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Wild Ungulate Influences on the Recovery of Willows, Black Cottonwood and Thin-leaf Alder Following Cessation of Cattle Grazing in Northeastern Oregon

Abstract

Restoration of degraded riparian ecosystems is of great importance for the recovery of declining and endangered stocks of Columbia River salmonids as well as riparian-obligate wildlife species. Willows (*Salix spp.*), thin-leaf alder (*Alnus incana*), and black cottonwood (*Populus trichocarpa*) are important features of western riparian ecosystems having multiple functional roles that influence biological diversity, water quality/quantity, and aquatic/terrestrial food webs and habitats. Removal of domestic livestock and the construction of big game exclosures have been hypothesized to be effective restoration techniques for riparian ecosystem as well as for salmonid habitat recovery. Following more than a century of livestock grazing, cattle were removed from Meadow Creek in 1991 and the rates of riparian shrub recovery were measured for the two years following. Elk and deer-proof exclosures were constructed to quantify the browsing influences of native large ungulates. The initial mean height of 515 deciduous trees and shrubs (14 species) was 47 cm. After two years in the absence of livestock, significant increases in height, crown area, crown volume, stem diameter and biomass were measured both outside and inside of the exclosures. Mean crown volume of willows increased 550% inside of wild ungulate exclosures and 195% outside. Black cottonwood increased 773% inside and 808% outside, while thin-leaf alder increased 1046% inside and 198% outside. Initial shrub densities on gravel bars were low averaging 10.7 woody plants/100m^2. Shrub numbers significantly increased ≈50% (to 15.8 plants/100m^2 m or one new shrub for every 9 meters of transect length) outside of elk and deer proof exclosures through both clonal and seedling establishment. At the beginning of the study (1991), catkin production on willows was low (i.e., only 10% produced catkins). Wild herbivores had a significant influence on the reproductive output of willows; in 1993 catkins were produced by 34% of the tagged willows within exclosures but only 2% outside of exclosures. Wild herbivores were found to have significant influences on the rate of height growth of black cottonwood. For willows, wild herbivores had a significant influence on the rate of growth for the parameters of height, crown area, crown volume, and standing biomass. Nevertheless, due to the inherent resilience and adaptions to natural disturbance processes displayed by the riparian species, there was a rapid and positive response to cessation of those land use activities (i.e, cattle grazing) that caused habitat degradation and/or were preventing recovery.

Introduction

Globally, riparian/aquatic ecosystems are being altered, degraded, or lost at a greater rate than any other time in recorded history and far faster than they are being restored (National Research Council 1992). Degradation of riparian/stream ecosystems by anthropogenic activities along with dams, fishing, and hatchery practices are suggested to have the most pronounced influences on the decline of salmonids in the Pacific Northwest. (Nehlson et al. 1991). In the semiarid forests and rangelands of the Pacific Northwest riparian zones are areas of high biological diversity and productivity (Thomas et al. 1979, Skovlin 1984, Kauffman and Krueger 1984). Records of explorers and fur trappers in the Columbia Basin described the land as abounding with wildlife, and streams rich with salmonids and beaver (*Castor canadensis*) (Thwaites 1905, Rollins 1935, Ogden 1950, Naimen et al. 1988). Streambanks were often noted to be dominated by dense stands of willow (*Salix spp.*), aspen (*Populus tremuloides*), and black cottonwood (*Populus trichocarpa*) (Elmore 1992). By the early 1900's, many of these stands had been severely degraded or eliminated in part due to the influences of livestock grazing (Chaney et al. 1990, Elmore and Kauffman 1994, Fleischner 1994). Because cattle (*Bos taurus*) tend to congregate within riparian areas (Roath and Krueger 1982, Gillen et al. 1984), many of these zones have remained overgrazed (Armour et al. 1991, Fleischner 1994). The biological consequences of overgrazing resulting in riparian degradation are disproportionately high because of the diversity

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and large number of aquatic and terrestrial organisms that utilize these ecosystems (Carothers 1977, Thomas et al. 1979).

In the semiarid west, intact riparian plant communities are typically diverse in terms of structure, composition, and edge:area ratios (Kauffman and Krueger 1984). Riparian plant communities are integral to stream function and aquatic productivity (Murphy and Meehan 1991, Swanston 1991, Li et al. 1994). For example, in low-order streams, riparian vegetation strongly influences stream temperature (Li et al. 1994), channel form, habitats of fish and aquatic invertebrates (Sullivan et al. 1986, Sedell and Beschta 1991, Elmore 1992), and are the predominant sources of nutrients and carbon via allocthonous inputs (Cummins 1974). Vegetation also protects streambank soils through root strength, deflection, and dissipation of stream flow energy. Acting as a roughness element, vegetation enhances sediment, debris, and nutrient retention, and hence, channel and floodplain formation (Meehan et al. 1977, Elmore and Beschta 1987, Gregory et al. 1991, Elmore 1992). Riparian vegetation also functions to increase channel diversity and aquatic habitats through creation of overhanging banks and coarse wood debris inputs (Sedell and Beschta 1991, Maser and Sedell 1994).

When livestock or wild ungulates are present during the growing season, the palatable willow or cottonwood shoots are vulnerable to excessive herbivory (Kovalchik and Elmore 1992). In addition, reproduction can be greatly diminished or non-existent when shrubs are repeatedly overgrazed (Kay and Chadde 1992). In Yellowstone National Park, Kay (1994) found a complete absence of catkins on willows which were <2.5m in height and a mean of 670 catkins on plants above this height. Without recruitment or stand establishment, riparian hardwood communities progressively senesce and eventually are eliminated from the ecosystem (Kauffman 1988, Busse 1989, Kauffman et al. 1993). Livestock are a major factor contributing to the decline of riparian communities (Platts 1978, Bull and Skovlin 1982, Fleischner 1994).

In order to survive the high frequency and variable magnitudes of disturbance events, vegetation of riparian ecosystems display adaptations facilitating rapid recovery following such events. A passive restoration approach has been suggested as the logical first step in successful ecological restoration. Passive restoration is defined as the removal of human activities that cause degradation or prevent recovery (Kauffman et al. 1995, Kauffman et al. 1996). Shrub recovery can be dramatic following the elimination of livestock grazing (Platts and Rinne 1985, Elmore and Beschta 1987, Green and Kauffman 1995), but few studies have quantified rates of individual shrub growth, reproductive effort and changes in riparian shrub density when herbivory by cattle, mule deer (*Odocoileus hemionus*) and elk (*Cervus elaphus*) have been eliminated (Crouse and Kindshy 1984, Platts and Rinne 1985).

This project was initiated to improve our understanding of the resilience of riparian hardwoods, in particular the natural ecosystem recovery process following implementation of a passive restoration approach associated with the removal or diminution of herbivore grazing pressure. Our objectives were to quantify rates of shrub growth and re-establishment following the cessation of livestock grazing alone and in combination with the removal of all large animal herbivory. Elk and deer-proof exclosures were utilized to quantify recovery with and without native ungulate influences.

Methods

Study Areas

The study site is located on Meadow Creek, at the Starkey Experimental Forest, Wallowa-Whitman National Forest in the Blue Mountains of northeast Oregon. Meadow Creek is a third order tributary of the Grande Ronde River in the lower Snake River Basin. Elevation of the site is 1130-1400 m. Mean annual precipitation is approximately 60 cm, with two-thirds falling as snow during winter. Stream annual average discharge varies between 0.056 and 5.60 $m^3 s^{-1}$, with annual peaks occurring most often in March, but can occur anytime between January and June (McIntosh 1992). Uplands are dominated by ponderosa pine (*Pinus ponderosa*) and grand fir (*Abies grandis*) on north slopes and by ponderosa pine and bunchgrass communities on the south slopes. Over the 4 km study reach, average channel gradient is 10 m km^{-1} and average sinuosity is approximately 1.13 m m^{-1} (Ganskopp 1978). Gravel bars consisting of coarse to fine-textured alluvium provide many suitable sites for the potential establishment of willows, thin-leaf alder (*Alnus incana*) and black cottonwood (Busse 1989).

The Meadow creek watershed and riparian zone have experienced a long history of logging, splash dams, roads, railroads, and livestock grazing. At the onset of this study in 1991, Meadow Creek was characterized by a depauperate composition of streambank shrubs and a relatively shadeless, simplified, riffle-dominated channel (Beschta et al. 1991, McIntosh 1992).

Domestic livestock grazing along Meadow Creek began in the 1860's with stocking rates varying between 0.81 ha and 3.0 ha/animal unit month (AUM) until 1975 (Harris 1954, Skovlin 1991). From 1975 until this project began in 1991, utilization levels were maintained at approximately 70% of herbaceous growth with a grazing intensity of 3.2 ha/AUM (Skovlin et al. 1977). Prior to livestock removal, Bryant and Skovlin (1982) reported that shrub utilization was 60 - 65% in all treatments and suggested that elk and deer browsing accounted for about 25% of this impact.

Riparian Vegetation

Planned livestock grazing was terminated along Meadow Creek following the 1990 grazing season. However, some trespass cattle grazed the area each year. Vegetation was measured in 1991 (reflective of a century of livestock grazing), and in 1992 and 1993 (one and two years without livestock).

Annual rates of growth, changes in structure, and reproductive effort were measured on a total of 244 shrubs consisting of 8 species that were permanently tagged and mapped. Measurements included height, elliptical crown area, number of basal stems, and mainstem diameter at the soil surface. Elliptical crown area (A) was calculated utilizing the formula:

$$A = (\pi \times w_1 \times w_2) / 4$$

where w_1 is the maximum crown diameter and w_2 is the perpendicular diameter at its midpoint. Crown volume was calculated by multiplying the elliptical crown area by height. Herbivory by beaver, insects, and large herbivores (deer and elk) was also recorded for each individual. All measurements were taken in late June each year.

Density and establishment rates of riparian hardwoods were quantified through measurement of 52 permanent belt transects that were 2 x 25 m in size. These permanent transects generally paralleled the stream course on gravel bars that are typical of habitats occupied by members of the Salicaceae (Busse 1989). The species and height of all shrubs within the transects were recorded. All field measurements were made from late June to early July each year.

In the Fall 1991, three large game exclosures were constructed on the upper, middle, and lower end of the study site. This allowed for the quantification of shrub growth with and without the influence of large herbivores. One fourth of all tagged shrubs (n=62) occurred within the elk exclosures. Eight of the 52 transects occurred within the exclosures.

Standing biomass and rates of accumulation were calculated from predictive equations developed specifically for these riparian shrubs and trees (Case 1995). Differences in shrub crown area, crown volume and biomass (inside the exclosures vs. outside) were tested using a co-variate analysis of variance with log transformed variables. Crown volume was used as the co-variate; significance levels were set at $p<0.10$. Between treatment comparisons of height, mainstem diameter, and number of stems were determined using analysis of variance with untransformed variables. Between years analyses of individual species characteristics were conducted using paired *t*-tests. Between year changes in density (transect data) were also tested using the paired *t*-test.

Results

Initial Shrub Composition and Stature

In 1991, a depauperate riparian composition and structure reflected a century or more of heavy utilization by domestic livestock. The initial mean height of the 244 tagged individual shrubs (8 species) was 57 cm. The mean width of black cottonwood exceeded its mean height (44 cm and 35 cm, respectively). The mainstem diameter and the general structure of the plants indicated that the majority were not seedlings, but individuals maintained in a low stature by grazing for many years. Sandbar willow (*Salix exigua* var. *melanopsis*), Mackenzie willow (*S. rigida* var. *mackensieana*), and Booth willow (*S. boothii*) were the most abundant of the tagged willows (Table 1). The structure of willows (i.e., height, and crown volume) also reflected extreme utilization by grazing animals (Figures 1 and 2).

TABLE 1. Change (%) in height, crown area, crown volume, mainstem diameter, number of stems and biomass for thin-leaf alder, black cottonwood, and willows outside (Out) and inside (In) of elk and deer exclosures at Meadow Creek following the termination of livestock grazing (1991-1993).

Species	Sample size		Height		Crown Area		Crown volume		Mainstem Diameter		Number of Stems		Biomass	
	Out	In	Out	In	Out	In	Out	In	Out	In	Out	In	Out	In
Thin-leaf alder	27	4	46^{*}	129^{*}	112^{*}	405^{*}	198^{*}	1046^{*}	$46^{*\#}$	105^{*}	43^{*}	-5	152^{*}	422^{*}
Black cottonwood	23	17	$72^{*\#}$	151^{*}	268^{*}	248^{*}	808^{*}	773^{*}	59^{*}	59^{*}	35^{*}	23	568^{*}	316^{*}
Willows combined:	117	41	$18^{*\#}$	86^{*}	$132^{*\#}$	228^{*}	$195^{*\#}$	550^{*}	44^{*}	38^{*}	49^{*}	49^{*}	$235^{*\#}$	439^{*}
Coyote willow	8	16	$2^{\#}$	102^{*}	$30^{\#}$	303^{*}	$47^{\#}$	705^{*}	$21^{\#}$	42^{*}	57	75^{*}	$48^{\#}$	745^{*}
Sandbar willow	29	5	$3^{\#}$	59^{*}	87^{*}	63^{*}	97^{*}	122^{*}	53^{*}	62^{*}	65^{*}	100^{*}	84^{*}	131^{*}
Whiplash willow	18	7	33^{*}	56^{*}	194^{*}	133	260^{*}	263^{*}	77^{*}	54^{*}	68^{*}	37	358^{*}	216^{*}
Lemon's willow	5	—	2	—	138^{*}	—	153	—	30^{*}	—	71	—	105	—
Mackenzie willow	33	2	14^{*}	107^{*}	106^{*}	175^{*}	161^{*}	488^{*}	49^{*}	36	58^{*}	58	415^{*}	606^{*}
Booth willow	38	11	$37^{*\#}$	80^{*}	$159^{*\#}$	240^{*}	270^{*}	586^{*}	38^{*}	20^{*}	$15^{\#}$	18^{*}	$157^{*\#}$	239^{*}

A superscripted * indicates a significant difference between years (1991-1993), but within the treatment. A superscripted # over data in the Out column indicates a significant difference in total growth when comparing plants inside and outside the exclosures ($p<0.10$).

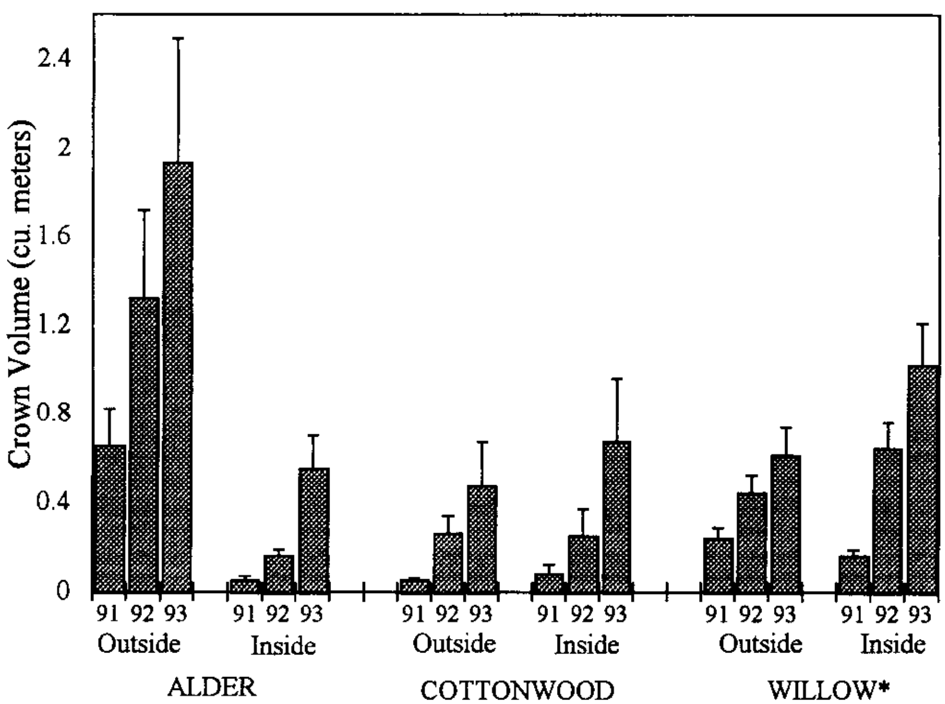

Figure 1. Mean crown volume outside and inside exclosures following livestock removal at Meadow Creek, Oregon (1991-1993). Thin bars represent one standard error. An asterisk (*) indicates a significant difference in growth outside (out) vs. inside (in) of elk and deer proof exclosures.

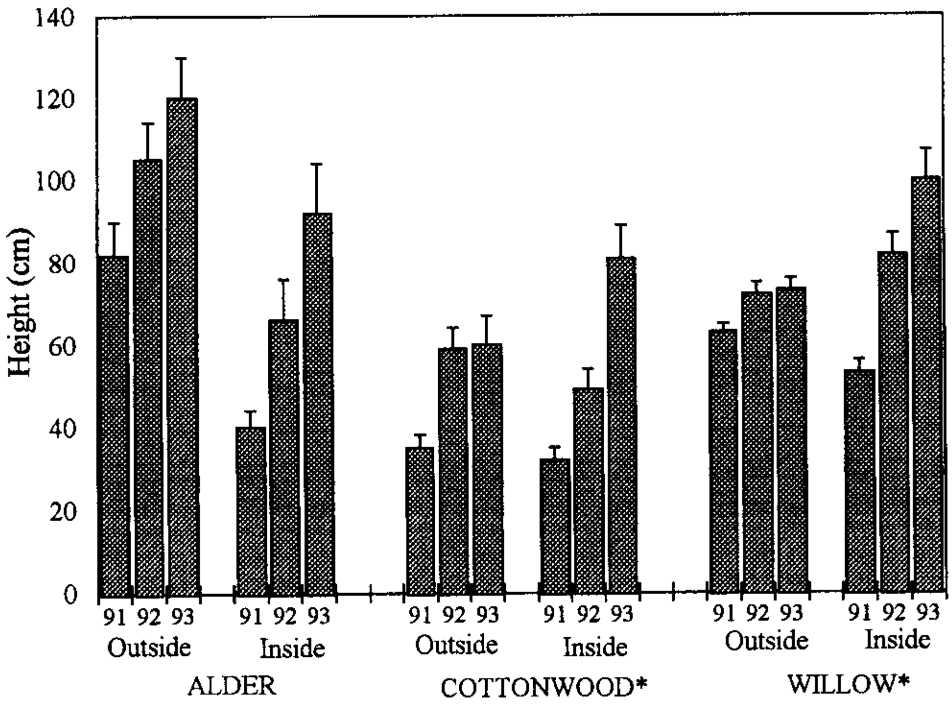

Figure 2. Mean height of thin-leaf alder, black cottonwood, and willow spp. outside and inside of exclosures following the cessation of livestock grazing at Meadow Creek, Oregon. Thin bars represent one standard error. An asterisk (*) indicates a significant difference ($p<0.10$) in height growth 1991-1993 outside vs inside of exclosures.

Initial shrub densities in 52 gravel bar transects was low with a mean of 10.7 plants/100m^2 (a range in density of 0-102 plants/100m^2). Only eight transects had densities >20 plants/100m^2. Density of all willows combined was 6.7 /100m^2, while density of black cottonwood and thin-leaf alder was 1.3 and 1.0 /100m^2, respectively. Density of all other species combined was 1.6 plants/100m^2) (Table 2).

Influences of Wild Herbivores on Riparian Shrubs

Initial shrub responses were dramatic following the removal of livestock but with the continued presence of wild ungulates. For thin-leaf alder, black cottonwood, and willow species combined, all of the measured shrub parameters (except the number of stems) significantly increased from 1991 to 1993. At the beginning of the study, mean biomass of thin-leaf alder, black cottonwood, and willow species outside of elk exclosures was 343 g, 38 g, and 84 g, respectively. Two years following the removal of livestock, increases in biomass ranged from 48% to 568% outside of exclosures and from 131% to 745% inside exclosures (Table 1). During this same time period, shrub crown volume increased 47-808% for all tagged species outside of exclosures (Table 1). Within the exclosures, the range in crown volume increase by species was 122-1046% (Table 1). Small shrubs displayed proportionately greater increases than larger shrubs, and for this reason a crown volume co-variate was used for between treatment analyses (inside vs outside of exclosures). Browsing by large ungulates was apparent on 72% of the tagged shrubs in 1992 and 73% in 1993.

There were significant differences in both the annual height growth and total height for willows and black cottonwood comparing inside and outside exclosures (Table 3 and Figure 2). In two years, black cottonwood within exclosures increased ≈50cm (151%) in height. Outside of exclosures cottonwood height increases were ≈25cm (72%). From 1991 to 1993, the mean height of willows increased from 63 to 73 cm outside exclosures and from 53 to 100cm inside of

TABLE 2. Shrub density on gravel bars at Meadow Creek, Oregon (1991- 1993). Numbers are mean and standard error (in parenthesis). Taxonomy follows that of Hitchcock and Cronquist (1973) and Brunsfeld and Johnson (1985).

	Density (#/100m²)		
Species	1991	1992	1993
Thin-leaf alder (*Alnus incana*)	0.96 (0.22)	1.19 (0.39)	1.58 (0.47)
Black cottonwood (*Populus trichocarpa*)	1.28 (0.67)	1.88 (0.78)	2 (0.94)
Coyote willow (*Salix exigua* var. *exigua*)	0.28 (0.17)	0.62 (0.33)	0.69 (0.41)
Sandbar willow (*Salix exigua* var. *melanopsis*)	5.00 (2.40)	6.23 (2.98)	6.96 (3.25)
Pacific willow (*Salix lasiandra var. caudata*)	0.16 (0.08)	0.38 (0.12)	0.38 (0.12)
Lemon's willow (*Salix lemmonii*)	0.04 (0.04)	0.04 (0.04)	0.04 (0.04)
MacKenzie willow (*Salix rigida var. mackenzieana*)	0.52 (0.17)	0.77 (0.23)	0.58 (0.18)
Booth willow (*Salix boothii*)	0.88(0.24)	0.81 (0.23)	0.88 (0.26)
Red-osier dogwood (*Cornus stolonifera*)	0.12 (0.09)	0.15 (0.17)	0.15 (0.09)
Black hawthorn (*Crataegus douglasii*)	0.12 (0.12)	0.15 (0.09)	0.12 (0.11)
Stinking currant (*Ribes hudsonianum*)	0.92 (0.46)	0.96 (0.15)	1.38 (0.51)
Common snowberry (*Symphoricarpos albus*)	0.36 (0.22)	0.58 (0.27)	0.88 (0.37)
Wax currant (*Ribes cerum*)	0.04 (0.04)	0.15 (0.32)	0.04 (0.04)
Pearhip rose (*Rosa woodsii*)	0.04 (0.04)	0.15 (0.15)	0.12 (0.08)
Service berry (*Amelanchier alnifolia*)	0	0.04 (0.11)	0.04 (0.04)
Red Raspberry (*Rubus idaeus*)	0	0.04 (0.04)	0
Totals	10.72	14.14	15.84

TABLE 3. Mean height growth (cm) during 2 seasons outside and inside of elk/deer exclosures, at Meadow Creek in northeast Oregon. Numbers in parenthesis are standard errors.

SPECIES	1991-1992		1992-1993	
	Outside	Inside	Outside	Inside
Thin-leaf alder	23 (3)	26 (9)	15 (4)	25 (6)
Black cottonwood	24 (4)	17 (4)	1 (5)	31 (5)*
Willow spp.	10 (2)	28 (3)*	1 (2)	18 (4)*
All spp. combined	13 (2)	25 (3)*	3 (2)	22 (3)*

*Indicates a significant difference inside vs. outside of exclosures ($p<0.10$)

exclosures (Figure 2). In addition, for willows we detected significant differences in the increases of crown area, crown volume and biomass between plants outside and within exclosures. The average biomass of all willows combined increased 439% (a range of 80- 439 g) inside the exclosures and 235% (83-178 g) outside of exclosures. Increases in crown volume of all willow species combined was 550% inside and 195% outside of exclosures. Crown volume of thin-leaf alder increased 198% outside of exclosures but 1046% inside the protected exclosures. The small sample size for thin-leaf alder inside the exclosures limited statistical interpretation. Black cottonwood crown volume growth was 808% outside and 773% inside exclosures during the 2 year period (Table 1).

Individually, coyote willow (*Salix exigua* var. *exigua*), and Booth willow were significantly influenced by wild ungulates for the variables of height, crown area and biomass (Table 1). For example, the increase in average biomass of Booth willow from 1991 to 1993 was 65 to 168 g/plant out of exclosures, and 91-308 g in the exclosures (an increase of 157% and 238%, respectively).

Rates of Establishment, Densities, and Seed Production

Two years following the removal of livestock, the total number of all shrubs encountered in all transects increased by 50% (i.e., from 271 in 1991 to 412 in 1993). This is equivalent to a one new shrub for every 9 meters of transect length. The greatest measured increase was for sandbar willow; 25% of the 144 new plants were of this species. Sandbar willow, which had the highest density of all species on the transects, increased from 5 to 7 plants/100m^2 (Table 2). Individuals from this species can readily establish from stems deposited on gravel bars during high flows. In addition, it was one of the few shrub species that commonly produced catkins (Table 4). The second most abundant species was black cottonwood which increased from 1.3 to 2.0 plants/100m^2, while the third most abundant, thin-leaf alder increased from 0.9 to 1.5 plants/100m^2. All other shrub species combined increased ≈0.5 plants/100m^2.

Most of the increases in density of individual species occurred within the same transects in which they were originally encountered; the increases in the frequency of species (i.e., the percent of transects in which individual species occurred) were slight following two years of grazing cessation. Sandbar willow occurred on more transects than any other (36% in 1991 and 38% in 1993). Frequency of thin-leaf alder was 24% in 1991 and increased to 29% by 1993. Black cottonwood occurred in 20% of the transects in 1991 and did not change during the 2 years. Frequency of whiplash willow (*Salix lasiandra var. caudata*) increased from 8% to 17%, and other willows showed no, or only slight increases or decreases in frequency of occurrence. Other common shrubs which increased in frequency included stinking currant (*Ribes husonianum*) (an increase from 20% to 28%), and common snowberry (*Symphoricarpos albus*), (an increase from 8% to 14%).

TABLE 4. Number of willow plants producing catkins at Meadow Creek, Oregon from 1991-1993.

	1991	1992	1993
Sandbar willow	5	8	4
Coyote willow	8	9	10
MacKenzie willow	1	2	2
Watson's willow	1	0	0
Whiplash willow	1	2	1
Total	16	20	17
% of all willows	9.2	11.4	9.8
Mean # catkins/plant	55 (16)	164 (48)	135 (32)

Limitations to shrub recovery may partially be explained by a lack of reproductive vigor; only 10% of willows produced catkins following livestock removal (Table 4). Sandbar willow and coyote willow were the most prolific making up 80% of the individuals that produced catkins. Of the individual plants that produced catkins, the mean number per plant increased three-fold from the first season (i.e., 55/plant in 1991 to 164 and 135/plant in 1992 and 1993). Most of the reproductive individuals in 1993 were inside of the exclosures; 34% of the willows inside exclosures produced catkins as compared to 2% outside of exclosures. From 1991 to 1993, the mean number of plants producing catkins outside of exclosures decreased from 6 to 3 while the number inside exclosures increased from 9 to 14.

Other ecosystem factors influencing shrub recovery included beaver which removed mainstems from 20% of willows, 11% of alder, and 4% of cottonwood during the study (Figure 3). Because they tend to remove the largest of stems, the magnitude of regrowth both within and outside of exclosures was likely diminished by their influences. High flows had discernable negative influences on the growth of 6% of the tagged shrubs in 1993.

Discussion

Interactions between biotic, hydrologic, and geomorphic features determine the rate of riparian recovery following natural or anthropogenic disturbances (Kauffman et al. 1996). On Meadow creek, these ecosystem features have been

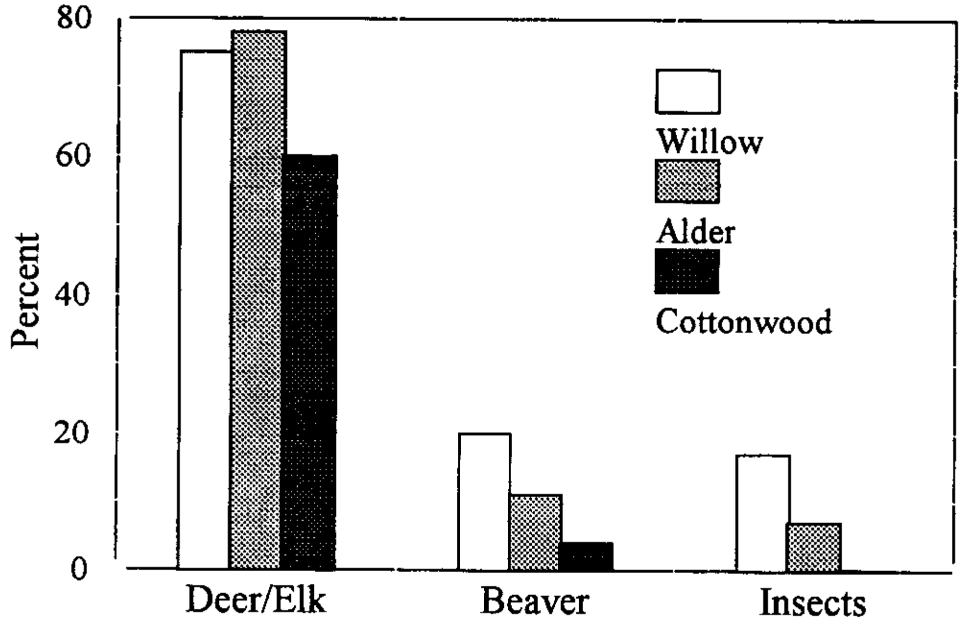

Figure 3. The percentage of individual shrubs browsed by deer and elk outside exclosures in 1993, and the percent browsed by beaver and insects outside during the 1991-1993 period at Meadow Creek.

dramatically influenced by the long period of heavy livestock use as well logging, roads, and other land use activities (McIntosh 1992). Results of these ecosystem perturbations are manifested in incised channels, simplified channel morphology, and changes in hydroperiod (Beschta et al. 1991, McIntosh 1992). A biotic disequilibrium was reflected in the depauperate riparian plant composition and structure (Tables 2, and Figures 1 and 2) as well as high densities of wild herbivores (B. Johnson, USFS La Grande, Oregon, personal communication). These physical and biotic features can combine to retard the restoration process. Nevertheless, significant increases in both the structure and density of the riparian hardwood component has occurred after only two years in the absence of livestock use.

While a 50% increase in shrub density may initially appear to be indicative of successful recovery following the cessation of livestock grazing, this increase must be viewed in terms of the very low initial densities. Density of black cottonwood, thin-leaf alder and willow species in 1993 was 2.0, 1.6, and 9.5/100m^2, respectively at Meadow creek. In contrast, density of these same species in a lightly grazed or cattle-excluded reach of Catherine creek (also an upper tributary of the Grande Ronde River, Northeast Oregon) was 35, 6, and 176 /100m^2, respectively (Green and Kauffman 1995). Irvine and West (1979) also reported similar high densities of Fremont cottonwood (*Populus fremontii*, ≈80/100m^2) and coyote willow (≈260/100m^2) in intact riparian ecosystems of Southeastern Utah.

Recovery of disturbance-dependent riparian shrub communities is a function of seed production or vegetative establishment from stem deposition in suitable substrates (Busse 1989, Kauffman et al. 1985). At the onset of the study, neither of these reproductive processes were occurring in measurable amounts at Meadow creek. The low total aboveground biomass of riparian shrubs as a result of livestock utilization would not have resulted in an abundant source of stem materials for vegetative establishment following flood deposition. In addition, only 10% of the willows produced catkins. Similarly, at Yellowstone National Park, Wyoming, catkin production only occurred on willows above the height of browsing animals (wild ungulates) (Kay 1994). On Meadow Creek, the increase in catkin production in exclosures and the increases in aboveground biomass following cessation of livestock grazing indicates a greater potential for increased rates of willow establishment in the future. For example, in areas with abundant seed and stem sources that were ungrazed by domestic livestock for 10 years, densities of willows and thin-leaf alder increased by 3 and 7-fold, respectively (Green and Kauffman 1995).

Over-utilization by domestic livestock resulting in the degradation of riparian and stream ecosystems is among the most significant threats to biological diversity in the western USA (Cooperider 1990, Kauffman and Krueger 1984, Fleischner 1994). A passive restoration approach which is simply the removal of those anthropogenic perturbations that are causing degradation or preventing recovery has been suggested to be the first necessary step in ecological restoration and probably the most important (Kauffman et al. 1995). The passive restoration approach in this study was the removal of high levels of herbivory by large ungulates which limited individual stature, reproduction, and establishment. The significant growth response of all hardwood species in the absence of cattle and/or wild ungulate herbivory is indicative of the success of this approach.

Similar responses of riparian vegetation to the removal of livestock has been recorded throughout the west (Crouse and Kindshy 1984, Platts and Rinne 1985, Busse 1989). In cattle exclosures on Catherine Creek, Oregon, height increases for willows and black cottonwood were 100 cm over a 10 year period, while thin-leaf alder increased 200 cm (Green and Kauffman 1995). In our study, the strongest height response was from black cottonwood protected from wild ungulates (i.e., a mean height growth of 31 cm in one season, 1992-1993) (Table 3). Some individual cottonwoods were observed to have grown >100 cm in a single season. In Utah, Behnke and Raleigh (1978) recorded height growth of willows at 50 cm after four years (12.5 cm/yr). In Montana, sandbar willow, a prolific sprouter, quickly re-established where it had not been totally removed by grazing (Hansen 1992). Also in Montana, shrub production was found to be 13 times greater inside of livestock exclosures (Marcuson 1977). Shrub canopy cover in north central Colorado was 8.5 times higher in protected areas (Schultz and Leininger 1990). In south central Washington after 10 years protection from livestock, woody plants had increased in stature and density to form a more or less continuous tree corridor along the entire length of the stream (Rickard and Cushing 1982).

Overuse of shrubs by wild ungulates can be severe when natural processes such as migration or predation are altered, or when management results in animal densities (domestic and native grazers) that are beyond the carrying capacity. It has been suggested that high densities of elk have reduced willow and cottonwood stands by 90% over the past decades in Yellowstone National Park (Kay 1994). In contrast, other studies reported that large wild ungulates had a minor influence on riparian shrub structure (Platts and Raleigh 1984). Rickard and Cushing (1982) and Green and Kauffman (1995) reported increased stature and density of woody plants when livestock were excluded but with the continued presence of wild herbivores.

Wild ungulates along Meadow Creek did influence early recovery after livestock were removed. Mean annual height growth of species outside of exclosures from 1991-1992 ranged from 10-24 cm (Table 3). During the second season (1992-1993) wild ungulate impacts were more severe, reducing height growth for willows and cottonwood to 1 cm. Height growth inside exclosures during this year was 18-31cm. However, crown volume and biomass are better indices to overall growth since they quantify the three-dimensional changes in shrub stature (Table 1 and Figure 1). In terms of total crown volume growth over two seasons, black cottonwood increased 773% inside elk exclosures and 808% outside, while willows increased 550% inside and 195% outside.

Our results may be a reflection of the high densities of deer and elk on the Starkey Experimental Forest ($\approx 5/km^2$ for deer and $6/km^2$ for elk). In addition, natural migrations have been altered through construction of the 77 km^2 big game enclosure surrounding the Experimental Forest. Because Meadow Creek is the only area not utilized by cattle within this enclosed area, wild ungulate influences on the riparian zone may be higher than would occur if cattle densities were lower throughout this forest.

We found that elk and deer use had greater influences on willows and black cottonwood than on thin-leaf alder. This preference for willows may be partly explained by its higher nutrient concentrations. Foliage of coyote and sandbar willow were found to contain higher concentrations of crude protein with a lower C:N ratio than black cottonwood or thin-leaf alder (Case 1995). Others have also reported that willows and cottonwood were preferred over alders (Kufeld 1972).

Conclusion

Many riparian zones in the west have been degraded since Euro-American settlement with deleterious consequences for fish, wildlife, and water quality. The continuous grazing effects of domestic livestock along with wild herbivore influences resulted in the degradation and/or have prevented the recovery of many western riparian systems. Because traditional livestock grazing strategies have focused on the ecology and production of upland communities, riparian/fisheries and wildlife values have often been overlooked (Wagner 1978, Nehlsen et al. 1991, Fleischner 1994). As in many other riparian zones of the interior west, the riparian-obligate hardwoods along Meadow Creek were all but eliminated. Given their value as providing critical habitat features for both the terrestrial and aquatic biota of these ecosystems, improved riparian management and ecological restoration are in need of implementation.

Riparian ecosystems are zones of frequent and varied disturbances at many temporal and spatial scales (Gregory et al. 1991). It was hypothesized that vegetation recovery following cessation of high intensities of grazing would be rapid in riparian ecosystems because of available water and nutrient resources, and because of the inherent plant adaptations for survival following frequent natural disturbances. We used a passive restoration approach and examined the inherent resilience and early regrowth responses following the removal of livestock, with and without wild ungulate browsing. Following more than a century of heavy grazing, growth increases were dramatic. However, wild herbivores had significant and negative influences on the rate of recovery on the height of black cottonwood, and height, crown area, crown volume, biomass and reproductive output of willows.

The rates of establishment were low at Meadow Creek due to the scarcity of mature-statured shrubs and the absence of reproduction. Nevertheless, the total number of all shrubs combined increased 50% in two seasons. If this trend continues artificial plantings will be unnecessary. We expect that as stature and vigor increase, more shrubs will produce seed and densities will continue to increase.

The restoration of endangered fisheries is a great concern for many western stream/riparian ecosystems (Nehlsen et al. 1991, Frissell 1993). Watershed and aquatic habitat restoration and sustainability include the re-establishment of streamside plant communities which provide many essential ecological functions. Over browsing by herbivores can severely reduce seed production, plant establishment, and plant vigor/survival. Due to their accessibility and palatability, recovering low-profile shrub communities remain vulnerable when herbivores are present. Although light seasonal livestock grazing in some systems may be compatible with some objectives, complete protection during the first 5-10 years of recovery offers the greatest likelihood of successful restoration (Kovalchik and Elmore 1992, Elmore and Kauffman 1994, Fleischner 1994). Where elk and/or deer populations are high, and the most rapid riparian recovery possible is desired, additional protection measures may be required at least during the initial recovery period.

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Literature Cited

Armour, C.L., D.A. Duff, and W. Elmore. 1991. The effects of livestock grazing on riparian and stream ecosystems. AFS position statement. Fisheries.16: 7-11.

Behnke, R.J., and R.F. Raleigh. 1978. Grazing and the riparian zone: Impact and management perspectives. p 184-189. In: Strategies for Protection and Management of Floodplain Wetlands and Other Riparian Ecosystems. USDA Forest Serv. GTR-WO-12.

Beschta, R.L., W.S. Platts, and J.B. Kauffman. 1991. Field review of fish habitat improvement projects in the Grande Ronde and John Day River Basins of Eastern Oregon. Project No. 91-069. Bonneville Power Administration, Division of Fish and Wildlife, Portland Oregon. 53pp.

Brunsfield, S.J. and F.D. Johnson. 1985. Field guide to the willows of east-central Idaho. Wildlife and Range Science Bulletin 39. Idaho Forest,Wildlife and Range Experiment Station. University of Idaho, Moscow ID. 95p.

Bryant, L D. and J.M. Skovlin. 1982. Effect of grazing strategies and rehabilitation on an eastern Oregon stream. IN: Habitat disturbance and recovery. Proceedings of a symposium. San Francisco, CA: California Trout, Inc: 27-30.

Bull, E.L., and J.M. Skovlin. 1982. Relationships between avifauna and streamside vegetation. Transactions of the Forty-seventh North American Wildlife and Natural Resources Conference 47: 496-506.

Busse K.G. 1989. Ecology of the Salix and Populus species of the Crooked River National Grasslands. MS Thesis, Oregon State University, Corvallis. 116p.

Carothers, S.W. 1977. Importance, preservation and management of riparian habitat: An overview. In: Importance, Preservation, and Management of Riparian Habitat. USDA Forest Serv. Gen. Tech. Rep. RM-43:2-4.

Case, R.L. 1995. The ecology of riparian ecosystems of northeast Oregon: Shrub recovery at Meadow Creek and the structure and biomass of headwater upper Grande Ronde ecosystems. MS Thesis, Oregon State University, Corvallis. 137p.

Chaney, E., W. Elmore and W.S. Platts. 1990. Livestock grazing on western riparian areas. U.S. Environmental Protection Agency, Region 8, Denver, CO.

Cooperider , A. 1990. Conservation of biological diversity on western rangelands. 55th North American Wildlife Conference, Denver, CO.

Crouse, M.R., and R.R. Kindshy. 1984. A method for predicting riparian vegetation potential. Pages 13-17 in Proc. 1984 Pacific Northwest range management short course. Oregon State Univ., Pendleton.

Cummins, K.W. 1974. Stream ecosystem structure and function. Bio Science 24: 631-641.

Elmore, W. 1992. Riparian responses to grazing practices, Chapter 17 IN: R.J. Naiman (ed) Watershed Management. Springer Verlag Berlin, p 442-457.

Elmore, W. and R.L. Beschta. 1987. Riparian areas: perceptions in management. Rangelands vol. 9 No. 6 260-265.

Elmore, W. and J.B. Kauffman. 1994. Riparian and watershed systems: degradation and restoration. (editors) Vavra M., W. A. Laycock, R. D. Pieper. IN: Ecological implications of livestock herbivory in the west. Society for Range Management.

Fleischner, T.L. 1994. Ecological costs of livestock grazing in western North America. Cons. Biol. 8 (3) 629-644.

Frissell, C.A. 1993. Topology of extinction and endangerment of native fishes in the Pacific Northwest and California (USA). Cons. Biol. 7:342-354.

Ganskopp, D.C. 1978. Plant communities and habitat types of the Meadow Creek experimental watershed. M.S. Thesis. Oregon State University, Corvallis.

Gillen, R.L., W.C. Krueger, and R.F. Miller. 1984. Cattle distribution on mountain rangeland in Northeastern Oregon. J. Range Manage. 37:549-553.

Green, D.M. and J.B. Kauffman. 1995. Succession and livestock grazing in a Northeast Oregon riparian ecosystems. Journal of Range Management 48:307-313.

Gregory, S.V., F.J. Swanson, W.A. McKee, and K.W. Cummins. 1991. An ecosystem perspective of riparian zones: focus on links between land and water. Bioscience Vol. 41 No. 8 540-551.

Hansen, P.L. 1992. Classification and management of riparian-wetland shrub sites in Montana: Proceedings—Symposium on ecology and management of riparian shrub communities. Sun Valley Idaho.USDA For. Sea. Intermountain Research Station, Ger. Tech. Rep. INT-289 130-137.

Harris , R.W. 1954. Fluctuations in forage utilization of ponderosa pine ranges in eastern Oregon. J. Range Manage. 7: 250-255.

Hitchcock, C.L. and A. Cronquist. 1973. Flora of the Pacific Northwest: an illustrated manual. University of Washington Press, Seattle. 730p.

Irvine, J.R. and N.E. West. 1979. Riparian tree species distribution and succession along the lower Escalante River, Utah. The Southwestern Naturalist. 24:331-346.

Kauffman, J.B. 1988. The status of riparian habitats in the Pacific Northwest forests IN: Streamside Management: Riparian Wildlife and Forestry Interactions (ed. Radeke K. J.). College of Forest Resources Univ. of Wash. Contribution No. 59 45-55.

Kauffman, J.B., R.L. Beschta, and W.S. Platts. 1993. Fish habitat improvement projects in the Fifteenmile and Trout Creek Basins of Central Oregon: Field review and management recommendations. US Department of Energy, Bonneville Power Administration, DOE/BP-18955-1. Portland, OR.

Kauffman, J.B., R.L. Case, D. Lytjen, N. Otting, and D.L. Cummings. 1995. Ecological approaches to riparian restoration in northeast Oregon. Restoration and Management Notes. 13:12-15.

Kauffman, J.B. and W.C. Krueger. 1984. Livestock impacts on riparian ecosystems and streamside management implications: a review. Journal of Range Man. 37:430-437.

Kauffman, J.B., W.C. Krueger, and M. Vavra. 1985. Ecology and plant communities of the riparian area associated with Catherine creek in Northeastern Oregon. Agriculture Experiment Station Bulletin 147, Oregon State University, Corvallis, OR.

Kauffman, J.B. N. Otting, D. Lytjen, and R.L. Beschta. 1996. Ecological principles and approaches to riparian restoration in the western United States. Chapter 5 IN: Healing the Watershed: a guide to the restoration of watersheds and native fish in the west. Pacific Rivers Council. Eugene, OR.

Kay, C.E. 1994. The impact of native ungulates and beaver on riparian communities in the Intermountain West. Nat. Res. and Envir. Issues Vol. I 23-44.

Kay, C.E. and S. Chadde. 1991. Reduction of willow seed production by ungulate browsing in Yellowstone National Park. IN: Proceedings— Symposium on ecology and management of riparian shrub communities. USFS Intermountain Research Station GTR-INT-289.

Kovalchik, B.L. and W. Elmore. 1992. Effects of cattle grazing systems on willow dominated plant associations in central Oregon. IN: Proceedings— Symposium on ecology and management of riparian shrub communities. USFS Intermountain Research Station GTR-INT-289.

Kufeld, R.C. 1973. Foods eaten by the Rocky Mountain elk. J. Range Manage. 26(2): 106-113.

Li, H.W., G.A. Lamberti, T.N. Pearsons, C.K. Tait, J.L. Li and J.C. Buckhouse. 1994. Cumulative effects of riparian disturbances along high desert trout streams of the John Day Basin, Oregon. Transactions of the American Fisheries Society 123:627-640.

Maser, C. and J.R. Sedell. 1994. From the forest to the sea: The ecology of wood in streams, rivers, estuaries and oceans St. Lucie Press. Delray Beach, FL.

Marcuson, P.E. 1977. The effect of cattle grazing on brown trout in Rock Creek, Montana. Fish and Game Fed. Aid Pro. F-20-R-21-11a.

McIntosh, B.A. 1992. Historical changes in anadromous fish habitat in the Upper Grande Ronde River, Oregon 1941-1990. MS Thesis. Oregon State University, Corvallis. 86p.

Meehan, W.R., F.J. Swanson, and J.R. Sedell. 1977. Influences of riparian vegetation on aquatic ecosystem with particular reference to salmonid fishes and their food supply. IN: Importance, Preservation and Management of Riparian Habitat. USDA Forest Service Gen. Tech. Rep. RM- 43. pp.137-643.

Murphy, M.L. and W.R. Meehan. 1991. Stream ecosystems: IN: Influences of Forest and Rangeland Management of Salmonid Fishes and Their Habitats. American Fisheries Society Special Publication 19:17-46.

National Research Council. 1992. Restoration of aquatic ecosystems. Committee on restoration of aquatic ecosystems-Science, technology and public policy. National Academy Press, Washington, D.C.

Nehlsen, W., J.E. Williams and J. Lichatowich. 1991. Pacific salmon at the crossroads: stocks at risk from California, Oregon, Idaho and Washington. Fisheries. 16:4-21.

Ogden, P.S. 1950. Ogden's snake country journals, 1824-26. Hudson Bay Records Society.

Platts, W.S. and R.F. Raleigh. 1984. Impacts of grazing on wetlands and riparian habitat. IN: Developing Strategies for Range Management. National Research Council. Westview Press. Boulder, CO.

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Platts, W.S. 1978. Livestock Interactions with Fish and Aquatic environments: Problems in Evaluation. IN: Transactions of the North American Wildlife and Natural Resources Conference 43: 462-476.

Platts, W.S. and J.N. Rinne. 1985. Riparian and stream enhancement management and research in the Rocky Mountains. N. Am. J. of Fish. Man. 5: 115-125.

Rickard, W.H. and C.E. Cushing. 1982. Recovery of streamside woody vegetation after exclusion of livestock grazing. J. Range Manage. 35:360-361.

Roath, L.R., and W.C. Krueger. 1982. Cattle grazing and behavior on a forested range. J. Range Manage. 35:332-338.

Rollins, P.A. (editor). 1935. Discovery of the Oregon trail (narratives of Robert Stuart). New York: Charles Scribners Sons. 391p.

Sedell, J.R. and R.L. Beschta. 1991. Bringing back the "bio" in bio-engineering. USDA Forest Service, American Fisheries Society Symposium 10:160-175.

Schutlz, T.T. and W.C. Leininger. 1990. Differences in riparian vegetation structure between grazed areas and exclosures. J. Range Manage. 43:295-299.

Skovlin, J.M.. 1991. Fifty Years of Research Progress: A Historical Document on the Starkey Experimental Forest and Range. USDA Forest Service Gen. Tech. Rep. PNW-GTR-266.

Skovlin, J.M. 1984. Impacts of grazing on wetlands and riparian habitats: A review of our knowledge. IN: Developing strategies for range management. Boulder, Colorado: National Research Council. Academy of Science, Westview Press. pp. 1001-1103.

Sullivan, K., T.E. Lisle, C.A. Dolloff, G.E. Grant and L.M. Reid. 1986. Stream channels: The link between forests and fishes. IN: Salo E.O. ,Cundy T.W. Streamside Management: Forestry and fish interactions. Inst. of Forest Resources. No. 57. University of Washington, Seattle. pp.39-97.

Swanston, D.N. 1991. Natural Processes: IN: Influences of Forest and Rangeland Management on Salmonid Fishes and Their Habitats. American Fisheries Society Special Publication 19:139-179.

Thomas, J.W., C. Maser, and J.E. Rodiek. 1979. Wildlife habitats in managed rangelands—The Great Basin of southeastern Oregon. Riparian Zones. USDA For. Serv. Gen. Tech. Rep. PNW-80.

Thwaite, R.G., ed. 1905. Early western travelers. Cleveland, OH: Arthur H. Clark.

Wagner, F.H. 1978. Livestock grazing and the livestock industry. pp. 121-145 IN: Wildlife and America contributions to an understanding of American wildlife and its conservation. ed. Howard Fleischner 1994 Brokaw, Council on Environmental Quality, US Fish and Wildlife Service, Forest Service, National Oceanic Atmospheric Administration.