced Exclosure (Treatment) vs Control
Response = observed # YOY coldwater fish per 30 m2 (84/129 = 65% zeroes)
Negative binomial model, with dummy variable for streams.

	Value	Std. Error	t-value
const(CAMAS)	5.42	2.72	
CAMP	0.21	0.81	0.26
CHESN	1.50	0.29	5.10 *****
MURD	0.48	0.15	5.50 *****
TEX	0.054	0.092	0.58
PHIPP	0.13	0.12	1.13
H2O.TEMP	0.26	0.14	1.80 (P=0.072)
Treat.Control	0.81	0.26	**3.13** ** (P=0.0023), B > A)

Null Deviance: 174 on 128 degrees of freedom
Residual Deviance: 89 on 121 degrees of freedom. Theta: 0.262 Std. Err.: 0.054

For all models (Tables 10 –13):
*'s indicate degree if significance, * <0.05, ** <0.01, ***<0.001 etc.
Results of Treatment or temperature do not depend on which stream is chosen as the 'constant'. A significant value for an individual stream indicates that it has a different density than the mean of the other streams excluding the stream represented by the constant.

Table 11. Density of YOY salmonids from BC types (3 streams) using all samples.
Analysis**:** 108 pool observations with Fenced Exclosure (Treatment) vs Control
Response = observed # YOY coldwater fish per 30 m2 (30/108 = 28% zeroes)
Negative binomial model, with dummy variable for streams.

	Value	Std. Error	t-value
Const(BEAR)	3.42	0.73	
DEVIL	0.96	0.29	3.28 ***
SUMM	0.25	0.12	2.16 *
H2O.TEMP	0.056	0.039	1.41
Treat.Control	0.14	0.14	1.04

Null Deviance: 149 on 107 degrees of freedom
Residual Deviance: 124 on 103 degrees of freedom. Theta: 0.667 Std. Err.: 0.115

Table 12. Density of YOY salmonids from combined AB and BC types (9 streams) from first-day samples only (Negative binomial model, with dummy variable for streams).
Analysis**:** 171 pool observations with Fenced Exclosure (Treatment) vs Control
Response = observed # YOY coldwater fish per 30 m2 (92/171 = 54% zeroes)

Deviance Residuals:	Min	1Q	Median	3Q	Max
	-2.27	-0.95	-0.52	0.16	2.10

	Value	Std. Error	t-value	
Const(BEAR)	3.13	1.41		
CAMAS	0.29	0.36	0.81	
CAMP	0.37	0.27	1.38	
CHESN	0.99	0.14	7.22	*****
DEVIL	0.62	0.14	4.38	*****
MURD	0.41	0.08	5.50	*****
TEX	0.095	0.048	2.00	*
PHIPP	0.13	0.07	1.71	
SUMM	0.047	0.047	0.99	
H2O.TEMP	0.115	0.076	1.52	
Treat.Control	0.98	0.15	**6.43**	***** p=3e-10

Null Deviance: 317 on 170 degrees of freedom
Residual Deviance: 144 on 160 degrees of freedom. Theta: 0.473 Std. Err.: 0.080

Table 13. Density of YOY salmonids from combined AB and BC types (8 streams - no Phipps) from first-day samples only (Negative binomial model, with dummy variable for streams).
Analysis**:** 150 pool observations with Fenced Exclosure (Treatment) vs Control
Response = observed # YOY coldwater fish per 30 m2 (79/150 = 53% zeroes)

Deviance Residuals:	Min	1Q	Median	3Q	Max
	2.23	-0.95	-0.58	0.25	2.31

					Effect size predictions by stream (% change = 150%)	
	Value	Std. Error	t-value		Control	Fenced
Const(BEAR)	1.14	0.16			3.1	7.8
CAMAS	0.42	0.30	1.40		2.1	5.1
CAMP	0.62	0.20	3.11	**	5.8	14.5
CHESN	0.84	0.12	6.88	*****	7.3	18.1
DEVIL	0.62	0.14	4.39	*****	5.8	14.5
MURD	0.44	0.07	5.92	*****	2.0	5.1
TEX	0.13	0.054	2.38	*	2.8	6.9
SUMM	0.10	0.054	1.90		2.8	7.0
Treat.Control	0.912	0.153	**5.96**	***** P = 4.8e-9		

Null Deviance: 278 on 149 degrees of freedom
Residual Deviance: 123 on 141 degrees of freedom. Theta: 0.46 Std. Err.: 0.08

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Appendix 1 A study of differences among snorkeler's observations.

This study was designed to detect any bias that may exist among observers and provide estimates of variance among surveys of the same pools at different times. It is not a calibration of the relationship of fish counts to actual fish densities. However, the project required comparisons between pairs of reaches and therefore needed comparable indices of population densities rather than actual fish densities. Because different snorkelers were involved with the treatment and control reaches (to control for temperature, e.g. Fig. 6), it was important that bias between observers could be discounted. We ran two field tests to compare counts among snorkelers: (1) on three adjacent pools surveyed by three snorkelers on same day, and (2) on twelve adjacent pools surveyed by two snorkelers on different days. The four snorkelers involved dominated the field observations in this study.

Test (1)

We chose a reach of the John Day Middle Fork River , just above the Middle Fork Campground, that contained three pools and a variety of fish species (Table A1). Each snorkeler carried out a standard survey, in turn, of each pool. Each pool was rested for 10 minutes between surveys. An independent reporter recorded data, and no communication was permitted among snorkelers. The pools did not contain young-of-year fishes. Pools 1,2, and 3 had maximum depths of 35, 65, and 45 cm, respectively, and had substrates with similar coverages of boulder, cobble, gravel, and sand. Snorkelers report visibility ranges reaching the shores in Pools 1 and 2, that had mean widths of 4.0 and 4.5 m, respectively, while a consensus of 2-m visibility on each side in the 3rd pool of mean width 4.5 m was reported. The observable area was calculated based on twice the lateral range of observation times the pool length (unless the stream margins could be observed), as with all the regular observations. Observed densities and composition differed significantly among pools, but differences in counts among snorkelers within pools were small and therefore biases that would imply one snorkeler was more accurate than another were not apparent (Table A1).

Fish counts, standardized on $30m^2$ of observable area, were analyzed in one-way ANOVAS that accounted for pool differences, in order to estimate the pooled, among-snorkeler error (residual). For ALL fish, the residual error (square root of mean square error) was 4.4, which corresponded to a coefficient of variation (CV) of 11% based on a mean count of 39.1. The corresponding mean error for salmonids (COLD) was 5.7 (CV = 37%) and for warm and coldwater fishes (WMCL) was 6.9 (CV = 29%). Time spent completing surveys was a significant covariate for ALL fish, but not for either of the two subgroups measured. In these tests, inter-pool variance explained 89-98% of total variance in counts.

The error we estimated mainly accounts for variation among snorkelers, while variation of observability due to differences in the spatial distribution of fish between surveys is expected to be a minor contributor because of the short time between surveys that were all completed within an afternoon. Additional variance due to time-related changes and YOY estimates, are were considered in Test (2).

Test (2)

This test ran replicate surveys on twelve consecutive pools on Sheep Creek (upstream of ODFW exclosure; Grande Ronde basin) on successive afternoons. The water remained low and

water temperatures ranged from 15.0-18.0 °C on 9/12/02 and 17.0-19.0 °C on 9/13/02. An independent reporter recorded data, and no communication was permitted among snorkelers. Fish counts were standardized on $30m^2$ of observable area for statistical analysis, and counts for common taxa are also reported (Table A2).

The paired t-test is a powerful statistical approach to test for bias between the surveys of each snorkeler, with paired samples on each of the twelve pools. No significant differences between surveys were observed for ALL fish (P=0.94, Means = 48.5, 47.6), coldwater YOY (P=0.62, Means = 3.49, 4.05), coldwater juveniles and adults (P=0.35, Means = 6.0, 10.3), warm and coolwater YOY (P=0.46, Means = 1.38, 0.76), and warm and coolwater juveniles and adults (P=0.66, Means = 37.6, 32.4).

Variances among surveys were higher than in Test (1), which is expected because the fish had 24h to redistribute themselves within and possibly among pools. In addition, the conditions for observing fish had more opportunity to vary. As in Test (1), the within-snorkeler, across-pool variance was estimated as the residual error from one-way ANOVAS that accounted for pool differences. For ALL fish, the residual error (square root of mean square error) was 27.2, which corresponded to a coefficient of variation (CV) of 57% based on a mean count of 48.0. The corresponding error for coldwater YOY was 2.57 (CV = 68%), for coldwater juveniles and adults was 10.8 (CV = 133%), for warm and coolwater YOY was 1.97 (CV = 184%), and for warm and coolwater juveniles and adults was 26.8 (CV = 76%). In these tests, inter-pool variance explained 53-77% of total variance in counts. The greater amount of unexplained variance than found in Test (1) was, as implied by the residual error and CV comparisons, due to greater variance among surveys in Test (2).

Conclusions

In both tests, no bias among snorkelers was detected, which validates a major assumption in the statistical tests (Tables 12 and 13). However, within-pool variance can be high, as shown by the residual errors (= square root of unexplained variance) and related coefficients of variation (CV) above. Because Test (2) variances were considerably higher than those in Test (1), much of the within pool variance can be ascribed to more spatial variation of fish in pools, and possibly movement between pools, due to the longer time lapse between surveys in Test (2). The high variance does not invalidate the analyses of exclosure versus control (Tables 12 and 13), but implies that a large part of the unexplained variance in those analyses (comprising about 45% of total variance for the YOY salmonid analyses) are probably attributable to within-pool variation in addition to variance of mean values among pools. With this sampling technique the onus on obtaining large numbers of samples is even stronger than with most other fish sampling techniques. Given a limited number of available paired stream sites, this provides a statistical argument for increasing the lengths of exclosures.

Table A1. Fish counts by three snorkelers from three pools in John Day Middle Fork River, 7/22/00.

				Total counts per 30m^2			Counts by species, by snorkeler, by pool								
Snorkeler	Pool #	Time (min)	Observ. Area-m^2	ALL	COLD	WMCL	CHI J	RBT A	RBT J	SPD A	SPD J	RSS A	RSS J	MNS A	TSC A
JL	1	3	32	18	9	8	10	0	0	0	9	0	0	0	0
JA	1	2	32	14	5	9	5	0	0	10	0	0	0	0	0
HL	1	3	32	17	7	10	5	0	2	0	10	0	0	0	1
JL	2	6	45	75	35	40	45	6	2	5	20	20	15	0	0
JA	2	5	45	75	17	57	15	5	6	25	25	15	20	1	0
HL	2	4	45	72	31	41	40	5	2	4	50	0	7	0	0
JL	3	6	80	21	12	9	25	4	4	3	20	0	1	0	0
JA	3	9	80	35	12	23	15	4	12	15	25	5	15	0	2
HL	3	6	80	25	9	15	20	2	3	3	30	0	8	0	0

ALL	All fish
COLD	All salmonids
WMCL	All warm and coolwater species
CHI	Chinook salmon
RBT	Rainbow/Steelhead trout
SPD	Speckled dace
RSS	Redside shiner
MNS	Mountain sucker
TSC	Torrent sculpin
J	Juveniles
A	Adults

Table A2. Independent fish counts by two snorkelers from the same 12 pools in Sheep Cr. (Grande Ronde) JH sampled on 9/12/00 (at 1234-1506h), JL sampled on 9/13/00 (at 1346-1513h).

				Total counts per 30m^2				Counts by selected species, by snorkeler, by pool							
Snorkeler	Pool #	Time (min)	Observ. Area-m^2	ALL	COLD JA	COLD YOY	WMCL JA	WMCL YOY	RBT JA	RBT YOY	RSS JA	NPM JA	BLS JA	SC A	MWF JA
JH	1	10	20	3	3	0	0	0	2	0	0	0	0	0	0
JL	1	8	20	58	5	0	54	0	3	0	30	5	0	0	0
JH	2	6	9	20	13	7	0	0	4	2	0	0	0	0	0
JL	2	7	9	103	23	13	67	0	7	4	20	0	0	0	0
JH	3	50	210	50	4	0	43	3	30	0	190	0	70	0	1
JL	3	23	315	36	2	1	32	0	25	15	240	10	70	2	18
JH	4	3	10	75	12	3	60	0	4	1	20	0	0	0	0
JL	4	2	10	69	63	3	3	0	21	1	1	0	0	0	0
JH	5	17	81	105	3	0	99	4	7	0	145	0	120	0	0
JL	5	11	81	90	6	0	84	0	15	0	70	30	120	0	8
JH	6	14	60	64	2	1	58	3	5	2	55	0	60	0	0
JL	6	6	60	56	2	0	54	0	3	0	40	8	55	1	4
JH	7	12	28	49	8	10	32	0	7	9	25	0	4	0	0
JL	7	5	28	45	10	7	28	0	9	6	10	10	6	0	0
JH	8	4	44	11	3	6	1	0	5	9	1	0	1	0	0
JL	8	4	44	1	0	1	0	0	0	2	0	0	0	0	0
JH	9	10	52	136	2	7	126	2	3	12	125	0	95	0	0
JL	9	11	52	55	3	5	48	0	5	8	30	15	30	4	5
JH	10	10	15	6	0	4	0	2	0	2	0	0	0	0	0
JL	10	4	15	20	2	12	6	0	1	6	0	1	0	0	2

JH	11	16	52	30	10	2	16	2	18	3	10	0	15	2	0
JL	11	6	52	11	3	5	3	0	6	8	5	0	0	0	1
JH	12		33	34	11	3	18	2	12	3	20	0	0	0	0
JL	12		33	25	5	2	9	9	6	2	10	0	0	0	0

ALL	All fish	JA	Juveniles and Adults
COLD	All salmonids (here RBT only)	YOY	Young-of-the-Year
WMCL	All warm and coolwater species		
RBT	Rainbow/Steelhead trout		
RSS	Redside shiner		
NPM	Northern pikeminnow		
BLS	Bridgelip sucker		
SC	Sculpin (torrent and Paiute)		
MWF	Mountain whitefish		